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Integrating Systematic Conservation Planning and Ecosystem Services: An Indicators Approach in the Hill Country of Central Texas

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Integrating Systematic Conservation Planning and Ecosystem Services: An Indicators Approach in the Hill Country of Central Texas

by

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Dedication

To my wife, Patty, and my children, Townes and Julian.

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Abstract

Integrating Systematic Conservation Planning and Ecosystem Services: An Indicators Approach in the Hill Country of Central Texas

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Ecosystem services are the aspects of the environment utilized to produce human well-being and are key elements of landscape sustainability. Increasingly, measures of ecosystem services are being incorporated into conservation decision making. However, a framework for evaluating systematic conservation planning ranked selection scenarios with indicators of ecosystem services has not been developed. Using the Central Texas counties of Blanco, Burnet, Hays, Llano, San Saba, and Travis as a study, a suite of spatially explicit modeling tools, Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST), are used to quantify carbon storage, soil conservation, and water provision. A fourth service metric, ecosystem richness, is derived using Texas Parks and Wildlife ecological systems classification data. The values of these four services are then used to evaluate four conservation scenarios, developed in conjunction with a local conservation non-profit, Hill Country Conservancy (HCC), and derived using Marxan decision-support software.

The evaluation process consists of both geographic information system (GIS) and statistical analysis. GIS based overlay analysis is used to identify areas of multiple ecosystem service overlap. Spearman correlation tables are used to test the spatial relationship among ecosystem services, as well as the relationship among each of the four conservation scenarios. Wilcox-Mann-Whitney U tests (WMW) are used to assess the statistical significance of each scenario's ecosystem service values as compared to the values of a random control scenario.

The results of this work reinforce the findings that there is often significant variability in the spatial congruence of multiple ecosystem services and their provision across a landscape. This work also supports the conclusion that the targeting of ecological phenomena for conservation concurrently targets areas supporting multiple ecosystem services. More distinctively, the results verify the capacity of ecosystem service indicators to effectively inform an iterative systematic conservation planning process.

At the local landscape-scale, this work provides HCC with defensible support of their conservation decisions based not only on organizational priorities, but also on ecosystem service values. More broadly, this work provides a framework for evaluating conservation scenarios with spatially explicit values of ecosystem services which can be replicated across a wide range of project scales and objectives.

Keywords: Ecosystem services, Systematic conservation planning, InVEST, Marxan, Landscape sustainability, and Central Texas

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INTRODUCTION

THE NEED FOR AN INTEGRATED CONSERVATION FRAMEWORK

The Hill Country of Central Texas is a mosaic of distinct natural and sociocultural landscapes. Like other important places globally, Hill Country landscapes are under threat from anthropocentric land-use transformations (Sanderson et al. 2002; Vitousek et al. 1997; Theobald 2005; Kjelland et al. 2007). The consequences of land-use transformations often include the degradation of ecological functionality, loss of biological diversity (biodiversity), the erosion of rural cultural identity, and overall decreased economic productive capacity (MA 2005; Foley et al. 2005; Balmford et al. 2002; Sorice et al. 2012). To sustain the well-being of Central Texans over future generations, despite significant projected population growth, rapid urbanization, and potential increases in natural disturbances due to changes in global climate (Nielsen-Gammon 2011), the integrity and diversity of these landscapes must be ensured (WCED 1987; MA 2005).

Conservation is one of the most cost-effective mechanisms of protecting both natural and sociocultural landscapes with a benefit to cost ratio estimated at 100:1 (Balmford et al. 2002; Costanza et al. 1997). Human societies, however, are complex adaptive systems embedded in even more complex adaptive ecosystems (Berkes et al. 2003; Liu et al. 2007; Walker et al. 2004). Choosing what to conserve involves making decisions under uncertainty, complexity, and substantial biophysical and monetary constraints, in addition to conflicting human values and interests (Dietz et al. 2003; Salafsky et al. 2002). The need to build consensus for action, despite these inherent difficulties, has evolved into the field of conservation planning.

Conservation planning is the process of identifying, setting, and implementing goals aimed at preserving biodiversity and natural resources within human influenced landscapes. Recently, two tools have emerged within the field to help facilitate decision-making and reasoning: 1) *systematic conservation planning*, and 2) the concept of *ecosystem services*.

Systematic conservation planning (Margules and Pressey 2000) is a structured step-wise approach to identifying conservation areas (Margules and Sarkar 2007). Because it explicitly incorporates human influence in determining how conservation goals can be achieved, it is also highly iterative with feedback, revision, and reiteration possible at any stage in the process (Margules and Sarkar 2007). The aim of systematic conservation planning is to establish quantifiable conservation goals, or targets, and, based on constraints, identify a network of optimal priority areas that achieve these targets in the most efficient, rigorous, and scientifically credible way possible.

Ecosystem services are the environmental aspects of a landscape utilized either directly or indirectly by society in order to produce human well-being (Fisher et al. 2009). For example, the ecological processes that result in clean water, the provision of food, and clean air are direct utilizations. Conversely, the ability of vegetation to sequester and store carbon is an indirect service. Regardless of whether they are utilized directly or indirectly, the promise of ecosystem services in regards to conservation is their potential to be quantified, mapped, and valued based on the benefits society receives from them (Costanza et al. 1997; Daily 1997; de Groot 1992).

Research in systematic conservation planning and in ecosystem services has advanced substantially in the past twenty years. The development of optimization algorithms and software packages, as well as their simultaneous application to multiple resource allocation problems has led to a fundamental shift in the who, what, where, how, and when of conservation decision-making and implementation (Sarkar et al. 2006). Similarly, the inherent interdisciplinary nature of the ecosystem service concept has facilitated new ways of defining, classifying, and utilizing them across a broad spectrum of natural, social, and economic science disciplines (Balmford and Bond 2005; Burkhard et al. 2010; Fisher et al. 2009; Wallace 2007).

Consequently, within the last decade research into the use of systematic conservation planning and ecosystem services in combination has begun. In most instances, the combined application has involved the use of ecosystem services as targets within a systematic conservation planning framework (Chan et al. 2006; Izquierdo and Clark 2012). Although this method of integration holds potential, empirical evidence suggests a sometimes weak, and even negative, spatial correlation between cultural preferences, biodiversity richness, and ecosystem services—as well as between discrete ecosystem services themselves (de Groot et al. 2010; Cimon-Morin et al. 2013; Anderson et al. 2009; Bai et al. 2011; Costanza et al. 2007). This variability can lead to conservation solutions biased toward areas with high concentrations of targeted services but poorly suited to protect untargeted services, biodiversity richness, cultural identity, or any combination thereof. Thus, structural integration remains a central challenge (Cimon-Morin et al. 2013; de Groot et al. 2010).

Utilizing ecosystem services as spatially explicit performance measures, or indicators, rather than as targets, provides a more flexible method of integration in which a number of alternative conservation solution scenarios can be compared (Müller and Burkhard 2012; Perrings et al. 2011). Indicators of ecosystem services have been used in evaluating scenarios of urban development in Oregon's Willamette Basin (Nelson et al. 2009) and rural land-use change scenarios in Minnesota (Polasky et al. 2011). However, there remains a need to develop a similar framework for use in quantitatively comparing scenarios of landscape-scale systematic conservation planning ranked selections,—hereafter referred to as *conservation scenarios*—specifically within the Hill Country of Central Texas.

RESEARCH OBJECTIVES

This thesis is situated within the field of conservation planning. With it I introduce a quantitative, synthesized framework for use by conservation practitioners and planners—in Central Texas and beyond—in evaluating conservation scenarios with indicators of ecosystem services.

My objectives in developing this framework are twofold. At the global level, my goal is to develop an easily reproducible means to making more informed, efficient, reliable, and persuasive conservation decisions in which synergies and tradeoffs between conservation goals and ecosystem services are more effectively balanced—ultimately facilitating the transition to sustainable landscapes. At the local level, my objective is to expand upon a strategic conservation plan commissioned by a local conservation non-profit, Hill Country Conservancy (HCC), by incorporating spatially explicit values of ecosystem services in an iterative decision-support context.

How best to structurally integrate systematic conservation planning and ecosystem services is the overarching question impelling this research. However, I specifically address the following questions:

- Where within the contiguous Central Texas counties of Blanco, Burnet, Hays, Llano, San Saba, and Travis should a local conservation non-profit, Hill Country Conservancy (HCC), focus conservation efforts in order to efficiently balance regional need with organizational goals?
- How can multiple ecosystem service values be integrated as indicators in a systematic conservation planning process?
- Can conservation scenarios be effectively evaluated with indicators of ecosystem services?
- How can ecosystem service indicators inform conservation decisions?

To answer these questions, I have organized this thesis as five chapters. In Chapter 1, I review the academic literature in order to situate this work within the context of its theoretical and applied underpinnings, including those on nature and its role in society, sustainability, and conservation. In Chapter 2, I review aspects of the ecological landscape, the sociocultural landscape, and conservation as presently found within the study area—the six Central Texas counties of Blanco, Burnet, Hays, Llano, San Saba, and Travis.

In Chapter 3, I present the materials and methods used to integrate systematic conservation planning and ecosystem service indicators within a combined framework. Here, I show how Marxan decision-support software is used to develop four conservation scenarios: *HCC*, *Water*, *Agriculture*, and *Ecology*. Additionally, I show how Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) models are used to quantify, and map three ecosystem service indicators: *Water Yield*, *Soil Conservation*, *Carbon Storage*; along with a separately calculated indicator of *Ecosystem Richness* derived using Texas Parks and Wildlife Department's (TPWD) ecological systems classification data. Moreover, I also demonstrate how geographic information systems (GIS) based overlay analysis, a random control scenario, and statistical analyses—including Spearman's correlations and the Wilcoxon-Mann-Whitney U tests (WMW)—are used to assess the application of ecosystem services as indicators in comparing conservation scenarios.

In Chapter 4, I review and discuss the analysis results. Within this chapter, I use the correlation results to reveal a rather weak spatial relationship among the four ecosystem service metrics. Despite this spatial incongruence, and trusting in the combined results of the overlay analysis and WMW tests, I also prove that the conservation scenarios are more efficient at targeting areas supporting multiple ecosystem services than the random scenario. I then discuss the implications of these findings, both in relation to similar research and also in terms of how they may be interpreted to inform conservation decisions.

Specifically, I elaborate on the inability to generalize ecosystem service relationships while also verifying the capacity of ecosystem service indicators to effectively inform an iterative systematic conservation planning process.

Finally, I close this thesis with Chapter 5. This chapter includes a discussion on the areas of further research uncovered during this process, explicitly in regards to ecosystem services, conservation planning, and conservation and sustainability. Also offered within this chapter are my final conclusions in which I reiterate not only the importance of continuing to develop efficient ways to incorporate ecosystem services in decision-support frameworks, but also on the relevance of this work in informing both broad-scale strategies and local planning decisions.

The field of conservation planning is progressing rapidly to address present and future environmental challenges. Likewise, the concept of ecosystem services is increasingly being utilized to justify conservation action. Yet how best to integrate ecosystem services within a landscape-scale systematic conservation planning process remains an open question. This thesis represents one small contribution to the quickly accelerating debate.

CHAPTER 1:

NATURE, SOCIETY, SUSTAINABILITY, AND CONSERVATION: A REVIEW OF THE LITERATURE

In this chapter, I review both the theoretical and applied research upon which this work is founded in an effort to situate it within the broader knowledge base. Key topics include the role of nature in society, the concept of sustainability, and the theory and practice of conservation. As a result, I have organized this chapter according to three primary sub-sections: 1) *Nature and Society*, 2) *Sustainability*, and 3) *Conservation*.

NATURE AND SOCIETY

Conservation is contingent upon an understanding of what constitutes nature. Yet the relationship between nature and society is not static but vacillates among cultures and time. In this sub-section, I review the literature to define nature—and nature's role in society—in a contemporary theoretical context.

What is Nature?

In the broadest sense, nature represents the entirety of the physical universe from the subatomic to the cosmic, including the geologic, hydrologic, and chemical cycles which allow for the existence of life here on earth. How humans actually relate to the physical world, and thus how they define nature, is commonly much more esoterically nuanced by culture, time, and scale (Cronon 1990; Nash 2001). For example some East Asian, and many indigenous cultures, have historically assumed a biocentric perspective in which humans are but a single component in a universal nature (Descola and Pálsson 1996). Conversely, throughout most of documented Western history, and specifically since the industrial revolution, there has been a notable dichotomy between civilization and nature—in this instance *nature* most often being identified as *wilderness* (Nash 2001).

Even within the Western perspective, the definition of nature, or wilderness, has evolved substantially within the last two-hundred years: from the "taming" of the rugged, desolate frontier wilderness of manifest destiny, to the sublime manifestation of natural perfection of Thoreau (1854); from the preservationist movement of John Muir (1901), to the wise consumption of nature's wild resources as espoused by the Resource Conservation Ethic of Gifford Pinchot (1910); from the shift to wilderness-as-ecological-system of Aldo Leopold's Conservationist Land Ethic (1949), to the modern environmental movement, inspired by Rachel Carson (1962), in which anthropogenic impacts on wilderness were broadly recognized; and finally, from the pristine wilderness of an indigenous pre-Columbian America (Sale 1990; Shetler 1991), to William Denevan's argument that wilderness itself is a myth and that indigenous Americans were hardly benign bystanders (1992).

The implication of the disparity among these perspectives is that the human relationship with nature varies according to sociocultural values and beliefs. In other words, our conception of nature is not universally built on immutable physical principles, but rather is conferred definition and form through cultural filters (Cronon 1990, 1996; Demeritt 2002; Gifford 1996; Greider and Garkovich 1994). In short, *nature* is socially constructed.

Nature as a Social Construction

If nature is defined largely by the cultural values ascribed to it, then redefining nature is possible through a reexamination of those values. Assuming the Western perspective of *nature*-as-*wilderness*—still arguably prevalent in the United States—as a point-of-reference, four sources provide compelling arguments for a novel, socially constructed definition of nature.

The first argument, offered by environmental historian William Cronon, focuses on how we temporally relate to nature. To Cronon (1996), nature, in the guise of wilderness, is a false premise encouraging us to adopt too high a standard for what we tend to view as exclusively nature. A more practical and beneficial approach would be to observe nature in the ordinary, thereby serving to temporally ground nature in the everyday (Cronon 1996). In this way, the daily societal interaction with nature is made manifest by its relative temporal closeness. In other words, a neighborhood park, or even the trees in the median on the drive to work are not only equal in nature to the wilderness of Big Bend National Park—or closer in context, The Balcones Canyonlands National Wildlife Refuge—but perhaps more so because they are encountered on a daily basis.

The second argument is provided by environmental philosopher Andrew Light and addresses nature in a spatial context. According to Light (2010), in order to be made relevant to everyone, nature must not only be seen in the ordinary—as opposed to the perception of nature only as extraordinarily wild—but that its generality, and ultimately its utility, must also be tangible through a specific relation to place—"a psychologically robust and even morally loaded conception of location imbued with a storied relationship between people and the things around them." This not only serves to locate nature spatially, it also emphasizes the local, social connection inherent within a given location. Thus, a focus on *place* offers the context in which specific problems can be identified, different values understood, conflicts resolved, and choices made (Potschin and Haines-Young 2013).

In coupling nature with a connotation of scale, sociologists Greider and Garkovich (1994) put forward a third argument. To Greider and Garkovich (1994), "landscapes" are "symbolic environments" which reflect our own self-definitions of identity and culture in relation to our specific biophysical surroundings. It is through the landscape that we incorporate elements of the physical environment into our cultural self-identity and through which biophysical changes to that environment may, in turn, result in a redefinition of ourselves. In short, *landscapes* are nature as an extension of ourselves.

Addressing the hierarchical relationship of people and nature, the fourth and final argument comes from Steven Vogel in his rejection of Deep Ecology—a still influential philosophy within the environmental movement. Dismissing human exceptionalism and the anthropocentric world-view of conservationists, Deep Ecology espouses a holistic, bio-centric ideology where humans are merely one biotic component of an egalitarian ecosystem (Naess 1995). To Vogel (1996), this "soft-science" based, bio-centric point-of-view is counterproductive. Because humans are the only species capable of global environmental change, an anthropocentric view of nature must be taken so that we may fully accept our role as responsible stewards of the planet (Vogel 1996).

In sum, and for the purposes of this work, nature is the physical universe and all the processes, cycles, and forms of life found on Earth, specifically as they relate to Central Texas. Yet, as a social construction, nature is also temporally located in the everyday, which is to say it is utilitarian (Cronon 1996). Nature is local, being spatially located through the idea of place (Light 2010). And nature is symbolically understood as landscapes (Greider and Garkovich 1994). Furthermore, because humans have the technological capacity to compress time and space on scales not seen anywhere else in nature—often to the detriment of natural systems which do not correspond to the same parameters—biocentrism is rejected in favor of a responsible anthropocentrism based on environmental stewardship (Vogel 1996).

Through this combined definition, society and nature are now integrated, not as a single system, but rather as separate, inter-linked systems. As such, the definition of nature remains malleable. It can be both transformed by society, and, reciprocally, used to transform society.

Of course, to assume the social construction of nature does not mean that nature is *created* by society. Ecological processes and biological life will continue to operate, adapt, and evolve with or without human presence. It also does not mean that all of nature is created equal in structure and function. Much as there exists an urban to rural gradient in which social diversity and services increase from largely homogenous rural areas to heterogeneous urban centers, nature also exists along a gradient, albeit inversely, from the homogenous, highly urbanized, limited, nature of the neighborhood park, to the heterogeneous, service rich, highly wild—referred to hereafter as wild nature (McDonnell et al. 1997). Rather, the social construction of nature is simply another mutation in the evolution of how society views and defines the natural systems with which it interacts.

Conversely, humanity is completely reliant upon nature for its welfare and survival (Guo et al. 2010; MA 2005). Society could not exist without the renewable and nonrenewable natural resources, or natural capital, that support the production of goods and services from which it derives benefit. As such, understanding nature solely in terms of cultural values is insufficient. To correctly reflect society's level of dependence, nature must also be understood in terms of the value of the benefits humanity receives from it. The need to also value nature under these terms has culminated, thus far, in the idea of ecosystem services.

Nature as Ecosystem Services

The concept of ecosystem services originated two decades ago in collaborative attempts between economists and ecologists to build a theoretical framework for quantifying and valuing natural capital (Costanza et al. 1997; Daily 1997; de Groot 1992). Since that time, numerous definitions and classification schemes of ecosystem services have been developed to meet various objectives (Balmford et al. 2011; Boyd and Banzhaf

2007; Costanza 2008; de Groot et al. 2002; Fisher et al. 2009; TEEB 2010; Wallace 2007; MA 2005).

The most widely adopted of these comes from the Millennium Ecosystem Assessment (MA) (2005) where ecosystem services were broadly defined as "the benefits people obtain from ecosystems" and classified into one of four service types: 1) "Provisioning services," which include the provision of food, materials, and drinking water; 2) "Regulating services" such as air quality, climate, water, erosion, and pest regulation; 3) "Cultural services" such as recreation, aesthetic values, and spiritual and religious values; and 4) "Supporting services" including soil formation, photosynthesis, and water and nutrient cycling.

The popularity of the general MA definition is understandable given the inherent interdisciplinary nature of the ecosystem service concept. However, Wallace (2007) argues that, in the context of the MA, the use of the term "service" is somewhat ambiguous with ecosystem function and processes, such as soil formation, being conceptually indistinct from services, such as food provision. In order to clarify the term "service," Fisher (2009) defines ecosystem services as "the aspects of ecosystems utilized (actively or passively) to produce human well-being." Defined this way, ecosystem phenomena, including structure and processes, become services if they are consumed by humans either directly or indirectly (Fisher et al. 2009). The Fisher definition is assumed for use in this study because of its capacity to broadly incorporate indirectly utilized ecological phenomena, such as carbon storage, within a "service" framework, yet also remain consistent with the MA classification scheme.

Under this framework, the concept of nature is characterized by an undeniably complex interaction between society and nature involving ecological processes, socio-economic processes, and cultural values. Consequently, what has been structured through the effort to define nature in terms of both its social construction and ecosystem services is essentially the ontological underpinnings of a new paradigm of environmental research known presently as *social-ecological systems* (SESs).

Social-Ecological Systems

SES theory is an emergent framework for describing and defining the interactions between the human and natural worlds. Although the origins of the SES concept trace back to C.S. Holling's (1973) adaptation of general systems theory as an explanatory device for the functioning of ecological processes, most recent theoretical advances have been made by interdisciplinary teams of natural and social scientists addressing solutions to common property resource conflicts (Berkes 1996; Goulder et al. 1997; Dietz et al. 2003; Ostrom 1990) and seeking to understand system resiliency (Gunderson and Holling 2002; Walker and Salt 2006).

SESs—also referred to as coupled human-environment systems—are defined as complex adaptive systems with interacting and interdependent physical, biological, and social components, characterized by reciprocal feedbacks, and emphasizing a "humans-in-nature" perspective (Carpenter and Folke 2006; Chapin, Kofinas, et al. 2009). In essence, human societies are complex adaptive systems embedded in even more complex adaptive ecosystems (Liu et al. 2007).

According to Chapin, Folke, et al. (2009), SESs demonstrate several fundamental characteristics: 1) SESs are self-organizing structures distinguished by the nonlinear interaction of a large number of physical components, including soil, water, and rocks; organisms, such as plants, microbes, and people; and the products of human activities, such as food, money, and buildings; 2) They have both amplifying and stabilizing feedbacks; 3) They are adaptive to change, whether through ecological processes or human agency. For instance, a significant drought may result in transitions of dominance in vegetative communities to more drought tolerant species. Or, in the damming of rivers to create reservoirs; 4) They may exist in alternative stable states in a given environment, such as a grassland that replaces a patch of forest destroyed by a wildfire; and 5) They are inherently unpredictable because of uncertainties arising from internal processes and their often non-linear response to external influences.

SESs can be defined at myriad scales, ranging from the entire planet to the microscopic (Chapin, Kofinas, et al. 2009; Levin and Harvey 1999). However, the degree to which the components interact is highly dependent on the regional system in which they are embedded, including cultural factors, which most often take shape at the landscape level (Chapin, Folke, et al. 2009; Greider and Garkovich 1994). In other words, it is at the landscape level that most people typically interact with and intuitively understand the natural world. Subsequently, landscapes have become pivotal scale domains in SES research (Wu 2013).

The assumed importance of the landscape-scale in SES research now begs the question: what exactly is a landscape?

What is a Landscape?

Given the wide range of land-forms and the fluid composition of biological communities, "landscape" is often imprecisely defined in terms of size, composition, or defining features. The intuitive, visceral understanding of the term is of "a large expanse of land and water" (Trombulak and Baldwin 2010). As vague as this innate definition may be, scholarly designations are often just as unstructured and varied. From an academic perspective, the term landscape typically implies a heterogeneous mosaic of local land-forms, plant and animal communities, and human land-uses combined over greater and greater areas (Trombulak and Baldwin 2010). While, some scholars try to place landscapes within a nested, hierarchical scale of land forms often below that of regions (Forman 1995), many other authors refer to an entire ecoregion, such as the Edwards Plateau, as a single landscape (Trombulak 2010).

More recently, a move has been underway to define a landscape by the parameters set forward within the research itself. For example, Cumming et al. (2013), define landscapes as "spatially bounded entities that are heterogeneous in many key elements and processes of interest." Under this definition, a landscape can be a single square-meter patch of grassland, or an entire continent, as long as the scale is appropriate to the patterns or processes under examination.

For the purposes of this work, I assume the Cumming et al. (2013), definition of landscape. With this classification, I am able to designate the geopolitically bounded, six-county study area a distinct landscape, despite the fact that it encompasses numerous other hierarchically-nested landscapes, contains only a portion of Edwards Plateau ecoregion, and also includes segments of adjacent ecoregions. What becomes important under this characterization is not the determination of the boundary itself, but rather the spatial-binding of the SES processes and interactions being examined within the landscape's borders. In other words, this work's six-county landscape provides a place-based framework for engaging questions involving social and ecological interactions and processes; including questions of sustainability in Central Texas.

SUSTAINABILITY

Much like the fluid definition of both nature and landscape, *sustainability* is also an amorphous concept. In the ensuing sub-section, I engage the broad ranging theoretical literature on sustainability in order to define, specifically, what that entails in terms of this work.

Sustainability in Social-Ecological Systems

In a perfect world, the flow of inputs and outputs in a social-ecological system would remain in a relatively stable, steady-state equilibrium between the consumption of the Earth's renewable natural resources, the rate at which those resources are replenished, and the equitable rationing of non-renewable resources. This highly simplified ideal represents the concept of sustainability broadly defined by the Brundtland Report (1987) as the ability of present generations to meet their needs without compromising the ability of future generations to meet their own needs.

Since its publication, there have been some notable refinements to the Brundtland definition. In the "triple-bottom-line" iteration, sustainability is based on three pillars: environment, society, and economy (Elkington 2004). To achieve sustainability under the triple-bottom-line is to simultaneously achieve environmental, economic, and social sustainability—the whole is the sum of its parts.

The concept of "strong sustainability" further refined both the Brundtland definition and the triple-bottom-line structure of sustainability (Daly 1995). In strong sustainability, rather than overlapping pillars, the components of sustainability are nested: economic activities are nested within the social domain, and both the economic and the social are nested within, and constrained by, the environment (Daly 1995). Strong sustainability represents a consistent, albeit, precautionary expression of the Brundtland definition in which the environment is necessarily given deference (Wu 2013). As such, it is the conceptualization of sustainability assumed in this work.

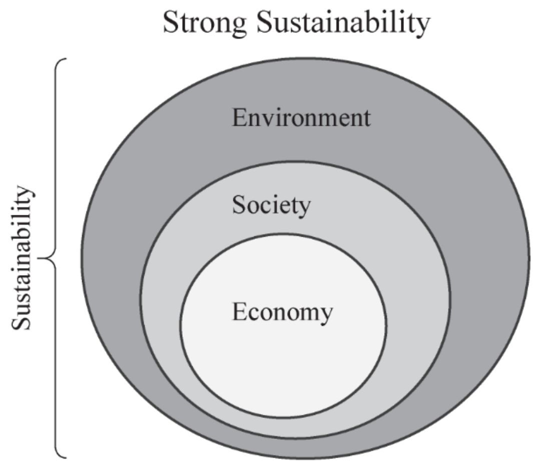

Figure 1.1 Conceptualization of Strong Sustainability
In the concept of strong sustainability, social-ecological sustainability is contingent on human activities not exceeding the capacity of ecosystems to provide services. In turn, the capacity of ecosystems to provide services is constrained by the earth's life support system. Redrawn from Chapin, Folke, et al. (2009).

In each of the three iterations of sustainability mentioned, it is assumed that society and nature can reach an optimal state through efficiency, eventually attaining an equilibrium between human population, natural resource use, the carrying capacity of the planet, and human well-being (Selman 2008). This definition of sustainability may be sufficient at the global scale where remaining within the carrying capacity of the planet is an absolute. However, attaining such an equilibrium at the landscape level is not likely.

Landscape scale SESs are highly dynamic and unpredictable with frequently occurring disturbances, both natural and anthropogenic in origin. The effects of both large-scale directional disturbances such as climate change, and non-linear abrupt perturbations such as drought, flood, fire, or economic crises, are most acutely felt at this level, leaving the affected SESs vulnerable to regime shifts, or the crossing of thresholds into novel, often less desirable states (Capra 2002; Chapin, Kofinas, et al. 2009; Walker et al. 2004; Walker and Salt 2006). As a result, the capacity of a landscape scale SES to provide ecosystem services is diminished, thereby increasing the likelihood of negative impacts in terms of the local economy (Farber et al. 2006; Haberl et al. 2006), poverty (Adams et al. 2004), community development (Roseland 2000; Thin 2002), and agricultural production (Bennett et al. 2006; Gabriel et al. 2010).

If ecosystem services, and thus, human well-being are to be maintained or improved within a local context and over the long term, the landscape level SES must be both resilient and adaptable to disturbances. Walker et al. (2004) define resilience as "the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks of the pre-disturbance system." Similarly, adaptability is the capacity of human actors in a system to influence or manage resiliency, whether intentionally or unintentionally (Walker et al. 2004). Together, resiliency and adaptability represent the mechanisms through which natural and social systems, respectively, respond to unpredictable change in order to dynamically maintain vital processes and services.

This is not to suggest that sustainability and resiliency are necessarily contrasting terms. Resiliency and adaptability, along with vulnerability—defined by Turner et al (2003) as the degree to which a system or system component is likely to experience harm as a result of disturbance—are all integral characteristics of a concept of sustainability which transcends environmental carrying capacity or the stability of ecological processes (Wu 2013). Rather, it is to suggest that in defining sustainability, notions of scale must also be taken into consideration. As a result, defining sustainability for a landscape level SES means defining *landscape sustainability*.

Landscape Sustainability

Numerous scholars have proposed definitions of landscape sustainability within recent years. Variations have been inspired by the Brundtland definition (Forman 1995;

Turner et al. 2013), as an extension of the TPL approach (Selman 2008), based on multifunctionality in landscape design and planning (Musacchio 2009), and focused on the provision of ecosystem services (Nassauer and Opdam 2008; Potschin and Haines-Young 2006). For the purposes of this study, a definition developed by Wu (2013), centered on landscape-specific ecosystem services, and incorporating resilience and adaptability in a manner consistent with the concept of strong sustainability, is assumed:

> Landscape sustainability is the capacity of a landscape to consistently provide long-term, landscape-specific ecosystem services essential for maintaining and improving human well-being in a regional context and despite environmental and sociocultural changes.

In accepting this definition, the question now becomes: how can the provision of landscape-specific ecosystem services be sustained within the Central Texas study area despite unpredictable environmental changes due to climate change or drought, and sociocultural changes from rapid urbanization and population growth?

There are two primary solutions to this question. One solution is ecological restoration. Although restoration will undoubtedly play an increasing role in the move toward landscape sustainability, it is limited in scale due to cost, often ineffective due to lack of complete ecological knowledge, less productive in terms of biodiversity and ecosystem services than lands left intact, and filled with uncertainty in outcomes and consequences (Benayas et al. 2009). At present, the most consistently effective answer to this question, in terms of both ecological function and socio-economic costs, is the alternative solution: conservation.

CONSERVATION

According to Smith and Wishnie (2000), conservation is defined as any action or practice that is designed to prevent or mitigate resource depletion, species extirpation, or habitat degradation. That conservation is an intrinsic component of landscape sustainability is largely undebated. However, conservation requires acceptance by a diverse set of participants, often with disparate views and interests. Thus, it is the socioeconomic motivations, implementation mechanisms, and planning strategies that beget the majority of conservation research. In the following sub-section, I explore the literature regarding these issues as a means to situate this study within the broader context of applied conservation science, specifically within the field of conservation planning.

