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ANALYSIS

Change in ecosystem service values in the San Antonio area, Texas☆

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Abstract

San Antonio is one of the fastest growing metropolitan areas in the USA. Urban sprawl may significantly impact ecosystem services and functions but such effects are difficult to quantify and watershed-level estimates are seldom attempted. The objective of the study reported here was to determine whether LANDSAT MSS could be used to quantify changes in land-use and ecosystem services due to urban sprawl in Bexar County, TX, in which San Antonio is centered. The size of six land cover categories in the summer of 1976, 1985, and 1991 were estimated in the 141 671 ha of three watersheds in Bexar County. Coefficients published by Costanza and co-workers in 1997 [Nature 387 (1997) 253] were used to value changes in ecosystem services delivered by each land cover category, and a sensitivity analysis was conducted to determine the effect of manipulating these coefficients on the estimated values. Although we estimated that there was a 65% decrease in the area of rangeland and a 29% increase in the area of urbanized land use between 1976 and 1991, there appeared to be only a 4% net decline in the estimated annual value of ecosystem services in the study area (i.e. \$5.58 ha^{-1} per year, with a 15-year cumulative total value of \$6.24 million for the whole study area). This relatively small decline could be attributed to the neutralizing effect of the estimated 403% increase in the area of the woodlands, which were assigned the highest ecosystem value coefficient. When we assumed that the shift of rangelands to woodlands produced no net change in the value of ecosystem services per hectare, the estimated annual ecosystem service value declined by 15.4% (\$23.22 ha^{-1} per year) between 1976 and 1991. When conducting time-series studies of ecosystem services, it is important to identify parallel changes in land cover types in order to quantify the potentially neutralizing influence of positive land cover changes on the negative effects of urban sprawl on ecosystem services.

Keywords: Urban sprawl; Land-use change; Remote sensing; Rangelands; Woodlands

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1. Introduction

Between 1982 and 1997, the amount of urbanized land in the USA increased by 47% to 30.75 million ha, while the population grew by 17% (Fulton et al., 2001). During the same time period, the conversion of land for development was estimated to have increased from about 500 000 ha per year between 1982 and 1992 to 1.3 million ha per year between 1992 and 1997 (NRCS, 1999). In general, urban sprawl in the south has been exacerbated by a decline in population density in urban centers, though to a lesser extent in Texas where urban population densities have decreased less than in other southern metropolitan areas (Fulton et al., 2001).

Population growth has been especially rapid in the states along the USA–Mexico broader (USCB, 1993). In Texas, a border state, the human population is projected to increase from 19 to 33 million by 2030, with over 70% of the growth expected to occur along the central and southern portions of the I-35 highway corridor and in the Lower Rio Grande Valley (Conner and James, 1996). As a result of this growth, San Antonio has become one of the fastest growing metropolitan areas in the USA, experiencing a 25.2% increase in population from 1990 to 1998, reaching approximately 1 million in 1996, and now being the eighth largest city in the country (SAEDF, 1999). This growth can be largely attributed to a steady growth in employment in the San Antonio area during the latter half of the 1990s when several large manufacturers moved into the area in response to the North American Free Trade Agreement (Rylander, 1997).

This population growth is increasingly impacting rural areas, especially those close to major urban centers in the southern part of Texas, by accelerating land subdivision and reducing the average size of land parcels (Conner and James, 1996). In addition, increase in urban sprawl generally leads to greater traffic volumes, increased pressure on local resources, less open space (Holtzclaw, 1999), and such land-use changes often have a significant negative impact on the affected ecosystems and the goods and services that they provide. Ecosystem services represent the benefits that living organisms derive from ecosystem functions that maintain the Earth's life support system, and include nutrient cycling, carbon sequestration, air and water filtration, and flood amelioration, to name a few (Costanza et al., 1997).

While changes in land use may significantly affect ecosystem processes and services, monitoring and projecting the impacts of such land-use changes are difficult for several reasons. Monitoring changes at the regional scale (where the impact of land-use changes on ecosystems often become noticeable) is difficult because of the large volume of data and interpretation required. In addition, accurately quantifying the impacts of urban sprawl on changes in ecosystem services is difficult because of the lack of information about the contribution of alternate landscapes to these services. Finally, in order to facilitate informed decision-making by comparing the impact of anthropogenic land-use changes with the effect of 'natural' ecosystem changes requires more explicit measures than simple value indices.

The objectives of this study were: (1) to evaluate the efficacy of using LANDSAT multispectral scanner (MSS) data to quantify land-use change in Bexar County, TX, from 1976 to 1991; and (2) to determine if generalized coefficients can be used to evaluate changes in ecosystem services at the watershed scale.

