SPATIAL AND TEMPORAL CHANGES IN BIODIVERSITY AND ECOSYSTEM SERVICES PROVISION IN THE SAN ANTONIO RIVER BASIN, TEXAS, FROM 1984 TO 2010

A Dissertation

by

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ABSTRACT

Land changes significantly alter biodiversity and ecosystem services (BES), the benefits people obtain from ecosystems. It is important to accurately quantify the land changes in order to understand the implications of these changes in a tightly coupled social-ecological context. The San Antonio River Basin (SARB) is an ecologically diverse region in South Texas. The city of San Antonio is located within the basin and is the hub of the North American Free Trade Agreement (NAFTA).

First, I apply the benefit transfer method to estimate multi-scale changes in ecosystem service values using two sets of widely cited valuation coefficients. The valuation results indicate that ecosystem service values decreases substantially since the NAFTA was enacted in 1994. More importantly, the results from sensitivity analyses indicate that the high value placed on urban areas, substantially overestimated the ESV of urban land.

Second, I apply the spatially explicit ecosystem approach based on the ecological production function method and find the synergistic spatial associations between biodiversity, carbon storage, and sediment retention over time at multiple scales using the nonparametric correlation analysis. The hotspot and overlap analyses indicate the continued decline in the biodiversity and ecosystem functions. The rates of biodiversity loss and carbon storage degradation have accelerated since the implementation of NAFTA in 1994 and the environmental consequences are negatively related to the urban sprawl in the San Antonio region. The sensitivity analyses indicate that the provision of

carbon stocks is the most sensitive to forest cover and significantly linked with biodiversity loss in the SARB.

Third, I examine the environmental inequity of land in Bexar County from the perspective of environmental justice. The results suggest the spatial socio-economic segregation in public health risks and disparities in the changes in biodiversity and ecosystem services. Finally, I synthesize my findings and contributions; I also propose several policy interventions to mitigate biodiversity loss and ecosystem degradation and to internalize the negative externalities of urban sprawl in the San Antonio region.

DEDICATION

To my loving wife, Jiyoun Lee and three beautiful children,

Dayoon, Gyuhong, and William Gyusang Lee.

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NOMENCLATURE

ACS	American Community Surveys
ARIES	Artificial Intelligence for Ecosystem Services
BES	Biodiversity and Ecosystem Services
BTM	Benefit Transfer Method
CBD	Convention on Biological Diversity
DEM	Digital Elevation Model
EA	Ecosystem Approach
EARZ	Edward Aquifer Recharge Zone
EBM	Ecosystem Based Management
EJ	Environmental Justice
EPFM	Ecological Production Function Method
ESV	Ecosystem Services Valuation
ESVs	Ecosystem Service Values
HUC	Hydrologic Unit Code
InVEST	Integrated Valuation of Environmental Services and Tradeoffs
IPBES	Intergovernmental Platform on Biodiversity and Ecosystem Services
ISODATA	Iterative Self-Organizing Data Analysis
MEA	Millennium Ecosystem Assessment
MIMES	Multi-scale Integrated Models of Ecosystem Services

NAFTA	North American Free Trade Agreement
NATA	National Air Toxics Assessment
NDVI	Normalized Difference Vegetation Index
NHD	National Hydrographic Dataset
NLCD	National Land Cover Database
PES	Payments for Ecosystem Services
SARB	San Antonio River Basin
SDR	Sediment Delivery Ratio
SolVES	Social Values for Ecosystem Services
TEEB	The Economics of Ecosystems and Biodiversity
TxDOT	Texas Department of Transportation
WTP	Willingness To Pay

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CHAPTER I

INTRODUCTION

1.1. Background

Human induced land changes to meet the rapidly growing demands of increasing population and economic growth affect biodiversity and ecosystem functioning at multiple scales with significant implications for sustainability (DeFries and Eshleman, 2004; Foley et al., 2005; Grimm et al., 2008). Recent studies have revealed that biodiversity and ecosystem services (BES) are being lost at unprecedentedly rapid rates and land change continue to be a major driver of biodiversity loss and ecosystem disruption (MEA, 2005; TEEB, 2010; IPBES, 2015). Urbanization is the most transformative form of land change significantly affecting biodiversity and ecosystem functioning (DeFries et al., 2010; Seto et al., 2012). For example, urban areas emit considerable amounts of carbon dioxide (CO_2) and greenhouse gases (GHG); thus urban expansion affects climate change in terms of the carbon sequestration across multiple spatial scales (Nowak et al., 2006; Hutyra et al., 2011; UN-Habitat, 2016). Furthermore, urban lands produce extensive ecological footprints leading to loss of habitats critical for biodiversity (UNEP, 2011).

The San Antonio region, together with other major metropolitan centers in Texas, has experienced rapid population and economic growth over the last thirty years, which accelerated after the implementation of NAFTA in 1994. However, a basin-wide evaluation of the effects of population and economic growth on land and associated BES has so far been lacking. This represents a critical knowledge gap for evaluating economic growth of the region in a larger context that incorporates potential effects on the biodiversity and ecosystem services.

1.2. Literature Review

Ecosystem services refer to the range of “conditions and processes through which natural ecosystems, and the species that they contain, help sustain and fulfill human life” (Daily, 1997) and “the benefits people derive from ecosystems” to support the sustainable human well-being (Costanza et al., 1997; MEA, 2005). The Millennium Ecosystem Assessment (MEA) defined a framework for four main categories and their linkages to human well-being in terms of security, basic material for good life, health, and good social relations from the freedom of choice and action (MEA, 2005).

First, provisioning services focus on the products and production of food, fuel, fiber, fresh water, and genetic resources. Second, regulating services are the benefits from the regulation of ecosystem processes in terms of climate regulation, erosion control, and water purification. Third, cultural services are the nonmaterial benefits from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences. Fourth, supporting services are necessary for the production of all other ecosystem services, such as primary production, production of oxygen, and soil formation (MEA, 2005).

Ecosystem services valuation (ESV) refers to assessing the contribution of ecosystem services to achieve a specific goal (Costanza and Folke, 1997). The ESV often refers to quantifying and assigning monetary values to ecosystem goods and services in a practical way. However, Costanza et al., (2014) argues that valuation is different from the commodification or privatization in conventional markets. Daly (1992) suggests three main goals of ESV in ecological economics in terms of 1) sustainable scale, 2) just distribution, and 3) efficient allocations. First, whereas neo-classical economics assumes no scale limits to the economy focusing on the quantity and throughput of material, ESV concerns the quality of benefits and recognizes the importance of scale. Also, it attaches importance to the concept of development instead of growth. Second, just distribution refers to the division of the resource flow in terms of fairness. Moreover, Daly (1992) argues that future generations should be included in the value accounting from the perspective of equality. Third, neo-classical economics utilize the market and price signaling to allocate resources. However, market failure is prevalent, especially in the field of public goods (i.e., natural resources, ecosystem services), and this market mechanism cannot be addressed through the traditional pricing systems without considering shadow prices and opportunity costs. In this regard, in ESV, an efficient allocation should be based on individual preferences in order to address the externality of conventional market.

In neo-classical economics, a commodity is valuable to the extent it provides consumptive utility and the market only efficiently works for the allocation of goods and services that are both rival and excludable. Ecosystem services, by contrast, have been considered public goods or common pool resources with non-rival and non-excludable characteristics (Costanza, 2008). Thus, many ecosystem services have been underestimated in economic terms and heavily discounted or ignored in environmental decision-making processes. This leads to the free rider problem for public goods in which beneficiaries pay little or nothing for ecosystem provision, leading to a social outcome not sufficient for Pareto optimality (Barrett, 2007). In the case of public goods, ecological economists have suggested that assigning property rights will eliminate the externality problem (Daly and Farley, 2010). Coase theorem states that in a perfectly competitive market with clearly defined property rights, allocative efficiency will be achieved whether property rights are given to the polluter or the victim of pollution as long as transaction costs are zero (Coase, 1960). Therefore, establishing appropriate property rights is necessary but not sufficient condition for implementing payments for ecosystem services (PES) (Engel et al, 2008). Ecosystem services also need to be assigned appropriate economic values (Costanza et al, 1997)

ESV provides the useful framework to assess and compare the values of different ecosystem services for the human welfare. The rationale for establishing ecosystem service values is to assess the contribution of these services to the sustainable, equitable and efficient use of ecosystems (Costanza and Folke, 1997). Furthermore, establishing defensible economic values for ecosystem services provides a useful approach for comprehensively evaluating tradeoffs among alternative land uses (Ingraham and Foster, 2008; de Groot et al., 2012). Daily (1997) emphasizes that there are still considerable challenges to determining and assigning economic values to ecosystem services. First, values of ecosystem services are typically underestimated due to the lack of information about their supply. Second, lack of clear marginal value for ecosystem services makes marginal analysis challenging. Third, assigning economic value to specific ecosystem services is complicated by the spatial and temporal interdependency of these services and potential double counting in the ecosystem services valuation process.

A variety of non-market valuation methods have been developed to reveal the preference and willingness to pay (WTP) for ecosystem services (Farber et al., 2002; Liu et al., 2010). First, the productivity method estimates economic values for ecosystem products or services that contribute to the production of commercially marketed goods. Second, the hedonic pricing estimates economic values for ecosystem or environmental services that directly affect market prices of some other goods. Third, the travel cost method estimates economic values associated with ecosystems or sites that are used for recreation. Fourth, the damage cost avoided, the replacement cost, and the substitute cost methods estimate economic values based on the avoided costs of diminishment of ecosystem services, the costs of replacing such services, or the costs of providing substitute services. Fifth, the contingent valuation method directly determines the public's WTP for the retention or restoration of environmental services. Each of these methods has advantages and drawbacks and cannot be applied universally to estimate values for all ecosystem services; the relevance, strengths and limitations across time and space must be considered whenever values are to be estimated for ecosystem services that elude market pricing (Daly and Farley, 2010).

Two approaches to ESV that have been most broadly used include: 1) the benefit transfer method (BTM) (Costanza et al., 1997; 2014) and 2) the ecological production function method (EPFM) (Tallis and Polasky, 2009; Nelson et al., 2009; Kareiva et al., 2011). BTM estimates economic values by extrapolating value estimates from one or more study sites to other areas that are assumed to be ecologically and socio-economically similar (Brouwer, 2000; Woodward and Wui, 2001; Kreuter et al., 2001; Plummer, 2009; Daly and Farley, 2010; Koschke et al., 2012; Foody, 2015). Because of the difficulty of obtaining marginal values of public goods, BTM has been used extensively to obtain first order estimates of changes in ESV over time. Costanza et al. (1997) conducted global valuation of ecosystem services and estimated global ecosystem values in the range of US$ 16-54 trillion per year, with an average of US$ 33 trillion per year as a minimum estimate. Recently, Costanza et al. (2014) updated their unit values and estimated the global values of the global ecosystem services in 2011 to be US$ 125-145 trillion per year. However, BTM has been criticized because unit values derived from one area are applied as average unit values in all areas and do not necessarily reflect the marginal value of the same public good in other areas (Toman, 1998).

In contrast to BTM, EPFM emphasizes the application of ecological production functions for economic and social valuation (Tallis and Polasky, 2009) in the context of ecosystem based management; this method facilitates integrated management by treating land, water, and living resources – including humans – as essential components of ecosystems (Secretariat of the Convention on Biological Diversity, 2004). The National Research Council asserts that "the fundamental challenge of valuing ecosystem services lies in providing an explicit description and adequate assessment of the links between the structures and functions of natural systems, the benefits (i.e., goods and services) derived by humanity, and their subsequent values" (NRC, 2005, p2). In this regard, EPFM generates spatially explicit biophysical supply of ecosystem services and habitat quality associated with land changes over time at multiple scales (Kareiva et al., 2011; Sharp et al., 2016).

Nelson et al. (2009) applied an EPFM-based modeling tool, Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST), to assess changes in biodiversity and ecosystem services (BES) in a spatially explicit manner. They found highly synergistic relationships between a range of ecosystem services and biodiversity and suggested the payments for carbon sequestration to moderate tradeoffs between different development situations and biodiversity and ecosystem services. However, practical modeling tools are still lacking and the validity and reliability of model predictions at multiple scales, as well as multiple ecosystem services, should be assessed (Tallis and Polasky, 2009; Kareiva et al., 2011; Polasky et al., 2011).

There are several valuation toolkits for the ecosystem service assessment (Drakou et al., 2015). First, InVEST is an open source ecosystem services valuation model (Sharp et al., 2016). An ecological production function approach is applied to quantify and value ecosystem services linking production functions to human benefits (http://www.naturalcapitalproject.org/invest/). The main advantage of InVEST is that it allows for spatially explicit evaluation of ecosystem services. On the other hand, several modules of this tool still require a GIS platform to operate. Second, ARIES (Artificial Intelligence for Ecosystem Services) is an open source modeling framework to map ecosystem services and is characterized by its online interface function without the need for a separate GIS platform (http://ariesonline.org/). However, data and models are only available for several western U.S. states; to be more broadly useful, the aerial coverage of its database needs to be expanded. Third, MIMES (Multi-scale Integrated Models of Ecosystem Services) is a dynamic modeling system for simulating and mapping ecosystem services (http://www.afordablefutures.com/services/mimes). The tool focuses on simulating interactions of coupled human and natural systems. Fourth, Social Values for Ecosystem Services (SolVES) generates value indices using the GIS system to map social values for ecosystem services from a combination of spatial and non-spatial responses to public value and surveys (https://solves.cr.usgs.gov/). In summary, each tool has its strengths and weaknesses. In Chapter 3 of this dissertation, I use InVEST to value biodiversity and ecosystem services for the analyses of terrestrial habitat quality, carbon sequestration, and sediment retention. I used InVEST because it employs a spatially explicit ecological production function approach for multiple ecosystem services and habitat quality. Furthermore, it provides quantitative information that facilitates analyses of tradeoffs or synergies among various ecosystem services and the ecological impacts of land changes over space and time (Polaksy et al., 2011; Sallustio et al., 2015).

Most studies that quantify ecosystem services do not disaggregate beneficiaries and consider the distribution of benefits between groups and individuals in society (Daw et al., 2011). Recently, it has been recognized that research on ecosystem services (ES) has significant potential to address environmental justice (EJ) issues for better socio-ecological decision-making (Marshall and Gonzalez-Meler 2016; USEPA, 2016). Moreover, Aragão et al. (2016) argues that environmental justice is essential to ecosystem services valuation. Thus, disaggregating BES beneficiaries into winners and losers in terms of geographic location, ethnicity, and socio-economic status can reveal critical gaps in moving towards sustainability. Most studies focus on accessibility to urban green space (UGS) or parks as a proxy for environmental justice (EJ) (Comber et al., 2008; Wolch et al., 2014). Also, Calderón-Contreras and Quiroz-Rosas (2017) states Normalized Difference Vegetation Index (NDVI) has not typically been used for assessing the provision of urban ecosystem services. Importantly, there have been relatively few studies to directly link the provision of biodiversity and ecosystem services to socio-economic characteristics in a context of urban expansion and economic development. This dissertation research addresses this insufficiently examined socio-ecological issue and provides a preliminary examination of the relationships among the distribution of BES estimates, NDVI, public health risks, and socio-economic variables, including the race/ethnic groups from the EJ perspective (Hetrick et al. 2013; Chakraborty et al., 2014; Jennings et al., 2016).

1.3. Research Questions and Objectives

This dissertation research addresses two main knowledge gaps in the literature. First, this research examines the effects of two sets of valuation coefficients on the changes in ecosystem service values at multiple scales using sensitivity analyses. Second, this research examines the spatial relationship between biodiversity and ecosystem services in a spatially explicit manner and compares the BTM and EPFM for the better understanding of changes in biodiversity and ecosystem services (BES).

The research presented in this dissertation addresses the following two questions: (1) How did the land class and the associated ecosystem service values change in the San Antonio River Basin (SARB) and Bexar County from 1984 to 2010? (2) How did the spatial and temporal relationship between biodiversity and ecosystem services (BES) change in the SARB and Bexar County during the same period? To address these two questions the dissertation research proposes a novel conceptual framework (Figure 1.1) that incorporates BTM and EPFM, as well as BES estimates to analyze spatio-temporal association among biodiversity and two ecosystem services (carbon storage, sediment retention/export) to contribute to developing comprehensive and integrated policy alternatives in the context of ecosystem approach.

This conceptual framework includes three primary objectives to answer each research question with its own research significance.

- Objective 1: Spatio-temporal analysis of land change at a period of rapid change in the SARB.

- Objective 2: Multiscalar analysis of changes in ecosystem service values using the benefit transfer method with two sets of valuation coefficients at multiple scales.

- Objective 3: Multiscalar analysis of spatial and temporal associations between biodiversity and ecosystem services using the ecological production function method.

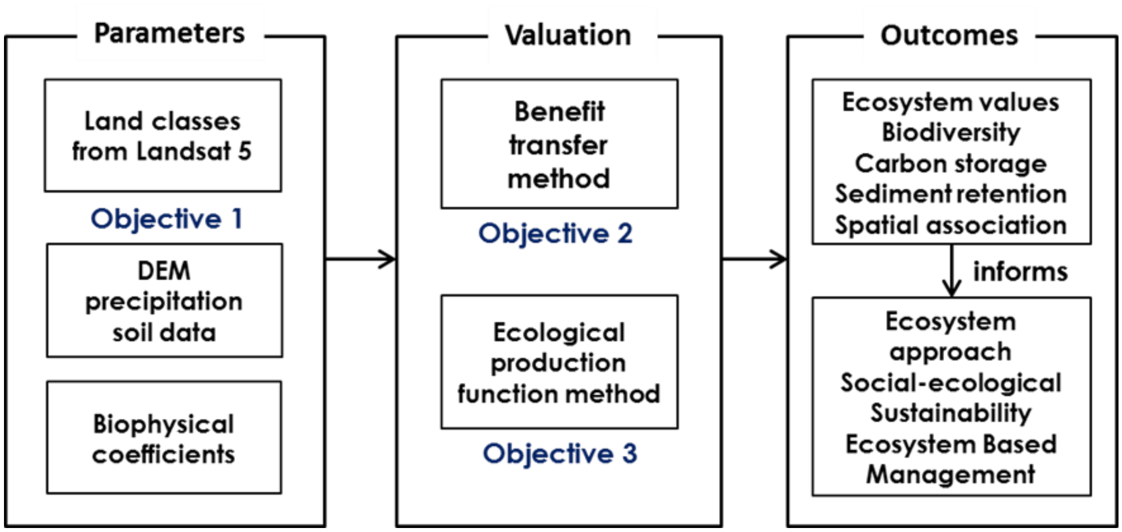

Figure 1.1. Conceptual framework of the dissertation research.

The rationale and significance of research is that it emphasizes the notion that biodiversity and ecosystem services contribute to the economy, livelihoods, good quality of human life, the long-term sustainability and resilience of society (IPBES, 2015). This research is novel for the study of ecosystem-based management in the region for the following three reasons. First, it systematically evaluates the effect of two types of unit value coefficients on total ESV in the San Antonio River Basin, contributing to the growing literature on BTM. Second, it investigates the spatial relationships between ecosystem services degradation and the loss of biodiversity in terms of spatially explicit EPFM in the context of NAFTA-induced urban expansion and economic development (Christensen et al., 1996). Third, the research makes a significant contribution to the development of more effective and feasible intervention strategies to improve overall ecosystem health, as well as urban sustainability and resilience by examining the these issues at multiple scales (Secretariat of the Convention on Biological Diversity, 2004).

The dissertation includes four chapters. Chapter 1 provides the background, literature review, research objectives and rationale pertaining to the research. Chapter 2 presents a land-change analysis using Landsat images from 1984, 1995, and 2010 to assess the environmental implications of this growth in the SARB. It also provides and analysis of spatiotemporal changes in ecosystem services across the SARB and within three watersheds in Bexar County, in which the city of San Antonio is located. Estimates of changes in ecosystem service values during this period are obtained through the combination of the results of the land-change analysis with BTM, using two sets of widely cited ecosystem-service valuation coefficients published in 1997 and 2014. Chapter 3 presents an analysis of changes in spatial and temporal associations among biodiversity, carbon storage, and sediment retention from 1984 to 2010 and uses EPFM in a spatially explicit manner. Chapter 4 concludes the dissertation by synthesizing the key findings, contributions, policy implications for the research for ecosystem based management, as well as the limitations of the study and suggested future research.

CHAPTER II

IMPACTS OF LAND CHANGE ON ECOSYSTEM SERVICES IN THE SAN ANTONIO RIVER BASIN, TEXAS, FROM 1984 TO 2010*

2.1. Introduction

A recent global assessment highlighted how massive urbanization is negatively impacting biodiversity and ecosystems around the world (Elmqvist et al., 2013). In particular, urban land expansion is one of the primary factors that affect the services humans derive from ecosystems (Millennium Ecosystem Assessment, MEA, 2005; Intergovernmental Panel on Climate Change, IPCC, 2007; Grimm et al., 2008). In the US where more than 80% of the population resides in urban areas, high rates of urban growth in the last several decades have led to various impacts on ecosystem services (Alberti, 2005; U.S. Census Bureau, 2010). Texas is one of the few states in the country where rapid urban growth is still prevalent. Over the past few decades, the state has experienced the largest increase in impervious surface cover in the US (Xian et al., 2011) concentrated around its three largest cities (Houston, San Antonio, and Dallas), which are among the ten largest US cities by population. Beyond these aggregate estimates, however, there is little understanding of how the growth of urban areas in the state impacted biodiversity and ecosystems.

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A major challenge in reducing the detrimental effects of economic development and urbanization on functional ecosystems is that many of the services these ecosystems provide are non-market public goods and, thus, economic values are poorly understood (Costanza et al., 2014; McDonald et al., 2014). The rationale for establishing ecosystem service values (ESVs) is to assess the contribution of these services to the sustainable, equitable and efficient use of ecosystems (Costanza and Folke, 1997). Additionally, establishing ESVs provides a useful approach for comprehensively evaluating tradeoffs among alternative land uses (Ingraham and Foster, 2008; de Groot et al., 2012).

The San Antonio River Basin (SARB) in south central Texas contains the rapidly urbanizing San Antonio Metropolitan Statistical Area. The city of San Antonio is the seventh most populous city in the US (U.S. Census Bureau, 2015) and a trade center of the North American Free Trade Agreement (NAFTA) (Brookings Institution, 2013). Since NAFTA was enacted in 1994, trade between the United States, Mexico, and Canada has grown significantly and reached $2.3 trillion in 2012. Bilateral trade between the United States and Mexico comprised 70% of this amount and increased 5-fold between 1993 and 2012 (U.S. Diplomatic Mission to Mexico, 2013). Currently, Mexico is the top country of origin for Texas imports (U.S. Census Bureau, 2016).

The population in the SARB has increased nearly 70% in the last 30 years due primarily to the economic growth in Bexar County, in which San Antonio is located. It is expected that the population will reach about 2.8 million by 2060, which would represent a 94% increase since 2000 (Texas Water Development Board, TWDB, 2011). Compared to a 1.63% annual population growth rate in Bexar County during the 10-year period leading up to the inception of NAFTA, the growth rate between 1994 and 2010 increased to approximately 1.90% per annum (Texas State Library and Archives Commission, TSLAC, 2015). Land change in this region has been associated to a large degree with the development of public transportation network and the NAFTA corridor including Interstate Highway (IH) 10, IH 35, IH 37, US Highway 281, and State Highway loop 1604. Among these highways, IH 35 represents the major freight road connecting San Antonio to Laredo and other southern border areas (Texas Department of Transportation, TxDOT, 2013).

Kreuter et al. (2001) investigated the impact on ESVs of urban expansion between 1976 and 1991 in Bexar County by combining landchange analysis with ecosystem services value coefficients provided by Costanza et al. (1997). They identified a 65% decrease in rangeland, 29% growth in urban areas and $6.24 million loss in ecosystem services within the county over the 15-year study period. In another study, American Forests (2002) estimated changes in forests and associated ESVs in the San Antonio region between 1985 and 2001. This study identified a 39% decrease in the woodlands with more than half canopy cover, which negatively affected storm water management and air quality, and boosted energy consumption. Beyond these two studies in Bexar County, no studies have been conducted in the SARB to evaluate the effects of population and economic growth on land and associated ecosystem services. This represents a critical knowledge gap for evaluating economic growth of the region in a larger context that incorporates potential effects on the provision of ecosystem services.

This study focuses on the SARB and Bexar County because of their central location in the corridor that has been the most affected by the implementation of NAFTA, with the City of San Antonio being a key trade center for this multinational agreement. In our study, we specifi- cally examined the effect of land change on the ESVs in the SARB between 1984 and 2010. We repeated this analysis on the three watersheds that cover most of Bexar County, which was the focus of the previous two studies. We selected "watershed" as the unit of analysis because it is fundamental to the provision of key ecosystem services including water purification, ground water and surface flow regulation, and erosion control (Brauman et al., 2007). We conducted our study to address two questions: (1) How did the land change dynamics unfold in the San Antonio River Basin (SARB) and Bexar County from 1984 to 2010? (2) How did the associated ESVs change in response to the land change in the SARB and Bexar County before and after the implementation of NAFTA?

2.2. Methods

2.2.1. Study Area

The SARB is one of the major basins located in South Texas, draining over 14,162 km of streams and covering 10,862 km^2 within 14 counties. It contains almost all of Bexar County (Figure 2.1). The city of San Antonio is centered in Bexar County and lies about 140 miles northwest of the Gulf of Mexico and 150 miles northeast of Laredo on the Mexican border (Figure 2.1; San Antonio Chamber of Commerce, 2015). The three watersheds, the Leon Creek Watershed, the Upper San Antonio River Watershed, and the Salado Creek Watershed, comprise Bexar County and cover 1579 km^2. The SARB transects five of the 10 ecoregions of Texas including Edwards Plateau, Texas Blackland Prairie, Post Oak Savannah, South Texas Plains, and Gulf Coast Prairies and Marshes (Gould et al., 1960). In addition, the SARB intersects with the Edwards Aquifer drainage and recharge zones (San Antonio River Authority, SARA, 2015). The climate in the SARB ranges from semi-arid in the upper northwestern part to subtropical in the lower southeastern part near the Gulf Coast.

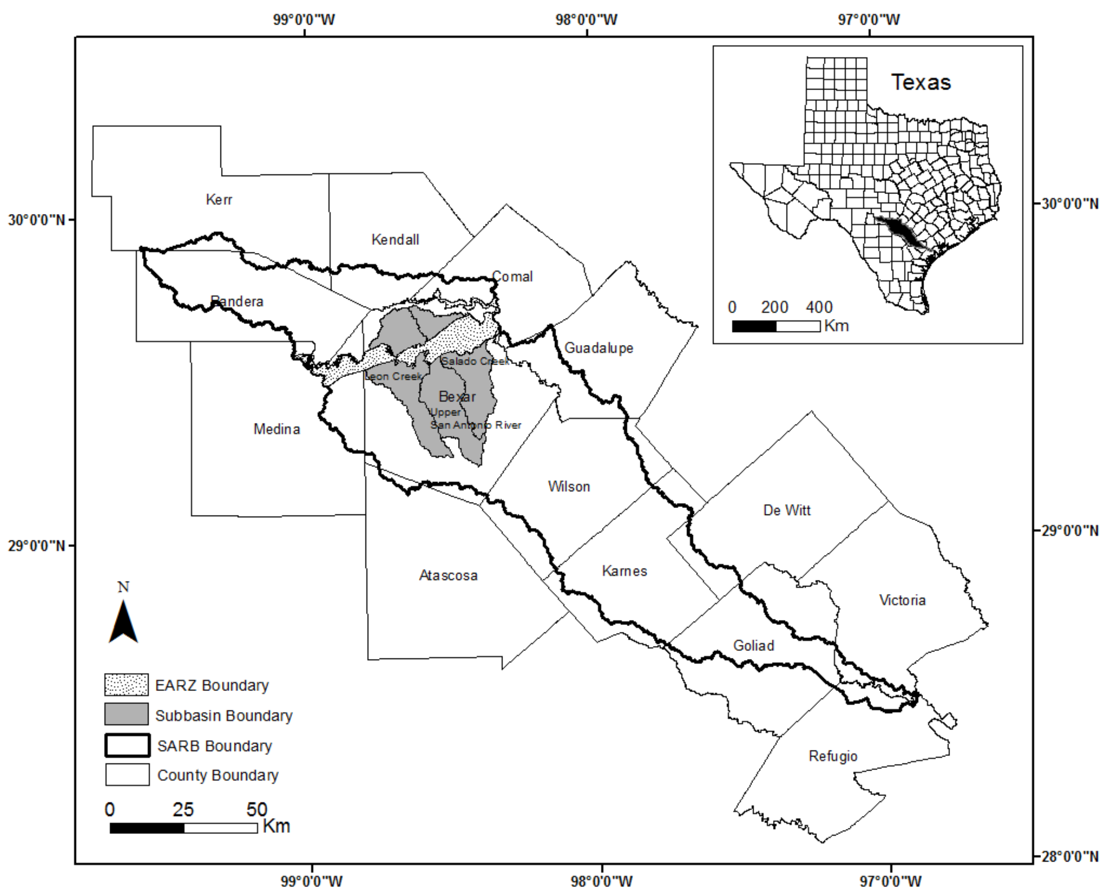


Figure 2.1. San Antonio River Basin (SARB) and three watersheds containing Bexar County.

2.2.2. Land-Change Analysis

We analyzed the land change using cloud-free, multitemporal Landsat 5 TM image data (30-meter spatial resolution, bands 1–5 and 7) acquired in November 1984, December 1995, and December 2010 (http:// earthexplorer.usgs.gov) (U.S. Geologic Survey (USGS) (2014)). We selected these image dates at time intervals that allow for pre- and postNAFTA analysis, and based on the availability of images from a consistent Landsat TM sensor, atmospheric conditions, and seasonal conditions under which land classes were expressed in a readily-interpretable manner. Multiscene data for each year consisted of four images (paths and rows 26/40, 27/39, 27/40, and 28/39 of the Worldwide Reference System (WRS)-2, respectively). We constructed mosaics and spatially subset the multiple images for each year to encompass the boundary of the SARB, based on geographic information system (GIS) boundary files (https:// tnris.org) (Texas Natural Resources Information System (TNRIS), 2015).

Table 2.1. Description of land classes utilized in this study. Urban class definitions are modified from the NLCD 2006 classification system (U.S. Geological Survey (USGS), 2015), and the other classes are from or modified from Anderson et al. (1976).