The Shifting Conservation Rationale

Beginning with Egypt at least 3000 years ago, and being noted in passages of the Bible's Old Testament, the concept of setting aside land or resources has been recorded through history and across cultures (Alison 1981). The historical motivations for conservation have been in the form of legal decrees citing privileged use, the preservation of flora and fauna, spiritual significance, and aesthetic value (Alison 1981; Diamond 2005). More recently, the call to conservation action has been most commonly associated with the loss of biological diversity (biodiversity)—or the variety of life on the planet.

By the end of the last century it had become apparent that global biodiversity was declining at unprecedented rates estimated at one thousand times higher than historic background levels, largely due to land-use transformations associated with expanding populations and economic growth (Hoekstra et al. 2005; Sanderson et al. 2002; Steffen et al. 2007; Vitousek et al. 1997). Since that time, tremendous effort and resources have been aimed at sustaining biodiversity through conservation action (Adams et al. 2010; Naidoo and Ricketts 2006).

Thanks to these efforts, the disproportionate benefits of biodiversity conservation to ecosystem health and human well-being—estimated at a benefit to cost ratio of 100:1

(Balmford et al. 2002)—are now well understood within the scientific community (Ehrlich and Wilson 1991; Naidoo and Ricketts 2006; Odling-Smee 2005; Pimentel et al. 1997). Still, these benefits are exceptionally difficult to translate into public policy because they are either couched in moral arguments, not fully captured in commercial markets or not comparable in terms of economic services and manufactured capital (Costanza et al. 1997). As a result, public concern for conservation action aimed exclusively at biodiversity seems to be waning (Novacek 2008; Pearce 2007). Moreover, as the Millennium Ecosystem Assessment (MA) (2005) made poignantly clear, the loss of biodiversity is not an end unto itself, but rather a symptom of more pervasive ecological degradation.

In terms of conservation, the MA has had a profound influence on the trajectory of research and practice. Developed by over 1,300 scientists, the MA used a SES framework for documenting, analyzing, and understanding the effects of environmental change on ecosystems and human well-being (Carpenter et al. 2009). Rather than view ecosystems exclusively in terms of biodiversity, the MA considered them through the lens of the services and benefits ecosystems provide society, of which biodiversity plays a key role at all levels of service production (Costanza et al. 2007). In doing so, the MA shifted the predominant rationale from one defined by the separation of nature and society, to a platform of "conservation for the people" (Kareiva and Marvier 2007).

This shift has opened the door to new lines of argument, innovative ways of identifying, quantifying and valuing nature, and conservation aimed not only at wild nature, but aspects of cultural identity such as agriculture, recreation, and aesthetics. In essence, conservation has morphed from a metaphorical wall, to a bridge constructed to ensure that the flow of ecosystem services remains unimpeded as it passes from the environment to society, and ultimately, the economy—as envisioned by nested hierarchical relationship of strong sustainability.

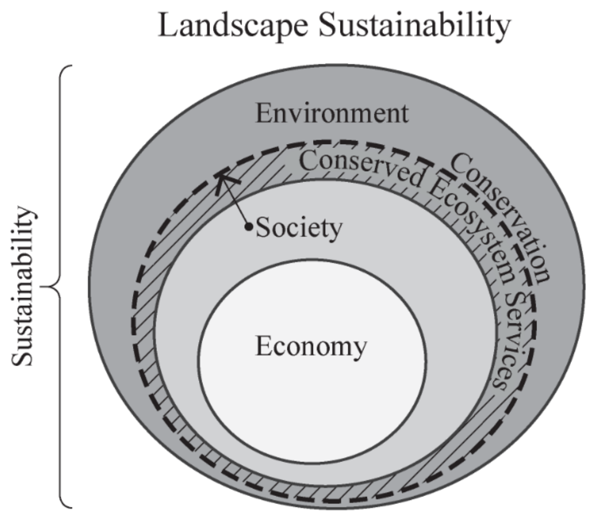

Figure 1.2 Conceptualization of Conservation's Role in Landscape Sustainability
This conceptualization of landscape sustainability is based on, and adapted from, the concept of strong sustainability as presented by Chapin, Folke, et al. (2009). In this version, conservation is a vital extension of the economic and social spheres, acting as a bridge into the environmental sphere, and thus, ensuring a sustained supply of ecosystem services.

Mechanisms of Conservation

At present, conservation action has largely been driven by public policy and enforceable laws aimed at preserving aesthetic quality, protecting natural resources, or preventing extinctions at a national level. For example, the United States has been a pioneer in federally mandated conservation with the world's first national park in 1872 (Yellowstone National Park), the first transnational treaty aimed exclusively at species protection in 1918 (Migratory Bird Treaty Act), the preservation of wild nature in 1964 (Wilderness Act), the protection of common pool resources in 1969 and 1972 (National Environmental Policy Act and the Clean Water Act, respectively), and the safeguarding of the Nation's most endangered and threatened species in 1973 (Endangered Species Act).

The proliferation of similar policies, both at the State level and around the globe, speak to the success of these measures in achieving their stated goals (Andreen 2003; Carson and Mitchell 1993; Schwartz 2008; Snape 1996; WDPA 2014). However, these top-down mechanisms, though arguably successful, remain insufficient (Rodrigues et al. 2004). As of 2012, approximately 12% of the Earth's terrestrial surface is currently under some form of protected status, yet biodiversity and ecological integrity continue to decline (Rands et al. 2010). It is now well recognized that to adequately protect wild nature, biodiversity, and the provision of ecosystem services, mechanisms aimed at the conservation of private lands must play an increasing role (Knight 1999; Langholz and Lassoie 2001; Wright and Czerniak 2000).

The methods of private land conservation vary. Federally mandated direct regulation—specifically as enforced under the Endangered Species Act (ESA)—has played a significant role in conserving private lands through the designation of critical habitat and the proliferation of multiple-species habitat conservation plans (Schwartz 2008). Although effective, some have viewed conservation mandated through the ESA as placing an undue burden on private landowners rendering it politically unpopular (Innes et al. 1998). In contrast, voluntary conservation programs, such as wildlife management cooperatives, are increasing in use, though their long-term effectiveness is still in question (Sorice 2008).

Some of the more novel approaches to conservation include incentive-based mechanisms. For example, tax policies encouraging low-intensity land management practices, such as wildlife management (Texas House Bill 1358) or water stewardship (Texas Senate Bill 449), in exchange for lower appraised property valuation options were passed by the State of Texas in 1995 and 2013, respectively. Furthermore, policies in which individuals or communities are directly compensated for actions that increase the provision of ecosystem services, referred to as payments for ecosystem services, also hold great promise (Jack et al. 2008). Nevertheless, at present, the effectiveness of incentive-based conservation mechanisms remain unclear (Kosoy and Corbera 2010; McElwee 2012).

Within the United States, the most common method of conservation on private land is now conservation easements (Kiesecker et al. 2007). Conservation easements are voluntary agreements entered into by property owners in which their development rights are transferred to a governmental entity or qualified conservation organization—such as HCC—in perpetuity in return for direct payment or tax breaks (Kiesecker et al. 2007). Although conservation easements are not free of criticism, their use has grown as a result of their flexibility in preserving not only biodiversity, as research suggests, but also working and cultural landscapes, yet still allowing private ownership and limited economic activity (Kiesecker et al. 2007; Morris 2008). Furthermore, as Kiesecker et al. (2007) indicates, the selection of lands for conservation easements has become increasingly deliberate with clear ecological objectives.

To facilitate landscape sustainability, it is highly likely that Texas, being over 95% privately owned, will continue to rely on conservation easements to protect its unique natural and sociocultural landscapes. HCC, along with similar organizations, will play an increasingly important role in the future of Texas conservation. This heightened responsibility must be balanced within a highly heterogeneous biophysical landscape, an equally diverse set of participants, and limited resources. Thus, to be effective, conservation on private land must be strategic.

Conservation Planning

Conservation planning has developed as a means to optimize conservation and land-use decisions within human dominated and natural landscapes. With roots in island biogeography (MacArthur and Wilson 1967), environmental planning (McHarg 1969), and conservation biology (Soule and Wilcox 1980), conservation planning explicitly incorporates human land-use, laws, regulations, economics, aesthetics, and multiple perspectives into a comprehensive framework for determining how conservation goals can be achieved (Trombulak and Baldwin 2010). The traditional aim of conservation planning has been to separate elements of biodiversity from processes that threaten their existence by establishing protected areas—also referred to as reserves.

However, questions regarding the overall effectiveness of reserves designed under the traditional conservation planning model have emerged. Empirical evidence began to confirm that selection of reserve sites was being driven largely by socioeconomic preferences rather than ecological need. For example, Joppa and Pfaff (2009) showed that protected areas tended to be concentrated on land that is too remote or unproductive—typically at higher elevations and steeper slopes—to be important economically. In contrast, the greatest number and most diverse assemblage of species is often found at lower elevation in areas valuable for agriculture or settlement (Rodrigues et al. 2004; Scott et al. 2001).

Furthermore, competition for limited resources has also led to a focus on grand scenery and wilderness over the complete representation of biodiversity at all levels of organization (Margules and Pressey 2000). As a result of these socio-economic/socio-political conflicts, newly established protected areas were likely to be relatively unproductive islands, functionally disconnected from landscape level ecological processes, and surrounded by largely unregulated human land-uses which threaten their persistence (Rodrigues et al. 2004; Margules and Pressey 2000; Scott et al. 2001).

Systematic Conservation Planning

Originally developed by Margules and Pressey (2000) to address inefficient, site-by-site policy and design of the traditional conservation planning model, systematic conservation planning (SCP) is a structured step-wise approach to solving conservation problems in a rigorous and scientifically credible manner. The aim of SCP is to establish quantifiable conservation goals, called targets, and, based on constraints, identify a network of priority areas that achieve these targets in either the smallest area possible (minimum set; Wilson et al. 2009) or to maximize the benefits for a given budget (maximal coverage; Church et al. 1996) according to a heuristic (non-exact) or optimal (exact) algorithm. Key advantages of the SCP framework include the fluid integration of multiple disciplines, a highly iterative and transparent process, clear goals, and quantifiable success or failure in achieving those goals (Margules and Sarkar 2007; Margules and Pressey 2000).

Following the precedent established by Margules and Pressey (2000), several SCP protocols have now been developed (Pressey and Cowling 2001; Groves et al. 2002; Margules and Sarkar 2007). Each include a number of non-unidirectional steps intended to guide planners through the complete conservation process; from participant identification and data gathering to reserve selection and management; with feedback, revision, and reiteration possible at any stage (Margules and Sarkar 2007). In practice, the inclusion of particular steps is highly adaptable and dependent on the nature of the conservation problem being addressed, the collective goals of the participants, and the intended use of the results.

Many of the SCP stages are basic elements of any planning process. However, for explicit problems of ranked selections, such as presented in this work, two fundamental steps distinguish SCP from the traditional conservation planning model (Margules and Sarkar 2007): 1) The identification of conservation targets; and 2) The selection of conservation priorities based on complementarity.

The setting of conservation targets is one of the unique characteristics of SCP in that it allows for measurable results. In general, targets are either species based or reflect social preferences for the landscape, such as multifunctional agriculture or ecosystem services (Wilson et al. 2009). For each biotic, environmental, or sociocultural variable—collectively termed conservation elements (Margules et al. 2002)—considered within a planning area, targets can be specified by absolute number, probability of occurrence, or proportional representation (Trombulak 2010).

For instance, species based targets may be specified as 1500 ha (3,706.6 acres) of each vegetation type in order to ensure a comprehensive and representative conservation network (Moilanen et al. 2009). For ecosystem services or other social objectives, targets may be at least 50% of all stored carbon (Chan et al. 2006), or to maximize a portfolio of cultural resources such as agricultural lands or recreational open space (Stoms et al. 2011).

Ideally, targets would be based exclusively on empirical evidence, but scientific justification is rarely available for all conservation elements. Thus, defining targets remains both art and science, often subjectively—albeit transparently—based on particular organizational goals.

The second distinguishing characteristic of SCP is complementarity. In order to solve either minimum set or maximal coverage conservation problems, SCP relies on the concept of complementarity—a step-wise, iterative process in which successive selections are prioritized, one-by-one, based on the representation of conservation elements that are not adequately represented in the existing selections, and, therefore, "complement" those elements already contained (Kirkpatrick 1983; Margules et al. 1988; Vane-Wright et al. 1991; Nicholls and Margules 1993). As a result, the sites selected at each iteration are not necessarily the most diverse, but those that add the most targeted conservation elements to the initial network (Margules and Pressey 2000).

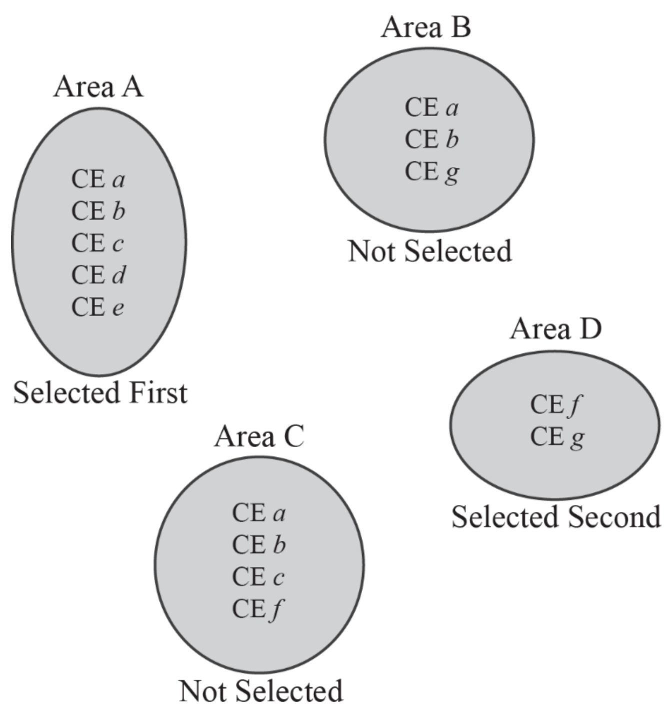

Figure 1.3 Simplified Complementarity Example
This example graphically depicts the concept of complementarity in regards to a minimum set problem. In this simplified version, targets are one occurrence for conservation elements (CE) *a* through *g*. Area A is selected first because it has both the greatest number of unique elements and the highest element richness. Area D is subsequently selected because it adds a greater "complement" of unique elements to the solution set than either Area C or Area B, despite containing fewer elements. Adapted from (Ogren 2008).

Numerous algorithms and corresponding software packages have been developed for use in computing complementarity including Marxan, C-Plan, and ResNet (Sarkar et al. 2006). Although each return broadly similar results (Carwardine et al. 2007), Marxan—which utilizes a spatially explicit heuristic algorithm termed "simulated annealing" to solve minimum set problems—has become the most commonly used. The reasons for Marxan's wide adoption are because it is: well documented and supported by its developers, continually improved for increased flexibility (Watts et al. 2009), integrated with geographic information systems, freely available in a format that runs on personal computers, and efficient in simultaneously handling multiple conservation elements and targets (Ball et al. 2009; Trombulak 2010).

The use of SCP and Marxan has most often been applied to the design of species based reserve networks (Moilanen et al. 2009). For example, Marxan has been used for the ranked selection of avian habitat within a 3200 km (~2,000 mile) corridor (Pearce et al. 2008), combined terrestrial and freshwater amphibian habitat (Becker et al. 2010), and coastal marine fisheries (Klein et al. 2008). The flexibility of Marxan in incorporating multi-criteria targets allow it also to be used to address other natural resource management problems such as multifunctional agriculture (Machado et al. 2006; Stoms et al. 2011), ecological restoration (McBride et al. 2010), water protection in Central Texas (Siglo 2012), and more importantly for this study, ecosystem services (Chan et al. 2011; Chan et al. 2006; Izquierdo and Clark 2012).

The Role of Ecosystem Services in Systematic Conservation Planning

The ecosystem service concept is inherently transdisciplinary, incorporating fundamentals of biology, ecology, sociology, and economics into a single conceptual framework. With the decrease in global ecosystem service supply, and recognition of the direct impact of ecosystem service loss on human well-being now broadly recognized (Guo et al. 2010), much recent research has been focused on quantifying, mapping, valuing, and assessing a wide range of ecosystem services, as well as their synergies and tradeoffs (Egoh et al. 2008; Reyers et al. 2013; Tallis et al. 2008; Tallis and Polasky 2009; Naidoo et al. 2008; Plieninger et al. 2013; van Berkel and Verburg 2014). As a result, measures of ecosystem services are increasingly being incorporated into decision-making processes such as natural resource management (Liu et al. 2013; Schmitt and Brugere 2013), land-use planning (Barral and Oscar 2012; Niemelä et al. 2010), sustainable development (Gren and Isacs 2009; Vidal-Legaz et al. 2013), design (Jones et al. 2012; Windhager et al. 2010), and, perhaps most profoundly, in conservation planning—as suggested by a recent review in which 153 peer-reviewed articles were identified as relating measures of ecosystem services to biodiversity conservation in some capacity (Cimon-Morin et al. 2013).

Research directly incorporating ecosystem services into a systematic conservation planning process, though showing great promise (Egoh et al. 2008), has thus far been limited to a few studies (Chan et al. 2011; Chan et al. 2006; Izquierdo and Clark 2012; Larsen et al. 2011; Naidoo et al. 2008; Onaindia et al. 2013; Thomas et al. 2013). Presently, there is no definitive approach for explicitly integrating ecosystem services into SCP problems, although developing frameworks compatible with popular tools of reserve design, such as Marxan, is widely viewed as beneficial (Chan et al. 2011; Chan et al. 2006; Izquierdo and Clark 2012).

In most previous SCP examples, ecosystem services have been integrated as targeted benefits. In the targeted benefit approach, the reserve-design algorithm considers ecosystem services as intrinsically important and attempts to minimize costs while maximizing benefit. The problem with this approach is that there is often a weak, and even negative, spatial correlation between ecosystem services and other common targeted benefits such as biodiversity, species, or ecosystems, as well as between discrete categories of ecosystem services and scales of measurement (Cimon-Morin et al. 2013).

For example, on a global scale, regulating services, such as carbon storage, tend to be positively correlated with wild nature and biodiversity, yet negatively correlated to provisioning and cultural services such as food production (Anderson et al. 2009; Egoh et al. 2008; Holland et al. 2011; Maes et al. 2012). Alternatively, when analyzed at the local scale, biodiversity and high carbon storage value may no longer spatially coincide (Anderson et al. 2009; Nelson et al. 2009). Thus, using a targeted benefit approach increases the risk of biased conservation solutions, unmet targets, or undesirable trade-offs between other conservation priorities.

In response to this issue, Chan et al. (2011) incorporated ecosystem services using a co-benefit/cost approach. Reserve-design algorithms, like Marxan, typically combine targeted benefits with a 'cost surface' in order to help specify reserve selections while minimizing 'costs' (Ball et al. 2009). In the method used by Chan et al. (2011), multiple ecosystem service values were combined into the cost function of Marxan creating a framework of *net-benefit maximization* in which the ecosystem services are *substitutable*. Although the results show promise in yielding less costly reserve networks, this approach has not been tested at fine scales and remains limited by the inability to simultaneously incorporate spatially variable cost values or threats—which are typically used to create the 'cost surface'—and co-benefits (Chan et al. 2011).

An alternative to either the targeted benefit or co-benefit/cost approach is to use ecosystem services as indicators to evaluate SCP ranked selections. In general, indicators are communication tools that enable a simplification of highly complex phenomena, typically for specific decision-making purposes. According to Heink and Kowarik (2010), an indicator in ecology and environmental planning is a "component or a measure of environmentally relevant phenomena used to depict or evaluate environmental conditions or changes or to set environmental goals." Significantly, the designation of ecosystem services as ecological indicators is supported by the Fisher et al. (2009) definition of ecosystem services as ecological phenomena.

There is recent precedent in using ecosystem service indicators to compare scenarios, albeit not within a systematic conservation planning context. Nelson et al. (2009) used InVEST derived ecosystem services to evaluate the impact of urban development scenarios in the Willamette Basin of Oregon. Similarly, Polasky et al. (2011), used ecosystem service indicators as a means of measuring the effect of land-use change scenarios in rural Minnesota. Yet there remains the opportunity to develop an integrated framework for use in evaluating conservation scenarios with spatially explicit values of ecosystem services.

With the remainder of this work, I take the opportunity to present such a framework; beginning with a social-ecological system review of the study area's natural and cultural landscapes in the proceeding chapter.

CHAPTER 2:

THE STUDY AREA: BLANCO, BURNET, HAYS, LLANO, SAN SABA, AND TRAVIS COUNTIES

The Texas Hill Country is a distinct biophysical and sociocultural region. Demarcated primarily by the Edwards Plateau ecoregion, the Hill Country encompasses twenty-one contiguous Central Texas counties. In 2011, the local conservation non-profit Hill Country Conservancy (HCC) initiated a strategic planning process intended to identify conservation priorities within these counties based on a thorough assessment of regional opportunities and threats. In the first phase of this process, variables such as boundaries, population growth, existing open space, land-use and land-cover patterns, land-market values, rare and threatened species, and both ground and surface water quality and quantity issues were evaluated and combined with expert opinion, organizational objectives, and landowner feedback in order to identify a landscape-scale area of focus for HCC's conservation efforts over the next twenty years (Siglo 2013).

Completed in January 2012, Phase I resulted in the selection of the following six Hill Country counties, as seen in Figure 2.1, comprising over 1.4 million hectares (3.5 million acres): Blanco (184,722 ha; 456,459 acres), Burnet (264,001 ha; 652,361 acres), Hays (175,660 ha; 434,065 acres), Llano (250,118 ha; 618,055 acres), San Saba (294,496 ha; 727,716 acres), and Travis (265,315 ha; 655,607 acres).

In this chapter, I provide a general overview of the natural and the sociocultural components and processes shaping the landscape of these six counties. In keeping with the social-ecological system framework, I have organized this chapter according to three subsections: 1) *The Natural Landscape,* which focuses on the ecological system; 2) *The Sociocultural Landscape,* which focuses on the social system and aspects of local culture; and 3) *Conservation,* which serves as a bridge between the two systems.

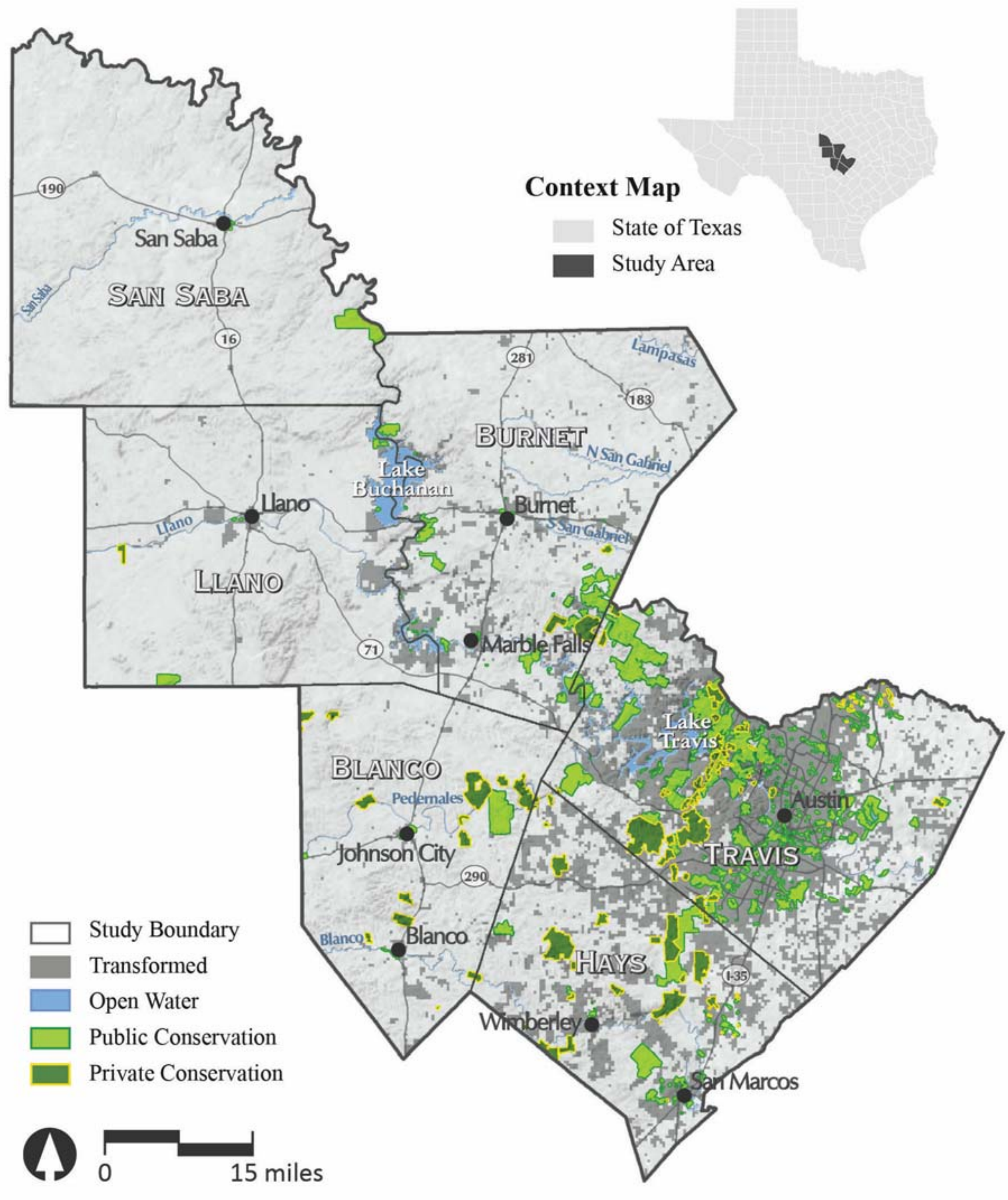

Figure 2.1 The Study Area
The study area includes Blanco, Burnet, Hays, Llano, San Saba, and Travis counties. Sources: USGS NHD (USGS 2013), TNRIS (2013), TPWD (2013), and TLTC (2013).

THE NATURAL LANDSCAPE

Coincidences of geography and geology have resulted in the Hill Country being one of the most ecologically diverse, and biologically important regions in Texas (Diamond et al. 1997). The six-county study area represents a significant area of interest critical to the sustained provision of ecosystem services to the region's population centers. In the following sub-section, I review the abiotic and biotic components of this vital natural landscape.

Climate and Weather

The climate of the study area is broadly defined as humid subtropical, being characterized by hot, humid summers and mild winters. It also represents a transitional zone between the humid subtropical east and the semi-arid steppe to the west. Average annual temperatures range between 20.2°C (68.4°F) and 17.6°C (63.7°F), generally following a southeast (warmest) to northwest (coolest) trend. Temperatures range from a mean of 28.8°C (83.34°F) to 26.1°C (78.98°F) in the warmest quarter to 11°C (51.8°F) to 8°C (46.4°F) in the coolest quarter.

Average annual precipitation follows a similar southeast (wettest) to northwest (driest) trend ranging from 890 mm (35") to 667 mm (26.3"). Typically the wettest periods are from April to June and September to October, while December and January are often the driest (Nielsen-Gammon 2011). Like most of Texas, the study area is highly vulnerable to short-term variations in rainfall for most of the year as a consequence of potential evapotranspiration far exceeding precipitation, particularly in summer (Nielsen-Gammon 2011).

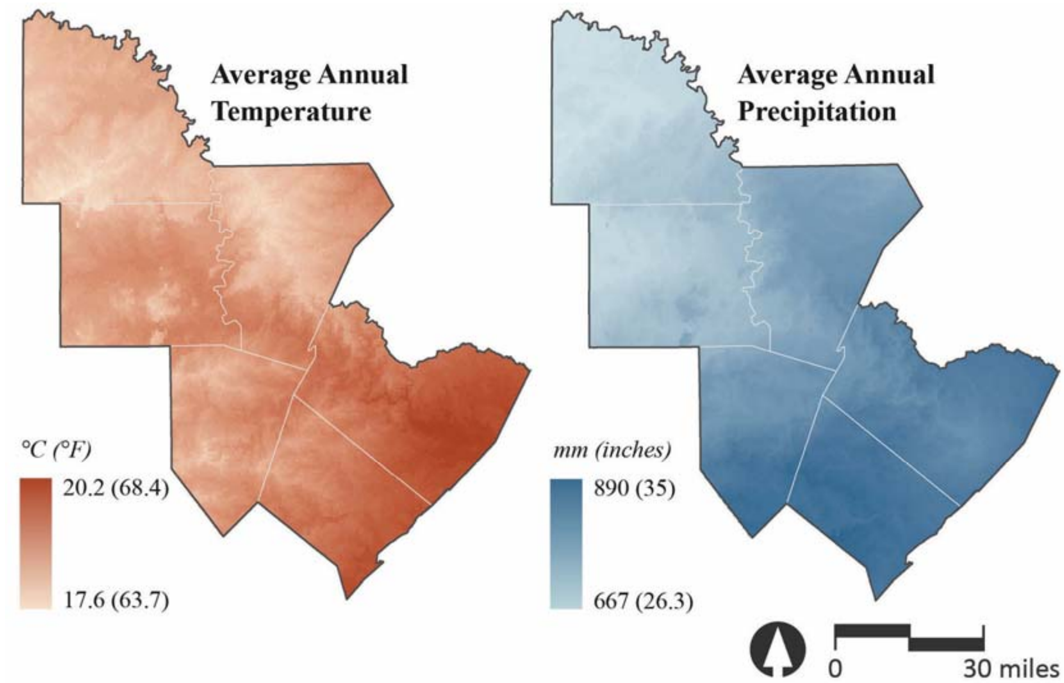

Figure 2.2 Average Annual Temperature and Average Annual Precipitation
Source: Worldclim: Global Climate Data: Bioclim (Hijmans et al. 2013).

Due to the geographic position of Texas on the North American continent, weather in the study area is highly variable, with severe and high impact events common. Examples of severe weather include: tornadoes, hail, and damaging thunderstorm winds. The area is also highly prone to flash flooding due to the potential of heavy rainfall from hurricanes, tropical storms, and moisture-laden gulf air, coupled with the Edward Plateau's rugged, rocky soil, and steep terrain (Nielsen-Gammon 2011).

Drought is also a common high impact event. Precipitation reconstruction from tree ring analysis dating back to 1650 has identified droughts lasting decades (Cleaveland 2006). Within the past century significant droughts have occurred in the 1910s, 1930s, and 1950s, with the drought of the 1950s now considered the drought of record (TWDB 2012). More recently, prolonged droughts have occurred in 1996, 1998, 2000, 2005-6, and 2011—the worst one year drought in the historical record (Hoerling et al. 2013).

The future climate of the study area is difficult to predict with any certainty. According to present-day climate models, variations in Texas climate over the past century, particularly for precipitation, do not correspond to changes expected from climate change (Nielsen-Gammon 2011). However, by the middle of this century, it is expected the processes of warming will overwhelm natural variability resulting in an increase of local temperature in the range of 2.2°C (4°F) (Nielsen-Gammon 2011). Subsequently, this increase in temperature is likely to result in rainfall events of heavier, but shorter duration, greater rates of evapotranspiration, and ultimately an increase in drought frequency and severity (Nielsen-Gammon 2011; TWDB 2012).

