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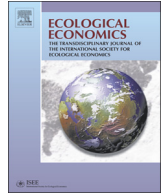


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# Impacts of Land Change on Ecosystem Services in the San Antonio River Basin, Texas, from 1984 to 2010



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## ABSTRACT

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). San Antonio, 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. To assess the environmental implications of this growth in the SARB, we first conducted a land-change analysis using Landsat images from 1984, 1995, and 2010. Then, we analyzed spatiotemporal changes in ecosystem services across the SARB and within three watersheds in Bexar County where the city of San Antonio is located. To estimate changes in ecosystem service values (ESV) during this period, we combined the results of the land-change analysis with a benefit transfer approach using two sets of widely cited ecosystem-service valuation coefficients published in 1997 and 2014 but we modified the urban coefficient from the 2014 publication for low-density and high-density urban areas. When 1997 coefficients were applied, the ESV in the SARB decreased, on average, by \$1.2 million/year during 1984–1995 and by \$1.8 million/year during 1995–2010. The ESV in Bexar County decreased, on average, by \$0.5 million/year and \$0.7 million/year during the first and second periods, respectively. When the 2014 coefficients and modified urban value coefficients were applied, the ESV in the SARB decreased, on average, by a 27% more during the first period than when the 1997 coefficients were applied, while, ESV increased during the second period by an average of \$2.2 million/year. This temporally opposite trend in ESV change did not occur in Bexar County, however. Using the 2014 coefficients, ESV in Bexar County decreased 5 times more during the first period and decreased 2.5 times more during the second period than when 1997 coefficients were applied. The differences in ESV trends resulting from the two sets of coefficients can be explained primarily by the different coefficients assigned to urban spaces (\$0/ha/year in the 1997 study and \$7005/ha/year in the 2014 study). Our results suggest that the value placed on urban areas in the 2014 publication, taken from a single case study and intended primarily for large urban parks, substantially overestimates the ESV of urban space. In our study areas, applying this value, even only to urban green space, led to the improbable conclusion that urbanization had a positive overall effect on the delivery of ecosystem services. While open spaces in urban areas do provide valuable ecosystem services, it is highly unlikely that their value exceeds those provided by less modified ecosystems. The ability to confidently use value coefficients when applying benefit transfer methods to estimate ESVs demands rigorous assessments of their broad applicability.

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## 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

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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.

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 land-change 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 specifically 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

