Selecting cost-effective areas for restoration of ecosystem services

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Abstract

Selection of areas for restoration should be based on cost-effectiveness analysis to attain the maximum benefit with a limited budget and overcome the traditional ad hoc allocation of funds for restoration projects. Restoration projects need to be planned on the basis of ecological knowledge and economic and social constraints. We devised a novel approach for selecting cost-effective areas for restoration on the basis of biodiversity and potential provision of three ecosystem services: carbon storage, water depuration, and coastal protection. We used Marxan, a spatial prioritization tool, to balance the provision of ecosystem services against the cost of restoration. We tested this approach in a mangrove ecosystem in the Caribbean. Our approach efficiently selected restoration areas that at low cost were compatible with biodiversity targets and that maximized the provision of one or more ecosystem services. Choosing areas for restoration of mangroves on the basis carbon storage potential, largely guaranteed the restoration of biodiversity and other ecosystem services.

Introduction

Because habitat for conservation is scarce, restoration has become an integral part of land management (Hobbs & Harris 2001). Increasingly, projects to restore endangered habitats are being established worldwide as a tool to increase the amount of area designated for the preservation of biodiversity (Dobson et al. 1997).

For restoration projects to be successful, they need to be planned on the basis of ecological knowledge and consider economic and social constraints (Miller & Hobbs 2007). Recently, the use of ecosystem services has been advocated for valuing the importance of ecosystems (Costanza et al. 1997). The term *ecosystem services* reflects the idea that ecosystems can be viewed as providers of services to society (MEA 2005). The services are classified as provisioning services (e.g. timber, fish), supporting services (e.g. nutrient cycling, primary production), regulating services (e.g. water purification, erosion regulation), and cultural services (e.g. recreation, aesthetic) (MEA 2005). The selection of ecosystem services as goals of restoration could facilitate the inclusion of economic and social interests in restoration projects, while increasing biodiversity (Benayas et al. 2009).

Restoration of ecosystems is expensive, and funds are usually limited. Defensible allocation of resources for restoration projects is critical to maximize benefits. To determine priority areas for restoration, spatial prioritization algorithms such as stochastic dynamic (Westphal et al. 2003) and integer programming (Crossman and Bryan 2006) have been used. Another spatial prioritization tool, Marxan, is a well-developed tool that has been extensively tested and is possibly the most widely used conservation planning software (Ball et al. 2009).

Marxan balances cost and benefit, allowing the inclusion of spatial requirements, such as level of connectivity and compactness of selected priority areas. Marxan was originally developed for planning the location of protected areas (Ball et al. 2009), but it has recently been used for establishing priority areas for revegetation (Renwick et al. 2014). We used Marxan to establish priority areas for restoration on the basis of ecosystem service potential. The study was performed in a degraded mangrove forest in the Mexican Caribbean.

Mangroves are one of the most valuable ecosystems in the world and are one of the most threatened; mangrove deforestation rates are higher than those of other tropical forests (Duke et al. 2007). Mangroves improve the water quality of creeks and rivers adjacent to them (Adame et al. 2010); stabilize and protect shorelines from physical disturbances such as wave surges, storms, and hurricanes (Barbier 2006); and contain large carbon (C) stocks (Donato et al. 2011). Expensive mangrove rehabilitation projects that aim to recuperate the ecosystem services these ecosystems provide have been implemented in many countries (e.g. Mexico, Zaldivar et al. 2010).

Measuring ecosystem services can be extremely challenging (Carpenter et al. 2008). It is difficult to select parameters that approximate the importance of a service and then translate the worth of the service into monetary value that can be applied in policy making (Bingham et al. 1995). Moreover, ecosystem services are not linear in space or time (Koch et al. 2009). However, some physical characteristics can be used as indicators of which ecosystem services can be expected (Ewel et al. 1998, Feller et al. 2010). For example, mangrove forests with tall trees and deep organic peat soils are characterized by large C stocks (Donato et al. 2011; Bouillon 2012;

