Ecological Economics 82 (2012) 23–32

Contents lists available at SciVerse ScienceDirect

Ecological Economics

journal homepage: www.elsevier.com/locate/ecolecon

Analysis

Land use change and its effects on the value of ecosystem services along the coast of the Gulf of Mexico

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ARTICLE INFO

Article history:
Received 15 January 2010
Received in revised form 17 July 2012
Accepted 23 July 2012
Available online 30 August 2012

Keywords:
Ecosystem services
Land use change
Ecosystem service value
Coast
Dunes
Mexico

ABSTRACT

In the central region of the Gulf of Mexico, urban growth occurs mainly to support tourism and results in loss of natural ecosystems and ecosystem services. Our objectives were to analyze land use changes and calculate the value of these changes in terms of lost ecosystem services. We selected three study sites with contrasting infrastructure for tourism: Boca del Río, Chachalacas and Costa Esmeralda. From 1995–2006, we found that urban sprawl was predominant, and occurred over mangroves, grasslands, croplands and the beach. Using the benefit transfer method, we calculated a net loss ($US 2006/ha/year) of $\$1.4 \times 10^3$ in Boca del Río, $\$7 \times 10^5$ in Chachalacas and $\$1 \times 10^5$ in Costa Esmeralda. Because the value of urban land is higher (from 45,000 USD/ha (2006) in Costa Esmeralda to 6 million in Boca del Río) than the total estimated Ecosystem Services Value (106,000 USD/ha, including all ecosystems and ecosystem services), land use change may seem economically profitable. However, after losing ecosystem services such as coastal protection or scenic value and recreation, the apparent gains from urban development are lost. Land use and policy making should consider ecosystem service losses so that ecosystems are preserved and society benefited.

1. Introduction

Natural ecosystems provide a variety of direct and indirect services and intangible benefits to humans and other living organisms (Costanza et al., 1997; Daily, 1997). Because of their relevance to society, these ecosystem services, as well as their economic value have become a focus of interest for scientists, policy makers and stakeholders over the last decade (Troy and Wilson, 2006). The provision of ecosystem services is directly related to the functionality of natural ecosystems upon which ecological processes and ecosystem structures depend (de Groot et al., 2002). Nevertheless, because of population growth, economic pressure, and urban sprawl, natural ecosystems are continuously being altered, destroyed or transformed, especially during the last decades (Vitousek et al., 1997). Globally, land use changes from natural ecosystems to croplands, grazing lands (grasslands), and urban areas have increased over time, resulting in reduced or modified biodiversity, altered functional processes and diminished provision of ecosystem goods and services to society (Balvanera et al., 2006; de Groot et al., 2002; Díaz, 2006; Li et al., 2007; McIntyre and Lavorel, 2007; Metzger et al., 2006). For instance, it has been observed that land use change into urban sites and agriculture is detrimental for several ecosystem services such as a) nutrient cycling, climate regulation, erosion control and genetic resources (Li et al., 2007; Peng et al., 2006; Wang et al., 2006; Zhao et al., 2004), b) recreation opportunities (Kreuter et al., 2001), c) climate regulation and erosion control (Portela and Rademacher, 2001) and d) soil fertility, water availability and increases the risk of forest fires (Schroter et al., 2005).

Although the impact of human activities is pervasive to all natural ecosystems in the world, the coasts have been particularly affected. For millennia humans have been attracted to the coastal areas and at present nearly 40% of the global population live within 100 km of the shoreline (Martínez et al., 2007). The impact of human activities near or at the coast is, therefore, intense. Mexico is no exception to this global trend: here, tourism has developed rapidly along the coast, especially during the last decades. In particular, the coast of the state of Veracruz (745 km long) is socially very important, with nearly 20% of its cities and 27% of its population (1,898,013) located less than 20 km away from the shoreline. Land use change has occurred rapidly in the state, which has lost more than 36% of its original forest cover since 1980 and more than 40% suffer from serious soil erosion (SEFIPLAN, 2005). Public and private investments in agriculture activities increased dramatically between 1940 and 1970; but have since changed to promote cattle ranching (CONABIO, 2006). State programs in Veracruz support and promote livestock activities in natural areas, thus exacerbating deforestation and pollution, and consequently, a degraded landscape

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0921-8009/$ – see front matter
http://dx.doi.org/10.1016/j.ecolecon.2012.07.018

(SEFIPLAN, 2005). Recently, tourism activities have increased along the coast, resulting in additional degradation and loss of natural ecosystems.

These development policies do not consider the environmental impacts of land use change in terms of ecosystem services. Indeed, the ecological relevance and the social pressure on the coastal ecosystems of Veracruz make it evident that there is an urgent need to: a) assess the current status of the natural ecosystems located along the coast of the state of Veracruz; b) to evaluate the ecosystem services provided by natural ecosystems located at the coast and c) assess how they have been altered with land use change. Because tourism and urban sprawl along the coast of Veracruz are increasing rapidly, in this study we aimed at assessing land use change in three coastal locations with contrasting tourism activities and later evaluated the impact of these changes in terms of ecosystem services. We used ecosystem service values to estimate the impact of regional land-use change associated with tourism development that has occurred over the last decade (1995–2006).

2. Methodological and Ideological Options

2.1. Study Sites

The state of Veracruz is located midway along the Gulf of Mexico between Tamaulipas and Tabasco (Fig. 1) and occupies 3.7% of the total surface of Mexico. The state's high weather and topographic diversity support a diverse array of atural ecosystems, ranging from tropical rain forests to alpine needle leaf forests (SEFIPLAN, 2005). Veracruz is the third most populated state in the country with nearly seven million inhabitants. The shoreline of Veracruz (745 km) represents 6.42% of the national coastline (11,593 km) (INEGI, 2007). In Mexico, the port of Veracruz is one of the most important ports for commerce between America and Europe. The state of Veracruz has a rather high agricultural productivity because of the abundant rivers interconnecting within the watersheds, which flow through the municipalities downhill and into the coastal zone (SEFIPLAN, 2005). The coastal ecosystems are highly diverse, and include mangroves, coral reefs, sea grasses, coastal dunes, tropical rain forests, among others. All along the coast, disorganized urban growth has generated deterioration, pollution and overexploitation of these diverse natural resources. As human conflicts compete for space and resources, coastal dunes and wetlands are transformed and degraded by human settlements (SEMARNAT, 2006).