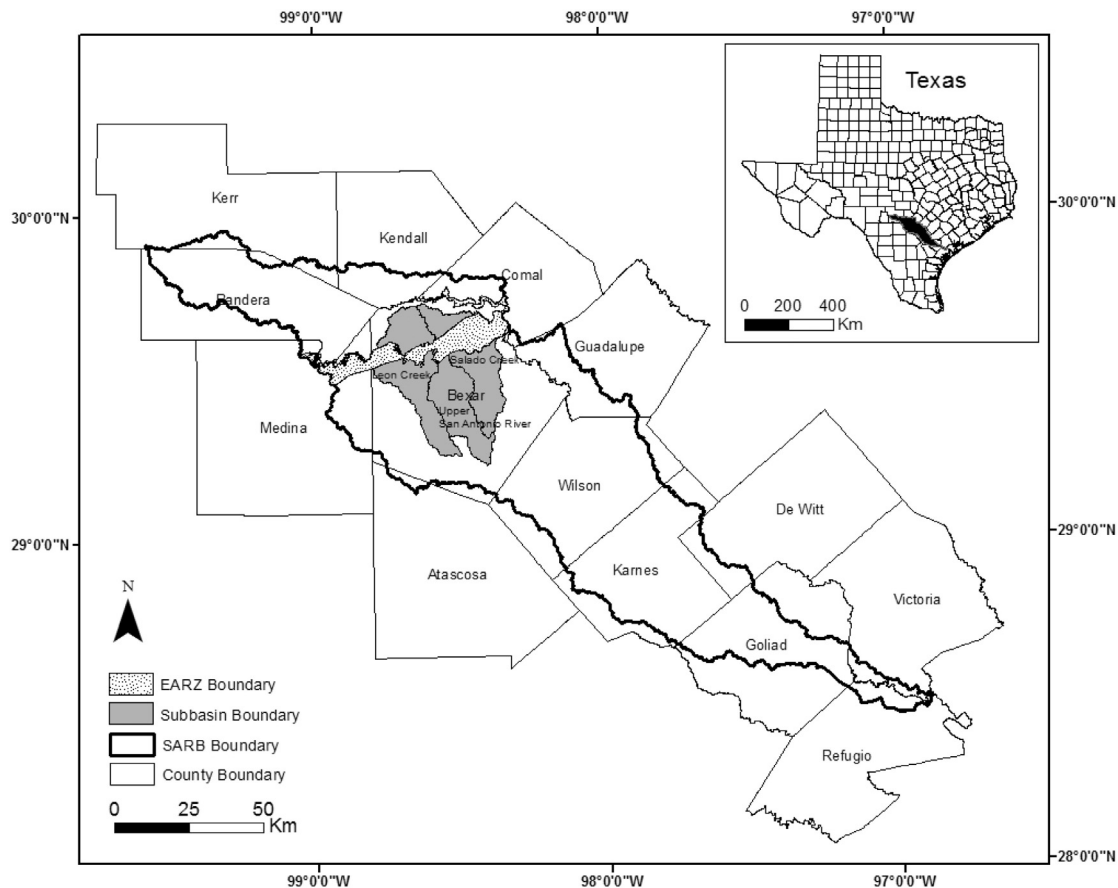


Fig. 1. San Antonio River Basin (SARB) and three watersheds containing Bexar County.

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. Methods

### 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<sup>2</sup> within 14 counties. It contains almost all of Bexar County (Fig. 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 (Fig. 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<sup>2</sup>. 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.

### 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 post-NAFTA 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 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 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 accuracy-assessment 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.

### 2.3. 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).

**Table 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 coefficients (2010 US\$/ha/yr)	2014 Modified coefficients (2010 US\$/ha/yr)	Percent difference from 1997 coefficients	<sup>b</sup> 1997–2014 Mean coefficients (2010 US\$/ha/yr)	Percent difference from 2014 modified coefficients
<sup>a</sup> Low density urban	Urban	0	5254	–	2627	–50.0%
<sup>a</sup> High density urban	Urban	0	1751	–	876	–50.0%
Agricultural Land	Cropland	132	5854	4334.8%	2993	–48.9%
Pasture	Grass/rangeland	337	4381	1200.0%	2359	–46.1%
Rangeland	Grass/rangeland	337	4381	1200.0%	2359	–46.1%
Forest Land	Temperate/boreal	438	3299	653.2%	1869	–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%

<sup>a</sup> Coefficients derived from urban coefficient in Costanza et al. (2014).

<sup>b</sup> 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:

$$ESV = \sum(A_k \times VC_k) \quad (1)$$

$$ESV_f = \sum(A_k \times VC_{fk}) \quad (2)$$

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). 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).

### 3. Results

#### 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 (Fig. 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 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