1.1. Using LANDSAT MSS data to measure land-use change

Potentially adverse ecological impacts of urban sprawl have increasingly prompted attempts to map and characterize urban and suburban growth. The US Geological Survey is developing a geo-referenced database of urban land-use change in selected metropolitan regions by merging information from historical maps, census statistics, commerce records, remotely sensed data, and digital land-use data (Acevedo et al., 1997), but this database is incomplete. As historical satellite imagery has become more readily available and less expensive, LANDSAT imagery has become an important tool for acquiring environmental data at spatial, temporal, and spectral resolutions appropriate for assessing broad land-use changes (Verstraete et al., 1996).

While the relatively low 80×80-m spatial resolution of the LANDSAT MSS data limits the detail that can be extracted from these data, ancillary data, such as maps reflecting land-use at the time that a satellite image was taken, can facilitate classification of coarse-resolution images. If coarse-resolution data and classification levels provide sufficient explanatory power for a given purpose, their use may be advantageous because they are less data intensive and provide better broad-scale uniformity than finer resolution data and classification levels (Bourgeron et al., 1999). Moreover, because LANDSAT MSS images were initially produced as early as 1972, MSS data represent the most comprehensive data set for analyzing large-scale land-use changes during the last 25 years.

While in some instances it is desirable to use high-resolution data to conduct detailed land-use analyses, such data cannot be used to quantify long-term land-use changes. Aerial photographs have been used since the 1940s and thus predate LANDSAT MSS, but such images are generally not available for a specified area at regular intervals. In order to study temporal changes, a time series of images for the location in question must be available. Satellite-based imaging (e.g. LANDSAT MSS, LANDSAT TM, etc.) was the first technology to routinely produce images at regular intervals. Digital land-use maps (based on a wide variety of data including LANDSAT images) can also facilitate analyses of land-use patterns. However, because they are composed of data averaged over some time period, such maps do not represent time-specific data and, therefore, cannot be used for time-series analyses of land-use change. We used LANDSAT MSS data to classify land-use during a 15-year period in Bexar County because: (1) they provided readily available and affordable time-specific digital data obtained at regular intervals since the early 1970s; (2) an objective of this study was to quantify long-term changes in land-use; and (3) the resolution of the data was sufficient for classifying land-use patterns at the watershed scale.

1.2. Estimating the value of ecosystem services

Abramovitz (1998) pointed out that ecosystem services have extensive economic value but that they are not credited for the non-market values they provide until they become depleted. While economic tools can be used to identify trade-offs between known ecological values, it remains challenging to link technical measures of ecosystem services to attributes that can be effectively evaluated by untrained individuals (Schaberg et al., 1999). Despite this and other challenges, several attempts have been made to estimate the worth of natural resources. Most notably, Costanza et al. (1997; 1998) presented a model for placing an economic value on different biomes and the services that they provided. Based on their model, they estimated that the global biospheric value of 17 identifiable ecosystem services provided by the 16 dominant global biomes is \$33 trillion per year, most of which is outside the market. However, because of uncertainties, they stated that this should be considered to be a minimum estimate.

While Costanza et al.'s article did focus debate on the importance of ecosystem services that are generally undervalued in standard economic analyses, their cross-sectional estimate based on average, often local, per-unit values, was widely criticized by economists for both theoretical and empirical reasons (Pimm, 1997; Toman, 1998; Masood and Garwin, 1998; Norgaard et al., 1998; Pearce, 1998). For example, because the last hectare of an ecosystem to disappear is likely to be worth much more than the first, simple multiplication of selected average values by all the units in the biosphere underestimated a potentially infinite social value of ecosystem services. Pearce's (1998) greatest concern was that Costanza et al.'s estimated \$33 trillion 'value of everything' is larger than the world GNP which is around \$18 trillion per year. Since 1997, additional studies conducted to quantify the value of ecosystem services have produced lower estimates. For example, Alexander et al. (1998) estimated that ecosystem services are 44–88% of global GNP and concluded that while this estimate is lower than Costanza et al.'s estimate, it nevertheless indicates that accounting for ecosystem service values would greatly alter current GNP estimates. In a regional study using locally derived data, Seidl and Moraes (2000) re-estimated the ecosystem contribution of the Pantanal sub-region Nhecolandia to global production and derived a value of \$15.5 billion per year, approximately 50% of Costanza et al.'s corresponding estimate.

Although Costanza et al.'s estimates of the value of ecosystem services are imperfect, and we lay no claim to their veracity, they do represent the most comprehensive set of first-approximations available for quantifying the change in the value of services provided by a wide array of ecosystems. Since one objective of our study was to determine the effectiveness of using generalized value coefficients to estimate watershed-level changes in ecosystem services, and because the scope of our project did not allow us to obtain area specific value coefficients, we used Costanza et al. (1997) estimates in our study despite their limitations.