Adame et al. 2012) (Table 1). Similarly, mangroves with high biomass are likely to absorb more nutrients than small mangrove trees and, thus, are more important in reducing nutrients, thereby improving water quality (Reich et al. 2006). In contrast, large stands of dense medium sized mangroves are the most likely to provide coastal protection (e.g. Tanaka et al 2007; Doyle et al. 2009; Koch et al. 2009). Finally, the location of a mangrove forest is also crucial in defining the services it provides. For example, for a mangrove forest to be important in water depuration, it would have to be flooded by water of poor quality (Verhoeven et al. 2006; Adame et al. 2010, 2012) (Table 1). We took field measurements of physical (forest structure, species composition, location) and chemical traits (nutrients, carbon) of mangroves that approximate the ecosystem services they deliver (Ewel et al. 1998). We used these data, in a novel approach in which provision of three ecosystem services (carbon storage, water depuration, and coastal protection) and the cost of restoration were used to identify priority areas for restoration .

Methods

Study Site

Nichupte Lagoon is on the Yucatan Peninsula, near Cancun City, in the Mexican Caribbean (Fig. 1). Large areas of mangroves surrounding Nichupte Lagoon are degraded, mainly as a result of the impacts of category-five hurricane Wilma (2005). Seven years after the hurricane, areas of mangroves are still highly degraded, which is unusual because mangroves generally start to regrow 1 year after a hurricane and can recover to pre-disturbance conditions within 3-7 years (Baldwin et al. 2001). It is likely that many of the ongoing stressors to the mangroves in the lagoon (water pollution, hydrological modifications, erosion) are impeding the

recovery of this forest. As a result, restoration efforts including hydrology restoration and afforestation were implemented in 2009.

Field sampling

Field sampling for this study was conducted in several trips from September 2010 to February 2012. We classified the mangroves of Nichupte as natural vegetation in good condition, moderately degraded vegetation, or severely degraded vegetation. The level of degradation was assessed as the proportion of dead trees and the tree density of the stand. Natural vegetation in good condition was further divided in three types: marsh with sparse mangrove trees, scrub mangroves (dense stands of trees < 1.5 m tall), and fringe or basin mangroves (stands of trees > 2 m tall at the edge or inland of the lagoon) (Lugo & Snedaker 1974). On the basis of this classification, we measured forest structure in 25 sites (5 marsh and mangroves, 10 scrub mangroves, and 10 fringe or basin mangroves) and soil characteristics in 11 sites (3 marsh and mangroves, 4 scrub mangroves, and 4 fringe or basin mangroves) (Fig. 1).

To measure tree biomass, tree density, and forest composition we used two approaches. In 7 sites, we used 50- to 100-m transects following the point-centred quarter method (Dahdouh-Guebas & Koedam 2006). In the other 18 sites, we used plots (10 x 10 m), where all the trees with diameter at breast height >2 cm were measured (Cintrón & Schaeffer-Novelli 1984). Allometric equations were used to determine tree biomass (Cintrón & Schaeffer-Novelli 1984; Smith & Whelan 2006; Hegazy et al. 2008; Komiyama et al. 2008).

Carbon stocks were calculated following a method similar to that of Kauffman and Donato (2012). To estimate tree C stocks, we multiplied aboveground biomass by 0.48 and

belowground biomass by 0.39 (Kauffman & Donato 2012). To calculate the soil C stock, we collected three soil cores at each sampling site with a 6.4-cm open-face peat auger. The core was systematically divided into depth intervals of 0-15 cm, 15-30 cm, and 30-50 cm. Samples of known volume were collected in the field and dried to constant mass to determine bulk density. The concentration of C was determined by dry combustion method (induction furnace) with an Elemental Analyzer (FlashEA 1112, Thermo Quest, Milan, Italy). We corrected for the amount of inorganic C from organic C with the loss of ignition method in a separate set of samples (Heiri et al. 2001).

We sampled water quality monthly from 2005 to 2006 at 30 stations (Supporting Information). Water was analyzed for ammonium (NH_4) with the phenol-hypochlorite method, nitrite (NO_2) with the sulfanilamide method, and nitrate (NO_3) as the reduction to NO_2 with a cadmium-copper column. Phosphorus as soluble reactive phosphorus (SRP) was measured with the mixture of reagents technique (Strickland & Parsons 1972). Chlorophyll *a* was determined by filtering 1 L of water through a membrane filter of 25 mm diameter (cellulose nitrate membrane filter of 0.45 μm pore size; Merck Millipore, MA, USA) and extracting the pigments with 90% acetone. Dissolved oxygen was measured in the field with a YSI 6600 multi-parameter probe (YSI, Xylem, Ohio, USA). A trophic index (TRIX), which is commonly used as an indication of water quality (Jorgensen et al. 2005), was calculated on the basis of the amounts of nutrients, Chlorophyll *a* and dissolved oxygen (Supporting Information).