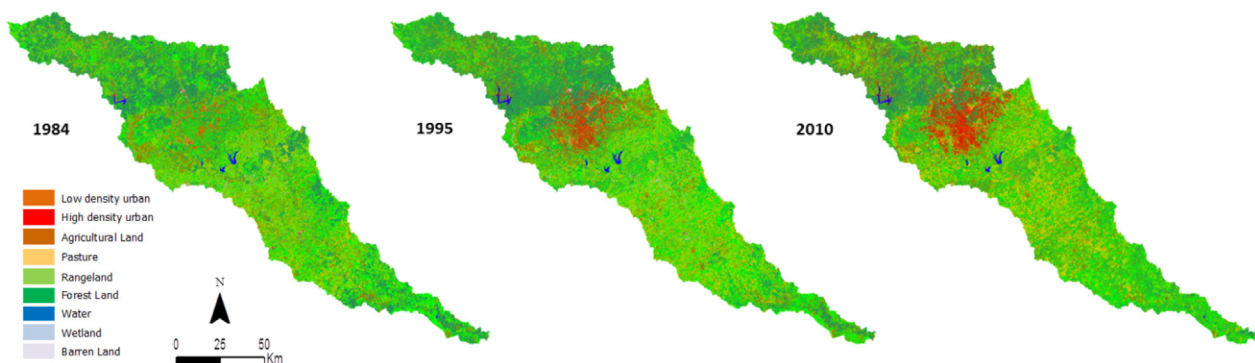


Fig. 2. Land change between 1984 and 2010 in the SARB.

**Table 3**  
Total estimated area (ha) and percent cover of each land class in the SARB from 1984 to 2010.

Land class	Total area (ha, %)					
	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	3672	0.3	4267	0.4	4015	0.4
Wetland	960	0.1	618	0.1	570	0.1
Barren Land	12,379	1.1	12,109	1.1	3345	0.3
No Data	807	0.1	1350	0.1	1563	0.1
Total	1,085,751	100	1,085,751	100	1,085,751	100

1.9% higher and that of high-density urban areas was 3.3% lower than during 1984–1995.

Rangeland and forest, which are the two largest land classes, declined markedly during our 26-year study period (Table 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.

In the three watersheds in Bexar County (Fig. 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 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

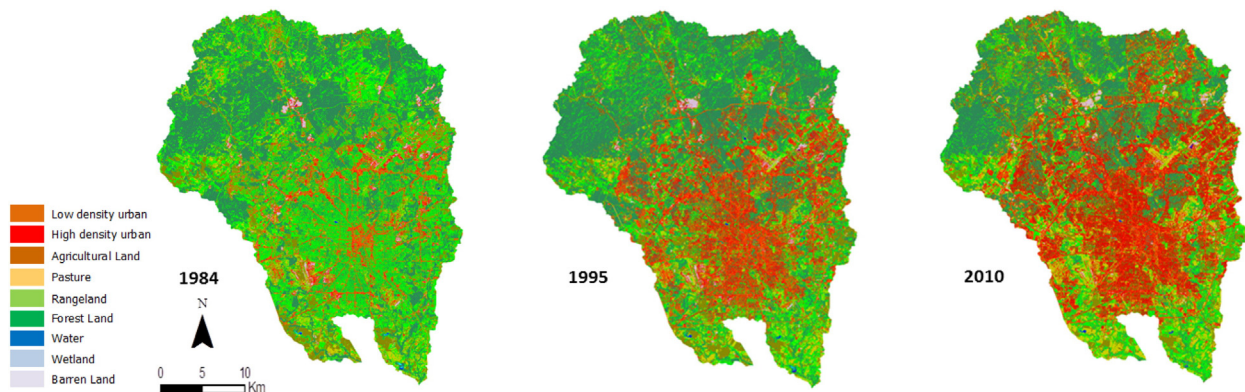
and 1995–2010 periods, respectively, and growth rates declined between the first and second time period for both high-density and low-density urban land (high-density growth decreased from 21.5% to 6.5% per annum and low-density decreased from 4.7% to 1.3% per annum). The combined area of the other land 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.

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.

### 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 (Fig. 4). When the 1997 coefficients 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 (Fig. 4).

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 (Fig. 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



**Fig. 3.** Land change between 1984 and 2010 in the three watersheds, Bexar County.

**Table 4**

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

Land class	Total area (ha, %)					
	1984		1995		2010	
	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	5127	3.2	17,227	10.9	33,965	21.5
Agricultural land	8752	5.5	9418	6.0	3756	2.4
Pasture	10,245	6.5	8919	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	2111	1.3	2115	1.3	1622	1.0
No data	2	0.0	0	0.0	0	0.0
Total	157,874	100	157,874	100	157,874	100

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).

