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Spatial and temporal changes in biodiversity and ecosystem services in the San Antonio River Basin, Texas, from 1984 to 2010

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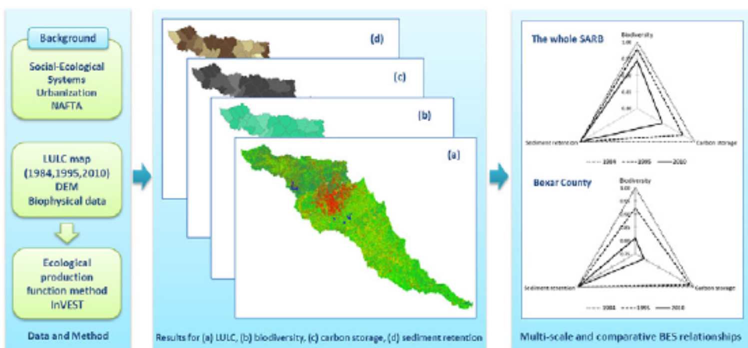
HIGHLIGHTS

- We conduct an analysis of urban impact on biodiversity and ecosystem services (BES).
- We find synergistic spatio-temporal relationships between BES at multiple scales.
- Loss of BES accelerated under the North American Free Trade Agreement (NAFTA).
- Green infrastructure to integrate BES is critical for the San Antonio River Basin.

GRAPHICAL ABSTRACT



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ABSTRACT

A fundamental premise of the Millennium Ecosystem Assessment is that biodiversity and ecosystem services are key determinants of long-term sustainability of social-ecological systems. With a continuing decline in local and global biodiversity and ecosystem services, it is crucial to understand how biodiversity and various ecosystem services interact and how land change may modify these interactions over time. However, few studies have been conducted to quantify these relationships. In this study, we present the first empirical comparative results to analyze how spatial associations between biodiversity and ecosystem services (BES) changed at multiple scales between 1984 and 2010 in the rapidly urbanizing San Antonio River Basin (SARB), Texas, USA. We found statistically significant positive spatial associations among biodiversity, carbon storage, and sediment retention both in the entire SARB and the urban watersheds in Bexar County. Overall, biodiversity and carbon storage declined across the SARB, while sediment retention remained relatively stable. Moreover, the rates of biodiversity loss and carbon storage degradation were negatively related to the urban expansion and have accelerated since the inception of the North American Free Trade Agreement (NAFTA) in 1994. During the pre- and post-NAFTA periods (1984–1995 and 1995–2010, respectively) the rates of biodiversity loss increased from 0.7% to 0.9%, and the rates of carbon-storage loss increased from 0.1% to 1.4% per annum in the urban watersheds. Our hotspot analyses indicate that the upstream watersheds in the Basin, which supply water to the critically important Edwards Aquifer, should be targeted for priority conservation to mitigate the adverse impacts of land change on BES. Our results suggest the strong need for green infrastructure policies that integrate biodiversity conservation and sustainable use of multiple ecosystem services to address the environmentally deleterious

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impacts of the extensive land change under the NAFTA and to ensure the long-term social-ecological sustainability of the rapidly urbanizing SARB.

1. Introduction

Land change to meet the increasing economic demands of rapidly growing human populations have significant implications for long-term sustainability and social-ecological resilience (DeFries and Eshleman, 2004; Foley et al., 2005; TEEB, 2010; IPBES, 2015). Urban expansion is one of the most critical alterations in land-use and land-cover affecting both biodiversity and ecosystem functions (MEA, 2005; Grimm et al., 2008; Güneralp and Seto, 2013), and threatening many species through habitat degradation and fragmentation (Czech et al., 2000; Mckinney, 2002; Hansen et al., 2005). However, the links between biodiversity and ecosystem services are poorly understood, especially in urbanizing areas (Chapin et al., 2000; Cardinale et al., 2012; Balvanera et al., 2014).

Nelson et al. (2009) found that a range of ecosystem functions are synergistically associated with biodiversity and, therefore, policy interventions that aim to simultaneously enhance both biodiversity and ecosystem services are justified. Based on this justification, they proposed payments for carbon sequestration to moderate development-related impacts on biodiversity and ecosystem services. Other studies also reported generally high association between biodiversity and ecosystem services (Turner et al., 2007; Harrison et al., 2014). However, some researchers have not found strong spatial congruence between biodiversity and ecosystem services (Egoh et al., 2009), as well as low overall correlation and relatively low pair-wise overlaps in spatial relationships between biodiversity and ecosystem services (Chan et al., 2006). Importantly, despite the growing recognition of significance of biodiversity and ecosystem services in moving social-ecological systems toward sustainability (Fischer et al., 2015; Virapongse et al., 2016), none of these studies investigated how the associations among biodiversity, carbon storage, and sediment retention were altered at multiple scales over time due to land change.

The San Antonio River Basin (SARB) is an ecologically diverse region in south-central Texas. The rapidly urbanizing San Antonio-New Braunfels Metropolitan Statistical Area (MSA) is located within this Basin, and San Antonio is the seventh most populous city in the US and ranked third in terms of net population increase in 2016 (USCB, 2015, 2017). The city, strategically located along the North American Free Trade Agreement (NAFTA) corridor, which has been facilitating trilateral trade among Canada, Mexico, and the United States (Brookings Institution, 2013), functions as a major multi-modal transportation hub within this corridor (TTI, 2007; TxDOT, 2013). The population in the SARB has increased from 1980 to 2010 by nearly 80% during the last 30 years due primarily to the economic growth in and around the City of San Antonio; its population is projected to further grow to about 2.5 million by 2050 (TWDB, 2010).

Our research was primarily motivated by growing concerns about biodiversity loss and degradation of ecosystem services including negative effects on the supply of high-quality water from the Edwards Aquifer, a regional karst aquifer replenished in part by the SARB (SCBD, 2010; Cardinale, 2011). For instance, the Golden-cheeked Warbler (*Setophaga chrysoparia*) is an endangered neotropic songbird that breeds in Ashe juniper (*Juniperus ashei*)-oak woodlands that are increasingly impacted by land-cover change in the Texas Hill Country (Kroll, 1980; Duarte et al., 2013; IUCN, 2016). Second, with a carbon-intensive economy and expanding NAFTA transportation corridor, Texas ranks first among US states in terms of total carbon dioxide (CO_2) emissions (USEIA, 2016). Third, sediment coming from urbanization-related soil erosion directly affects watershed health and water quality (Bush et al., 2000). Thus, we address the following questions: (1) What are the quantitative changes in biodiversity and ecosystem services in the SARB in response to the land changes from 1984 to 2010?; and (2) how did the spatial associations between biodiversity and ecosystem services in the SARB change during the same period?

2. Methods

2.1. Study area

The San Antonio River Basin (SARB), one of the major drainage basins in South Texas, covers an ecologically diverse area of 10,862 km^2. The central part of the basin includes part of the heavily urbanized San Antonio-New Braunfels Metropolitan Statistical Area (MSA). The City of San Antonio is located about 140 miles northwest of the Gulf of Mexico and 150 miles northeast of Laredo on the Mexican border (Fig. 1; San Antonio Chamber of Commerce, 2015). The upper half of the SARB intersects with the environmentally-sensitive Edwards Aquifer drainage and recharge zones (SARA, 2015), whereas the lower part of the basin is mostly rural and flows southeastward through the Gulf Coastal Plains. The Edwards Aquifer recharge zone (EARZ) is a limestone area where surface water recharges the aquifer. The SARB is characterized by higher elevation in the north-west near the EARZ (Fig. 1). The upper reaches of the SARB are also characterized by steep elevation and periodic flash flooding; it has been referred to as Flash Flood Alley along the Balcones Escarpment (TWRI, 2016).