The Terrestrial Surface

The study area's predominant geologic foundations were formed during the Early Cretaceous period approximately 130 to 90 million years ago. At that time calcium carbonate from marine organisms was deposited in a shallow sea, eventually morphing into layers of limestone. Over the last 40 million years, Upper Cretaceous chalks, soft to hard limestones, shales, and claystones have eroded away revealing the resistant Lower Cretaceaous limestone of the Glen Rose and Edwards groups and creating the characteristic karst topography of caverns, sinkholes, and subsurface hydrology found throughout the Hill Country (Woodruff and Wilding 2008).

The Balcones Escarpment and the Llano Uplift are two other distinguishing geologic characteristics of the study area. Approximately 20 million years ago, the Balcones Fault system developed along a northeast-southwest arc stretching from Del Rio along the Mexican border north beyond Waco. Bisecting Hays and Travis counties nearly in half, the well-expressed escarpment forms an abrupt transition zone between the Gulf Coastal Plain, and its gently rolling Blackland Prairies to the south, and the steep limestone canyons and tablelands of the Edwards Plateau to the north (Woodruff and Wilding 2008). Located largely within Llano County, the Llano uplift is characterized by the central exposure of Precambrian igneous rocks—primarily granite—in a roughly circular geologic dome formation.

In general, both elevation and slope within the study area are variable. High elevations of 605 meters (~2,000 ft.) above sea level (asl) found along the western edge of the area descend into river valleys and canyons at ~350 meters (~1,150 ft.), off of the Balcones Escarpment at ~250 meters (~820 ft.), to a low of 112 meters (~367.5 ft.) in the Colorado River floodplain in the area's southeast corner. Average slope over the area is ~5%. Slopes between 15% and 60% occur across 9% of the study area and are generally localized to the steep-sided canyons found along waterways to the west of the Balcones Escarpment. Accounting for only .11% of the study area, significant slopes greater than 60% are not common, but do occur within these canyons.

The physical and chemical properties of area soils vary greatly depending largely on the underlying bedrock. For example, the Glen Rose and Edwards limestone based soils typically have a slightly to moderately alkaline pH in the range 7.5 to 8.2, whereas soils formed from the granite of the Llano Uplift are typically slightly acidic to neutral with a pH ranging between 6.5 and 7.0.[1] In general, limestone based soils north of the Balcones Escarpment are rocky, shallow, alkaline, relatively highly erodible, and poorly suited for cultivated crops (Wrede 2010). In contrast, alluvial soils along river bottoms within the plateau, and especially below the escarpment, are often silty-clay-loam up to forty inches thick with both high natural fertility and high available water content, making them well suited for agricultural production (Woodruff and Wilding 2008).

[1] Based on USGS STATSGO 2 Soil database. The USGS pH scale ranges from 'Ultra Acidic' at pH < 3.5 and 'Very Strongly Alkaline' at pH > 9.0. Neutral values range from pH 6.6 to 7.3.

Hydrology

The central hydrologic feature of the study area is the Colorado River Basin. Over 81% of the study area's land surface drains into the Colorado River through either the Middle Colorado-Concho Basin (14%), Middle Colorado-Llano Basin (62%), or the Lower Colorado Basin (5%). Major tributaries within the area include the San Saba River, the Llano River, and the Pedernales River. Sections of each of these rivers, including the Colorado, have been designated as "ecologically significant streams" by the Texas Parks and Wildlife Department (TPWD 2014), thus protecting them from further impoundment.

Although no segment of either the Guadalupe River or the Brazos River is located within its boundary, the study area does include portions of their basins and tributaries. Located in the eastern half of Burnet County, the Lampasas River, the North San Gabriel and the South San Gabriel all drain into the Little Basin, part of the Brazos River Basin. An important section of the Blanco River, a major tributary of the Guadalupe River, is found in southwest corner of Hays County.

No significant natural lakes exist in the study area, though several large reservoirs along the Colorado River—referred to as the *Highland Lakes*—have been created to store fresh water, aid in flood control, and provide hydroelectric power. Beginning in the north with Lake Buchanan and following the river south, additional reservoirs include Inks Lake, Lake LBJ, Lake Marble Falls, Lake Travis, Lake Austin, and Lady Bird Lake. In total, these reservoirs comprise ~20,500 hectares (50,722 acres) of the study area.

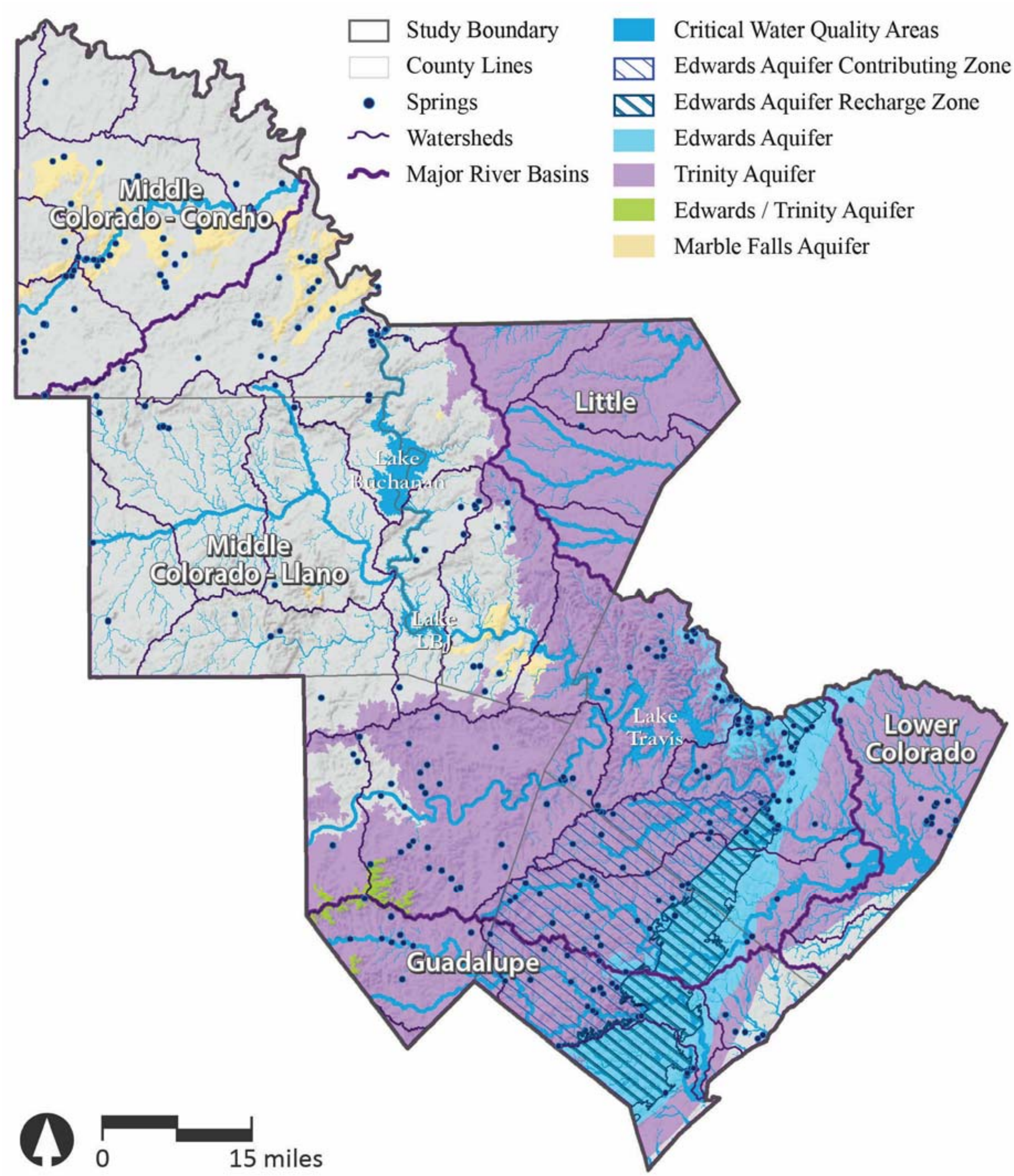

Figure 2.3 Hydrology
Sources: USGS NHD (2013), and TNRIS (2013).

Underground aquifers are one of the Hill Country's fundamental hydrologic characteristics. The most prominent aquifers found within the study area include the Edwards, the Trinity, and to a lesser extent the Marble Falls. With recharge zones covering 4.2% and 31.5%, respectively, both the Edwards and the Trinity occur in the southern half of the study area directly parallel with and above the Balcones Escarpment, while the Marble Falls, covering 2.8%, is located in the northern half, predominately within San Saba County.

A large number of artesian springs spread the ecological influence of the aquifers well beyond the study boundary. In total, there are over 322 documented springs found within the area (USGS 2013). Of these, Barton Springs in Austin, Jacobs Well in Wimberley, and Aquarena Springs in San Marcos, are the most well-known. As is the case with Aquarena Springs and the San Marcos River, many of these springs serve as the primary water source of area streams and tributaries, providing relatively consistent water flow across much of South-Central Texas despite unreliable rainfall patterns (Abbott 1975). Moreover, karst springs and caves provide specialized habitat leading to the evolution of rare biological life forms (Bowles and Arsuffi 1993; Culver and Sket 2000).

Vegetation and Wildlife

The study area is home to a rich diversity of plant and animal communities, many of which are regionally endemic (TNC 2004; Diamond et al. 1997). This diversity is largely the result of high variability in climate, geology, topography, and hydrology creating numerous ecotones, or areas of steep transition between ecological communities, ecosystems, or ecological regions (ecoregions), where species richness and abundance tend to peak (Kark 2013).

For example, assuming the United States Environmental Protection Agency's (EPA) ecoregional classification system (EPA 2013), four Level III ecoregions converge within the study area: 1) the Edwards Plateau (76% of the study area); 2) the Cross Timbers (12.3%) at its north-eastern tip; 3) Texas Blackland Prairies (11.2%) at its southern tip; and 4) a small portion of the East Central Texas Plains (0.5%). At the more detailed Level IV classification, the number of distinct regions increases to nine with the Balcones Canyonlands (31.7%), Llano Uplift (19.7%), and Edwards Plateau Woodland (24.5%) dominating, as seen in Figure 2.3 and Table 2.1.

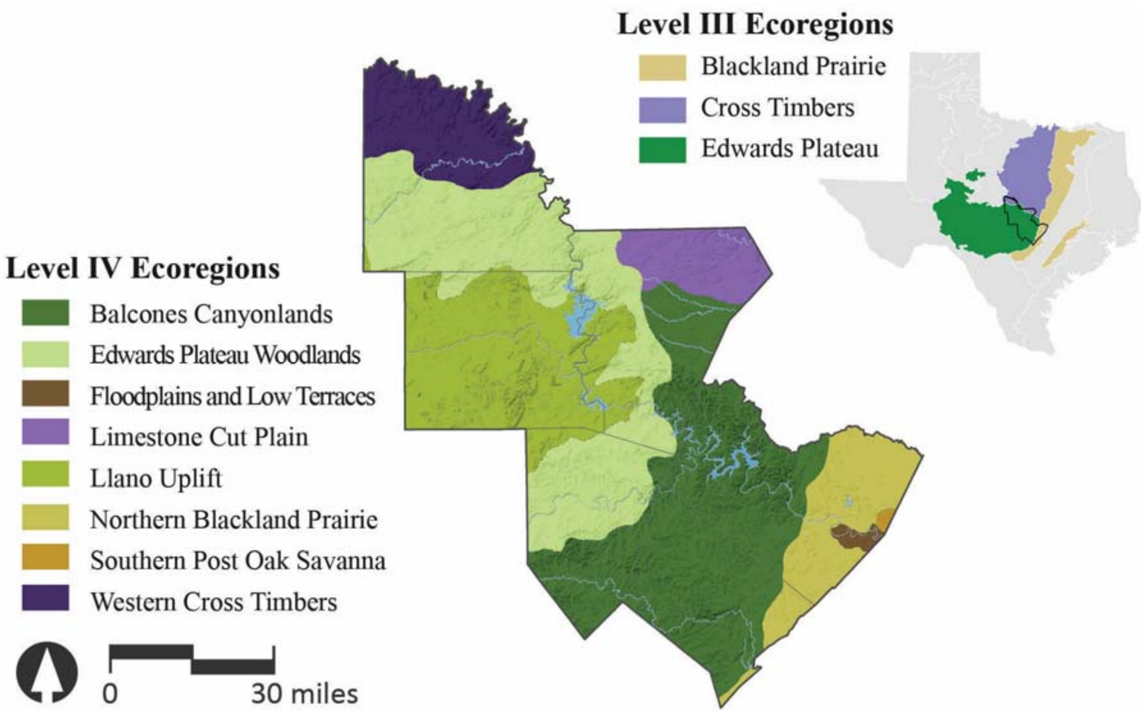

Figure 2.4 Level III and Level IV Ecoregions
Source: US EPA (2013).

Table 2.1 Level III and Level IV Ecoregions
Source: US EPA (2013)

Level III Ecoregions	*Acres*	*Hectares*	*% Total Area*
Cross Timbers (CT)	435,818	176,370	12.3%
East Central Texas Plains (ECTP)	16,984	6,873	0.5%
Edwards Plateau (EP)	2,682,810	1,085,700	76%
Texas Blackland Prairies (TBP)	396,465	160,444	11.2%

Level IV Ecoregions	*Level III*	*Acres*	*Hectares*	*% Total Area*
Western Cross Timbers	CT	266,698	107,929	7.6%
Limestone Cut Plain	CT	169,119	68,441	4.8%
Southern Post Oak Savanna	ECTP	14,439	5,843	0.4%
Floodplains and Low Terraces	ECTP	2,546	1,030	0.1%
Edwards Plateau Woodland	EP	865,817	350,385	24.5%
Llano Uplift	EP	695,978	281,654	19.7%
Balcones Canyonlands	EP	1,121,010	453,660	31.7%
Northern Blackland Prairie	TBP	372,857	150,890	10.6%
Floodplains and Low Terraces	TBP	23,609	9,554	0.7%

In general, the vegetation of the study area is a mix of evergreen savanna, upland deciduous, and lowland riparian plant communities (Wrede 2010). Texas Parks and Wildlife (TPWD) has classified these vegetation assemblages and communities into 79 distinct ecosystems (TPWD 2010). According to size of distribution, the most prominent of these include *Edwards Plateau Savanna Grassland* (15.66% of the study area), *Edwards Plateau Ashe Juniper Motte and Woodland* (8%), *Llano Uplift Grassland* (7%), and

Edwards Plateau Live Oak Motte and Woodland (7%). Largely the result of recent land-use patterns—specifically, high levels of herbivory by domestic animals—and wildfire suppression (Van Auken 2000), a large percentage, ~13% in total, is covered in native invasives such as Ashe juniper (*Juniperus ashei*) shrubland and mesquite (*Prosopis glandulosa*) shrubland.

As seen in Figure 2.4 and Table 2.2, combining the TPWD ecosystem classes into land-cover types yields a transitional landscape composed primarily of forest (38%) and grassland (32%).

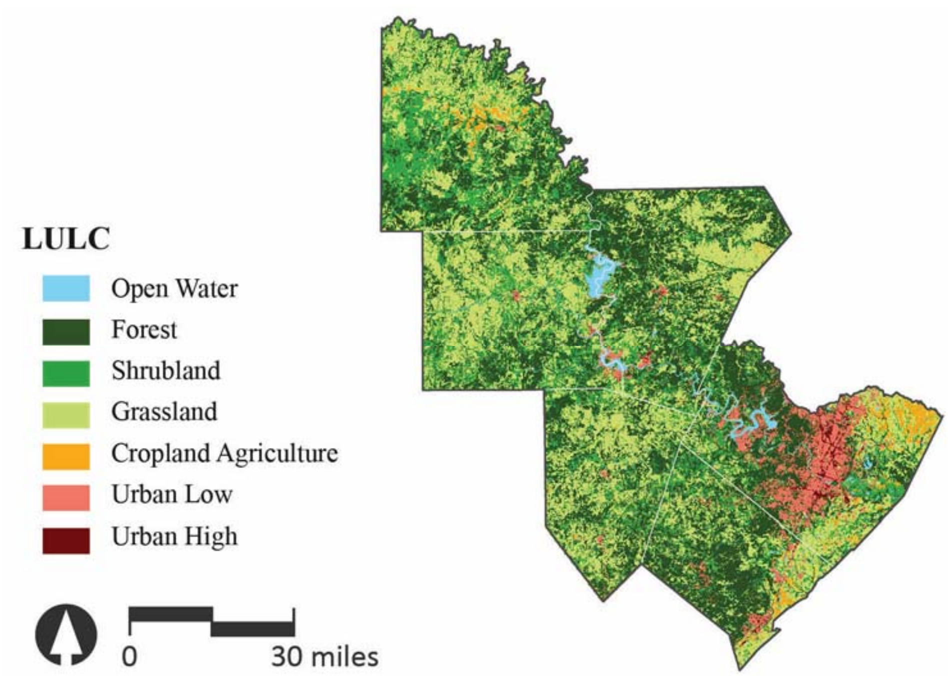

Figure 2.5 Land-Use and Land-Cover (LULC)
Source: modified from TPWD (2010).

Table 2.2 Land-Use and Land-Cover (LULC)
Source: modified from TPWD (2010).

LULC Class	*Acres*	*Hectares*	*% Total Area*
Open Water	65,507	26,510	1.8%
Forest	1,349,870	546,274	38%
Shrubland	706,008	285,712	19.9%
Grassland	1,135,070	459,349	32%
Cropland Agriculture	79,553	32,194	2.2%
Urban Low Density	183,433	74,233	5.2%
Urban High Density	31,338	12,682	0.9%

Each of the study area's ecosystem types provides habitat to a diverse range of biological life. At least 400 game and non-game species are presently found within the Edwards Plateau (TPWD 2014). Furthermore, because it is located within the North American Central Flyway, over 420 species of resident and migratory birds, as well as the migratory monarch butterfly (*Danaus plexippus*), can be seen in the area (Lockwood 2001).

A number of exceptionally rare species are also found within the study boundaries. Of these rare species, 21 are currently listed as either endangered or threatened under the US Endangered Species Act (ESA). Most of these are endemic inhabitants of specific karst caves and springs and are found nowhere else on earth. Examples include amphibians such as the Barton Springs salamander (*Eurycea sosorum*), arachnids such as the Bone Cave harvestman (*Texella reyesi*), fishes such as the San Marcos gambusia (*Gambusia georgei*), insects such as the Comal Springs riffle (*Heterelmis comalensis*), and plants such as Texas wild-rice (*Zizania texana*).

In terms of amount and distribution of potential habitat within the study area, arguably the most important ESA listed species is the golden-cheeked warbler (*Setophaga* *chrysoparia*). The golden-cheeked warbler is a small migratory songbird that nests exclusively in old-growth Ashe juniper and live oak slope forests within the Edwards Plateau. According to a species distribution model developed by Loomis Austin, Inc. (2008), 411,238 acres, or 11.6% of the study area, is classified as high quality golden-cheeked warbler habitat, with another 1,108,674 acres (31.3%) listed as moderate quality habitat.

THE SOCIOCULTURAL LANDSCAPE

The sociocultural landscape of the study area is undergoing significant changes. Rapid population growth, demographic shifts, urbanization, and development are altering local economies and regional culture. In turn, these changes influence patterns of land-use and land-tenure across the rural-urban gradient, ultimately affecting the ecological functionality of the natural landscape. In this section, I review some of the patterns and processes presently shaping the sociocultural landscape.

Cultural History and Land-Use

Archeological evidence suggests that the study area has been continuously inhabited by humans for at least the last 11 thousand years (Hester 1986), and perhaps as far back as 15.5 thousand years (Waters et al. 2011). Given resource and technology constraints, the indigenous populations remained small prior to the 1500, though there is some indication that they did actively transform the land to meet their needs (Hester 1986).

The arrival of the Spanish in the 16th century not only brought a new culture and permanent settlements to the San Antonio area by 1718, they also introduced new technology: the horse (Palmer 1986). With the quick adoption of the horse, the nomadic indigenous tribes of the Southern Great Plains, specifically the Comanche, greatly expanded their territory, controlling all of the present-day Hill Country and forcing many of the remaining, more sedentary peoples to relocate from the region (Palmer 1986). Although the frequency and extent of prescribed burning by bison-based nomadic cultures is unknown, there is evidence to suggest the technique was wide-spread enough to have some impact on land-cover within the study area (Kimmerer and Lake 2001).

Despite the short-lived establishment of Spanish missions as far north as present day San Saba, permanent settlements did not begin to increase within the Hill Country until the arrival of European settlers in the mid-1800s. A number of ethnic groups settled in and around the region, but by far the most influential to Hill Country culture has been the Germans. Their language, settlement patterns, and land-uses dominated the area until the late 19th century, and remain important to the local culture to this day (Palmer 1986).

While slowing during the Civil War, settlement and immigration began to both increase and diversify after 1870. During this time, waves of immigrant ethnic groups, including recently-freed slaves, Mormons, and Mexicans began to settle within the region. Over the last century, aspects of these disparate cultures have combined to form a distinctive regional culture collectively bound by its rural identity (Palmer 1986).

The rural cultural identity of the Hill Country has many aspects, but can be defined by several broad characteristics. The first characteristic is a clear dichotomy between rural and urban. The second is the raising of livestock as the dominant economic driver. For example, as recently as the early 1980s ranching accounted for over 90% of all agricultural income within the Hill Country (Palmer 1986). The third characteristic is the ability to be economically self-sufficient through agriculture. Encompassing the previous three, the fourth characteristic is a pattern of landownership predicated on privately holding relatively large, contiguous tracts of land.

This distinctive cultural identity remained largely unchanged for more than a century. However, beginning in the 1980s, new patterns began to emerge. Rapid growth in suburban development increased neighboring land values, altering economic incentives away from agriculture and toward the increased subdivision of large properties (Palmer 1986). As a result, landownership has shifted from traditional ranchers and farmers towards owners who manage the land for wild-game production, or who want to experience a rural lifestyle—often on a limited basis—but do not use the land primarily for agricultural purposes (Sorice et al. 2012).

The total effect of the landownership and land-use changes taken place within the Hill Country are not yet fully understood. From an ecological perspective, the shift in landownership and the subsequent decrease in land-use intensity may provide certain benefits as degradation from agricultural land-uses has been evident across the Hill Country from as early as the late 1800s (Palmer 1986; Sorice et al. 2012). From a cultural perspective, it is apparent that the changes taking place have an erosive effect on the distinctive rural cultural identity found across the Hill Country and within the study area, but to what extent, remains unclear (Sorice et al. 2012). What is evident from area-wide projections of population and density, is that these same patterns are likely to continue in the coming decades.

Present Population and Density

Today, the study area represents a major portion of one of the fastest growing metropolitan areas in the United States (USCB 2013). According to the most recent United States census, as of 2010 the total area-wide population is 1,260,052 (TSDC 2013). Representing 81% of the total, Travis County (1,024,266) dominates. The other five counties combined account for only 19% of the total population, with Hays County

(157,107) being the second most populous, followed by Burnet (42,750), Llano (19,301), Blanco (10,497), and San Saba (6,131).

Table 2.3 Areas and Population Trends
Source: TSDC (2013)

	Hectares	*% Total Area*	*Pop. 1950*	*Pop. 2010*	*Pop. 2050*	*% Increase 2010-2050*
Blanco	184,723	13%	3,780	10,497	17,672	68%
Burnet	264,001	18%	10,356	42,750	82,668	93%
Hays	175,660	12%	17,840	157,107	952,790	506%
Llano	250,118	17%	5,377	19,301	22,035	14%
San Saba	294,496	21%	8,666	6,131	6,722	10%
Travis	265,315	18%	160,980	1,024,266	1,990,820	94%
Total	1,434,314	100%	206,999	1,260,052	3,072,707	144%

As projected by the Office of the State Demographer, all counties are expected to experience population growth in the coming decades (TSDC 2013). The most significant growth is set to occur within the Austin-San Marcos area, specifically along the I-35 corridor, up the Colorado River, and around the Highland Lakes (EPA 2009). In terms of sheer number, Travis County is expected to nearly double its population from a 2010 estimate of 1,024,266 to 1,990,820 by 2050. If viewed in terms of percent increase in population between 2010 and 2050, growth in Hays County stands out at 506%, followed by Travis (94%), Burnet (93%), Blanco (68%), Llano (14%), and San Saba (10%). Overall, total area wide population is estimated to increase 144% over its present 2010 population, nearly tripling to over 3 million by 2050.

In general, increased population correlates to an increase in natural resource use and a decrease in ecological functionality (Guo et al. 2010; MA 2005). However, the impact of a population is not simply based on number, but also on its distribution. Over the last two centuries, population density, typically defined as people per acre, has been increasing globally as society has transformed en masse from largely rural to largely urban (Meyerson et al. 2007).

More recently, and particularly here in the United States, population growth, coupled with generally unrestricted private property rights, has pushed urban growth back out beyond the urban fringe into suburban, and increasingly exurban areas of low relative density (Theobald 2005, 2001). The result is a cultural landscape transformed by increased subdivision of large, potentially agriculturally productive parcels into smaller, more expensive, low-density, residential and commercial purposed tracts (Sorice et al. 2012). Consequently, the natural landscape is increasingly degraded by roads and urban infrastructure and is less able to provide ecosystem services (Lindenmayer and Fischer 2006; MA 2005; McKinney 2002).

Based on analysis of the US EPA's ICLUS model projections (EPA 2009), and using a population density classification scheme developed by Theobald (2005), similar patterns of suburban and exurban transformation are expected to continue within the study area.

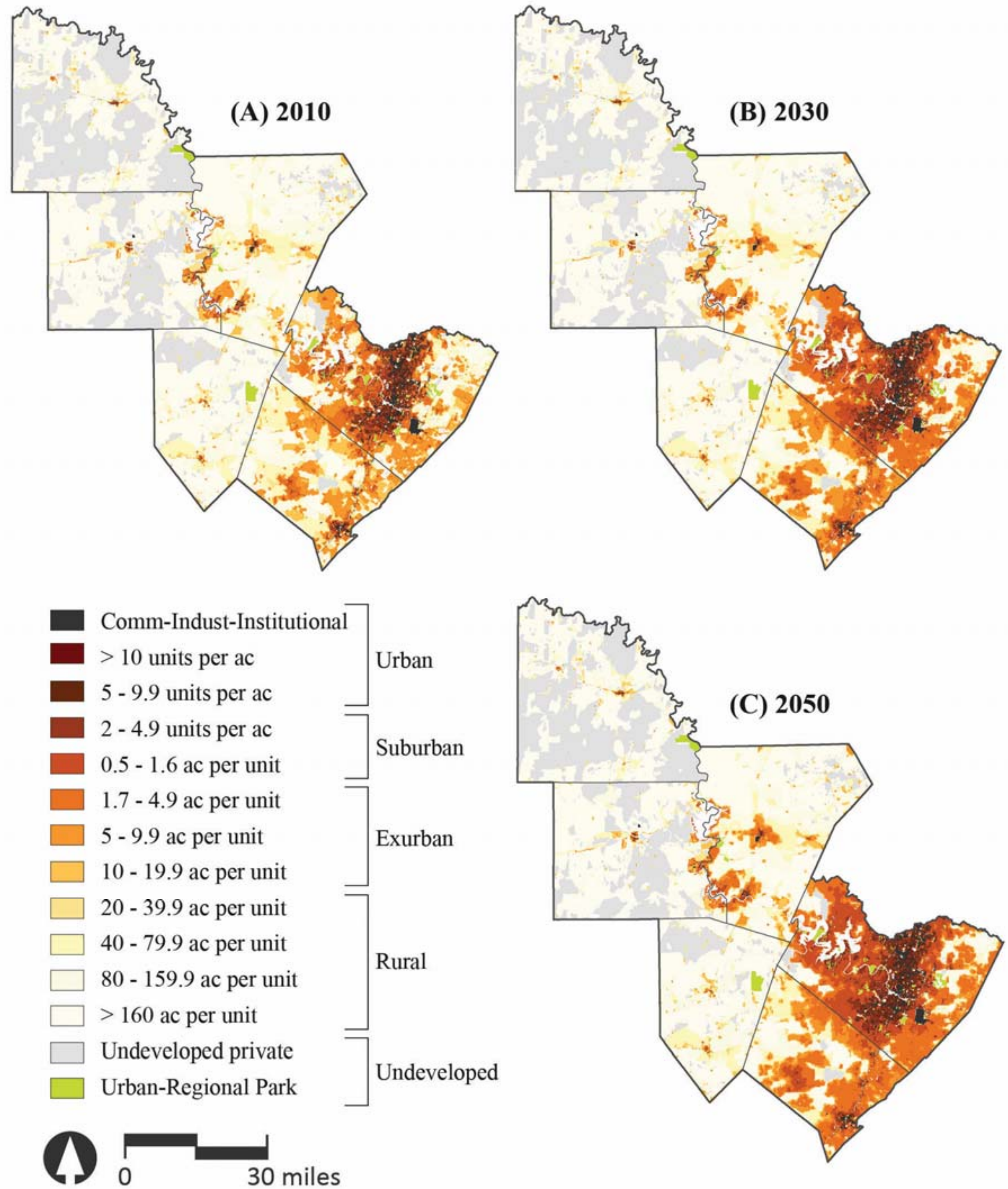

Figure 2.6 Projected Population Density
Projected population density within the study area for the years (A) 2010, (B) 2030, and (C) 2050. Source: US EPA (2009).

As of 2010, areas classified as *Urban* (5 units per acre to >10 units per acre including commercial, industrial, and institutional) equaled 2% of the study area; *Suburban* (1.6 acres per unit to 2 units per acre) equaled 3%; *Exurban* (1.7 to 19.9 acres per unit) equaled 14%; *Rural* (20 to >160 acres per unit) equaled 59%; and *Undeveloped* (undeveloped private and urban and regional parks) equaled 20%. By 2050 the amount of acres classified as *Urban*, *Suburban*, and *Exurban* are expected to increase by 13%, 183%, and 12%, respectively. Meanwhile, *Rural* and *Undeveloped* land densities are projected to decrease by -11% and -2%, respectively.

Unless current trends are altered, these projections suggest the continued loss of native habitat and rural cultural identity to urban development, specifically within the locales most critical to the production of water related ecosystem services—the aquifer recharge zones and reservoir watersheds of Travis, Hays, and Burnet counties. This declining capacity to provide ecosystem services could have negative implications, not only in terms of local ecosystems, but also for local economies (Farber et al. 2006; Haberl et al. 2006).

Local Economics

According to 2008 data, local economic measures such as “Gross Regional Product” (GRP) and “Top Industries” tend to reflect these same patterns of population growth, development, and land-use within the study area (IMPLAN 2008). Overall, the study area has a GRP of almost $70 billion with technology manufacturing the leading industry by output, and state and local governments foremost by employment.