The TRIX values were used to produce a map of water quality of the lagoon (kriging extrapolation as gridding method, Surfer v10.1.561, Golden Software, Colorado, USA)

(Supporting Information), which was extrapolated to the adjacent mangrove forests in order to estimate the quality of the water flooding them.

Vegetation mapping

To create a vegetation map, we used SPOT images of 10 x 10 m resolution from 2006. The images were analyzed with the geospatial analysis software TNTmips (Microimages, NE, USA). We based the classification of vegetation types on the non-supervised categories obtained from the fractional cover algorithm means on the green and medium infrared bands and normalized difference vegetation index (NDVI), which includes cover and mangrove density (Giri et al. 2007). This index gives a good separation of dry and wet soils, which helps differentiate among vegetation types that are usually a result of differing flood regimes and represent distinct biological communities (Lugo & Snedaker 1974). We also used the transformed vegetation index (TVI) that considers vigour or the capacity of the aerial biomass to provide a spectral signal (Ghormley & Merri 2011). Vegetation classes were assigned on the basis of field experience and field data. Most of the lagoon was composed of areas of graminoids (*Cladium jamaicense*, *Typha domingensis*) with sparse mangroves, mostly *Rhizophora mangle*, but also *Conocarpus erectus* (1,786 ha, 40% of total wetland vegetation area). Scrub mangroves of *R. mangle* were the second most common wetland type within the lagoon (1,329 ha, 29%). Fringe and basin mangroves were dominated by *R. mangle* but had some patches of *Languncularia racemosa* and *Avicennia germinans*. This association accounted for 12% of the total vegetation area (539 ha)(Fig. 1). The next most abundant association was low-density stands of *R. mangle* and *A. germinans* (450 ha, 10%), which are considered successive vegetation

of degraded mangroves. Severely degraded mangroves were stands of dead trees, which accounted for 9% of the total vegetation (417 ha) (Supporting Information).

Spatial analyses

We used the vegetation type map as a template for mapping field data. We created a map for C stocks, aboveground biomass, tree height, tree density, and forest width by selecting the value of the closest sampling point and assigning it to the vegetation type unit. For each map, we added a grid of 225 m² (15 x 15 m) with ET GeoWizard extension. Each of the resulting squares was considered a planning unit. For each planning unit, we calculated with Zonal Statistics (ArcGis v10.1, ESRI, CA, USA) the vegetation type (proportion of each planning unit covered by each type), C stock, aboveground biomass, mean tree height, tree density, and the width of the forest in which that unit was found. For units with degraded vegetation, we calculated their potential height, basal area, and biomass on the basis of adjacent healthy areas. For water quality, we assigned the dominant TRIx for that unit. These calculations allowed us to estimate the rehabilitation potential for ecosystem services (carbon storage, water depuration, and coastal protection) for each unit. We measured carbon storage as the C stock for each unit and used 2 surrogates of water depuration, water quality and tree biomass. The potential for water depuration is highest where tree biomass is largest and water quality is poor (Reich et al. 2006; Adame et al. 2010). However, where nutrient enrichment is very high, trees have low production, high mortality, and high soil subsidence, all which will negatively affect the successful restoration of a site (Verhoeven et al. 2006; Lovelock et al. 2009). Thus, in this project, we prioritized restoration units where tree biomass was high and nutrients in the floodwater were intermediate.