Land class	Class description
Urban	Areas with a mixture of constructed materials and vegetation
Low-density urban	Areas with less than 50% impervious surfaces per pixel
High-density urban	Areas with 50% or more impervious surfaces per pixel
Agricultural Land	Areas used for the production of cultivated crops
Pasture	Areas of grasses, grass-legume mixtures for grazing or the production of hay crops
Rangeland	Areas dominated by herbaceous vegetation
Forest Land	Deciduous forest, evergreen forest, and mixed forest
Water	Areas of open water, lakes, and rivers
Wetland	Soil or substrate periodically saturated with or covered with water
Barren Land	Bedrock, desert pavement, sand dunes, and other accumulations of earthen material

After conducting atmospheric and radiometric corrections (Appendix A.1), we classified the images using unsupervised Iterative Self-Organizing Data Analysis (ISODATA) (Jensen, 2005). For each image, we conducted a maximum of 100 iterations to generate no more than 50 spectral clusters. Using reference aerial photography (discussed below) and National Land Cover Database (NLCD) data (http://www. mrlc.gov) (U.S. Geological Survey, USGS, 2015) that were available near the time of Landsat image acquisition, we then merged these clusters into nine land classes: low-density urban, high-density urban, (cultivated) agricultural land, pasture, rangeland, forest land, water, wetland, and barren land (Table 2.1). Except for the pasture and urban classes, the seven other land classes generally correspond to the USGS land classification system (Anderson et al., 1976). We differentiated pasture from agricultural land (i.e., cultivated agriculture). Urban areas can generally be defined by the percentage of impervious surfaces (Schueler, 1994; Arnold and Gibbons, 1996).We differentiated between low- and high-density urban areas as follows: low-density urban consists of areas with less than 50% impervious surfaces, whereas high-density urban is comprised of areas with 50% or more impervious surfaces. We generalized these urban classes from the “developed” class definition of the NLCD 2006 classification system (http://www.mrlc. gov) (U.S. Geological Survey, USGS, 2015).

We assessed the accuracy of the Landsat-derived land classifications based on visual interpretation of aerial photographs (http:// earthexplorer.usgs.gov) (U.S. Geologic Survey, USGS, 2014) and temporally-proximal NLCD data (http://www.mrlc.gov) (U.S. Geological Survey, USGS, 2015), when available

(Appendix A.1). We used stratified random sampling (Congalton and Green, 1999) to select 50 accuracyassessment points for each of the nine land classes (a total of 450 points) in each classified image. We conduct the classification accuracy assessment based on confusion matrices (Congalton and Green, 1999; Jensen, 2005), where overall classification accuracies are 85.11%, 87.33%, and 85.78% for the 1984, 1995, and 2010 images, respectively.

2.2.3. Ecosystem Service Value (ESV) Estimation

For the valuation of ecosystem services, we used the benefit transfer method (BTM), a widely used approach for valuing ecosystem services. BTM extrapolates the value estimates from one or more study sites to other areas that are assumed to be ecologically and socio-economically similar (Brouwer, 2000; Woodward and Wui, 2001; Plummer, 2009; Daly and Farley, 2010; Koschke et al., 2012; Foody, 2015). In their seminal study, Costanza et al. (1997) used values from other studies and applied BTM to develop a set of unit values for several ecosystem services and estimated the global value of ecosystem services. A subsequent assessment updated unit ESVs based on a larger database of case studies (Costanza et al., 2014). The value coefficients derived in this later study were based primarily on those reported by de Groot et al. (2012), the most comprehensive set of aggregate values for 22 ecosystem services based on 665 value estimates collected from over 300 case studies around the world. Specifically, the 2014 value coefficients used for the representative land classes in our study were aggregates of estimates from numerous case studies as shown here in parentheses:

Wetland (139); Water (36); Forest (109); Rangeland (36); Pasture (36); Agriculture (33); Barren (3); Low and High Density Urban (1).

Costanza et al. (2014) claimed that the underlying data and models they used for their assessment could be applied at multiple scales to assess changes in several ecosystem services. We used BTM based on value coefficients published by Costanza et al. (1997) (hereafter 1997 coefficients) and by Costanza et al. (2014) (hereafter 2014 modified coefficients) to estimate the changes in ESVs in the SARB and Bexar County between 1984 and 2010. We adjusted these coefficients to 2010 U.S. dollar values using Consumer Price Index Inflation Calculator from the U.S. Bureau of Labor Statistics, 2016 (http://data.bls.gov) for the land classes in our study (Table 2.2).

Table 2.2. Land classes used in this study, equivalent to biomes presented by Costanza et al. (1997, 2014), and three sets of value coefficients for each of the land classes.

Land class	Equivalent biome	1997 coefficents (2010 US$/ha/yr)	2014 Modified coefficients (2010 US$/ha/yr)	Percent difference from 1997 coefficients	b 1997-2014 Mean coefficients (2010 US$/ha/yr)	Percent difference from 2014 modified coefficients
a Low density urban	Urban	0	5,254	-	2,627	-50.0%
a High density urban	Urban	0	1,751	-	876	-50.0%
Agricultural Land	Cropland	132	5,854	4334.8%	2,993	-48.9%
Pasture	Grass/rangeland	337	4,381	1200.0%	2,359	-46.1%
Rangeland	Grass/rangeland	337	4,381	1200.0%	2,359	-46.1%
Forest Land	Temperate/boreal	438	3,299	653.2%	1,869	-43.4%
Water	Lakes/rivers	12,332	13,158	6.7%	12,745	-3.1%
Wetland	Flood plains	28,417	27,008	-5.0%	27,713	2.6%
Barren Land	Desert	0	0	0.0%	0	0.0%

a Coefficients derived from urban coefficient in Costanza et al. (2014).
b Coefficients for sensitivity analysis.

We modified the 2014 urban land coefficient based on low- and high-density urban development (<50% and ≥50% impervious cover, respectively). In order to better capture this dichotomy of urban class, and based on an inspection of orthophotos used for our accuracy assessment, we assumed an average of 75% and 25% green space for low and high-density urban areas, respectively. We assigned green space value of \$6111/ha/year (Brenner et al., 2010), adjusted to 2010 US\$ values, to the green space fraction of each urban class and \$0 to the rest of each urban class. This produces urban ecosystem value coefficients of \$5254 and \$1751/ha/year for low- and high-density urban areas, respectively.

The value coefficients in Costanza et al. (1997) and Costanza et al. (2014) differ substantially. These differences are, in part, due to a larger number of local cases being used in 2014 to obtain values for each biome. Increases in coefficient values between the two studies are especially large in biomes with relatively low values in 1997, including agricultural land (~4300%), grassland and rangelands (~1200%), and forests (~650%). Most notable, however, is the increase in ecosystem service value of urban areas from \$0 in 1997 to \$7005/ha/year in 2014. We obtained the total ESV in the SARB and Bexar County for 1984, 1995, and 2010:

$$ESV = \Sigma\,(A_k \times VC_k) \tag{1}$$

$$ESV_f = \Sigma\,(A_k \times VC_{fk}) \tag{2}$$

where ESV is the estimated ecosystem service value in a given year, A_k is the area (ha), and VC_k is the value coefficient (\$/ha/year) for the considered land class 'k' (Kreuter et al., 2001). ESV_f is the estimated ecosystem service value of function f in the study area and VC_{fk} is the value coefficient of function f ($/ha/year) for the considered land class 'k' (Zhao et al., 2004). We calculated the temporal changes in ESV from the differences between estimated values for each land class in 1984, 1995, and 2010.

To address uncertainties in the unit value of each land class, we examined the sensitivity of our total ESV estimations based on the 2014 modified coefficients by applying the mean value of the 1997 and 2014 modified coefficients (Table 2.2). Additionally, we used the lower unit value ($1836/ha/year in 2010 US$) for urban green space reported by Brander and Koetse (2011). Using this latter unit results in urban ecosystem value coefficients of $1377 and $459/ha/year for low- and high-density urban areas, respectively, (since we assume, on average, 75% and 25% of urban land cover is green space for low- and high-density urban areas, respectively). These values are 73.8% less than the 2014 modified coefficients. We calculated the coefficient of sensitivity (CS) as follows:

$$CS = \frac{(ESV_j - ESV_i)/ESV_i}{(VC_{jk} - VC_{ik})/VC_{ik}} \quad (3)$$

where ESV is the estimated ecosystem service value, VC is the value coefficient, 'i' and 'j' represent the 2014 and 1997-2014 mean coefficient values, respectively, and 'k' represents the land class (Kreuter et al., 2001).

2.3. Results

2.3.1. Land Change in the SARB and Bexar County

Our land classification indicates substantial urban growth between 1984 and 2010 in the SARB, particularly around San Antonio (Figure 2.2). The proportion of the SARB that is urban increased steadily during our study period from 4.3% in 1984 to 7.0% in 1995 and then to 13.3% in 2010 (Table 2.3, Table A1). This corresponds to a total increase of 97,327 ha from 1984 to 2010. Overall, the annual growth rate of urban areas (~6%) remained consistent during the two periods. However, during 1995–2010, the annual growth rate of low-density urban areas was 1.9% higher and that of high-density urban areas was 3.3% lower than during 1984–1995.

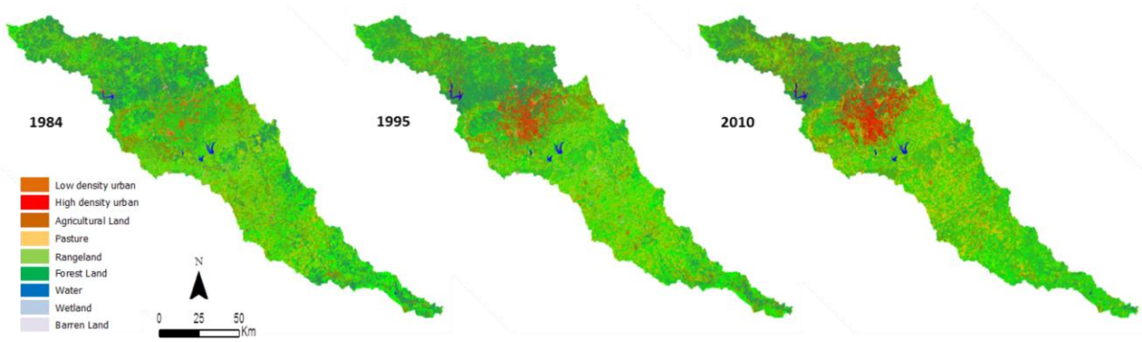


Figure 2.2. Land change between 1984 and 2010 in the SARB.

Rangeland and forest, which are the two largest land classes, declined markedly during our 26-year study period (Table 2.3). The percent cover of rangelands decreased from 37.9% in 1984, to 36.1% in 1995, and 35.8% in 2010, resulting in a total loss of 22,075 ha over the entire study period. The forest land decreased from 29.9% to 27.7%

and then to 23.2% during the study period, with a total loss of 73,146 ha. There was also a decline in agricultural (cultivated) land from 10.3% to 9.7% and then to 8.5%, resulting in a total loss of 19,224 ha.

The annual decrease in the percent cover of these three land classes was 0.3% greater during 1995–2010 than 1984–1995 (Table A1). Contrasting with these declines is the increase in pasture from 16.0% in 1984 to 17.8% in 1995 and 18.4% in 2010, with a total gain of 25,443 ha during the study period. This pattern is consistent with analyses indicating the increasing trend in both hay production and prices in Texas (Acheampong et al., 2010). In combination, the other land classes, including wetland, water, and barren areas, comprise <2% of the SARB in all three years of analysis.

Table 2.3. Total estimated area (ha) and area percentage of each land class in the SARB from 1984 to 2010.

	Total area (ha, %)					
Land class	1984	%	1995	%	2010	%
Urban	46,602	4.3	76,095	7.0	143,929	13.3
Low density urban	31,327	2.9	45,312	4.2	85,764	7.9
High density urban	15,275	1.4	30,783	2.8	58,165	5.4
Agricultural Land	111,835	10.3	104,841	9.7	92,611	8.5
Pasture	173,895	16.0	193,128	17.8	199,338	18.4
Rangeland	411,210	37.9	392,479	36.1	389,135	35.8
Forest Land	324,391	29.9	300,864	27.7	251,245	23.2
Water	3,672	0.3	4,267	0.4	4,015	0.4
Wetland	960	0.1	618	0.1	570	0.1
Barren Land	12,379	1.1	12,109	1.1	3,345	0.3
No Data	807	0.1	1,350	0.1	1,563	0.1
Total	1,085,751	100	1,085,751	100	1,085,751	100

In the three watersheds in Bexar County (Figure 2.3), the forest land represented the largest land class in 1984 covering 37.5%; it increased to 39.7% in 1995 but then decreased to 26.6% in 2010, resulting in a total loss of 17,163 ha (Table 2.4, Table A1). Rangelands in Bexar County decreased substantially from 36.4% in 1984 to 22.1% in 1995 and then remained relatively unchanged by 2010 with a total loss of 22,804 ha. By contrast, the area of the two urban land classes more than tripled during the 26-year study period growing from 12.6% in 1984 to 25.1% in 1995 and 38.4% in 2010, by which time urban land represented the largest land class in Bexar County covering 60,663 ha. These increases represent annual growth rates of 9% and 3.5% during the 1984–1995 and 1995–2010 periods, respectively, and growth rates declined between the first and second time period for both high-density and low density urban land (high-density growth decreased from 21.5% to 6.5% per annum and low-density decreased from 4.7% to 1.3% per annum). The combined area of the other land classes (pasture, agriculture, barren land, wetlands and water bodies) was about 13% of the three rapidly urbanizing watersheds and remained relatively unchanged during the study period.

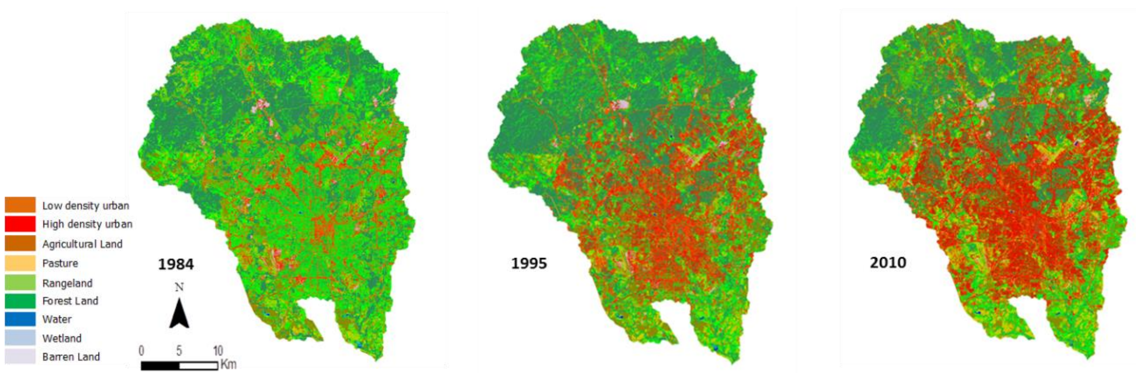


Figure 2.3. Land change between 1984 and 2010 in the three watersheds, Bexar County.

Table 2.4. Total estimated area (ha) and area percentage of each land class in the three watersheds, Bexar County from 1984 to 2010.

	Total area (ha, %)					
Land class	1984	%	1995	%	2010	%
Urban	19,894	12.6	39,666	25.1	60,663	38.4
Low density urban	14,767	9.4	22,439	14.2	26,698	16.9
High density urban	5,127	3.2	17,227	10.9	33,965	21.5
Agricultural Land	8,752	5.5	9,418	6.0	3,756	2.4
Pasture	10,245	6.5	8,919	5.6	14,996	9.5
Rangeland	57,496	36.4	34,858	22.1	34,692	22.0
Forest Land	59,175	37.5	62,640	39.7	42,012	26.6
Water	82	0.1	133	0.1	77	0.1
Wetland	117	0.1	125	0.1	56	0.0
Barren Land	2,111	1.3	2,115	1.3	1,622	1.0
No Data	2	0.0	0	0.0	0	0.0
Total	157,874	100	157,874	100	157,874	100

Land change matrix for the SARB reveals that more forest land and rangeland was lost to low-density urban areas than to high-density urban areas (Table A2). Of all forest lands, respectively, 24,015 ha and 10,810 ha were converted to low-density and high-density urban land, whereas of all rangelands 50,907 ha turned into low-density and 15,673 ha into high-density urban land. These patterns in land change emphasize that the low-density urban land was growing at a more rapid rate than high-density urban land in the SARB. In Bexar County, losses of forest cover and rangelands to the high-density and low-density urban areas were more even: respectively, 8937 ha and 6750 ha of forest land were lost to low-density and high-density urban land, whereas rangelands lost 16,434 ha to low-density and 10,393 ha to high-density urban land development.

2.3.2. Changes in ESVs

Temporal changes in estimated ESVs for each land class in the SARB mirrored the changes in the area of each class but varied substantially according to the value coefficients applied (Figure 2.4). When the 1997 coef- ficients were used, the total ESV per annum in the SARB decreased from $426 million in 1984 to $413 million in 1995 (3.1% decrease) and then to $386 million in 2010 (6.5% decrease) (Table A3). By contrast, when the 2014 modified coefficients were applied, estimated overall annual ESV in the SARB was an order of magnitude higher, decreasing from $4553 million in 1984 to $4536 million in 1995 (0.4% decrease) but then increasing to $4569 million in 2010 (0.7% increase). These differences in the rate and direction of change can be explained by the proportionately greater increase in the ESV of urban areas in the second evaluation period (1995–2010) when the 2014 modified coefficients were used (Figure 2.4).

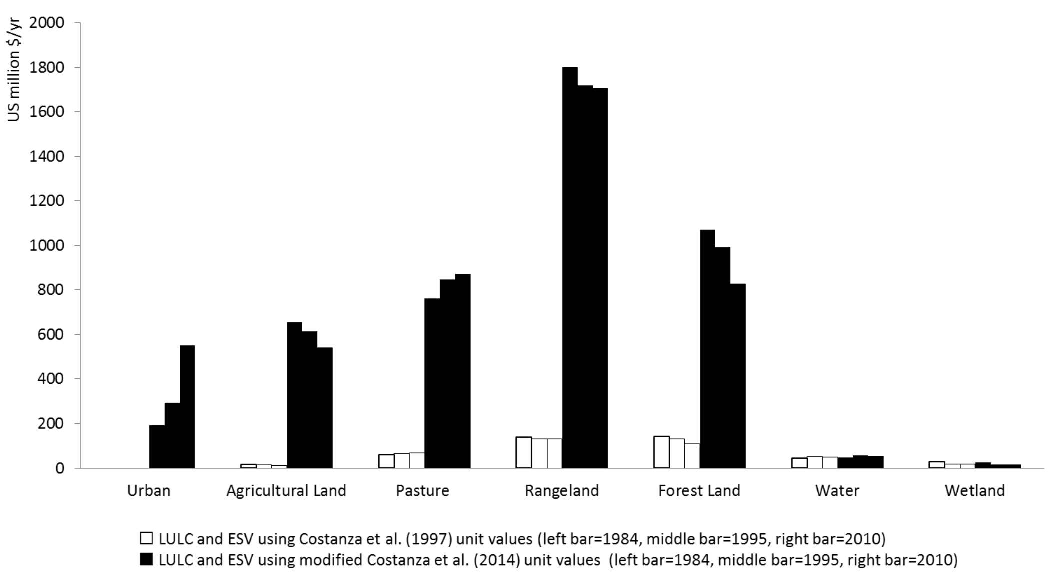


Figure 2.4. Changes in the ESV by land class between 1984 and 2010 in the SARB.

As with the SARB, temporal changes in estimated ESVs for each land class in Bexar County mirrored the changes in the area of the land classes (Figure 2.5). Estimates of total ESV in the three watersheds based on 1997 coefficients decreased from $54.23 million in 1984 to $48.62 million in 1995 (10.4% decrease) and to $38.18 million in 2010 (21.5% decrease) (Table A3). As with the SARB analysis, when 2014 modified coefficients were used, total annual ESV estimates are an order of magnitude higher and decreased at a slower rate during the 26-year period. In this case, the estimated total annual ESV in Bexar County decreased from $634 million to $606 million between 1984 and 1995 (4.3% decrease) and, contrary to the basin-wide analysis, continued to decrease to $580 million in 2010 (4.3% decrease).

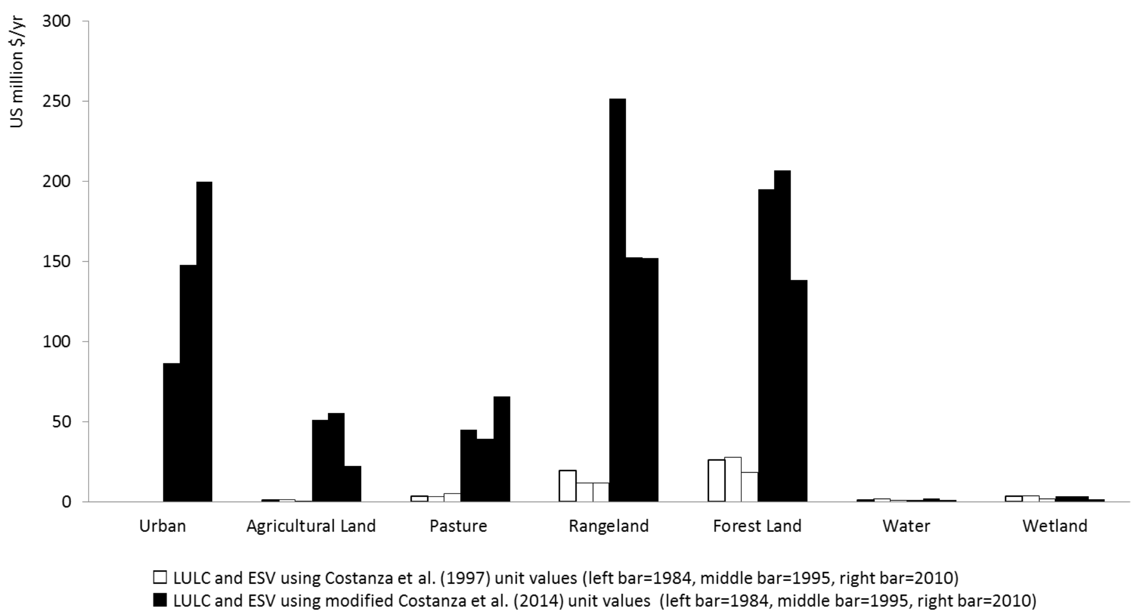


Figure 2.5. Changes in the ESV by land class between 1984 and 2010 in the three watersheds, Bexar County.

The results from the sensitivity analysis indicate that estimated ESVs for both scales of analysis are relatively inelastic (i.e., CS substantially b1) (Table A4). Adjusting value coefficients (VC) for wetland and water had little impact on the estimated ESV, primarily because these land classes covered negligible proportions of the total land area and changes in their value coefficients between 1997 and 2010 are small (b7%). The CS for rural land classes in the SARB is highest for rangelands (SARB = 0.40 to 0.37; Bexar = 0.40 to 0.26), followed by forest lands (SARB = 0.23 to 0.18; Bexar = 0.31 to 0.23), pasture (SARB = 0.17 to 0.19; Bexar = 0.07 to 0.11) and agricultural lands (SARB = 0.14 to 0.12; Bexar = 0.08 to 0.04).

When the value coefficients for low- and high-density urban space were reduced by 73.8% (based on the lower unit value reported in Brander and Koetse (2011)) and 50% (rows 1–2 in Table 2.2), the corresponding CSs were relatively small at both scales of analysis (low-density: SARB = 0.04 to 0.10; Bexar = 0.12 to 0.24; high density: SARB = 0.01 to 0.02; Bexar = 0.01 to 0.10). Additionally, although the CSs for urban land did increase over the 26-year study period (Table A4), they were generally lower than for the other land classes, and all CS values were ≤0.40. Based on these sensitivity analyses, the ESV estimates for all three years of analysis (1984, 1995 and 2010) appear to be relatively robust.

2.3.3. Changes in ESV Functions

We also quantified and compared the contributions of each ecosystem function to the overall ESV in the SARB and Bexar County (Figures 2.6–2.7, Table A5). At both spatial scales, the value of individual ecosystem services was higher when the 2014 modified value coefficients rather than the 1997 coefficients were applied, but the difference varies substantially among ecosystem services.

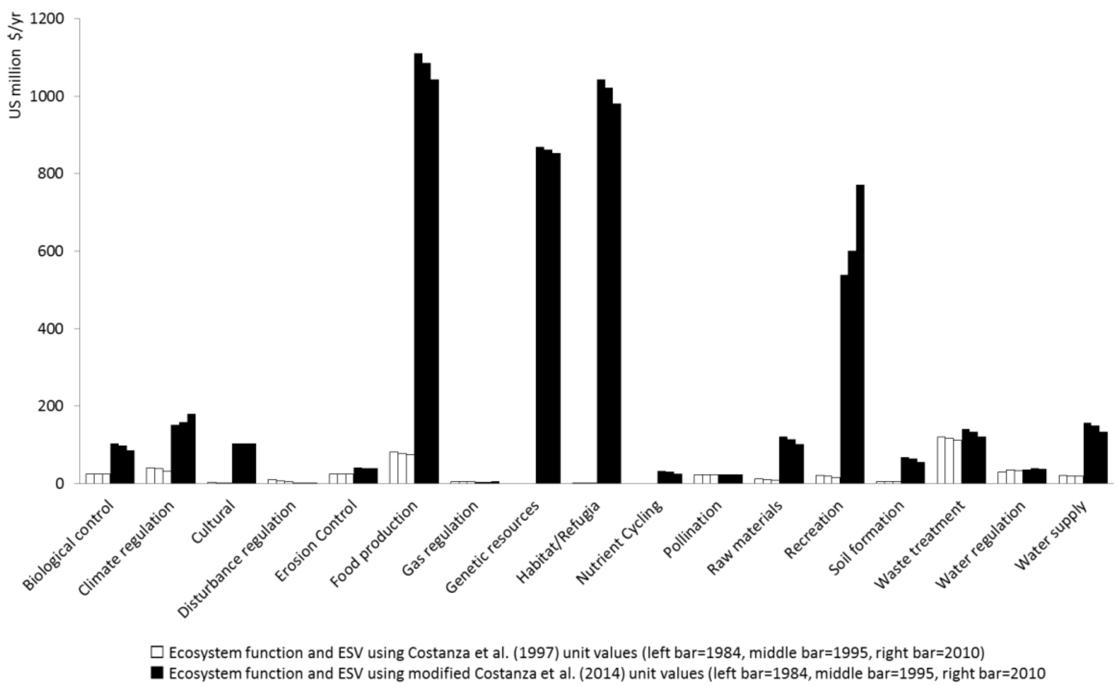


Figure 2.6. Changes in ESV by functions between 1984 and 2010 in the SARB.

Genetic resources and habitat/refugia were assigned minimal value in 1997; however, using the 2014 value coefficients, these two services each contribute 11–23% of the total ESV in both the SARB and Bexar County. The other two ecosystem services that contributed more than 10% to overall ESV at both spatial scales are food production (15% and 25% in the SARB and Bexar County) and recreation (12% and 38% in the SARB and Bexar County). By contrast, while value of waste treatment services changed little at either scale when 1997 and 2014 coefficients were used, their contribution to total ESV dropped from 28% and 31% in the SARB and Bexar County, respectively, to around 3% in both when the 2014 coefficients were applied due primarily to the large increase in value coefficients of other ecosystem services. Similarly, gas and disturbance regulation services, which contributed 1.0% and 2.7%, respectively, when 1997 coefficients were used, dropped to almost zero when the 2014 coefficients were applied.

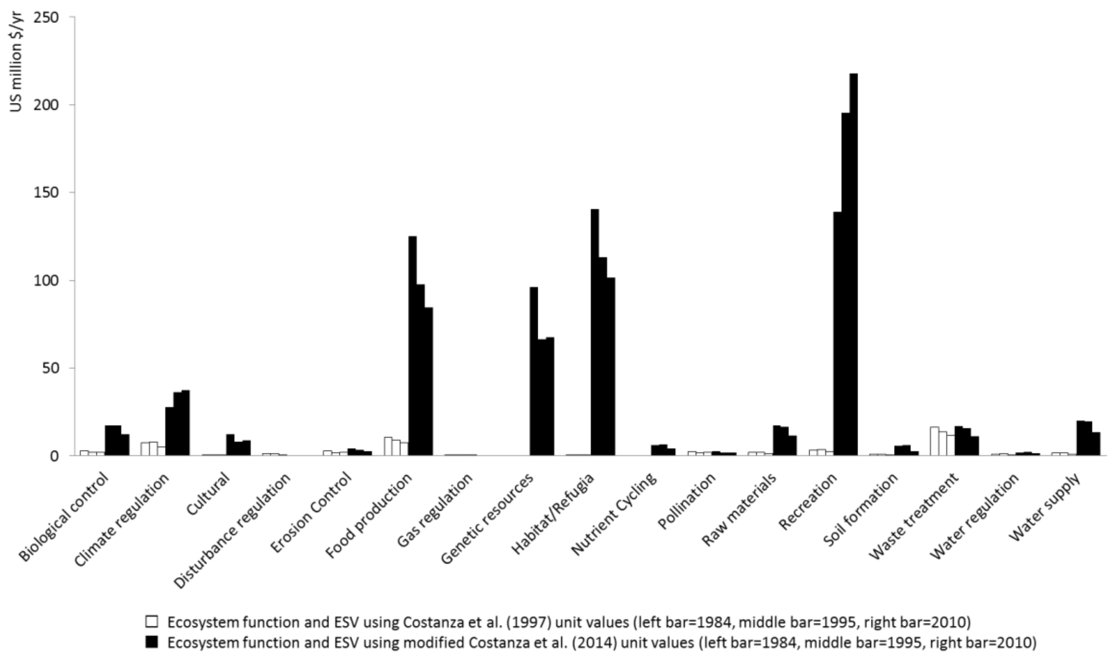


Figure 2.7. Changes in ESV by functions between 1984 and 2010 in the three watersheds, Bexar County.

Regardless of the value coefficients used, the patterns of temporal change in the values of ecosystem functions declined during the 26-year study period, with two exceptions, recreation and climate regulation (Figures 2.6–2.7). Both recreation and climate regulation services decreased or stayed approximately constant in value and in percent contribution to overall ESV throughout the 26-year period when the 1997 coefficients were used but they increased in value and percent contribution when the 2014 coefficients were applied. Notably, in Bexar County, recreation accounted for 38% of the overall ESV in 2010.When the 2014 coefficients were used, the increases in the values of recreation and climate regulation and the decreases in the values of other ecosystem services were greater during the 1995–2010 period than the 1984–1995 period.