The coast of Veracruz is an important destination for national and international tourism. We selected three study sites with contrasting tourism activities and infrastructure (Fig. 1), all located in the central region of the Gulf of Mexico in the state of Veracruz: Boca del Río, Chachalacas and Costa Esmeralda. The three study sites cover similar areas, although Costa Esmeralda was marginally larger (Table 1). They all share a hot-humid weather with mean annual precipitation ranging from 1018 mm/year in Chachalacas to 1694 in Boca del Río. Boca del Río is located to the south and adjacent to the Port of Veracruz. It is mostly urbanized and includes large hotels and urban infrastructure. Here, commercial activities are predominant. This is the most densely populated location from our three study sites. Chachalacas is located in the municipality of Úrsulo Galván, it is the least urbanized area and offers suburban infrastructure in a more natural setting. Costa Esmeralda encompasses 3 municipalities: Tecolutla, San Rafael and Nautla. It is a farming area with limited urban infrastructure. Fishing, cattle ranching and croplands are more important in Chachalacas and Costa Esmeralda. All three study sites have popular beaches that are frequently visited by tourists. Natural ecosystems and habitats in all study sites have been fragmented owing to land use change and the increasing impacts of human activities.

2.2. Land Use Changes

We used high resolution aerial images of the same areas from different time periods (1995 and 2006), to assess the changes that have occurred in each site over a decade. Land use polygons were created based on 1:75,000 orthophotos from 1995 that were obtained from the National Institute of Statistics, Geography and Informatics (INEGI, 2007). Polygons were also created over high resolution aerial photography of 0.80 m of pixel from 2006. ArcView 3.2 was used to digitize by hand two vector maps for each study area. In each location, the study area comprised the alongshore political limits of the municipalities, and a 2.5 km wide band inland from the shoreline. Land use was classified for two time periods (1995 and 2006), and was verified in the field. Finally, we calculated transition matrices of land use change, by summarizing the cover of each land use type from 1995 and calculating how each one changed in the following decade (2006).

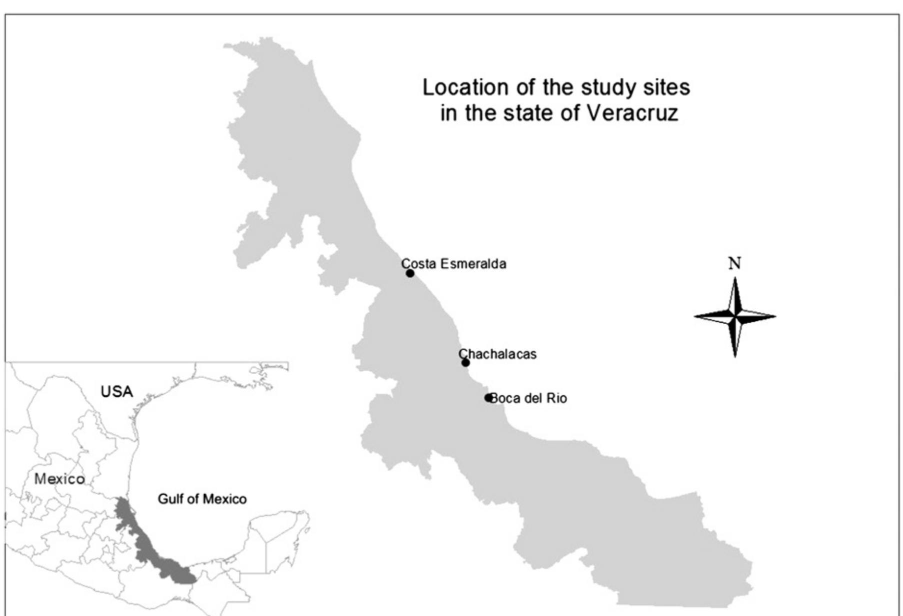

Fig. 1. Location of the study areas in the state of Veracruz.

Table 1
Location, weather and social features of the three study sites.

	Boca del Río	Chachalacas	Costa Esmeralda
Municipalities	Boca del Rio	Ursulo Galvan	Tecolutla, San Rafael, Nautla
Latitude	19° 07′	19° 24′	20° 11′
Longitude	96° 06′	96° 18′	96° 51′
Surface (ha)	2290	2630	3060
Climate	hot-sub humid	hot-sub humid	hot-humid
Mean yearly temperature (°C)	25	25	25.5
Mean total yearly precipitation (mm)	1694	1018	1494
Main activities	Commerce, tourism activities	Agriculture, livestock, sugar production, tourism activities, fishing, commerce	Agriculture, livestock, fishing, commerce, tourism activities
Rivers	Jamapa	Actopan	Bobos, Tecolutla, Nautla, Misantla
Population	141,906	26,909	25,680
Beaches	Mocambo, Santa Ana y Mandinga	Chachalacas	La Guadalupe, Ricardo flores Magon, La vigueta, Playa oriente, Monte gordo, Casitas, Maracaibo

2.3. Estimation of Economic Ecosystem Services Values (ESV)

For each natural ecosystem occurring at our three study sites we calculated the mean economic value of all recognized ecosystem services provided by local natural ecosystems, in order to estimate the potential impact of land use change in terms of lost or diminished ecosystem services (SEFIPLAN, 2005; Troy and Wilson, 2006). Several methods are used to estimate the monetary value for each specific service provided by natural ecosystems. One of these valuation techniques is the benefit transfer method, which transfers the monetary value determined from one place and time (policy site) to make inferences about the economic value of environmental goods and services at another place and time (study site) (Rosenberger and Stanley, 2006; Wilson and Hoehn, 2006). The economic information collected at the policy site is derived from various methodologies and tools. Once the value has been transferred to the study site, an estimated ecosystem service value can be calculated. Thus, transfer valuation technique enables the valuation of ecosystem services in different locations where the role that ecosystems have on economic and social systems is recognized, but local valuations are lacking.