For coastal protection, we calculated a coastal protection index (CPI). We assumed that vegetation efficient at attenuating wave, wind, and flooding is characterized by dense stands of medium height trees in continuous stands of forests that are of at least 100 m wide (Table 1 and references therein). Thus, planning units with a mean tree height < 2 m or > 4 m were assigned a tree height index value of 0, whereas units with a mean height of 2-4 m were assigned a value of 1. Planning units that were part of a continuous forest (>100 m in width [Mazda et al. 1997a]) were given higher ranks on the forest width index (which was a continuous value that ranged from 0 to 1) relative to units that were part of isolated patches of vegetation.

$$\text{CPI} = \text{forest width index} * \text{tree height index} * \text{tree density.} \quad (1)$$

Cost-effective areas for restoration

We used Marxan (Ball et al. 2009) to identify cost-effective areas for mangrove restoration that would achieve conservation targets for biodiversity while securing the maintenance of ecosystem services (similar to Chan et al. 2006). We used our vegetation types as surrogates for biodiversity (Lugo and Snedaker 1974; Ball et al. 2009). The cost of restoration was assumed to be US\$5077 ha⁻¹, which is the cost of hydrological restoration and afforestation of projects in the region (Herrera et al. 2012). The cost of restoration was balanced on the basis of accessibility and degradation, assuming that the most expensive sites are the most degraded and least accessible.

Rehabilitation targets for ecosystem services

To evaluate rehabilitation needs for restoration or maintenance of each ecosystem service, we ran multiple scenarios with different combinations of targets. We started from a scenario where we set a constant target of restoring or maintaining 20% of the mangrove area that is suitable for the maintenance of biodiversity. We then increased sequentially the target mangrove area suitable for the delivery of the three ecosystem services combined (C storage, water depuration and costal protection) from 30% to 40% to 50%. In a second scenario, we set again a constant target of mangrove area for the maintenance of biodiversity of 20%, but in this case we selected a target of 50% for one ecosystem service at a time and 0% for the rest of the services. In this way, we highlighted priority restoration areas for each ecosystem service.

For each project, we calibrated the data set for the species penalty factor, which refers to the penalty applied when an ecosystem service or biodiversity indicator does not meet its target. We also calibrated for the boundary length modifier, which controls the boundary length or compactness of the solution. The results are shown as maps of best solutions (i.e., the selection of units that best reached all target areas) and as maps of irreplaceability (i.e., how often each unit was selected and thus how irreplaceable that unit was to accomplishing the best solution).

We calculated the cost of restoration to achieve different target areas of mangroves to deliver all ecosystem services. Finally, we tested the surrogacy value of each ecosystem service for the remaining services and biodiversity conservation. We ran three independent planning scenarios, including one for each ecosystem service (target=50%) at a time, and then checked how well the remaining services and biodiversity values were represented within the best solution. An ecosystem service could be considered a good surrogate for other services and biodiversity if a high proportion of the targets for these features could be achieved in a conservation plan where they had not been directly included.

Results

Ecosystem services

The C stocks (including soils 50 cm deep) calculated had a mean (SE) of 215.0 Mg·ha⁻¹ (92.1); the highest values were for fringe mangroves (380.4 Mg·ha⁻¹ [34.7]) and lowest were for marsh (62. Mg·ha⁻¹ [13.3]) (Table 2). Highest C stocks were measured in the fringe forests in the mid- and northwest of the lagoon, followed by those in the southwest of the lagoon (Supporting Information).

During the sampled period, the lagoon had a mean water quality index (TRIX) of 5.6 (0.1) (range of 4.7-7.3). The highest values were at Bojorquez Lagoon, where water quality was the poorest (Supporting Information). Poor water quality was also found in the north and southwest portions of the lagoon, the former is where untreated wastewater from Cancun City is discharged and the latter is where the residues from an old garbage dump and discharges from septic tanks are being flushed into the lagoon (Granel-Castro & and Gález-Hita 2002). Mean (SE, range) dissolved NO₃⁻ and NO₂⁻ were 4.62 μmol·l⁻¹ (0.14, 1.25-9.95 μmol·l⁻¹) and 0.86 μmol·l⁻¹ (0.04, 0.32-3.98 μmol·l⁻¹), respectively, while mean NH₄⁺ concentrations were 8.04 μmol·l⁻¹ (0.23, 1.54-15.09 μmol·l⁻¹). Mean (SE, range) SRP was 0.45 μmol·l⁻¹ (0.10, 0.20-1.08 μmol·l⁻¹). Chlorophyll-*a* values had a mean of 1.41 mg·m⁻³ (0.08, 0.18-4.65 mg·m⁻³). Finally, during the sampling period, dissolved oxygen was 4.3 mg·l⁻¹ (0.1, 1.0-7.8 mg·l⁻¹).