2.4. Discussion

In both the SARB and Bexar County, urbanization has been characterized over the last 30 years by rapid socio-economic and land changes caused by the increasing population and economic development. On the other hand, the two dominant land classes, native rangelands and woodlands/forests that provide a diverse set of ecosystem services, have decreased significantly. In the SARB, the loss of rangelands has been largely due to the urban expansion whereas, in the case of the forests, conversions both to urban and to rangelands have been significant (Table A2). Across the three watersheds in Bexar County, the increase in the forest cover between 1984 and 1995 is likely due to the pervasive expansion of junipers (*Juniperus ashei*) in the northern part of the watersheds due to long-term fire suppression polices.

This increase was, however, followed by a decrease in woody plant cover by 2010 primarily due to the urban expansion in the watersheds (Table A2), and possibly also a significant die back of woody plants as a result of one of the driest seasons on record in 2008 (Twidwell et al., 2014). Overall, these results are consistent with the findings that the San Antonio area in Bexar County experienced one of the greatest losses in forest in the southern USA between 2001 and 2006 (World Resources Institute, WRI, 2011). The decrease in the rangeland and woodland/forest cover indicates a substantial loss of the ecosystem services, including a decrease in surface water infiltration, wildlife habitat and biodiversity, microclimate regulation, and carbon sequestration provided by native vegetation in the SARB.

Comparing the pre-NAFTA period (1984–1995) and the post-NAFTA period (1995–2010), our analysis shows that while overall rate of urban expansion in the SARB remained fairly consistent, the rate of expansion of low-density urban accelerated after NAFTA went into effect (Table A1). Notably, the expansion of low-density urban has been concentrated around the San Antonio area reflecting the sprawling nature of urban development in the region (Figure 2.2). This expansion of low-density urban growth is creating more widespread impact on the delivery of ecosystem services, especially those provided by rangelands and woodlands/forests. These findings are consistent with Alig et al. (2004) who reported that land change affecting forests since 1990 have been mainly centered in southern US posing significant threats to ecosystems.

The reduction in water infiltration services due to increasing impervious space in urbanizing areas within the SARB has particular significance for the Edwards Aquifer that provides 90% of San Antonio's water needs (San Antonio Water Systems, SAWS, 2016). This is because the recharge zone of this karst aquifer runs from west to east through northern Bexar County and northeastern Medina County (Figure 2.1). The minimal filtration capacity of karst aquifers results in the quality of their water being determined by the quality of water entering the recharge zone. Thus, the conversion of perennial plant cover with high filtration capacity, provided by rangelands and forests, to impervious surfaces in the recharge zone detrimentally affects the quality of water used by the residents of the San Antonio metropolitan area.

A key consideration in terms of future land change and resulting impacts on the ecosystem services in the SARB is the development of transportation infrastructure, which represents large portions of impervious urban surfaces, especially near Interstate Highway (IH) corridors (Nowak et al., 2005; Alig et al., 2010). San Antonio's future growth will be especially affected by the continued development of the so-called NAFTA corridor (Texas Department of Transportation, TxDOT, 2014). This includes construction and expansion of numerous highways in the region including IH-35, the major freight road connecting San Antonio to the Mexican border, and other Interstate Highways and railroads are expected to converge in San Antonio region in 2030 to connect Texas NAFTA gateways (Texas Department of Transportation, TxDOT, 2013). These expanding transportation networks will likely further degrade the ecosystem services in the region through land change, air pollutant emissions and water contamination (American Forests, 2002). Additionally, as low-density urban development radiates outwards from the urban centers (e.g., San Antonio-New Braunfels Metropolitan Statistical Area in the northern segment of the SARB), the demand for more road infrastructure from these automobile dependent communities will also increase in other parts of the SARB (Filion et al., 1999).

Our study determined much higher ESVs using the 2014 modified coefficients (Costanza et al., 2014) than those using the 1997 coefficients (Costanza et al., 1997) at both spatial scales of analysis, the SARB and the three watersheds in Bexar County. Temporal patterns of change also differed when we applied these two sets of value coefficients. This is primarily because the 1997 coefficients assumed zero ecosystem service value in urban areas whereas the 2014 modified coefficients included a high value to the urban green space ($7005/ha/year in 2010 US$). The zero value assigned to urban space in Costanza et al. (1997), failed to recognize ecosystem services provided by urban green spaces, such as carbon sequestration, air filtration, or recreation opportunities (Bolund and Hunhammar, 1999; Kreuter et al., 2001). However, the ecosystem service value assigned to urban areas by Costanza et al. (2014) seems equally unrealistic. The value of urban green space in that study was derived from a single study (Brenner et al., 2010). Moreover, Costanza et al. (2014) extrapolated this greenspace value to all urban space regardless of the various uses of land characterizing urban landscapes.

Another study used the opportunity cost of not developing Central Park in New York to estimate the value of the "myriad ecosystem services to New Your City" of the 341-hectare green space (Sutton and Anderson, 2016, p. 87). In this way they determined that Central Park provided over $70million/ha/year in ecosystem services. As the authors point out, "the very high value of the ecosystem services provided by Central Park result from an inter-action of social, natural, human, and built capital". However, in general, it seems unreasonable that green space in highly developed areas is more valuable in terms of ecosystem services delivery than less fragmented and less developed areas. For example, based on a meta-analysis of 20 studies using contingent valuation to estimate the value of urban green space, Brander and Koetse (2011, p 2767) estimated “the value of open space with ‘average’ characteristics” to be approximately $1550/ha/year ($1836/ha/year in 2010 US$).

We attempted to partially address the apparent overestimate of the value of urban space in Costanza et al. (2014) by, at least, modifying the proportion of land in low- and high-density urban space that was assigned this high value (75% and 25% in low- and high-density urban space, respectively). However, based on Brander and Koetse (2011), the value assigned in this way to these two urban classes may still be high. We addressed this concern in the sensitivity analysis by applying the Branner and Koetse “average” value to urban green space. We found the coefficient of sensitivity for the ESVs were quite low when these adjustments were made (0.01 to 0.24). This provides a reasonable level of confidence that our ESV estimates for the SARB and Bexar County were not overly distorted by the value coefficients we used for the urban classes.

Based on the 1997 coefficients, our assessment of changes in overall ESV from 1984 to 2010 revealed the same overall negative effect of urbanization on the value of ecosystem services in the SARB and Bexar County as an earlier study in Bexar County (Kreuter et al., 2001) and other case studies (Liu et al., 2012; Su et al., 2012, 2014; Estoque and Murayama, 2013; Wu et al., 2013). However, the use of 2014 modified coefficients resulted in a proportionately slower decline in the ESVs than previous studies during the pre-NAFTA period (1984–1995) at both scales of analysis, as well as a reduction in ESV decline in Bexar County and a slight ESV increase in the SARB during the post-NAFTA period (1995–2010). This suggests that the increase in ecosystem services due to urban expansion more than offset the decrease in ecosystem services due to the loss of forests and rangelands within this period.

A closer look reveals that the increase in value of ecosystem services in the SARB during the post-NAFTA period is due to the high values assigned to recreation and climate regulation services in urban areas (Costanza et al., 2014). These high values mask the loss of other essential ecosystem services provided by natural vegetation classes, including sediment retention, water filtration, and waste assimilation. Clearly, this is problematic, because regulatory services provided by properly functioning ecosystems (e.g., carbon sequestration, water filtration, and provision of wildlife habitat) cannot simply be substituted by cultural services, such as recreation. Our comparative study suggests that the value assigned by Costanza et al. (2014) to ecosystem services provided by urban land, particularly recreation, is a substantial overestimate, especially, compared to those values assigned to other ESs. When applied at the regional scale (SARB) or the local scale (Bexar County), this results in an underestimate of the degradation of ecosystems resulting from the urban expansion.

Our findings illuminate issues associated with scaling up and scale dependence of the validity of value coefficients when BTM analyses are conducted to evaluate ESVs. This underscores the importance of ensuring that the transferred unit value derived from the primary evaluation study is compatible with the site to which it is applied, with respect to both the scale and characteristics of the reference and study sites, in order to avoid misinterpretation of land change effects on the value of ecosystem services delivered. The effect of inaccurate estimation of per unit ESVs due to urban expansion is likely negligible at the global analyses of Costanza et al. (1997) and Costanza et al. (2014) because urban lands constitute a very small percentage of global land area. In contrast, urban land covers significantly larger proportions of our study area at basin scale and especially at the smaller county scale of analysis. The ability to confidently use such value coefficients as proxies for ESVs demands rigorous assessment of their broad applicability. This is especially critical for studies intended to identify changes in ESVs resulting from the implementation of development instruments, such as NAFTA.

Given the ongoing economic growth pressures of NAFTA, it is expected that continued demand for land conversion to meet the needs of a rapidly growing human population will significantly impact ecosystems within the SARB as well as outside of the basin along the NAFTA corridor. It is thus imperative to implement proper land-use policies to safeguard forests and rangelands from urban land expansion. From an international perspective, NAFTA provisions for environmental protection should be reinforced through multi-scale cooperative environmental impact assessments in Mexico, the US and Canada. At the national and regional scale, smart growth supported by the U.S. Environmental Protection Agency (USEPA) could help balance economic development and conservation (Smart Growth Network, 2006). At the regional scale, forests and rangelands are especially vulnerable to rapid urbanization within the SARB. The payments for establishing and maintaining conservation easements and implementing best management practices for ensuring watershed health motivate landowners to maintain intact properties that provide open space, support biodiversity and facilitate effective ecosystem functions. At the local scale, adverse impacts of urbanization can be minimized and ecosystem services and biodiversity can be safeguarded through Low Impact Development, which is a functional landscape strategy to mimic the pre-development hydrologic regime through conservation and use of natural features of the landscape (U.S. Environmental Protection Agency, USEPA, 2000).

Numerous indirect valuation methods have been developed for public goods that are subject to market externalities, such as in situ ecosystem services (Farber et al., 2002; Costanza, 2008). BTM has been criticized because unit values derived from one area are applied as average unit values in all areas and do not necessarily reflect the marginal value of the same public good in other areas (Toman, 1998). For example, air filtration by trees may be marginally more beneficial in urban than rural areas where trees are more abundant. However, in time series analyses, such as the ones we conducted at two spatial scales, applying absolutely accurate ecosystem service value coefficients is likely less critical than for one-time cross-sectional analyses; in our time series analyses we were more interested in the directional change in ESVs than absolute values at specific points in time. Such directional changes are generally affected less by the assumed value coefficients than point in-time values (Kreuter et al., 2001). For this reason and the difficulty of obtaining marginal values of public goods, BTM has been used extensively to obtain first order estimates of changes in ESV over time. Another approach to addressing the limitations of BTM is performing sensitivity analyses, as we did in this study, to determine the effect of assumed value coefficients on total ESV estimates (Kreuter et al., 2001; Liu et al., 2012).

A final limitation of our study is uncertainty of land classifications. The proxies we used for each land class were not perfect matches for transferring values. For example, temperate/boreal forests are not equivalent to oak-juniper woodlands that dominate much of the upper SARB but this was the closest proxy we could identify. Similarly, agricultural lands were not classified to reflect different cropping systems. These limitations occur due to the characteristics of the sensor (Landsat 5 TM). Thus, a more detailed classification of forested lands by categories of species and of agricultural lands by cropping systems would allow for more accurate valuation of ecosystem services. More importantly for our study was uncertainty and ambiguity of imperviousness (i.e., percentage of impervious surfaces in a unit area). Because impervious surfaces consist of spatially mixed and spectrally heterogeneous features, it is often difficult to distinguish target objects from other land classes. Urban space classifications based on varying levels of imperviousness would be more appropriate for estimating urban ESV than the 50% imperviousness criterion we applied to differentiate low- and high-density urban areas. To overcome these limitations, satellite imagery of higher spatial resolution, preferably from hyperspectral sensors, are needed (Weng, 2012).

2.5. Conclusion

In our study we examined the impacts of land change and urbanization on ecosystem services at two scales, the SARB and Bexar County. Substantial land changes occurred in the study area between 1984 and 2010. Most notable are the large increase in low-density urban land occurring after NAFTA went into effect in 1994. Most of this low-density urban expansion occurred in and around Bexar County where the city of San Antonio is located. The changes in the ESVs during the study period indicate that the urban expansion in the SARB had significant impacts on the ecosystem services. Our findings also highlight the problematic nature of the urban coefficients included in two widely cited studies that include aggregated ecosystem service value coefficients for numerous biomes and which have frequently been applied to “analogous” land classes. Given value coefficients in these studies are based on multiple studies in different parts of the world, they may have some utility for approximating ESV trends of over time and space. However, we caution against the use of either of the two urban value coefficients ($0/ha/year and $7005/ha/year in 2010 US$) even for preliminary trend analysis. More place-based studies are needed to improve the estimate for the ESV of, in particular, urban areas at regional and local scales in order to more comprehensively and accurately characterize the potential effects of development polices, such as NAFTA, on the delivery of ecosystem services in the affected areas.

CHAPTER III

SPATIAL AND TEMPORAL CHANGES IN BIODIVERSITY AND ECOSYSTEM SERVICES AND THE SOCIAL-ECOLOGICAL IMPLICATIONS IN THE SAN ANTONIO RIVER BASIN, TEXAS, FROM 1984 TO 2010

3.1. Introduction

Human induced land changes to meet the rapidly growing demands of increasing population and economic growth affect biodiversity and ecosystem functioning at multiple scales with significant implications for sustainability (DeFries and Eshleman, 2004; Foley et al., 2005; Grimm et al., 2008). Recent studies have revealed that the nature's carrying capacity for biodiversity and ecosystem services (BES) are being impaired at unprecedentedly rapid rates, and land changes continue to be a driving force of biodiversity loss and ecosystem disruption (MEA, 2005; TEEB, 2010; IPBES, 2015). Urbanization is one of the most transformative forms of land change affecting biodiversity and ecosystem functioning (DeFries et al., 2010; Seto et al., 2012). For example, urban areas emit considerable amounts of carbon dioxide (CO_2) and greenhouse gases (GHG); thus urban expansion affects climate change in terms of the carbon sequestration across multiple spatial scales (Nowak et al., 2006; Hutyra et al., 2011; UN-Habitat, 2016). In addition, urbanization endangers species by replacing their habitat directly and leads to the degradation of habitats through extraction of natural resources to support urban activities by producing extensive ecological footprints (Czech and Krausman, 1997; Czech et al., 2000; Hansen et al., 2005; MEA, 2005; UNEP, 2011; Vimal et al., 2012; Mackintosh et al., 2015).

Biological diversity, or biodiversity means "the variability among living organisms from all sources including, *inter alia,* terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (UN, 1992), while ecosystem services refers to "benefits to humans from natural ecosystems" (MEA, 2005). Despite the improved the understanding of how the loss of biodiversity affects ecosystem functioning and human well-beings (Loreau et al. 2001; Hooper et al. 2005; Mace et al., 2012), accelerating rates of biodiversity loss and ecosystem degradation have been moving toward abrupt tipping points that would irreversibly reduce the carrying capacity of ecosystems for current and future human well-being (Secretariat of the Convention on Biological Diversity, 2010).

The San Antonio River Basin (SARB) is an ecologically diverse region in south central Texas. Rapidly urbanizing San Antonio-New Braunfels Metropolitan Statistical Area (MSA) is located in this basin. The city of San Antonio is the seventh most populous city in the US (U.S. Census Bureau, 2015) and is a trade center of the North American Free Trade Agreement (NAFTA) to facilitate a trilateral trade (i.e., Canada, Mexico, and the United States) (Brookings Institution, 2013), functioning as a major multi-modal transportation hub for the NAFTA (TTI, 2007; TxDOT, 2013). The population in the SARB has increased nearly 70% in the last 30 years due primarily to the economic growth in San Antonio region. It is expected that the population will reach about 2.8 million by 2060, which would represent a 94% increase since 2000 (TWDB, 2011).

Beyond these aggregate estimates, however, there is little understanding of how the land change and urban expansion impacted the biodiversity and ecosystems in the region at multiple scales over time. Furthermore, the links between biodiversity and ecosystem services is still poorly understood (Chapin III et al., 2000; Zhao et al., 2006; de Groot et al., 2010; Cardinale et al., 2012; Balvanera et al., 2014; McHale et al., 2013). Nelson et al. (2009) found that a range of ecosystem functions are synergistically associated with biodiversity conservation and policy interventions are justified to enhance the biodiversity and ecosystem services. For example, they suggested the payments for carbon sequestration to moderate the tradeoffs between development situations and different of biodiversity and ecosystem services. Others revealed a generally high overlap between biodiversity and ecosystem services (Turner et al., 2007; Bai et al., 2011; Harrison et al., 2014). However, the results from Chan et al. (2006) indicated that there are potential tradeoffs in spatial associations between biodiversity and ecosystem services with overall low correlation and relatively low pair-wise overlaps. Likewise, Egoh et al. (2009) reported that the match between biodiversity and ecosystem services is not strong.

None of these studies investigated how historical relationship between biodiversity, carbon storage, and sediment retention changed at multiple scales due to land changes. Renard et al. (2015) emphasize that a spatio-temporal approach of multiple ecosystem services should be taken into account to better understand the complex dynamics and interaction of ecosystem services and to identify the future trajectories for human well-being.

Moreover, integrated analysis of tightly coupled social-ecological systems (SES) is getting more important and essential to better understand the links between biodiversity and ecosystem services beyond the biophysical relations (Liu et al, 2007; Carpenter et al., 2009; Ostrom, 2009; Reyers et al. 2013; De Groot et al., 2014). With respect to social-ecological context, linkage between ecosystem services (ES) and environmental justice (EJ) has critical importance in urban areas for sustainability goals, such as urban resilience, public health, and fair provision of urban green space (UGS) (Sister et al. 2010; Wolch et al., 2014). Environmental justice is defined as "the fair treatment and meaningful involvement of all people regardless of race, color, national origin or income with respect to the development, implementation, and enforcement of environmental laws, regulations and policies" (USEPA, 2016), and includes disproportionate exposure to environmental risks in terms of socio-economic status (Chakraborty et al., 2014). Thus, the incorporation of socio-economic variables is critical for a comprehensive understanding of social and ecological resilience and urban sustainability, as well as the links between biodiversity and ecosystem services (McPhearson et al., 2014; Kremer et al., 2015).

Furthermore, despite the growing emphasis of integrating ecosystem services and environmental justice (Aragão et al. 2016), little has been done to include urban ecosystem services (UES) into an environmental justice framework that incorporates social-ecological feedbacks (Marshall et al., 2016). This represents a critical knowledge gap in that urbanization is a leading cause of habitat degradation and deterioration of ecosystem services (Mckinney, 2002; Grimm et al., 2008; Secretariat of the Convention on Biological Diversity, 2010; Elmqvist et al., 2013) and coupled concept of BES and EJ significantly increases ecological resilience and social justice in the increasing interconnected urban realm (Ernstson, 2013).

Thus, in this study, I ask the following questions: (1) What are the quantitative changes in biodiversity and ecosystem services in the San Antonio River Basin (SARB) and Bexar County in response to land changes from 1984 to 2010? (2) How did the spatial relationships between biodiversity and ecosystem services change over time? (3) What are the sustainability implications of these changes from the environmental justice perspective in a tightly coupled social-ecological context?

3.2. Methods

3.2.1. Study Area

The San Antonio River Basin (SARB), one of the major basins in South Texas, covers an ecologically diverse area of 10,862 km^2. The upper part of the SARB begins in the northeast corner of Bandera County. This portion of the Basin is dominated by the Edwards Plateau ecoregion in Texas Hill Country characteristics (SARA, 2014). The central part of the basin includes the heavily urbanized San Antonio-New Braunfels Metropolitan Statistical Area (MSA). The city of San Antonio is centered in Bexar County and lies about 140 miles northwest of the Gulf of Mexico and 150 miles northeast of Laredo on the Mexican border (Figure 3.1; San Antonio Chamber of

Commerce, 2015). The upper half of SARB intersects with the environmentally sensitive Edwards Aquifer drainage and recharge zones. The lower part of the basin is mostly rural and flows southeastward through the Gulf Coastal Plains. Especially, the surface and groundwater in the Edward Aquifer Recharge Zone (EARZ) is essential water resources for urban residents and the quantity and quality of water are critically linked with the proliferation of impervious surfaces, sediment retention and export in the region. The climate in the SARB ranges from semi-arid in the upper northwestern part to subtropical in the lower southeastern part near the Gulf Coast.

I selected "watershed" as the unit of analysis because it is fundamental to the provision of key ecosystem services for ground water and surface flow regulation and often represent the minimum ecological management unit (Kreuter et al., 2001; Troy and Wilson, 2006; Brauman et al., 2007; Zank et al., 2016). The Basin consists of 107 sub-watersheds at Hydrologic Unit Code (HUC) 12 level and contains almost all of Bexar County (Figure 3.1). I divided Bexar County into two parts to characterize the detailed changes in BES over time setting up at least one subwatershed buffer from urban watersheds along the boundary of Bexar County to identify the urban-rural gradient and impacts of urban sprawl on BES. Three urban watersheds; the Leon Creek Watershed, the Upper San Antonio River Watershed, and the Salado Creek Watershed, comprise Bexar County with 16 subwatersheds covering 1,579 km^2. The suburban watersheds consist of 22 subwatersheds along the boundary of Bexar County, while upstream watersheds and downstream watersheds consist of 20 and 49 sub-watersheds, respectively.

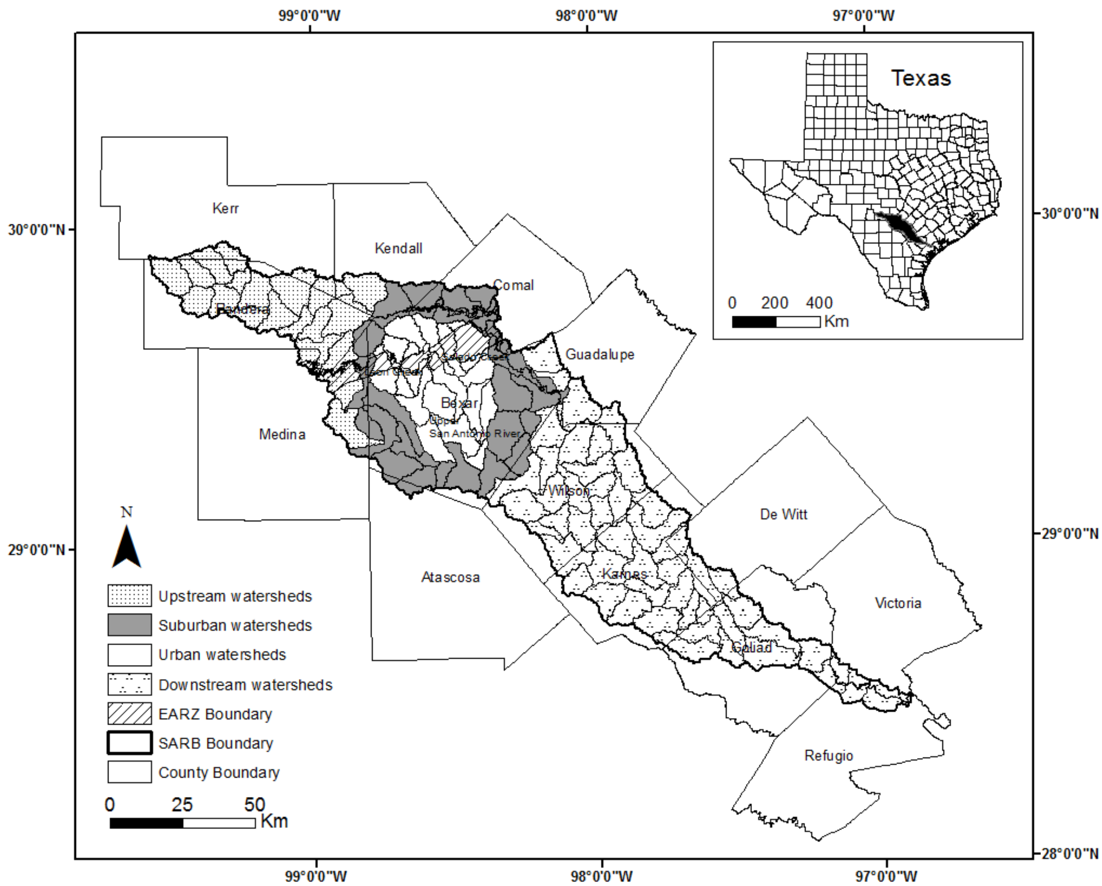


Figure 3.1. San Antonio River Basin (SARB) and 107 subwatersheds at Hydrologic Unit Code (HUC) 12 level.

3.2.2. Data Analysis.

I utilized cloud-free, multi-temporal Landsat 5 TM image data (30-meter spatial resolution, bands 1-5 and 7) acquired in November 1984, December 1995, and December 2010 (USGS, 2014) (http://earthexplorer.usgs.gov). I selected these image dates at time intervals that allow for the pre- and post-NAFTA analyses, and based on the availability of images from a consistent Landsat TM sensor, atmospheric conditions, and seasonal conditions under which land classes were expressed in a readily-interpretable manner. I used the watershed boundary dataset in terms of Hydrologic Unit Code (HUC) 12 level, which is delineated and georeferenced to U.S. Geological Survey (USGS) 1:24,000 scale topographic base maps from Texas Natural Resources Information Systems (TNRIS, 2015) (https://tnris.org/). I applied both spatial and numerical data for the analyses (Table 3.1, Table B2) and utilized the ecological production function based Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) 3.3 (Sharp et al., 2016) and ArcGIS 10.2 (ESRI, 2015). I modeled biodiversity and ecosystem services on a spatial grid at a 30 meter resolution and conducted analyses by an average per-hectare estimate based on a subwatershed level using the Zonal Statics function in the ArcGIS.

Table 3.1. Data and InVEST model characterization of biodiversity, carbon storage, and sediment retention.

	Method	Unit	Purpose	Data for the analysis
Biodiversity	Habitat quality index	Unitless	To assess the conditions of habitat	Land class Threat table
Carbon storage	Sum of four carbon pools	Mg C per hectare	To estimate terrestrial carbon stocks	Land class Carbon pool table
Sediment retention	Sediment delivery ratio	Ton per hectare	To estimate retained sediment	Land class DEM, K factor, R factor C and P factor table

3.2.3. Land-Change Analysis

I classified the satellite images using unsupervised Iterative Self-Organizing Data Analysis (ISODATA) (Jensen, 2005) into nine land classes: low-density urban, high-density urban, (cultivated) agricultural land, pasture, rangeland, forest land, water, wetland, and barren land. Urban areas can generally be defined by the percentage of impervious surfaces (Schueler, 1994; Arnold and Gibbons, 1996). I differentiated between low- and high-density urban areas to measure the level of imperviousness, biodiversity, and ecosystem services characteristics in response to urban expansion as follows: low-density urban consists of areas with less than 50% impervious surfaces, whereas high-density urban is comprised of areas with 50% or more impervious surfaces. I generalized these urban classes and other class definition of the NLCD 2006 classification system (http://www.mrlc.gov) (USGS, 2015). I conduct the classification accuracy assessment based on confusion matrices (Congalton and Green, 1999), where overall classification accuracies are 85.11%, 87.33%, and 85.78% for the 1984, 1995, and 2010 images, respectively.

3.2.4. InVEST Biodiversity and Ecosystem Services Quantification

Quantifying the terrestrial ecosystem services is required to estimate the impacts associated with land changes for the ecological production function method (EPFM) (Tallis and Polasky, 2009). The InVEST uses land class maps and tabular data together with environmental information (e.g., soil, topography, and climate) to generate spatially explicit biophysical supply of ecosystem services and habitat quality under the temporal land changes at multiple scales. (Kareiva et al., 2011; Sharp et al., 2016). Specifically, I quantified the BES estimates in the SARB and the 16 subwatersheds in Bexar County and derived the changes in BES, by calculating the difference between 1984 and 2010.

Secretariat of the Convention on Biological Diversity (2010) states that increasing pressures on biodiversity have accelerated the impairment of BES in spite of the continued measures of biodiversity conservation and ecosystem maintenance across the spatial scales. The analyses address these increasing concerns about the rapid decline in biodiversity and degradation of ecosystem services, which are critically related to water quality (Cardinale, 2011), as well as species habitat and multiple ecosystems functioning in the SARB. For instance, the Golden-cheeked Warbler (*Setophaga chrysoparia*), also known as the gold finch of Texas, is a key endangered species of bird that breeds in Ashe juniper (*Juniperus ashei*) and woodlands of the Texas Hill Country and the SARB (Kroll, 1980; Engels and Sexton 1994; Duarte et al., 2013; IUCN, 2016). Furthermore, Texas is a prime example for greenhouse gas (GHG) emissions as a high carbon and gray infrastructure intensified economy identified by the U.S. Energy Information Administration (2015), in that Texas ranks first in terms of total carbon dioxide (CO_2) emissions and transportation emissions by 642 and 222 million metric tons in 2014, respectively with increasing trends. Thus, carbon storage and sequestration, is of critical importance for the mitigation and adaptation of regional climate change in Texas as well as in the US (IPCC, 2007).

3.2.4.1. Biodiversity

InVEST calculates the habitat quality score as a proxy of biodiversity in Equations 4–5. Habitat quality is a function of the land class in the grid cell and the sensitivity of the habitat to the threats, which is posed by the surrounding land class. The function of habitat quality consists of four factors: 1) the relative impact of each threat, 2) the relative sensitivity of each habitat type to each threat, 3) the distance between habitats and sources of threats, and 4) the extent to which the land is legally protected (Terrado et al., 2016). Three threat factors, such as high density urban, low density urban, agricultural land were considered as potential threats in the SARB. The quality of habitat (Q_{xj}) in a grid cell x that is in $LULC_j$ can be calculated as follows.

$$D_{xj} = \sum_{r=1}^{R} \sum_{y=1}^{Y_r} \frac{w_r}{\sum_{r=1}^{R} w_r} r_y i_{rxy} S_{jr} \tag{4}$$

$$Q_{xj} = H_j \left(1 - \left(\frac{D_{xy}^z}{D_{xy}^z + k^z}\right)\right) \tag{5}$$

where D_{xj} is the sum of the total threat level in a grid cell x of habitat type j in Equation 4, Q_{xj} in Equation 5 is the quality of habitat in the grid cell x, H_j is Boolean map based on the user's definition by which land class can provide habitat for the conservation objective; and z (z = 2.5) and k are scaling parameters; and half saturation constant with 0.5 (Sharp et al., 2016). The proximity to land class and the intensity of human land use, such as economic activities and urban impervious surfaces impact the habitat quality (Czech, et al., 2000, McKinney, 2002). The habitat quality analyses in the SARB were conducted using the input parameter values in Tables B3–B4.