We selected "watershed" as the unit of analysis because it is fundamental to the provision of key ecosystem services for surface-flow regulation and groundwater recharge, and often, it represents the scale of management needed to sustain properly-functioning ecosystems (Kreuter et al., 2001; Troy and Wilson, 2006; Brauman et al., 2007). The SARB consists of 107 sub-watersheds at Hydrologic Unit Code (HUC) 12 level and contains almost the entire Bexar County (Fig. 1). We divided Bexar County into two parts as urban and peri-urban watersheds in order to characterize detailed changes over time in biodiversity and ecosystem services (collectively referred to as BES henceforth). Three urban watersheds (i.e., the Leon Creek, the Upper San Antonio River, and the Salado Creek Watersheds) with 16 sub-watersheds covering 1579 km^2 comprise Bexar County; the peri-urban watersheds consist of 22 sub-watersheds surrounding the urban watersheds of the Bexar County; and the upstream watersheds and downstream watersheds consist of 20 and 49 sub-watersheds, respectively (Fig. 1).

2.2. Quantification of biodiversity and ecosystem services

In our study we targeted the period from 1984 to 2010 in order to focus on the potentially NAFTA-related land changes. Our study period also allows us to exclude the confounding effects of the rapid development of the Eagle Ford Shale as a result of oil and gas extraction that started around 2010 (Railroad Commission of Texas, 2016). In part, the Eagle Ford Shale underlies Wilson and Karnes Counties, which incorporate the south-central part of the SARB. We used the watershed boundary dataset in terms of Hydrologic Unit Code (HUC) 12 level, which is delineated and geo-referenced to U.S. Geological Survey (USGS) 1:24,000-scale topographic base maps (https://tnris.org) (TNRIS, 2015). We also employed Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) (Sharp et al., 2016) to analyze changes in biodiversity, carbon storage, and sediment retention (Table 1). The ecological production function method (EPFM) used in InVEST

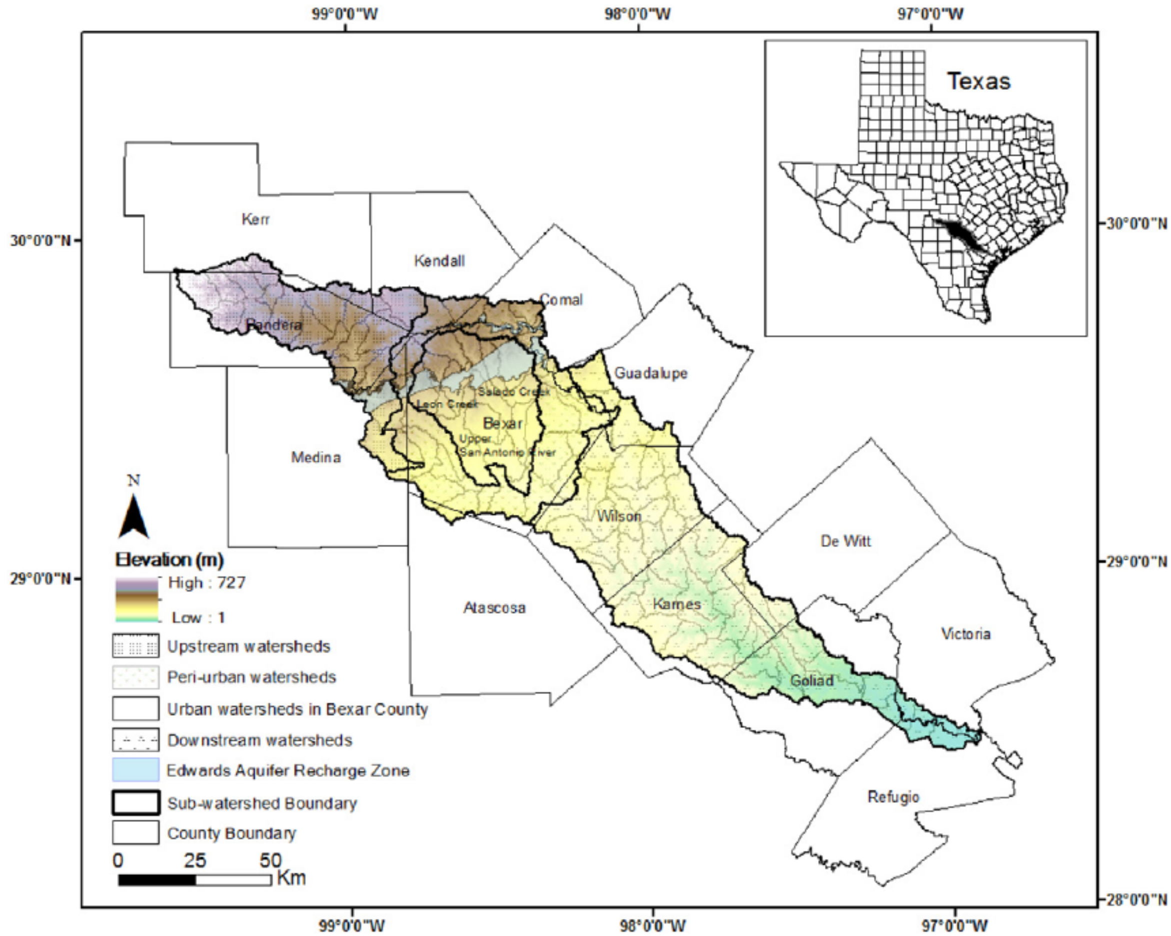

Fig. 1. The San Antonio River Basin (SARB) and sub-watersheds.

quantifies the biodiversity and terrestrial ecosystem services to estimate the impacts associated with land changes (Tallis and Polasky, 2009). We used land-use and land-cover maps of the SARB from 1984, 1995, and 2010 (Yi et al., 2017) and tabular data, together with environmental information including soil, topography, and climate (Appendix Tables A2–A6) to generate spatially-explicit biophysical supply of ecosystem services and habitat quality under the temporal land changes at multiple scales (Kareiva et al., 2011). Specifically, we modeled biodiversity, carbon storage, and sediment retention using a 30 × 30-meter spatial grid to quantify and compare the BES relationships between the entire SARB and the urban watersheds in Bexar County (i.e., regional BES and BES in rapidly urbanizing watersheds), and we also conducted analyses of average per-hectare estimates for individual sub-watersheds.

2.2.1. Biodiversity

Habitat quality is used in InVEST as a proxy for biodiversity (Eqs. (1)–(2)). It is a function of the land class in the grid cell and the sensitivity of the habitat to threats that are posed by the surrounding land classes (Tables A3–A4). Specifically, habitat quality depends on 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). We considered high-density urban, low-density urban, and agricultural land as potential threats in the SARB.

$$D_{xj} = \sum_{r=1}^{R} \sum_{y=1}^{Y_r} \left(\frac{w_r}{\sum_{r=1}^{R} w_r} \right) r_y i_{rxyx} S_{jr} \tag{1}$$

$$Q_{xj} = H_j \left(1 - \left(\frac{D_{xj}^z}{D_{xj}^z + k^z} \right) \right) \tag{2}$$

where D_{xj} is the total threat level in a grid cell x of habitat type j, and w_r indicates the relative negative impact of a threat. R indexes all modeled

Table 1
Data and InVEST model characterization of biodiversity, carbon storage, and sediment retention.[a]

	Method	Unit	Purposes	Data for the analysis
Biodiversity	Habitat quality index	Unitless	To assess the conditions of habitats	Land class Threat table
Carbon storage	Sum of four carbon pools	Mg C/ha	To estimate terrestrial carbon stocks	Land class Carbon pool table
Sediment retention	Sediment delivery ratio	tons ha^{-1}	To estimate retained sediment	land use land cover DEM, K factor, R factor C and P factor table

[a] Each table provides the tabular information for the input variables of BES quantification.

degradation sources, and y indexes all grid cells on r's raster map. Y_r indicates the set of grid cells on r's raster map. The impact of threat r that originates in grid cell y, r_y, on habitat in grid cell x is given by i_{rxy}. β_x indicate the level of accessibility in grid cell x, where 1 indicates complete accessibility. S_{jr} indicates the sensitivity of land class (i.e., habitat type) j to threat r, where values closer to 1 indicate greater sensitivity. Q_{xj} is the quality of habitat in the grid cell x, and H_j is Boolean map based on the user's definition by which land class can provide habitat for the conservation objective. z ($z = 2.5$) and k are scaling parameters, and half saturation constant is 0.5 (Sharp et al., 2016).