2. Study area and estimation approach

San Antonio is centered in Bexar County (29°27′ N, 98°31′ W) near the head of the San Antonio River Basin, which traverses the Edwards Plateau, the Texas Blackland Prairies, and the Western Gulf Coastal Plain eco-regions (Fig. 1). Bexar County was chosen for the study in order to maximize the probability of detecting changes in ecosystem services due to urban

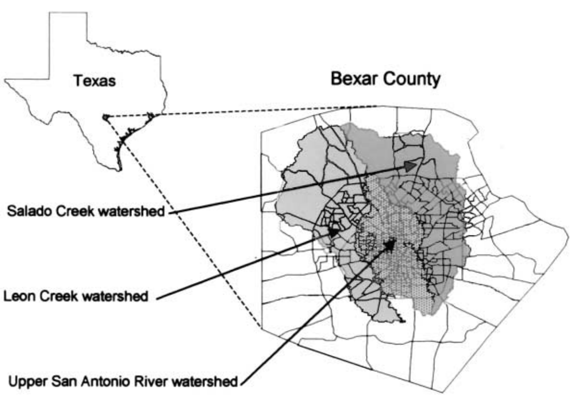

Fig. 1. Location of three watersheds in Bexar County, TX.

sprawl. The three major streams running through the county are Salado Creek, the Upper San Antonio River, and Leon Creek. Since watersheds often represent the minimum ecological management unit and the purpose of our study was to estimate changes in the value of ecosystem services in the wake of land use conversion, the watersheds associated with the three streams were used as the analytical units in Bexar County. The size of the three watersheds comprising the study area was 141 671 ha.

Satellite imagery and remote sensing analytical software were used to determine the area of six land-use classes described below in each of the three watersheds in Bexar County. These estimates were then incorporated into an economic valuation model to quantify changes in the value of ecosystem services in each watershed over time. In addition, a sensitivity analysis was conducted to determine how sensitive the estimates of the value of ecosystem services were to the applied valuation coefficients.

2.1. Land-use classification

Data sets incorporating Bexar County during relatively cloud-free days in August 1976 (the earliest available MSS data set of the study area), August 1985 (the next available cloud-free MSS data set in a year with rainfall similar to 1976), and June 1991 (the most recent available cloud-free MSS data set in a similar rainfall year) were obtained from the US Geological Survey EROS Data Center. The data sets were geo-referenced and Bexar County was extracted as a separate data set using the ERDAS Imagine® software (ERDAS, 1997), which incorporates functions for both image processing and the use of geographic information systems (GIS). Groupings of the spectral properties of the pixels in each of the three data sets of Bexar County were obtained through unsupervised classification using the Iterative Self-Organizing Data Analysis Technique (ISODATA) (Jensen, 1996) that is available in ERDAS Imagine®.

After the unsupervised classifications of the 1976, 1985, and 1991 data sets were completed, the resulting 100 spectral signature classes in each image were grouped into six land cover categories. This was accomplished using the US Geological Survey Land-use/Land Cover Classification System for Use with Remote Sensor Data, Level I (Jensen, 1996) in combination with a 1991 ground-truthed land-use map of Bexar County, developed for zoning purposes by the City of San Antonio Planning Department. The six land cover categories were: rangeland, woodland, bare soil, residential, commercial, and transportation. In Bexar County, the rangeland category corresponds to relatively open grasslands, while *Juniper*, *Quercus* (oak) and *Ulmus* (elm) species dominate the woodlands. During the time period covered by the LANDSAT MSS data used in this study, cultivated areas would have been largely bare and thus bare soils were assumed to reflect mainly post-harvest croplands. Residential (mainly suburban) and commercial (mainly urban) areas were differentiated by assuming that commercial areas have a reflective value more closely resembling paving materials, which occurs extensively in urban areas, while residential areas were assumed to be intermediate to the commercial and bare soil spectral signatures.

Once images of the six land categories were derived for each of the 3 years, vector files delineating the Leon Creek, Salado Creek, and the Upper San Antonio River Watersheds were projected onto these images in order to estimate the area of each of the six land cover types within each watershed. Since no true reference data were available for the 1976–1991 time periods of investigation, uncertainty associated with land-use classification change was not measured (see Congalton and Green, 1999).

2.2. Assignment of ecosystem service values

In order to obtain ecosystem services values for various ground cover types, the six land cover categories used to classify the LANDSAT MSS data sets were compared with the 16 biomes identified in Costanza et al.'s (1997) ecosystem services valuation model. The most representative biome was used as a proxy for each land cover category, including grass/rangelands for rangelands, temperate/boreal forest for woodland, cropland for bare soil, and urban for the commercial, residential, and transportation categories (which were not differentiated by Costanza et al.), as shown in Table 1. The total value of terrestrial ecosystem services in the study area in 1976, 1985, and 1991 was obtained as follows:

Table 1
Costanza et al. (1997) biome equivalents for the six land-use categories, and the corresponding ecosystem values

Land cover categories	Equivalent biome	Ecosystem service coefficient ($ ha^{-1} per year)
Rangeland	Grass/rangelands	232
Woodland	Temperate/boreal forest	302
Bare soil	Cropland	92
Residential	Urban	0
Commercial	Urban	0
Transportation	Urban	0

$$\mathrm{ESV} = \sum (A_k \times \mathrm{VC}_k) \quad (1)$$

where ESV is the estimated ecosystem service value, A_k is the area (ha) and VC_k the value coefficient ($ ha^{-1} per year) for land use category 'k'. The change in ecosystem service values was estimated by calculating the difference between the estimated values for each land cover category in 1976, 1985 and 1991.