Several environmental valuation databases provide the information that is necessary to complete a benefit transfer valuation. Three valuation databases stand out among the many databases that are currently available: "Environmental Valuation Reference Inventory" (EVRI), "Envalue" and "Ecosystem Services Database" (ESD) (McComb et al., 2006). We used these databases to perform our transfer valuation. We also used applicable gray literature, such as Mexican government statistics and one PhD. thesis (Shuang, 2007). Based on these databases and published literature (e.g. Costanza et al., 1997; de Groot et al., 2002), we determined specific ecosystem services that are provided by the ecosystems that are found at the study site.

The applicability of the benefit transfer method relies entirely upon the assumption that the ecosystem service being provided in the location where the original study was conducted and the area to which the estimated value is being applied, are sufficiently similar. Because of this, we were careful to choose only ecosystem service values calculated for Mexico and other Latin American countries which are similar to Mexico in terms of their ecological, cultural, social and economic characteristics. Overall, we found 27 valuation studies from Latin America, of which 17 (63%) were performed in Mexico (see Appendix 1). However, we mostly used monetary values that were estimated in Mexico for 9 ecosystem services provided by the ecosystems found in our study sites and that have been studied previously. Two exceptions to this were medicine in agricultural lands and ecosystem services provided by grasslands. Here we used the economic valuations from Belize and Brazil respectively, to fill out the missing information from Mexico. GDP and Gini index as well as the studied ecosystems of both countries are similar to those from Mexico. The remaining valuation studies from Latin America were used only for comparison between Mexico and Latin America.

In addition to the above, we used direct market price values to calculate the recreation and protection values of coastal dunes and beaches for which we had not found information for either Mexico or Latin America. For recreation we used the prices of 4-wheel drive (4WD) and sand-board rentals in Mexican and Chilean sand dunes, respectively. The calculations were performed as follows:

4WD rentals.- In Chachalacas, tourists frequently rent 4WD to drive around the beach and on the coastal dunes (the environmental impact of these activities will not be addressed in this study). In the summer of 2011, we gathered information from all six businesses that offer this service to tourists. We calculated the mean cost of 4WD rentals (29.09 USD/h) that ranged from 17.03 to 46.83. Because most tourists visit Chachalacas during the weekends (except for specific holidays such as Christmas and Easter vacation), we estimated that, in optimal circumstances, all 52 vehicles that are available in Chachalacas are rented for 8 h a day, every weekend of the year (104 days). Finally, upon asking where tourists drive these rented vehicles, we estimated from aerial photos that 4WD users drive over approximately 100 ha from the nearly 2000 ha of coastal dunes found in this site. With this information we were able to calculate yearly per hectare value of coastal dunes in Chachalacas, in terms of recreation (see Appendix 1).

Sand-board rentals.- In the coastal dunes of Concón, Chile, sand surfing is a favorite activity of local inhabitants of Valparaiso as well as tourists. Here, the hourly cost of sand-board rental in 2010 was 2.08 USD. Similar to the 4WD rentals, we estimated that all 40 sand-boards from the two local businesses were rented 8 h a day, every weekend of the year. Sand surfing takes place in 2 out of the 19 ha of this nature sanctuary. We only used this recreation cost for comparison with Mexican values.

Protection value.- In Costa Esmeralda we found a landowner who had built an artificial permanent "foredune" using concrete instead of sand. This "dune" was located in front of the property, between the house and the beach, for protection against the impact of storm surges and hurricanes (the effectiveness of this strategy will not be discussed in this study). In 2006, this artificial dune cost 5103 USD and covered 37.5 m^2. With this, and considering that this structure will probably need to be replaced every 20 years (from our field observations), and trying to be conservative in our estimations, we calculated the protection value per hectare per year (see Appendix 1).

In order to standardize all these economic valuations (Ecosystem Service Value - ESV), the economic values estimated for ecosystem services from different ecosystems were adjusted to US$ currency using the Consumer Price Index (CPI) and the Purchasing Power Parity (PPP) for 2006, obtained from US government statistics (US Department of Labor). We thus adjusted the original values estimated in the 19 studies we used (Appendix 1), to US dollars (2006), using the following formula (Envalue, 2007).

$$\mathrm{ESV} = \frac{(\mathrm{value}/CPI) \times 100}{PPP} \times \mathrm{USA}\ \ PPP$$

Where:

Value is the value in the original year in the original currency.
CPI is an index of inflation of the source data, with a base year in 2006.
PPP is the Purchasing Power Parity between the original currency and US$ in 2006.

Finally, we calculated the total value of ecosystem services provided by each ecosystem by adding the value of each individual service. Area data from our three study sites was then multiplied by these values per hectare to obtain potential total values for each ecosystem in both 1995 and 2006. To estimate the changes in ESV owing to land use change, values from 2006 were subtracted from those in 1995. These calculations were performed by estimating yearly change rate with the next formula (Márquez, 2008):

$$\delta n = (S_2/S_1)^{1/n} - 1$$

Where:

δn	Change rate
S1	Surface of each ecosystem in 1995
S_2	Surface on each ecosystem in 2006
n	Number of years in the study period

Because land area data was available for multiple years, we were able to compare past and current changes on ecosystem surface and learn how important that gain or loss of area has been in terms of ecosystem services at the three study sites (Rosenberger and Stanley, 2006; Troy and Wilson, 2006).

3. Results

3.1. Land Use-Cover Change

Based on plant cover types we established a classification of eight general land use-cover classes that were found at the three study areas with the GIS analysis (beach, dune, mangrove, scrubland, tropical forest, cropland, grassland for cattle, urban). In Boca del Río we found four land use cover types: beach, mangrove, grassland, and urban (Fig. 2). The beach became urbanized and lost 7 ha (19%). Mangroves and grassland had the largest reduction (41 ha and 77 ha lost, respectively), which represented 16% and 35% of their original 1995 surface. Urban areas occupied the largest surface in both 1995 and 2006 (Table 2). Overall, urban areas increased 125 ha in 2006 that represents a 7% expansion replacing grasslands, mangroves and beach previously present in 1995 (Table 2). The transition matrix reveals that all land use changes that took place resulted in urban sprawl, showing the high urbanization trend in Boca del Río even though it was already mostly urbanized in 1995 (Table 2).