2.2.2. Carbon storage and valuation

The carbon storage model in InVEST estimates the amount of carbon stored in a landscape based on the carbon density of each land class (Eq. (3); Table A5).

$$TCS = A_k \times (C_{ak} + C_{bk} + C_{sk} + C_{dk}) \tag{3}$$

where terrestrial carbon storage (TCS) (Mg C/ha per year) is carbon stored in the study area for a particular year. A_k is area (ha) for land use category 'k' and carbon density consists of four factors: 1) C_{ak} is the aboveground carbon density (Mg C/ha per year); 2) C_{bk} is the belowground carbon density (Mg C/ha per year); 3) C_{ck} is the soil carbon density (Mg C/ha per year); 4) C_{dk} is the dead mass carbon density (Mg C/ha per year).

We assumed the equilibrium or steady-state level for carbon storage of each land class, which means the changes in our carbon-storage estimates are linearly related to the land changes from 1984 to 2010 (Polasky et al., 2011; Sallustio et al., 2015). The economic value of carbon storage was estimated in terms of the social cost of carbon (SCC). The SCC is based on the axiom that it works as a Pigouvian tax that could be placed on the over-production of CO_2, thereby internalizing the social costs associated with carbon emissions (Pigou, 1920). Thus, the value of SCC reflects the marginal cost of CO_2 damage to society. We used 37 US$/tonne of carbon as the estimated SCC (OIRA, 2013), and measured the value of carbon storage in each year in terms of 2015 constant US dollars.

2.2.3. Sediment retention

We 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 unit cell scale and calculates the SDR that reaches the stream. The amount of annual soil loss in a grid cell I, $USLE_i$ (tons·ha^{-1} year^{-1}), is estimated by the revised universal soil loss equation (RUSLE) (Eq. (4)).

$$USLE_i = R_i \times K_i \times LS_i \times C_i \times P_i \tag{4}$$

where R_i is the rainfall erosivity index (MJ·mm·(ha·h)$^{-1}$), K_i is the soil erodibility (tons·ha·h·(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. An index of connectivity (IC) is calculated by the upslope contribution and flow path to the stream (Vigiak et al., 2012). The sediment delivery ratio for a grid cell i, SDR_i, is computed from the index in terms of a sigmoid function (Eq. (5)) (Sharp et al., 2016).

$$SDR_i = \frac{SDR_{\max}}{1 + \exp\left(\frac{IC_0 - IC_i}{k}\right)} \tag{5}$$

$SDR_{\max}$ is the maximum theoretical SDR, indicating the maximum proportion of soil loss that reaches the stream. We used a value of 0.3, based on observational data in the SARB (USDA, 2016). To estimate the flow accumulation threshold (tfac), we utilized the ArcHydro tool to obtain a raster of flow accumulation from the digital elevation model (DEM) (ESRI, 2016). Then, we overlaid the stream network from the National Hydrographic Dataset (NHD, 2016) (http://nhd.usgs.gov) to match the corresponding threshold. A threshold value of 300 pixels generated the best fit with >90% congruence. IC_0 and k_b indicate two calibration parameters to define the relationship between the index of connectivity and sediment delivery ratio (Vigiak et al., 2012). The sediment load from a unit grid cell i, E_i (tons·ha^{-1} year^{-1}), is estimated by annual soil loss and sediment delivery ratio (Eq. (6)).

$$E_i = USLE_i \times SDR_i \tag{6}$$

The model produces three main outputs: 1) total amount of sediment exported from each pixel that reaches the stream (tons/pixel); 2) total potential soil loss per pixel in the land class calculated from the USLE equation (tons/pixel); 3) sediment retention, based on the difference in the amount of sediment in terms of the original watershed and a hypothetical watershed where all land classes are supposedly bare soil (tons/pixel) (Sharp et al., 2016). We derived the values for the cover-management factor (C) and support factor (P) for each land-cover type from Wischmeier and Smith (1978) and Hamel et al. (2015). We used the default value of 0.5 for IC_0, and adjusted value of 1.3 for k_b to calibrate parameters for SDR-IC relationship. The estimate of 431,261 (tons/year) for sediment export in 2010 from InVEST (Table A8) is in agreement with the sediment-transport data amounting to ~449,000 (tons/year) collected at the confluence of the San Antonio River with the Guadalupe River (Banta and Ockerman, 2014).

2.3. Analysis of spatio-temporal associations among biodiversity and ecosystem services

We analyzed the spatio-temporal associations among biodiversity and ecosystem services across the entire SARB and the urban watersheds in Bexar County. To determine whether our data meet the normality assumption, we reviewed the Quantile-Quantile (Q-Q) plots and conducted the Shapiro-Wilk normality test, which indicated that the data are not normally distributed and exhibit non-linear patterns from 1984 to 2010 ($p < 0.05$). Due to the non-normality in the data, we applied non-parametric statistical tests.

We first applied the Kruskal-Wallis test to identify statistically-significant differences in land change-related temporal shifts in biodiversity, carbon storage, and sediment retention within four geographically distinct areas of the SARB (i.e., upstream, peri-urban, urban, and downstream watersheds). For post-hoc comparisons, we conducted the Mann-Whitney U test to test the statistical differences between pairs of geographic areas. Second, we applied the Wilcoxon signed rank test to examine the impact of urban land expansion within the four geographic areas on biodiversity and ecosystem services during the pre- (1984–1995) and the post-NAFTA (1995–2010) periods.

Third, we calculated Spearman's ρ across 107 sub-watersheds in the SARB and 16 sub-watersheds in Bexar County to estimate the temporal changes at multiple spatial scales. We used spider diagrams to illustrate comparative associations between biodiversity and ecosystem services (DeFries et al., 2004). To this end, we normalized BES values from 0 to 1 to better capture BES interactions in terms of synergies or trade-offs. Finally, to identify the biodiversity and ecosystem services hotspots and overlaps at the sub-watershed level, we used the Getis-Ord G_i^* statistic (Getis and Ord, 1992). We used zonal statistics to calculate the mean BES values in each sub-watershed, which we then used to examine statistically-significant spatial clusters of high values (hotspots) and low values (cold spots) of BES, as well as the degrees of overlap among them (Egoh et al., 2009; Reyers et al., 2009).

3. Results

3.1. Land changes in the SARB, Bexar County, and the EARZ

There was substantial urban growth between 1984 and 2010 in the SARB, particularly around San Antonio, while rangeland and forest areas, declined markedly (Table 2; Yi et al., 2017). In the urban watersheds in Bexar County, forest land, the largest land class in 1984, and rangelands also decreased substantially by 2010; by contrast, the combined area of the two urban land classes more than tripled by 2010 and became the largest. In the SARB portion of the Edwards Aquifer Recharge Zone (EARZ), urban growth was more than twice as rapid as that in the SARB or in Bexar County and, by 2010, over 20% of the land was either low- or high-density urban space (Tables 2, A1, Fig. A1).