Home to the state capital, several universities, including the University of Texas at Austin, and a majority of the state’s technology sector, the Travis County economy accounts for over 92% of the area’s GRP. Hays County, with a significant public university, large food service industry, and considerable construction sector, follows at 5.2% of area GRP. Burnet (1.6%), Llano (.5%), Blanco (.5%), and San Saba (.15%) combined account for the remaining ~3% of GRP, with the agricultural sector being a top five industry, in terms of employment, in each.

However, when viewed by economic output, agriculture is a top five industry within San Saba County alone. Rental activity, real estate, construction, and food service—together with unique sectors like manufacturing in Burnet and electrical power generation in Blanco—tend to now be the biggest economic drivers within these four, predominately rural counties.

These broad economic data tend to support the research of Sorice et al. (2012), who found that over one-third (39%) of Central Texas landowners hold their property for amenities such as recreation, aesthetic quality, or to experience the rural lifestyle, rather than exclusively for agricultural production (24%). This suggests the study area's rural economies are increasingly less dependent on agriculture, but perhaps more dependent on sustaining a rural cultural identity in order to maintain their growing service based industries (Sorice et al. 2012).

CONSERVATION

As I argue in Chapter 1, conservation is the bridge between the ecological and social systems and is therefore an intrinsic component of landscape sustainability. Approximately 60,727 hectares (150,060 acres), or 4.2% of the study area is presently under some form of legal conservation status with the primary drivers being: endangered species and biodiversity, recreation and aesthetics, cultural heritage, and increasingly, ecosystem services—principally in terms of water. In the following sub-section, I review the state of conservation within the study area in regard to both public and private conservation.

Table 2.4 Conservation Holdings
Source: TPWD (2013) and TLTC (2013)

Land Holder	*Hectares*	*Acres*	*% Total Area*
Federal	9,016	22,279	0.63
State of Texas	5,935	14,666	0.41
County	6,052	14,955	0.42
Municipal	11,771	29,087	0.82
River Authority	4,475	11,058	0.31
Private	23,478	58,015	1.64
Total	60,727	150,060	4.23

Public Conservation

In total, public conservation accounts for over 37 thousand hectares (92,044 acres), or ~2.6% of the study area. In this work, *public conservation* includes publicly owned parks, preserves, fee simple holdings, and conservation easements held by public institutions at any level of government. Although the property under conservation easements technically remains privately owned, if the easement is owned by a public entity it is considered public conservation for the purposes of this work.

Federal holdings within the study area are not numerous, but they are substantial. For example, the US Fish and Wildlife Service holds approximately ~9,015 hectares (~22,279 acres) of the Balcones Canyonlands National Wildlife Refuge, which was established to protect critical habitat of the endangered golden-cheeked warbler (*Setophaga chrysoparia*) and black-capped vireo (*Vireo atricapilla*).

The State of Texas is also a large contributor to public conservation. State owned holdings account for almost 6,000 hectares (14,700 acres). Examples include state parks such as Colorado Bend and Pedernales Falls, state natural areas such as Enchanted Rock, nature preserves such as Discovery Well Cave, and state historic sites such as the Lyndon B. Johnson homestead.

However, when combined within the study area, county, municipal, and quasi-governmental authorities account for the largest percentage of public conservation. Together these institutions have holdings of over 17 thousand hectares (42,050 acres), including over 10,500 hectares (26,000 acres) held by the City of Austin, and almost 4,500 hectares (11,057 acres) owned by the Lower Colorado River Authority (LCRA).

Private Conservation

In total, private conservation accounts for over 23 thousand hectares (58,015 acres, or ~1.6% of the study area. In terms of this work, *private conservation* is defined as privately owned parks and preserves, and fee simple holdings and conservation easements held by non-profit land trusts. Although private owners, home owners associations (HOAs), and non-profit organizations such as the Travis County Audubon Society all play a role in private conservation, it is the non-profit land trust that is the primary contributor.

Based solely on hectares conserved, land trusts and conservation easements are the principal components of private conservation within the study area—and are likely to continue to be so in the future. Including fee simple ownership and conservation easements, land trust holdings combine for almost 13,100 hectares (32,341 acres) of conserved land. Land trust held conservation easements alone account for over 11,300 hectares (~28,000 acres).

Over the last fifteen years HCC has acquired upwards of 3,200 hectares (8,000 acres) in both fee simple ownership and conservation easements. In an effort to strategically expand their conservation footprint, HCC commissioned a study in the fall of 2013 with the explicit intent of identifying conservation priorities, based on organizational goals, inside which HCC could proactively focus its resources over the next twenty years (Siglo 2013). As presented in Chapter 3, the systematic conservation planning methods of this thesis are an expression of HCC's objective to continue to increase private conservation within the Hill Country of Central Texas.

CHAPTER 3:
MATERIALS AND METHODS

In this chapter, I describe the materials and methods used to answer the following two research questions: 1) Where within the contiguous Central Texas Counties of Blanco, Burnet, Hays, Llano, San Saba, and Travis should Hill Country Conservancy focus conservation efforts in order to efficiently balance regional conservation need with organizational goals; and 2) How can multiple ecosystem services be integrated as indicators in a systematic conservation planning process?

Three primary areas of research characterize this work. For organizational clarity, each is addressed in its own subsection: 1) *Systematic Conservation Planning*; 2) *Ecosystem Services*; and 3) *Analysis*.

SYSTEMATIC CONSERVATION PLANNING

This portion of the study was developed in conjunction with a strategic conservation plan commissioned by Hill Country Conservancy (HCC) in the Fall of 2013 (Siglo 2013). Following the protocols established by Margules and Pressey (2000), and using Marxan decision-support software (Ball et al. 2009), a systematic conservation planning (SCP) ranked selection process was applied across the six Central Texas counties of Blanco, Burnet, Hays, Llano, San Saba, and Travis. The process was conducted iteratively over several months with the explicit intent of identifying priority areas, over a range of conservation scenarios, within which HCC could focus its resources over the next twenty years. Similar to The Nature Conservancy's conservation approach (TNC 2004), the strategy of HCC was not to emphasize rarity, but rather to target a comprehensive representation of intact and functional ecosystems, specifically, those critical to maintaining water supply and quality.

Here I use the data, selected conservation elements, targets, scenarios, and software parameters developed during the HCC planning process as a foundation for additional integrative research. This serves not only to justify potentially subjective conservation decisions, such as target levels, but also to ground this study in real-world applicability.

Data

A broad range of geographic information systems (GIS) data and procedures were used during the SCP portion of this study. State, county, and municipal boundaries, populated places, and public open space data were sourced from the Texas Natural Resources Information System (TNRIS 2013). Data regarding private open space were provided by the Texas Land Trust Council in the form of the Conservation Land Inventory database (TLTC 2013). The environmental and biotic data used were acquired from a variety of sources (Table 3.1). Once obtained, all data were converted to the NAD1983 UTM zone 14N (meter) projection, evaluated for consistency, and tested for usability using ESRI's GIS software ArcMAP 10.0—hereafter referred to as ArcGIS. The data were then categorized as either *Water*, *Ecology*, or *Culture* depending on the most relevant representation.

Table 3.1 Environmental and Biotic Data

Category	*Description*	*Source*	*Type*
Water	Significant Waterway Buffer	TCEQ, TPWD	Vector
Water	Public Drinking Water Intakes	TFS	Vector
Water	Colorado River Watersheds	USGS NHD	Vector
Water	Aquifer Areas	TWDB	Vector
Water	Springs	USGS NHD	Vector
Water	Wetland	USFWS NWI	Vector
Ecology	Texas Ecological Systems Phase 1	TPWD	Vector
Ecology	Significant Slopes	USGS NED	Raster
Ecology	GCWA Habitat	Loomis Partners	Vector
Culture	Viewshed	Texas State University	Vector
Culture	Prime Farmland Soils	USDA-SSURGO	Vector
Culture	Available Water Content	NRCS-STATSGO2	Vector
Culture	Soil Depth	NRCS-STATSGO2	Vector

Planning Units

Selecting priority conservation areas requires the use of an evaluation unit to make comparisons between locales. These spatial units are known as *planning units*. Planning units can either be uniform shapes, such as square or hexagonal grids, or irregular shapes derived from land tenure parcels, watersheds, or habitat remnants. Previous research has shown that hexagonal units are slightly more efficient than square, and smaller units tend to be more efficient than larger ones—albeit similarly precise (Nhancale and Smith 2011). In general, the differences in size and shape are less important than selecting planning units which take into account underlying land-tenure patterns, intended implementation of conservation action, and computer processing constraints (Nhancale and Smith 2011).

Planning units composed of 500m by 500m, or 25 hectare (~62 acres), grid cells were selected for use in this study. At 25 hectares, the planning units are a compromise between the study area's mean parcel size of 13 hectares (~32 acres), HCC's preferred minimum of 40 hectares (~100 acres) for potential conservation priority areas, and the generation of solution sets that remain computationally flexible. Moreover, square grid cells were selected over hexagonal, or irregular cells, because of the integrative nature of this work. Specifically, raster data—spatially explicit data represented as a grid of square cells—is more precisely scaled and integrated, with less overlap error, to square planning units.

In total the study area has been divided into 57,479 planning units. Before ranking the planning units, it is necessary to categorize each individual unit according to their present land use and land management status. This study uses three categories: *available*, *open-space* ("locked-in"), and *transformed* ("locked-out").

Planning units with greater than 50% of their area in existing reserves, parks, conservation easements, or other conservation management status were defined as *open-space* and "locked-in" the solution. These open-space sites are critical to the ranking process because they act as catalysts for the selection of other sites near their boundary. Utilizing a selection threshold based on a simple majority of a planning unit's area ensures that units covering a portion of a reserve, but less than a majority, remain available for selection. These *adjoining* units, and the landholdings they represent, are high priorities for conservation organizations, including HCC, because they can be combined with existing reserves to produce large contiguous areas of open-space.

Under this reasoning, it could be argued that increasing the open-space threshold to greater than 60% would allow a larger number of adjacent units to be available for selection. However, at this threshold only 51,385 hectares (126,950 acres), or 85% of the 60,727 hectares (150,060 acres) of actual open-space, is accounted for in the planning units. Whereas a greater than 50% threshold represents 57,500 hectares (142,086 acres), or 95% of the actual area. Thus, a greater than 50% threshold represents a balance between area accuracy and selection availability of reserve-adjacent planning units.

Conversely, a large percentage of the study area has already been developed, or significantly altered from its natural state, and should not be considered for conservation purposes. These areas are considered *transformed*. Parcels smaller than 20 acres, the TPWD ESD Urban high and low land-use classifications (2010), and a manual evaluation of the 2010 NAIP aerial imagery were used in combination to designate areas as transformed. Planning units including 30% or more transformed areas were excluded from consideration and "locked-out" of the solution set. The use of 30% (~20 acres) represents a precautionary approach based on spatial diffusion theory and the assumption that continued development is more likely to occur given proximity to existing development (Theobald and Hobbs 1998). In other words, if a third of a planning unit is presently transformed, then the probability of further development within that planning unit is greater, thus reducing its overall conservation value, as per HCC's criteria.

In addition, planning units with 50% or more of their area classified as *open water* were also "locked-out." The decision to "lock-out" open water was made after preliminary ranked selections demonstrated that, if classified as available, open water disproportionately accounted for the majority of chosen units. Similarly, if classified as "locked-in", results indicated a heavy bias toward units directly adjacent to open water, to the exclusion of other targets.

Table 3.2 Planning Unit Classification

	Planning Units	*Ha*	*Acres*	*% of Area*
Study Area	57,479	1,436,975	3,550,840	100.00
Open Space[1]	2,300	57,500	142,085	4.00
Open Water[2]	616	15,400	38,054	1.07
Transformed[2]	10,201	255,025	630,180	17.75
Available	44,362	1,109,050	2,740,520	77.18

[1] Planning Units classified as Open Space are "Locked-in" and are included as part of every solution.
[2] Planning Units classified as Open Water and Transformed are "Locked-out" of any potential solution.

Area-of-Occurrence and Conservation Elements

Area-of-occurrence is the overlap of planning units with mapped conservation elements. Derived from the environmental and biotic spatial data (Table 3.1), conservation elements are subsets of the geographic, ecological, or cultural features to be targeted during the selection process. The total number of conservation elements used in this work ranged from 58 to 62, depending on the scenario.

The area-of-occurrence for each conservation element was calculated using the "Tabulate Area" tool in ArcGIS. The final value given representing the total area each conservation element occupied in a particular planning unit with a range from 0 to 25 hectares (0 to ~62 acres). For instance, if a particular planning unit contained 10 hectares (~25 acres) of the Edwards aquifer recharge zone, then its area-of-occurrence for the Edwards conservation element also equals a value of 10. Determining the area-of-occurrence for each conservation element, per each planning unit, is the first step in developing quantifiable conservation targets.

Targets

Conservation targets are the desired percentage of the total area-of-occurrence of a conservation element to be included in the solution set. For example, if a conservation element occurs over 100 hectares (~247 acres), and the target for that element is set at 50%, the solution set should contain 50 (~124 acres) hectares of that conservation element.

Target levels for individual conservation elements were developed in collaboration with local experts and HCC board and staff members in an iterative fashion over several months. Several variables influenced target determination including: 1) *Total area*: HCC was interested in identifying approximately 25% of the study area as a conservation priority; 2) *Organizational priorities*: HCC's primary project objective was isolating ecologically functional land critical to sustaining the City of Austin's water supply and quality. Secondary concerns included maintaining cultural heritage and aesthetics, as well as protecting rare and endangered species and ecosystems; 3) *Relative area*: because the targets applied in this work are percentage based, target levels for each conservation element were proportionately scaled to relative area-of-occurrence; 4) *Relative rarity*: a rarity value for all TPWD derived ecosystem types was obtained from botanist and rare plant specialist Jason Singhurst of TPWD. This value, ranging from 1 (common) to 5 (rare), represents the relative rarity of the ecosystem type across its entire range, not exclusively within the study area; 5) S*olution feasibility*: At least 95% of all targets must be met in the Marxan solution for it to be considered feasible; and finally, 6) *Indispensable patterns*: indispensable patterns are the top-priority ecological patterns whose benefits have no known substitute (Forman 1995). These include "a few large natural vegetation patches, wide vegetation corridors protecting water courses, connectivity for movement of key species among patches, and small patches and corridors providing heterogeneous bits of nature throughout developed areas" (Forman 1995).

The final target set, represented here as the *HCC Target Set* (*HCC*), was selected from multiple preliminary target ranges according to how well they fit HCC's criteria. Final values in the *HCC* set range from 5% to 80%, as seen in Table 3.3. The *HCC* target set provided the base from which additional, alternative scenarios were generated. Refer to Appendix A for a full range of conservation targets.

Table 3.3 Abridged Conservation Targets

Category	*Conservation Element*	*HCC*	*Water*	*Agriculture*	*Ecology*
Water	Aquifer Areas	10 - 40%	15 - 60%	10 - 40%	10 - 40%
Water	Spring Buffers	40%	60%	40%	40%
Water	Wetlands	30%	45%	30%	30%
Water	Significant Stream Buffers	50%	75%	50%	50%
Water	Major Waterway Buffer	30%	45%	30%	30%
Water	Public Drinking Water Intake Watersheds	30%	45%	30%	30%
Water	Colorado River Watersheds	30 - 40%	30 - 40%	30 - 40%	30 - 40%
Water/ Ecology	Texas Ecological Systems Database	5 - 80%	5 - 80%	5 - 80%	7.5 - 80%
Water/ Ecology	Significant Slopes	15 - 40%	23 - 60%	15 - 40%	23 - 60%
Ecology	GCWA Habitat	10 - 30%	10 - 30%	10 - 30%	15 - 45%
Culture	Viewshed	25%	25%	25%	25%
Culture	Prime Farmland Soil	30%	30%	45%	30%
Culture	Available Water Content	5%	5%	7.5%	5%
Culture	Soil Depth	5%	5%	7.5%	5%

Scenarios

In total, four scenarios, each targeting 58 distinct conservation elements, were developed for analysis: 1) *HCC*; 2) *Water*; 3) *Agriculture*; and 4) *Ecology*. Building on the *HCC* target set, each additional scenario was created by raising the targets within a discreet category, i.e., water, culture, or ecology, by a fixed factor of 50%, while maintaining all other target values (Table 3.2).

To determine the factor of increase, a sensitivity analysis was conducted examining a range of increase values (i.e., 10%, 25%, 50%, and 75%) per each scenario. The 50% value was ascertained by charting the standardized Z-scores of *target shortfall*[2] and *spatial correlation*[3]. As the range of target values increases from 10% to 75%, *target shortfall* increases while *spatial correlation* decreases, with the point of intersection, i.e., the point of optimal compromise between minimizing target shortfall and maximizing scenario differences, around the 50% value in all three scenarios.

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[2] Target shortfall is the amount of targets not achieved in the Marxan solution. Calculated as the average shortfall of 100 solutions.

[3] Spatial correlation is the relative difference in spatial distribution and overlap between the most frequently selected 11,050 units in the *HCC* scenario and the most frequently selected 11,050 units per scenario (x). Calculated using Spearman's correlation with R statistical software.

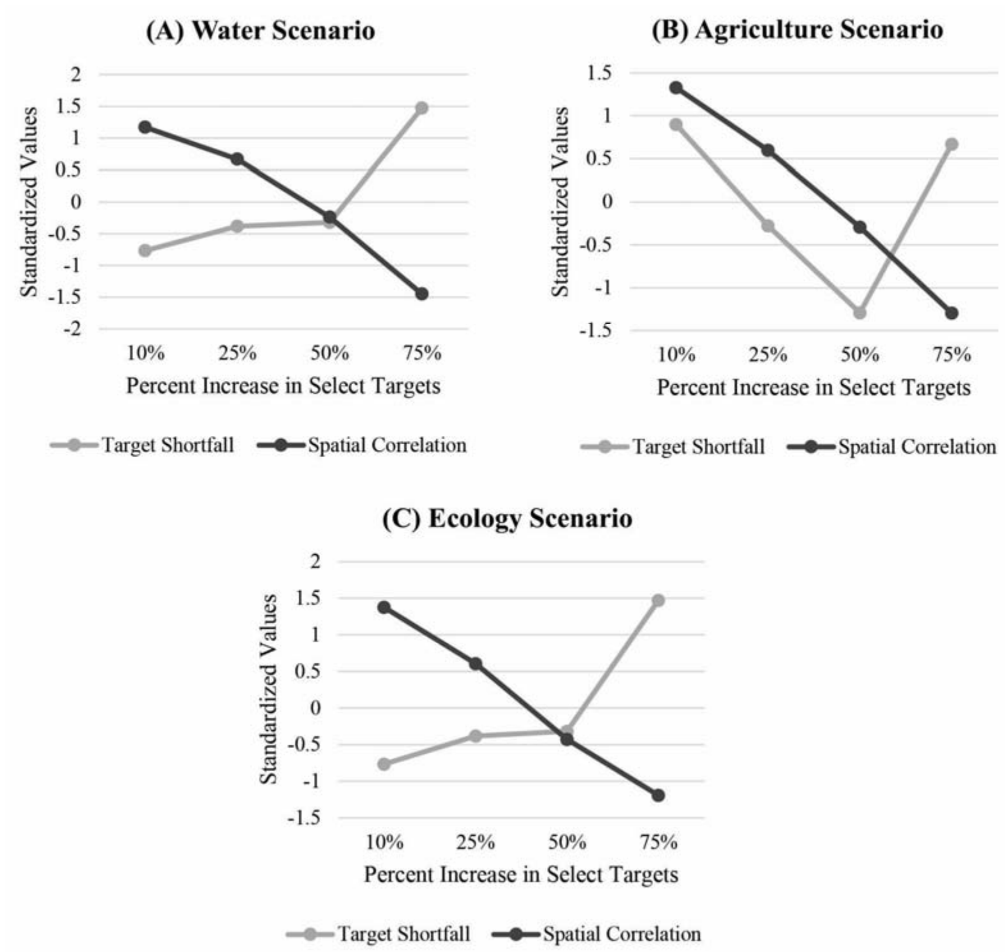

Figure 3.1 Sensitivity Analysis Results
Sensitivity analysis results testing range of target increases for conservation scenarios: A) Water Scenario, B) Agriculture Scenario, and C) Ecology Scenario.

Scenario 1: HCC

The *HCC* scenario represents the baseline where all of HCC's project criteria were met. It must be noted that because water protection was HCC's primary concern in this study, the *HCC* scenario was developed to be water-centric.

Scenario 2: Water

Pushing the *HCC* scenario's focus on water a step further, the *Water* scenario is intended to maximize the conservation of sensitive water elements and features such as aquifer recharge zones, significant stream buffers, and wetlands. All targets within the water and water/ecology categories, excluding the coarse filter *Colorado River Watersheds*, were raised by a factor of 50% above the *HCC* target levels.

Scenario 3: Agriculture

The *Agriculture* scenario represents HCC's priority to conserve rural cultural heritage within the Central Texas Hill Country. In this scenario, targets for elements such as grasslands (a surrogate for potential livestock production), and targets associated with agricultural production, such as prime farmland soil, were raised by a factor of 50% above the *HCC* target levels.

Scenario 4: Ecology

In the *Ecology* scenario, priority was given to rare and sensitive ecosystems, endangered species habitat, and critical biophysical components such as significant slopes. All ecology and water/ecology related targets were raised by a factor of 50% above the *HCC* target levels.

Marxan Parameterization

One of the advantages to using Marxan is the wealth of both general documentation and published research. As such, the parameters used here were applied with a heavy reliance on the *Marxan Good Practices Handbook v2* (Ardron et al. 2010) and reinforced through the academic literature (Chan et al. 2006; Izquierdo and Clark 2012; Nackoney and Williams 2013). Significant parameters include the "Boundary Length Modifier" (BLM), the "Penalty Factor" (PF), and "Planning Unit Cost" (Cost) value (Table 3.4).

Table 3.4 Marxan Parameterization

Parameter	*Value*	*Description*
Boundary Length Modifier (BLM)	20	Controls the amount of clustering
Penalty Factor (PF)	1-15	General weight given to CEs
Planning Unit Cost (Cost)	1	Cost value assigned per planning unit
Iterations	1 mil	# of Simulated Annealing iterations
Repetitions, or Restarts	100	# of separate runs

The BLM controls the level of planning unit aggregation, or "clustering," of the solution by minimizing the reserve system boundary length. A low BLM results in more fragmented solutions. As BLM values increase, solutions become more compactly clustered with lower perimeter-to-area ratios. Here, a BLM of 20 was determined by following a process outlined in Ardron et al. (2010). Starting at a value of 0, the BLM is increased incrementally by factors of 10 until a desired level of clustering is reached, i.e., clusters smaller than 40 hectares (~100 acres) are minimized, without incurring large increases in cost.

The PF determines the size of the penalty for not meeting a conservation element's targets. The higher the PF value, the greater importance Marxan places on ensuring the element's targets are met. PF values in this analysis range from 1 to 15. Again following Ardron et al. (2010), these values were calibrated by first finding a uniform SPF value for which all targets are met, and a lower value in which most are not. The difference between these values represents the range explored for each conservation element. All PF values are then set to the low value (1), and, for those features missing their targets, increased iteratively in increments of 5 until all targets are met.

The "Planning Unit Cost" (Cost) is a relative value assigned to each planning unit. Cost values can denote socioeconomic cost, relative area, level of degradation, distance from threat, or a combination of multiple metrics combined into a single cost surface (Ardron et al. 2010). Here, a cost value of (1) was universally applied across all planning units. The primary reason being to reduce complexity in the selection algorithm. Thus, because the conservation targets are the sole driving determinate of selection, any differences in results can be more easily understood. According to Marxan best practices, this is a common technique, especially when using uniform planning units (Ardron et al. 2010).

Additional Marxan parameters of note include "iterations" and "repetitions." Iterations represent the number of times the simulated annealing algorithm is applied during a single run and determines how close the algorithm gets to the optimal solution. For all but the largest of datasets, 1 million iterations is sufficient (Ardron et al. 2010). Repetitions, also referred to as restarts, are the number of independent solutions to the reserve problem Marxan will generate. Using 100 repetitions is an intuitive value from which to calculate selection frequency, thus it was used here. All other parameters were left as the default values.

Solution Sets

Marxan results are typically viewed and analyzed in one of two ways: as a "Best Solution" or as the "Summed Solution." The "Best Solution" represents the single run that achieved the most targets for the least cost. However, the "Best Solution" is only one of many potential good solutions, any one of which would create an adequate reserve network. Instead of representing target achievement per cost, the "Summed Solution" is a measure of the frequency of selection across all runs—essentially Marxan's measure of "irreplaceability." For example, assuming 100 restarts, a planning unit with a "Summed Solution" value of 90 indicates a unit that was selected 90 out of 100 restarts, thus representing a 90% selection frequency.

The nature of the conservation problem being addressed typically dictates whether the "Best" or "Summed" solution is used. The "Summed Solution" is best suited to address the problem presented in this study because the aim of HCC is to identify conservation priority areas with the most unique and irreplaceable features across a broad suite of categories on an ad hoc basis, rather than to isolate a near optimal reserve network based on constraints.

Using Marxan's "Summed Solution" output, the final solution sets represent 11,050 planning units (25% of all "available" units) selected with the highest frequency across 100 runs per each scenario, i.e., *HCC*, *Water*, *Agriculture*, and *Ecology*, respectively.

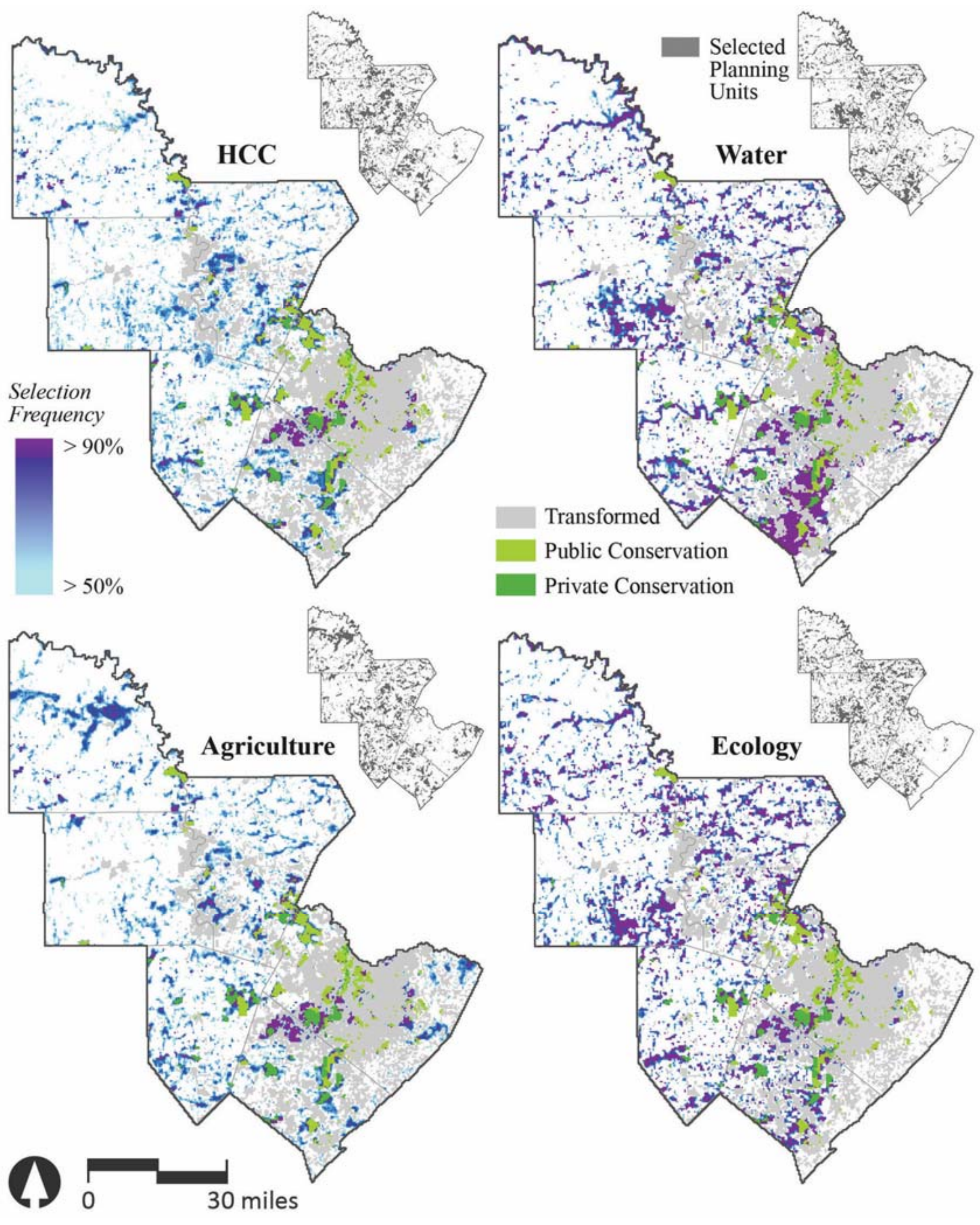

Figure 3.2 Systematic Conservation Planning Scenarios
The four systematic conservation planning scenarios derived using Marxan decision-support software (Ball et al. 2009). Large maps illustrate selection frequency, the smaller maps show the top 11,050 planning units selected per each scenario.

Limitations

Key limitations exist for both systematic conservation planning in general, and Marxan in particular. In terms of SCP, the problems typically addressed are inherently complex involving highly interactive biological, environmental, social, and economic variables. These variables are often represented by incomplete or limited datasets. Consequently, there is a large degree of uncertainty at every stage in the process, with the quality of solutions a reflection of both the level of present knowledge and the quality of the data used (Margules and Sarkar 2007). An inability to easily integrate stochastic or temporally dynamic data, such as the ecological consequences of climate change, as well as the restriction to only a single cost surface are constraints specific to Marxan (Ardron et al. 2010). As a result of these limitations, it is important to stress that both SCP and Marxan are decision-support tools, not decision-making tools. Their use must be viewed with a critical eye as part of a dynamic, iterative process in which real world decisions are made under knowledge, data, time, and resource constraints.

Nevertheless, SCP and Marxan remain well-suited to answering the questions posed in this work for a number of reasons. First, because they are based on area-of-occurrence, the conservation elements being assessed are comparatively simple and do not explicitly include such hard to quantify measures as future economic land values or exact number and locations of rare species. Second, the study happens to be located within an area where accurate, standardized data is easily accessible, detailed ecological interactions are well-documented, and many dynamic processes, such as population growth and urbanization, are relatively predictable. Finally, although the study assumes a twenty year time horizon, its precise purpose is to locate priority conservation areas as they exist in the present. Thus, although the potential consequences of climate change are important to understand and consider, they do not necessarily impact the selection of conservation areas as they relate to the parameters set forth in this work. Moreover, because SCP and Marxan are designed to be a part of an iterative process, new or revised variables can be incorporated as they become available.