3.2.4.2. Carbon Storage and Sequestration

Carbon storage model in InVEST estimates the amount of carbon stored in a landscape based on the carbon density of each LULC category (Equation 6).

$$TCS = \Sigma A_k \times (C_{ak} + C_{bk} + C_{sk} + C_{dk}) \tag{6}$$

where TCS (Mg C/ha per year) is the terrestrial carbon storage in the study area for a particular year and A_k is the area (ha) for land use category 'k' and C_{ak} is the aboveground carbon density (Mg C/ha per year), C_{bk} is the belowground carbon density (Mg C/ha per year), C_{ck} is the soil carbon density (Mg C/ha per year), C_{dk} is the dead mass carbon density (Mg C/ha per year). Storage refers to the mass of carbon in an ecosystem at specific time, while sequestration means the change in carbon storage over time. In the analyses four types of carbon values were mainly based on the InVEST (Table B5). For example, I used the four types of unit carbon values for forest class, assigning 63 Mg C/ha for aboveground biomass, 53 Mg C/ha for belowground biomass, 43 Mg C/ha for soil carbon, 1 Mg C/ha for dead mass, respectively, which amount to total 160 Mg C/ha (Health et al., 2011; COLE, 2016).

In order to better characterize the ecosystem services with regard to urban areas, I assumed an average of 75% and 25% green space for low- and high-density urban areas, based on an inspection of orthophotos used for the accuracy assessment, respectively. I also assumed the steady-state level for carbon storage for each land class, which means the change in carbon storage is solely due to the land change between 1984 and 2010 (Polasky et al. 2011, Sallustio et al. 2015). The economic value of carbon sequestration was estimated in terms of the social cost of carbon (SCC). The SCC is based on the axiom in the Ecological Economics that it works as the Pigouvian tax that could be placed on the over-production of carbon dioxide (CO_2), internalizing resulting air pollution and social costs associated with carbon emissions (Pigou, 1920). Thus, the values of SCC means the marginal damage cost of carbon dioxide (CO_2) for society to estimate the value of reducing carbon emissions to address the market failure and climate change. I used the 37 US$ per metric ton of carbon as the estimated social cost of carbon (OIRA, 2013), and measured the values of annual sequestration in 2015 constant US dollars following Polasky et al. (2011).

3.2.4.3. Sediment Retention and Export

I used the sediment delivery ratio (SDR) model to quantify the sediment retention in the SARB. The model quantifies the amount of eroded sediment at the grid cell and calculates the SDR reaching the catchment outlet. The amount of annual soil loss on grid cell *i* is given by the revised universal soil loss equation (RUSLE) (Equation 7).

$$USLE_i = R_i \times K_i \times LS_i \times C_i \times P_i \tag{7}$$

where R_i is the rainfall erosivity index (*MJ·mm·(ha·hr)*$^{-1}$), K_i is the soil erodibility (*ton·ha·hr·(MJ·ha·mm)*$^{-1}$), LS is the slope length-gradient factor (unitless), C is the crop management factor (unitless), and P is the support practice factor (unitless).

The sediment delivery ratio (SDR) is quantified as a function of the hydrologic connectivity of the area. The algorithm computes an index of connectivity (IC), which determines the degree of hydrological connectivity of a grid cell to the stream, based on its upslope contribution and flow path to the stream. The sediment delivery ratio for a grid cell *i* is then directly derived from the conductivity index, using a sigmoid function (Equation 8) (Sharp et al., 2016).

$$SDR_i = \frac{SDR_{max}}{1+exp\left(\frac{IC_0-IC_i}{k}\right)} \tag{8}$$

SDR_{max} is the maximum theoretical SDR, defined as the maximum proportion of fine sediment which can travel to the stream. I used the value of 0.3, based on the observational data in the SARB (https://www.nrcs.usda.gov/) (USDA, 2016). To estimate the flow accumulation threshold (tfac), I utilized the ArcHydro tool from the ArcGIS toolbox to obtain a raster of flow accumulation from the digital elevation model (DEM) (ESRI, 2015). Then, I overlaid the stream network from the National Hydrographic Dataset (NHD; http://nhd.usgs.gov) to match the corresponding threshold. A threshold value of 300 generated the best fit resulting in more than 90% matching. IC_0 and k_b are calibration parameters that define the shape of the sigmoid function SDR–IC relationship (Vigiak et al., 2012). The sediment yield from a given grid cell i is a function of the soil loss and SDR factor (Equation 9).

$$Pixel_i = USLE_i \times SDR_i \tag{9}$$

The model produces three main outputs: i) total amount of sediment exported to the stream (tons/year); ii) total amount of potential soil loss calculated by the USLE (tons/year); iii) sediment retention as the difference in the amount of sediment delivered by the current land cover and a hypothetical watershed where all land use types have been cleared to bare soil (tons/year) (Sharp et al., 2016). The values for the cover management factor (C) and support factor (P) for each land cover type were derived from Wischmeier and Smith (1978) and Hamel et al. (2015). IC_0 and kb are two calibration parameters that determine the shape of the relationship between hydrologic connectivity (i.e., the degree of connection from patches of land to the stream) and the sediment delivery ratio. The default value of 0.5 was used for IC_0, while the kb was adjusted to 1.3 for calibration. The result of 431,261 (tons/year) from the InVEST sediment export in 2010 (Table 3.3, Table B6) was compared and matched by the sediment transport data at confluence with Guadalupe River in the SARB (Banta and Okerman, 2014; Okerman et al., 2015).

3.2.4.4. Sensitivity Analysis for the BES estimates

To address uncertainties in the unit value of each parameter, values of the InVEST process-related parameters were further evaluated by a set of sensitivity analyses to determine the effects of changes in input parameters on the affected biodiversity, carbon storage, and sediment retention (export) results. Each sensitivity calculation was made by adjusting a input parameter of the InVEST model by the ± 50 percent changes while keeping other input parameters unchanged (Sánchez-Canales et al. 2015; Hamel et al., 2015). The results from sensitivity analyses were compared and evaluated in terms of the coefficient of sensitivity (CS) for all three years (i.e., 1984, 1995, and 2010) (Equation 10).

$$CS = \frac{(ESV_j - ESV_i)/ESV_i}{(VC_{jk} - VC_{ik})/VC_{ik}} \tag{10}$$

where ESV is the estimated biodiversity and ecosystem service values, VC is the parameter value coefficient, '*i*' and '*j*' represent the default and adjusted parameter value, respectively, and '*k*' represents the input parameter category (Kreuter et al., 2001).

3.2.5. Analysis of Spatio-temporal Relationships

I compared BES estimates in the whole SARB and urban watersheds in Bexar County. To determine whether the parametric or non-parametric approach to be applied in the study area I conducted the Shapiro-Wilk normality test for the data distribution using SPSS 18.0 (IBM, 2016). The results from the normality test indicates that the data are not normally distributed with the non-linear patterns from 1984 to 2010 ($p<0.05$). Thus, I applied the non-parametric tests due to the unbalanced distribution BES estimates for each model, non-linearity, and skewness, which are commonly characterized in terms of the quantification of biodiversity and ecosystem services.

3.2.5.1. Changes in Spatial and Temporal Relationships

Correlations for BES estimates in each year were analyzed using the non-parametric approach to alleviate the normality assumption in the data. Thus, I calculated Spearman's ρ across the 107 sub-watersheds and 16 sub-watersheds to estimate the spatial characteristics over time. A spider diagram was applied to illustrate relationships between biodiversity and ecosystem services for the purpose of relating biodiversity to multiple ecosystem services. It is a useful analytical framework to depict relative relationships at multiple scales over time to inform environmental decision-making processes to examine the tradeoffs and synergies of multiple ecosystem services (DeFries et al., 2004; Kroll et al. 2012). I normalized BES into the scale from 0 to 1 in terms of dividing each value by the maximum value per year to facilitate comparisons of interactions between biodiversity and ecosystem services. Thus, the scale on the spider diagram means either a win-win (i.e., synergy) or a trade-off for the particular biodiversity and ecosystem services.

3.2.5.2. GIS based BES Hotspot and Overlap Analyses

Getis-Ord Gi* statistic (Getis and Ord 1992) was applied to identify the biodiversity and ecosystem services hotspots and overlaps in the subwatershed level. I utilized the Hot Spot Analysis in ArcGIS 10.2 with mean BES estimates as the input values (Timilsina et al., 2013). This tool measures how concentrated the high or low values are in the study area and identifies statistically significant spatial clusters of high values (hot spots) and low values (cold spots) ($p<0.1$) (ESRI, 2016).

The overlap analyses between biodiversity and ecosystem service hotspots provide more relevant the information for the conservation targets and the suitability (Chan et al., 2006). For the BES hotspot overlap, I calculated the overlaps of BES hotspots, which measure shared areas in terms of proportional overlap (Prendergast et al. 1993; Egoh et al., 2009). The assessment of BES was conducted by the hotspot areas of BES provision and overlap of hotspots with hotspots for each subwatershed where biodiversity and ecosystem services co-occur (Reyners et al., 2009).

3.2.5.3. Spatio-temporal Land Change Impacts on BES

I applied the non-parametric Kruskal-Wallis test (Kruskal and Wallis, 1952) to determine if there are statistically significant differences due to land change among the 4 sub-areas (i.e., upstream, suburban, urban, downstream watersheds) in terms of biodiversity and ecosystem services (i.e., carbon storage, sediment retention (export)) in the SARB. For post-hoc comparison, I conducted the Mann-Whitney U test (Mann and Whitney, 1947) to test the differences between two independent samples of non-parametric data and to find the variation in how different sub-areas responded each other. In addition, I applied the non-parametric Wilcoxon signed rank test (Wilcoxon, 1945) to examine the impact of urban sprawl among the four sub-areas on the conditions of biodiversity and ecosystem services during the pre- (1984-1995) and the post-NAFTA (1995-2010) periods.

3.2.6. Environmental Justice Analysis in BEXAR County

Most studies that quantify ecosystem services do not disaggregate beneficiaries and consider the distribution of benefits between groups and individuals in society (Daw et al., 2011). Recently, it has been recognized that research on ecosystem services (ES) has significant potential to address environmental justice (EJ) issues for better socio-ecological decision-making (Marshall and Gonzalez-Meler 2016; USEPA, 2016). Moreover, EJ analysis is necessary to disaggregate BES beneficiaries into winners and losers in terms of ethnicity, geography, and socioeconomic status. I examined the relationships between the distribution of environmental benefits, such as BES estimates, Normalized Difference Vegetation Index (NDVI), public health risks, and socio-economic variables including the race/ethnic groups from the EJ perspective (Chakraborty et al., 2014; Grineski et al., 2015). Specifically, the median income, poverty rate, unemployment rate, race/ethnic percentage (i.e., Hispanic, White, African American) from the American Community Surveys (ACS) 2010 (US Census Bureau, 2010) in the 361 census tracts in Bexar County were used. For public health risks, I used the data for the diesel particulate matter, total respiratory hazard index (HI), and total air toxics cancer risk from the National Air Toxics Assessment (NATA) 2011 (US EPA, 2015), considering air pollution caused by transportation is a growing health concern in the San Antonio region (AACOG, 2015).

3.3. Results

3.3.1. Land Changes in the SARB, Bexar County, and the EARZ

Land classification indicates substantial urban growth between 1984 and 2010 in the SARB, particularly around San Antonio. The urban proportion of the SARB increased steadily during the study period, while rangeland and forest, which are the two largest land classes, declined markedly (Table 3.2). In the urban watersheds in Bexar County, the forest land represented the largest land class in 1984 but then decreased to 26.6% in 2010 (Table B1). Rangelands, likewise, decreased substantially. By contrast, the combined area of the two urban land classes more than tripled in 2010. In the Edward Aquifer Recharge Zone (EARZ) (Figure B1), the results indicate that the urban growth is more than double the rate in the SARB and in Bexar County and the proportion of urban land consists of more than 20% (Table 3.2). Importantly, the impervious surfaces exceeded overall 10% in the environmentally sensitive EARZ.

Table 3.2. Total estimated area (ha) and the percent cover of each land class in the SARB (a) and in the EARZ of the SARB (b).

(a)

	Total area (ha, %)					
Land class	1984 (ha)	1984 (%)	1995 (ha)	1995 (%)	2010 (ha)	2010 (%)
Urban in the SARB	46,602 (19,894)	4.3 (12.6)	76,095 (39,666)	7.0 (25.1)	143,929 (60,663)	13.3 (38.4)
Low density urban	31,327 (14,767)	2.9 (9.4)	45,312 (22,439)	4.2 (14.2)	85,764 (26,698)	7.9 (16.9)
High density urban	15,275 (5,127)	1.4 (3.2)	30,783 (17,227)	2.8 (10.9)	58,165 (33,965)	5.4 (21.5)
Agricultural Land	111,835 (8,752)	10.3 (5.5)	104,841 (9,418)	9.7 (6.0)	92,611 (3,756)	8.5 (2.4)
Pasture	173,895 (10,245)	16.0 (6.5)	193,128 (8,919)	17.8 (5.6)	199,338 (14,996)	18.4 (9.5)
Rangeland	411,210 (57,496)	37.9 (36.4)	392,479 (34,858)	36.1 (22.1)	389,135 (34,692)	35.8 (22.0)
Forest Land	324,391 (59,175)	29.9 (37.5)	300,864 (62,640)	27.7 (39.7)	251,245 (42,012)	23.2 (26.6)
Water	3,672 (82)	0.3 (0.1)	4,267 (133)	0.4 (0.1)	4,015 (77)	0.4 (0.1)
Wetland	960 (117)	0.1 (0.1)	618 (125)	0.1 (0.1)	570 (56)	0.1 (0.0)
Barren Land	12,379 (2,111)	1.1 (1.3)	12,109 (2,115)	1.1 (1.3)	3,345 (1,622)	0.3 (1.0)
No Data	807 (2)	0.1 (0.0)	1,350 (0)	0.1 (0.0)	1,563 (0)	0.1 (0.0)
Total (ha)	1,085,751 (157,874)	100.0 (100.0)	1,085,751 (157,874)	100.0 (100.0)	1,085,751 (157,874)	100.0 (100.0)

( ) denotes the values of urban watersheds in Bexar County of the SARB.

Table 3.2. Total estimated area (ha) and the percent cover of each land class in the SARB (a) and in the EARZ of the SARB (b) (Continued).

(b)

	Total area (ha, %)					
Land class	1984 (ha)	1984 (%)	1995 (ha)	1995 (%)	2010 (ha)	2010 (%)
Urban in the EARZ	1,556	3.3	3,481	7.2	9,926	20.7
Low density urban	1,082	2.3	2,316	4.8	5,687	11.9
High density urban	474	1.0	1,165	2.4	4,239	8.8
Agricultural Land	1,236	2.6	1,509	3.2	493	1.0
Pasture	1,897	4.0	1,361	2.8	2,550	5.3
Rangeland	14,365	30.0	8,647	18.1	10,529	22.0
Forest Land	28,161	58.8	32,128	67.1	23,495	49.1
Water	51	0.1	55	0.1	45	0.1
Wetland	23	0.1	20	0.1	19	0.1
Barren Land	577	1.2	671	1.4	814	1.7
No Data	11	0.0	5	0.0	6	0.0
Total (ha)	47,877	100.0	47,877	100.0	47,877	100.0

3.3.2. Quantitative Analyses of BES Estimates and Spatial Distribution

Biodiversity quantification indicates decreasing trend within the north-western region from 1984 to 2010 in the SARB, particularly around San Antonio (Figure 3.2). The annual rate of biodiversity loss was 0.1% higher in the post-NAFTA (1995-2010) period compared to the pre-NAFTA (1984-1995) period (Table 3.3). However, the annual rate of biodiversity loss per year was 0.2% higher during the post-NAFTA period (Table 3.4) in the urban watersheds in Bexar County. Carbon storage indicates high carbon stock distribution in the upper part of the SARB with decreasing patterns (Figure 3.2). The total carbon storage decreased by 4.0% during the pre-NAFTA period and by 7.8% during the post-NAFTA period, amounting to 75.8 million Mg C in 2010. The annual rate of carbon storage loss was 0.1% higher during the post-NAFTA period in the SARB (Table 3.3). In the urban watersheds in Bexar County, carbon storage decreased much greater, indicating the annual rate of carbon storage loss was 1.3% higher during the post-NAFTA period (Table 3.4), which amounts to 10.7 million Mg C in 2010.

Table 3.3. Changes in biodiversity, carbon storage, sediment retention (export) in the SARB.

	BES estimation per year			1984-1995			1995-2010			1984-2010		
Biodiversity Ecosytem Services	1,984	1,995	2,010	Change	%	%/year	Change	%	%/year	Change	%	%/year
Biodiversity (Unitless)	9,471,443	9,259,334	8,908,040	-212,109	-2.2	-0.2	-351,294	-3.8	-0.3	-563,403	-5.9	-0.2
Carbon storage (Mg C)	85,669,518	82,243,805	75,844,795	-3,425,713	-4.0	-0.4	-6,399,010	-7.8	-0.5	-9,824,723	-11.5	-0.4
Sediment retention (ton)	57,373,771	57,379,906	57,303,720	6,135	0.0	0.0	-76,186	-0.1	0.0	-70,051	-0.1	0.0
Sediment export (ton)	361,210	355,075	431,261	-6,135	-1.7	-0.2	76,186	21.5	1.4	70,051	19.4	0.7

Table 3.4. Changes in biodiversity, carbon storage, and sediment retention (export) in Bexar County.

	BES estimation per year			1984-1995			1995-2010			1984-2010		
Biodiversity Ecosytem Services	1,984	1,995	2,010	Change	%	%/year	Change	%	%/year	Change	%	%/year
Biodiversity (Unitless)	1,380,205	1,268,591	1,102,905	-111,614	-8.1	-0.7	-165,686	-13.1	-0.9	-277,300	-20.1	-0.8
Carbon storage (Mg C)	13,752,537	13,604,787	10,707,880	-147,750	-1.1	-0.1	-2,896,907	-21.3	-1.4	-3,044,657	-22.1	-0.9
Sediment retention (ton)	4,591,570	4,584,057	4,574,630	-7,513	-0.2	0.0	-9,427	-0.2	0.0	-16,940	-0.4	0.0
Sediment export (ton)	46,878	54,391	63,818	7,513	16.0	1.5	9,427	17.3	1.2	16,940	36.1	1.4

Sediment retention results indicate that the spatial distribution has been stable from 1984 to 2010 (Figure 3.2). The total sediment retention slightly decreased, amounting to 57.3 million ton in 2010 (Table 3.3). On the other hand, the total sediment export increased by 0.7% per year, indicating the annual rate of sediment export was 1.6% higher during post-NAFTA period in the SARB (Table 3.3). In the urban watersheds in Bexar County, The total sediment retention slightly decreased amounting to 4.5 million ton in 2010. On the other hand, the total sediment export increased by 1.4% from 1984 to 2010. The annual rate of sediment export was 0.3% lower during post-NAFTA period (Table 3.4).

3.3.3. Carbon Storage Valuation

The total amount of money value in carbon storage in constant 2015 US$ decreased, which amounts to 2,806 million US$ in 2010, indicating decrease by 4.0% during the pre-NAFTA period and by 7.8% during post-NAFTA period in the SARB. On the other hand, the total money values in the urban watersheds in Bexar County indicate that total money values in 2015 US$ decreased amounting to 396 million US$ in 2010 (Table 3.5). The monetary valuation based on the ecosystem approach indicates considerable carbon loss in terms of dollar values in both the SARB and the urban Bexar County.

Table 3.5. Changes in values of carbon storage in the SARB and Bexar County.

	Carbon value (2015 US million $ per year)			1984-1995			1995-2010			1984-2010		
	1,984	1,995	2,010	Change	%	Change/year	Change	%	Change/year	Change	%	Change/year
SARB	3169.7	3043.0	2806.2	-126.7	-4.0	-11.5	-236.7	-7.8	-15.7	-363.5	-11.5	-13.9
Bexar County	508.8	503.3	396.1	-5.4	-1.1	-0.4	-107.1	-21.3	-7.1	-112.6	-22.1	-4.3

(a)

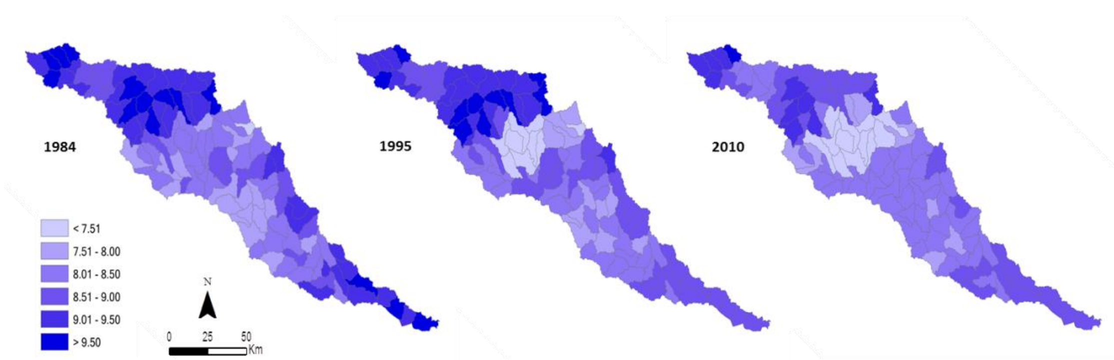
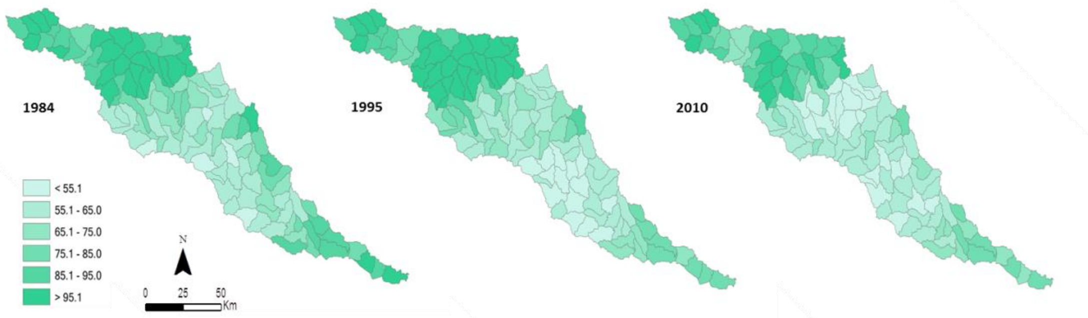
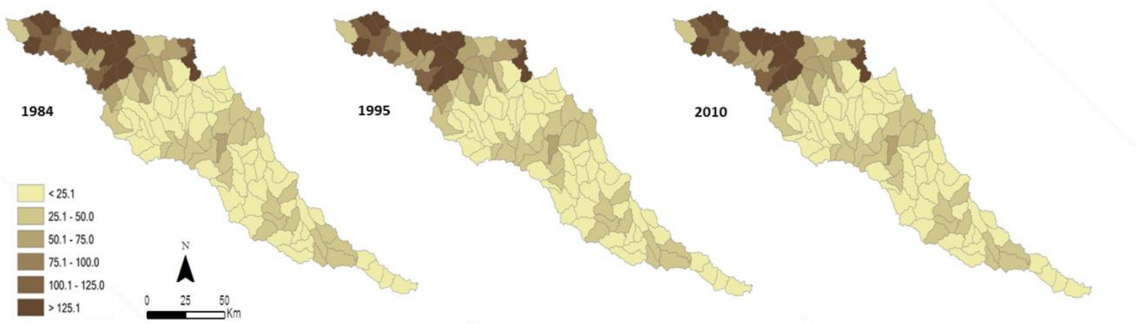


Figure 3.2. Spatial distribution of biodiversity (score/ha) (a), carbon storage (Mg C/ha) (b), sediment retention (ton/ha) (c), and sediment export (ton/ha) (d) in the SARB.

(b)



(c)



(d)

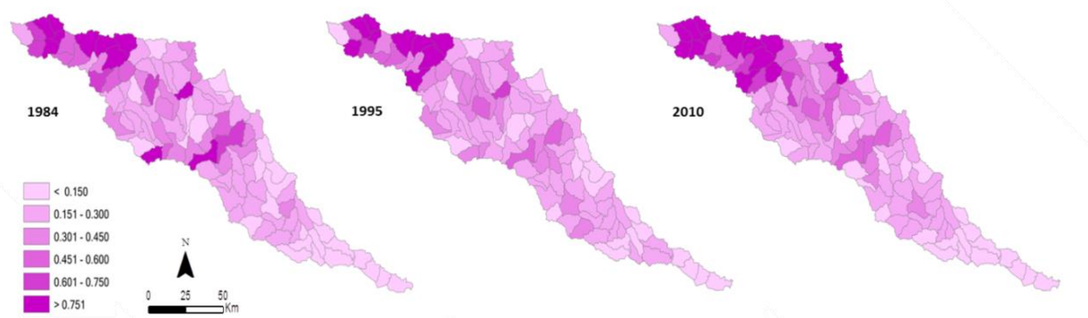


Figure 3.2. Continued.

3.3.4. Sensitivity Analyses for the BES Estimates

The results from the sensitivity analysis of biodiversity indicate that changed habitat scores at both scales of analyses are relatively inelastic (i.e., CS close to 0) (Table B7). Adjusting value coefficients (VC) for low density urban, high density urban, agricultural land had little impact on the estimated ESV, primarily because these land classes are substantially saturated to the potential impacts areas in terms of the default parameter values. The results from the sensitivity analyses of carbon storage indicate that simulated habitat scores at both scales of analyses are relatively inelastic (i.e., CS substantially < 1) except for the rangeland and forest classes (Figures 3.3–3.4, Table B8). Adjusting value coefficients (VC) for forest class had high impact on the estimated ESV, followed by rangeland. The results from the sensitivity analyses of sediment retention indicate that simulated ESV at both scales of analyses are relatively inelastic (i.e., CS close to 0) except for the kb and IC_0 of calibration parameters (Figures 3.3–3.4, Table B9). Adjusting value coefficients (VC) for kb had high impact on the estimated ESV, followed by IC_0. Similarly, the results from sediment export indicate that adjusting value coefficients (VC) for kb had high impact on the estimated ESV, followed by low density urban, and IC_0 (Figures 3.3–3.4, Table B10).

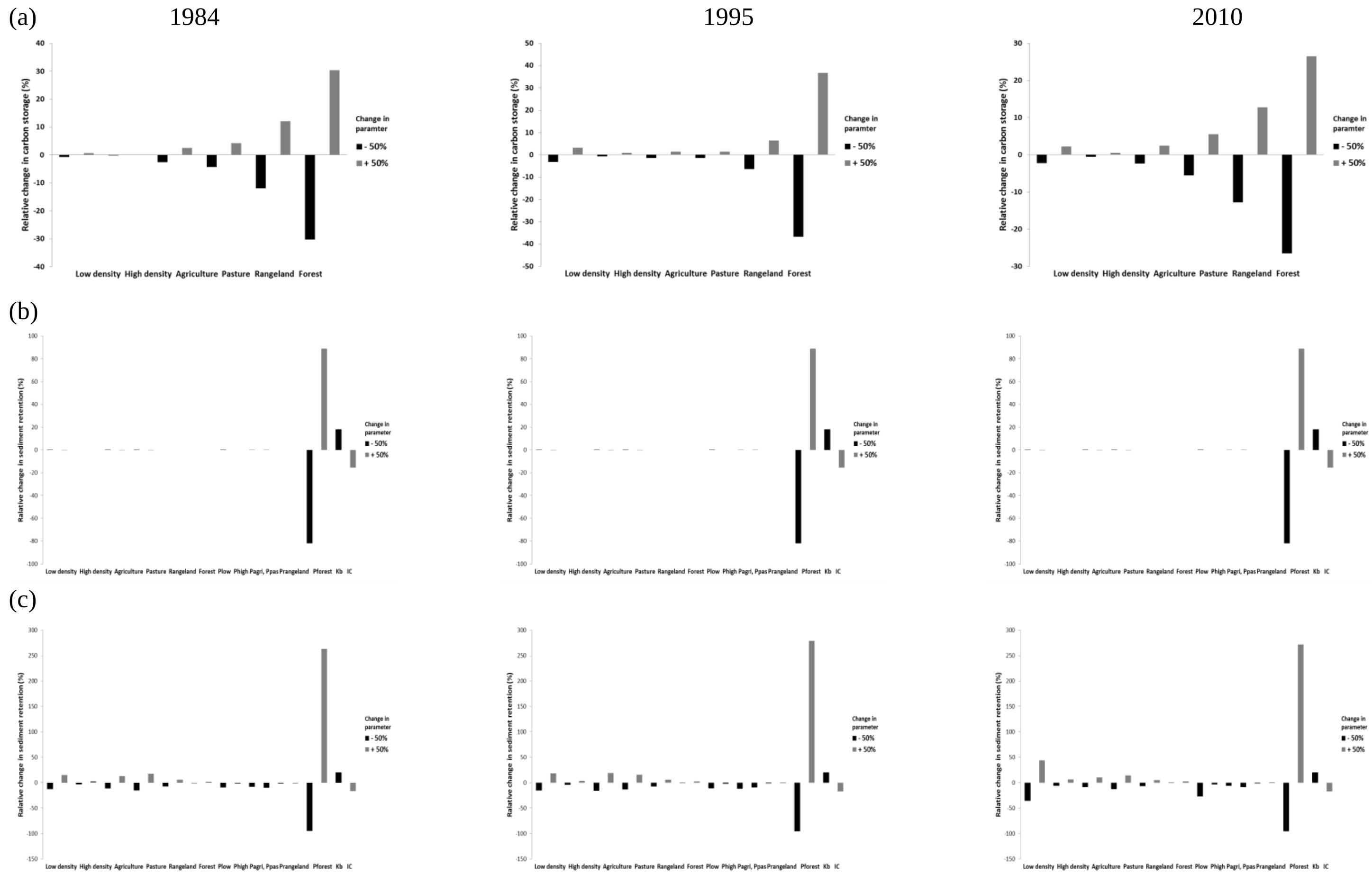


Figure 3.3. Sensitivity results for carbon storage (a) with inputs of low density, high density, agriculture, pasture, rangeland, forest from left to right, and sediment retention (b) and sediment export (c) in the SARB with inputs of low density, high density, agriculture, pasture, rangeland, forest, low density (P factor), high density (P factor), agriculture (P factor), pasture (P factor), rangeland (P factor), forest (P factor), Kb, IC from left to right.