3.2. Quantitative analyses of BES estimates and spatial distribution

Overall, biodiversity and ecosystem services spatially varied substantially across the SARB and showed distinct geographic distributions ($p < 0.01$) (Table A7). Within upper portion of the SARB, biodiversity decreased between 1984 and 2010, particularly around San Antonio (Fig. 2). Compared to the pre-NAFTA period, during the post-NAFTA period the annual rate of biodiversity loss was 0.1% higher in the SARB and 0.2% higher in Bexar County (Table 3).

Our assessment also identified a decrease in carbon stocks in the upper part of the SARB, especially in and around the urban watersheds (Fig. 2). The total carbon storage decreased by 4.0% and 7.8% during the pre- and post-NAFTA periods, respectively, amounting to a 0.1% greater annual decrease in the latter period and a total decline of 9.8 million Mg C from 85.6 million Mg C in 1984 (Table 3). In the urban watersheds in Bexar County, the rate of decrease in carbon storage was greater than in the SARB. It dropped by 3.0 million Mg C from 13.7 million Mg C in 1984, and the rate of decrease was 1.3% per annum greater during the post-NAFTA period (Table 3).

Based on the InVEST parameters, total amount of sediment retention was found to change very little from 1984 to 2010 both in the SARB and in Bexar County (Table 3; Fig. 2). The change in the total sediment export, as expected, was equal to the change in sediment retention. The change in sediment export was, however, not consistent across the SARB; sediment export was greater in the steeper terrain of the upper reaches of the drainage basin and lower in the flatter middle reaches of the drainage basin (Fig. A2). Although overall sediment export throughout the study period increased with a rate of 0.7% per year, it decreased at a rate of 0.2% per year during the pre-NAFTA period, but then increased 1.4% per year during the post-NAFTA period. Total sediment export in the urban watersheds of Bexar County also increased throughout the study period, but the pattern of change was different than that across the SARB; in the urbanized areas, the annual rate of change was slower during the post- than the pre-NAFTA period (Table A8).

3.3. Carbon storage valuation

Based on the ecological production function method, the estimated monetary value of carbon storage decreased during our study period in the SARB and especially in the urban Bexar County (Table 4). Moreover, the estimated value of carbon storage in the SARB decreased twice as fast after NAFTA came into effect. The total value of carbon storage in constant 2015 US$ decreased by $363.5 million in the SARB during our study period, with 4.0% and 7.8% decreases during the pre- and post-NAFTA periods, respectively. In Bexar County, the value of carbon storage decreased by $112.6 million during the same time period with a 1.1% decrease during the pre-NAFTA period and a remarkably high 21.3% decrease during the post-NAFTA period.

3.4. Land change impacts on BES in the four sub-watersheds

Our study revealed statistically significant differences in all three years (i.e., 1984, 1995, and 2010) among the four sub-watersheds (i.e., upstream, peri-urban, urban, downstream watersheds) in the SARB in terms of biodiversity, carbon storage, and sediment retention (Fig. 3). The Kruskal-Wallis test indicated the statistically significant differences ($p < 0.01$) in biodiversity ($\chi^2 = 16.01$, 23.12, and 25.49, respectively), carbon storage ($\chi^2 = 30.38$, 43.61, and 27.95, respectively), and

Table 2
Total area (ha) and the percent cover of each land class in the SARB (a) and in the EARZ of the SARB (b).

Land class	Total area					
	1984 (ha)	1984 (%)	1995 (ha)	1995 (%)	2010 (ha)	2010 (%)
(a)						
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 (5127)	1.4 (3.2)	30,783 (17,227)	2.8 (10.9)	58,165 (33,965)	5.4 (21.5)
Agricultural land	111,835 (8752)	10.3 (5.5)	104,841 (9418)	9.7 (6.0)	92,611 (3756)	8.5 (2.4)
Pasture	173,895 (10,245)	16.0 (6.5)	193,128 (8919)	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	3672 (82)	0.3 (0.1)	4267 (133)	0.4 (0.1)	4015 (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 (2111)	1.1 (1.3)	12,109 (2115)	1.1 (1.3)	3345 (1622)	0.3 (1.0)
No data	807 (2)	0.1 (0.0)	1350 (0)	0.1 (0.0)	1563 (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)
(b)						
Urban in the EARZ	1556	3.3	3481	7.2	9926	20.7
Low density urban	1082	2.3	2316	4.8	5687	11.9
High density urban	474	1.0	1165	2.4	4239	8.8
Agricultural land	1236	2.6	1509	3.2	493	1.0
Pasture	1897	4.0	1361	2.8	2550	5.3
Rangeland	14,365	30.0	8647	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

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

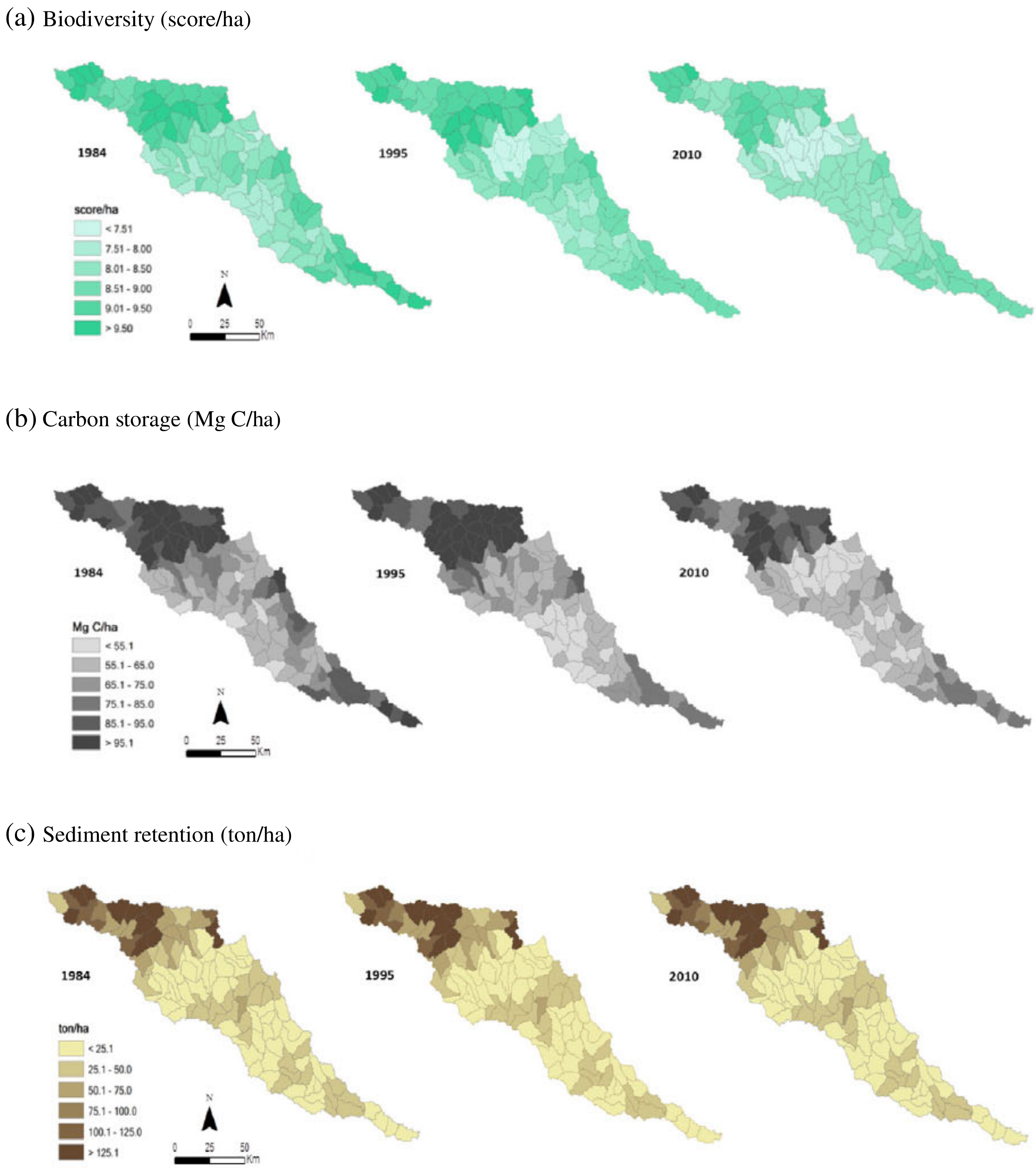

Fig. 2. Spatial distributions of biodiversity (score/ha) (a), carbon storage (Mg C/ha) (b), and sediment retention (ton/ha) (c) in the SARB from 1984 to 2010.