Areas of vegetation most likely to provide protection from hurricanes and other tropical storms were large stands of dense medium-sized (2-4 m) trees (Table 1). Fringe and basin

mangroves were the tallest, while scrub mangroves were the densest (Table 2). The tallest mangroves were in fringe and basin forests in the mid portion of the lagoon (up to 8 m tall), while the densest scrub forest were in the north of the lagoon (23,900 trees ha⁻¹). The largest uninterrupted stands were in the west and southwest, where marshes and scrub mangroves form extensive patches of vegetation (>150 ha). In the east, vegetation was sparse; there were small patches of <5 ha of marsh and mangroves.

Cost-effective areas for restoration

When including all ecosystem services in the analysis (Fig. 2), mangroves in the north and mid-western areas were cost-effective for restoration or conservation. Maps for both best solution and irreplaceability show similar results, highlighting sites where restoration efforts should be focused or, alternatively, if mangroves are not very degraded, where conservation would efficiently maintain ecosystem services.

When paying attention to a particular ecosystem service (Fig. 3), the mangroves from the northwestern and mid-western portion of the lagoon appeared to be especially important for the maintenance of C stocks. These areas represented large expanses of the tallest fringe or basin mangroves, were highly accessible, and were where the mangroves were partially degraded. For water depuration, mangroves in the northwest were especially important sites for restoration. The selection of such an area was the result of relatively poor water quality and accessible tall mangroves that were moderately degraded. As restoration areas with high potential for coastal protection, Marxan selected a variety of units that included fringe or basin and scrub mangroves situated close to access points (Fig. 3). Mangroves west of the lagoon appeared to be especially

important because they comprised a large expanse of dense forest of medium height that was moderately to severely degraded.

Cost of restoration versus ecosystem service delivery

Increasing the ecosystem service target from 10% to 20% of the area resulted in an almost 3-fold increase in the cost of restoration, while the increase from 20% to 30% of the area increased costs by <20% (Fig. 4).

Surrogacy of ecosystem services

When testing for surrogacy among ecosystem services and biodiversity, when we planned for restoration areas on the basis of carbon storage, a high proportion of the targets were met for the other two ecosystem services (water depuration and coastal protection) without their being directly included. This suggests that if restoration or conservation areas are selected on the basis of carbon storage the delivery of the other ecosystem services are largely guaranteed. Carbon storage was associated with the three most abundant vegetation types (Table 3).

Discussion

With our approach, we selected cost-effective areas for restoration of vegetation on the basis of potential ecosystem service provision. Our method can help managers select areas for restoration that can efficiently comply, at low cost, with the goal of delivering the highest level of ecosystem services while protecting biodiversity. Our results show that it is possible to implement restoration actions to both protect biodiversity and achieve high benefits for society, thus increasing financial incentives for restoration (Balvanera et al. 2001).

In some cases, the selection of one ecosystem service over the other results in a trade-off of increasing one service at the expense of another service (Carpenter et al. 2008). However, we found that selecting cost-effective areas for restoration of mangroves on the basis of C storage largely guaranteed the provision of two other services: coastal protection and water depuration. Thus, the inclusion of restoration projects in C markets or other trading schemes could provide multiple ecosystem services in addition to C storage. For example, Cancun has a population of 676,000 (2009), with a mean annual emission of 3.72 t CO₂ per capita (IEA 2011). Thus, Cancun produces at least 2.5 million Mg CO₂ every year. Nichupte stores at least 0.9 million Mg C (3.4 million Mg CO₂) in 4,520 ha of wetlands. A target of conserving or restoring 30% of the mangrove area of Nichupte Lagoon would store 40% of the annual emission of Cancun (1.0 million Mg·C). The target could be accommodated if the C sequestration rates of different vegetation communities were known and a specific target (e.g., neutral emissions) was required. For example, soil C sequestration rates in Southern Mexico are approximately 1.2-1.5 Mg C·ha⁻¹·yr⁻¹ (M.F.A, unpublished data). Therefore, the financing of a restoration project of 100 ha by a hotel or tourist company could annually offset emissions of 500 Mg CO₂·yr⁻¹ and provide the organization with a zero-emissions label.