The results from the sensitivity analysis indicate that estimated ESVs for both scales of analysis are relatively inelastic (i.e., CS substantially <1) (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 (<7%). 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), 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

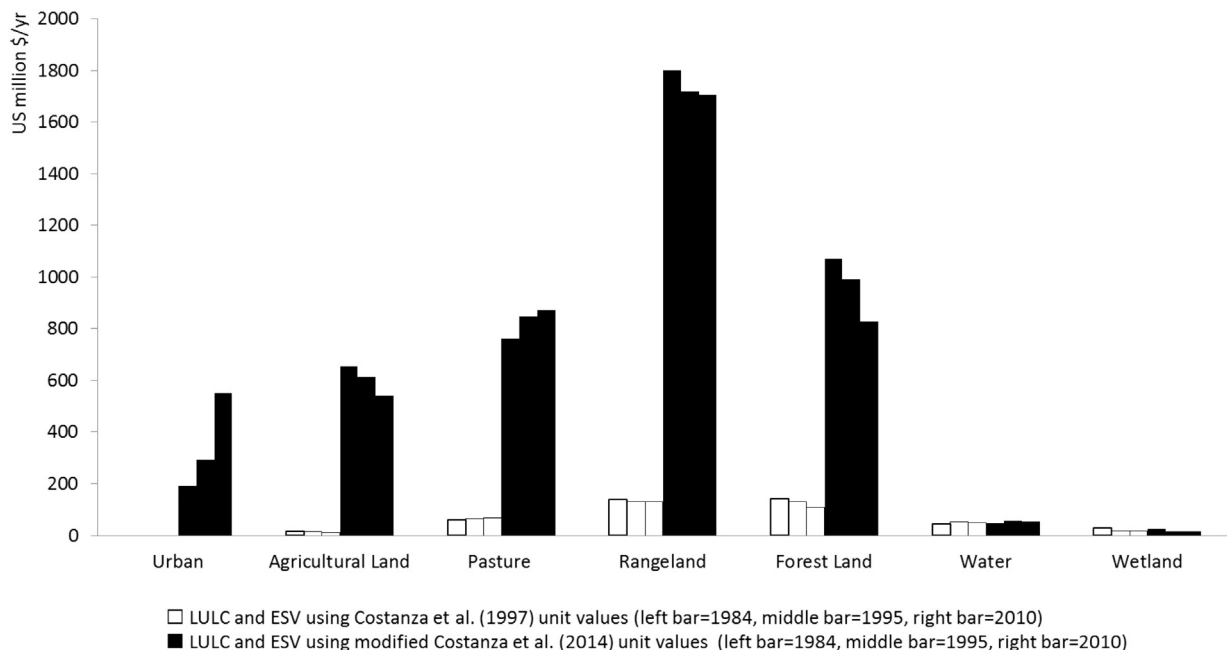
were generally lower than for the other land classes, and all CS values were  $\leq 0.40$ . Based on these sensitivity analyses, the ESV estimates for all three years of analysis (1984, 1995 and 2010) appear to be relatively robust.

### 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 (Figs. 6 and 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.

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.

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 (Figs. 6 and 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