The biomes used as proxies for the land cover categories are clearly not perfect matches in every case. Specifically, the juniper/mesquite/elm dominated woodlands in the study area may not be well represented by Costanza et al.'s temperate/boreal forest biome because of the different climatic conditions under which they occur, but in terms of the ecosystem services that they provide, the woodlands of Texas and temperate/boreal forests do have some similarities. For example, Texas woodlands can increase soil nutrient concentrations and contribute to gas regulation through their roles as carbon sinks (e.g. Boutton et al., 1999), and they can provide recreation opportunities. Due to the uncertainties about the representativeness of the proxies used for each land cover category as well as the veracity of Costanza et al.'s value coefficients, sensitivity analyses were conducted to determine the dependence of temporal changes in ecosystem service values on the applied valuation coefficients. The ecosystem value coefficients for rangeland, woodlands, and bare soil categories were each adjusted by 50%. Pervasive woody plant invasion throughout much of Texas, including Bexar County, resulted in a large portion of the area identified as rangeland in 1976 being classified as woodland in 1991. However, because of the uncertainty about the representatives of Costanza et al.'s temperate/boreal forest biome for woodlands, one additional sensitivity analysis was conducted by reducing the ecosystem value coefficient for woodlands (\$302 ha^{-1} per year) by 30% to that of rangelands (\$232 ha^{-1} per year), thereby assuming that woodlands are a variant of rangelands.

In each analysis, the coefficient of sensitivity (CS) was calculated using the standard economic concept of elasticity, i.e. the percentage change in the output for a given percentage change in an input (Mansfield, 1985):

$$CS = \frac{(ESV_j - ESV_i)/ESV_i}{(VC_{jk} - VC_{ik})/VC_{ik}} \qquad (2)$$

where ESV is the estimated ecosystem service value, VC is the value coefficient, 'i' and 'j' represent the initial and adjusted values, respectively, and 'k' represents the land use category.

If the ratio of the percentage change in the estimated total ecosystem value (ESV) and the percentage change in the adjusted valuation coefficient (VC) is greater than unity, then the estimated ecosystem value is elastic with respect to that coefficient, but if the ratio is less than one, then the estimated ecosystem value is considered to be inelastic. The greater the proportional change in the ecosystem service value relative to the proportional change in the valuation coefficient, the more critical is the use of an accurate ecosystem value coefficient.

3. Estimated changes

3.1. Land-use change estimates

It is important to emphasize that, due to a lack of reference data, a limitation of retrospective land-use classification is uncertainty about the accuracy of the estimated size of land-use categories (Congalton and Green, 1999). Therefore, observation of changes in the size of land-use categories must be treated with caution. However, if the magnitude of the estimated changes in land use is substantial, it may still be possible to draw general inferences about the effect of perceived changes in land use patterns on ecosystem services.

In 1976, the rangeland category dominated all three watersheds within the study areas and totaled an estimated 80 497 ha (Fig. 2), but by 1991, the total area of this category was estimated to have decreased by about two-thirds to 27 896 ha. The effective annual rate of decrease appears to have accelerated from about 3% per year during the 1976–1985 period to 12% per year during the 1985–1991 period (Table 2). The woodland category, by contrast, increased substantially in all three watersheds and overall more than quadrupled from 8886 ha in 1976 to 44 654 ha in 1991 (Fig. 2), with an estimated annual growth rate of 10–12% per year (Table 2). The sizeable decrease in open rangelands and the concomitant increase in the size of the woodland category is consistent with the widespread and rapid encroachment of woody plants in the region (Smeins and Merrill, 1988; Smeins et al., 1997). Intensive livestock grazing and reduced fire frequency have resulted in a change in the grass–woody plant interaction on many rangelands in the USA (Scifres, 1980; Archer and Stokes, 2000), which has led to widespread transformation of grasslands to shrublands, and savannas and woodlands (Archer, 1989, 1994; Schlesinger et al., 1990). As a result of these changes and the associated decline in water yields in some areas (Thurow et al., 2000), brush control has become a dominant issue in rangeland management throughout the Edwards Plateau and the Edwards Aquifer Recharge Zone.