In contrast with Boca del Río, the landscape at Chachalacas, showed seven land use types: beach, dune, scrubland, tropical forest, cropland, grassland, and urban (Fig. 3). The landscape at Chachalacas showed the largest changes because, originally, it had the largest surface covered by natural ecosystems in comparison with the other sites. The beach, one of the most widely used ecosystems, only represents less than 1% of the total area. This ecosystem expanded during the last decade owing to a prograding shoreline during the observed period, due to sediments that are accumulating towards the coral reef located offshore. Mobile coastal dunes were the second largest polygon in 1995, and changed mostly into scrubland (owing to natural vegetation succession) (Table 3). Scrublands also expanded over grasslands, and so did a small fragment (less than 1% in the landscape) of Tropical Forest which increased from 1995 to 2006. The transition matrix reveals that a large proportion of natural ecosystems changed into both grasslands for cattle ranching and scrubland (due to natural succession). We found that grasslands for cattle occupied the largest area in both 1995 and 2006. Croplands expanded slightly, while urban system almost doubled. Urban sprawl mostly took place on the beach and coastal dunes (foredunes) (Table 3).

Finally, the landscape at Costa Esmeralda revealed six different land use types: beach, mangrove, scrubland, cropland, grassland, and urban systems (Fig. 4). The beach covered the smallest surface in both years (Table 4). Six percent of the beach and foredunes was urbanized, while 14% of originally mangrove covered areas changed into croplands (49 ha). A large proportion of the scrublands (43%) were transformed into croplands (Table 4). Croplands and grassland occupied the largest area in both 1995 and 2006 (Table 4). Nevertheless, a large area (76 ha) of the croplands became grasslands for cattle (Table 4). Urban areas increased markedly over the last decade at the expense of beach, grasslands, croplands and mangroves (Table 4).

Table 2
Transition matrix of land use change (in ha) from 1995 to 2006, at Boca del Río. Numbers in parenthesis show percentage, which refers to the total addition of each land use in 1995 (rows). Bold shows the highest transition values.

1995	2006				Total 1995 (ha)
	Beach	Mangrove	Grassland	Urban	
Beach	30 (81)	0	0	**7 (19)**	37
Mangrove	0	213 (84)	0	**41 (16)**	254
Grassland	0	0	140 (65)	**77 (35)**	217
Urban	0	0	0	1776 (100)	1776
Total 2006 (ha)	30	213	140	1901	
Total transition (ha)	−7	−41	−77	125	
Total transition (h%)	−19	−16	−35	7	

3.2. Ecosystem Services Values (ESV)

In our literature search we found ecosystem service values estimated for the eight land use types occurring in our study sites. These values have been estimated for 7 Latin American countries, most of them in Mexico (Appendix 1). The ecosystem services comprised in this set of estimations include food production, medicine and extractive plants, recreation, protection, raw material, fuel, habitat, gas regulation, and waste treatment. The highest ecosystem services values were found for the beach and coastal dunes, tropical forest and mangroves. We only used the ecosystem service values estimated for Mexico, except for two cases.

As shown in the transition matrices, land use at the three study sites has changed over the last decade and, consequently, so have their ecosystem services values. Table 5 shows the values for each ecosystem service provided by the natural ecosystems found in our studies sites. It is evident that wetlands (mangroves) and rain forest have the largest number of ecosystem services that have been valued. However, from our own calculations we found that the ecosystem services provided by coastal dunes showed the highest values (80,459 USD/ha/year) because of the very high relevance of disturbance regulation and recreation (67,874 USD/ha/year and 12,585 USD/ha/year, respectively), which are considered to be very valuable (Appendix 1). In the case of scrublands we used the same ESVs that we found for tropical rain forest, because we did not locate any study on the valuation of ecosystem services provided by scrublands. We think that this is a reasonable combination given the fact that scrublands and tropical rainforests are similar, in terms of vegetation, since they share several species and their structure (and hence functioning) is also similar.

Because of the intense urbanization at Boca del Río, this location had no gains in ESV, but losses (Table 6). This is due to the fact that, in our case, ecosystem services values for urban areas were considered as ESV = 0. In addition, urban areas increased at the expense of mangroves, beach and dunes with the largest ESV, which meant important losses in terms of ecosystem services and their economic values. Similar results were observed in Chachalacas. Significant ESV losses were estimated because of loss of particularly valuable ecosystems, such as coastal dunes and the beach. The large loss of coastal dunes into grasslands for cattle yielded a relatively high loss, again, owing to their very high ESV. Finally,

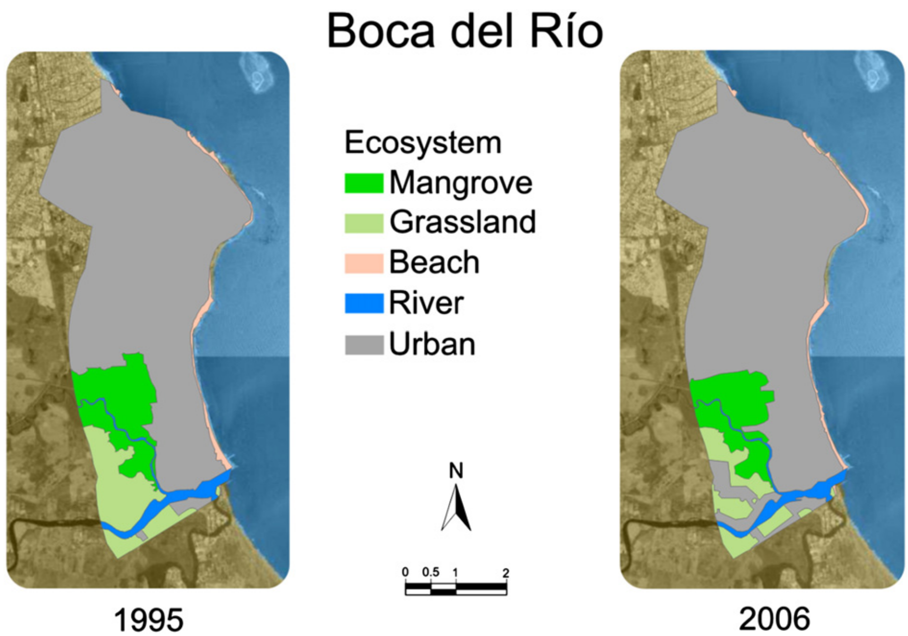

Fig. 2. Land use changes over time at Boca del Rio, Veracruz.

land use changes in Costa Esmeralda from mangrove and shrubland into grasslands for cattle and urban settlements also resulted in net losses of ESV which were relatively high because of the high ESV of mangroves. Overall, a decreased ESV resulted in an important net loss (Table 6) in Costa Esmeralda.