ECOSYSTEM SERVICES

Since the publication of the Millennium Ecosystem Assessment in 2005, there has been a steady increase in research aimed at developing tools to quantify and geospatially map ecosystem services. In the following subsection, I use one of these tools to map three measures of ecosystem services—along with a separately derived measure of ecosystem richness. The aim is to use these metrics as performance measures, or indicators, of the four conservation scenarios described in the preceding subsection.

InVEST

Developed as part of the Natural Capital Project, InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) is a suite of spatially explicit simulation models used to generate estimates of the levels and economic values of ecosystem services (Tallis et al. 2013). In the InVEST models, inputs of land-use, land cover, land management, and environmental conditions are combined with biophysical and/or economic information in ecological production functions from which, and depending on research need and data availability, the spatially explicit biophysical, or monetary, ecosystem service estimates are obtained. InVEST was selected over other options because it is freely downloaded, integrates easily with ArcGIS, requires relatively few inputs, and is well documented in both use guidelines (Tallis et al. 2013) and applied research (Izquierdo and Clark 2012; Nelson et al. 2009; Tallis and Polasky 2009).

Ecosystem Service Metrics

In this study, InVEST is used to estimate and map ecosystem services across the six-county Central Texas study area. Three terrestrial models were chosen for this application: 1) *carbon storage*, 2) *soil conservation*, and 3) *water provision.* These three services were selected because their use is well represented in the academic literature (Bai et al. 2011; Chan et al. 2006; Izquierdo and Clark 2012; Sánchez-Canales et al. 2012; Nelson et al. 2009), they tend to show a positive correlation between services (Bai et al. 2011; Chan et al. 2006), and they are emblematic of the social and ecological phenomena HCC is targeting in its *Strategic Conservation Plan* (Siglo 2013). Although each InVEST model also estimates economic value, the valuation estimates tend to be highly variable, with little empirical reinforcement. Thus, they were not included in this analysis.

In addition to the three InVEST ecosystem services metrics, a measure of *Ecosystem Richness* (ER) was also calculated to test the relationship between ecosystem services and biodiversity within the study area. This is beneficial to the analysis because prior research has demonstrated that measures of ecosystem services and biodiversity are not always positively correlated, with weak, and even negative, correlations common (Chan et al. 2006; Cimon-Morin et al. 2013). Therefore, understanding the relationship between ecosystem services and biodiversity is critical to making informed decisions.

Data

Each InVEST model requires numerous data inputs, including biophysical value tables and spatially explicit environmental raster and vector data. Prior to use in InVEST, and using ArcGIS, all spatial input data were projected to NAD1983 UTM zone 14N, and clipped to the dissolved planning unit outline. Additionally, all raster data were aligned with the planning unit cells. For a complete list of ecosystem service metric data and data sources, refer to Appendix B.

The InVEST model allows the user to determine the cell size of the output rasters. For this study, all rasters were given a cell size of 100 m^2 (~1,076 ft^2), or exactly 1 hectare (~2.5 acres). This cell size was chosen for two reasons. First, because most InVEST services are calculated as value of service (x) per hectare, regardless of raster cell size, it is easy to evaluate the output per cell. Second, it incrementally fits the planning unit size, i.e., 25 InVEST raster output cells fit into 1 planning unit cell.

Carbon Storage

In order to estimate the amount of stored carbon in the landscape, the InVEST "Carbon Storage and Sequestration" model couples land-use and land-cover maps with stocks in four carbon pools: aboveground biomass, belowground biomass, soil, and dead organic matter. In this application, sequestration—or the continued accumulation of carbon in plants and soil over time—was excluded in favor of the more simple carbon storage value because of the uncertainty regarding sequestration and the importance of preventing the loss of stored carbon (García-Oliva and Masera 2004; Lövbrand 2004). Outputs from the InVEST carbon storage model are given as tons of carbon stored per hectare and range in value from a low of 0 to a high of 90.

Soil Conservation

The InVEST soil conservation model estimates the capacity of a land parcel to retain sediment. Based on the Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1978), the rate of soil retention is a function of geomorphology, climate, vegetation and management practices (Tallis et al. 2013). Outputs from the soil conservation model are given as tons of soil retained per hectare and range in value from a low of 0 to a high of 4,996.

Water Provision

Using an approximation of the Budyko curve (Zhang et al. 2001), InVEST calculates water provision as the difference between precipitation and actual evapotranspiration on each pixel derived from data on mean annual precipitation, annual reference evapotranspiration, and correction factors for vegetation type, soil depth, and plant-available water content (Tallis et al. 2013). Outputs from the water provision model are given as cubic millimeters of water per hectare and range in value from a low of 22 to a high of 781.

Ecosystem Richness

Ecosystem richness was calculated for use as a biodiversity surrogate using the TPWD Ecosystems Classification Phase 1 Dataset in raster format (TPWD 2010). Using the "Zonal Statistics" ArcGIS tool, the number of TPWD ecosystem types were summed per 25 hectare planning unit. The ER metric offers a straightforward method to identify areas with the largest range of ecosystem types. This is based on the assumption that a large number of ecosystem types represent greater overall biodiversity, as other studies have demonstrated (Ferrier and Watson 1997; Rodrigues and Brooks 2007). Ranging in value from 1 to 28, output from the ecosystem richness metric is given as the number of ecosystem types per planning unit.

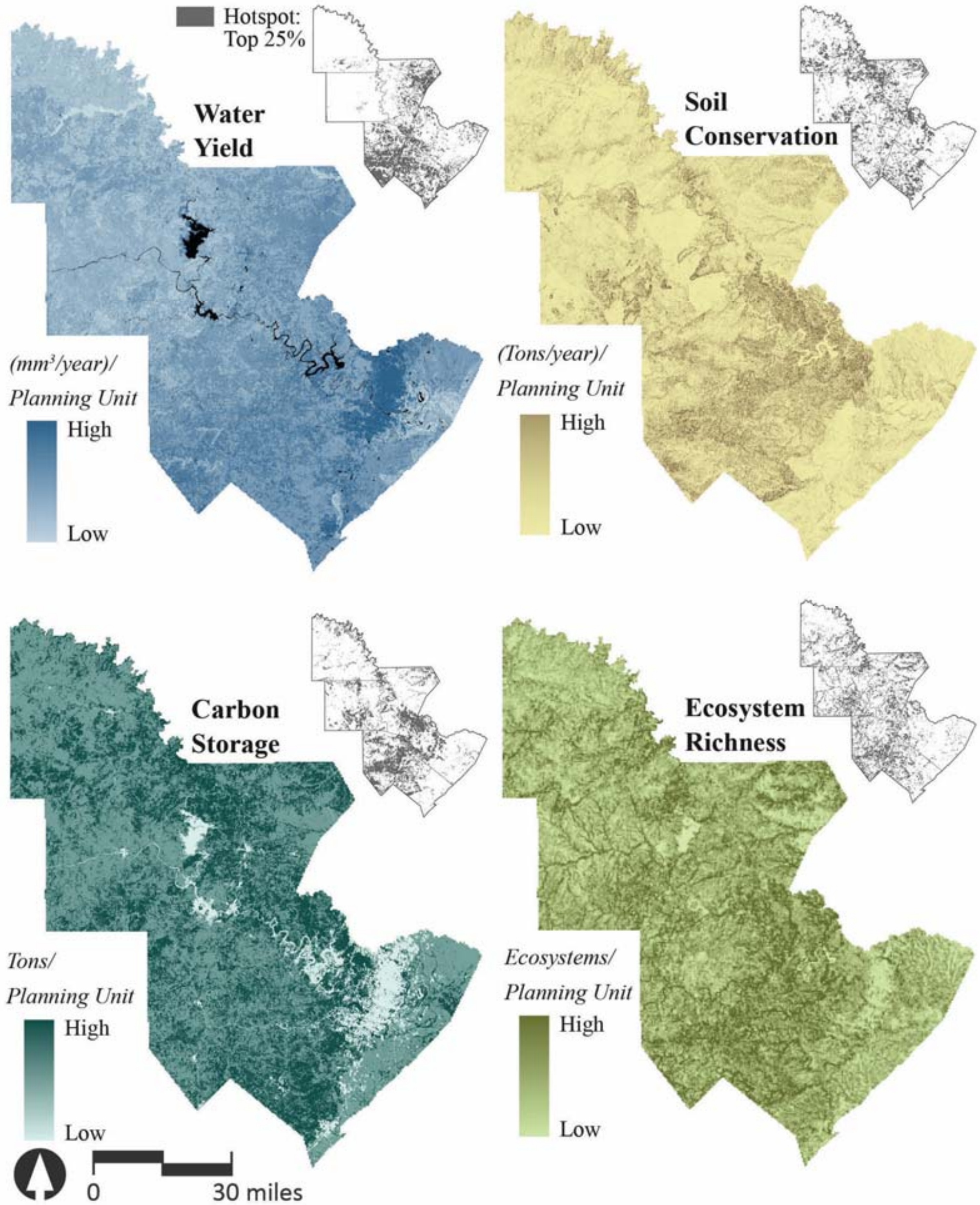

Figure 3.3 Ecosystem Service Metrics
The three InVEST derived ecosystem service metrics and "hotspot" maps, along with the separately measured "Ecosystem Richness" metric and "hotspot" map (Tallis et al. 2013). Hotspots are classified as the richest 25% of planning units for each service.

Indicators

Once calculated, the outputs from the ecosystem service rasters can be tabulated to compare the performance of each of the four conservation scenarios. Using the "Zonal Statistics as Table" ArcGIS tool, each ecosystem service metric was tallied within the planning units of each respective conservation scenario. To use *carbon storage* as an example, the "Zonal Statistics as Table" tool was used to sum the values of the cells in the carbon storage output within each of 11,050 planning units of the *HCC* scenario. The sum of all *HCC* planning units were then totaled. This process was repeated for each scenario until all had a summed carbon storage total, thus allowing a straightforward comparison. The entire process was repeated again for each of the remaining three ecosystem service metrics.

Limitations

The InVEST suite of models have well recognized limitations (Tallis et al. 2013). This is largely the result of a reliance on the mathematical simplification of complex ecological functions and non-standardized data. In general, all InVEST models show high sensitivity toward the LULC classes used in input maps, including spatial resolution and accuracy. Second, the carbon cycle is greatly oversimplified assuming fixed storage levels according to LULC type without considering gains or losses between types or over time. Third, the USLE method used in the soil retention model has only been verified on slopes of 1 to 20 percent and only considers the individual effect of each variable, while in reality, factors often interact, potentially altering erosion rates. And fourth, based on annual averages, the water yield model overlooks extreme events and does account for precipitation seasonality.

Despite these limitations, the InVEST suite of models is useful to this application for several reasons. First, the models are flexible, requiring relatively little data inputs, yet are also well-grounded in science (Tallis and Polasky 2009). Second, although steep slopes and extreme events do occur within the study area, average slope and seasonality are both relatively low. Finally, this analysis is not intended for devising detailed ecosystem service management plans, but rather for their potential use in evaluating conservation scenarios.

ANALYSIS

In this subsection, I describe the methods of analysis used to evaluate the conservation scenarios and the ecosystem service indicators. These methods include overlay analysis using ArcGIS, the creation of a random scenario, and statistical analyses using Spearman's correlations and Wilcoxon-Mann-Whitney U tests (WMW).

Overlay Analysis

In order to determine the spatial distribution of ecosystem service overlaps, both in total across the study area, and as a comparative measure between conservation scenarios, an overlay analysis was performed using ArcGIS. The first step in the process is to isolate the areas with the highest concentration of a particular service. In the literature these areas are often termed *hotspots*, with hotspot generally being characterized as "an area that provides large components of a particular service" (Bai et al. 2011).

Although hotspots have been specifically defined as "5% or less" in some studies (Egoh et al. 2009; Orme et al. 2005), and as the "richest 10% of grid cells for each service" in others (Bai et al. 2011), using either of these thresholds in this study resulted in an insignificant amount of hotspot overlaps—specifically of four-service overlaps. In order to obtain a sufficient percentage of four-service overlaps, the threshold was decreased in increments of five until a more desirable distribution was achieved. Thus, *hotspots* are defined here as the richest 25% of planning units for each service.

In order to calculate the amount of overlap, the sum of the four hotspots were tallied. To do so, all planning units representing a hotspot for each of the four ecosystem service metrics were given a value of (1), with all non-hotspot planning units given a value of (0). The values between each of the four metrics were then summed per planning unit. The result is 57,479 planning units with a value ranging between zero and four. The values from the summed overlap tally were then extracted per each of the 11,050 planning units of the four conservation scenarios, as well as the random scenario, for purposes of comparison.

Random Scenario

In comparing conservation scenarios, it is important to have either a baseline measure, or a control, from which to judge the impact of alternatives. Borrowing a common quantitative method in landscape ecology (Turner and Gardner 1991), in this study a neutral alternative of 11,050 planning units was randomly selected from the 44,362 "available" units—or 25% of all "available" units—using the 'Subset Features' tool in ArcGIS. Hereafter this random selection is referred to as the *Random* scenario.

Statistical Analysis

In addition to the overlay analysis, statistical analyses are also performed. To test the spatial relationship among variables in this study, Spearman's rank correlation is calculated using R statistical software. Spearman's rank correlation is a non-parametric measure of statistical dependence between at least two variables. Separate Spearman correlations are calculated among the four ecosystem service metrics, the four ecosystem service hotspots, as well as among the random scenario and the four Marxan conservation scenarios per each ecosystem service metric. Prior to calculating the correlations among the ecosystem service metrics, the values for each metric were standardized with Z-scores.

Standardizing the values for the scenario correlations is not necessary because they already reflect a standard unit, i.e., the measure of a particular ecosystem service.

Again employing R statistical software, the Wilcoxon-Mann-Whitney U test (WMW) is used to test the hypothesis that the Marxan conservation scenarios are statistically distinct from the random scenario in regards to the ecosystem service indicators. Null hypothesis being: there is no significant shift between the median of the random control and the median of the targeted scenario. The WMW was selected over a conventional t-test because the distribution of most variables are non-normal and a non-parametric test, such as the WMW, is considered more appropriate, and more powerful, in such circumstances (Fay and Proschan 2010). Furthermore, although the distribution of the data could be normalized through a transformation, it is unnecessary when using the WMW test. This is because the WMW test is based only on the rank order of the observations, not on their value. Therefore, any transformation of the values that does not change the order will not affect the statistic.

Limitations

As there were with both systematic conservation planning and ecosystem service quantification, there are also limitations to the methods of analysis. One of the general limitations includes the selection of a study area based on political boundaries, i.e. Texas counties. Because ecological processes do not typically correspond with geopolitical lines, the analyses are applicable only to the defined boundary, not to the broader scale at which the specific processes are taking place. This scale mismatch is especially germane to the analysis of ecosystem services. For example, if the overlay analysis was conducted at the ecoregional level (*Edwards Plateau*), the spatial congruence of ecosystem services may be significantly different, and perhaps the relaxation of the hotspot threshold would not have been necessary.

Nevertheless, the analytic constraints imposed by scale-mismatched boundaries are not specific to this project. Instead, they are typical of planning processes in which areas must be defined according to the scale of the decisions being made and actions being implemented. In this study, the counties selected were based on their inclusion of watersheds deemed critical to Austin-area water quality, as well as the areas within which HCC is most likely to find expansion opportunities outside its historic base of operations. Thus, even though the six-county study area may not be fully representative of the interactions occurring within the Edwards Plateau as a whole, the results remain relevant to the decision-making of HCC.

Similarly, general limitations also exist in terms of the data upon which the analyses are based. Namely, each analysis technique is constrained by the assumption that the underlying data is accurate. Since varying scales of fine-grain spatial data were aggregated into larger areal units, specifically in regards to the creation of the ecosystem service metrics, this assumption must be tempered with the understanding that aggregation of data in this manner may lead to different values and inferences. In geography this is known as the modifiable areal unit problem (MAUP) (Jelinski and Wu 1996). However, because each of the InVEST outputs were aggregated to the planning units individually, and based strictly on the sum of the underlying cells, the MAUP has little—if any—affect in altering the overall spatial relationship of variables in this application.

Limitations specific to the statistical analyses are also noted in this study. Foremost is the random selection of planning units for use as a control (*Random Scenario*). Land-use decisions, including those intended for conservation, do not occur randomly. Patterns of location, time, and scale can be determined from past trends. In turn, projected futures of past trends can be used as plausible baselines for comparing proposed alternatives—a technique common in scenario planning (Vanston et al. 1977). As a result, the random control is limited by its implausibility in regard to actual land-use planning decisions.

However, this limitation does not mean that the random scenario is inappropriate for the purposes of this work. The present rate of conservation acquisition within the study area is insufficient for the creation of a projected, past-trend baseline scenario covering the same amount of area—within the same twenty-year timeframe—as the HCC conservation scenarios, rendering a plausible, yet comparable baseline infeasible. Alternatively, a conservation scenario developed from the criteria of an organization other than HCC—The Nature Conservancy, for example—could be used for purposes of comparison but would otherwise be inadequate as a baseline measure. More importantly, the question being asked is not which conservation scenario provides the most services, but instead, are indicators of ecosystem services effective comparative tools of conservation scenarios. As used in this work, the random control offers an unconstrained starting point from which to answer the latter question.

An additional limitation of the statistical analysis methods is the lack of singular explanatory power. The WMW test only proves whether a shift in the median between the control and conservation scenario is statistically significant. Spearman's correlation only shows the spatial proximity of values. On their own, neither provide conclusive enough evidence to inform land-use decisions. When used in combination—along with spatially explicit techniques such as overlay analysis—the separate limitations of each are mitigated.

CHAPTER 4:

RESULTS AND DISCUSSION

The final research questions posed by this study ask: 1) Can conservation scenarios be effectively evaluated with indicators of ecosystem services; and 2) How can ecosystem service indicators inform conservation decisions? In this chapter, I attempt to answer these questions through a review of the overlay and statistical analysis results, as well as a subsequent discussion of the results and their conservation implications. Consequently, I have divided this chapter into two subsections: 1) *Results,* and 2) *Discussion*.

RESULTS

The work described in this sub-section is the result of multiple methods of analysis. These methods include both ArcGIS based overlay analysis, and statistical analyses in the form of Spearman's correlation and Wilcoxon-Mann-Whitney U tests (WMW). The results from these analyses are presented in the following three categories: 1) *Overlay Analysis*, 2) *Statistical Analysis*, and 3) *Comparing Scenario Results*

Overlay Analysis

Overlay analysis provides a means to determine the spatial distribution of ecosystem service hotspots as well as a method of comparison between conservation scenarios. In total, 23% of the study area include hotspot overlaps ≥ 2, 8% ≥ 3, and less than 1% with 4 the maximum of four overlaps. As seen in Figure 4.1, the greatest proportion of overlaps ≥ 3 are located just west of the Balcones fault zone, beginning at the northwest corner of both Hays and Travis County and extending into the southern portions of Blanco and Burnet. When viewed as a percentage of county area (Figure 4.2), Blanco (20%), Hays (16%), and Burnet (8%) have the greatest proportion of ecosystem service hotspot overlaps ≥ 3. At 3% of its area, Hays represents the greatest proportion of four overlaps, followed by Blanco with 1%.

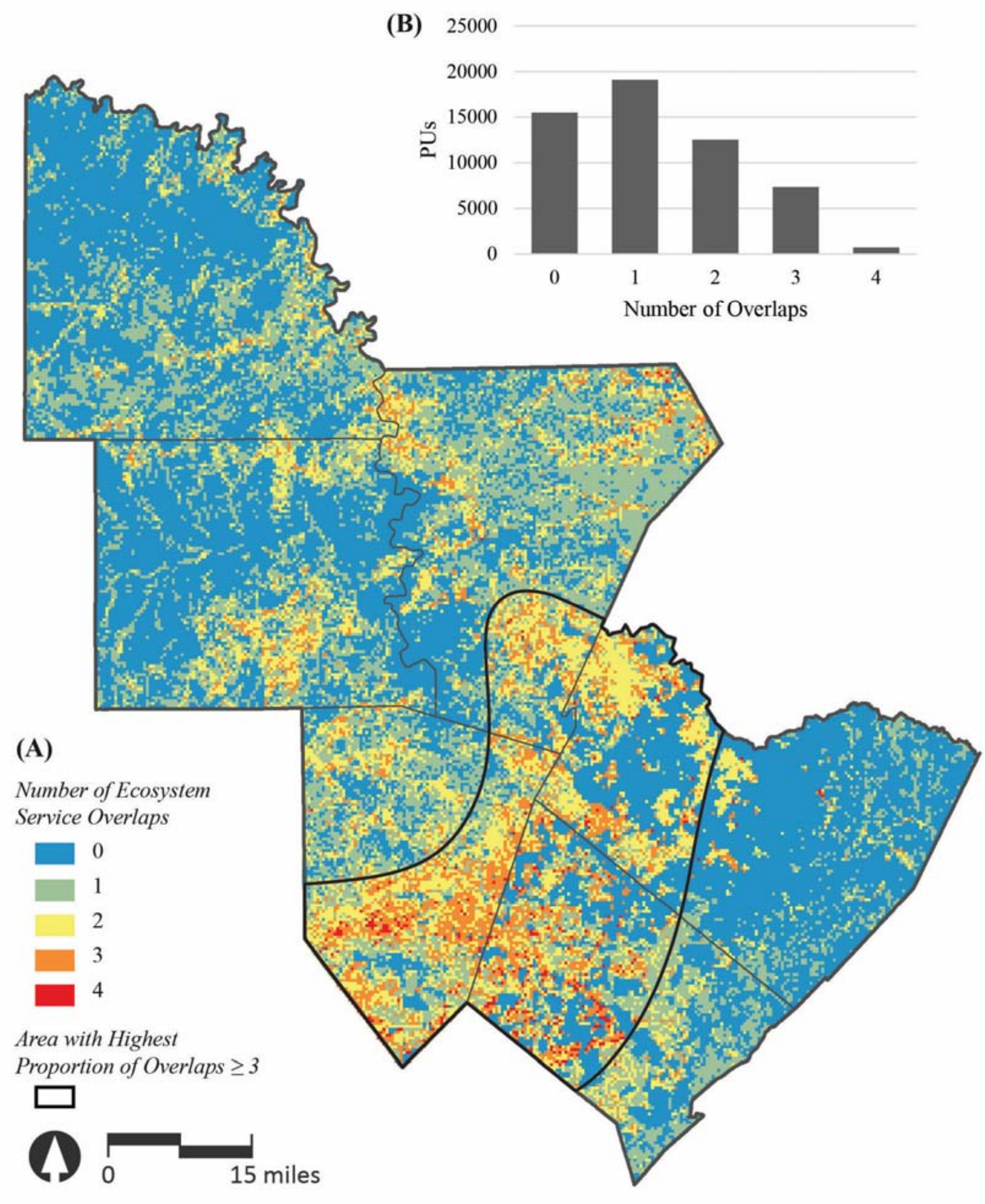

Figure 4.1 Overlay Analysis Results
A) Spatially explicit representation of ecosystem service “hotspot” overlaps. B) Number of overlaps by Planning Units (PUs).

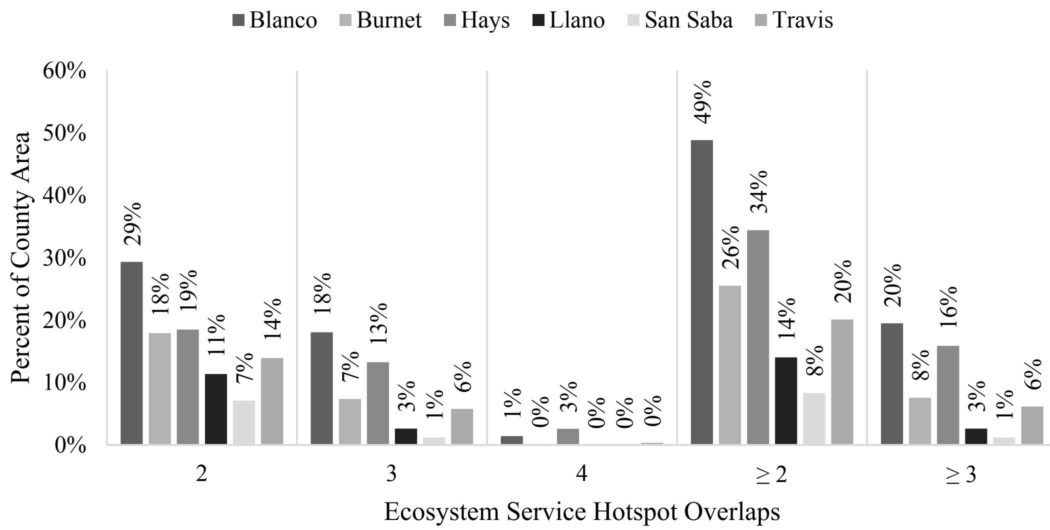

Figure 4.2 Ecosystem Service Hotspot Overlaps by Percent of County Area

In terms of the conservation scenarios, each share a similar hotspot overlap distribution range, as seen in Table 4.1. Hotspot overlaps ≤ 1 constitute approximately 60% or more of each scenario, with overlaps ≥ 3 representing less than 20% for all scenarios, and overlaps of four representing less than 2% for all scenarios. For hotspots ≥ 2, *Water* represents the greatest percentage (43.7%), followed by *Ecology* (42.4%), and *HCC* (41.1%). For hotspots ≥ 3, *Ecology* becomes the top scenario (17.2%), followed by *Water* (17%), and *HCC* (16.3%). Of all four scenarios, *Agriculture* represents the smallest percentage of overlaps ≥ 2 (32.8%), and ≥ 3 (13.2%). Most significantly, all four conservation scenarios were found to have higher overlap percentages ≥ 2 and ≥ 3 when compared with *Random*.

Table 4.1 Percentage of Conservation Scenario by Number of Ecosystem Service Hotspot Overlaps

Overlaps	*Random*	*HCC*	*Water*	*Agriculture*	*Ecology*
None	39.4	24.1	20.7	33.6	22.4
1	33.7	34.8	35.7	33.6	35.2
2	17.3	24.8	26.7	19.6	25.2
3	8.8	14.9	15.3	12.0	15.5
4	0.8	1.4	1.7	1.2	1.7
Total	100	100	100	100	100
≥ 2	26.9	41.1	43.7	32.8	42.4
≥ 3	9.6	16.3	17	13.2	17.2

Statistical Analysis

Statistical correlation analysis tests the spatial correspondence between variables. The results presented here indicate that the four ecosystem service functions, i.e., *Water Yield*, *Soil Conservation*, *Carbon Storage*, and *Ecosystem Richness*, have relatively distinct spatial distributions across the total study area (Table 4.2). The overall average correlation is positive but low (0.09), with the correlation between *Soil Conservation* and *Ecosystem Richness* (0.33), and *Soil Conservation* and *Carbon Storage* (0.32) found to be highest, though still relatively weak. In contrast, a negative correlation was found between *Water Yield* and *Soil Conservation* (-0.29) and between *Water Yield* and *Ecosystem Richness* (-0.19). While unexpected, this spatial incongruence can be explained by the known positive relationship between forested land-cover and both *Soil Conservation* and *Ecosystem Richness*, and the negative relationship between forest cover and *Water Yield* due to increased evapotranspiration and soil retention (Chan et al. 2006).

Table 4.2 Spearman's Correlation between Ecosystem Services

	Water Yield	*Soil Conservation*	*Carbon Storage*	*Ecosystem Richness*
Water Yield	1			
Soil Conservation	-0.29	1		
Carbon Storage	0.16	0.32	1	
Ecosystem Richness	-0.19	0.33	0.17	1

The spatial correlation of the four ecosystem service function hotspots show a similar outcome (Table 4.3). Overall average correlation remains both positive and low (0.15). Notable exceptions include an increase in correlation between the hotspots of *Carbon Storage* and *Ecosystem Richness* (0.32) and a reversal from a negative overall correlation between *Water Yield* and *Ecosystem Richness* (-0.19) to a positive correlation between the hotspots of those same functions (0.14). These deviations can be understood as a matter of geography, namely that the increase in topographic variation and slope found just west of the Balcones escarpment, when coupled with rainfall patterns which increase from northwest to southeast, facilitates greater ecological richness, discourages wholesale clearing of forest cover, while also increasing water yield.

Table 4.3 Spearman's Correlation between Ecosystem Service Hotspots

	Water Yield	*Soil Conservation*	*Carbon Storage*	*Ecosystem Richness*
Water Yield	1			
Soil Conservation	-0.12	1		
Carbon Storage	0.21	0.24	1	
Ecosystem Richness	0.14	0.12	0.32	1

Testing the spatial correlation between scenarios is also crucial to ensure outcomes are based on variation in conservation targets, rather than influenced largely through spatial distribution. Correlation between conservation scenarios, not including the *Random* scenario, is positive with a relatively high average (0.59). Because each scenario was based on the *HCC* target set, this outcome is to be expected. Not surprisingly, the *Water* and *Ecology* scenarios, which share a number of cross categorical targets, has the highest correlation (0.71). More critically, the spatial correlation between *Random* and the four conservation scenarios is insignificant. With values ranging from a low of 0.06 to a high of 0.07, statistical insignificance is verified.

Table 4.4 Spearman's Correlation between Scenarios

	Random	*HCC*	*Water*	*Agriculture*	*Ecology*
Random	1				
HCC	0.07	1			
Water	0.06	0.65	1		
Agriculture	0.07	0.60	0.46	1	
Ecology	0.07	0.66	0.71	0.46	1

In addition to the correlation analysis, hypothesis tests, in the form of the Wilcoxon-Mann-Whitney U test, offer a verification tool to ensure that each of the four conservation scenarios are sufficiently distinct from *Random* for each of the four ecosystem service indicators, based on the difference in respective sums (Table 4.5). In eleven of the sixteen tests, the null hypothesis—being that there is no statistical difference between *Random* and conservation scenario (*x*)—could be rejected with over 99% confidence. In an additional two tests, both within *Water Yield* (Table 4.5A), the null hypothesis for both the *Agriculture* and the *Ecology* scenarios could be rejected with over 95% confidence. It is worth noting, however, that the *Ecology* scenario is statistically significant, not by its increase over *Random*, but rather in its decrease below *Random,* in regards to *Water Yield*.

In contrast, the null cannot be rejected in three of the sixteen tests, including both the *HCC* and the *Water* scenario in *Water Yield* (Table 4.5A), and the *Agriculture* scenario within *Carbon Storage* (Table 4.5C). Because of the spatial incongruence found in the correlation analysis, these outcome can be explained by the focus on forest cover and riparian ecosystems in the *HCC* scenario, the intensification of that focus in the *Water* and *Ecology* scenarios, and a decrease in targeted forest cover, and hence carbon storage, within the *Agriculture* scenario.