Table 3
Changes in biodiversity, carbon storage, and sediment retention in the SARB.

Biodiversity ecosystem services	BES estimates per year			1984–1995			1995–2010			1984–2010		
	1984	1995	2010	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
	(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)	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
	(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)	57,373,771	57,379,906	57,303,720	6135	0.0	0.0	−76,186	−0.1	0.0	−70,051	−0.1	0.0
	(4,591,570)	(4,584,057)	(4,574,630)	(−7513)	(−0.2)	(0.0)	(−9427)	(−0.2)	(0.0)	(−16,940)	(−0.4)	(0.0)

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

Table 4
Changes in the monetary values of carbon storage in the SARB and the urban watersheds in Bexar County.

	Carbon value (2015 US million $ per year)			1984–1995		1995–2010		1984–2010	
	1984	1995	2010	Change	Annual change	Change	Annual change	Change	Annual change
SARB	3169.7	3043.0	2806.2	−126.7	−11.5	−236.7	−15.7	−363.5	−13.9
Bexar County	508.8	503.3	396.1	−5.4	−0.4	−107.1	−7.1	−112.6	−4.3

sediment retention ($\chi^2 = 42.70$, 42.62, and 42.65, respectively) within four geographically distinct areas of the SARB (Table A9). The post hoc Mann-Whitney *U* test determined the statistical differences between pairs of geographic areas. Estimates of biodiversity, carbon storage, and sediment retention were significantly greater in the upstream sub-watershed than in the other three sub-watersheds in each year (i.e., 3 > 1, 2, 4), whereas differences among the peri-urban, urban, and downstream sub-watersheds were not statistically significant for either biodiversity or any of the ecosystem services (Table A9). The differences in BES between the upstream and other sub-watersheds are mainly associated with the biophysical characteristics and the dominant land cover in each sub-watershed.

The variation in low-density urban area is statistically significant ($p < 0.01$) in three sub-watersheds (i.e., upstream, peri-urban, and downstream), which indicates low-density urban expansion intensified during the post-NAFTA period of the study (1995–2010) (Tables 5, A10). The results also indicate that growth of low-density urban land is prevalent beyond the boundary of the urban watersheds in Bexar County. Furthermore, peri-urban watersheds with the most significant variation in both low- and high-density urban classes experienced the most

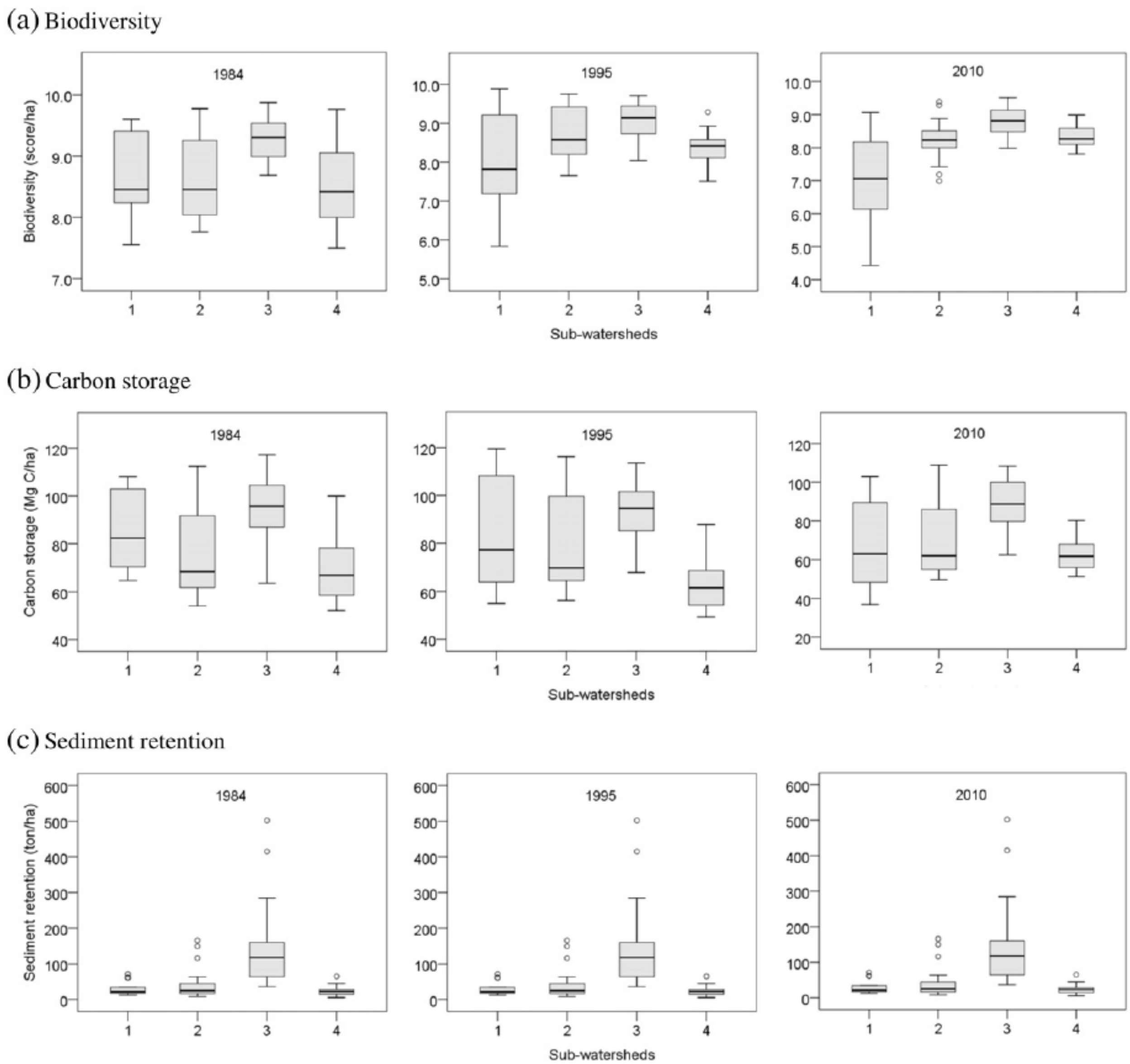

Fig. 3. Boxplots of biodiversity (a), carbon storage (b), and sediment retention (c) for each year (1984, 1995, 2010) in the four sub-watersheds of the SARB (box: 25–75 percentiles, line: median, whiskers: minimum and maximum values, circles: individual outliers). Numbers from left to right indicate the four sub-watersheds, respectively (1: urban watersheds (n = 16), 2: peri-urban watersheds (n = 22), 3: upstream watersheds (n = 20), 4: downstream watersheds (n = 49)).

Table 5
Wilcoxon signed rank test for urban classes and BES estimates between 1984–1995 and 1995–2010 (pre- and post-NAFTA periods, respectively).