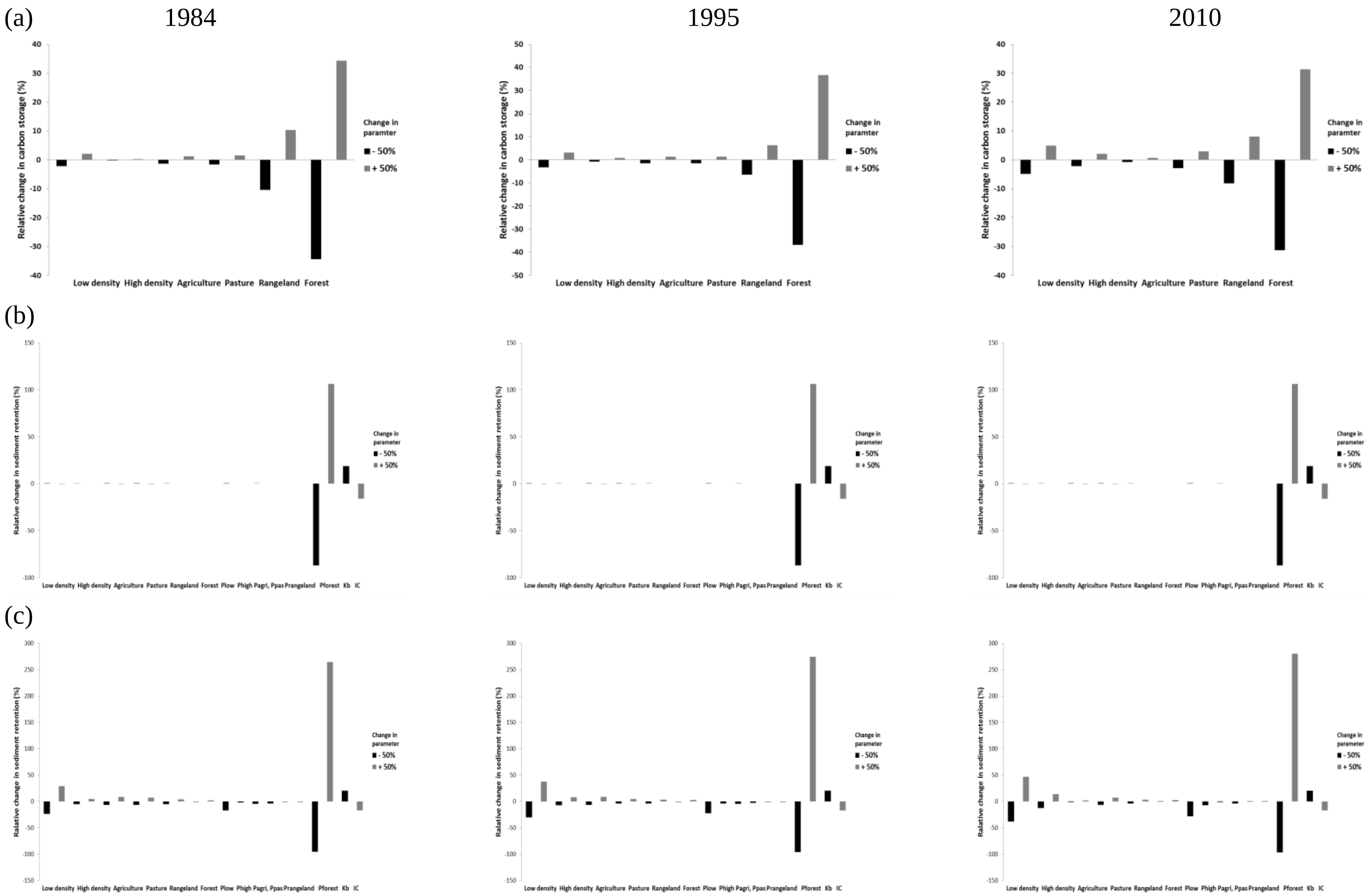


Figure 3.4. Sensitivity results for carbon storage (a) with inputs of low density, high density, agriculture, pasture, rangeland, forest from left to right, and sediment retention (b) and sediment export (c) in Bexar County with inputs of low density, high density, agriculture, pasture, rangeland, forest, low density (P factor), high density (P factor), agriculture (P factor), pasture (P factor), rangeland (P factor), forest (P factor), Kb, IC from left to right.

3.3.5. Changes in Spatial and Temporal Relationships

BES estimates per hectare at 95% confidence interval (CI) are described in Figure 3.5. The biodiversity scores decreased from 8.71 score/ha per year to 8.22 score/ha per year in the SARB and from 8.71 score/ha per year to 7.06 score/ha per year in Bexar County indicating urban watersheds were more negatively impacted by the urbanization. Carbon storage also decreased from 78.63 Mg C/ha per year to 69.89 Mg C/ha per year in the SARB and from 86.05/ha per year to 67.74/ha per year in Bexar County.

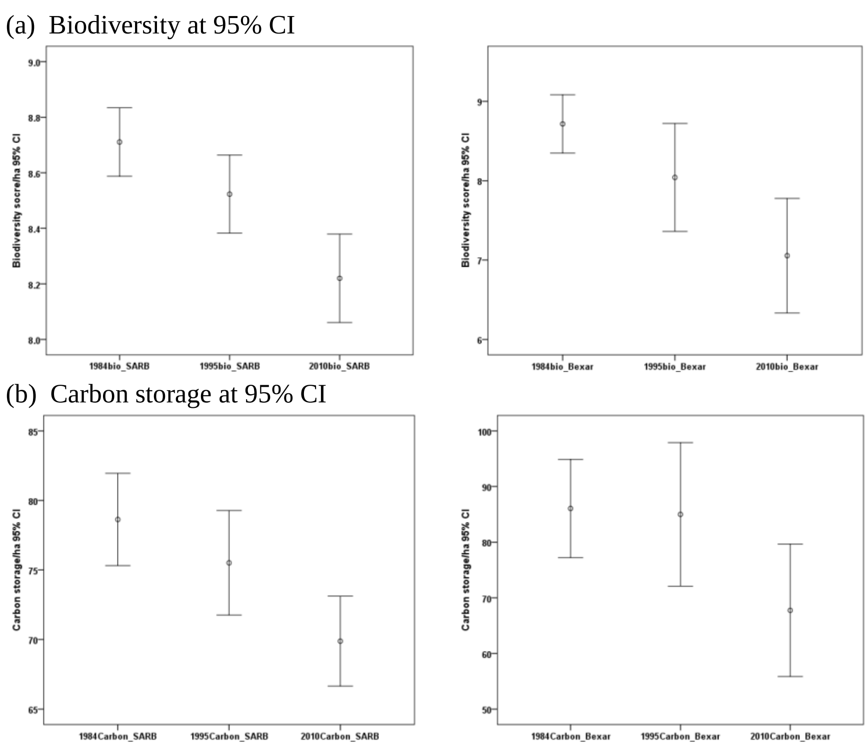
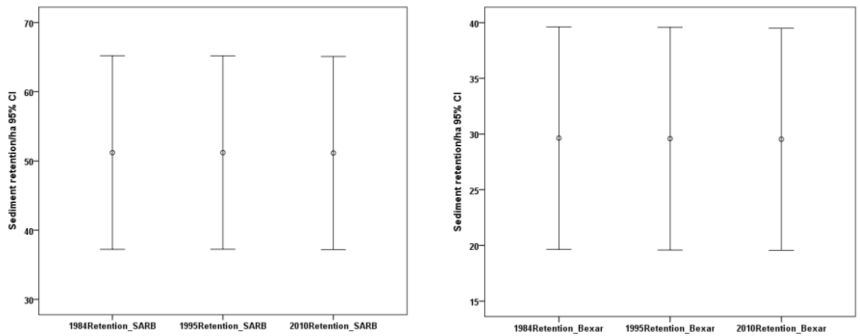


Figure 3.5. Biodiversity (score/ha) (a), carbon storage (Mg C/ha) (b), sediment retention (ton/ha) (c), and sediment export (ton/ha) (d) at 95% CI in the SARB and Bexar County (left=1984, middle=1995, right=2010).

(c) Sediment retention at 95% CI



(d) Sediment export at 95% CI

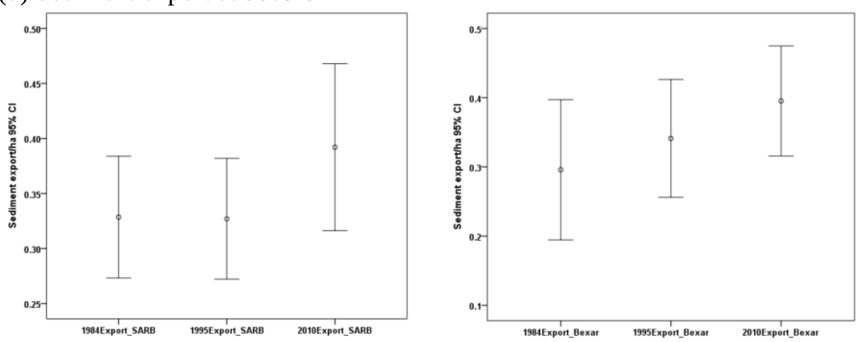


Figure 3.5. Continued.

Sediment retention was stable around 51.8 ton/ha per year in the SARB and 29.6 ton/ha per year in Bexar County. On the other hand, sediment export slightly increased from 0.33 ton/ha per year to 0.39 ton/ha per year in the SARB and from 0.3 ton/ha per year to 0.4 ton/ha per year in Bexar County. The results from nonparametric analyses indicate that the positive correlations between the biodiversity–carbon storage pair and between the biodiversity–sediment retention pair are statistically significant ($p<0.01$) at both spatial scales of analyses over time (Table 3.6). The historical correlation between the carbon storage–sediment retention was also statistically significant ($p<0.01$ in the SARB; $p<0.05$ in Bexar County).

Table 3.6. Nonparametric Spearman's correlations among biodiversity, carbon storage, and sediment retention.

	1984		1995		2010	
	SARB	Bexar County	SARB	Bexar County	SARB	Bexar County
Biodiversity and carbon storage	0.922**	0.949**	0.826**	0.944**	0.805**	0.906**
Biodiversity and sediment retention	0.538**	0.674**	0.694**	0.718**	0.539**	0.785**
Carbon storage and sediment retention	0.568**	0.574*	0.639**	0.671**	0.707**	0.771**

** Correlation is significant at the 0.01 level (2-tailed)

* Correlation is significant at the 0.05 level (2-tailed)

(a) (b)

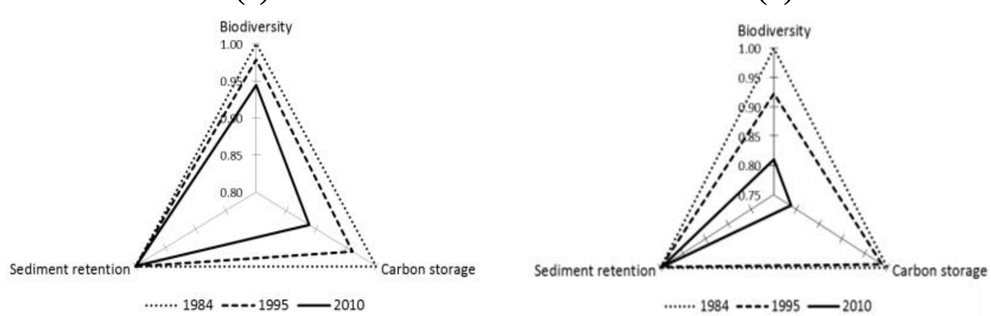


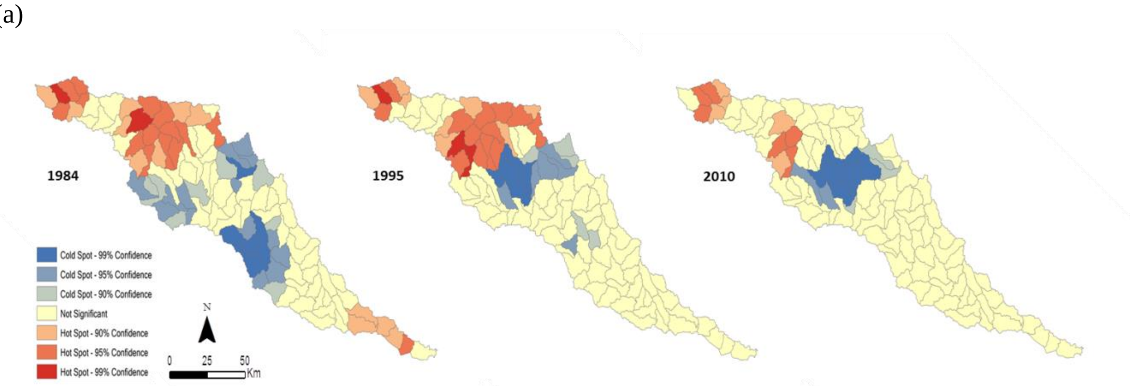
Figure 3.6. Diagrams for the historical relationships between 1984 and 2010 in the SARB (a) and Bexar County (b).

The normalized spider diagrams for BES estimates show the relationships in a single frame and summarize the temporal dynamics for the BES (Figure 3.6). Rapid decline in carbon storage and biodiversity is identified in urbanizing Bexar County, compared to the SARB with the same direction but different degrees and relationships in the diagram. On the other hand, sediment retention service is relatively stable, compared to biodiversity and carbon storage both in the SARB and Bexar County.

3.3.6. GIS based Hotspot and Overlap Analyses.

Moran's I results indicated that biodiversity and ecosystem services varied substantially across the SARB and showed distinct geographic distributions ($p<0.01$) (Table B11). Hotspot Analysis (Getis-Ord Gi*) identified spatially clustered subwatersheds in term of biodiversity, carbon storage, sediment retention (export) (Figure 3.7). Statistically significant hotpots were mostly identified in the upstream watersheds mainly covered with forest/woodland classes in the SARB from 1984 to 2010. Cold spots are located in and around the urban watersheds and agricultural land.

(a)



(b)

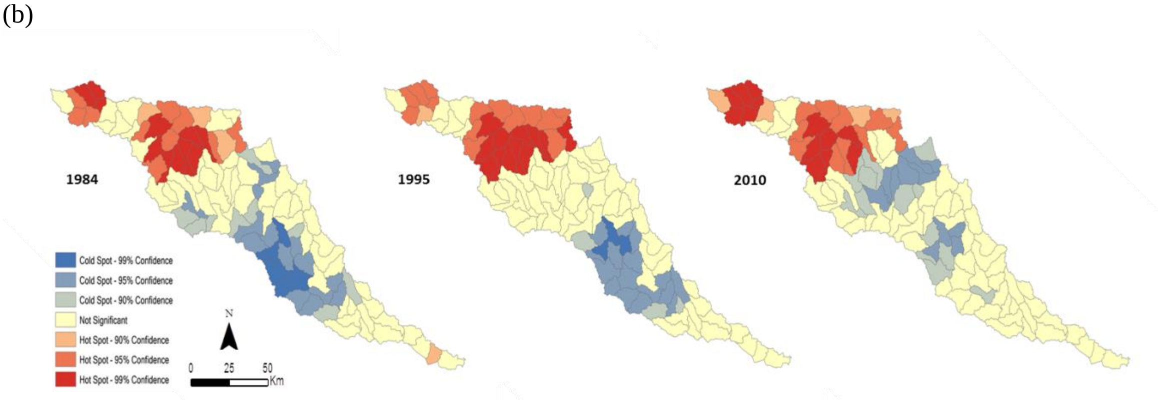
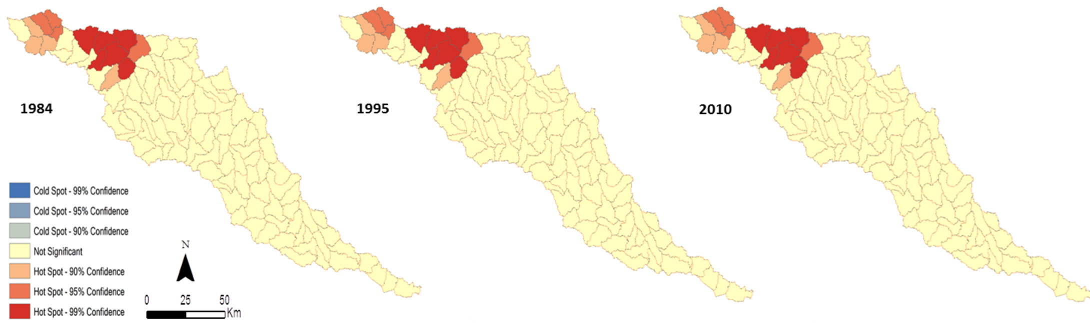


Figure 3.7. Hotspots/cold spots of biodiversity (a), carbon storage (b), sediment retention (c), and sediment export (d) in the SARB.

(c)



(d)

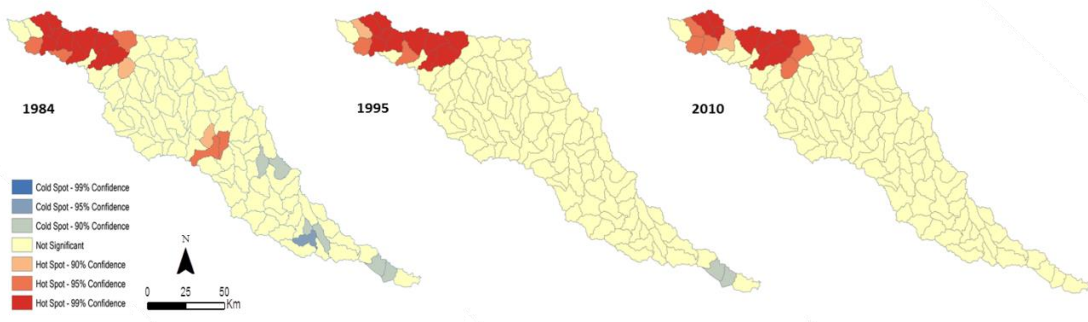


Figure 3.7. Continued.

Table 3.7 presents the extent of BES hotspots at the 90%, 95%, and 99% confidence level and overlaps across the SARB. The proportional overlap measures the shared area between biodiversity and ecosystem services expressed as a percentage of the SARB area (Reyers et al., 2009). The result indicates that the hotspot proportion of biodiversity and ecosystem services is limited to a few areas. The extent of biodiversity hotspots occupy 27% - 10% of the region, while the biodiversity decreased from 31.7% to zero in Bexar County.

Table 3.7. BES hotspots and overlap analyses in the SARB and the urban watersheds in Bexar County.

	1984		1995		2010	
Hotspot extent and overlap (ha, %)	SARB	Bexar County	SARB	Bexar County	SARB	Bexar County
Biodiversity	301,651 (27.8)	50,080 (31.7)	246,806 (22.7)	50,080 (31.7)	102,671 (9.5)	0.0 (0.0)
Carbon storage	258,228 (23.9)	73,588 (46.6)	277,347 (25.5)	73,588 (46.6)	276,857 (25.5)	50,080 (31.7)
Sediment retention	142,596 (13.1)	0.0 (0.0)	142,596 (13.1)	0.0 (0.0)	142,596 (13.1)	0.0 (0.0)
Biodiversity and carbon storage overlap	217,293 (20.0)	73,588 (46.6)	234,611 (21.6)	73,588 (46.6)	102,671 (9.5)	0.0 (0.0)
Biodiversity and sediment retention overlap	126,503 (11.7)	0.0 (0.0)	107,275 (9.9)	0.0 (0.0)	82,058 (7.6)	0.0 (0.0)

( ) denotes the percent overlap of BES hotspots ($p<0.1$) in the SARB and Bexar County.

The overlap analyses indicate the rapid decline in biodiversity in 2010. Likewise, carbon storage had an overlap with the biodiversity ranging from 20% to 9.5%, while sediment retention had an overlap with the biodiversity ranging from 11.7% to 7.6% in the SARB. This implies that BES are spatially linked and positively related over time. Thus, the results indicate that integration of BES can be effective as environmental policy alternatives for conversing and managing biodiversity and multiple ecosystem services simultaneously at multiple scales in the SARB.

3.3.7. Land Change Impacts on BES in the Four Sub-areas.

A Kruskal-Wallis test was conducted to determine if there were statistically significant differences in the mean ranks among four sub-areas in the SARB. The results showed that there were statistically significant differences ($p<0.01$) in terms of biodiversity, carbon storage, sediment retention, and sediment export for all three years (Table 3.8). The results indicate that these differences are mainly due to the dominant land classes and biophysical characteristics in the four sub-areas (i.e., upstream, suburban, urban, downstream watersheds), which most likely to affect spatio-temporal provision of biodiversity and ecosystem services.

The Mann Whitney U test for a post hoc test was conducted to determine where any differences lied between the two sub-areas in each analysis. The test showed that biodiversity scores of the upstream watersheds are statistically higher than those of the other three watershed groups for all three years ($p<0.01$), while biodiversity in the urban watersheds turned into the lowest in 2010 with mean rank of 26.41. The Mann Whitney U test for carbon storage indicates that the carbon storage of the upstream watersheds are statistically higher than those of the other three watershed groups for all three years ($p<0.01$), while carbon storage in the downstream watersheds was the lowest for all three years. In Mann Whitney U test for the sediment retention and export indicates that the upstream watersheds are statistically higher than those of the other three watershed groups for all three years ($p<0.01$). However, the other three watershed groups were not statistically significant each other in terms of sediment retention. On the other hand, sediment export in the downstream watersheds was the lowest for all three years ($p<0.01$) (Table. 3.8). Given urban expansion and land conversion are expected to continue into the north-western part of the SARB, the results indicate that upstream watersheds with Texas Hill Country characteristics should be prioritized for BES conservation policies to protect the biodiverse ecosystem services and to sustain well-being of residents in the region.

Table 3.8. Kruskal-Wallis test for biodiversity (a), carbon storage (b), sediment retention (c), and sediment export in the four sub-areas of the SARB.

(a)

Kruskal-Wallis test for biodiversity	Mean rank (1984)	Mean rank (1995)	Mean rank (2010)
1. Urban watersheds (n=16)	55.13	43.63	26.41
2. Suburban watersheds (n=22)	48.98	59.98	50.23
3. Upstream watersheds (n=20)	78.20	80.97	78.53
4. Downstream watersheds (n=49)	46.01	43.69	54.69
χ^2	16.01	23.12	25.49
p	p<0.01	p<0.01	p<0.01
Post hoc Mann Whitney U test	3 > 1, 2, 4	3 > 1, 2, 4 and 2 > 4	3 > 2, 4 > 1

(b)

Kruskal-Wallis test for carbon storage	Mean rank (1984)	Mean rank (1995)	Mean rank (2010)
1. Urban watersheds (n=16)	69.13	66.94	47.19
2. Suburban watersheds (n=22)	48.14	62.64	51.80
3. Upstream watersheds (n=20)	81.95	83.70	86.50
4. Downstream watersheds (n=49)	40.29	33.78	43.95
χ^2	30.38	43.61	27.95
p	p<0.01	p<0.01	p<0.01
Post hoc Mann Whitney U test	3 > 2, 4 and 1 > 2, 4	3 > 2, 4 and 1, 2 > 4	3 > 1, 2, 4

(c)

Kruskal-Wallis test for sediment retention	Mean rank (1984)	Mean rank (1995)	Mean rank (2010)
1. Urban watersheds (n=16)	46.97	46.88	46.41
2. Suburban watersheds (n=22)	52.34	52.43	52.34
3. Upstream watersheds (n=20)	93.83	93.78	93.80
4. Downstream watersheds (n=49)	40.79	40.80	40.80
χ^2	42.70	42.62	42.65
p	p<0.01	p<0.01	p<0.01
Post hoc Mann Whitney U test	3 > 1, 2, 4	3 > 1, 2, 4	3 > 1, 2, 4

(d)

Kruskal-Wallis test for sediment export	Mean rank (1984)	Mean rank (1995)	Mean rank (2010)
1. Urban watersheds (n=16)	56.25	66.00	68.56
2. Suburban watersheds (n=22)	55.36	47.07	56.89
3. Upstream watersheds (n=20)	88.38	88.68	90.43
4. Downstream watersheds (n=49)	38.62	39.04	33.08
χ^2	36.74	39.89	53.60
p	p<0.01	p<0.01	p<0.01
Post hoc Mann Whitney U test	3 > 1, 2, 4 and 1, 2 > 4	3 > 1, 2, 4 and 1 > 4	3 > 1, 2, 4 and 1, 2 > 4

3.3.8. Urban Sprawl Impacts on BES.

I conducted the Wilcoxon signed-rank test to evaluate whether the four sub-areas experienced statistically significant changes in terms of urban sprawl and BES estimates between the pre-NAFTA (1984-1995) and the post-NAFTA (1995-2010) periods in the SARB. The results indicate that there were statistically significant differences in each sub-area on the BES estimates during the two periods (Table 3.9, Table B12). Most notably, the change in low density urban class was statistically significant in three watershed groups (i.e., suburban, upstream, and downstream) ($p<0.01$), which means the low density urban expansion or urban sprawl occurred during the post-NAFTA (1995-2010) period. The results indicate that low density urban sprawl is prevalent beyond the boundary of the urban watersheds in the SARB. Furthermore, suburban watersheds experienced the most significant changes in all the BES estimates compared to other three watershed groups (i.e., urban, upstream, and downstream) ($p<0.01$), which also means suburban watersheds are the most vulnerable to land changes in the region.

The results confirm that there was a significant increase of urban classes mostly in the periphery of urban center during the NAFTA period; simultaneously biodiversity and ecosystem services were significantly lost and impaired along the rural-urban gradient. Thus, the analyses empirically demonstrate the current sprawling development patterns in the rapidly urbanizing San Antonio region negatively impacted the integrity of biodiversity and functioning of ecosystems.

Table 3.9. Wilcoxon signed rank test for two urban classes and BES between the pre- and post-NAFTA periods.

	Urban watersheds		Suburban watersheds		Upstream watersheds		Downstream atersheds	
Wilcoxon signed rank test	z	p	z	p	z	p	z	p
Low density urban post NAFTA - pre NAFTA	-1.581	0.119	-3.937	0.000**	-3.762	0.000**	-4.080	0.000**
High density urban post NAFTA - pre NAFTA	-0.801	0.440	-3.571	0.000**	-1.134	0.453	-1.000	0.317
Biodiversity post NAFTA - pre NAFTA	-0.535	0.614	-3.736	0.000**	-2.237	0.025*	-2.738	0.005**
Carbon storage post NAFTA - pre NAFTA	-2.792	0.003**	-4.075	0.000**	-1.923	0.054	-5.272	0.000**
Sediment retention post NAFTA - pre NAFTA	-0.369	0.833	-2.994	0.002**	-2.553	0.009**	-0.290	0.772
Sediment export post NAFTA - pre NAFTA	-0.122	0.897	-3.210	0.001**	-0.255	0.009**	-0.145	0.884

* $p < .05$, ** $p < .01$

3.3.9. Environmental Justice Analysis in Bexar County

The EJ analysis in this dissertation disaggregates BES beneficiaries into different groups in terms of ethnicity, geography, and socio-economic status in a spatially explicit way. In Bexar County, Hispanic population consists of 58.7% and White consists of 30.3% and African American consists of 7.5%, respectively according to 2010 Census in Bexar County (USCB, 2010) (http://www.census.gov/quickfacts/). I first attempt to apply the quantitative BES estimates to directly relate to race percentage in each census tract in Bexar County in order to address the environmental justice in a tightly coupled social-ecological context.

The results of spatial analysis regarding the distribution of social-ecological variables from each census tract in Bexar County indicates that the BES benefits in Bexar County is not distributed equitably from the perspective of environmental justice and the long-term sustainable development in the region. Most environmental benefits in terms of BES estimates and NDVI are concentrated in the northern part of Bexar County and the EARZ, where most high median income communities and high percentage of white population lives (Figure B2). On the other hand, neighborhoods with higher percentage of Hispanic residents with higher poverty rate and higher unemployment rate are exposed to higher environmental hazards, such as transportation related diesel particulate matter, total respiratory hazard index (HI), and total cancer risk from air toxics compared to White and African American population. The results indicate that Hispanic population is negatively correlated to BES benefits and lower socioeconomic status ($p<0.01$), while positively correlated to environmental hazards ($p<0.01$), which is opposite results of White population (Table 3.10, Table B13).

Table 3.10. Nonparametric Spearman's correlation of race/ethnicity percentage with socio-ecological variables in Bexar County.

Socio-ecological variables	Hispanic	White	African American
Biodiversity	-0.638**	0.638**	0.125*
Carbon storage	-0.720**	0.750**	0.108*
Sediment retention	-0.309**	0.284**	0.272**
Normalized Difference Vegetation Index	-0.533**	0.601**	0.092
Median household income	-0.746**	0.805**	0.091
Poverty rate	0.719**	- 0.773**	- 0.089
Unemployment rate	0.544**	- 0.601**	- 0.013
Diesel particulate matter	0.376**	- 0.405**	- 0.034
Total respiratory hazard index (HI)	0.205**	- 0.220**	- 0.114*
Total cancer risk from air toxics	0.264**	- 0.299**	- 0.052

** Correlation is significant at the 0.01 level (2-tailed)

* Correlation is significant at the 0.05 level (2-tailed)

3.4. Discussion

In this research I quantified spatio-temporal changes in BES estimates at multiple scales in the SARB and examined their dynamic interactions and spatial relationships in terms of the ecological production function method (EPFM) using InVEST. The research contributes to advancing the valuation of ecosystem services and implementation of the ecosystem approach (EA) by presenting novel findings compared to the benefit transfer method (BTM) by Costanza et al. (2014) and informing policy for socio-ecological sustainability. No studies have been conducted in the SARB to analyze the impacts of land change on ecosystem services. This dissertation fills this knowledge gap through a land-change analysis that provides useful information for spatio-temporal land change patterns in the SARB including the environmental sensitive Edward Aquifer Recharge Zone (EARZ).

The results indicate that the urban expansion around the San Antonio area occurred mostly at the expense of woodlands/forests and ecosystem service values decreased substantially, in particular, since the North American Free Trade Agreement (NAFTA) was enacted in 1994. The result is consistent with Nowak et al. (2005) who reported that most of the urban expansion across the United States occurred in forested land between 1990 and 2000. Notably, the expansion of low-density urban has been concentrated around the San Antonio area, reflecting the sprawling nature of urban development in the region (Yi et al., 2017).