The bare soil category, which was assumed to predominantly represent post-harvest cropping areas, was estimated to have increased in all three watersheds from 6353 ha in 1976 to 13 057 ha in 1991 (Fig. 2). However, while the size of this land cover category increased by an estimated 113% between 1976 and 1985, it appeared to have decreased by about 3% during the subsequent 6 years (Table 2). The initial increase may have been

Table 2
Total estimated area (ha) of each land-use category in the San Antonio study area, and changes in land-use from 1976 to 1991

Land-use category	Total area (ha)			1976–1985			1985–1991			1976–1991		
	1976	1985	1991	ha	%	% per year	ha	%	% per year	ha	%	% per year
Rangeland	80 497	59 126	27 896	−21 371	−27	−3.4	−31 230	−53	−12.0	−52 601	−65	−6.8
Woodland	8886	25 336	44 654	16 450	185	12.0	19 319	76	9.9	35 769	403	11.0
Bare soil	6353	13 514	13 047	7161	113	8.7	−467	−3	−0.6	6694	105	4.9
Residential	11 499	10 087	16 655	−1412	−12	−1.4	6568	65	8.7	5156	45	2.5
Commercial	6116	10 457	15 362	4341	71	6.1	4905	47	6.6	9246	151	6.3
Transportation	25 748	23 060	23 857	−2687	−10	−1.2	797	3	0.6	−1891	−7	−0.5

associated with clearing of rangelands for crop production, as well as suburban development, but it may also have been partly due to misclassification because of seasonal variations in vegetative ground cover at the time that the three satellite data sets were captured. For example, more rapid spring growth due to above average spring rains in 1976 could have resulted in an underestimate of cropland due to misallocation of crop land (bare soil) to rangeland in the 1976 image (NCDC, 1994).

Although the estimated size of the residential land cover category increased in all three watersheds from 11 449 ha in 1976 to 16 655 ha in 1991 (Fig. 2), the rate of growth averaged only 3% per year during this 15-year period, and most of the growth appears to have occurred during the 1985–1991 period (9% per year) (Table 2). The estimated size of this category actually decreased slightly in the Salado Creek and Leon Creek watershed between 1976 and 1985. The apparently small change in the residential land cover categories may be an underestimate of the actual growth for two reasons. First, because asphalt shingles are widely used for roofing and residential area road networks are dense, some residential areas could have been misallocated to the urban and transportation categories. Secondly, the potentially increasing masking effect of turf and tree canopy cover in residential areas could have resulted in some misallocation of maturing residential areas to the rangeland or woodland land cover categories in 1985 and 1991.

According to our analysis, there was a 151% increase in the area of the urban land use category from 1976 to 1991 (Table 2). By contrast, the transportation category appeared to decrease marginally during the 15-year period of our study, mainly from 1976 to 1985, but reduction in transportation is inconsistent with expanding urban sprawl. This anomaly may be explained by the fact that there are inevitable inaccuracies in delineations of urban and transportation land uses when using remotely sensed spectral reflectance data because asphalt in parking lots, roofs, and roads produces identical reflectance, and concrete has the same spectral signature regardless of its location. Thus it is better to consider changes in the urban and transportation categories together. In combination, these two land use categories increased 23% from an estimated 31 864 ha in 1976 to 39 219 ha in 1991, and the average annual rate of growth appeared to have accelerated from 0.6% per year during the 1976–1985 period to 3.5% per year during the 1985–1991 period.

3.2. Estimation of changes in ecosystem services

Using the estimated change in the size of each land cover category, together with the ecosystem service value coefficients reported by Costanza et al. (1997), we found that land-use changes in the

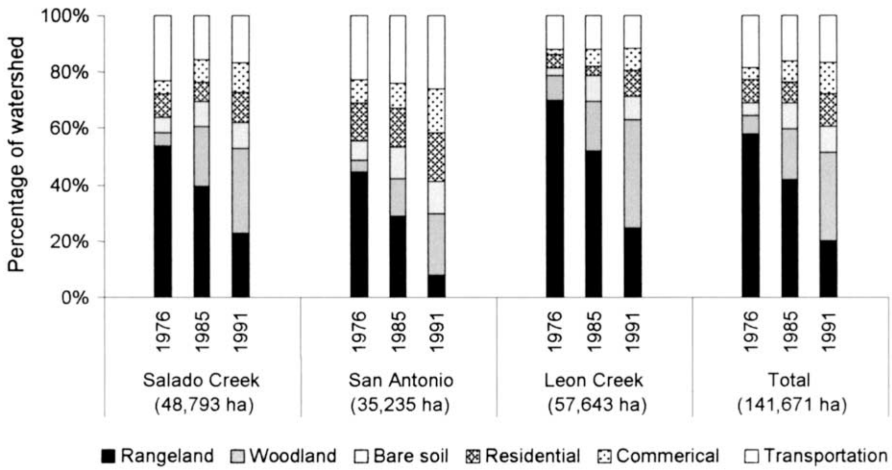

Fig. 2. Size and percentage contribution of each of six land cover categories in each watershed and the total study area.

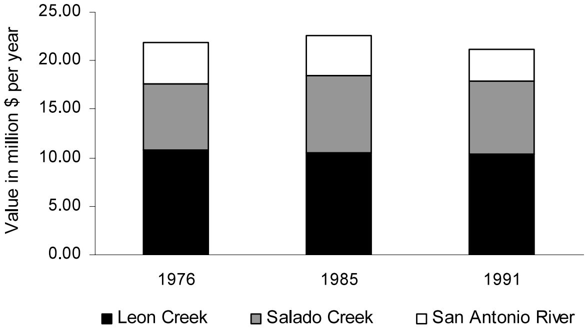

Fig. 3. Ecosystem service values estimated for each watershed using Costanza and co-author's 1997 coefficients.