Table 4.5 Descriptive Statistics and p-values by Ecosystem Service Indicator

	Sum	*Mean*	*Min.*	*Max.*	*p-value*[1]
(A**) Water Yield** (mm^3 of water/year)					
Random	50,822,785	4,599	591	10,590	—
HCC	50,864,169	4,603	691	10,590	0.636
Water	50,585,473	4,578	591	10,590	0.441
Agriculture	51,454,142	4,656	691	10,590	0.03*
Ecology	50,115,106	4,535	591	10,590	0.04*
(B) Soil Conservation (tons of retained soil/year)					
Random	30,929,857	2,799	12	32,408	—
HCC	38,200,665	3,457	15	32,408	< 2.2e-16**
Water	37,591,045	3,402	12	32,408	< 2.2e-16**
Agriculture	31,410,333	2,843	12	32,408	7.011e-6**
Ecology	37,236,677	3,370	26	37,712	< 2.2e-16**
(C) Carbon Storage (tons carbon/year)					
Random	16,474,224	1,491	258	2250	—
HCC	17,804,934	1,611	258	2250	< 2.2e-16**
Water	18,234,465	1,650	218	2250	< 2.2e-16**
Agriculture	16,616,807	1,504	258	2250	0.057
Ecology	18,288,368	1,655	258	2250	< 2.2e-16**

(D) Ecosystem Richness (ecosystem types/planning unit)					
Random	88,753	8.032	1	26	—
HCC	110,993	10.04	1	27	< 2.2e-16**
Water	112,826	10.21	1	27	< 2.2e-16**
Agriculture	101,351	9.172	1	26	< 2.2e-16**
Ecology	113,925	10.31	1	26	< 2.2e-16**

[1] Wilcox-Mann-Whitney non-parametric two-sample test comparing each conservation scenario to the random scenario.
* > 95% Confidence
** > 99% Confidence

Comparing Scenario Results

To be informative within a broader systematic conservation planning process, the analysis results must be easily comparable between alternative options. Here the *Random* scenario provides the baseline from which the conservation scenarios can be compared according to the overlay and statistical analysis results.

Concerning ecosystem service hotspots, the performance of each of the four conservation scenarios represents a significant improvement over *Random* in terms of percent increase (Figure 4.3). For example, in overlaps ≥ 3, *Ecology* (79%), *Water* (76%), and *HCC* (70%) each represent at least a 70% increase over *Random.* Although the *Agriculture* scenario (37% ≥ 3) does not meet the same standard, it nonetheless represents an almost 40% increase over *Random*. Compared solely in terms of the maximum number of overlaps (4), the percent increase over *Random* is even more exaggerated, with *Ecology* (126%) and *Water* (121%) showing the greatest increase, followed by *HCC* (82%) and *Agriculture* (52%).

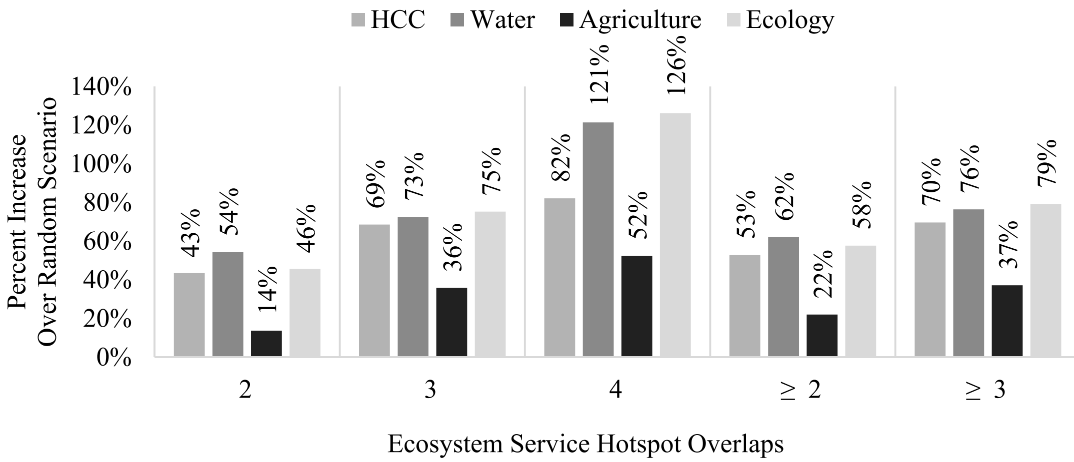

Figure 4.3 Ecosytem Service Hotspot Overlaps by Percent Increase over Random Scenario

In terms of statistical analysis, the conservation scenarios, including *Random*, can be directly compared by the statistical sum of each ecosystem service. Here the sum represents the total amount of each ecosystem service calculated across the 11,050 planning units comprising each scenario. In *Soil Conservation* (Table 4.5B), *Carbon Storage* (Table 4.5C), and *Ecosystem Richness* (Table 4.5D), each of the conservation scenarios have sums greater than *Random*. In all but the *Agriculture* scenario in *Carbon Storage*, these differences are statistically significant. Conversely, in *Water Yield* (Table 4.5A) the sums of *Ecology* and *Water* are less than the sum of *Random*, while *HCC* and *Agriculture* are greater than *Random*, with only *Agriculture* significantly so.

As illustrated in Table 4.5, the leading scenarios, by statistical sum, vary between each ecosystem service. In *Water Yield*, *Agriculture* (51.45 million mm^3) denotes the top service provider, followed by *HCC* (50.86 million mm^3). Whereas *HCC* (38.2 million tons) is the top providing scenario of *Soil Conservation*, followed by *Water* (37.6 million tons). In terms of *Carbon Storage*, *Ecology* (18.29 million tons) is the leading scenario, trailed by *Water* (18.23 million tons). This same pattern holds for *Ecosystem Richness* as well, with *Ecology* (10.31 mean ecosystem types per planning unit) the top scenario, followed again by *Water* (10.21 mean ecosystem types per planning unit).

Each ecosystem service sum noted above represents a different unit of analysis thereby making intra-service comparisons, and specifically visualizations between scenarios, challenging. For comparative purposes, standardized values, or z-scores, of the statistical sums of each ecosystem service are calculated by subtracting the ecosystem service mean of all scenarios from an individual scenario and then dividing the difference by the standard deviation. The resultant z-score is the number of standard deviations a scenario is above (reflected as a positive number) or below (reflected as a negative number) the mean (Figure 4.4).

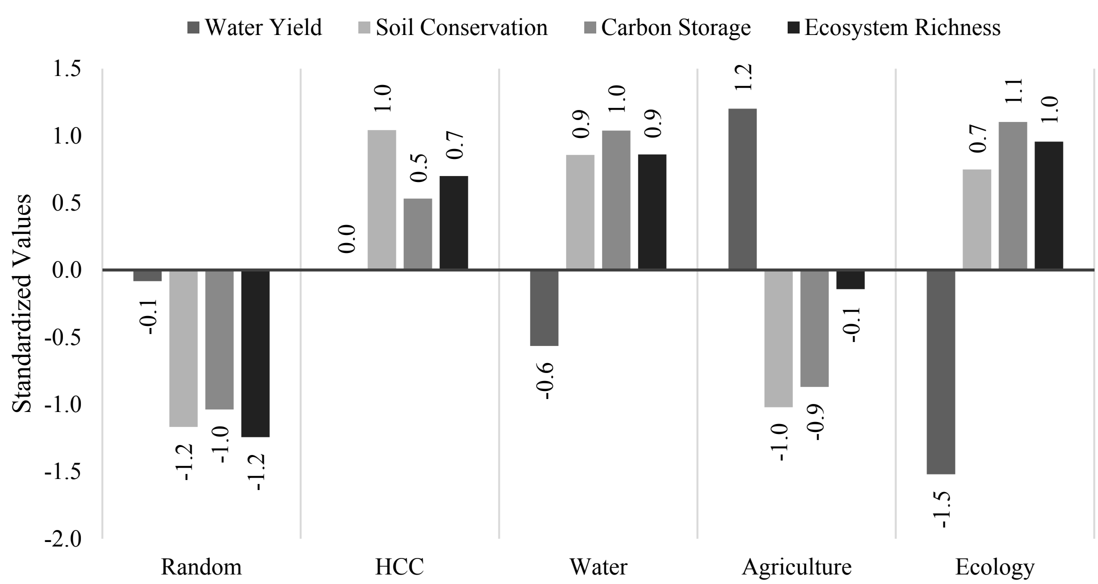

Figure 4.4 Ecosystem Service Standardized Values by Conservation Scenario

Furthermore, a standard overall "score" is obtained by summing the ecosystem service z-scores of each scenario. As seen in Figure 4.5, although *HCC* was the top scenario in only one of the four ecosystem service categories, its overall score of 2.28 makes it the leading scenario followed by *Water* (2.19), *Ecology* (1.29), and *Agriculture* (-0.83). As should be expected, *Random*'s overall score of -3.53 is significantly lower than the scores of all four targeted conservation scenarios.

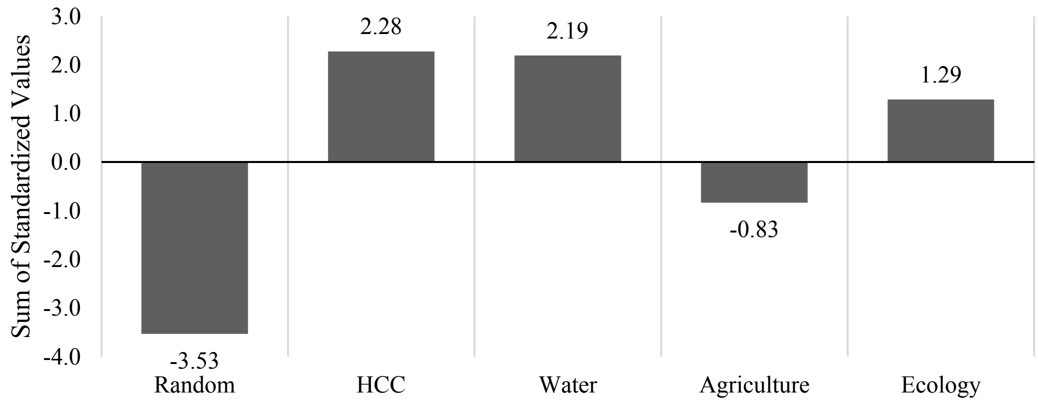

Figure 4.5 Summed Ecosystem Service Standardized Values by Conservation Scenario

DISCUSSION

The results presented in the preceding sub-section contribute to an emerging literature in which multiple ecosystem services are not only quantified at the landscape-scale through ecological production functions, but also incorporated into decision-making contexts. In contrast to previous work, this research is distinguished by its use of ecosystem service indicators to evaluate a range of conservation scenarios. In the following sub-section I discuss the results of this research, and their implications, in terms of the following three categories: 1) *Spatial Congruence*, 2) *Statistical Significance*, and 3) *Conservation Implications and the Decision-Making Process.*

Spatial Congruence

This study offers evidence to the inherent complexity in attempting to quantify spatially explicit ecological phenomena for use in decision-making. Although the four ecosystem service functions, i.e., *Water Yield*, *Soil Conservation*, *Carbon Storage*, and *Ecosystem Richness*, were selected because previous research had indicated a high positive correlation between them (Bai et al. 2011; Chan et al. 2006), this same pattern was not replicated across the Central Texas study area.

Similar to the findings of Izquierdo and Clark (2012), I found negative correlations between *Water Yield* and biodiversity. Yet unlike Izquierdo and Clark (2012), who also noted a negative correlation between "Water" and "Carbon," I found *Water Yield* to be negatively correlated with *Soil Conservation* when measured across the entire site. Adding further complexity, when compared in terms of hotspots, I discovered the negative correlation between *Water Yield* and *Ecosystem Richness* became positive. Additional analysis indicates this shift is partially attributable to the relaxation of the hotspot threshold from 10% to 25%, as a result of the tendency of correlation to increase in conjunction with threshold. For example, assuming the top 10% hotspot, *Water Yield* and *Ecosystem*

Richness have a correlation value of 0.10—with 1.0 being perfect correlation. At 15%, the correlation rises to 0.12. Even with a tightened threshold of 5%, the correlation remains positive with a value of 0.05. This evidence suggests a true, albeit weak, correlational shift from negative to positive for *Water Yield* and *Ecosystem Richness*.

Regardless of whether the correlations are positive or negative, they remain relatively weak in all instances. This is in direct contrast to the work of Bai et al (2011), who found average correlation between ecosystem service values equal to 0.56. Conversely, I found average positive correlation values of 0.25, when measured across the entire study area. When measured in terms of ecosystem service hotspots, the positive correlation became even less pronounced, with an average value of 0.21.

These weak or negative spatial correlations support the general conclusion that ecosystem services, including biodiversity, often co-occur with certain services, but do not co-occur with other services (Naidoo et al. 2008; Turner et al. 2007). This conclusion can be taken a step further to suggest that ecosystem services which tend to co-occur at one location or scale, may not co-occur at another location or scale (Anderson et al. 2009). Further still to suggest that ecosystem services may co-occur in total across a single location and scale, but that the hotspots of those same services may not co-occur. Thus, these findings underscore the inability to broadly generalize ecosystem service relationships, and, therefore, the necessity of conducting place-specific, comprehensive analyses when incorporating ecosystem services into any decision-making context.

However, the ability of each conservation scenario to outperform *Random* in terms of ecosystem service provision supports another general conservation conclusion, despite the lack of spatial congruence. Namely, that the targeting of biophysical attributes for conservation concurrently targets areas supporting multiple ecosystem services (Chan et al. 2011; Chan et al. 2006; Larsen et al. 2011; Onaindia et al. 2013; Thomas et al. 2013).

Statistical Significance

Although the use of a random, or null, model is commonly used to test the significance of hypotheses in landscape ecology (Turner and Gardner 1991), to date it has not been applied in a multi-scenario comparison of ecosystem service provision. In the context of this study, I found the method to be a beneficial verification tool which yielded unexpected, yet decisive results. Specifically, the failure to reject the null hypothesis in two of the four scenarios in *Water Yield* indicate that the conservation scenarios are no better at selecting units of high value for *Water Yield* than a random sampling of planning units.

This result is partially explained by the negative spatial correlation found between *Water Yield* and other services, though a negative correlation should be expected to result in statistically significant sums *below* the random scenario sum. This occurred in only one of the four scenarios (*Ecology*), and only with 95% confidence. Without further analysis—such as a comparison between alternative measures of *Water Yield*, a sensitivity analysis of the variables used by the InVEST model (Sánchez-Canales et al. 2012), or testing the impact of recalibrating the conservation targets—to verify the results, it cannot be said with confidence that the conservation scenarios perform better than random in this instance. Thus, at the least, greater caution must be taken in the incorporation of *Water Yield* into conservation decisions made within the study area. Moreover, to avoid adding increased uncertainty to the process, and potential undesired outcomes, it may be more prudent to simply omit *Water Yield* as a decision consideration.

Conservation Implications and the Decision-Making Process

As with any planning process, the ultimate goal is to use analysis results to inform actual decisions. Subsequently, the results of this integrated research framework suggest a range of conservation implications at both the landscape-specific, local level, and more broadly at the generalized, global level.

The primary driver of this work is the identification of conservation priority areas for a local conservation non-profit, Hill Country Conservancy (HCC), across the six-county Central Texas study area. As such, the research results are intended to bolster the understanding of complex ecological phenomena, specifically as they relate to ecosystem services, in order to more robustly compare conservation alternatives and thereby inform conservation decisions. Each method of analysis, i.e. ecosystem service hotspot overlap, spatial correlation, and statistical significance, convey distinct implications in terms of the study area, and the conservation planning process set within it.

The ecosystem service hotspot analysis provides some of the most salient results of this work, explicitly in terms of landscape sustainability. As shown in Figure 4.1, the spatial distribution of hotspot overlaps is prominently situated within the same geographic area—namely northwestern Travis County, all of Hays County, and southern portions of both Blanco and Burnet counties—projected to undergo the greatest land-use changes in the coming decades (EPA 2009), largely as a result of increased urbanization and development (refer to Figure 2.6). The implication being that the portion of the landscape most critical to the regional provision of ecosystem services, and therefore landscape sustainability, is also the most vulnerable to transformation.

With this understanding, the systematic conservation planning process can be adapted in one of two ways. First, conservation targets can be revised to select either at-risk areas or biophysical conservation elements correlated with ecosystem service hotspots.

Alternatively, or in conjunction, cost values can be calculated for each planning unit, with the planning units in the threatened area considered most likely to undergo land-use change issued the lowest cost values. Incorporating a cost surface in this manner would increase the likelihood that the most vulnerable planning units would be selected by the Marxan minimum set algorithm.

The lack of spatial congruence between ecosystem services, as identified in the spatial correlation analysis, also has conservation implications. Namely, the negative correlation between *Water Yield* and other ecosystem services compounds the likelihood of significant tradeoffs occurring between services. More importantly, the lack of strong spatial congruence across all ecosystem services increases the difficulty in maximizing a suite of ecosystem services within a single conservation scenario. Because this study utilizes ecosystem services as indicators rather than as targets, this does not pose a structural problem and the effects of spatial incongruence can still be mitigated in a number of ways.

First, discreet ecosystem services can be isolated and conservation targets adapted to maximize that specific service. For example, if viable incentives for carbon storage and sequestration—such as a carbon market or a cap-and-trade policy—are implemented as a response to climate change, conservation targets can be adapted to maximize the provision of carbon storage across the landscape.

In contrast, additional metrics of ecosystem services can be calculated, and subsequently "bundled" according to their spatial correlations in a similar method to that proposed by Bai (2011). Conservation targets, and ultimately alternative scenarios, can then be developed to either emphasize a specific bundle of services, or to minimize tradeoffs between groups—both of which are comparable using the overall summed ecosystem service "score" developed in this work (Figure 4.5 represents a comparison of the minimization of tradeoffs by conservation scenario; Figure 4.6 the scenario best able to provide a spatially correlated bundle of services).

Finally, the statistical analysis and random control can be used to identify any ecosystem service function which does not provide statistically significant results. Consequently, those services which are not reliable can be omitted from decision consideration. This action can significantly alter the rank order of the conservation scenarios, thereby affecting side-by-side comparison, and ultimately, conservation decisions.

For example, by excluding *Water Yield* from the overall summed ecosystem service "score," the negative correlation between *Water Yield* and the other services is ameliorated. Whereas with *Water Yield* included, *HCC* (2.28) was the leading scenario, followed by *Water* (2.19), and *Ecology* (1.29); excluding *Water Yield* results in *Ecology* (2.81) leading, followed by *Water* (2.76), and *HCC* (2.28), as illustrated in Figure 4.6. This new order now mirrors the results of the hotspot overlay analysis.

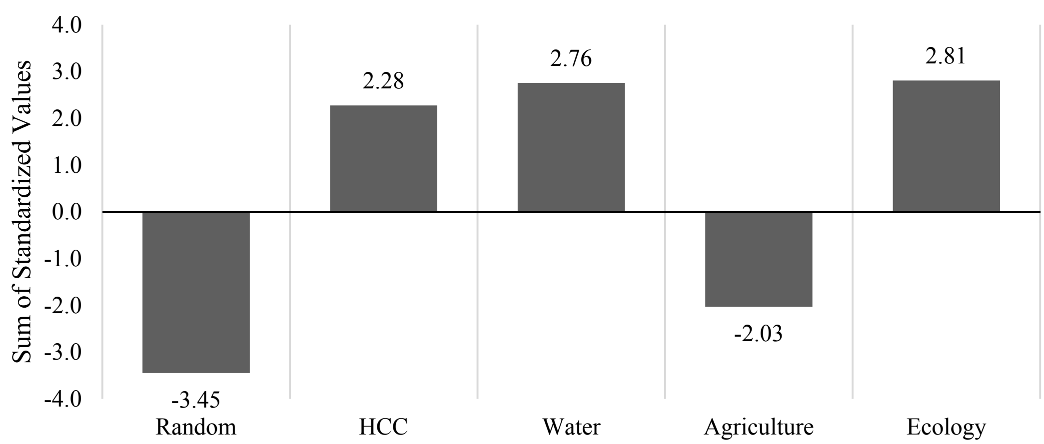

Figure 4.6 Summed Ecosystem Service Standardized Values (excluding *Water Yield*) by Conservation Scenario

Thus, if recommendations are ultimately required, they can be given based on which solution is most relevant to the problem being asked. For instance, if *Water Yield* is omitted, it is possible to confidently recommend the *Ecology* and *Water* scenarios as optimum solutions in terms of both their inclusion of ecosystem service hotspots, and their bundled spatially correlated ecosystem service summed “scores.” Conversely, based on the summed ecosystem service “score”—tabulated using all four ecosystem service indicators—*HCC* represents the scenario best able to successfully balance a range of service trade-offs. However, because the *Water Yield* results were not statistically significant, any recommendation of the *HCC* scenario would also include a cautionary caveat. Regardless, this framework is flexible and adaptable enough to strategically assist conservation decisions on the basis of the provision of ecosystem services.

CHAPTER 5:
AREAS OF FURTHER RESEARCH AND CONCLUSIONS

In the preceding chapters, I reviewed this work's theoretical foundations (Chapter 1), analyzed the six-county Central Texas study area (Chapter 2), described the multiple methods of analysis (Chapter 3), and presented and discussed the analysis results, along with the conservation implications of those results (Chapter 4). The subsequent product is a research framework that offers a novel and robust approach for integrating ecosystem services into conservation planning methods by utilizing them as indicators to compare the performance of conservation scenarios. In this, the final chapter, I explore the areas of further research uncovered through the process, as well as offer my final conclusions, in two respective subsections: 1) *Areas of Further Research*, and 2) *Conclusions*.

AREAS OF FURTHER RESEARCH

As with most scientific research, the search for one answer often leads to myriad additional questions. In the following sub-section I discuss the areas of further research identified during the development of this work. I have classified these areas of additional inquiry into three distinct categories to be discussed in turn: 1) *Ecosystem Services*, 2) *Conservation Planning*, and 3) *Conservation and Sustainability*.

Ecosystem Services

Through the course of this work, several broad categories of research opportunities were identified in relation to the concept of ecosystem services. These opportunities include additional research specific to the work presented here, and more general studies concerning the quantification and mapping of services, as well as those related to their economic valuation.

Several research areas were identified as possible extensions to the framework developed in this study. Foremost, the framework is intended to be highly adaptive, therefore a greater quantity of ecosystem services can be incorporated, and their effect on outcomes and potential tradeoffs compared. For example, additional ecosystem service metrics such as *pollination*, *recreation*, *crop production*, or *forage production* can be measured, mapped, analyzed, and bundled according to patterns of spatial correlation. Subsequently, alternative scenarios intended to accentuate specific *bundles* of services can also be developed and compared. Furthermore, the use of indictors, as developed in this project, could be compared to alternative methods of ecosystem service integration such as targets (Chan et al. 2006; Izquierdo and Clark 2012), or as costs (Chan et al. 2011).

Research opportunities concerning the quantification and mapping of ecosystem services were also discovered. Although the InVEST models used in this study are well-recognized tools for spatially explicit ecosystem service quantification, they are not without limitations. Further sensitivity analysis of all InVEST terrestrial models across a broad range of environments and land-uses—similar to the *water yield* sensitivity analysis conducted by Sanchez-Canales et al. (2012)—is necessary to better gauge the relative importance of individual biophysical parameters and their effect on final outputs. More specific to this work, the InVEST *water yield* model needs to be tested for accuracy and precision when applied to unique geologic substructures, such as the highly porous karst limestone dominate throughout Central Texas. Moreover, the InVEST models are only one method, among many, attempting to quantify and map ecosystem services. A study directly comparing multiple quantification methods in terms of overall accuracy in output values, and precision in providing them at the place-specific, landscape level, is also needed.

Perhaps most importantly, more research is needed regarding the economic valuation of ecosystem services. The promise of the ecosystem service concept is the potential to value ecological phenomena in standard economic terms such as the US dollar. Presently, most ecosystem services do not have explicit market value, and thus comparison remains difficult. For this reason, economic valuation was omitted from this work, despite being an optional component of the InVEST models. As markets become more universal and standardized, as demonstrated by the continued rise in transaction volumes in the global carbon market (Kossoy and Guigon 2012), and valuation models become more proficient at appraising externalities, the economic valuation of ecosystem services can be more easily incorporated into multiple research contexts. Until that time, more place-specific economic analyses identifying the net social value—defined as the aggregate benefit to society, not simply its market value, of a particular service (Tol 2011)—of multiple ecosystem services are needed. Furthermore, potential options for payments for ecosystem services, such as tax-based incentives, subsidies, or direct payments to communities, institutions, or individuals, must continue to be developed (Jack et al. 2008).

Conservation Planning

Further research opportunities, relating more specifically to the field of conservation planning, are also identified in this work. The incorporation of dynamic processes, conservation targets, and integrative methods of output are all areas of potential inquiry.

One of the main limitations of systematic conservation planning methods is the inability to fully account for dynamic processes, such as urban growth and development or the impacts of climate change on biotic communities, within a selection algorithm. More research is needed to develop and compare methods which incorporate these elements in some capacity—whether through existing methods such as using a Marxan "cost" layer, or through novel integrative methods using future land-use scenarios or ecological modeling techniques to represent dynamic processes.

Conservation targets also present a range of potential research opportunities. Overall, to mitigate subjectivity in the selection process, more research is needed to develop both generalized and place-specific conservation target protocols derived from empirical evidence. More specifically, an additional opportunity includes a direct comparison between multiple target sets based on distinct ecological functions or structures. For example, targets based on maintaining ecological functionality, such as in this work, can be compared to alternative sets based on ecosystem service hotspots, maintaining geologic diversity, referred to as "geodiversity" (Gray 2013), or based on the concept of "land facets"—recurring landscape units with uniform topographic and soil attributes (Beier and Brost 2010). In turn, they can be evaluated according to their provision of ecosystem services using the framework developed in this work.

Furthermore, there remains the prospect of integrating systematic conservation planning results with other geographic information system (GIS) based scenario planning tools. For instance, Envision Tomorrow is an open-source GIS based suite of urban and regional planning software tools and indicators developed by Fregonese Associates (Fregonese 2014). Using the Envision platform, Marxan selection outputs could be automatically linked to indicators of ecosystem services. As a result, the impacts of scenarios of urban growth—or increased conservation measures—on the provision of ecosystem services could be ascertained in real-time, thereby increasing the potential engagement of private clients or the public and greatly aiding informed decision-making capabilities.

Conservation and Sustainability

Finally, this research also highlighted a range of research potentials relating more broadly to the concepts of conservation itself, as well as to sustainability.

Increasingly, the impacts of urban development and, thus, the need for conservation action, are well understood. However, the dynamics of land-use change are inherently complex. More research is needed to understand the drivers of large-scale, land-cover change, particularly as they relate to land-use preference and landowner motivations. More specifically, research addressing the effectiveness of current incentives for conservation, the feasibility and continued development of potential incentives for conservation such as payments for ecosystem services, individual landowner motivations toward conservation, and governmental motivations toward conservation—at the national, state, county, and municipal levels—are all essential.

Moreover, as I have argued previously, the ultimate purpose of conservation is to not only protect threatened species and ecosystems, but also to sustain human well-being. A number of research opportunities relating to landscape sustainability were uncovered in this work. Namely, more study is needed to understand the dynamics of localized natural disturbances such as floods, drought, and wildfire, localized human disturbances resulting from population growth and urbanization, and global disturbances related to climate change, on the provision of ecosystem services. Subsequently, more effort is needed to understand the attributes that contribute to resiliency, as well as how best to enhance those attributes through planning and design. Place specific research comparing ecosystem service outputs to proxies of human demand is also needed as it would help develop an understanding of whether the ratio between landscape level services and social demand are sustainable. Lastly, because equity is intrinsic to sustainability, more research is needed to evaluate the social equity of conservation, specifically in regards to the conservation of private lands and the resultant impact on local economies in terms of employment, food cost, and property values.

CONCLUSIONS

Within the field of conservation planning, the integration of ecosystem services is an emergent area of research with the potential to significantly shift the foundation of conservation from the separation of wild nature and society to that of "conservation for the people" (Kareiva and Marvier 2007). Inherent in this shift lies the promise of a sustainable landscape shared by both.

With this work, I offer a novel contribution to this growing body of literature. Although there are recognized limitations in both the materials and methods, the integration of spatially explicit indicators of ecosystem services into a systematic conservation planning process results in a flexible, adaptable framework well suited to supporting the iterative, data-driven nature of decision-making processes.

As the results suggest, this framework is highly place-specific. Yet, in its use of readily available data, it is easily replicated across a wide range of project scales and objectives. This work reinforces the finding that there is often a substantial amount of variability in the spatial congruence of multiple ecosystem services and their provision across a landscape (Cimon-Morin et al. 2013; Izquierdo and Clark 2012), while also supporting the conclusion that the targeting of ecological phenomena for conservation concurrently targets areas supporting multiple ecosystem services (Bai et al. 2011; Chan et al. 2006).

Ultimately, conservation planning is about fostering decisions which lead to informed action. As such, the most important contributions of this work are at the local, landscape-scale. In the application of this framework to the six-county Central Texas study area, it is possible to not only determine where HCC should allocate their resources based on a range of organizationally defined conservation scenarios, but also to verify which scenarios best provide a range of ecosystem services—whether that is the scenario best able to optimize the tradeoffs between spatially incongruent services or to maximize an amount of bundled services. Having this knowledge allows HCC to make, and iteratively adapt, conservation decisions as well as lends transparent, scientific defense to those choices.

As we progress further into the twenty-first century, it is likely that our current environmental challenges and land-use conflicts will continue to increase along with human population and global temperature. If landscape sustainability is to be achieved, in Central Texas and beyond, research and conservation action will have to keep pace. The integration of ecosystem services into conservation planning methods has tremendous potential for introducing and developing new tools for researchers, planners, practitioners, and decision-makers. Yet that integration is just beginning, with many questions left unanswered. This research provides one solution for integration in which indicators of ecosystem services contribute to strategic conservation decisions, thus taking into account the ecological phenomena necessary for human well-being and sustainable landscapes.