An advantage of our methods is that it can be used with relatively simple data obtained in the field (e.g. forest structure), which provides increased confidence in results, the potential for community involvement in assessments, and avoidance of the uncertainty associated with inaccurate databases or predicted data (Wilson et al. 2005). However, analyzing ecosystem services is complex, and there are limitations in generalizing provision of services on the basis of

physical or measureable traits (Chan et al. 2006). For example, C sequestration rates vary with tidal inundation and forest age and are likely to be altered substantially by events such as hurricanes (Alongi 2011). Similarly, the service of water depuration will be modified if the quality of the floodwater changes as a result of, for example, an increase in sewage treatment (an improvement) or high rainfall events (a detriment) (Adame et al. 2012). However our analysis provides an improved alternative to the common practice of selecting areas for vegetation restoration solely on the basis of accessibility.

Our method can be modified to accommodate particular needs in different locations. For example, the cost of restoration can include the price of the land, if the land must be purchased, or the opportunity cost (e.g. logging, tourist development), as has been done in other studies focused on selecting areas for conservation (e.g. Lourival et al. 2009). Other ecosystem services could be incorporated on the basis of social preferences rather than biological traits, such as the value for tourism activities. Marxan provides different solutions with different targets for conservation or restoration and thus allows flexibility in the criteria used in the selection process and the parameters that approximate the cost (Ball et al. 2009).

There are two sites in Nichupte under restoration. One of the sites (mid-west) has high ecosystem service potential. However, the other site (mid-east) (Supporting Information) has low to medium ecosystem services provision because it is an isolated patch of mangroves that provides little protection against hurricanes. Additionally, this restoration site is flooded with relatively high-quality water and thus provides little service in water depuration. Future

alternative sites for new projects are highlighted in our maps. For example, the mangroves in the northwest of the lagoon had the overall highest score for ecosystem services provision.

The cost of restoring vegetation such as mangroves can be as high as \$510,00 ha⁻¹ when including the costs of staff salaries, technical equipment, and transportation (Spurgeon 1998). Therefore, global rehabilitation of ecosystem services requires continuous innovation and learning (Carpenter et al. 2008). We show that it is possible to systematically plan restoration projects to preserve both biodiversity and ecosystem services in a cost-effective and defensible way. Our approach could be used to prioritize areas for restoration in many other degraded ecosystems in both terrestrial and marine landscapes.

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Supporting Information

The map of carbon stocks (Appendix S1) and water quality (Appendix S2) are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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Tables

Table 1. Physical traits of mangroves that can be associated with ecosystem services.

Forest trait	Level of ecosystem service delivery ^a			Reference ^b
	low	medium	high	
Carbon stock				
tree biomass	low biomass	medium	high biomass	1,2
peat depth	shallow peat	medium	deep peat	1,2,3
soil C content	low C	medium	high C	1,2,3
Water depuration				
tree biomass	low biomass	medium	high biomass	4
nutrient exposure	low nutrients	high	medium	5, 6
Coastal protection				
Protection against wind				
tree density	sparse stand	medium	dense stand	7
tree height	short trees	tall trees	medium trees	7,8,9,10
vulnerability to damage	<i>La</i> dominated	<i>Av</i> dominated	<i>Rh</i> dominated	9, 11,12
Protection against waves				
tree density	sparse stand	medium	dense stand	13
tree height	short trees	medium	tall trees	14
forest width	narrow forest	medium	wide forest	15,16
root complexity	<i>La</i> and <i>Av</i>		<i>Rh</i> dominated	14, 15

dominated

Protection against flooding

forest width	narrow forest	medium	wide forest	14,16
resilience to flooding	<i>Av</i> and <i>La</i>		Rh dominated	11
	dominated			

^a Abbreviations: *La*, *Laguncularia*; *Rh*, *Rhizophora*; *Av*, *Avicennia*; C, carbon.