In brief, our calculations show that land use change from 1995 to 2006 has resulted in significant ESV losses in the three study sites. Chachalacas lost the most because of fast urbanization during the last decade, while Boca del Río lost the least, because it was already mostly urbanized in 1995. Furthermore, losses in Chachalacas are due to loss of very valuable natural ecosystems (in terms of their ESVs) to urban areas with zero ESVs.

4. Discussion

We observed that land use has changed in all three study sites, especially so in Chachalacas and Costa Esmeralda. Urban sprawl and loss of natural ecosystems into croplands or grasslands for cattle ranching have been the most recurrent land use changes. In our three study sites, mangroves and coastal dunes have decreased notoriously resulting in losses of ecosystem services and their economic values, although these changes occurred at different rates and intensities. Overall, the net yearly reduction in the total value of ecosystem services was: $\$1.4 \times 10^3$ in Boca del Río, $\$7 \times 10^5$ in Chachalacas and $\$1 \times 10^5$ in Costa Esmeralda (2006

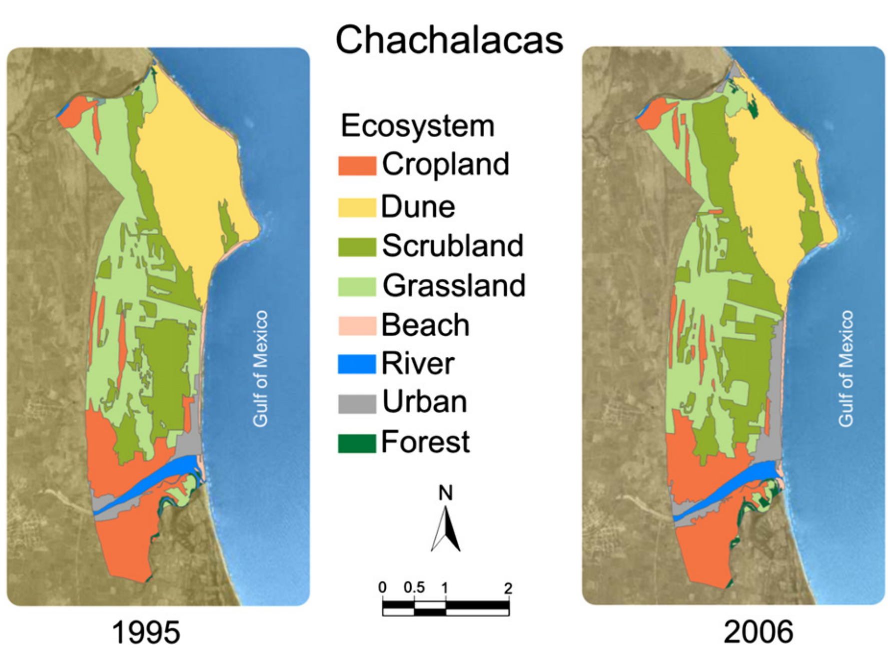

Fig. 3. Land use changes over time at Chachalacas, Veracruz.

Table 3
Transition matrix of land use change (in ha) from 1995 to 2006, at Chachalacas. Numbers in parenthesis show percentage, which refers to the total addition of each land use in 1995 (rows). Bold shows the highest transition values.

1995	2006							Total 1995 (ha)
	Beach	Dune	Scrubland	Forest	Cropland	Grassland	Urban	
Beach	17 (71)	0	0	0	0	0	**7 (29)**	24
Dune	10 (1)	577 (85)	75 (11)	6 (1)	0	11 (1)	3 (1)	682
Scrubland	0	4 (1)	415 (81)	0	9 (2)	**75 (15)**	4 (1)	507
Forest	0	0	0	9 (60)	1 (7)	**4 (26)**	1 (7)	15
Crop	0	0	3 (1)	3 (1)	448 (92)	21 (4)	10 (2)	485
Grassland	0	0	139 (17)	8 (1)	34 (4)	585 (73)	37 (5)	803
Urban	0	0	0	0	0	0	80 (100)	80
Total 2006 (ha)	27	581	632	26	492	696	142	
Total transition (ha)	3	−101	125	11	7	−107	62	
Total transition (h%)	13	−15	25	73	1	−13	78	

USD/ha/year). These values represent a significant percentage of total ESV estimated for coastal regions in Latin America where ESV was 21,636 USD/ha/year for mangroves and 80,459 USD/ha/year for beaches and coastal dunes.

These trends are the result of different factors. On the one hand, the loss of natural ecosystems whose ecosystem services have been estimated to be very valuable (such as wetlands, coastal dunes and beaches) yielded significant losses in terms of ESV, especially in Costa Esmeralda and Chachalacas. On the other hand, we were unable to locate monetized ecosystem service values for urban systems in this region. Furthermore, on top of missing information, ecosystem services in the urban areas of our study sites are minimal or non-existent since urbanization has not included natural areas such as parks within the cities, which could provide additional ecosystem services. Thus, in our case, urbanization meant zero (or insignificant) ESV. Hence, losing valuable ecosystems and increasing urban areas resulted in net loss in ecosystem services, although some ecosystem services may be locally provided by urban areas when natural areas are included within the urban landscape (i.e. city parks). ESV losses in Boca del Río were particularly low because here, the landscape was already mostly urbanized in 1995, our initial state in our analyses. That is, most natural ecosystems – and ecosystem services in consequence – had already been lost prior to the decade that we analyzed. Land use change could only be minimal in this already urbanized area, and hence, ESV were low.

Because of global population growth, deforestation, agriculture expansion and biodiversity decline, similar results have been observed in different parts of the world. For example, in China, Zhao et al. (2004) assessed ESV change in Chongming Island using 5 land use types. Following a methodology similar to ours, they found that ESV declined by 62%, mostly because of important wetland and marsh losses. In a different study, Li et al. (2007) found a net loss of US$24.1 $\times 10^6$ of ESV for tropical forest in Pingbian Miao Autonomous County, China. In San Antonio, Texas (Kreuter et al., 2001) estimated a 65% loss in the area used for rangeland and a 29% increase in urbanized land between 1976 and 1991 and a cumulative loss of ESV valued at US$6 $\times 10^6$. All these studies mirror our findings that land use change and loss of natural habitats have resulted in significant losses of ES and their corresponding economic values (Table 6). Similar to the above mentioned studies, we found that ESV's decreased 16% in Boca del Rio, 13% in Chachalacas and 10% in Costa Esmeralda (Table 6).