141 671 ha of our study area resulted in a \$0.78 million per year (i.e. 4% or \$5.58 ha^{-1} per year) net decline in ecosystem services from 1976 to 1991 (Fig. 3, Table 3). Assuming a linear decline in ecosystem services from 1976 to 1991, this decline represents a cumulative loss of \$6.24 million in ecosystem services over the 15-year period of the study. However, this cumulative estimate must be regarded with caution because ecosystem services may fluctuate somewhat; our analysis detected a slightly higher ecosystem service value in 1985 than in 1976. In order to accurately estimate the cumulative loss of ecosystem services, it would be necessary to measure ecosystems' services at regular intervals, preferably annually.

The estimated change in the annual value of ecosystem services was small in all three watersheds between 1976 and 1991. However, while the change was negative in both the Leon Creek and San Antonio River watersheds (\$0.41 and 0.96 million per year, respectively), it was positive in the Salado Creek watershed (\$0.59 million per year). The primary reason for these small changes is that the effect of the estimated decline in rangelands, valued at \$232 ha^{-1} per year, was largely offset by the increase in woodlands that were valued at \$302 ha^{-1} per year, 30% higher than rangelands. Thus, while the value of ecosystem services provided by the rangeland category was estimated to have declined by \$12.0 million (65%) between 1976 and 1991, the value of the woodland land cover category was estimated to have increased by \$10.8 million (403%) during the same period (Table 3). The estimates also indicated a small increase (\$0.62 million per year) in the contribution of bare soil to the total ecosystem service value during this same time period.

Our estimated 15-year conversion of 9511 ha of rangeland, woodland and bare soils to urban and suburban land-uses (residential, commercial and transportation) and the associated loss in ecosystem services appeared to have a small net effect on the annual value of ecosystem services provided within the study area. This is a surprising result given that the residential, commercial and transportation land cover categories were assigned no ecosystem service value while the ecosystem services provided by rangelands, woodlands, and bare soils were assigned values of \$232, 302, and 92 ha^{-1} per year, respectively. One explanation for the apparent small net effect of land use conversion on the value of ecosystem services in the study area is that the loss of ecosystem services on land being developed was offset by the apparent conversion of ecologically 'less' valuable bare soil (cropland) and rangelands to ecologically 'more' valuable woodlands. This ecological conversion is consistent with the increase in distribution and density of woody plants throughout much of central Texas (Smeins et al., 1997).

3.3. Ecosystem services sensitivity analyses

The effects of using alternative coefficients to estimate total ecosystem service values in the study area in 1976 and 1991 are shown in Table 4. The CS of these analyses were less than unity in all cases. The estimated value of the ecosystem service value for the study area increased from a low of 0.03–0.06% for 1% increase in the value of the bare soil coefficient, to a high of 0.31–0.85% for a 1% increase in the value of the rangeland coefficient. This indicates that the total ecosystem values estimated for the study area are relatively inelastic with respect to the ecosystem service coefficients. While this implies that our estimates were robust, highly under or over valued coefficients can substantially affect the veracity of estimated changes in ecosystem service values over time even when the CS are less than unity. For this reason, we also report on the effect of large variations in coefficient values on the estimated value of land-use related changes in ecosystems services.

Table 3
Total ecosystem service values (ESV in US$ × 10^6 per year) estimated for each land cover category in the study area using Costanza et al. coefficients, and the overall change and rate of change between 1976 and 1991

Land-use category	ESV (US$ × 10^6 per year)			Difference in ESV between the first and last year of each time period and the annual change rate that this difference represents								
	1976	1985	1991	1976–1985			1985–1991			1976–1991		
				$ × 10^6	%	% per year	$ × 10^6	%	% per year	$ × 10^6	%	% per year
Rangeland	18.68	13.72	6.47	−4.96	−27	−3	−7.25	−53	−12	−12.20	−65	−7
Woodland	2.68	7.65	13.49	4.97	185	12	5.83	76	10	10.80	403	11
Bare soil	0.58	1.24	1.20	0.66	113	9	−0.04	−3	−1	0.62	105	5
Urban categories[a]	0.00	0.00	0.00	0.00	–	–	0.00	–	–	–	–	–
Total	21.94	22.61	21.16	0.67	3	0	−1.45	−6	−1	−0.78	−4	0

[a] Includes residential, commercial and transportation land cover categories.