¹Donato et al. 2011, ²Adame et al. 2013, ³Bouillon 2011, ⁴Reich et al. 2006, ⁵Adame et al. 2010, ⁶Verhoeven et al. 2006, ⁷Dahdouh-guebas et al 2005, ⁸Roth 1992, ⁹Doyle et al. 1995, ¹⁰Doyle et al 2009, ¹¹McCoy et al. 1996, ¹²Sherman and Fahey 2001, ¹³Mazda et al 1997b, ¹⁴Mazda et al. 1997a, ¹⁵Badola and Hussein 2005, ¹⁶Kathiresan and Rajendran 2005, ¹⁷Tanaka et al 2007, ¹⁸Pezeshki et al. 1990.

Table 2. Mean (SE, range) forest structure (tree height and density, aboveground biomass) and carbon stocks (trees, soil up to 50 cm deep, and total C) of mangrove vegetation types within Nichupte Lagoon.

	Tree height (m)	Density (trees·ha ⁻¹)	Aboveground biomass (Mg·ha ⁻¹)	C stock (MgC·ha ⁻¹)		
				trees	soil	total
Marsh	2.1 (0.3,	7561 (461,				62.0
mangroves	1.4-3.0)	7100- 8022)	11.5 (1.3)*	8.1 (0.8)	47.5 (19.1)	(13.3)*
Scrub	1.7 (0.1,	15,889 (4020,			169.6	202.6
mangroves	1.2-2.6)	11,300-23,900)	14.7 (3.6)	10.0 (1.6)	(68.8)	(46.2)
Fringe or						
basin	4.2 (0.4)	7649 (2456)		120.4	301.8	380.4
mangroves	(2.0-8.1)	(3100-11,898)	64.9 (11.1)	(48.0)	(52.4)	(34.7)

* Aboveground biomass of marsh associated to mangroves was not measured in this study but is likely to add 16-23 MgC·ha⁻¹ to the aboveground biomass of this ecosystem and 8-12 MgC·ha⁻¹ to the total carbon stock (Adame et al. 2013).

Table 3. Surrogacy value of each ecosystem service for other services and vegetation types.^a

<i>Ecosystem service</i>	Carbon storage	Water depuration	Coastal protection
carbon storage	1.00	0.75	0.83
water depuration	0.94	1.00	0.80
coastal protection	0.92	0.71	1.00
<i>Vegetation Type</i>			
Marsh-mangroves	0.56	0.39	0.66
Fringe or basin mangroves <i>Rm/ Lr</i> ^b	0.52	0.44	0.33
Fringe or basin mangroves <i>Rm/Ag</i> ^b	0.18	0.17	0.03
Scrub mangroves <i>Rm</i> ^b	0.65	0.36	0.63

^aThe surrogacy value represents the proportion of the target achieved in a solution where only one ecosystem service was included at a time. High values (close to 1) indicate strong surrogacy of a given ecosystem for other services or vegetation types.

^bAbbreviations: *Rm*, *Rhizophora mangle*; *Lr*, *Laguncularia racemosa*; *Ag*, *Avicennia germinans*.

Figure 1. Vegetation communities within Nichupte Lagoon, Mexico (symbols, sampling points; stars, forest structure and soil characteristics (nutrients, carbon stocks) measured; squares, forest structure assessed).. To facilitate visualization, we grouped moderately and severely degraded mangroves into one category.

Figure 2. The best cost-effective areas for mangrove restoration, where ecosystem services (ES; carbon stocks, water depuration, and coastal protection) are estimated as the highest and cost of restoration (accessibility, degradation level) is relatively low, and importance (irreplaceability) of areas for achieving all the targets. We used a common target of 20% of the area for the restoration or maintenance of biodiversity and a target of 30%, 40%, and 50% of the area to provide all ecosystem services (ES) combined. .

Figure 3. The best cost-effective areas for mangrove restoration where one ecosystem service (carbon stocks, water depuration or coastal protection) is highest and where the cost of restoration (accessibility, degradation level) is relatively low, and importance (irreplaceability) of areas for achieving all the targets. We used a common target of 20% of the area for restoration or maintenance of biodiversity and a target of 50% area for one ecosystem service at a time and a null target for the rest of the services.

Figure 4. Cost of restoration (10^6 US dollars) to meet 10-50% of the area required for providing ecosystem service targets.