Renard et al. (2015) emphasize that spatio-temporal provisions of multiple ecosystem services should be taken into account to better understand the complex dynamics in multiple ecosystem services and to identify the future trajectories for human well-being. In this context the research analyzes changes in spatial and temporal associations between biodiversity and ecosystem services in terms of habitat quality, carbon storage, and sediment retention in the SARB and in Bexar County. The results from nonparametric correlation analyses in Table 3.6 indicate that synergistic relationships between the biodiversity–carbon storage pair and between the biodiversity–sediment retention pair are statistically significant ($p<0.01$) over time in the SARB and in Bexar County. The historical correlation between the carbon storage–sediment retention was also statistically significant ($p<0.01$ in the SARB; $p<0.05$ in Bexar County).

The normalized diagrams for the BES estimates in Figure 3.6 illustrate that the rapid decline for carbon storage and biodiversity is identified in urbanizing Bexar County compared to the SARB and the temporal dynamics for the BES. On the other hand, sediment retention service is relatively stable compared to biodiversity and carbon storage both in the SARB and Bexar County. The hotspot overlap analysis indicates the rapid decline in biodiversity in 2010. Carbon storage had an overlap with the biodiversity ranging from 20% to 9.5%, while sediment retention had an overlap with the biodiversity ranging from 11.7% to 7.6% in the SARB. These results are consistent with the findings of Bai et al. (2011) who reported the synergistic relationship in the conservation of multiple ecosystem services.

The sensitivity analyses indicate that the provision of carbon stocks is the most sensitive to forest cover and is significantly linked with biodiversity loss in the SARB.

Moreover, the rates of biodiversity loss and carbon storage degradation have accelerated since the NAFTA went into effect in 1994, and the declining trends are negatively related to the urban sprawl in the San Antonio region. The results from the Wilcoxon signed rank test in Table 3.9 demonstrate that the degradation of biodiversity and ecosystem services is mainly associated with low density urban sprawl around the San Antonio area. Especially, suburban watersheds experienced the most significant changes in environmental degradation ($p<0.01$) between the pre- and post-NAFTA periods in the BES estimates compared to other watershed groups (i.e., urban, upstream, and downstream), which means these areas were significantly impacted and impaired in the form of urban sprawl along the urban-rural interface.

The research employs EPFM using InVEST to quantify the BES and examine the changes in spatial associations among biodiversity, carbon storage, and sediment retention in the rapidly urbanizing San Antonio River Basin. The results indicate the rate of biodiversity loss and carbon storage degradation have accelerated since the North American Free Trade Agreement (NAFTA) implementation in 1994, especially in the highly urbanized watersheds. The results also demonstrate the varying impact of urban sprawl on biodiversity and ecosystem service across urban-rural gradient. Moreover, to characterize the detailed spatio-temporal relationship between biodiversity and ecosystem services, this study attempts to investigate the ecological relationships in terms of nonparametric correlation and hotspot overlap analysis, which have not been analyzed before at multiple scales over time. The findings indicate the synergistic spatial associations between biodiversity and ecosystem services in the SARB and Bexar County, respectively.

On the other hand, the accelerating rate of biodiversity loss and ecosystem service degradation is of particular concern to the region's future sustainability. Carbon storage within ecosystems has the effect of removing carbon dioxide (CO_2) from the atmosphere (UNFCCC, 2015). For example, forests have the ability to remove and store carbon from the atmosphere through the process of photosynthesis. When the natural vegetation cover is converted to agriculture or urban land, carbon is released to the atmosphere as carbon dioxide, exacerbating climate change. Carbon storage in the 1995-2010 period significantly decreased compared to the 1984-1995 period, amounting to the loss of 236.7 and 126.7 million US$ in the SARB and Bexar County, respectively. Considering carbon storage is important for climate regulation in the urban landscape, carbon market, payments for these ecosystem services should be considered to facilitate the low carbon and climate resilient economy (UNEP, 2011). In this context, Nelson et al. (2009) suggest the payments for carbon sequestration to moderate the trade-off between different development policies and biodiversity and ecosystem services.

EPFM using InVEST produces robust valuation of quality of habitat and supplies of ecosystem services at the local and regional scales compared to BTM. The results from EPFM indicate that carbon storage degradation has accelerated since the North American Free Trade Agreement (NAFTA) implementation in 1994, which is directly related to climate regulation. EPFM also demonstrate the significant negative impacts of urban sprawl on biodiversity and ecosystem services. To illustrate, the monetary valuation based on InVEST results indicates considerable carbon loss in terms of dollar values in both the SARB and the urban Bexar County. On the contrary, the benefit transfer method (BTM) using the parameter from Costanza et al. (2014) in the same areas showed the increasing trend in terms of dollar values of climate regulation functions at multiple scales (Yi et al., 2017). On the other hand, the EPFM requires many assumptions and considerable investment in terms of time and data acquisition with financial constraints to fully capture the available information of ecosystem functioning in the study area (Maes et al. 2012; Ruckelshaus et al., 2015). Admittedly, there are a range of methodologies to value the biodiversity and ecosystem services across the scales. However, there is no established consensus about the best method for the valuation of biodiversity and ecosystem services. Thus, the advantages and drawbacks of each valuation strategy should be considered carefully based on the type of biodiversity and ecosystem services across time and space (Daly and Farley, 2010).

Findings from this study shed new light on urban ecosystem service (UES), ecosystem approach (EA), and ecosystem based management (EBM), which provide an improved understanding of dynamic interaction in a tightly coupled social and ecological context and suggest reorientation of environmental policy interventions for policy makers, researchers, and stakeholders (Ruckelshaus et al., 2015). The results from land change analysis in the Edward Aquifer Recharge Zone (EARZ) indicate that the proportion of the urban land increased six-fold, which is more than double the rate of urban growth in the SARB as a whole and in Bexar County. Forest declined the most during the urban expansion among land classes. Given forest ecosystems in the EARZ mediate air and water flows, groundwater recharge and contributes to biodiversity and regional climate stability, diminishing forest cover poses increasing challenges for biodiversity and ecosystem protection especially for water quantity and quality in the recharge zone. Importantly, according to the analysis, the proportion of impervious surfaces in the EARZ exceeded the 10% threshold by 2010.

The Edward Aquifer provides positive ecological functions, such as habitat for endangered species, including the nine karst invertebrates in Bexar County (USFWS, 2012). In the north-western area of the basin, the Golden-cheeked Warbler (*Setophaga chrysoparia*) and the Black-capped Vireo (*Vireo atricapillus*) have been identified as endangered by the State of Texas due to urban development and habitat loss. Forest loss and fragmentation are one of the primary reasons for the decline in population of endangered species, and urban land uses have important impacts on bird populations (Engels and Sexton 1994; TPWD, 2003). Furthermore, the environmentally sensitive EARZ will be more prone to flash flooding and groundwater contamination due to the proliferation of impervious surfaces (GEAA, 2014; Flood safety, 2015). The results provide the rationale for collective action to internalize these negative externalities associated with urban sprawl and increasing social costs to protect BES from environmental degradation in the region. Likewise, urgent policy interventions should be implemented to address the urban expansion in the Edward Aquifer Recharge Zone, which satisfies most of San Antonio's water needs and is critically important for the quantity and quality of groundwater supplies.

The San Antonio region is the significant multimodal transportation hub of the North American Free Trade Agreement (NAFTA). Land change in this region has been significantly associated with the development of a transportation network. To illustrate, seven highway corridors, such as IH-35, 10, 20, 30, US 59, US 281, and US77, in Texas carry more than 80 percent of all NAFTA trade. Among them, the IH-35 corridor carries the largest portion of NAFTA truck vehicle miles traveled (VMT) at over 35 percent of the total Texas NAFTA VMT connecting San Antonio to Laredo and other southern border areas in the near future. Moreover, it is estimated that NAFTA trade tonnage to and through Texas will be more than double and the value of trade is expected to increase by 280 percent by 2030 (Cambridge Systematics Inc., 2007). This means that the continued gray infrastructure investment is expected for the management and expansion of numerous highways in the region. These expanding transportation networks will likely further degrade the ecosystem services in the region through land change, air pollutant emissions and water contamination (American Forests, 2002; AACOG, 2015).

Increased impervious surfaces cause hydrologic flow changes, detrimental ecological consequences, such as habitat degradation, impairment of aquatic communities, and poor water quality by the delivery of pollutants to the stream, which include sediment, pesticides, organic pollutants, oil and grease (Paul and Meyer 2001). In addition, urban stream is vulnerable to nonpoint source pollution (NPS), contaminated sewer infrastructure, and sedimentation from construction. Schueler (1994) found that streams whose watersheds have over 10 percent impervious cover showed significant degradation, proposing 10 percent impervious land cover as a maximum level of imperviousness to maintain the health of the stream. The results indicate that the urban proportion in the EARZ exceeds 10% criteria, and unprecedented growth in the EARZ as well as the SARB is expected to continue at the expense of forest and rangeland by the forces of current economic NAFTA policy and population growth under the development pressure. Thus, reducing impervious surfaces is critical to reduce the carbon footprint and mitigate the climate change and land fragmentation in the region.

The rapid expansion of low density urban class in the SARB is closely related to the urban sprawl beyond the boundary of the urban center in Bexar County. The analysis shows that the SARB experienced significant suburban expansion. This expansion of low-density urban growth is creating more widespread impact on the biodiversity and ecosystem services, especially those provided by rangelands and woodlands/forests in the SARB (Table 3.9). Urban sprawl is characterized by negative externalities, such as increasing automobile dependence, transportation network, gray infrastructure, the spatial segregation in terms of socio-economic status, and loss of environmental qualities (Ewing et al., 2003; Bhatta, 2010). San Antonio is a prime example of urban sprawl with a growing population moving further outward. Because urban sprawl is strongly related with higher consumption of energy, urban heat island effect, and GHG emissions, San Antonio region is exposed to increasing risks of air pollution and potential high ground level ozone coupled with NAFTA transportation corridors. Importantly, this points to the growing health concerns to ensure the region meets national air quality standards and to protect the public health and the intact environment (AACOG, 2015). Accordingly, these negative socio-economic effects of sprawl should be considered as market failures in that the resulting traffic congestion, public health concerns, and carbon locked-in urban growth are neither sustainable nor economically efficient from the perspective of sustainable development. For example, NAFTA induced economic development in the region has been increasingly locked-into fossil fuel and gray infrastructure-based transportation networks through the path dependent and urban expanding processes driven by the economy of scale with increasing returns to scale.

The mitigation and adaptation of climate change are closely linked to integrated policy interventions and strategies (Gill et al., 2007). The urban green spaces (UGS) contribute significantly to ecosystem services with the role of urban green infrastructure in terms of clean air and water provision, biodiversity and habitat provision for wildlife, reduction of greenhouse gas (GHG) emissions, microclimate regulation, stormwater runoff reduction, and recreation (Bolund and Hunhammar, 1999; Morancho, 2003; McPherson et al., 2005). The findings suggest that the policy interventions reoriented for low carbon and climate resilient development should be implemented to mitigate the rapid biodiversity loss and ecosystem degradation and to internalize the negative externalities of urban sprawl. Such policies also contribute to mitigating disparities in the allocation of biodiversity and ecosystem services for human health and quality of life in the rapidly urbanizing watershed, in that humans are recognized as one of the critical components of ecosystem (Secretariat of the Convention on Biological Diversity, 2004). Given urban green space (UGS) provides multifunctional benefits in terms of recreation, ecological function, aesthetic value, public health, and social interaction, the concept of green infrastructure presents growing importance to improve the public health and environmental justice for the ecosystem based management (EBM) strategy in flexible and cost-effective ways (EEA, 2011; Pataki et al., 2011; USEPA, 2013; Hansen and Pauleit, 2014). Human well-being are tightly coupled with ecosystem services and the sustainable development only occurs within the integration of ecological, social, economic, and institutional perspectives. Urban expansion comes at the expense of many other ecosystem services and biodiversity. Thus, a regional payment for ecosystem services (PES) programs for carbon sequestration and watershed, and market based carbon pricing and taxes can help mitigate the unintended environmental costs from urban sprawl and land uptake in the region. Compact and infill development should be used to mitigate the negative effects of land fragmentation in the SARB.

The environmental justice analysis in Bexar County, based on the BES estimates and the American Community Survey (2010) and National Air Toxics Assessment (2015) census tract socio-economic and health risk characteristics, indicates that neighborhoods with a higher percentage of Hispanic population are positively correlated with lower BES, lower Normalized Difference Vegetation Index, lower median household income, higher poverty rate, and significantly greater exposure to public health risks including diesel particulate matter, total respiratory hazard index (HI), and total cancer risk from air toxics. By contrast, neighborhoods with a predominantly White population were found to have the opposite correlation ($p<0.01$). These results are consistent with the findings that examined distributive inequality associated with vegetative cover, air pollution, and the proximity or accessibility of urban green space

(UGS) from the perspective of environmental justice (Comber et al., 2008; Wolch et al., 2014; Jennings et al., 2016). The findings indicate that there is an ethnically unequal distribution of ecosystem service benefits and social costs of land change that segregates predominantly Hispanic and White populations in the San Antonio region.

The findings from this study contribute to understanding dynamic relationships between land changes and BES estimates and inform research on the impacts of urbanization on the environment with spatially explicit data and information. However, it is still challenging to understand the spatial distributions and the relationships of multiple ecosystem services (Qui and Turner, 2013). More place-based studies with sensitivity and robustness analyses are needed to further consider the dynamic interactions between biodiversity and ecosystem services. Collecting spatially explicit local data and information on many of the parameters would be a next step to further test these spatio-temporal relationships.

There are many alternate measures of BES estimates that could be used for future research in a variety of decision-making processes. However, there is no perfect way to estimate the shadow (or true economic) value of biodiversity and ecosystem services. For example, qualitative local research could be a useful approach to complement quantitative valuation methods and would allow for a more comprehensive and augmented understanding of the relationship between BES to enhance the human well-being in a sustainable way. Future research should use data at different spatio-temporal scales to more accurately capture the impact of land change on ecosystem services. Incorporating the impact of population and economic growth on land and ecosystem services in the valuation model can also be considered a future research avenue. In addition, different classification methods of remotely sensed imagery as well as finer spatial and temporal resolution of the imagery may increase the classification accuracy to analyze land change and its impacts.

Given the complex and dynamic relationship among biodiversity and multiple ecosystem services at various scales, holistic conservation policies against biodiversity and ecosystem service loss are urgently needed to mitigate the downward spiral of biodiversity loss and ecosystem degradation and internalize the negative externalities of urban sprawl. In this regard, decision-making processes should focus on the integration of biodiversity and ES provision, considering adaptive management, non-linearity, time-lags of ecosystem processes, equitable sharing of BES benefits, stakeholder involvement, and international cooperation for BES conservation in the region. Policy interventions should also focus on low carbon and climate resilient infrastructure, urban green space, and inclusive development for sustainable use of biodiverse ecosystem services.

3.5. Conclusion

In this study I first examined the changes in relationships among biodiversity, carbon storage, and sediment retention in the rapidly urbanizing watersheds at multiple scales over time in the SARB and in Bexar County. The results indicate that the substantial impact of land changes on BES occurred from 1984 to 2010 across 107 subwatersheds. The declines in spatio-temporal biodiversity mirror carbon storage losses, especially in the urban subwatersheds in Bexar County. The results indicate that overall biodiversity and ecosystem services are spatially linked and positively correlated in the SARB and in Bexar County, improving our understanding of historical BES dynamics for the future trajectory of sustainable use of biodiverse ecosystem services.

Importantly, the results confirms the finding of Yi et al. (2017) for the overestimation of ESV of Costanza et al. (2014) and highlight the decreasing pattern of carbon sequestration, which is opposite trend in terms of the increasing pattern of climate regulation from BTM in the SARB. The results from EPFM indicate that the urban sprawl in suburban watersheds around Bexar County significantly impacted the loss of biodiversity and ecosystem degradation during the post NAFTA (1995-2010) period compared to other three subwatersheds. Thus, the potential threats to long-term sustainability in the SARB are linked to the proliferation of developed areas and impervious surfaces and the increasing level of social costs at the expense of natural resources and well-functioning ecosystem services. Considering the positive and strong relationship between biodiversity and ES, the findings indicate that land use policy and decision-making processes for suburban watersheds are particularly important in managing the SARB for the future along the urban-rural gradient. Finally, the quantification of biodiversity and ecosystem services and spatial analysis in a coupled social-ecological context significantly contribute to advancing ecosystem based management for socio-ecological resilience and sustainability as a methodological complement. I believe this integrative approach will not only improve our understanding of the critical role of biodiversity and ecosystem services but also capture the socio-ecological connotations across time and space.

CHAPTER IV

CONCLUSIONS

This chapter provides a synthesis of the main findings and implications of the research, contributions of the dissertation research, limitations and future work. The overarching goal of the research was to provide a comprehensive and comparative investigation of spatio-temporal land change impacts on biodiversity and ecosystem services, utilizing two main valuation approaches, the Benefit Transfer Method (BTM) and the Ecological Production Function Method (EPFM), in the San Antonio River Basin, Texas, from 1984 to 2010.

4.1. Main Findings and Implications of the Research

The results from the BTM suggest that the value placed on urban areas in the 2014 publication substantially overestimates the ecosystem service values of urban space in terms of climate regulation and recreation (Yi et al., 2017). By contrast, the results from the EPFM confirms the overestimation of ESV of Costanza et al. (2014), indicating that the degradation of carbon storage function has accelerated since the implementation of NAFTA in 1994, which is intrinsically related to the function of climate regulation. The EPFM also demonstrates the significant negative impacts of urban sprawl on biodiversity and ecosystem services.

Objective 1 – Spatio-temporal analysis of land change during a period of rapid economic development and urbanization-driven change within the SARB: The research employed multi-temporal Landsat 5 TM data to examine the land-change patterns at two scales; SARB and Bexar County. The results indicate that more forest land and rangeland was lost to low-density urban areas than to high-density urban areas. These patterns in land change emphasize that the low-density urban land was growing at a more rapid rate than high-density urban land in the SARB. However, at the smaller spatial scale of analysis in Bexar County, losses of forest cover and rangelands to the high-density and low-density urban areas were more even.

The results also indicate that the rate of low-density urban expansion accelerated around the San Antonio area after the North American Free Trade Agreement (NAFTA) went into effect, reflecting the sprawling nature of urban development in the region. The growth in low-density urban development is creating more widespread impacts on the delivery of ecosystem services, especially those provided by rangelands and woodlands/forests. These results are consistent with the findings that the San Antonio area in Bexar County experienced one of the greatest losses in forest in the southern USA between 2001 and 2006 (World Resources Institute, WRI, 2011) and that land change affecting forests since 1990 have been mainly centered in southern US posing significant threats to ecosystems (Alig et al., 2004).

Objective 2 – Multiscalar analysis of changes in ecosystem service values using BTM with two sets of valuation coefficients at multiple scales: The research analyzed spatio-temporal changes in ecosystem service values and functions and applied the BTM using two sets of valuation coefficients at two spatial scales, SARB and Bexar County. The results from sensitivity analyses indicate that estimated ecosystem service values for both scales of analysis are relatively inelastic (i.e., CS substantially <1), which suggests that the estimates of ecosystem service values appear to be relatively robust for all three years of analysis (1984, 1995 and 2010).

The contributions of each ecosystem function to the overall ESV in the SARB and Bexar County were also quantitatively compared. At both spatial scales, the value of individual ecosystem services was much higher when the 2014 modified value coefficients rather than the 1997 coefficients were applied, but the difference varied substantially among ecosystem services. Regardless of the value coefficients used, the patterns of temporal change in the values of ecosystem functions declined during the 26-year study period, with two exceptions, recreation and climate regulation. The results suggest the value placed on urban areas in the 2014 publication (which was taken from a single case study that focused primarily for large urban parks and not urban areas as a whole), substantially overestimates the ESV of urban space. The application of this value coefficient, even only to urban green space, led to the improbable conclusion that urbanization had a positive overall effect on the delivery of ecosystem services.

Objective 3 – Multiscalar analysis of spatial and temporal associations between biodiversity and ecosystem services using EPFM: The research analyzed changes in spatial and temporal associations between biodiversity and ecosystem services in terms of habitat quality, carbon storage, and sediment retention in the SARB and in Bexar County. The results from nonparametric analyses indicate that the correlations between the biodiversity–carbon storage pair and between the biodiversity–sediment retention pair are statistically significant ($p<0.01$) at both spatial scales of analysis. The correlation between the carbon storage–sediment retention was also statistically significant ($p<0.01$ in the SARB; $p<0.05$ in Bexar County). The normalized spider diagrams for BES estimates show that the rapid decline in carbon storage and biodiversity is identified in urbanizing Bexar County compared to the SARB. On the other hand, compared to biodiversity and carbon storage, sediment retention was found to be relatively stable during the 26-year period of the study both in the SARB and Bexar County. The hotspot and overlap analyses indicate the rapid decline in biodiversity in 2010. These results are consistent with the findings of Bai et al. (2011) who reported the synergistic relationship in the conservation of multiple ecosystem services.

The sensitivity analyses indicated that the provision of carbon stocks is most sensitive to forest cover and significantly linked with biodiversity loss in the SARB. Moreover, rates of biodiversity loss and carbon storage degradation have accelerated since the NAFTA went into effect in 1994, and the declining trends are negatively related to the urban sprawl in the San Antonio region. The nonparametric Wilcoxon signed rank test indicates that suburban watersheds at the urban-rural interface experienced the most significant change in environmental degradation ($p<0.01$) between the pre- and post-NAFTA periods in the SARB and demonstrates the environmental impact of urban sprawl on BES over time.

The results from the environmental justice analysis indicate that there is an ethnically unequal distribution of ecosystem service benefits and social costs of land change degradation that segregates predominantly Hispanic and White populations in the

San Antonio region. Furthermore, the results advance the understanding of the distributive inequality associated with BES estimates, vegetative cover, air pollution, and the proximity or accessibility of urban green space (UGS) in a socio-ecological context (Comber et al., 2008; Wolch et al., 2014; Jennings et al., 2016).

4.2. Contributions of Dissertation Research

By presenting novel findings and informing policy for socio-ecological sustainability, this dissertation research contributes to the advancement of ecosystem services valuation and implementation of an inclusive ecosystem management approach. First, no studies have been conducted in the whole of the SARB to analyze the impacts of land change on ecosystem services. The research presented in this dissertation fills that knowledge gap through a land-change analysis that provides useful information for spatio-temporal land change patterns in the SARB. Importantly, the analysis shows that the urban expansion, especially low-density suburban sprawl, around San Antonio occurred mostly at the expense of woodlands/forests and ecosystem service values decreased particularly since NAFTA was enacted in 1994. The result is consistent with Nowak et al. (2005) who reported that most of the urban expansion across the United States occurred in forested land between 1990 and 2000.

The results from land change analysis in the Edward Aquifer Recharge Zone (EARZ) indicate that the proportion of the urban land increased six-fold, which is more than double the rate of urban growth in the SARB as a whole and in Bexar County. Forest declined the most during the urban expansion among land classes. Given forest ecosystems in the EARZ mediate air and water flows, groundwater recharge and contributes to biodiversity and regional climate stability, diminishing forest cover poses increasing challenges for biodiversity and ecosystem protection especially for water quantity and quality in the recharge zone. Importantly, according to the analysis, the proportion of impervious surfaces in the EARZ exceeded the 10% threshold by 2010.

The associated proliferation of impervious surfaces makes the environmentally sensitive EARZ more prone to flash flooding and groundwater contamination (GEAA, 2014; Flood safety, 2015). These results provide the rationale for collective action to internalize the negative externalities associated with urban sprawl and increasing social costs to protect BES from environmental degradation in the region. Likewise, the results also point to the urgent need for policy interventions to address further urban expansion in the EARZ, which affects the quantity and quality of groundwater recharge in the Edwards Aquifer upon which San Antonio depends almost exclusively for its water supply.

Second, the ability to confidently use value coefficients when applying the BTM to estimate ecosystem service values demands rigorous assessments of their broad applicability. The research presented in this dissertation is the first to compare the effects of two sets of widely cited ecosystem-service valuation coefficients, published in 1997 and 2014, on the estimates of changes in ecosystem services values over time. Moreover, the urban coefficient from the 2014 publication was modified for low-density and high-density urban areas to more robustly characterize urbanization-related changes of ecosystem service values. The results from the sensitivity analyses provide valuable new knowledge associated with scale dependence of the validity of value coefficients when BTM analyses are conducted to evaluate the ecosystem service values.

This underscores the importance of ensuring that the transferred unit value derived from the primary evaluation study is compatible with respect to the scale and characteristics of the study site, in order to avoid misinterpretation of land change effects on the value of ecosystem services delivered. The research results presented in this dissertation indicate that the revised urban coefficient reported in Costanza et al. (2014) is highly overestimated. Thus, to more comprehensively and accurately characterize potential effects of development polices, such as NAFTA, on the delivery of ecosystem services in affected areas, more place-based studies are needed to improve the ESV coefficients for different categories of urban areas at regional and local scales.

Third, the research employs the EPFM to quantify the BES and examine the changes in spatial associations among biodiversity, carbon storage, and sediment retention in the rapidly urbanizing SARB. The results indicate the rate of biodiversity loss and carbon storage degradation has accelerated since the implementation of NAFTA in 1994 especially in the highly urbanized watersheds. The results also demonstrate the varying impact of urban sprawl on biodiversity and ecosystem service across urban-rural gradient. Moreover, to characterize the detailed spatio-temporal relationship between biodiversity and ecosystem services, this study was novel in applying nonparametric correlation, and the hotspot and overlap analyses to investigate ecological relationships at multiple scales over time. The findings indicate synergistic spatial associations between biodiversity and ecosystem services in the SARB and in Bexar County.

This study is also the first to investigate how BES estimates are spatially unevenly correlated with socio-economic variables in Bexar County. Despite the significant potential of ecosystem services to address environmental justice issues for better socio-ecological decision-making, the majority of quantitative studies focus on social inequalities of anthropogenic air pollution and hazardous waste risks. However, few have explored the direct link between BES and socio-economic characteristics. The dissertation research addresses this novel and under-examined aspect of the uneven spatial distribution of changes in BES benefits. Thus, the study contributes to understanding of the unequal distribution of environmental benefits and risks in a tightly coupled social and ecological context. In addition, this approach is transferable to other rapidly urbanizing watersheds to disaggregate BES beneficiaries.

4.3. Limitations and Future Work

Numerous indirect valuation methods have been developed for public goods that are subject to market externalities, such as *in situ* ecosystem services (Farber et al., 2002; Costanza, 2008). As mentioned previously, the limitation of BTM is mainly related to unit value coefficients. On the other hand, the EPFM requires many assumptions and considerable investment in terms of time and data acquisition. This dissertation research on spatio-temporal relationships among BES using InVEST may produce potentially uncertain and biased outputs because the analyses rest on the assumptions embedded in the parameterization of InVEST. More place-based studies with sensitivity and robustness analyses are needed to further consider the dynamic interactions between biodiversity and ecosystem services. Collecting spatially explicit local data and information on many of the parameters would be a next step to further test these spatio-temporal relationships.

There are many alternate measures of BES estimates that could be used for future research in a variety of decision-making processes. However, there is no perfect way to estimate the shadow (or true economic) value of biodiversity and ecosystem services. For example, qualitative local research could be a useful approach to complement quantitative valuation methods and would allow for a more comprehensive and augmented understanding of the relationship between BES to enhance the human well-being in a sustainable way. Future research should use data at different spatio-temporal scales to more accurately capture the impact of land change on ecosystem services. Incorporating the impact of population and economic growth on land and ecosystem services in the valuation model can also be considered a future research avenue. In addition, different classification methods of remotely sensed imagery, as well as finer spatial and temporal resolution of the imagery may increase the classification accuracy to analyze land change and its impacts.

Finally, there are increasing environmental concerns since the Eagle Ford Shale development in the southern part of the SARB. Continued economic development associated with the oil and gas productions is expected to increase biodiversity loss and ecosystem degradation coupled with changes in public health and human well-being. Thus, combining the oil and gas productions with the land change would provide a more integrated interpretation of BES change in the region. Future research should include this environmental change in the Eagle Ford Shale to accurately characterize the dynamic and complex interactions between land change and BES.

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APPENDIX A

CHAPTER II SUPPLEMENTARY MATERIAL*

A.1. Land-change analysis

A.1.1. Preprocessing

We utilize the Standard Terrain Correction (Level 1T, precision and terrain correction; WGS 1984, UTM zone 14N) Landsat product, which is pre-processed with a systematic geometric correction/accuracy via incorporation of ground control points (GCPs) and digital elevation model (DEM) data (USGS, 2011). We use Landsat TM radiometric calibration coefficients given in Chander et al. (2009), and employ the Fast Line-of-sight Atmospheric Analysis of Spectral Hypercube (FLAASH®) radiative transfer model (Perkins et al., 2005) to remove atmospheric absorption and scattering effects. This facilitates multitemporal analysis of the resultant surface reflectance (Jensen, 2005). Furthermore, for each image date, we stack an ancillary Normalized Difference Vegetation Index (NDVI) (Rouse et al., 1974) image with its corresponding multispectral Landsat TM image from which it is derived to enhance detection of vegetation (Gamon et al., 1995).

* Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.ecolecon.2016.11.019.

A.1.2. Accuracy assessment

We assess the accuracy of the Landsat-derived land classifications based on manual/visual interpretation of aerial photographs obtained from the USGS (http://earthexplorer.usgs.gov) (U.S. Geological Survey (USGS), 2014), as well as temporally-proximal NLCD data (http://www.mrlc.gov/) (U.S. Geological Survey (USGS), 2015), when available. We geometrically rectify a total of 28 aerial photographs acquired in 1981 and 1983 via an average of 78 GCPs for each photo (minimum and maximum number of GCPs were 56 and 120, respectively) and a first-order polynomial transformation, with root-mean-square error (RMSE) values less than 0.5 pixel (pixel size = 5 m) for every image. We use these images as reference data for classified Landsat image data in 1984. Seven of these aerial photographs are collected in 1981, whereas 21 are acquired in 1983, and the aerial photos are systematically and regularly distributed across the SARB study area. In addition, we utilize county orthophoto mosaics from the U.S. Department of Agriculture (USDA) Geospatial Data Gateway (https://gdg.sc.egov.usda.gov) at one-meter spatial resolution (i.e., USDA-Digital Ortho County Mosaics) and NLCD data to evaluate the thematic accuracy of the 1995 and 2010 classified images.