**Fig. 4.** Changes in the ESV by land class between 1984 and 2010 in the SARB.

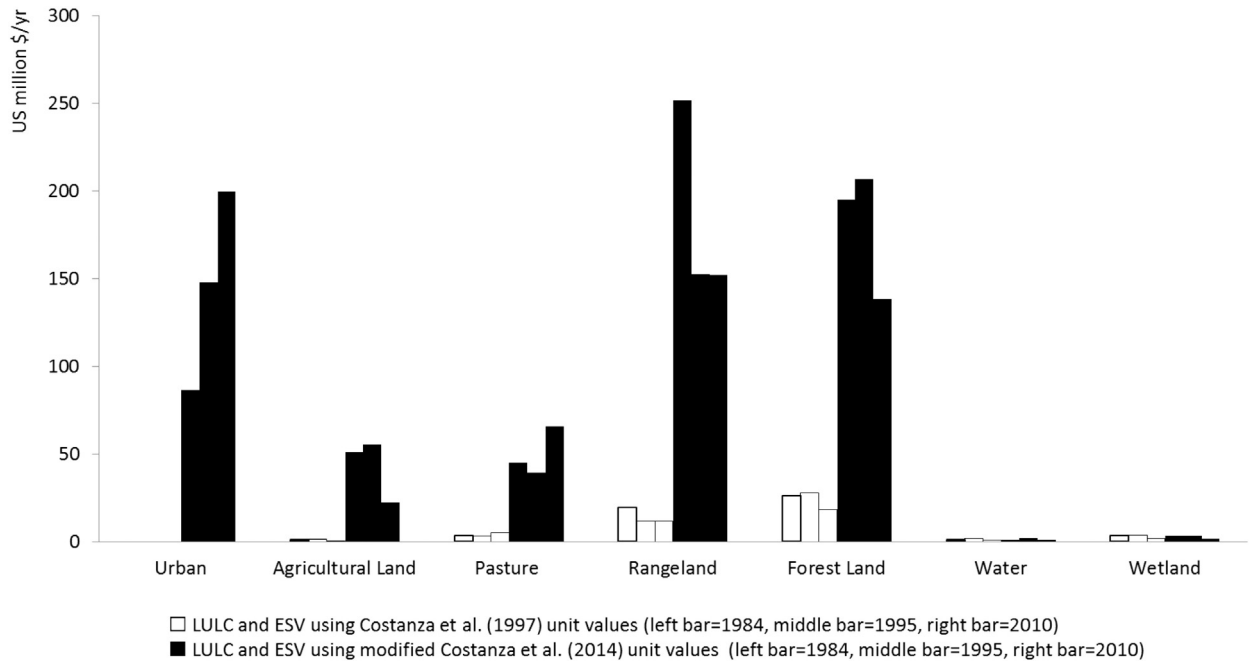


Fig. 5. Changes in the ESV by land class between 1984 and 2010 in the three watersheds, Bexar County.

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.