	Urban watersheds		Peri-urban watersheds		Upstream watersheds		Downstream watersheds	
	z	p	z	p	z	p	z	p
Low density urban 1995 to 2010 vs. 1984 to 1995	−1.581	0.119	−3.937	0.000**	−3.762	0.000**	−4.080	0.000**
High density urban 1995 to 2010 vs. 1984 to 1995	−0.801	0.440	−3.571	0.000**	−1.134	0.453	−1.000	0.317
Biodiversity 1995 to 2010 vs. 1984 to 1995	−0.535	0.614	−3.736	0.000**	−2.237	0.025*	−2.738	0.005**
Carbon storage 1995 to 2010 vs. 1984 to 1995	−2.792	0.003**	−4.075	0.000**	−1.923	0.054	−5.272	0.000**
Sediment retention 1995 to 2010 vs. 1984 to 1995	−0.369	0.833	−2.994	0.002**	−2.553	0.009**	−0.290	0.772

* $p < 0.05$.
** $p < 0.01$.

significant changes during the pre- and post-NAFTA periods in biodiversity, carbon storage, and sediment retention estimates compared to the other three sub-watersheds (i.e., upstream, urban, and downstream) ($p < 0.01$). Importantly, these empirical results demonstrate that biodiversity loss, degradation of carbon storage, and sediment retention function were negatively related to the urban expansion in the SARB and have accelerated since the inception of NAFTA in 1994.

3.5. Spatial and temporal associations among biodiversity and ecosystem services

Normalized spider diagrams indicate the comparative relationships among biodiversity, carbon storage, and sediment retention estimates and synergistically summarize their temporal dynamics during the study period (Fig. 4). Moreover, the diagrams highlight the BES relationships in urban watersheds in Bexar County compared to the regional BES relationships in the entire SARB. 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. These declines in biodiversity and carbon storage were especially marked during the post-NAFTA period, and were more pronounced in the Bexar County than the entire SARB, especially with respect to biodiversity. By contrast, the spider diagrams show that sediment retention remained relatively stable in both the SARB and Bexar County. The correlations for biodiversity–carbon storage, biodiversity–sediment retention, and carbon storage–sediment retention are positive at both spatial scales of analyses over time ($p < 0.01$, except for carbon storage–sediment retention in Bexar County where $p < 0.05$) (Table A11).

Using Hotspot Analysis (Getis-Ord G_i^*), we identified spatially-clustered sub-watersheds in terms of biodiversity, carbon storage, and sediment retention (Fig. 5). Statistically-significant hotspots were mostly identified in the upstream watersheds, which were mainly covered by forests/woodlands during our study period. By contrast, most cold spots were located in and around the urban watersheds and agricultural lands in the central portions of the SARB. Table 6 presents the extent of hotspots at ≥90% confidence level and hotspot overlaps among biodiversity, carbon storage, and sediment retention throughout the SARB.

Proportional overlap between hotspots measures the shared area among biodiversity and multiple ecosystem services expressed as a percentage of the study area (Prendergast et al., 1993; Reyers et al., 2009). Our analyses indicate biodiversity, carbon storage, and sediment retention hotspots are confined to a few areas in the upstream sub-watersheds of the SARB. Sediment export indicated similar patterns across the SARB (Fig. A2). During our study period, the percent cover of biodiversity hotspots decreased from 28% to 10% in the SARB and from 32% to zero in Bexar County. In the SARB, percent cover of carbon-storage hotspots increased little (24% to 25%), but decreased substantially (46% to 32%) in Bexar County. Sediment-retention hotspots were relatively stable at 13% in the SARB, but no hotspots were identified in Bexar County. We found 20% and 9.5% overlaps in carbon storage and biodiversity hotspots and 11.7% and 7.6% overlaps in sediment retention and biodiversity hotspots in 1984 and 2010, respectively. This implies that the overlapping watersheds at HUC 12 level significantly decreased by 2010 at multiple scales, indicating biodiversity and ecosystem services are not only positively related but also spatially linked over time.

4. Discussion

A fundamental premise of the Millennium Ecosystem Assessment is that biodiversity and ecosystem services are key determinants of long-term sustainability of social-ecological systems (MEA, 2005). With a continuing decline in local and global biodiversity and ecosystem services, it is crucial to understand how biodiversity and various ecosystem services interact and how land change may modify these interactions over time. In this study, we employed the spatially-explicit ecological production function method (EPFM) using InVEST to quantify

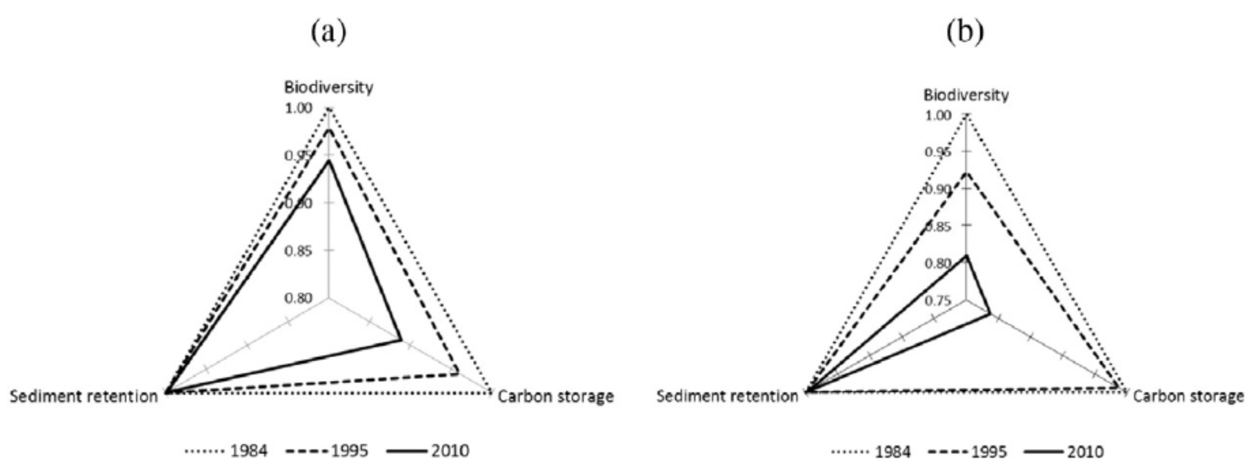

Fig. 4. Spider diagrams for comparative BES relationships between 1984 and 2010 in the SARB (a) and Bexar County (b).