The overall classification accuracy, based on confusion matrices (Congalton and Green, 1999; Jensen, 2005), are 85.11%, 87.33%, and 85.78% for the 1984, 1995, and 2010 images, respectively; the corresponding Kappa (KHAT) accuracies are 83.25%, 85.75%, and 84.00%, respectively, where the KHAT statistic is an estimate of KAPPA, which is a measure of agreement between the classified image and the reference data

(Congalton and Green, 1999). Producer’s and user’s accuracies are generally relatively high across classes and image-acquisition dates, with some variability. For example, over the study time period, producer’s accuracy for low-density urban varies between 76.00% and 94.59%, whereas the values for high-density urban are somewhat lower. User’s accuracies for the low-density urban class are similar to its producer’s accuracies, whereas high-density urban entails a lower minimum and a higher maximum user’s accuracy over the 1984 to 2010 time period. Regarding some other classes, for agricultural land and pasture, producer’s and user’s accuracies are high for both classes across all dates (many values >90%), and this is generally the case for forest land as well. Rangeland posts relatively low producer’s accuracies across all dates though it accrues high user’s accuracies (>90%) for all multitemporal classifications.

Table A1. Land changes in the SARB and Bexar County from 1984 to 2010

The SARB

	Total area (ha)						1984-1995			1995-2010			1984-2010		
Land class	1984	%	1995	%	2010	%	ha	%	%/year	ha	%	%/year	ha	%	%/year
Urban	46,602	4.3	76,095	7.0	143,929	13.3	29,493	63.3	5.7	67,834	89.1	5.9	97,327	208.8	8.0
Low density urban	31,327	2.9	45,312	4.2	85,764	7.9	13,985	44.6	4.1	40,452	89.3	6.0	54,437	173.8	6.7
High density urban	15,275	1.4	30,783	2.8	58,165	5.4	15,508	101.5	9.2	27,382	89.0	5.9	42,890	280.8	10.8
Agricultural Land	111,835	10.3	104,841	9.7	92,611	8.5	-6,994	-6.3	-0.6	-12,230	-11.7	-0.8	-19,224	-17.2	-0.7
Pasture	173,895	16.0	193,128	17.8	199,338	18.4	19,233	11.1	1.0	6,210	3.2	0.2	25,443	14.6	0.6
Rangeland	411,210	37.9	392,479	36.1	389,135	35.8	-18,731	-4.6	-0.4	-3,344	-0.9	-0.1	-22,075	-5.4	-0.2
Forest Land	324,391	29.9	300,864	27.7	251,245	23.2	-23,527	-7.3	-0.7	-49,619	-16.5	-1.1	-73,146	-22.5	-0.9
Water	3,672	0.3	4,267	0.4	4,015	0.4	595	16.2	1.5	-252	-5.9	-0.4	343	9.3	0.4
Wetland	960	0.1	618	0.1	570	0.1	-342	-35.6	-3.2	-48	-7.8	-0.5	-390	-40.6	-1.6
Barren Land	12,379	1.1	12,109	1.1	3,345	0.3	-270	-2.2	-0.2	-8,764	-72.4	-4.8	-9,034	-73.0	-2.8
No Data	807	0.1	1,350	0.1	1,563	0.1									
Total (ha)	1,085,751	100.0	1,085,751	100.0	1,085,751	100.0									

Bexar County

	Total area (ha)						1984-1995			1995-2010			1984-2010		
Land class	1984	%	1995	%	2010	%	ha	%	%/year	ha	%	%/year	ha	%	%/year
Urban	19,894	12.6	39,666	25.1	60,663	38.4	19,772	99.4	9.0	20,997	52.9	3.5	40,769	204.9	7.9
Low density urban	14,767	9.4	22,439	14.2	26,698	16.9	7,672	52.0	4.7	4,259	19.0	1.3	11,931	80.8	3.1
High density urban	5,127	3.2	17,227	10.9	33,965	21.5	12,100	236.0	21.5	16,738	97.2	6.5	28,838	562.5	21.6
Agricultural Land	8,752	5.5	9,418	6.0	3,756	2.4	666	7.6	0.7	-5,662	-60.1	-4.0	-4,996	-57.1	-2.2
Pasture	10,245	6.5	8,919	5.6	14,996	9.5	-1,326	-12.9	-1.2	6,077	68.1	4.5	4,751	46.4	1.8
Rangeland	57,496	36.4	34,858	22.1	34,692	22.0	-22,638	-39.4	-3.6	-166	-0.5	0.0	-22,804	-39.7	-1.5
Forest Land	59,175	37.5	62,640	39.7	42,012	26.6	3,465	5.9	0.5	-20,628	-32.9	-2.2	-17,163	-29.0	-1.1
Water	82	0.1	133	0.1	77	0.1	51	62.2	5.7	-56	-42.1	-2.8	-5	-6.1	-0.2
Wetland	117	0.1	125	0.1	56	0.0	8	6.8	0.6	-69	-55.2	-3.7	-61	-52.1	-2.0
Barren Land	2,111	1.3	2,115	1.3	1,622	1.0	4	0.2	0.0	-493	-23.3	-1.6	-489	-23.2	-0.9
No Data	2	0.0	0	0.0	0	0.0									
Total (ha)	157,874	100.0	157,874	100.0	157,874	100.0									

Table A2. Land change matrix of the SARB and Bexar County from 1984 to 2010.

The SARB

		1984 land classes (ha)									
		Low density urban	High density urban	Agricultural Land	Pasture	Rangeland	Forest Land	Water	Wetland	Barren Land	*Total
1995 land class (ha)	Low density urban	14,619	0	1,272	6,287	18,077	3,846	28	16	1,163	45,312
	High density urban	3,163	15,275	171	202	9,085	2,769	24	23	70	30,783
	Agricultural Land	1,280	0	34,660	19,063	42,151	7,214	31	32	410	104,841
	Pasture	2,657	0	37,542	71,739	72,189	6,356	27	19	2,599	193,128
	Rangeland	6,497	0	29,524	66,101	180,533	105,194	93	177	4,306	392,479
	Forest Land	2,468	0	6,201	8,173	85,421	197,746	307	165	159	300,864
	Water	203	0	72	100	283	158	2,965	342	136	4,267
	Wetland	56	0	13	11	170	132	49	172	13	618
	Barren Land	374	0	2,374	2,207	3,035	572	15	5	3,522	12,109
	*Total	31,327	15,275	111,835	173,895	411,210	324,391	3,672	960	12,379	

* No data excluded

The SARB

		1995 land classes (ha)									
		Low density urban	High density urban	Agricultural Land	Pasture	Rangeland	Forest Land	Water	Wetland	Barren Land	*Total
2010 land class (ha)	Low density urban	16,590	0	8,655	6,838	32,830	20,169	54	53	502	85,764
	High density urban	11,211	30,783	749	621	6,588	8,041	12	12	130	58,165
	Agricultural Land	396	0	25,453	25,798	30,880	8,933	7	5	1,131	92,611
	Pasture	7,495	0	26,889	80,853	67,091	8,999	15	11	7,969	199,338
	Rangeland	6,977	0	35,639	72,090	173,927	98,942	39	22	1,240	389,135
	Forest Land	2,306	0	7,156	6,551	79,970	154,156	329	327	96	251,245
	Water	29	0	31	32	122	138	3,504	48	12	4,015
	Wetland	10	0	27	13	79	105	194	134	6	570
	Barren	297	0	226	324	767	701	4	2	1,024	3,345
	*Total	45,312	30,783	104,841	193,128	392,479	300,864	4,267	618	12,109	

* No data excluded

Bexar County

		1984 land classes (ha)									
		Low density urban	High density urban	Agricultural Land	Pasture	Rangeland	Forest Land	Water	Wetland	Barren Land	*Total
1995 land class (ha)	Low density urban	8,618	0	533	1,485	9,185	2,056	2	2	558	22,439
	High density urban	2,586	5,127	113	123	7,172	2,056	1	3	46	17,227
	Agricultural Land	301	0	3,004	1,519	3,598	945	2	3	46	9,418
	Pasture	432	0	1,882	2,629	3,251	575	3	2	127	8,919
	Rangeland	1,696	0	2,088	3,391	15,434	11,856	6	17	369	34,858
	Forest Land	933	0	998	921	18,399	41,295	9	22	52	62,640
	Water	8	0	0	0	13	15	52	36	7	133
	Wetland	26	0	3	1	34	22	3	32	4	125
	Barren	166	0	132	176	409	326	4	0	902	2,115
	*Total	14,767	5,127	8,752	10,245	57,496	59,175	82	117	2,111	

* No data excluded

Bexar County

		1995 land classes (ha)									
		Low density urban	High density urban	Agricultural Land	Pasture	Rangeland	Forest Land	Water	Wetland	Barren Land	*Total
2010 land class (ha)	Low density urban	9,261	0	1,627	1,326	7,249	6,881	6	22	326	26,698
	High density urban	8,051	17,227	385	278	3,221	4,694	1	4	104	33,965
	Agricultural Land	17	0	1,851	405	692	770	2	0	19	3,756
	Pasture	2,075	0	2,074	3,823	4,158	1,952	3	4	907	14,996
	Rangeland	1,750	0	2,784	2,751	11,586	15,689	9	4	119	34,692
	Forest Land	1,086	0	638	261	7,645	32,246	31	69	36	42,012
	Water	1	0	0	0	2	4	58	3	9	77
	Wetland	1	0	1	0	7	7	21	18	1	56
	Barren	197	0	58	75	298	396	3	1	594	1,622
	*Total	22,439	17,227	9,418	8,919	34,858	62,640	133	125	2,115	

* No data excluded

Table A3. Valuation in the SARB and Bexar County from 1984 to 2010.

Costanza et al. (1997) in the SARB

Land class	ESV (US million $ per year)			1984-1995			1995-2010			1984-2010		
	1984	1995	2010	USD	%	%/year	USD	%	%/year	USD	%	%/year
Urban	0.00	0.00	0.00	0.00	0.0	0.0	0.00	0.0	0.0	0.00	0.0	0.0
Agricultural Land	14.76	13.83	12.22	-0.92	-6.3	-0.6	-1.64	-11.7	-0.8	-2.53	-17.2	-0.7
Pasture	58.60	65.08	67.17	6.48	11.1	1.0	2.09	3.2	0.2	8.57	14.6	0.6
Rangeland	138.57	132.26	131.13	-6.31	-4.6	-0.4	-1.12	-0.9	-0.1	-7.43	-5.4	-0.2
Forest Land	142.08	131.77	110.04	-10.3	-7.3	-0.7	-21.73	-16.5	-1.1	-32.03	-22.5	-0.9
Water	45.28	52.62	49.51	7.33	16.2	1.5	-3.10	-5.9	-0.4	4.22	9.3	0.4
Wetland	27.28	17.56	16.19	-9.71	-35.6	-3.2	-1.36	-7.8	-0.5	-11.08	-40.6	-1.6
Barren Land	0.00	0.00	0.00	0.00	0.0	0.0	0.00	0.0	0.0	0.00	0.0	0.0
Total	426.58	413.14	386.29	-13.43	-3.1	-0.3	-26.85	-6.5	-0.4	-40.29	-9.4	-0.4

Constanza et al. (2014) Modified VC in the SARB

Land class	ESV (US million $ per year)			1984-1995			1995-2010			1984-2010		
	1984	1995	2010	USD	%	%/year	USD	%	%/year	USD	%	%/year
Urban	191.33	291.97	552.45	100.63	52.6	4.8	260.48	89.2	5.9	361.11	188.7	7.3
Low intensity urban	164.59	238.06	450.60	73.47	44.6	4.1	212.53	89.3	6.0	286.01	173.8	6.7
High intensity urban	26.74	53.90	101.84	27.15	101.5	9.2	47.94	89.0	5.9	75.10	280.8	10.8
Agricultural Land	654.68	613.73	542.14	-40.94	-6.3	-0.6	-71.59	-11.7	-0.8	-112.53	-17.2	-0.7
Pasture	761.83	846.09	873.29	84.25	11.1	1.0	27.2	3.2	0.2	111.46	14.6	0.6
Rangeland	1,801.51	1,719.45	1,704.80	-82.06	-4.6	-0.4	-14.65	-0.9	-0.1	-96.71	-5.4	-0.2
Forest Land	1,070.16	992.55	828.85	-77.61	-7.3	-0.7	-163.69	-16.5	-1.1	-241.30	-22.5	-0.9
Water	48.31	56.14	52.82	7.82	16.2	1.5	-3.31	-5.9	-0.4	4.51	9.3	0.4
Wetland	25.92	16.69	15.39	-9.26	-35.6	-3.2	-1.29	-7.8	-0.5	-10.53	-40.6	-1.6
Barren Land	0.00	0.00	0.00	0.00	0.0	0.0	0.00	0.0	0.0	0.00	0.0	0.0
Total	4,553.77	4,536.64	4,569.77	-17.13	-0.4	0.04	33.13	0.7	0.05	16.00	0.4	0.0

Costanza et al. (1997) in Bexar County

Land class	ESV (US million $ per year)			1984-1995			1995-2010			1984-2010		
	1984	1995	2010	USD	%	%/year	USD	%	%/year	USD	%	%/year
Urban	0.00	0.00	0.00	0.00	0.0	0.0	0.00	0.0	0.0	0.00	0.0	0.0
Agricultural Land	1.15	1.24	0.49	0.08	7.6	0.7	-0.74	-60.1	-4.0	-0.65	-57.1	-2.2
Pasture	3.45	3.00	5.05	-0.44	-12.9	-1.2	2.04	68.1	4.5	1.60	46.4	1.8
Rangeland	19.37	11.74	11.69	-7.62	-39.4	-3.6	-0.05	-0.5	0.0	-7.68	-39.7	-1.5
Forest Land	25.91	27.43	18.40	1.44	5.9	0.5	-9.03	-32.9	-2.2	-7.51	-29.0	-1.1
Water	1.01	1.64	0.94	0.62	62.2	5.7	-0.69	-42.1	-2.8	-0.06	-6.1	-0.2
Wetland	3.32	3.55	1.59	0.22	6.8	0.6	-1.96	-55.2	-3.7	-1.73	-52.1	-2.0
Barren Land	0.00	0.00	0.00	0.00	0.0	0.0	0.00	0.0	0.0	0.00	0.0	0.0
Total	54.23	48.62	38.18	-5.61	-10.4	-0.9	-10.44	-21.5	-1.4	-16.05	-29.6	-1.1

Constanza et al. (2014) Modified VC in Bexar County

Land class	ESV (US million $ per year)			1984-1995			1995-2010			1984-2010		
	1984	1995	2010	USD	%	%/year	USD	%	%/year	USD	%	%/year
Urban	86.56	148.05	199.74	61.49	71.0	6.5	51.68	34.9	2.3	113.18	130.7	5.0
Low intensity urban	77.58	117.89	140.27	40.38	52.0	4.7	22.37	19.0	1.3	62.68	80.8	3.1
High intensity urban	8.97	30.16	59.47	21.18	236.0	21.5	29.30	97.2	6.5	50.49	562.5	21.6
Agricultural Land	51.23	55.13	21.98	3.89	7.6	0.7	-33.14	-60.1	-4.0	-29.24	-57.1	-2.2
Pasture	44.88	39.07	65.69	-5.80	-12.9	-1.2	26.62	68.1	4.5	20.81	46.4	1.8
Rangeland	251.88	152.71	151.98	-99.17	-39.4	-3.6	-0.72	-0.5	0.0	-99.90	-39.7	-1.5
Forest Land	195.21	206.64	138.59	11.43	5.9	0.5	-68.05	-32.9	-2.2	-56.62	-29.0	-1.1
Water	1.07	1.75	1.01	0.67	62.2	5.7	-0.73	-42.1	-2.8	-0.06	-6.1	-0.2
Wetland	3.15	3.37	1.52	0.21	6.8	0.6	-1.86	-55.2	-3.7	-1.64	-52.1	-2.0
Barren Land	0.00	0.00	0.00	0.00	0.0	0.0	0.00	0.0	0.0	0.00	0.0	0.0
Total	634.02	606.75	580.53	-27.27	-4.3	-0.3	-26.21	-4.3	-0.2	-53.48	-8.4	-0.3

Table A4. Sensitivity analysis.

The SARB

Change in valuation coefficient (VC)	1984 %	1984 CS	1995 %	1995 CS	2010 %	2010 CS
*Low density urban VC-73.8%	-2.67	0.04	-3.87	0.05	-7.28	0.10
*High density urban VC-73.8%	-0.43	0.01	-0.88	0.01	-1.64	0.02
Low density urban VC-50%	-1.81	0.04	-2.62	0.05	-4.93	0.10
High density urban VC-50%	-0.29	0.01	-0.59	0.01	-1.11	0.02
Agricultural Land VC-48.9%	-7.02	0.14	-6.61	0.14	-5.80	0.12
Pasture VC-46.1%	-7.72	0.17	-8.60	0.19	-8.82	0.19
Rangeland VC-46.1%	-18.25	0.40	-17.49	0.38	-17.21	0.37
Forest Land VC-43.4%	-10.19	0.23	-9.48	0.22	-7.86	0.18
Water VC-3.1%	-0.03	0.01	-0.04	0.01	-0.04	0.01
Wetland VC+2.6%	0.01	0.01	0.01	0.00	0.01	0.00
Barren Land VC+0.0%	0.00	0.00	0.00	0.00	0.00	0.00

*Urban open space coefficients (Brander and Koetse, 2011)

Bexar County

Change in valuation coefficient (VC)	1984 %	1984 CS	1995 %	1995 CS	2010 %	2010 CS
*Low density urban VC-73.8%	-9.03	0.12	-14.34	0.19	-17.83	0.24
*High density urban VC-73.8%	-1.04	0.01	-3.67	0.05	-7.56	0.10
Low density urban VC-50%	-6.12	0.12	-9.72	0.19	-12.08	0.24
High density urban VC-50%	-0.71	0.01	-2.49	0.05	-5.13	0.10
Agricultural Land VC-48.9%	-3.95	0.08	-4.44	0.09	-1.85	0.04
Pasture VC-46.1%	-3.27	0.07	-2.97	0.06	-5.22	0.11
Rangeland VC-46.1%	-18.33	0.40	-11.61	0.25	-12.08	0.26
Forest Land VC-43.4%	-13.35	0.31	-14.77	0.34	-10.35	0.23
Water VC-3.1%	-0.01	0.00	-0.01	0.00	-0.01	0.00
Wetland VC+2.6%	0.01	0.00	0.01	0.01	0.01	0.00
Barren Land VC+0.0%	0.00	0.00	0.00	0.00	0.00	0.00

*Urban open space coefficients (Brander and Koetse, 2011)

Table A5. ES function and ESV.

The SARB

	1984				1995				2010				Rank		Trend	
Ecosystem function	ESV'	%	ESV"	%	ESV'	%	ESV"	%	ESV'	%	ESV"	%	'	"	'	"
Biological control	25.16	5.9	104.30	2.3	24.79	6.0	97.92	2.2	24.16	6.2	85.28	1.9	5	10	↓	↓
Climate regulation	41.52	9.7	151.28	3.3	38.51	9.3	158.02	3.5	32.15	8.3	180.29	3.9	4	5	↓	↑
Cultural	3.10	0.7	104.72	2.3	2.18	0.5	104.07	2.3	1.95	0.5	104.42	2.3	14	9	↓	↓
Disturbance regulation	10.08	2.4	3.01	0.1	6.49	1.6	1.94	0.0	5.98	1.5	1.78	0.0	11	17	↓	↓
Erosion Control	23.98	5.6	42.18	0.9	24.00	5.8	40.47	0.9	24.12	6.2	39.09	0.9	6	12	↑	↓
Food production	81.14	18.9	1109.82	24.4	78.93	19.1	1085.82	23.9	74.59	19.3	1043.89	22.8	2	1	↓	↓
Gas regulation	5.63	1.4	5.26	0.1	5.50	1.4	5.27	0.1	5.51	1.5	5.29	0.1	12	16	↓	↑
Genetic resources	0.00	0.0	869.73	19.1	0.00	0.0	862.68	19.0	0.00	0.0	852.94	18.7	16	3	–	↓
Habitat/Refugia	0.61	0.1	1043.87	22.9	0.39	0.1	1022.29	22.5	0.36	0.1	980.83	21.5	15	2	↓	↓
Nutrient Cycling	0.00	0.0	33.52	0.7	0.00	0.0	30.59	0.7	0.00	0.0	25.64	0.6	16	14	–	↓
Pollination	23.88	5.6	24.22	0.5	23.76	5.7	24.07	0.5	23.62	6.1	23.90	0.5	7	15	↓	↓
Raw materials	11.74	2.7	121.25	2.7	10.87	2.6	115.00	2.5	9.08	2.3	102.90	2.3	10	8	↓	↓
Recreation	20.53	4.9	538.28	11.8	19.23	4.7	600.51	13.2	16.57	4.3	771.73	16.9	9	4	↓	↑
Soil formation	6.03	1.4	68.51	1.5	5.68	1.4	64.29	1.4	4.94	1.3	56.71	1.2	13	11	↓	↓
Waste treatment	120.45	28.2	140.31	3.1	117.30	28.4	133.96	3.0	111.05	28.8	122.44	2.7	1	7	↓	↓
Water regulation	30.80	7.2	36.86	0.8	35.49	8.6	39.78	0.9	33.51	8.7	38.11	0.8	3	13	↑	↓
Water supply	21.87	5.1	156.54	3.4	19.92	4.8	149.88	3.3	18.62	4.8	134.44	2.9	8	6	↓	↓
Total	426.58	100.0	4553.77	100.0	413.14	100.0	4536.64	100.0	386.29	100.0	4569.77	100.0				

' Ecosystem function and ESV using Costanza et al. (1997) unit values (2010 US million $/ha/yr)

" Ecosystem function and ESV using modified Costanza et al. (2014) unit values (2010 US million $/ha/yr)

Bexar County

	1984				1995				2010				Rank		Trend	
Ecosystem function	ESV'	%	ESV"	%	ESV'	%	ESV"	%	ESV'	%	ESV"	%	'	"	'	"
Biological control	2.89	5.3	17.27	2.7	2.15	4.4	17.37	2.9	2.02	5.3	12.20	2.1	6	7	↓	↓
Climate regulation	7.57	14.0	27.91	4.4	8.01	16.5	36.11	6.0	5.37	14.0	37.60	6.5	3	5	↓	↑
Cultural	0.41	0.8	12.15	1.9	0.44	0.9	7.98	1.3	0.22	0.6	8.85	1.5	14	10	↓	↓
Disturbance regulation	1.22	2.3	0.36	0.1	1.31	2.7	0.39	0.1	0.58	1.5	0.17	0.0	10	17	↓	↓
Erosion Control	2.77	5.1	4.22	0.7	1.79	3.7	3.42	0.6	2.03	5.3	2.86	0.5	5	13	↓	↓
Food production	10.62	19.5	124.99	19.7	8.94	18.3	97.66	16.1	7.48	19.6	84.72	14.6	2	3	↓	↓
Gas regulation	0.65	1.3	0.60	0.1	0.44	0.9	0.39	0.1	0.46	1.3	0.44	0.1	13	16	↓	↓
Genetic resources	0.00	0.0	96.10	15.2	0.00	0.0	66.22	10.9	0.00	0.0	67.57	11.6	16	4	–	↓
Habitat/Refugia	0.07	0.1	140.47	22.1	0.07	0.2	113.04	18.6	0.03	0.1	101.70	17.5	15	2	↓	↓
Nutrient Cycling	0.00	0.0	6.00	0.9	0.00	0.0	6.36	1.0	0.00	0.0	4.21	0.7	16	11	–	↓
Pollination	2.68	4.9	2.70	0.4	1.80	3.7	1.83	0.3	1.91	5.0	1.92	0.3	7	14	↓	↓
Raw materials	2.13	3.9	17.18	2.7	2.26	4.6	16.63	2.7	1.51	3.9	11.71	2.0	8	8	↓	↓
Recreation	3.39	6.3	139.18	22.0	3.52	7.3	195.33	32.2	2.39	6.4	217.80	37.5	4	1	↓	↑
Soil formation	1.02	1.9	5.91	0.9	1.02	2.1	6.29	1.0	0.72	1.9	2.82	0.5	12	12	↓	↓
Waste treatment	16.35	30.2	16.90	2.7	13.83	28.5	15.80	2.6	11.76	30.8	11.03	1.9	1	9	↓	↓
Water regulation	0.85	1.6	1.73	0.3	1.18	2.5	2.25	0.4	0.75	2.0	1.54	0.3	11	15	↓	↓
Water supply	1.54	2.8	20.05	3.2	1.78	3.7	19.62	3.2	0.85	2.2	13.32	2.3	9	6	↓	↓
Total	54.23	100.0	634.02	100.0	48.62	100.0	606.75	100.0	38.18	100.0	580.53	100.0				

' Ecosystem function and ESV using Costanza et al. (1997) unit values (2010 US million $/ha/yr)

" Ecosystem function and ESV using modified Costanza et al. (2014) unit values (2010 US million $/ha/yr)

APPENDIX B

CHAPTER III SUPPLEMENTARY MATERIAL

B.1. Biodiversity and ecosystem Services (BES) quantification

B.1.1. Data and modeling process

This appendix describes the data sources and input parameters applied to the InVEST biodiversity, carbon storage, and sediment retention (export) models in the San Antonio River Basin. The process of land change analysis in terms of preprocessing and accuracy assessment is described by Yi et al. (2017) for the details. Table B1 describes the result of land change analysis in the study area. Table B2 lists data sources used for the study area in terms of InVEST models, which include land classes, DEM, rainfall erosivity index (R), soil erodibility (K), threshold flow accumulation (tfac) etc. (Sharp et al., 2016).

Input parameters for each modeling processes and ranges for the sensitivity analyses are described in Tables B3–B6. The results from sensitivity analyses in the SARB and in Bexar County for biodiversity, carbon storage, and sediment retention (export) are described in Tables B7–B10 from 1984 to 2010. Table B11 indicates the result of Moran's I in terms of biodiversity, carbon storage, sediment retention (export) in the SARB. Table B12 describes the result of Wilcoxon signed rank test in four sub-areas (i.e., upstream, suburban, urban, downstream watersheds) to examine the statistical differences during the pre-NAFTA (1984-1995) period and post-NAFTA (1995-2010) period in terms of biodiversity, carbon storage, sediment retention (export) in the SARB. B13 describes the Nonparametric Spearman's correlation matrix among BES estimates, NDVI, Air Toxics, and socio-economic variables based on the census tracts (USCB, 2010) in Bexar County (n=361).

B.1.2. Assessment of land changes and environmental justice analysis.

I assess the land changes in the EARZ and environmental justice analysis in Bexar County. Figure B1 shows land change between 1984 and 2010 in the Edward Aquifer Recharge Zone of the SARB. Figure B2 illustrates the spatial distribution of social-ecological variables for the environmental justice analysis in Bexar County in 2010 using the ACS 2010 (USCB, 2010) and the NATA 2011 (EPA, 2015).

Table B1. Land changes in the SARB and urban watersheds in Bexar County from 1984 to 2010.

The SARB

	Total area (ha)						1984-1995			1995-2010			1984-2010		
Land class	1984	%	1995	%	2010	%	ha	%	%/year	ha	%	%/year	ha	%	%/year
Urban	46,602	4.3	76,095	7.0	143,929	13.3	29,493	63.3	5.7	67,834	89.1	5.9	97,327	208.8	8.0
Low density urban	31,327	2.9	45,312	4.2	85,764	7.9	13,985	44.6	4.1	40,452	89.3	6.0	54,437	173.8	6.7
High density urban	15,275	1.4	30,783	2.8	58,165	5.4	15,508	101.5	9.2	27,382	89.0	5.9	42,890	280.8	10.8
Agricultural Land	111,835	10.3	104,841	9.7	92,611	8.5	-6,994	-6.3	-0.6	-12,230	-11.7	-0.8	-19,224	-17.2	-0.7
Pasture	173,895	16.0	193,128	17.8	199,338	18.4	19,233	11.1	1.0	6,210	3.2	0.2	25,443	14.6	0.6
Rangeland	411,210	37.9	392,479	36.1	389,135	35.8	-18,731	-4.6	-0.4	-3,344	-0.9	-0.1	-22,075	-5.4	-0.2
Forest Land	324,391	29.9	300,864	27.7	251,245	23.2	-23,527	-7.3	-0.7	-49,619	-16.5	-1.1	-73,146	-22.5	-0.9
Water	3,672	0.3	4,267	0.4	4,015	0.4	595	16.2	1.5	-252	-5.9	-0.4	343	9.3	0.4
Wetland	960	0.1	618	0.1	570	0.1	-342	-35.6	-3.2	-48	-7.8	-0.5	-390	-40.6	-1.6
Barren Land	12,379	1.1	12,109	1.1	3,345	0.3	-270	-2.2	-0.2	-8,764	-72.4	-4.8	-9,034	-73.0	-2.8
No Data	807	0.1	1,350	0.1	1,563	0.1									
Total (ha)	1,085,751	100.0	1,085,751	100.0	1,085,751	100.0									

Urban watersheds in Bexar County

	Total area (ha)						1984-1995			1995-2010			1984-2010		
Land class	1984	%	1995	%	2010	%	ha	%	%/year	ha	%	%/year	ha	%	%/year
Urban	19,894	12.6	39,666	25.1	60,663	38.4	19,772	99.4	9.0	20,997	52.9	3.5	40,769	204.9	7.9
Low density urban	14,767	9.4	22,439	14.2	26,698	16.9	7,672	52.0	4.7	4,259	19.0	1.3	11,931	80.8	3.1
High density urban	5,127	3.2	17,227	10.9	33,965	21.5	12,100	236.0	21.5	16,738	97.2	6.5	28,838	562.5	21.6
Agricultural Land	8,752	5.5	9,418	6.0	3,756	2.4	666	7.6	0.7	-5,662	-60.1	-4.0	-4,996	-57.1	-2.2
Pasture	10,245	6.5	8,919	5.6	14,996	9.5	-1,326	-12.9	-1.2	6,077	68.1	4.5	4,751	46.4	1.8
Rangeland	57,496	36.4	34,858	22.1	34,692	22.0	-22,638	-39.4	-3.6	-166	-0.5	0.0	-22,804	-39.7	-1.5
Forest Land	59,175	37.5	62,640	39.7	42,012	26.6	3,465	5.9	0.5	-20,628	-32.9	-2.2	-17,163	-29.0	-1.1
Water	82	0.1	133	0.1	77	0.1	51	62.2	5.7	-56	-42.1	-2.8	-5	-6.1	-0.2
Wetland	117	0.1	125	0.1	56	0.0	8	6.8	0.6	-69	-55.2	-3.7	-61	-52.1	-2.0
Barren Land	2,111	1.3	2,115	1.3	1,622	1.0	4	0.2	0.0	-493	-23.3	-1.6	-489	-23.2	-0.9
No Data	2	0.0	0	0.0	0	0.0									
Total (ha)	157,874	100.0	157,874	100.0	157,874	100.0									

The EARZ of SARB

	Total area (ha)						1984-1995			1995-2010			1984-2010		
Land class	1984	%	1995	%	2010	%	ha	%	%/year	ha	%	%/year	ha	%	%/year
Urban	1,556	3.3	3,481	7.2	9,926	20.7	1,925	123.7	11.2	6,445	185.1	12.3	8,370	537.9	20.7
Low density urban	1,082	2.3	2,316	4.8	5,687	11.9	1,234	114.0	10.4	3,371	145.5	9.7	4,605	425.6	16.4
High density urban	474	1.0	1,165	2.4	4,239	8.8	691	145.6	13.2	3,074	263.8	17.6	3,765	793.4	30.5
Agricultural Land	1,236	2.6	1,509	3.2	493	1.0	273	22.1	2.0	-1,016	-67.3	-4.5	-743	-60.1	-2.3
Pasture	1,897	4.0	1,361	2.8	2,550	5.3	-537	-28.3	-2.6	1,189	87.4	5.8	652	34.4	1.3
Rangeland	14,365	30.0	8,647	18.1	10,529	22.0	-5,719	-39.8	-3.6	1,882	21.8	1.5	-3,837	-26.7	-1.0
Forest Land	28,161	58.8	32,128	67.1	23,495	49.1	3,967	14.1	1.3	-8,632	-26.9	-1.8	-4,666	-16.6	-0.6
Water	51	0.1	55	0.1	45	0.1	4	7.4	0.7	-10	-17.9	-1.2	-6	-6.1	-0.5
Wetland	23	0.1	20	0.1	19	0.1	-4	-15.4	-1.4	-1	-1.4	-0.1	-4	-52.1	-0.6
Barren Land	577	1.2	671	1.4	814	1.7	94	16.3	1.5	143	21.3	1.4	236	41.0	1.6
No Data	11	0.0	5	0.0	6	0.0									
Total (ha)	47,877	100.0	47,877	100.0	47,877	100.0									

Table B2. Data sources used for the SARB application of the InVEST models.