When the rangelands coefficient was increased by 50%, the ecosystem services in the study area decreased in value by 22.0% (− \$6.89 million per year) between 1976 and 1991, which was a substantially greater decrease than the initial 4% decline. A 50% decrease in the rangeland coefficient resulted in a 43.6% (\$5.62 million per year) gain in ecosystem services during the same time period. Increasing or decreasing the rangeland coefficient by 50% affected the estimated 1976 ecosystem service value more (± 42.6%) than the 1991 value (± 15.3%). Conversely, when the coefficient for woodlands was increased by 50% the value of ecosystem services grew by 19.8% (\$4.62 million per year) between 1976 and 1991, but a 50% decrease in the woodland coefficient led to a 28.10% (− \$5.88 million per year) decrease in value. In contrast to rangelands, increasing the woodland coefficient by 50% affected the estimated 1976 ecosystem service value less (± 6.1%) than the 1991 value (± 31.9%). These differences between land cover types can be accounted for by the decrease in size of the ecologically 'less valuable' rangelands between 1976 and 1991 and a concomitant increase in size of the 'more valuable' woodlands.

Due to the uncertainty about the relative ecological value of rangelands and woodlands, perhaps most relevant was the result of the sensitivity analysis in which the woodland coefficient was equated to that of rangelands (\$232 ha^{-1} per year), the underlying assumption being that the transformation of rangelands to woodlands results in no overall change in ecosystem services. In this scenario, the annual value of ecosystem services declined by 15.4% (\$3.29 million per year in the 141 671 ha of our study area, i.e. \$23.22 ha^{-1} per year) between 1976 and 1991. This is a substantially greater loss than the originally estimated 4% (\$5.58 ha^{-1} per year) decline in annual ecosystem service values. Assuming a straight-line decline, this would equate to a cumulative loss of \$26.32 million over the 15-year period of our study compared with the estimated cumulative loss of \$6.24 million using the original coefficient values.

Increasing the value coefficient of bare soils by 50% resulted in a reduced loss of annual ecosystem services from 4 to 2.1% between 1976 and 1991, while decreasing the coefficient by 50% resulted in an increased loss of 5%. These comparatively small changes were mainly because of the small size of this land cover category.

4. Discussion

Remote sensing from satellites may be the only economically feasible way to regularly gather information with high spatial, spectral, and temporal

Table 4
Estimated total ecosystem service values (ESV in US$ $\times 10^6$) in the San Antonio study area, after adjusting ecosystem service valuation coefficients (VC), the magnitude of changes in the ESV following the adjustments, and the coefficient of sensitivity (CS) associated with these adjustments

Change in valuation coefficient (VC)	ESV[a]		1976–1991 change		Effect of changing CV from original value[a]			
	1976	1991	$ \times 10^6$	%	1976		1991	
					%	CS	%	CS
Rangeland VC + 50%	31.28	24.39	−6.89	−22.0	42.6	0.85	15.3	0.31
Rangeland VC − 50%	12.61	17.92	5.62	43.6	−42.6		−15.3	
Woodland VC + 50%	23.28	27.90	4.62	19.8	6.1	0.12	31.9	0.64
Woodland VC − 50%	20.60	14.41	−5.88	−28.1	−6.1		−31.9	
Woodland VC = Rangeland VC	21.32	18.03	−3.29	−15.4	−2.8		−14.8	
Bare soil VC + 50%	22.24	21.76	−0.48	−2.1	1.3	0.03	2.8	0.06
Bare soil VC − 50%	21.65	20.56	1.09	−5.0	−1.3		−2.8	

[a] Total ecosystem service values before adjusting the ecosystem service coefficients were: \$21.94 million and \$21.16 million in 1976 and 1991, respectively.

resolution over large areas (Verstraete et al., 1996). This advantage will increase as the cost of obtaining such data declines and computational power to cope with larger data sets from higher resolution sensors increases. However, one limitation for conducting time series analyses of land-use changes using remotely sensed data is that satellite data from high-resolution detectors have a relatively short history. Even LANDSAT data cannot be used for analyzing land-use changes prior to 1972. Due to the lack of recorded historical land cover data, one further limitation of using remotely sensed data to determine changes in land-use patterns is the difficulty of measuring uncertainty about land-use classification. Congalton and Green (1999) state “To date, no standard accuracy assessment technique for change detection has been developed”. They propose the use of an error matrix, but this assumes that some reference data exist. Without such reference data, the investigator is left with two choices: do no analysis, or conduct the analysis with the full knowledge that there is uncertainty associated with historical land use classification. Even with uncertainty, simple classification schemes, such as those used in this study, are reasonable because the distinction between gross land covers is relatively large (Congalton and Green, 1999). Thus, despite the uncertainty of classification, the relatively course spectral resolution of early sensor technology, such as LANDSAT MSS, can be useful for estimating broad scale land-use changes within watersheds.