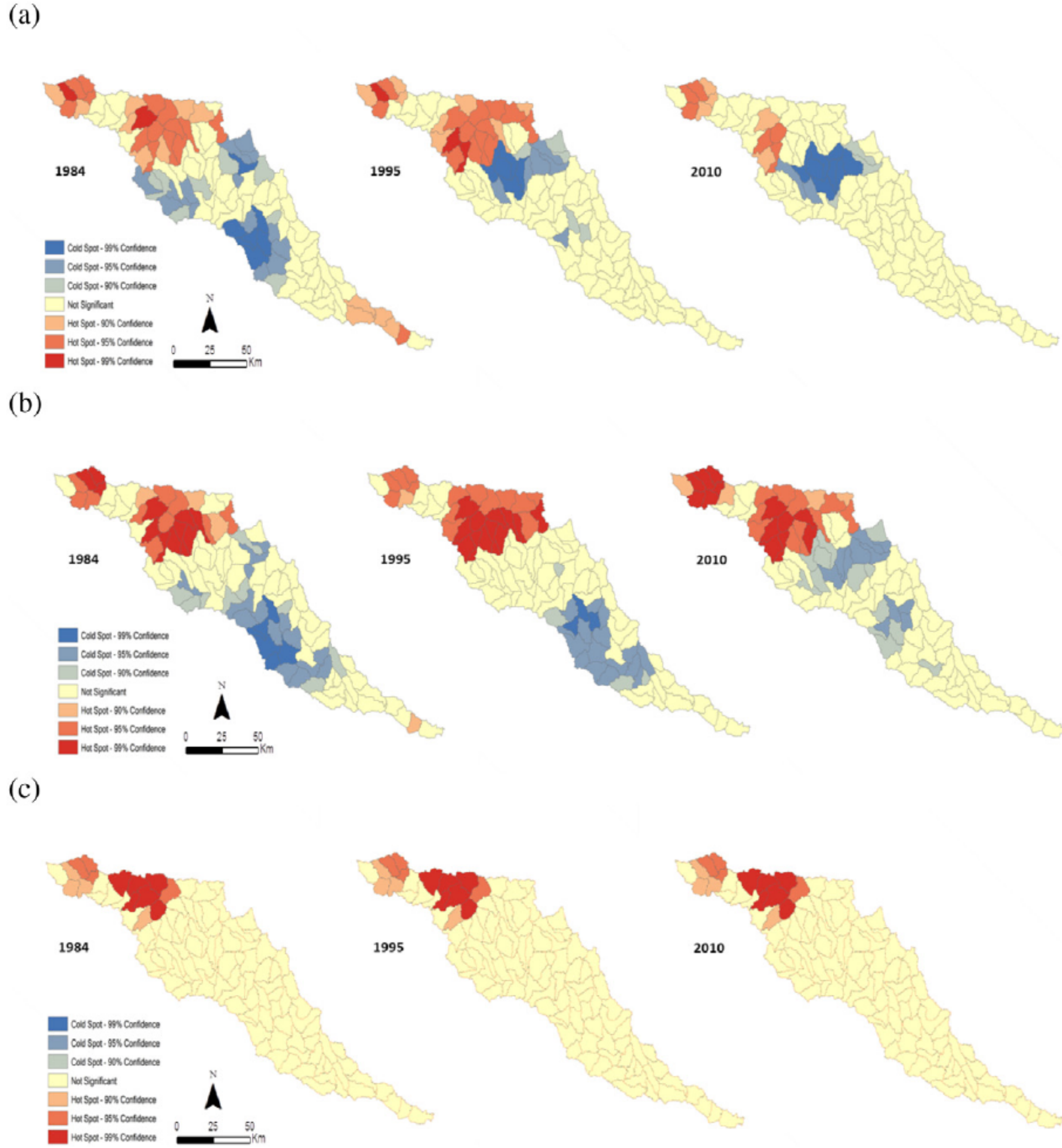

Fig. 5. Hotspots and cold spots of biodiversity (a), carbon storage (b), and sediment retention (c) in the SARB from 1984 to 2010.

biodiversity and selected ecosystem services, including carbon storage and sediment retention, and examined spatio-temporal associations among them at multiple scales in the rapidly urbanizing SARB. Our findings point out that degradation of biodiversity and ecosystem services is mainly associated with the expansion of low-density urban land, specifically in the peri-urban watersheds around the Bexar County which experienced the most significant changes after the implementation of the NAFTA. Of particular concern are the post-NAFTA declines in

Table 6
BES hotspots and overlap analyses in the SARB and the urban watersheds in Bexar County.

Hotspot extent and overlap (ha, %)	1984		1995		2010	
	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.

biodiversity and carbon storage. Since the ecosystem function of carbon storage is critical for CO_2 sequestration and climate regulation, carbon markets and payments by developers for these ecosystem services should be given serious consideration by the governments to facilitate the low-carbon and climate-resilient economy (Engel et al., 2008; UNEP, 2011; Brookings Institution, 2016). In this context, Nelson et al. (2009) recommended payments for carbon sequestration to moderate trade-offs between the impacts of pro-development policies and the need to retain biodiversity and multiple ecosystem services for the well-being of current and future generations.

The portion of the Edwards Aquifer recharge zone (EARZ) within the SARB experienced a six-fold increase in urban land at the rate of 20.7% growth per year from 1984 to 2010, indicating more than double the rate of urban growth in the entire SARB (8.0%) and in Bexar County (7.9%). Forest/woodland was the land-cover type that was the most affected by urban expansion in this environmentally-sensitive zone. The EARZ is critical for replenishing the aquifer upon which the City of San Antonio depends predominantly for its water supply, and it is more prone to flash flooding and groundwater contamination due to the proliferation of impervious surfaces (GEAA, 2014; Earl and Vaughan, 2015). In addition, the EARZ provides habitat for several endangered species, including the nine karst invertebrates in Bexar County (USFWS, 2012). Moreover, the upper portion of the SARB contains habitat for two species of bird, the Black-capped Vireo (*Vireo atricapillus*) and the Golden-cheeked Warbler (*Setophaga chrysoparia*), which were federally-listed under the US Endangered Species Act in 1987 and 1990, respectively, in part due to habitat loss driven by urban sprawl in the upper SARB (Engels and Sexton, 1994; Campbell, 2003). The results provide the rationale for sustainable collective action to internalize these negative externalities associated with urban sprawl and increasing social costs to reduce the detrimental effects of environmental degradation in the coupled human and environment systems (Ostrom, 2000). Accordingly, policy interventions to effectively address urban expansion and to sustain bio-diverse ecosystems in the EARZ should be urgently implemented.

San Antonio-New Braunfels MSA is identified as one of the most sprawling large metro areas in the US (Ewing and Hamidi, 2014). Changes in hydrology caused by the expanding impervious surfaces increase surface runoff into streams, resulting in poor water quality and degraded aquatic habitats (Paul and Meyer, 2001) as well as increasing vulnerability to nonpoint source pollution, malfunctioning sewer infrastructure, and construction-related sedimentation (Chin, 2006). Such risks have led to the recommendation that, in order to maintain stream health, impervious land cover should not exceed 10% of total watershed area (Schueler, 1994). In our study, we found that urban land occupied within watersheds of the EARZ has been approaching this threshold and the "urban stream syndrome" (Walsh et al., 2005) is likely to become an increasingly serious issue as urban land cover continues to expand. This poses a major challenge for the City of San Antonio because poor water-filtration capacity of the karstic geology results in water quality in the aquifer being directly determined by water quality in the streams traversing the EARZ.

The San Antonio region constitutes the strategic transportation hub of the NAFTA. In particular, the Interstate Highway (IH)-35 corridor carries the largest portion of NAFTA-related traffic, with over 35% of total Texas truck vehicle miles traveled, and connecting San Antonio to Laredo, the largest land port of Texas, and other southern border areas. Moreover, it is estimated that NAFTA trade tonnage will more than double by 2030 to support the increasing exports from Texas (TxDOT, 2007). These expanding transportation networks to strengthen the US-Mexico supply chains for the economy of scale will likely further degrade the biodiversity and ecosystem services in the region through direct and induced land fragmentation, edge effects, air pollutant emissions, and water contamination (Forman and Alexander, 1998; Trombulak and Frissell, 2000; American Forests, 2002; AACOG, 2015).

The rapid growth of low-density urban space in the SARB, especially within the EARZ, is directly linked to urban sprawl beyond the City of San Antonio in Bexar County. Our findings suggest that sprawling low-density urban development in the region gained pace after the implementation of NAFTA, which results in carbon locked-in transportation networks and gray infrastructure-intensified economy with diminishing environmental quality (Ewing et al., 2003; Bhatta et al., 2010). In our study this was exemplified by reduced biodiversity and carbon storage capacity due primarily to the reduction and fragmentation of rangelands and forests/woodlands. In this context, mitigation and adaptation to regional climate change have been closely linked to policy interventions and strategies for green infrastructure, defined as a cost-effective and resilient approach to deliver multiple benefits at watershed scale compared to conventional gray infrastructure (Mell, 2010; USEPA, 2017). To limit carbon footprint induced by urban expansion and land fragmentation, investing in green infrastructure and urban green spaces is critical to improving ecological, social, and economic benefits, and contributing to sustainable development in the region (De Ridder et al., 2004; Benedict and McMahon, 2006; Gill et al., 2007; EEA, 2011).