#### 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 policies.

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.,

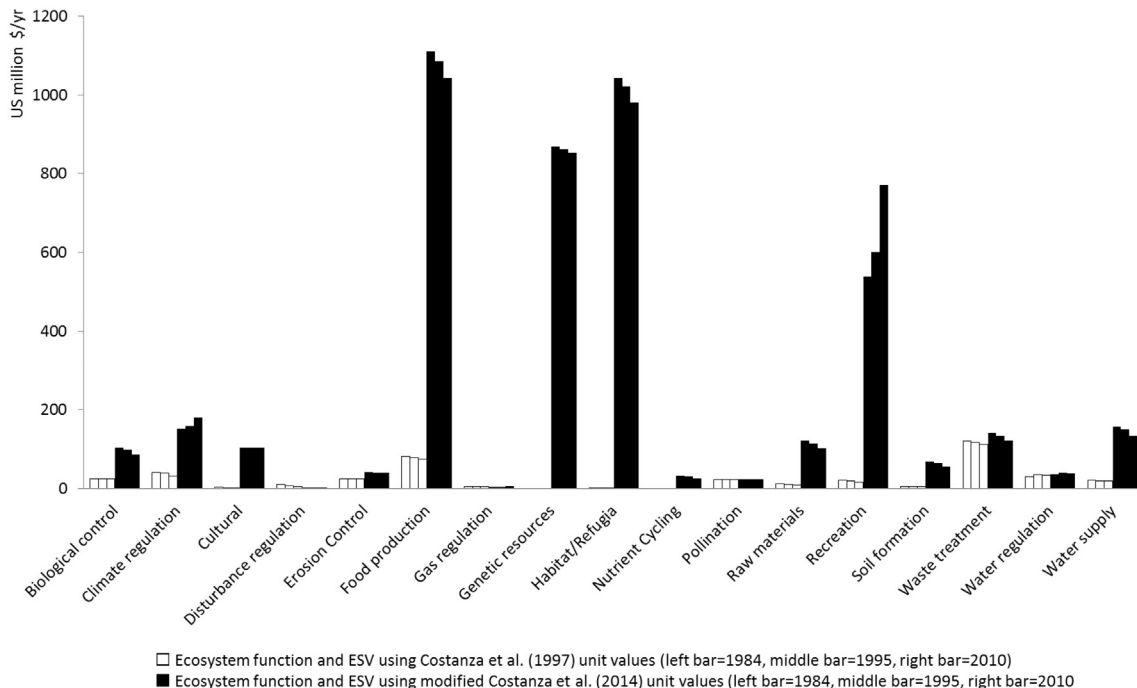


Fig. 6. Changes in ESV by functions between 1984 and 2010 in the SARB.



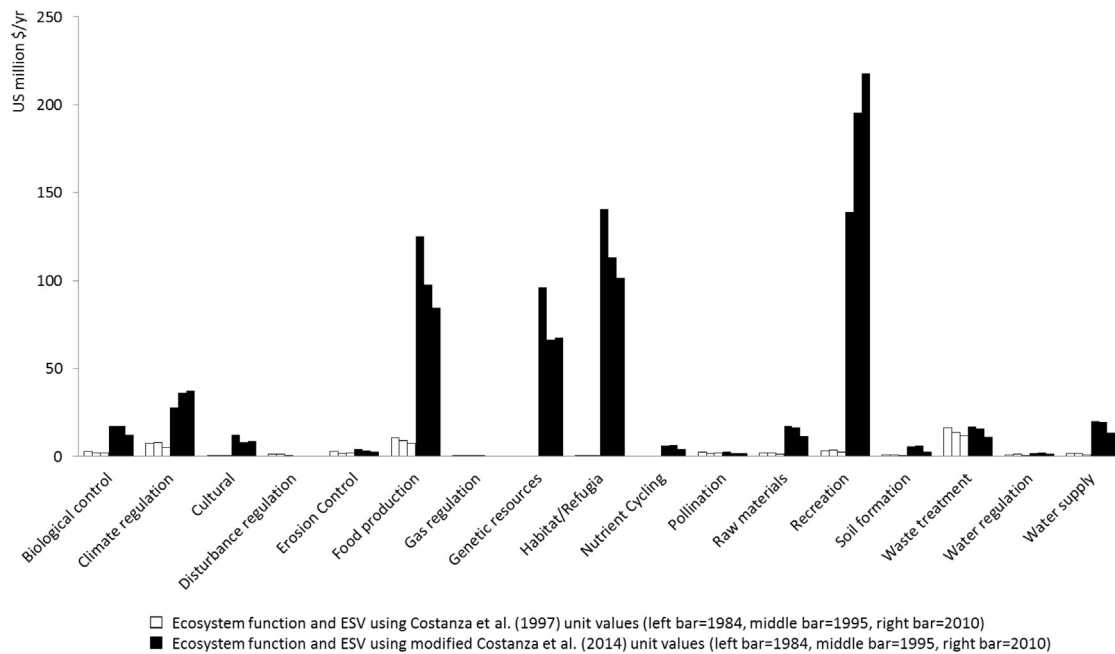


Fig. 7. Changes in ESV by functions between 1984 and 2010 in the three watersheds, Bexar County.

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 (Fig. 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 (Fig. 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 green-space 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 York City” of the 341-hectare green space (Sutton and Anderson, 2016, p. 87). In this way they determined that Central Park provided over \$70 million/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).

## 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 policies, such as NAFTA, on the delivery of ecosystem services in the affected areas.

## Appendix A. Supplementary data

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

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