Frequently and for obvious reasons, food production and raw material extraction are important drivers of land use change in different parts of the world (Lambin et al., 2003). For example Zhao et al. (2004), showed that fishing activities in Dongtan, China, during centuries, have currently resulted in declining wetlands and marshes, favoring a few ecosystem services (i. e. food production) at the expense of additional services, such as disturbance and gas regulation, habitat and refuge, pollination, recreation, waste treatment, water regulation and water provision. Similarly, Li et al. (2007) found that the most common land use change drivers were agriculture expansion and human population growth (urban sprawl), which resulted in deforestation and loss of biodiversity, in coincidence with our observations.

The environmental cost of land use changes is slowly beginning to be considered in the decision-making process regarding land use. For instance, among the goals of the development plan of the state of Veracruz 2005–2010 the relevance of protecting "ecosystems that provide goods

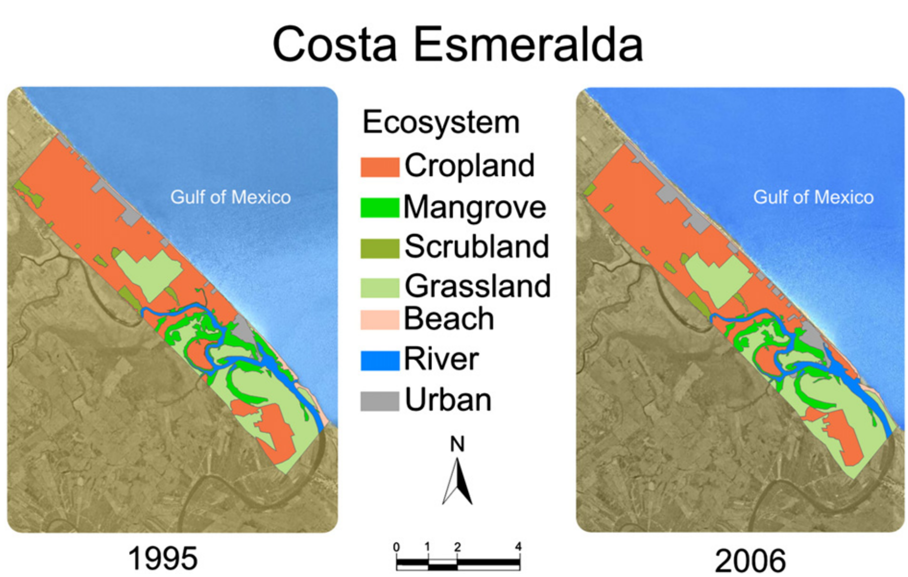

Fig. 4. Land use changes over time at Costa Esmeralda, Veracruz.

Table 4
Transition matrix of land use change (in ha) from 1995 to 2006, at Costa Esmeralda. Numbers in parenthesis show percentage, which refers to the total addition of each land use in 1995 (rows). Bold shows the highest transition values.

1995	2006						Total 1995 (ha)
	Beach	Mangrove	Scrubland	Crop	Grassland	Urban	
Beach	51 (94)	0	0	0	0	3 (6)	54
Mangrove	0	278 (79)	0	49 (14)	19 (5)	6 (2)	352
Scrubland	0	0	30 (57)	**23 (43)**	0	0	53
Crop	0	0	8 (1)	1559 (91)	76 (4)	74 (4)	1717
Grassland	0	0	15 (2)	11 (1)	775 (96)	5 (1)	806
Urban	0	0	0	0	0	132 (100)	132
Total 2006 (ha)	51	278	53	1642	870	220	
Total transition (ha)	−3	−74	0	−75	64	88	
Total transition (h%)	−6	−21	0	−4	8	67	

and services such as water supply, soil fertility, weather regulation and habitat support, with the goal of improving economic activities in the state" was clearly stated. This plan recognized the need to evaluate ecosystem services in order to identify the costs and benefits of state development policies (SEFIPLAN, 2005). However, although the relevance was acknowledged, so far, ecosystem services valuation have not been considered for these plans. Because of this, the calculation of the economic value of the ecosystem services provided by natural ecosystems is particularly important given the fact that the coast of Veracruz is becoming increasingly developed in order to promote tourism. No doubt, these plans will result in high impact on the natural remnants of coastal ecosystems. Thus, the contradiction stands out between the state development policy aiming at sustaining ES and the development projects for the coast which are intensive and extensive and vaguely consider the conservation of natural ecosystems.

The loss of mangroves, coastal dunes and beach was perhaps the most important change in the last decade. Certainly, it seems like protection against storms and floods will become more relevant given current climate change scenarios that predict increased impact and frequency of hurricanes (Emanuel, 2005; Webster et al., 2005). Veracruz is very susceptible to the impact of storms and floods, thus, making the conservation of these ecosystems highly relevant to society (Pérez-Maqueo et al., 2007). Additionally, the scenic beauty of the beaches at Veracruz (Mendoza González, 2009) provides further evidence of the relevance of preserving these ecosystems with very high economic and ecological relevance.

4.1. Methodology Caveats

A range of methodologies is available to value ecosystem services as well as changes in them. These valuations take into account both use and non-use values that society and individuals gain from ecosystem services or lose when they are degraded or lost. Most ecosystem services are not traded in markets and thus, need to be valued using non-market pricing techniques because they remain unpriced. Non-market values of goods and services are difficult to measure. Therefore, more indirect means of valuation must be used. A spectrum of valuation techniques frequently used to establish values in the absence of direct market values is described in Table 7. Each valuation methodology has its own strengths and limitations which restricts its use to a range of ecosystems and ecosystem services. The type of valuation technique to be used, therefore, depends on the type of ecosystem service to be valued, the ecosystem itself and the quality and quantity of the information available.