One challenge for identifying land-use changes was that different land uses might produce similar spectral signatures. For example, it was impossible to determine whether changes in the extent of bare ground were due to changes in the area of cropland, denuded rangeland resulting from drought or overgrazing, or new construction. In Bexar County, most crops are harvested by August, the month in which two of the three images used in the study were taken. This resulted in a lack of distinction between post-harvest cropland and other bare soil surfaces. In addition, areas designated as rangelands in 1976 appear to have converted to woodland in the later images. Such increases in woody plant distributions have been associated with long-term overgrazing in many areas in Texas since the 1880s, as well as fire suppression, which is exacerbated in areas with rapid development (Smeins and Merrill, 1988). Nevertheless, there was some uncertainty about the true extent of the rangeland to woodland conversion. As satellite data with greater spectral and spatial resolution becomes more readily available such land-use ambiguities are likely to decrease.

While Costanza et al.’s ecosystem service values that we used in our analysis have been challenged on theoretical and empirical grounds, they represent the only set of valuation coefficients for a wide array of biomes, each of which encompass several related ecosystems. Also important is the realization that absolutely accurate coefficients are often less critical for time series than cross-sectional analyses because coefficients tend to affect estimates of directional change less than estimates of the magnitude of ecosystem values at specific points in time. We were primarily interested in changes in ecosystem services over time, and the scope of our project precluded us from deriving area-specific ecosystem value coefficients.

Based on the estimated size of six land cover categories and Costanza et al.’s ecosystem services values for related biomes, we determined that the total annual ecosystem service values in Bexar County declined from \$21.94 to 21.16 million per year from 1976 to 1991. Thus, while there appeared to be a 65% decrease in the size of rangelands and a 29% increase in the area of the urban land-use categories, we estimated there to have been only a 4% (\$5.58 ha^{-1} per year) loss in the annual value of ecosystem services. Assuming a straight-line decrease, this represents a \$6.24 million 15-year cumulative loss. This relatively small decline is largely attributable to the fact that the estimated \$12.2 million per year loss in ecosystem services delivered by rangelands was largely offset by the estimated \$10.8 million per year increase in ecosystem services from woodlands. When we assumed that the shift of rangelands to woodlands produced no net change in the value of ecosystem services per hectare, the loss in the estimate annual ecosystem service value between 1976 and 1991 grew to 15.4% (\$23.22 ha^{-1} per year), representing a \$26.32 million 15-year cumulative loss if the decline is linear.

While our study showed that urban sprawl in Bexar County resulted in a decline in the value of ecosystem services delivered by the affected land, it also showed that changes in the value of ecosystem services over time depend on the interaction of changes in various land cover types. Our study suggests that urban spread may not necessarily lead to a large net decline in ecosystem services if there is a concomitant increase in size of other land cover types that provide a greater level of ecosystem services. This is not to say that urban sprawl is beneficial for the delivery of ecological services, but that the negative impacts of the spread of urban and suburban land uses can be potentially offset by other mitigating changes in land cover. This is important because it is unlikely that with increasing human population pressure, the conversion of land to urban and suburban land use will cease or even dissipate in the near future.

Our study showed that LANDSAT data can be used to obtain coarse estimates of changes in ecosystem values at the watershed level. However, in order for this type of analysis to become valuable for policy formulation affecting land-use, it is imperative to obtain a wider array of value coefficients for ecosystem services that more accurately reflect local conditions. This is no easy or costless exercise. One approach could be to identify benchmark ecosystem service values for dominant ecosystem types within a region and then to evaluate the ecosystem services provided at specific locations relative to the representative benchmark. Since ecosystem services are not traded and, therefore, have no market-based price, indirect valuation techniques will be required to obtain these estimates. Such an approach would be analogous to the evaluation procedure developed by Dyksterhuis (1949) for determining the ecological condition of rangelands, and it would allow systematic estimation of changes in the ecological services delivered at specific locations over time. Such information could be extrapolated to larger watershed and regional scales though the use of remotely sensed data and GIS tools to classify land into representative ecosystems for which benchmark values have been established.

Tools for this sort of analysis are becoming increasingly available in common GIS systems. While we limited the use of GIS tools to quantifying land-use change, such tools can also be used to conduct a variety of additional analysis because the data are spatially explicit. Ecological economists could incorporate variables such as patch size, edge effect, contiguity, wildlife corridors, biodiversity potential, to name a few, in their analyses to better evaluate changes in land-use from an ecosystem service perspective. For example, in future watershed level studies, quantifying the location and sequence of different land cover types relative to watercourses could be used to facilitate ecosystem value estimates. Thus, increasingly sophisticated GIS tools could be used in conjunction with ecosystem benchmarking to efficiently and accurately estimate to impact of land use conversion on the services provided by the affected ecosystems.

From a policy standpoint it is also critical that the factors facilitating urban sprawl be clearly understood and that rural landowners be provided with incentives that minimize the negative impacts from such changing land use. For example, weak local government, developer pressure, high estate taxes for rural landowners, and greater urban affluence are major causes of rapid rural land fragmentation and urban sprawl west of San Antonio and Austin. Unless landowners have positive incentives to sustain rural land-uses, urban sprawl will increasingly impact ecosystem services in these areas even when we are able to accurately quantify the loss of ecosystem services associated with such changes in land-use.

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