APPENDICES

Appendix A

Systematic Conservation Planning: Conservation Elements and Targets

Conservation Elements			*Conservation Targets*			
Conservation Element	**Category**	**Source**	**General[1]**	**Water[2]**	**Agriculture[3]**	**Ecology[4]**
Edwards Aquifer Recharge Zone	Water	USGS NHD	0.4	0.6	0.4	0.4
Trinity Aquifer Recharge Zone	Water	USGS NHD	0.15	0.225	0.15	0.15
Marble Falls Aquifer Recharge Zone	Water	USGS NHD	0.1	0.15	0.1	0.1
Major Waterway 1200ft buffer	Water	USGS NHD	0.3	0.45	0.3	0.3
Significant Stream 1200ft buffer	Water	TPWD	0.5	0.75	0.5	0.5
Springs 350ft buffer	Water	TPWD	0.4	0.6	0.4	0.4
Wetlands 150ft buffer	Water	TFS	0.3	0.45	0.3	0.3
Public Water Supply Well 1200ft buffer	Water	TWDB	0.3	0.45	0.3	0.3
Public Water Supply Surface Intake HUC 12 Watersheds	Water	TWDB	0.3	0.45	0.3	0.3
Colorado River-Lake Buchanan HUC 10 Watershed	Water	USGS NHD	0.35	0.35	0.35	0.35
Sandy Creek HUC 10 Watershed	Water	USGS NHD	0.35	0.35	0.35	0.35
Inks Lake-Lake Lyndon B Johnson HUC 10 Watershed	Water	USGS NHD	0.375	0.375	0.375	0.375
Hickory Creek-Llano River HUC 10 Watershed	Water	USGS NHD	0.375	0.375	0.375	0.375
San Fernando Creek-Llano River HUC 10 Watershed	Water	USGS NHD	0.325	0.325	0.325	0.325
Little Llano River-Llano River HUC 10 Watershed	Water	USGS NHD	0.35	0.35	0.35	0.35
Lake Marble Falls-Lake Travis HUC 10 Watershed	Water	USGS NHD	0.35	0.35	0.35	0.35
Cow Creek-Lake Travis HUC 10 Watershed	Water	USGS NHD	0.325	0.325	0.325	0.325
City of Austin-Colorado River HUC 10 Watershed	Water	USGS NHD	0.325	0.325	0.325	0.325

Conservation Elements			*Conservation Targets*			
Conservation Element	**Category**	**Source**	**General[1]**	**Water[2]**	**Agriculture[3]**	**Ecology[4]**
Onion Creek-Colorado River HUC 10 Watershed	Water	USGS NHD	0.3	0.3	0.3	0.3
North Grape Creek-Pedernales River HUC 10 Watershed	Water	USGS NHD	0.4	0.4	0.4	0.4
Pedernales River-Lake Travis HUC 10 Watershed	Water	USGS NHD	0.3	0.3	0.3	0.3
Edwards Plateau: Ashe Juniper Motte and Woodland	Ecology	TPWD ESD	0	0	0	0.025
Edwards Plateau: Live Oak Motte and Woodland	Ecology	TPWD ESD	0	0	0	0.05
Edwards Plateau: Deciduous Oak / Evergreen Motte and Woodland	Ecology	TPWD ESD	0	0	0	0.075
Edwards Plateau: Oak / Hardwood Motte and Woodland	Ecology	TPWD ESD	0	0	0	0.075
Edwards Plateau: Post Oak Motte and Woodland	Ecology	TPWD ESD	0.075	0.075	0.075	0.1125
Edwards Plateau: Savanna Grassland	Ecology	TPWD ESD	0.05	0.05	0.075	0.075
Edwards Plateau: Ashe Juniper Slope Forest	Ecology	TPWD ESD	0.05	0.075	0.05	0.075
Edwards Plateau: Live Oak Slope Forest	Ecology	TPWD ESD	0.8	0.8	0.8	1.2
Edwards Plateau: Oak / Ashe Juniper Slope Forest	Ecology	TPWD ESD	0.1	0.15	0.1	0.15
Edwards Plateau: Oak / Hardwood Slope Forest	Ecology	TPWD ESD	0.2	0.3	0.2	0.3
Llano Uplift: Live Oak Woodland	Ecology	TPWD ESD	0.15	0.15	0.15	0.225
Llano Uplift: Post Oak Woodland	Ecology	TPWD ESD	0.05	0.05	0.05	0.075
Llano Uplift: Grassland	Ecology	TPWD ESD	0.05	0.05	0.075	0.075
Edwards Plateau: Floodplain Ashe Juniper Forest	Water/ Ecology	TPWD ESD	0.4	0.6	0.4	0.6
Edwards Plateau: Floodplain Live Oak Forest	Water/ Ecology	TPWD ESD	0.5	0.75	0.5	0.75
Edwards Plateau: Floodplain Hardwood / Ashe Juniper Forest	Water/ Ecology	TPWD ESD	0.5	0.75	0.5	0.75

Conservation Elements			*Conservation Targets*			
Conservation Element	**Category**	**Source**	**General[1]**	**Water[2]**	**Agriculture[3]**	**Ecology[4]**
Edwards Plateau: Floodplain Hardwood Forest	Water/ Ecology	TPWD ESD	0.25	0.375	0.25	0.375
Edwards Plateau: Floodplain Ashe Juniper Shrubland	Water/ Ecology	TPWD ESD	0.25	0.375	0.25	0.375
Edwards Plateau: Floodplain Deciduous Shrubland	Water/ Ecology	TPWD ESD	0.4	0.6	0.4	0.6
Edwards Plateau: Floodplain Herbaceous Vegetation	Water/ Ecology	TPWD ESD	0.5	0.75	0.5	0.75
Edwards Plateau: Riparian Ashe Juniper Forest	Water/ Ecology	TPWD ESD	0.25	0.375	0.25	0.375
Edwards Plateau: Riparian Live Oak Forest	Water/ Ecology	TPWD ESD	0.5	0.75	0.5	0.75
Edwards Plateau: Riparian Hardwood / Ashe Juniper Forest	Water/ Ecology	TPWD ESD	0.5	0.75	0.5	0.75
Edwards Plateau: Riparian Hardwood Forest	Water/ Ecology	TPWD ESD	0.4	0.6	0.4	0.6
Edwards Plateau: Riparian Ashe Juniper Shrubland	Water/ Ecology	TPWD ESD	0.2	0.3	0.2	0.3
Edwards Plateau: Riparian Deciduous Shrubland	Water/ Ecology	TPWD ESD	0.4	0.6	0.4	0.6
Edwards Plateau: Riparian Herbaceous Vegetation	Water/ Ecology	TPWD ESD	0.35	0.525	0.35	0.525
Edwards Plateau: Ashe Juniper / Live Oak Shrubland	Ecology	TPWD ESD	0.05	0.05	0.05	0.075
Edwards Plateau: Shin Oak Shrubland	Ecology	TPWD ESD	0.075	0.075	0.075	0.1125
Edwards Plateau: Ashe Juniper / Live Oak Slope Shrubland	Ecology	TPWD ESD	0.05	0.075	0.05	0.075
Edwards Plateau: Shin Oak Slope Shrubland	Ecology	TPWD ESD	0.1	0.15	0.1	0.15
Edwards Plateau: Wooded Cliff/Bluff	Ecology	TPWD ESD	0.4	0.6	0.4	0.6
Significant Slope 15 to 60%	Ecology	USGS NED	0.15	0.225	0.15	0.225
Significant Slope over 60%	Ecology	USGS NED	0.4	0.6	0.4	0.6

Conservation Elements			*Conservation Targets*			
Conservation Element	**Category**	**Source**	**General[1]**	**Water[2]**	**Agriculture[3]**	**Ecology[4]**
Golden-cheeked Warbler Moderate Quality Habitat	Ecology	Loomis	0.1	0.1	0.1	0.15
Golden-cheeked Warbler High Quality Habitat	Ecology	Loomis	0.3	0.3	0.3	0.45
Endangered Species Critical Habitat	Ecology	US FWS	0.5	0.5	0.5	0.75
Prime Farmland Soil	Cultural	NRCS SSURSGO	0.3	0.3	0.45	0.3
Scenic Views	Cultural	Texas State	0.25	0.25	0.25	0.25
Available Water Content	Cultural	NRCS SSURSGO	0.05	0.05	0.075	0.05
Soil Depth	Cultural	NRCS SSURSGO	0.05	0.05	0.075	0.05

[1] General Scenario targets developed in consultation with Hill Country Conservancy
[2] Water Scenario increased selected water related conservation element targets (dark grey cells) by a factor of 50%
[3] Agriculture Scenario increases selected agriculture related conservation element targets (dark grey cells) by a factor of 50%
[4] Ecology Scenario increases selected ecology related conservation element targets (dark grey cells) by a factor of 50%

Appendix B

InVEST Data and Sources

Model	*Source*	*Type*
Carbon Storage		
LULC	TPWD: Texas Ecological Systems Database: Phase 1 (TPWD 2010) http://www.tpwd.state.tx.us/gis/gallery/	Raster ~10m
Future LULC	EPA ICLUS model for year 2030 (EPA 2009) http://cfpub.epa.gov/ncea/global/recordisplay.cfm?deid=205305	Raster ~1km
Carbon Pools	Intergovernmental Panel on Climate Change (IPCC) 2006 methodology (IPCC 2006) http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html	values
Carbon Value: Social Cost of Carbon	The economic effects of climate change (Tol 2009)	values
Water Provision		
DEM	USGS NED DEM http://ned.usgs.gov/index.asp	Raster ~30m
Soil Depth	NRCS STATSGO2; NRCS Soil Data Viewer http://soils.usda.gov/survey/geography/ssurgo/description.html; http://soils.usda.gov/sdv/	Vector
Precipitation	Prism Climate Data http://www.wcc.nrcs.usda.gov/climate/prism.html	Raster ~1km
Plant Available Water Content (AWC)	NRCS STATSGO2; NRCS Soil Data Viewer http://soils.usda.gov/survey/geography/ssurgo/description.html; http://soils.usda.gov/sdv/	Vector
Average Annual Potential Evapotranspiration (PET)	MODIS 16 Global Evapotranspiration Project http://www.ntsg.umt.edu/project/mod16	Raster ~1km
LULC	TPWD: Texas Ecological Systems Database: Phase 1 (TPWD 2010) http://www.tpwd.state.tx.us/gis/gallery/	Raster ~10m
Watersheds	USGS, National Hydrography Dataset (NHD): HUC 8 http://nhd.usgs.gov/index.html; http://nhd.usgs.gov/wbd_data_citation.html accessed via TNRIS: www.tnris.org	Vector
Subwatersheds	USGS, National Hydrography Dataset (NHD): HUC 12 http://nhd.usgs.gov/index.html; http://nhd.usgs.gov/wbd_data_citation.html accessed via TNRIS: www.tnris.org	Vector
Stream Flow Lines	USGS, National Hydrography Dataset (NHD): flowlines http://nhd.usgs.gov/index.html; http://nhd.usgs.gov/wbd_data_citation.html accessed via TNRIS: www.tnris.org	Vector
Biophysical Table	Adapted from the INVEST 2.5.5 example dataset http://www.naturalcapitalproject.org/ On mf laptop > C:\InVEST_2_5_5_x64\Base_Data\Freshwater\biophysical table	values

Model	*Source*	*Type*
Water Purification Threshold (Nutrient Loading)	Modeling Phosphorus loading and lake response under uncertainty: A manual and compilation of export coefficients (Reckhow et al. 1980)	values
Soil Conservation		
DEM	USGS NED dem http://ned.usgs.gov/index.asp	Raster ~30m
Rainfall erosivity index (R)	USDA Predicting Soil Erosion by Water (USDA 1997) ESRI shapefile created from hardcopy map; converted to raster	Raster ~30mS
Soil Erodibility (K)	NRCS STATSGO2; NRCS Soil Data Viewer http://soils.usda.gov/survey/geography/ssurgo/description.html; http://soils.usda.gov/sdv/	Vector
LULC	TPWD: Texas Ecological Systems Database: Phase 1 (TPWD 2010) http://www.tpwd.state.tx.us/gis/gallery/	Raster ~10m
Watersheds	USGS, National Hydrography Dataset (NHD): HUC 12 http://nhd.usgs.gov/index.html; http://nhd.usgs.gov/wbd_data_citation.html accessed via TNRIS: www.tnris.org	Vector
Subwatersheds	USGS, National Hydrography Dataset (NHD): HUC 12 http://nhd.usgs.gov/index.html; http://nhd.usgs.gov/wbd_data_citation.html accessed via TNRIS: www.tnris.org	Vector
Biophysical Table	P and C coefficients: (USDA 1997) Sediment retention value: Adapted from the INVEST 2.5.5 example dataset, http://www.naturalcapitalproject.org/	values
Sediment Threshold Table	NA; values of (1) were given for each variable	

REFERENCES

Abbott, Patrick L. 1975. On the Hydrology of the Edwards Limestone, South-Central Texas. *Journal of Hydrology* 24 (3):251-269.

Adams, Vanessa M., Robert L. Pressey, and Robin Naidoo. 2010. Opportunity Costs: Who Really Pays for Conservation? *Biological Conservation* 143 (2):439-448.

Adams, William M., Ros Aveling, Dan Brockington, Barney Dickson, Jo Elliott, Jon Hutton, Dilys Roe, Bhaskar Vira, and William Wolmer. 2004. Biodiversity Conservation and the Eradication of Poverty. *Science* 306 (5699):1146-1149.

Alison, Robert M. 1981. The Earliest Traces of a Conservation Conscience. *Natural History* 90 (5):72-77.

Anderson, Barbara J., Paul R. Armsworth, Felix Eigenbrod, Chris D. Thomas, Simon Gillings, Andreas Heinemeyer, David B. Roy, and Kevin J. Gaston. 2009. Spatial Covariance between Biodiversity and other Ecosystem Service Priorities. *Journal of Applied Ecology* 46 (4):888-896.

Andreen, William L. 2003. Water Quality Today-Has the Clean Water Act Been a Success? *Alabama Law Review* 55 (3):537.

Ardron, Jeff A., Hugh P. Possingham, and Carissa J. Klein. 2010. Marxan Good Practices Handbook, Version 2. Pacific Marine Analysis and Research Association, Victoria, BC, Canada. 165 pages.

Arendt, Randall. 1994. *Rural by Design*. Chicago: American Planning Association.

Bai, Yang, Changwei Zhuang, Zhiyun Ouyang, Hua Zheng, and Bo Jiang. 2011. Spatial Characteristics between Biodiversity and Ecosystem Services in a Human-Dominated Watershed. *Ecological Complexity* 8 (2):177-183.

Ball, Ian R., Hugh P. Possingham, and M. Watts. 2009. Marxan and Relatives: Software for Spatial Conservation Prioritisation. In *Spatial Conservation Prioritisation: Quantitative Methods and Computational Tools*, edited by A. Moilanen, K. A. Wilson and H. P. Possingham. Oxford: Oxford University Press.

Balmford, Andrew, and William Bond. 2005. Trends in the State of Nature and Their Implications for Human Well-Being. *Ecology Letters* 8 (11):1218-1234.

Balmford, Andrew, Aaron Bruner, Philip Cooper, Robert Costanza, Stephen Farber, Rhys E. Green, Martin Jenkins, Paul Jefferiss, Valma Jessamy, and Joah Madden. 2002. Economic Reasons for Conserving Wild Nature. *Science* 297 (5583):950-953.

Balmford, Andrew, Brendan Fisher, Rhys E. Green, Robin Naidoo, Bernardo Strassburg, R. Kerry Turner, and Ana S.L. Rodrigues. 2011. Bringing Ecosystem Services into the Real World: An Operational Framework for Assessing the Economic Consequences of Losing Wild Nature. *Environmental and Resource Economics* 48 (2):161-175.

Barral, María Paula, and Maceira Néstor Oscar. 2012. Land-Use Planning Based on Ecosystem Service Assessment: A Case Study in the Southeast Pampas of Argentina. *Agriculture, Ecosystems & Environment* 154:34-43.

Becker, C. G., R. D. Loyola, C. F. B. Haddad, and K. R. Zamudio. 2010. Integrating Species Life-History Traits and Patterns of Deforestation in Amphibian Conservation Planning. *Diversity and Distributions* 16 (1):10-19.

Beier, Paul, and Brian Brost. 2010. Use of Land Facets to Plan for Climate Change: Conserving the Arenas, Not the Actors. *Conservation Biology* 24 (3):701-710.

Benayas, José M. Rey, Adrian C. Newton, Anita Diaz, and James M. Bullock. 2009. Enhancement of Biodiversity and Ecosystem Services by Ecological Restoration: A Meta-Analysis. *Science* 325 (5944):1121-1124.

Bennett, A.F., J.Q. Radford, and A. Haslem. 2006. Properties of Land Mosaics: Implications for Nature Conservation in Agricultural Environments. *Biological Conservation* 133 (2):250-264.

Berkes, F. 1996. Social Systems, Ecological Systems and Property Rights. In *Rights to Nature. Ecological, Economic, Cultural, and Political Principles of Institutions for the Environment*, edited by S. S. Hanna, C. Folke and K. G. Maler. Washington D.C.: Island Press.

Berkes, F., J. Colding, and C. Folke. 2003. *Navigating Social-Ecological Systems: Building Resilience for Complexity and Change*. Cambridge: Cambridge University Press.

Bowles, David E., and Thomas L. Arsuffi. 1993. Karst Aquatic Ecosystems of the Edwards Plateau Region of Central Texas, USA: A Consideration of Their Importance, Threats to Their Existence, and Efforts for Their Conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems* 3 (4):317-329.

Boyd, James, and Spencer Banzhaf. 2007. What are Ecosystem Services? The Need for Standardized Environmental Accounting Units. *Ecological Economics* 63 (2–3):616-626.

Burkhard, Benjamin, Irene Petrosillo, and Robert Costanza. 2010. Ecosystem Services– Bridging Ecology, Economy and Social Sciences. *Ecological Complexity* 7 (3):257-259.

Capra, F. 2002. *The Hidden Connections: A Science of Sustainability*. New York: Anchor Books.

Carpenter, Stephen R., and Carl Folke. 2006. Ecology for Transformation. *Trends in Ecology & Evolution* 21 (6):309-315.

Carpenter, Stephen R., Harold A. Mooney, John Agard, Doris Capistrano, Ruth S. DeFries, Sandra Díaz, Thomas Dietz, Anantha K. Duraiappah, Alfred Oteng-Yeboah, Henrique Miguel Pereira, Charles Perrings, Walter V. Reid, José Sarukhan, Robert J. Scholes, and Anne Whyte. 2009. Science for Managing Ecosystem Services: Beyond the Millennium Ecosystem Assessment. *Proceedings of the National Academy of Sciences* 106 (5):1305-1312.

Carson, Rachel. 1962. *Silent Spring*. Boston: Houghton Mifflin.

Carson, Richard T., and Robert Cameron Mitchell. 1993. The Value of Clean Water: The Public's Willingness to Pay for Boatable, Fishable, and Swimmable Quality Water. *Water Resources Research* 29 (7):2445-2454.

Carwardine, J., W.A. Rochester, K.S. Richardson, Kristen J. Williams, R.L. Pressey, and H.P. Possingham. 2007. Conservation Planning with Irreplaceability: Does the Method Matter? *Biodiversity and Conservation* 16 (1):245-258.

Chan, Kai M.A., Lara Hoshizaki, and Brian Klinkenberg. 2011. Ecosystem Services in Conservation Planning: Targeted Benefits vs. Co-Benefits or Costs? *PloS ONE* 6 (9):e24378.

Chan, Kai M.A., M. Rebecca Shaw, David R. Cameron, Emma C. Underwood, and Gretchen C. Daily. 2006. Conservation Planning for Ecosystem Services. *PLoS Biology* 4 (11):e379.

Chapin, F.S., C. Folke, and G.P. Kofinas. 2009. A Framework for Understanding Change. In *Principles of Ecosystem Stewardship: Resilience-Based Natural Resource Management in a Changing World*, edited by F. S. Chapin, G. P. Kofinas and C. Folke. Cambridge: Cambridge University Press.

Chapin, F.S., G.P. Kofinas, C. Folke, and M.C. Chapin. 2009. *Principles of Ecosystem Stewardship: Resilience-Based Natural Resource Management in a Changing World*. Cambridge: Cambridge University Press.

Church, Richard L., David M. Stoms, and Frank W. Davis. 1996. Reserve Selection as a Maximal Covering Location Problem. *Biological Conservation* 76 (2):105-112.

Cimon-Morin, Jérôme, Marcel Darveau, and Monique Poulin. 2013. Fostering Synergies between Ecosystem Services and Biodiversity in Conservation Planning: A Review. *Biological Conservation* 166:144-154.

Cleaveland, Malcolm K. 2006. Extended Chronology of Drought in the San Antonio Area. In *Report to the Guadalupe-Blanco River Authority*.

Costanza, Robert. 2008. Ecosystem Services: Multiple Classification Systems are Needed. *Biological Conservation* 141 (2):350-352.

Costanza, Robert, Ralph d'Arge, Rudolf De Groot, Stephen Farber, Monica Grasso, Bruce Hannon, Karin Limburg, Shahid Naeem, Robert V. O'Neill, Jose Paruelo, Robert G. Raskin, Paul Sutton, and Marjan van den Belt. 1997. The Value of the World's Ecosystem Services and Natural Capital. *Nature* 387:253-260.

Costanza, Robert, Brendan Fisher, Kenneth Mulder, Shuang Liu, and Treg Christopher. 2007. Biodiversity and Ecosystem Services: A Multi-Scale Empirical Study of the Relationship between Species Richness and Net Primary Production. *Ecological Economics* 61 (2–3):478-491.

Cronon, William. 1990. Modes of Prophecy and Production: Placing Nature in History. *The Journal of American History* 76 (4):1122-1131.

Cronon, William. 1996. The Trouble with Wilderness: Or, Getting Back to the Wrong Nature. *Environmental History* 1 (1):7-28.

Culver, David C., and Boris Sket. 2000. Hotspots of Subterranean Biodiversity in Caves and Wells. *Journal of Cave and Karst Studies* 62 (1):11-17.

Cumming, Graeme S., Per Olsson, F. S. Chapin, and C. S. Holling. 2013. Resilience, Experimentation, and Scale Mismatches in Social-Ecological Landscapes. *Landscape Ecology* 28 (6):1139-1150.

Daily, Gretchen. 1997. *Nature's Services: Societal Dependence on Natural Ecosystems*. Washington, D.C.: Island Press.

Daly, Herman E. 1995. On Wilfred Beckerman's Critique of Sustainable Development. *Environmental Values* 4 (1):49-55.

de Groot, Rudolf S. 1992. *Functions of Nature: Evaluation of Nature in Environmental Planning, Management, and Decision Making*. Amsterdam Wolters-Noordhoff.

de Groot, Rudolf S., Rob Alkemade, Leon Braat, Lars Hein, and Louise Willemen. 2010. Challenges in Integrating the Concept of Ecosystem Services and Values in Landscape Planning, Management and Decision Making. *Ecological Complexity* 7 (3):260-272.

de Groot, Rudolf S., Matthew A. Wilson, and Roelof M. J. Boumans. 2002. A Typology for the Classification, Description and Valuation of Ecosystem Functions, Goods and Services. *Ecological Economics* 41 (3):393-408.

Demeritt, D. 2002. What is the 'Social Construction of Nature'? A Typology and Sympathetic Critique. *Progress in Human Geography* 26 (6):767-790.

Denevan, William M. 1992. The Pristine Myth: The Landscape of the Americas in 1492. *Annals of the Association of American Geographers* 82 (3):369-385.

Descola, Philippe, and Gísli Pálsson. 1996. *Nature and Society: Anthropological Perspectives*. New York: Routledge.

Diamond, David D., C. Diane True, and Kan He. 1997. Regional Priorities for Conservation of Rare Species in Texas. *The Southwestern Naturalist* 42 (4):400-408.

Diamond, Jared. 2005. *Collapse: How Societies Choose to Fail or Succeed*. New York: Viking Press.

Dietz, T., E. Ostrom, and P.C. Stern. 2003. The Struggle to Govern the Commons. *Science* 302 (5652):1907.

Egoh, Benis, Belinda Reyers, Mathieu Rouget, Michael Bode, and David M. Richardson. 2009. Spatial Congruence between Biodiversity and Ecosystem Services in South Africa. *Biological Conservation* 142 (3):553-562.

Egoh, Benis, Belinda Reyers, Mathieu Rouget, David M. Richardson, David C. Le Maitre, and Albert S. van Jaarsveld. 2008. Mapping Ecosystem Services for Planning and Management. *Agriculture, Ecosystems & Environment* 127 (1):135-140.

Ehrlich, Paul R., and Edward O. Wilson. 1991. Biodiversity Studies: Science and Policy. *Science* 253:758-761.

Elkington, John. 2004. Enter the Triple Bottom Line. In *The Triple Bottom Line: Does it All Add Up*, edited by A. Henriques and J. Richardson. London: Earthscan.

EPA. *Integrated Climate and Land Use Scenarios (ICLUS) Project*. United States Environmental Protection Agency 2009. Available from http://cfpub.epa.gov/ncea/global/recordisplay.cfm?deid=257306.

———. 2009. Land-Use Scenarios: National-Scale Housing-Density Scenarios Consistent with Climate Change Storylines. Washington, D.C.: U.S. Environmental Protection Agency.

———. *Ecoregions of North America*. U.S. Environmental Protection Agency 2013. Available from http://www.epa.gov/wed/pages/ecoregions/na_eco.htm#Downloads.

Farber, S., R. Costanza, D.L. Childers, J. Erickson, K. Gross, M. Grove, C.S. Hopkinson, J. Kahn, S. Pincetl, and A. Troy. 2006. Linking Ecology and Economics for Ecosystem Management. *BioScience* 56 (2):121-133.

Fay, Michael P., and Michael A. Proschan. 2010. Wilcoxon-Mann-Whitney or t-test? On Assumptions for Hypothesis Tests and Multiple Interpretations of Decision Rules. *Statistics Surveys* 4:1.

Ferrier, Simon, and Graham Watson. 1997. An Evaluation of the Effectiveness of Environmental Surrogates and Modelling Techniques in Predicting the Distribution of Biological Diversity. Environment Australia, Canberra.

Fisher, Brendan, R. Kerry Turner, and Paul Morling. 2009. Defining and Classifying Ecosystem Services for Decision Making. *Ecological Economics* 68 (3):643-653.

Foley, J.A., R. DeFries, G.P. Asner, C. Barford, G. Bonan, S.R. Carpenter, F.S. Chapin, M.T. Coe, G.C. Daily, and H.K. Gibbs. 2005. Global Consequences of Land Use. *Science* 309 (5734):570.

Forman, Richard T.T. 1995. *Land Mosaics: The Ecology of Landscapes and Regions*. Cambridge: Cambridge University Press.

Forman, Richard T.T. 1995. Some General Principles of Landscape and Regional Ecology. *Landscape Ecology* 10 (3):133-142.

Fregonese. *Envision Tommorrow*. Fregonese Associates 2014. Available from http://www.envisiontomorrow.org/.

Gabriel, D., S.M. Sait, J.A. Hodgson, U. Schmutz, W.E. Kunin, and T.G. Benton. 2010. Scale Matters: The Impact of Organic Farming on Biodiversity at Different Spatial Scales. *Ecology Letters* 13 (7):858-869.

García-Oliva, Felipe, and Omar R. Masera. 2004. Assessment and Measurement Issues Related to Soil Carbon Sequestration in Land-Use, Land-Use Change, and Forestry (LULUCF) Projects Under the Kyoto Protocol. *Climatic Change* 65 (3):347-364.

Gifford, T. 1996. The Social Construction of Nature. *Interdisciplinary Studies in Literature and Environment* 3 (2):27-35.

Goulder, Lawrence H., Donald Kennedy, and G.C. Daily. 1997. Valuing Ecosystem Services: Philosophical Bases and Empirical Methods. In *Nature's Services: Societal Dependence on Natural Ecosystems.*, edited by G. C. Daily. Washington, D.C.: Island Press.

Gray, Murray. 2013. *Geodiversity: Valuing and Conserving Abiotic Nature*. Chichester: John Wiley & Sons.

Greider, T., and L. Garkovich. 1994. Landscapes: The Social Construction of Nature and the Environment. *Rural Sociology* 59 (1):1-24.

Gren, Ing-Marie, and Lina Isacs. 2009. Ecosystem Services and Regional Development: An Application to Sweden. *Ecological Economics* 68 (10):2549-2559.

Groves, C.R., D.B. Jensen, L.L. Valutis, K.H. Redford, M.L. Shaffer, J.M. Scott, J.V. Baumgartner, J.V. Higgins, M.W. Beck, and M.G. Anderson. 2002. Planning for

Biodiversity Conservation: Putting Conservation Science into Practice. *BioScience* 52 (6):499-512.

Gunderson, L.H., and C.S. Holling. 2002. *Panarchy: Understanding Transformations in Human and Natural Systems*. Washington, D.C.: Island Press.

Guo, Zhongwei, Lin Zhang, and Yiming Li. 2010. Increased Dependence of Humans on Ecosystem Services and Biodiversity. *PloS ONE* 5 (10):e13113.

Haberl, H., V. Winiwarter, K. Andersson, R.U. Ayres, C. Boone, A. Castillo, G. Cunfer, M. Fischer-Kowalski, W.R. Freudenburg, and E. Furman. 2006. From LTER to LTSER: Conceptualizing the Socioeconomic Dimension of Long-Term Socioecological Research. *Ecology and Society* 11 (2):13.

Heink, Ulrich, and Ingo Kowarik. 2010. What are Indicators? On the Definition of Indicators in Ecology and Environmental Planning. *Ecological Indicators* 10 (3):584-593.

Hester, Thomas R. 1986. Early Human Populations Along the Balcones Escarpment. In *The Balcones Escarpment-Geology, Hydrology, Ecology and Social Development in Central Texas*, edited by P. L. Abbott and C. M. Woodruff, Jr.: Geological Society of America.

Hijmans, Robert J., Susan Cameron, Juan Parra, Peter Jones, Andrew Jarvis, and Karen Richardson. *WorldClim: Global Climate Data: Bioclim* 2013. Available from http://www.worldclim.org/bioclim.

Hoekstra, J.M., T.M. Boucher, T.H. Ricketts, and C. Roberts. 2005. Confronting a Biome in Crisis: Global Disparities of Habitat Loss and Protection. *Ecology Letters* 8 (1):pp. 23-29.

Hoerling, Martin, Arun Kumar, Randall Dole, John W. Nielsen-Gammon, Jon Eischeid, Judith Perlwitz, Xiao-Wei Quan, Tao Zhang, Philip Pegion, and Mingyue Chen. 2013. Anatomy of an Extreme Event. *Journal of Climate* 26 (9):2811-2832.

Holland, Robert A., Felix Eigenbrod, Paul R. Armsworth, Barbara J. Anderson, Chris D. Thomas, Andreas Heinemeyer, Simon Gillings, David B. Roy, and Kevin J. Gaston. 2011. Spatial Covariation between Freshwater and Terrestrial Ecosystem Services. *Ecological Applications* 21 (6):2034-2048.

Holling, C. S. 1973. Resilience and Stability of Ecological Systems. *Annual Review of Ecology and Systematics* 4:1-23.

IMPLAN. 2008. IMPLAN System: Data and Software. IMPLAN Group.

Innes, Robert, Stephen Polasky, and John Tschirhart. 1998. Takings, Compensation and Endangered Species Protection on Private Lands. *The Journal of Economic Perspectives* 12 (3):35-52.

IPCC. 2006. IPCC Guidelines for National Greenhouse Gas Inventories. In *Volume 4: Agriculture, Forestry and Other Land Use*, edited by H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara and K. Tanabe. Hayama, Japan: The Intergovernmental Panel on Climate Change.

Izquierdo, Andrea E., and Matthew L. Clark. 2012. Spatial Analysis of Conservation Priorities Based on Ecosystem Services in the Atlantic Forest Region of Misiones, Argentina. *Forests* 3 (3):764-786.

Jack, B. Kelsey, Carolyn Kousky, and Katharine R.E. Sims. 2008. Designing Payments for Ecosystem Services: Lessons from Previous Experience with Incentive-Based Mechanisms. *Proceedings of the National Academy of Sciences* 105 (28):9465-9470.