In the absence of site-specific valuation information the benefit transfer method is an alternative to estimating non-existing values. All the different valuation approaches described in Table 7 can be used in benefit transfer methods. This method adapts the existing valuation information or data to new policy contexts. That is, the benefit transfer method involves obtaining non-market values from previously estimated calculations for similar goods, services and ecosystems, through the analysis of a single study or a group of studies (with either similar or different valuation methodologies). The transfer refers to the application of values and other information from the original study site to a new policy site (DEFRA, 2007; Desvousges et al., 1992). This method is especially useful when collection of primary data is not feasible due to budget and time constraints (Kreuter et al., 2001 –NJ; Wilson and Hoehn, 2006). In addition, the benefit transfer method provides useful information especially when ecosystem service values are considered as either negligible or zero because they have been ignored.

We are aware of the limitations of the benefit transfer method, which have been greatly debated in the environmental economics literature (Brouwer, 2000; Smith et al., 2002). Addressing these discussions is outside the scope of this paper. Nevertheless, we will mention some of these limitations and explain the strategies followed to minimize them. The critical underlying assumption of the transfer method is that the economic value of ecosystem goods or services at the study site can be inferred with sufficient accuracy from the analysis of existing valuation studies at other sites. That is, this method assumes that the study site is sufficiently similar to the reference site, particularly in terms of consumer preferences and environmental

Table 5
ESV calculated for each land cover type. The number in parenthesis is the number of references found for each ES, evaluated in Mexico (except for Medicine in Agricultural systems and Grasslands – total Ecosystem service values). The values are in US dollars/2006 per ha per year.

Service/ecosystem	Dune/beach	Mangrove	Tropical forest/scrubland	Grassland	Agriculture	Urban	Total
Food production		1225 (2)	167 (1)		875 (1)		2267 (4)
Fuel			24 (1)				24 (1)
Gas regulation			121 (2)				121 (2)
Habitat			7 (1)				7 (1)
Medicine and extractive plant			84 (1)		487 (1)		571 (2)
Raw materials		1.45 (1)	33 (1)				34.45 (2)
Recreation	12,585 (1)	155 (1)					12,740 (2)
Protection	67,874 (1)						67,874 (1)
Waste treatment		569 (2)					569 (2)
Total value		21,636 (1)		385 (1)			22,021 (2)
Total	80,459 (2)	11,793 (7)	436 (7)	385 (1)	1362 (2)		

Table 6
Total ESV changes (USD 2006) estimated for all ecosystems found in Boca del Río, Chachalacas and Costa Esmeralda, Veracruz, from 1995 to 2006.

Land use	Total ESV of Boca del Río (USD)	Total ESV of Chachalacas (USD)	Total ESV of Costa Esmeralda (USD)
Cropland	0	9462	−107,048
Mangrove	−1896	0	−776,750
Dune	0	−7,881,379	0
Grassland	−148	−40,657	21,636
Beach	−12,093	202,810	−195,810
Urban	0	0	0
Forest	0	3293	0
Scrubland	0	43,713	0
Total	−14,137	−7,662,757	−1,057,972

quality and conditions. The technique also assumes that the reference site has the same market structure, substitute services, and access to those services at the current study site (Ready and Navrud, 2006).

There are different ways of transferring values from the study context to the study context: a) direct transfer from one primary study; b) to transfer the estimated values after adjusting for economic (income) differences; c) to transfer values from meta-analyses which take the results from a number of studies and adjusts for differences in local income (DEFRA, 2007; Pascual et al., 2010). In our case, and being aware of the limitations of the benefit transfer method, we were careful to use it under specific conditions. Firstly, we searched the refereed literature to find as much information as possible on the economic value of ecosystem services provided by the natural ecosystems found in our study sites. From this, we chose the studies that were useful to us, after considering: a) that the good or service being valued was similar between the reference site and the current study sites and b) the site where they are applied and that the populations affected had similar socioeconomic characteristics. c) Of course, the original estimates being transferred had themselves to be reliable in order for any attempt at transfer to be meaningful. In concordance with this, we used mostly studies from Mexico and from refereed literature – sometimes local literature. In addition to the transfer value method, we used direct market prices for some ecosystem services provided by coastal dunes and beaches in Mexico, in the Gulf of Mexico. It became evident that these values are very high, as well as those calculated for Chile (see Appendix 1). Furthermore, and in order to account for potential economic differences between the study site and the policy site, our estimations were always adjusted to US dollars 2006 using the Consumer Price Index (CPI) and the Power Purchasing Parity (PPP) of US dollars of 2006. We tried to minimize the limitation explained above by transferring the original values to the US economy to make the final estimations, which made our results more readily comparable with other similar studies.

Certainly, the perfect method for valuing ecosystem services may not exist. But still, these non-precise valuations are useful for many appraisal purposes. These practical appraisals provide the means to compare the relative magnitude of changes in the provision of ecosystem services under different scenarios and options as, for example, land use change. In this case, despite the acknowledged limitations such as the context sensitivity of value estimates that affect the benefit transfer method, when applied carefully this method provides a credible basis for policy decisions involving sites other than the study site for which values were originally estimated.

Finally, a gap analysis of the studies that we used for our ESV calculations shows that not all ecosystems have been well studied and some have not been studied at all. An incomplete coverage is a relevant issue. A more complete coverage of ecosystem services would almost certainly increase the aggregate values of ecosystem services per ecosystem. All in all, additional estimates of ESV using local information would probably reveal a higher ESV, especially when people realize that ecosystems provided more services than they had previously been aware of.

Table 7
Benefits and limitations of the different valuation methods used to estimate the economic value of ecosystem services (DEFRA, 2007; Pascual et al., 2010).

Valuation method	Description	Benefits	Limitations
Avoided cost	Value based on the avoided costs of damages due to lost services (storm protection).	Market data available and robust.	It varies largely from place to place because properties values vary from place to place. Can overestimate actual value.
Replacement cost	It is based on the price of the cheapest alternative way of obtaining that service (water treatment).	Market data available and robust.	The technology to replace many ecosystem services does not always exist.
Travel cost	Willingness to pay to have access to specific ecosystem services (visiting Monarch butterflies). Willingness to pay to travel to specific locations.	Based on observed behavior.	Highly sensible to cultural and social differences Limited to recreational benefits. Difficult to apply when trips include multiple destinations.
Hedonic pricing	Prices people are willing to pay for ecosystem services such as scenic beauty (housing prices; hotel fares).	Based on market data, readily available and robust.	Highly sensible to cultural and social differences
Contingent valuation	Willingness to pay for specific scenarios (preserve ecosystem services)	Allows to capture use and non-use values.	Highly sensible to cultural economic and social differences. Biased responses; hypothetical nature of the market.
Marginal product	Modeling to estimate output value in response to material input.	Allows to capture use and non-use values.	Requires good information.