Data type		Source
DEM	A Raster dataset with an elevation value for each cell	USDA https://gdg.sc.egov.usda.gov/ 30 meter resolution
Rainfall erosivity index (R)	A Raster dataset, with an erosivity index value for each cell (MJ·mm·(ha·hr)$^{-1}$)	NOAA, https://coast.noaa.gov/digitalcoast/ 30 meter resolution
Soil erodibility (K)	K is a raster dataset, with a soil erodibility value for each cell (ton·ha·hr·(MJ·ha·mm)$^{-1}$)	http://websoilsurvey.sc.egov.usda.gov/ Soil Survey Geographic Database (SSURGO) 30 meter resolution
Land class	A raster dataset	Landsat ISODATA classification 30 meter resolution
Watershed	A shapefile of polygons	TNRIS https://tnris.org/ NHD http://nhd.usgs.gov/
Biophysical table	Cover management and support practice factor for the USLE	Wischmeier and Smith (1984) Hamel et al. (2015)
Threshold flow accumulation (tfac)	The number of upstream cells that must flow into a cell	DEM based stream network and NHD river stream matching by ArcHydro Tool; 300
k_b and IC_0	Calibration parameters for k_b and Index of connectivity (IC_0)	Vigiak et al. (2012), k_b;1.3, IC_0; 0.5
SDR_{max}	Sediment delivery ratio (SDR) max	Cropland Modeling Documentation, SDR_{max}; 0.3 USDA https://www.nrcs.usda.gov/

Table B3. Parameters for biodiversity habitat score and sensitivity to each threat (0 to 1).

Land class	Habitat score	Source	Relative sensitivity to agriculture	Source	Relative sensitivity to low density urban	Source	Relative sensitivity to high density urban	Source
Low density urban	0.46	InVEST (2016)	0.21	InVEST (2016)	0.26	InVEST (2016)	0.28	InVEST (2016)
High density urban	0.15	Assumed from Low density urban	0.07	Assumed from Low density urban	0.08	Assumed from Low density urban	0.09	Assumed from Low density urban
Agricultural Land	0.48	InVEST (2016), Terrado et al. (2016)	0.31	InVEST (2016)	0.39	InVEST (2016)	0.44	InVEST (2016)
Pasture	0.72	InVEST (2016)	0.32	InVEST (2016)	0.41	InVEST (2016)	0.42	InVEST (2016)
Rangeland	0.8	InVEST (2016)	0.38	InVEST (2016)	0.47	InVEST (2016)	0.5	InVEST (2016)
Forest Land	1	InVEST (2016), Leh et al. (2013)	0.65	InVEST (2016)	0.72	InVEST (2016)	0.75	InVEST (2016)
Water	0.9	InVEST (2016)	0.72	InVEST (2016)	0.8	InVEST (2016)	0.83	InVEST (2016)
Wetland	0.7	InVEST (2016)	0.75	InVEST (2016)	0.9	InVEST (2016)	0.7	InVEST (2016)
Barren Land	0	InVEST (2016), Leh et al. (2013)	0	InVEST (2016)	0	InVEST (2016)	0	InVEST (2016)

Table B4. Sensitivity of distance to each threat.

LULC	Maximum distance (km) from threat	Source	Weight (0 to 1); impact on habitat quality	Source	Type of decay over space	Source	Distance range for sensitivity analysis
Low density urban	10	InVEST (2016)	1	InVEST (2016), Terrado et al (2016)	Exponential	InVEST (2016)	± 50% (5, 15)
High density urban	10	InVEST (2016)	1	InVEST (2016), Polasky et al (2011)	Exponential	InVEST (2016)	± 50% (5, 15)
Agricultural Land	8	InVEST (2016)	0.7	InVEST (2016)	Exponential	Polasky et al. (2011)	± 50% (4, 12)

Table B5. Parameters for carbon density and sensitivity analysis.

	Aboveground mass (Mg C/ha)		Belowground mass (Mg C/ha)		Soil (Mg C/ha)		Dead mass (Mg C/ha)		Total (Mg C/ha)	Sensitivity analysis
Land class	Value	Source	Value	Source	Value	Source	Value	Source	Value	Value
Low density urban	5	InVEST (2016)	6	InVEST (2016)	27	InVEST (2016)	1	InVEST (2016)	39	± 50% (19.5, 58.5)
High density urban	2	Assumed from Low density	2	Assumed from Low density	9	Assumed from Low density	0	Assumed from Low density	13	± 50% (6.5, 19.5)
Agricultural Land	13	InVEST (2016)	3	InVEST (2016)	24	IPCC (2006)	0	InVEST (2016)	40	± 50% (20, 60)
Pasture	2	InVEST (2016)	2	InVEST (2016)	38	IPCC (2006)	0	InVEST (2016)	42	± 50% (21, 63)
Rangeland	5	InVEST (2016)	5	InVEST (2016)	38	IPCC (2006)	2	InVEST (2016)	50	± 50% (25, 75)
Forest Land	63	COLE (2016)	53	COLE (2016)	43	COLE (2016)	1	COLE (2016)	160	± 50% (80, 240)
Water*	0	InVEST (2016)	0	InVEST (2016)	0	InVEST (2016)	0	InVEST (2016)	0	
Wetland*	9	InVEST (2016)	4	InVEST (2016)	23	InVEST (2016)	0	InVEST (2016)	36	
Barren Land*	0	InVEST (2016)	0	InVEST (2016)	0	InVEST (2016)	0	InVEST (2016)	0	

*Sensitivity not analyzed

Table B6. Parameters for sediment retention and sensitivity analysis.

Input data	Land class	Value	Source	Sensitivity analysis
USLE C factor	Low density urban	0.3	Wischmeier and Smith (1978)	± 50% (0.15, 0.45)
	High density urban	0.1	Wischmeier and Smith (1978), Hamel et al.(2015)	± 50% (0.05, 0.15)
	Agricultural Land	0.2	Wischmeier and Smith (1978)	± 50% (0.1, 0.3)
	Pasture	0.1	Wischmeier and Smith (1978)	± 50% (0.05, 0.15)
	Rangeland	0.01	Wischmeier and Smith (1978), Hamel et al.(2015)	± 50% (0.005, 0.015)
	Forest Land	0.001	Wischmeier and Smith (1978), Hamel et al.(2015)	± 50% (0.0005, 0.0015)
	Water*	0.001	Wischmeier and Smith (1978), Hamel et al.(2015)	
	Wetland*	0.001	Wischmeier and Smith (1978), Hamel et al.(2015)	
	Barren Land*	1	Wischmeier and Smith (1978)	
USLE P factor	Nine land classes	1	Wischmeier and Smith (1978), Hamel et al.(2015)	- 50% (0.5)
Threshold cell numbers		300	National Hydrography Dataset (NHD)	
Kb		1.3	Vigiak et al. (2012)	± 50% (0.65, 1.95)
Index of connectivity (ICo)		0.5	Vigiak et al. (2012)	± 50% (0.25, 0.75)
Sediment delivery ratio (SDR)max		0.3	Vigiak et al. (2012), USDA (2016)	

*Sensitivity not analyzed

Table B7. Biodiversity sensitivity and value change in the SARB and Bexar County from 1984 to 2010.

	1984		1995		2010	
Change in value coefficient (VC) in the SARB	%	CS	%	CS	%	CS
Low density urban VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
Low density urban VC + 50%	0.0	0.00	0.0	0.00	0.0	0.00
High density urban VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
High density urban VC + 50%	0.0	0.00	0.0	0.00	0.0	0.00
Agricultural land VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
Agricultural land VC + 50%	0.0	0.00	0.0	0.00	0.0	0.00

	1984		1995		2010	
Change in value coefficient (VC) in Bexar County	%	CS	%	CS	%	CS
Low density urban VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
Low density urban VC + 50%	0.0	0.00	0.0	0.00	0.0	0.00
High density urban VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
High density urban VC + 50%	0.0	0.00	0.0	0.00	0.0	0.00
Agricultural land VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
Agricultural land VC + 50%	0.0	0.00	0.0	0.00	0.0	0.00

Table B8. Carbon storage sensitivity and value change in the SARB and Bexar County from 1984 to 2010.

	1984		1995		2010	
Change in value coefficient (VC) in the SARB	%	CS	%	CS	%	CS
Low density urban VC - 50%	-0.7	0.01	-1.1	0.02	-2.2	0.04
Low density urban VC + 50%	0.7	0.01	1.1	0.02	2.2	0.04
High density urban VC - 50%	-0.1	0.00	-0.2	0.00	-0.5	0.01
High density urban VC + 50%	0.1	0.00	0.2	0.00	0.5	0.01
Agricultural land VC - 50%	-2.6	0.05	-2.5	0.05	-2.4	0.05
Agricultural land VC + 50%	2.6	0.05	2.5	0.05	2.4	0.05
Pasture VC - 50%	-4.3	0.09	-4.9	0.10	-5.5	0.11
Pasture VC + 50%	4.3	0.09	4.9	0.10	5.5	0.11
Rangeland VC - 50%	-12	0.24	-11.9	0.24	-12.8	0.26
Rangeland VC + 50%	12	0.24	11.9	0.24	12.8	0.26
Forest VC - 50%	-30.3	0.61	-29.3	0.59	-26.5	0.53
Forest VC + 50%	30.3	0.61	29.3	0.59	26.5	0.53

	1984		1995		2010	
Change in value coefficient (VC) in Bexar County	%	CS	%	CS	%	CS
Low density urban VC - 50%	-2.1	0.04	-3.2	0.06	-4.9	0.10
Low density urban VC + 50%	2.1	0.04	3.2	0.06	4.9	0.10
High density urban VC - 50%	-0.2	0.00	-0.8	0.02	-2.1	0.04
High density urban VC + 50%	0.2	0.00	0.8	0.02	2.1	0.04
Agricultural land VC - 50%	-1.3	0.03	-1.4	0.03	-0.7	0.01
Agricultural land VC + 50%	1.3	0.03	1.4	0.03	0.7	0.01
Pasture VC - 50%	-1.6	0.03	-1.4	0.03	-2.9	0.06
Pasture VC + 50%	1.6	0.03	1.4	0.03	2.9	0.06
Rangeland VC - 50%	-10.4	0.21	-6.4	0.13	-8.1	0.16
Rangeland VC + 50%	10.4	0.21	6.4	0.13	8.1	0.16
Forest VC - 50%	-34.4	0.69	-36.8	0.74	-31.3	0.63
Forest VC + 50%	34.4	0.69	36.8	0.74	31.3	0.63

Table B9. Sediment retention sensitivity and value change in the SARB and Bexar County from 1984 to 2010.

	1984		1995		2010	
Change in value coefficient (VC) in the SARB	%	CS	%	CS	%	CS
Low density urban VC - 50%	0.1	0.00	0.1	0.00	0.3	0.00
Low density urban VC + 50%	-0.1	0.00	-0.1	0.00	-0.3	0.00
High density urban VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
High density urban VC + 50%	0.0	0.00	0.0	0.00	0.0	0.00
Agricultural land VC - 50%	0.1	0.00	0.1	0.00	0.1	0.00
Agricultural land VC + 50%	-0.1	0.00	-0.1	0.00	-0.1	0.00
Pasture VC - 50%	0.1	0.00	0.1	0.00	0.1	0.00
Pasture VC + 50%	-0.1	0.00	-0.1	0.00	-0.1	0.00
Rangeland VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
Rangeland VC + 50%	0.0	0.00	0.0	0.00	0.0	0.00
Forest VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
Forest VC + 50%	0.0	0.00	0.0	0.00	0.0	0.00
Low density P factor VC - 50%	0.1	0.00	0.1	0.00	0.2	0.00
High density P factor VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
Agricultural land P factor VC - 50%	0.1	0.00	0.1	0.00	0.0	0.00
Pasture P factor VC - 50%	0.1	0.00	0.1	0.00	0.1	0.00
Rangeland P factor VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
Forest P factor VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
Kb VC - 50%	-82.0	1.64	-82.0	1.64	-82.0	1.64
Kb VC + 50%	89.0	1.78	89.0	1.78	88.8	1.78
IC - 50%	18.0	-0.36	18.0	-0.36	18.0	-0.36
IC + 50%	-15.6	-0.31	-15.6	-0.31	-15.6	-0.31

	1984		1995		2010	
Change in value coefficient (VC) in Bexar County	%	CS	%	CS	%	CS
Low density urban VC - 50%	0.2	0.00	0.4	0.00	0.5	0.00
Low density urban VC + 50%	-0.3	0.00	-0.4	0.00	-0.6	0.00
High density urban VC - 50%	0.1	0.00	0.1	0.00	0.2	0.00
High density urban VC + 50%	0.0	0.00	-0.1	0.00	-0.2	0.00
Agricultural land VC - 50%	0.1	0.00	0.1	0.00	0.0	0.00
Agricultural land VC + 50%	-0.1	0.00	-0.1	0.00	0.0	0.00
Pasture VC - 50%	0.1	0.00	0.0	0.00	0.1	0.00
Pasture VC + 50%	-0.1	0.00	0.0	0.00	-0.1	0.00
Rangeland VC - 50%	0.1	0.00	0.1	0.00	0.1	0.00
Rangeland VC + 50%	0.0	0.00	0.0	0.00	0.0	0.00
Forest VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
Forest VC + 50%	0.0	0.00	0.0	0.00	0.0	0.00
Low density P factor VC - 50%	0.2	0.00	0.3	0.00	0.4	0.00
High density P factor VC - 50%	0.0	0.00	0.0	0.00	0.1	0.00
Agricultural land P factor VC - 50%	0.1	0.00	0.1	0.00	0.0	0.00
Pasture P factor VC - 50%	0.0	0.00	0.0	0.00	0.1	0.00
Rangeland P factor VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
Forest P factor VC - 50%	0.0	0.00	0.0	0.00	0.0	0.00
Kb VC - 50%	-87.1	1.74	-87.1	1.74	-87.1	1.74
Kb VC + 50%	106.0	2.12	105.6	2.11	105.2	2.10
IC - 50%	18.7	-0.37	18.7	-0.37	18.7	-0.37
IC + 50%	-16.0	-0.32	-16.0	-0.32	-16.0	-0.32

Table B10. Sediment export sensitivity and value change in the SARB and Bexar County from 1984 to 2010

	1984		1995		2010	
Change in value coefficient (VC) in the SARB	%	CS	%	CS	%	CS
Low density urban VC - 50%	-12.5	0.25	-15.5	0.31	-35.5	0.71
Low density urban VC + 50%	15.1	0.30	18.5	0.37	43.7	0.87
High density urban VC - 50%	-3.6	0.07	-4.0	0.08	-5.8	0.12
High density urban VC + 50%	3.3	0.07	4.1	0.08	6.2	0.12
Agricultural land VC - 50%	-11.1	0.22	-15.8	0.32	-8.5	0.17
Agricultural land VC + 50%	13.2	0.26	19.0	0.38	10.3	0.21
Pasture VC - 50%	-15.2	0.30	-13.5	0.27	-12.6	0.25
Pasture VC + 50%	17.4	0.35	15.6	0.31	14.5	0.29
Rangeland VC - 50%	-7.3	0.15	-7.6	0.15	-6.6	0.13
Rangeland VC + 50%	5.5	0.11	5.8	0.12	5.0	0.10
Forest VC - 50%	-0.1	0.00	-0.1	0.00	-0.1	0.00
Forest VC + 50%	2.0	0.04	2.6	0.05	2.7	0.05
Low density P factor VC - 50%	-9.3	0.19	-11.4	0.23	-27.2	0.54
High density P factor VC - 50%	-1.6	0.03	-2.3	0.05	-3.6	0.07
Agricultural land P factor VC - 50%	-8.0	0.16	-11.8	0.24	-6.3	0.13
Pasture P factor VC - 50%	-10.3	0.21	-9.3	0.19	-8.4	0.17
Rangeland P factor VC - 50%	-1.4	0.03	-1.5	0.03	-1.3	0.03
Forest P factor VC - 50%	-0.1	0.00	-0.1	0.00	-0.1	0.00
Kb VC - 50%	-95.0	1.90	-95.6	1.91	-95.8	1.92
Kb VC + 50%	263.4	5.27	278.7	5.57	271.8	5.44
IC - 50%	20.2	-0.40	20.3	-0.41	20.3	-0.41
IC + 50%	-16.9	-0.34	-17.0	-0.34	-17.0	-0.34

	1984		1995		2010	
Change in value coefficient (VC) in Bexar County	%	CS	%	CS	%	CS
Low density urban VC - 50%	-23.7	0.47	-30.7	0.61	-38.0	0.76
Low density urban VC + 50%	28.9	0.58	37.2	0.74	46.3	0.93
High density urban VC - 50%	-5.3	0.11	-7.7	0.15	-12.7	0.25
High density urban VC + 50%	4.6	0.09	7.7	0.15	13.6	0.27
Agricultural land VC - 50%	-6.7	0.13	-6.9	0.14	-1.7	0.03
Agricultural land VC + 50%	8.2	0.16	8.2	0.16	2.0	0.04
Pasture VC - 50%	-6.4	0.13	-3.9	0.08	-7.0	0.14
Pasture VC + 50%	6.9	0.14	4.2	0.08	7.1	0.14
Rangeland VC - 50%	-5.6	0.11	-4.2	0.08	-4.4	0.09
Rangeland VC + 50%	4.1	0.08	3.0	0.06	3.0	0.06
Forest VC - 50%	-0.1	0.00	-0.1	0.00	-0.1	0.00
Forest VC + 50%	1.8	0.04	2.4	0.05	2.3	0.05
Low density P factor VC - 50%	-17.4	0.35	-22.3	0.45	-28.7	0.57
High density P factor VC - 50%	-2.1	0.04	-4.0	0.08	-7.3	0.15
Agricultural land P factor VC - 50%	-5.0	0.10	-5.0	0.10	-1.2	0.02
Pasture P factor VC - 50%	-4.1	0.08	-2.5	0.05	-3.9	0.08
Rangeland P factor VC - 50%	-0.9	0.02	-0.6	0.01	-0.6	0.01
Forest P factor VC - 50%	-0.1	0.00	-0.1	0.00	-0.1	0.00
Kb VC - 50%	-95.8	1.92	-96.2	1.92	-96.7	1.93
Kb VC + 50%	264.4	5.29	274.5	5.49	280.6	5.61
IC - 50%	20.3	-0.41	20.4	-0.41	20.5	-0.41
IC + 50%	-17.0	-0.34	-17.0	-0.34	-17.1	-0.34

Table B11. Moran's I results in the SARB.

	Years	Moran's I	Variance	Z-scores	p-values
Biodiversity	1984	0.766	0.0071	9.198	<0.001
	1995	0.697	0.0069	8.474	<0.001
	2010	0.680	0.0066	8.431	<0.001
Carbon storage	1984	0.823	0.0071	9.874	<0.001
	1995	0.891	0.0070	10.706	<0.001
	2010	0.800	0.0070	9.636	<0.001
Sediment retention	1984	0.537	0.0057	7.184	<0.001
	1995	0.538	0.0057	7.191	<0.001
	2010	0.538	0.0057	7.193	<0.001
Sediment export	1984	0.550	0.0068	6.759	<0.001
	1995	0.521	0.0063	6.655	<0.001
	2010	0.533	0.0059	7.043	<0.001

Table B12. Wilcoxon signed rank test in four sub-areas of the SARB.

		Urban watershed				Suburban watershed				Upstream watershed				Downstream watershed			
Wilcoxon signed rank test		N	Mean rank	z	p	N	Mean rank	z	p	N	Mean rank	z	p	N	Mean rank	z	p
Low density 1995 to 2010 - 1984 to1995	Negative ranks	10	10.19	-1.581	0.119	2	3.00	-3.937	0.000**	0	0.00	-3.762	0.000**	7	23.21	-4.080	0.000**
	Positive ranks	6	6.81			20	12.35			18	9.50			38	22.96		
	Ties	0				0				2				4			
High density 1995 to 2010 - 1984 to1995	Negative ranks	4	9.85	-0.801	0.440	0	0.00	-3.571	0.000**	2	4.00	-1.134	0.453	0	0.00	-1.000	0.317
	Positive ranks	11	6.25			15	8.00			5	4.00			1	1.00		
	Ties	1				7				13				48			
Biodiversity 1995 to 2010 - 1984 to1995	Negative ranks	7	8.71	-0.535	0.614	18	10.42	-3.736	0.000**	13	9.46	-2.237	0.025*	14	18.71	-2.738	0.005**
	Positive ranks	7	6.29			1	2.50			4	7.50			30	24.27		
	Ties	2				3				3				5			
Carbon storage 1995 to 2010 - 1984 to1995	Negative ranks	12	10.17	-2.792	0.003**	21	12.00	-4.075	0.000**	12	13.04	-1.923	0.054	6	12.33	-5.272	0.000**
	Positive ranks	4	3.50			1	1.00			8	6.69			42	26.24		
	Ties	0				0				0				1			
Sediment retention 1995 to 2010 - 1984 to1995	Negative ranks	5	8.10	-0.369	0.833	13	8.62	-2.994	0.002**	14	8.36	-2.553	0.009**	8	12.19	-0.290	0.772
	Positive ranks	8	6.31			2	4.00			2	9.50			12	9.38		
	Ties	3				7				4				29			
Sediment export 1995 to 2010 - 1984 to1995	Negative ranks	7	5.36	-0.122	0.897	1	4.50	-3.210	0.001**	2	9.50	-0.255	0.009**	11	8.95	-0.145	0.884
	Positive ranks	5				14				14				8			
	Ties	4	8.10			7	8.25			4	8.36			30	11.44		

* p < .05, ** p < .01

Table B13. Nonparametric Spearman's correlation matrix among BES estimates, NDVI, Air Toxics, and socio-economic variables based on the Census tracts in Bexar County (n=361).

	Biodiversity	Carbon storage	Sediment retention	NDVI	Total Respiratory HI	Diesel PM	Total Cancer risk	Poverty rate	Median household income	Unemployment rate
Biodiversity										
Carbon storage	0.924**									
Sediment retention	0.317**	0.346**								
NDVI	0.656**	0.812**	0.281**							
Total Respiratory HI	-0.466**	-0.329**	-0.165**	-0.271**						
Diesel Particulate Matter	-0.622**	-0.516**	-0.159**	-0.447**	0.836**					
Total Cancer risk	-0.557**	-0.411**	-0.164**	-0.311**	0.946**	0.889**				
Poverty rate	-0.547**	-0.643**	-0.181**	-0.587**	0.327**	0.443**	0.378**			
Median household income	0.648**	0.718**	0.199**	0.655**	-0.431**	-0.548**	-0.487**	-0.856**		
Unemployment rate	-0.351**	-0.469**	-0.157**	-0.441**	0.118*	0.253**	0.160**	0.579**	-0.596**	
Hispanic %	-0.638**	-0.720**	-0.309**	-0.533**	0.205**	0.376**	0.264**	0.719**	-0.746**	0.544**
White %	0.638**	0.750**	0.284**	0.601**	-0.220**	-0.405**	-0.299**	-0.773**	0.805**	-0.601**
African American %	0.125*	0.108*	0.272**	0.092	-0.114*	-0.034	-0.052	-0.089	0.091	-0.013

** Correlation is significant at the 0.01 level (2-tailed)

* Correlation is significant at the 0.05 level (2-tailed)

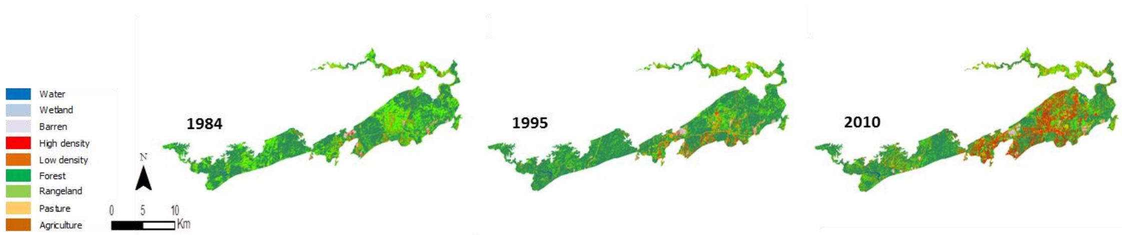
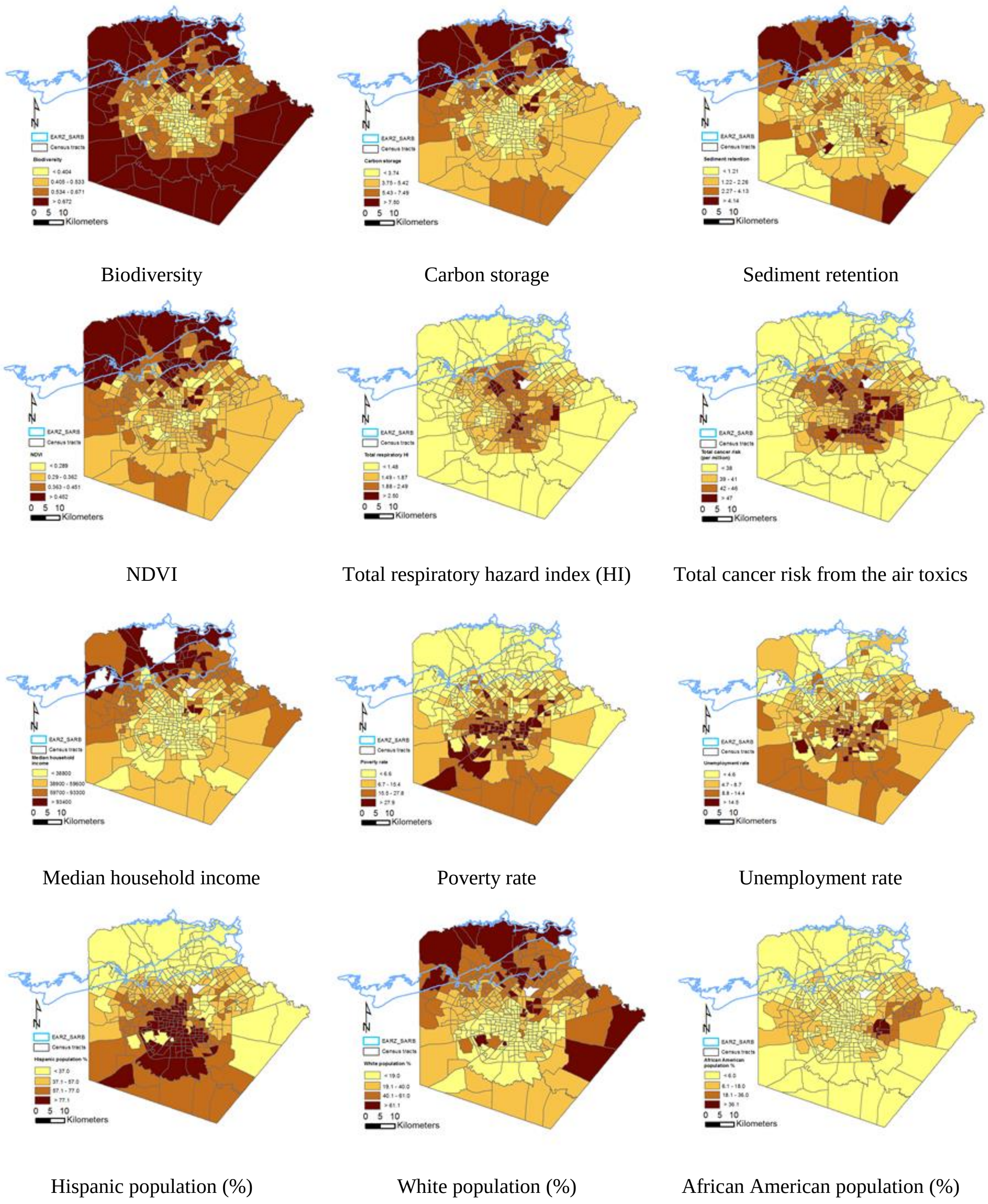


Figure B1. Land change between 1984 and 2010 in the Edward Aquifer Recharge Zone (EARZ) of the SARB.



Biodiversity | Carbon storage | Sediment retention

NDVI | Total respiratory hazard index (HI) | Total cancer risk from the air toxics

Median household income | Poverty rate | Unemployment rate

Hispanic population (%) | White population (%) | African American population (%)

Figure B2. Census tract based spatial distribution of social-ecological variables for the environmental justice analysis in Bexar County in 2010 using ACS 2010 (USCB, 2010) and the NATA 2011 (EPA, 2015).