Jelinski, Dennis E., and Jianguo Wu. 1996. The Modifiable Areal Unit Problem and Implications for Landscape Ecology. *Landscape Ecology* 11 (3):129-140.

Jones, K. Bruce, Giovanni Zurlini, Felix Kienast, Irene Petrosillo, Thomas Edwards, Timothy G. Wade, Bai-lian Li, and Nicola Zaccarelli. 2012. Informing Landscape Planning and Design for Sustaining Ecosystem Services from Existing Spatial Patterns and Knowledge. *Landscape Ecology* 28 (6):1175-1192.

Joppa, Lucas N., and Alexander Pfaff. 2009. High and Far: Biases in the Location of Protected Areas. *PloS ONE* 4 (12):e8273.

Kareiva, Peter, and Michelle Marvier. 2007. Conservation for the People. *Scientific American* 297 (4):50-57.

Kark, Salit. 2013. Effects of Ecotones on Biodiversity. In *Encyclopedia of Biodiversity*. New Jersey: Elsevier.

Kiesecker, Joseph M., Tosha Comendant, Terra Grandmason, Elizabeth Gray, Christine Hall, Richard Hilsenbeck, Peter Kareiva, Lynn Lozier, Patrick Naehu, and Adena Rissman. 2007. Conservation Easements in Context: A Quantitative Analysis of Their Use by The Nature Conservancy. *Frontiers in Ecology and the Environment* 5 (3):125-130.

Kimmerer, Robin Wall, and Frank Kanawha Lake. 2001. The Role of Indigenous Burning in Land Management. *Journal of Forestry* 99 (11):36-41.

Kirkpatrick, J.B. 1983. An Iterative Method for Establishing Priorities for the Selection of Nature Reserves: An Example from Tasmania. *Biological Conservation* 25 (2):127-134.

Kjelland, Michael E., Urs P. Kreuter, George A. Clendenin, R. Neal Wilkins, X. Ben Wu, Edith Gonzalez Afanador, and William E. Grant. 2007. Factors Related to Spatial Patterns of Rural Land Fragmentation in Texas. *Environmental Management* 40 (2):231-244.

Klein, C. J., A. Chan, L. Kircher, A.J. Cundiff, N. Gardner, Y. Hrovat, A. Scholz, B.E. Kendall, and S. Airame. 2008. Striking a Balance between Biodiversity Conservation and Socioeconomic Viability in the Design of Marine Protected Areas. *Conservation Biology* 22 (3):691-700.

Knight, Richard L. 1999. Private Lands: The Neglected Geography. *Conservation Biology* 13 (2):223-224.

Kosoy, Nicolás, and Esteve Corbera. 2010. Payments for Ecosystem Services as Commodity Fetishism. *Ecological Economics* 69 (6):1228-1236.

Kossoy, Alexandre, and Pierre Guigon. 2012. State and Trends of the Carbon Market 2012. Washington, D.C.: World Bank.

Langholz, Jeffrey A., and James P. Lassoie. 2001. Perils and Promise of Privately Owned Protected Areas. *BioScience* 51 (12):1079-1085.

Larsen, Frank W., Maria C. Londoño-Murcia, and Will R. Turner. 2011. Global Priorities for Conservation of Threatened Species, Carbon Storage, and Freshwater Services: Scope for Synergy? *Conservation Letters* 4 (5):355-363.

Leopold, Aldo. 1949. *A Sand County Almanac*. New York: Oxford University Press.

Levin, Simon A., and Jeffrey A. Harvey. 1999. *Fragile Dominion: Complexity and the Commons*. Cambridge, MA: Perseus

Light, Andrew. 2010. The Moral Journey of Environmentalism: From Wilderness to Place. In *Pragmatic Sustainability: Theoretical and Practical Tools*, edited by S. A. Moore. London: Routledge.

Lindenmayer, David B., and Joern Fischer. 2006. *Habitat Fragmentation and Landscape Change: An Ecological and Conservation Synthesis*. Washington, D.C.: Island Press.

Liu, J., T. Dietz, S.R. Carpenter, M. Alberti, C. Folke, E. Moran, A.N. Pell, P. Deadman, T. Kratz, and J. Lubchenco. 2007. Complexity of Coupled Human and Natural Systems. *Science* 317 (5844):1513.

Liu, Shuang, Neville D. Crossman, Martin Nolan, and Hiyoba Ghirmay. 2013. Bringing Ecosystem Services into Integrated Water Resources Management. *Journal of Environmental Management* 129:92-102.

Lockwood, Mark. 2001. *Birds of the Texas Hill Country*. Austin: The University of Texas Press.

Loomis Austin, Inc. 2008. Mapping Potential Golden-Cheeked Warbler Breeding Habitats Using Remotely Sensed Forest Canopy Cover Data. Austin, Texas.

Lövbrand, Eva. 2004. Bridging Political Expectations and Scientific Limitations in Climate Risk Management–On the Uncertain Effects of International Carbon Sink Policies. *Climatic Change* 67 (2-3):449-460.

MA. 2005. *Millienium Ecosystem Assessment: Ecosystems and Human Well-Being*. Vol. 5. Washington, D.C.: Island Press.

MacArthur, Robert H., and Edward O. Wilson. 1967. *The Theory of Island Biogeography*. Princeton: Princeton University Press.

Machado, E.A., D.M. Stoms, F.W. Davis, and J. Kreitler. 2006. Prioritizing Farmland Preservation Cost-Effectively for Multiple Objectives. *Journal of Soil and Water Conservation* 61 (5):250-258.

Maes, J., M.L. Paracchini, G. Zulian, M.B. Dunbar, and R. Alkemade. 2012. Synergies and Trade-offs between Ecosystem Service Supply, Biodiversity, and Habitat Conservation Status in Europe. *Biological Conservation* 155 (0):1-12.

Margules, C.R., A.O. Nicholls, and R.L. Pressey. 1988. Selecting Networks of Reserves to Maximise Biological Diversity. *Biological Conservation* 43 (1):63-76.

Margules, C.R., and R.L. Pressey. 2000. Systematic Conservation Planning. *Nature* 405 (6783):243-253.

Margules, C.R., R.L. Pressey, and P.H. Williams. 2002. Representing Biodiversity: Data and Procedures for Identifying Priority Areas for Conservation. *Journal of Biosciences* 27 (4):309-326.

Margules, C.R., and S. Sarkar. 2007. *Systematic Conservation Planning*. Cambridge: Cambridge University Press.

McBride, Marissa F., Kerrie A. Wilson, Jutta Burger, Yi-Chin Fang, Megan Lulow, David Olson, Mike O'Connell, and Hugh P. Possingham. 2010. Mathematical Problem Definition for Ecological Restoration Planning. *Ecological Modelling* 221 (19):2243-2250.

McDonnell, Mark J., Steward T.A. Pickett, Peter Groffman, Patrick Bohlen, Richard V. Pouyat, Wayne C. Zipperer, Robert W. Parmelee, Margaret M. Carreiro, and Kimberly Medley. 1997. Ecosystem Processes Along an Urban-to-Rural Gradient. *Urban Ecosystems* 1 (1):21-36.

McElwee, Pamela D. 2012. Payments for Environmental Services as Neoliberal Market-Based Forest Conservation in Vietnam: Panacea or Problem? *Geoforum* 43 (3):412-426.

McHarg, Ian L. 1969. *Design with Nature*. New York: John Wiley and Sons.

McKinney, M.L. 2002. Urbanization, Biodiversity, and Conservation. *BioScience* 52 (10):883-890.

Meyerson, Frederick A.B., Leticia Merino, and Jorge Durand. 2007. Migration and Environment in the Context of Globalization. *Frontiers in Ecology and the Environment* 5 (4):182-190.

Moilanen, Atte, Kerrie A. Wilson, and Hugh P. Possingham. 2009. *Spatial Conservation Prioritization: Quantitative Methods and Computational Tools*. Oxford: Oxford University Press.

Morris, Amy Wilson. 2008. Easing Conservation? Conservation Easements, Public Accountability and Neoliberalism. *Geoforum* 39 (3):1215-1227.

Muir, John. 1901. *Our National Parks*. Boston: Houghton Mifflin.

Müller, Felix, and Benjamin Burkhard. 2012. The Indicator Side of Ecosystem Services. *Ecosystem Services* 1 (1):26-30.

Musacchio, Laura R. 2009. The Scientific Basis for the Design of Landscape Sustainability: A Conceptual Framework for Translational Landscape Research and Practice of Designed Landscapes and the Six E's of Landscape Sustainability. *Landscape Ecology* 24 (8):993-1013.

Nackoney, Janet, and David Williams. 2013. A Comparison of Scenarios for Rural Development Planning and Conservation in the Democratic Republic of the Congo. *Biological Conservation* 164:140-149.

Naess, Arne. 1995. The Shallow and the Deep, Long-Range Ecology Movements: A Summary. In *Deep Ecology for the 21st Century: Readings on the Philosophy and Practice of the New Environmentalism*, edited by G. Sessions. Boston: Shambala.

Naidoo, R., A. Balmford, R. Costanza, B. Fisher, R.E. Green, B. Lehner, T.R. Malcolm, and T.H. Ricketts. 2008. Global Mapping of Ecosystem Services and Conservation Priorities. *Proceedings of the National Academy of Sciences* 105 (28):9495-9500.

Naidoo, R., and T.H. Ricketts. 2006. Mapping the economic costs and benefits of conservation. *PLoS Biology* 4 (11):e360.

Nash, Roderick. 2001. *Wilderness and the American Mind*. New Haven: Yale University Press.
Nassauer, J.I., and P. Opdam. 2008. Design in Science: Extending the Landscape Ecology Paradigm. *Landscape Ecology* 23 (6):633-644.
Nelson, E., G. Mendoza, J. Regetz, S. Polasky, H. Tallis, D.R. Cameron, K.M.A. Chan, G.C. Daily, J. Goldstein, and P.M. Kareiva. 2009. Modeling Multiple Ecosystem Services, Biodiversity Conservation, Commodity Production, and Tradeoffs at Landscape Scales. *Frontiers in Ecology and the Environment* 7 (1):4-11.
Nhancale, Bruno A., and Robert J. Smith. 2011. The Influence of Planning Unit Characteristics on the Efficiency and Spatial Pattern of Systematic Conservation Planning Assessments. *Biodiversity and Conservation* 20 (8):1821-1835.
Nicholls, A.O., and C.R. Margules. 1993. An Upgraded Reserve Selection Algorithm. *Biological Conservation* 64 (2):165-169.
Nielsen-Gammon, John W. 2011. The Changing Climate of Texas. In *The Impact of Global Warming on Texas*, edited by J. Schmandt, G. R. Iorth and J. Clarkson. Austin: The University of Texas Press.
Niemelä, Jari, Sanna-Riikka Saarela, Tarja Söderman, Leena Kopperoinen, Vesa Yli-Pelkonen, Seija Väre, and D. Johan Kotze. 2010. Using the Ecosystem Services Approach for Better Planning and Conservation of Urban Green Spaces: a Finland Case Study. *Biodiversity and Conservation* 19 (11):3225-3243.
Novacek, Michael J. 2008. Engaging the Public in Biodiversity Issues. *Proceedings of the National Academy of Sciences* 105 (Supplement 1):11571-11578.
Odling-Smee, Lucy. 2005. Conservation: Dollars and Sense. *Nature* 437 (7059):614-616.
Ogren, Jonathan 2008. Conservation Planning in Central Texas. Thesis, Geography and the Environment, The University of Texas at Austin.
Onaindia, Miren, Beatriz Fernández de Manuel, Iosu Madariaga, and Gloria Rodríguez-Loinaz. 2013. Co-benefits and Trade-offs between Biodiversity, Carbon Storage and Water Glow Regulation. *Forest Ecology and Management* 289:1-9.
Orme, C. David L., Richard G. Davies, Malcolm Burgess, Felix Eigenbrod, Nicola Pickup, Valerie A. Olson, Andrea J. Webster, Tzung-Su Ding, Pamela C. Rasmussen, Robert S. Ridgely, Ali J. Stattersfield, Peter M. Bennett, Tim M. Blackburn, and Kevin J. Gaston. 2005. Global Hotspots of Species Richness are not Congruent with Endemism or Threat. *Nature* 436:1016 - 1019.
Ostrom, Elinor. 1990. *Governing the Commons: The Evolution of Institutions for Collective Action*. Cambridge: Cambridge University Press.
Palmer, E. Charles. 1986. Land Use and Cultural Change along the Balcones Escarpment: 1718-1986. In *The Balcones Escarpment-Geology, Hydrology, Ecology and Social Development in Central Texas*, edited by P. L. Abbott and C. M. Woodruff, Jr.: Geological Society of America.
Pearce, David. 2007. Do We Really Care about Biodiversity? *Environmental and Resource Economics* 37 (1):313-333.
Pearce, Jennie L., David Anthony Kirk, Cynthia P. Lane, Marguerite H. Mahr, John Walmsley, Daniel Casey, Judy E. Muir, Susan Hannon, Andrew Hansen, and

Kingsford Jones. 2008. Prioritizing Avian Conservation Areas for the Yellowstone to Yukon Region of North America. *Biological Conservation* 141 (4):908-924.

Perrings, Charles, Shahid Naeem, Farshid S. Ahrestani, Daniel E. Bunker, Peter Burkill, Graciela Canziani, Thomas Elmqvist, Jed A. Fuhrman, Fabian M. Jaksic, and Zen'ichiro Kawabata. 2011. Ecosystem Services, Targets, and Indicators for the Conservation and Sustainable Use of Biodiversity. *Frontiers in Ecology and the Environment* 9 (9):512-520.

Pimentel, David, Christa Wilson, Christine McCullum, Rachel Huang, Paulette Dwen, Jessica Flack, Quynh Tran, Tamara Saltman, and Barbara Cliff. 1997. Economic and Environmental Benefits of Biodiversity. *BioScience* 47 (11):747-757.

Pinchot, Gifford. 1910. *The Fight for Conservation*. New York: Doubleday, Page & Company.

Plieninger, Tobias, Sebastian Dijks, Elisa Oteros-Rozas, and Claudia Bieling. 2013. Assessing, Mapping, and Quantifying Cultural Ecosystem Services at Community Level. *Land Use Policy* 33:118-129.

Polasky, Stephen, Erik Nelson, Derric Pennington, and Kris A. Johnson. 2011. The Impact of Land-Use Change on Ecosystem Services, Biodiversity and Returns to Landowners: A Case Study in the State of Minnesota. *Environmental and Resource Economics* 48 (2):219-242.

Potschin, Marion, and Roy Haines-Young. 2006. “Rio+10”, Sustainability Science and Landscape Ecology. *Landscape and Urban Planning* 75 (3–4):162-174.

Potschin, Marion, and Roy Haines-Young. 2013. Landscapes, Sustainability and the Place-Based Analysis of Ecosystem Services. *Landscape Ecology* 28 (6):1053-1065.

Pressey, R.L., and R.M. Cowling. 2001. Reserve Selection Algorithms and the Real World. *Conservation Biology* 15 (1):275-277.

Rands, Michael R.W., William M. Adams, Leon Bennun, Stuart H.M. Butchart, Andrew Clements, David Coomes, Abigail Entwistle, Ian Hodge, Valerie Kapos, Jörn P.W. Scharlemann, William J. Sutherland, and Bhaskar Vira. 2010. Biodiversity Conservation: Challenges Beyond 2010. *Science* 329 (5997):1298-1303.

Reckhow, Kenneth H., Michael N. Beaulac, and Jonathan T. Simpson. 1980. Modeling Phosphorus Loading and Lake Response Under Uncertainty: A Manual and Compilation of Export Coefficients. Washington, D.C.: U.S. Environmental Protection Agency.

Reyers, Belinda, Reinette Biggs, Graeme S. Cumming, Thomas Elmqvist, Adam P. Hejnowicz, and Stephen Polasky. 2013. Getting the Measure of Ecosystem Services: A Social-Ecological Approach. *Frontiers in Ecology and the Environment* 11 (5):268-273.

Rodrigues, A.S.L., S.J. Andelman, M.I. Bakarr, L. Boitani, T.M. Brooks, R.M. Cowling, L.D.C. Fishpool, G.A.B. da Fonseca, K.J. Gaston, and M. Hoffmann. 2004. Effectiveness of the Global Protected Area Network in Representing Species Diversity. *Nature* 428 (6983):640-643.

Rodrigues, A.S.L., and T.M. Brooks. 2007. Shortcuts for Biodiversity Conservation Planning: The Effectiveness of Surrogates. *Annual Review of Ecology, Evolution, and Systematics* 38:713-737.

Roseland, M. 2000. Sustainable Community Development: Integrating Environmental, Economic, and Social Objectives. *Progress in Planning* 54 (2):73-132.

Salafsky, N., R. Margoluis, K.H. Redford, and J.G. Robinson. 2002. Improving the Practice of Conservation: A Conceptual Framework and Research Agenda for Conservation Science. *Conservation Biology* 16 (6):1469-1479.

Sale, Kirkpatrick. 1990. *The Conquest of Paradise: Christopher Columbus and the Columbian Legacy*. New York: Penguin.

Sánchez-Canales, María, Alfredo López Benito, Ana Passuello, Marta Terrado, Guy Ziv, Vicenç Acuña, Marta Schuhmacher, and F. Javier Elorza. 2012. Sensitivity Analysis of Ecosystem Service Valuation in a Mediterranean Watershed. *Science of The Total Environment* 440:140-153.

Sanderson, E.W., M. Jaiteh, M.A. Levy, K.H. Redford, A.V. Wannebo, and G. Woolmer. 2002. The Human Footprint and the Last of the Wild. *BioScience* 52 (10):891-904.

Sarkar, S., R.L. Pressey, D.P. Faith, C.R. Margules, T. Fuller, D.M. Stoms, A. Moffett, K.A. Wilson, K.J. Williams, and P.H. Williams. 2006. Biodiversity Conservation Planning Tools: Present Status and Challenges for the Future. *Annu. Rev. Environ. Resour.* 31:123-159.

Schmitt, Laetitia Helene Marie, and Cecile Brugere. 2013. Capturing Ecosystem Services, Stakeholders' Preferences and Trade-Offs in Coastal Aquaculture Decisions: A Bayesian Belief Network Application. *PloS ONE* 8 (10):e75956.

Schwartz, Mark W. 2008. The Performance of the Endangered Species Act. *Annual Review of Ecology, Evolution, and Systematics* 39:279-299.

Scott, J. Michael, Frank W. Davis, R. Gavin McGhie, R. Gerald Wright, Craig Groves, and John Estes. 2001. Nature Reserves: Do They Capture the Full Range of America's Biological Diversity? *Ecological Applications* 11 (4):999-1007.

Selman, P. 2008. What Do We Mean by Sustainable Landscape? *Sustainability: Science Practice and Policy* 4 (2):23-28.

Shetler, Sidney. 1991. Three Faces of Eden. In *Seeds of Change: A Quincentennial Commemoration*, edited by H. I. Viola and C. Margolis. Washington, D.C.: Smithsonian Institution Scholarly Press.

Siglo. 2012. Land for Water: A Conservation Initiative: A Strategic Framework for Landscape-Level Changes. Texas Land Conservancy.

———. 2013. Strategic Conservation Plan. Hill Country Conservancy.

Smith, Eric Alden, and Mark Wishnie. 2000. Conservation and Subsistence in Small-Scale Societies. *Annual Review of Anthropology* 29:493-524.

Snape, William J. 1996. *Biodiversity and the Law*. Washingtion, D.C.: Island Press.

Sorice, M.G. 2008. Understanding Participation in Wildlife Conservation Programs on Private Lands. Dissertation, Wildlife and Fisheries Sciences, Texas A&M University, College Station.

Sorice, M.G., U.P. Kreuter, B.P. Wilcox, and W.E. Fox III. 2012. Classifying Land-Ownership Motivations in Central, Texas, USA: A First Step in Understanding Drivers of Large-Scale Land Cover Change. *Journal of Arid Environments* 80:56-64.

Soule, Michael E., and Bruce A. Wilcox. 1980. *Conservation Biology: An Evolutionary-Ecological Perspective*. Sunderland: Sinauer Associates.

Steffen, Will, Paul J. Crutzen, and John R. McNeill. 2007. The Anthropocene: Are Humans Now Overwhelming the Great Forces of Nature. *AMBIO: A Journal of the Human Environment* 36 (8):614-621.

Stoms, David M., Jason Kreitler, and Frank W. Davis. 2011. The Power of Information for Targeting Cost-Effective Conservation Investments in Multifunctional Farmlands. *Environmental Modelling & Software* 26 (1):8-17.

Tallis, H.T., P. Kareiva, M. Marvier, and A. Chang. 2008. An Ecosystem Services Framework to Support both Practical Conservation and Economic Development. *Proceedings of the National Academy of Sciences* 105 (28):9457-9464.

Tallis, H.T., and S. Polasky. 2009. Mapping and Valuing Ecosystem Services as an Approach for Conservation and Natural-Resource Management. *Annals of the New York Academy of Sciences* 1162 (1):265-283.

Tallis, H.T., T. Ricketts, A.D. Guerry, S.A. Wood, R. Sharp, E. Nelson, D. Ennaanay, S. Wolny, N. Olwero, K. Vigerstol, D. Pennington, G. Mendoza, J. Aukema, J. Foster, J. Forrest, D. Cameron, K. Arkema, E. Lonsdorf, C. Kennedy, G. Verutes, C.K. Kim, G. Guannel, M. Papenfus, J. Toft, M. Marsik, and J. Bernhardt. 2013. InVEST 2.5.6 User's Guide. The Natural Capital Project, Stanford University.

TEEB. 2010. The Economics of Ecosystems and Biodiversity: Mainstreaming the Economies of Nature: A Synthesis of the Approach, Conclusions and Recommendations of TEEB. The Economics of Ecosytems and Biodiversity.

Theobald, D.M. 2001. Land-Use Dynamics Beyond the American Urban Fringe. *Geographical Review* 91 (3):544-564.

Theobald, D.M. 2005. Landscape Patterns of Exurban Growth in the USA from 1980 to 2020. *Ecology and Society* 10 (1):32.

Theobald, D.M., and N.T. Hobbs. 1998. Forecasting Rural Land-Use Change: A Comparison of Regression and Spatial Transition-Based Models. *Geographical and Environmental Modelling* 2:65-82.

Thin, Neil. 2002. *Social Progress and Sustainable Development*. West Hartford: Kumarian Press.

Thomas, Chris D., Barbara J. Anderson, Atte Moilanen, Felix Eigenbrod, Andreas Heinemeyer, Tristan Quaife, David B. Roy, Simon Gillings, Paul R. Armsworth, and Kevin J. Gaston. 2013. Reconciling Biodiversity and Carbon Conservation. *Ecology Letters* 16:39-47.

Thoreau, Henry D. 1854. *Walden*. Edited by J. L. Shanley. 150th Anniversary ed. Princeton: Princeton University Press. Original edition, 1854.

TLTC. *Conservation Lands Inventory*. Texas Land Trust Council 2013. Available from http://www.texaslandtrustcouncil.org/conservation-lands-inventory.

TNC. 2004. A Biodiversity and Conservation Assessment of the Edwards Plateau Ecoregion. The Nature Conservancy.
TNRIS. *Maps and Data*. Texas Natural Resources Information System 2013. Available from http://www.tnris.org/get-data.
Tol, Richard S.J. 2009. The Economic Effects of Climate Change. *The Journal of Economic Perspectives* 23 (2):29-51.
Tol, Richard S.J. 2011. The Social Cost of Carbon. *Annual Review of Resource Economics* 3 (1):419-443.
TPWD. 2010. Texas Ecological Systems Project: Phase 1. Austin: Texas Parks and Wildlife Department.
———. *GIS Lab Data Downloads: GIS Data*. Texas Parks and Wildlife Department 2013. Available from http://www.tpwd.state.tx.us/landwater/land/maps/gis/data_downloads/.
———. *Ecologically Significant Stream Segments*. Texas Parks and Wildlife Department 2014. Available from http://www.tpwd.state.tx.us/landwater/water/environconcerns/water_quality/sigsegs/.
———. *Nongame Wildlife in the Texas Hill Country*. Texas Parks and Wildlife Department 2014. Available from http://www.tpwd.state.tx.us/landwater/land/habitats/hillcountry/nongame/.
Trombulak, Stephen C. 2010. Assessing Irreplaceability for Systematic Conservation Planning. In *Landscape-Scale Conservation Planning*, edited by S. C. Trombulak and R. F. Baldwin. New York: Springer.
Trombulak, Stephen C., and Robert F. Baldwin. 2010. Introduction: Creating a Context for Landscape-Scale Conservation Planning. In *Landscape-Scale Conservation Planning*, edited by S. C. Trombulak and R. F. Baldwin. New York: Springer.
TSDC. *Texas Population Projections Program*. Texas State Data Center 2013. Available from http://txsdc.utsa.edu/Data/TPEPP/Projections/Index.aspx.
Turner, B. L., Roger E. Kasperson, Pamela A. Matson, James J. McCarthy, Robert W. Corell, Lindsey Christensen, Noelle Eckley, Jeanne X. Kasperson, Amy Luers, Marybeth L. Martello, Colin Polsky, Alexander Pulsipher, and Andrew Schiller. 2003. A Framework for Vulnerability Analysis in Sustainability Science. *Proceedings of the National Academy of Sciences* 100 (14):8074-8079.
Turner, Monica G. , Daniel C. Donato, and William H. Romme. 2013. Consequences of Spatial Heterogeneity for Ecosystem Services in Changing Forest Landscapes: Priorities for Future Research. *Landscape Ecology* 28 (6):1081-1097.
Turner, Monica G., and Robert H. Gardner. 1991. *Quantitative Methods in Landscape Ecology*. New York: Springer.
Turner, Will R., Katrina Brandon, Thomas M. Brooks, Robert Costanza, Gustavo A.B. Da Fonseca, and Rosimeiry Portela. 2007. Global Conservation of Biodiversity and Ecosystem Services. *BioScience* 57 (10):868-873.
TWDB. 2012. Water for Texas: 2012 State Water Plan. Texas Water Development Board.

TXCIP. *Texas County Profiles*. Texas Association of Counties. Available from http://www.txcip.org/tac/census/CountyProfiles.php.

USCB. 2013. Cumulative Estimates of Resident Population Change for Incorporated Places over 50,000, Ranked by Percent Change: April 1, 2010 to July 1, 2012. U.S. Census Bureau.

USDA. 1997. Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE). edited by K. G. Renard, G. R. Foster, G. A. Weesies, D. K. McCool and D. C. Yoder: U.S. Department of Agriculture.

USGS. 2013. National Hydrography Dataset (NHD). U.S. Geologic Survey.

Van Auken, O.W. 2000. Shrub Invasions of North American Semiarid Grasslands. *Annual Review of Ecology and Systematics* 31 (1):197-215.

van Berkel, Derek B., and Peter H. Verburg. 2014. Spatial Quantification and Valuation of Cultural Ecosystem Services in an Agricultural Landscape. *Ecological Indicators* 37:163-174.

Vane-Wright, R.I., C.J. Humphries, and P.H. Williams. 1991. What to Protect?—Systematics and the Agony of Choice. *Biological Conservation* 55 (3):235-254.

Vanston, J.H., W.P. Frisbie, S.C. Lopreato, and D.L. Boston. 1977. Alternate Scenario Planning. *Technological Forecasting and Social Change* 10 (2):159-180.

Vidal-Legaz, Beatriz, Julia Martínez-Fernández, Andrés Sánchez Picón, and Francisco I. Pugnaire. 2013. Trade-offs between Maintenance of Ecosystem Services and Socio-Economic Development in Rural Mountainous Communities in Southern Spain: A Dynamic Simulation Approach. *Journal of Environmental Management* 131:280-297.

Vitousek, P.M., H.A. Mooney, J. Lubchenco, and J.M. Melillo. 1997. Human Domination of Earth's Ecosystems. *Science* 277 (5325):494-499.

Vogel, Steven. 1996. Introduction. In *Against Nature: The Concept of Nature in Critical Theory*. Albany: State University of New York Press.

Walker, B.H., C.S. Holling, S.R. Carpenter, and A. Kinzig. 2004. Resilience, Adaptability and Transformability in Social-Ecological Systems. *Ecology and Society* 9 (2):5.

Walker, B.H., and D.A. Salt. 2006. *Resilience Thinking: Sustaining Ecosystems and People in a Changing World*. Washington, D.C.: Island Press.

Wallace, Ken J. 2007. Classification of Ecosystem Services: Problems and Solutions. *Biological Conservation* 139 (3–4):235-246.

Waters, M.R., S.L. Forman, T.A. Jennings, L.C. Nordt, S.G. Driese, J.M. Feinberg, J.L. Keene, J. Halligan, A. Lindquist, and J. Pierson. 2011. The Buttermilk Creek Complex and the Origins of Clovis at the Debra L. Friedkin Site, Texas. *Science* 331 (6024):1599.

Watts, M.E., I.R. Ball, R.S. Stewart, C.J. Klein, K. Wilson, C. Steinback, R. Lourival, L. Kircher, and H.P. Possingham. 2009. Marxan with Zones: Software for Optimal Conservation Based Land and Sea-Use Zoning. *Environmental Modelling & Software* 24 (12):1513-1521.

WCED. 1987. Our Common Future. New York: United Nations.

WDPA. *World Database on Protected Areas* 2014. Available from http://www.wdpa.org/Statistics.aspx.

Wilson, K.A., J. Carwardine, and H.P. Possingham. 2009. Setting Conservation Priorities. *Annals of the New York Academy of Sciences* 1162 (1):237-264.

Windhager, Steven, Frederick Steiner, Mark T. Simmons, and David Heymann. 2010. Toward Ecosystem Services as a Basis for Design. *Landscape Journal* 29 (2):107-123.

Wischmeier, Walter H., and Dwight David Smith. 1978. Predicting Rainfall Erosion Losses: A Guide to Conservation Planning. U.S. Department of Agriculture.

Woodruff, C. M., Jr., and L. P. Wilding. 2008. Bedrock, Soils, and Hillslope Hydrology in the Central Texas Hill Country, USA: Implications on Environmental Management in a Carbonate-Rock Terrain. *Environmental Geology* 55 (3):605-618.

Wrede, Jan. 2010. *Trees, Shrubs, and Vines of the Texas Hill Country*. College Station: Texas A&M University Press.

Wright, John B., and Robert J. Czerniak. 2000. The Rising Importance of Voluntary Methods of Land Use Control in Planning. *Journal of Planning Education and Research* 19 (4):419-423.

Wu, Jianguo. 2013. Landscape Sustainability Science: Ecosystem Services and Human Well-Being in Changing Landscapes. *Landscape Ecology* 28 (6):999-1023.

Zhang, Lu, W.R. Dawes, and G.R. Walker. 2001. Response of Mean Annual Evapotranspiration to Vegetation Changes at Catchment Scale. *Water Resources Research* 37 (3):701-708.

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