5. Conclusions

Our study shows that, over time, the expansion of agriculture, livestock and urban sprawl has had a direct impact on ecosystem services and their non-market economic values. Beaches, dunes and wetlands were the ecosystems whose surface was reduced the most, and yet, they were the ecosystems with the highest estimated ESV, such as water supply, recreation and disturbance regulation. It is interesting to note that it could be thought that the ESV that are lost due to land use change are over-compensated by the gained economic value of urban land, where per hectare prices may fluctuate from 45,000 to 600,000 USD/ha (2006) in Costa Esmeralda, to 190,000 to 910,000 USD/ha (2006) in Chachalacas and 1×10^6 to 6×10^6 USD/ha (2006) in Boca del Río. The estimated Ecosystem Service Values in this study, even when added altogether, are much lower than these: 106,000 USD/ha. Thus, it could make sense to change land use from natural to urban, because it may seem as economically convenient. However, when ecosystem services such as coastal protection or scenic value and recreation are lost, then these apparent gains from urban development are also lost, because the quality of the beaches deteriorates or because the impact of hurricanes and storms may result in large economic losses. When this occurs, the losses are surpassed by the gains. Certainly, economic gains that are aimed to be achieved in development projects along the coasts should consider the important non-market economic losses that occur as natural ecosystems are lost or transformed. This is especially true for scenic beauty and protection.

Finally, in addition to the economic valuation of ecosystem services and of considering these services during decision making processes, it is also to include socio-cultural preferences toward ecosystem services to identify the impact of different management options on the society and the ecosystem's capacity to deliver services (Martín-López et al., 2012).

In an ever increasing agricultural and urban setting worldwide (especially at the coast), land use change may seem economically profitable. However, when land use changes deplete the ecosystem's capacities to deliver ecosystem services, medium and long-term losses may exceed these short-term gains. Land use and policy making should aim at balancing society's needs and preferences, while considering ecosystem service losses since in the long run, it will be beneficial for all of us, while natural ecosystems are preserved and used adequately.

Acknowledgements

This study was financed by grant FOMIX Veracruz-Conacyt (37009) and with the financial support of becas mixtas from CONACyT (204461) to GMG. GMG and DL are thankful for their MSc scholarships CONACyT (204461 and 275429/224619 respectively). We also would like to thank the support from the Gund Institute of Ecological Economics, University of Vermont, especially to Robert Costanza for his insightful contributions.

Appendix 1

Detailed ecosystem service values calculated in different Latin American countries for the ecosystems that are also found in our study sites. For comparison, GDP and Gini index values are provided. We mostly used values estimated for Mexico, except otherwise stated.

Land cover	Ecosystem service	Reference	Study site	2006 US dollars per ha/year	GDP per capita/year 2006	GINI index/year
Beach and dunes						
	Recreation	This study	Chile	$8659	8864.73	54.92 (2003)
	Recreation	This study	Mexico	$12,585	8051.92	46.05 (2004)
	Protection	This study	Mexico	$67,874	8051.92	46.05 (2004)
Total Mexico				$80,459		
Total Latin America				$8659		
Mangrove						
	Food production	Agüero, 1999	Mexico	$1259	8051.92	46.05 (2004)
	Food production	Yáñez-Arancibia and Lara-Dominguez, 1995	Mexico	$1191	8051.92	46.05 (2004)
		Mean food production		$1225		
	Raw materials	Agüero, 1999	Mexico	$1.45	8051.92	46.05 (2004)
	Recreation	Sanjurjo, 2004	Mexico	$155	8051.92	46.05 (2004)
	Waste treatment	Agüero, 1999	Mexico	$952	8051.92	46.05 (2004)
	Waste treatment	Lara-Domínguez et al., 1998	Mexico	$186	8051.92	46.05 (2004)
		Mean waste treatment		$569		
	Total value	Lara-Domínguez et al., 1998	Mexico	$21,636	8051.92	46.05 (2004)
Total Mexico (mean of total value - Lara-Domínguez et al., 1998 - and total from all other studies)				$11,793		
Tropical forest						
	Food production	Adger et al., 1995	Mexico	$167	8051.92	46.05 (2004)
	Fuel	Adger et al., 1995	Mexico	$24	8051.92	46.05 (2004)
	Gas regulation	Adger et al., 1995	Mexico	$127	8051.92	46.05 (2004)
	Gas regulation	Adger et al., 1995	Mexico	$115	8051.92	46.05 (2004)
	Gas regulation	Smith et al., 1998	Peru	$68	3287.74	
		Mean gas regulation (Mexico)		$121		
	Habitat	Herrador and Dimas., 2000	Costa Rica	$47	5046.90	49.76 (2003)
	Habitat	Adger et al., 1995	Mexico	$7	8051.92	46.05 (2004)
	Medicine	Adger et al., 1995	Mexico	$84	8051.92	46.05 (2004)
	Raw material	Balick and Mendelson, 1992	Belize	$810		
	Raw material	Godoy et al., 1993	Honduras	$13	1255.6	53.84 (2003)
	Raw material	Adger et al., 1995	Mexico	$33	8051.92	46.05 (2004)
	Raw material	Baltodano, 2005	Nicaragua	$978	1022.81	43.11 (2001)
	Raw material	Peters et al., 1989	P	$797	3287.74	52.02 (2003)
		Mean raw material (Latin America)		$649		
Total Mexico				$436		
Total Latin America				$764.5		
Agriculture						
	Food production	Economic Research Service, USDA, 2006	Mexico	$875	8051.92	46.05 (2004)
	Food production	Barsev, 2002	Nicaragua	$34	1022.81	43.11 (2003)
	Medicine and extractive plant	Balick and Mendelson, 1992	Belize	$487	4094.42	
Total Mexico				$1362		
Total Latin America				$521		
Grassland						
	Total value	Seidl and Moraes, 2000	Brazil	$385	5659.74	56.99 (2004)
Total Latin America				$385		

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