Human well-being is tightly coupled with ecosystem services in social-ecological systems (Liu et al., 2007; Reyers et al., 2013), and sustainable development is interconnected with social, economic, ecological, and institutional dimensions (Daily et al., 2009; Ostrom, 2009; Fischer et al., 2015; Virapongse et al., 2016). As indicated previously, biodiversity and ecosystem services are often positively correlated (Turner et al., 2007; Nelson et al., 2009; Harrison et al., 2014), although this is not ubiquitous (Chan et al., 2006; Egoh et al., 2009). Our research in the SARB found that biodiversity was positively related to two ecosystem services, carbon storage and sediment retention, in all three years during our study period. Moreover, our results indicate that biodiversity and the two ecosystem services in Bexar County are more significantly impacted by continuing urban growth compared to BES in the SARB.

As the urban expansion occurs predominantly at the expense of biodiversity and many ecosystem services, an integrated regional program for smart and sustainable growth that offsets the loss of ecosystem services due to ongoing urban expansion is imperative in the SARB (Yi et al., 2017). Numerous instruments have been implemented in Texas and elsewhere to conserve wildlife habitat, water quality, and also carbon storage. These include publically funded and/or market-based programs aimed at reducing land fragmentation and protecting biodiversity and ecosystem services. Mechanisms for implementing such programs may include carbon pricing and taxes (Engel et al., 2008), voluntary perpetual conservation easements that remove development rights from private property ownership (Stroman and Kreuter, 2014, 2015), and conservation credit or habitat exchange programs (Kreuter et al., 2017). Here we focus on biodiversity conservation (protection of endangered species) as a proxy for the delivery of ecosystem services.

One informative example of an integrated endeavor to protect both water quality and wildlife habitat from the deleterious effects of urban sprawl, is the City of Austin's program for Water Quality Protection Land (WQPL; http://www.austintexas.gov/department/water-quality-protection-land) and the Balcones Canyonlands Preserve (BCP; http://www.austintexas.gov/bcp). Both programs were implemented following voter-approved municipal bonds to purchase land and acquire conservation easements within the contributing and recharge zones of the Barton Springs segment of the Edwards Aquifer. These bonds are recouped by the City of Austin through supplemental utility fees. Under the WQPL, land and conservation easements were acquired to protect water quality and aquatic habitat for two endangered salamander species associated with Barton Springs, while under the BCP the same was done to protect the endangered Golden-cheeked Warbler habitat from the ravages of urban development.

Conservation-credit and habitat-exchange programs have also been beneficially applied in Texas and in other states to protect and increase

endangered species habitat. Of specific relevance is the Recovery Credit System that was developed in central Texas to offset the deleterious effects of military-training activities on the Golden-cheeked Warbler habitat on Fort Hood (Kreuter et al., 2017). Under this program, the Department of Defense used federal funds to purchase term-limited (10–25 years) conservation credits offered for sale by surrounding landowners whose properties contained suitable habitat. This program was operationalized through a reverse bidding process whereby landowners offered their previously established credits for sale subject to their cost-sharing requirement for land improvement and the number of years they were willing to protect the habitat.

Our hotspot and cold spot analyses indicate that upstream watersheds, with predominantly forest/woodland land cover, contain proportionately more of the hotspots (statistically-significant spatial clusters of high values) for biodiversity, carbon storage, and sediment retention than the rest of the watershed. Paradoxically, these upper-basin watersheds also exhibited proportionately high sediment export characteristics, which is nevertheless consistent with previous work conducted in forested areas (Bogdan et al., 2016). One explanation for this might be that the upper watersheds have relatively shallow soils overlying karst geology that evolve under predominantly grassland cover but have been substantially overtaken by woody plants due to long-term fire suppression (Twidwell et al., 2013). In such cases, when grass cover declines due to increasing tree dominancy, sediment export may increase.

As the urban expansion and land conversion are expected to continue into the north-western part of the SARB within the contributing and recharge zones of the Edwards Aquifer, our results indicate that, in order to sustain the future well-being of residents in the region, upstream watersheds should be prioritized for the types of biodiversity and ecosystem services conservation programs discussed above. One option would be to develop a biodiversity and/or ecosystem services mitigation banking mechanism, similar to wetland mitigation banks. Under such a mechanism, developers would be required to purchase conservation credits to offset the long-term social costs of biodiversity and ecosystem service impacts of urban expansion and the infrastructural developments from which they are making short-term profits. In the long term, society cannot afford to subsidize the short-term profits of development without a mechanism to offset the deleterious impacts of accelerating expansion of urban areas.

Furthermore, our results suggest that the integration of various cost-effective green infrastructure in development plans is critical to ensure conservation and future resilience of both biodiversity and ecosystem services in the SARB (USEPA, 2017). Specifically, our assessment of BES relationships in the SARB in the wake of NAFTA's implementation provide a template for better incorporating green infrastructure in more environmentally friendly urban development. These empirical results create an analytical framework with respect to large-scale economic development initiatives, such as NAFTA, to better address the resilience and sustainability of social-ecological systems at watershed scales and to better inform environmentally friendly policy alternatives (e.g., conservation-credit and habitat-exchange program, and BES mitigation banking). Importantly, NAFTA is the first significant international trade agreement to include environmental side agreements. Through the North American Agreement on Environmental Cooperation, NAFTA includes provisions to promote environmental monitoring and cooperation (Villarreal and Fergusson, 2017). This cooperation agreement represents an important policy shift that incorporates regional economic growth within a social-ecological systems framework.

While this study is a first-of-a-kind analysis for spatio-temporal associations among BES at multiple scales and provides insights into potential synergies between biodiversity and ecosystem services as well of trade-offs between these biophysical relationships and urban characteristics (e.g., low- and high-density), challenges remain in reaching a more nuanced understanding of the spatial distributions of, and the relationships among, multiple ecosystem services (Qui and Turner, 2013). There is substantial collective evidence on high spatial congruence between biodiversity attributes and ecosystem services (Harrison et al., 2014). However, in cases where associations between biodiversity and ecosystem services present a more complicated dynamic, conclusive evidence may remain elusive (Lyytimäki and Sipilä, 2009). Therefore, more place-based studies with sensitivity analyses are needed to our further understanding of the dynamic interactions between biodiversity and ecosystem services (Ruckelshaus et al., 2015; Yi, 2017).

5. Conclusion

We present a novel multi-scale analysis of spatio-temporal associations between biodiversity, carbon storage, and sediment retention. We found that the spatio-temporal associations among biodiversity and these two ecosystem services were broadly positive and statistically significant in the SARB and in Bexar County. Our findings highlight that the land change in the SARB between 1984 and 2010 negatively impacted biodiversity, carbon storage and sediment retention, more significantly in the urban watersheds in Bexar County compared to the entire SARB. Moreover, urban land expansion in the peri-urban watersheds around Bexar County, especially near the EARZ in the upper portion of the SARB, led to particularly significant losses in biodiversity and degradation of these ecosystem services during the post-NAFTA period of our study (1995–2010).

Given the level of spatial congruence among, and the negative consequences of urban development on, BES across the SARB, green infrastructure and conservation policies are urgently needed to ensure the long-term social-ecological sustainability, especially in the upper portion of this drainage Basin. This requires the establishment of a development impact off-set mechanism, such as mitigation banking with conservation credits that are produced by land conservation in hotspots of highest potential for biodiversity, carbon storage, and sediment retention, which we identified as occurring predominantly in the upper portion of the SARB. Furthermore, our results suggest the strong need for green infrastructure policies that can integrate biodiversity conservation and sustainable use of multiple ecosystem services to address the environmentally deleterious impacts of the extensive land change in the region under the NAFTA.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2017.10.302.

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