Landscape Changes in a Critical Subtropical Coastal Wetland in Northwestern Mexico: Is Shrimp Farming a Driver of Concern?

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Research Article

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Abstract

Coastal wetlands are critical ecosystems that are under intense pressure due to anthropogenic activities. In addition to urban growth and agriculture, shrimp farming has become one of the main drivers behind the loss of tropical and subtropical coastal wetlands. Despite its socio-economic importance, shrimp farming has high environmental costs worldwide. Consequently, it is essential to monitor shrimp farming at regional and local scales to determine if the resulting pressure on coastal wetlands is increasing. We analyzed land use/land cover (LULC) in the Bahía Santa Maria-La Reforma (BSMR) lagoon system in northwestern Mexico using remote sensing data to determine landscape and surface cover changes. We also projected future scenarios based on stochastic models and evaluated landscape metrics considering the effects of shrimp farming. Four LULC thematic maps (1985, 1994, 2002, and 2017) with overall accuracy values > 80% and two projected maps (2027 and 2037) were produced. Agriculture was the dominant LULC class in the BSMR coastal lagoon system, although saltmarshes appear as the most critical wetland type in the area. Shrimp farming, which was nonexistent in 1985, represented 4% of the total landscape in 2017. By 2037, this value is expected to increase to 5%. Saltmarshes showed negative trends due to the expansion of shrimp farming. Considering the importance of wetlands given their ecosystem services, this study highlights LULC changes due to economic activities and the need to improve management strategies to protect the wetlands of the BSMR coastal lagoon system.

1 Introduction

Coastal wetlands are among the most productive, dynamic, and valuable ecosystems. Despite their limited geographic distribution, coastal wetlands provide a wide range of ecosystem services, including purifying water, protecting shorelines, and buffering coastal areas from storm surges, among others. Coastal wetlands also play central roles in biogeochemical cycles, productivity, the maintenance of biodiversity, and the regulation of local climatic conditions (Camacho-Valdez et al., 2013; Lebreton et al., 2019; Moomaw et al., 2018; Sun et al., 2021; Wu, Zhou, & Tian, 2017). However, coastal wetlands are under intense pressure. Since the 18th century, ~ 46–50% of wetlands have been lost worldwide, with much of these losses occurring during the 20th and early 21st centuries (Davidson, 2014). These losses result from direct and indirect anthropogenic activities, which have also come with substantial social and environmental costs (Davidson, 2014; Moomaw et al., 2018).

In tropical and subtropical coastal regions, land use/land cover change (LULCC) due to the expansion of shrimp farms and other activities associated with aquaculture production has been frequently identified as a principal driver of wetland loss since the 1950s (Ottinger et al., 2016). Since then, shrimp production has increased from 1000 Mt to more than 4 million Mt per year (Anderson et al., 2018; Ashton, 2010; FAO, 2019). However, severe environmental problems, such as water pollution, habitat and biodiversity loss, and landscape fragmentation, have accompanied this impressive increase in shrimp production (Berlanga-Robles et al., 2011a; Miranda et al., 2009; Ottinger et al., 2016; Paez-Osuna et al., 2003). In many countries like Thailand, the Philippines, and Ecuador, 50–80% of shrimp culture ponds have been built in either mangroves or associated areas (Aksornkoae & Tokrisna, 2004; Kongkeo, 1997; Terchunian et al., 1986). Consequently, the shrimp farming industry has been named as the main cause of mangrove deforestation in the world (Ashton, 2010; Primavera, 2006; Valiela et al., 2009).

In Mexico, shrimp farming began in 1985, and 70 t of shrimp were produced in that year. By 2018, the shrimp farming industry had grown to produce 158,000 t (Berlanga-Robles et al., 2011a; CONAPESCA, 2018). This growth is primarily due to the expansion of shrimp farming in northwestern Mexico, particularly in the state of Sinaloa, were at least 60% of the shrimp farming ponds in the country are currently concentrated (CESASIN, 2022; CONAPESCA, 2018). Although most of the shrimp ponds in Mexico were not built on mangrove areas, as was the case in other countries, other types of coastal wetlands have been severely affected by LULCC due to shrimp farming (Berlanga-Robles et al., 2011b).

Studies focused on aquaculture mapping and monitoring in northwestern Mexico using Remote Sensing (RS) and Geographical Information System (GIS) data have described and quantified LULCC (Alonso-Pérez et al., 2003; Berlanga-Robles & Ruiz-Luna, 2006; Berlanga-Robles et al., 2011a; 2011b), although little attention has been paid to the impacts of shrimp aquaculture on the spatial structure of coastal landscapes as a whole. Thus, modeling information and landscape metrics can be used to evaluate structural landscape characteristics with quantifiable parameters. This information can also be used to monitor the structural complexity of landscapes and to detect LULCC patterns over a variety of spatial scales that are the result of prior land use and management actions (Berlanga-Robles et al., 2011a).

Understanding the drivers behind LULCC and the resulting trends and environmental impacts is essential for the successful management of coastal resources and ecosystems, which requires specific strategies to address different risks (Bertolo et al., 2012; Lam, 2008; Lambin & Geist, 2008; Tewabe & Fentahum, 2020). As such, an urgent need exists to inventory and monitor aquaculture areas at regional and local scales to assess the potential consequences of shrimp farming on critical wetland ecosystems (Ottinger et al., 2016).

In this study, we analyzed LULCC in the Bahia Santa Maria-La Reforma (BSMR) coastal lagoon system. This system has been heavily transformed by the local shrimp aquaculture industry, which has constantly grown over the last three decades. We analyzed the impacts of this growth on the spatial structure of the BSMR coastal lagoon system using remote sensing techniques to identify existing trends and predict future LULCC scenarios based on stochastic models. Additionally, we evaluated the impacts of shrimp farming on the spatial complexity and connectivity of the BSMR coastal lagoon system through an analysis of landscape metrics.

2 Methods

2.1 Study area

The BSMR region is located in the state of Sinaloa in northwestern Mexico (24° 70'–25° 30' N, 107° 90'–108° 40' W; Fig. 1). The landscape is dominated by agriculture and a wetland mosaic composed of a lagoon system, intertidal marshes, sandy beaches, and mangrove forests. In the BSMR, the shrimp farming ponds are generally situated near natural wetlands.

The BSMR coastal lagoon system has been designated as a Ramsar Site (Num. 1340) and an Important Bird Area by BirdLife International. It is also a Priority Wetland of the North America Wetlands Conservation Act, a Site of Hemispheric Importance of the Western Hemisphere Shorebird Reserve Network, and a Mexican Priority Marine and Terrestrial Region (Enríquez-Andrade et al., 2005).

2.2 Mapping LULCC

Thematic maps of the study area were produced for 1985, 1994, 2002, and 2017 based on supervised classifications of Landsat images. The dates were selected based on the dynamics of the regional shrimp farming industry and the conservation designations assigned to the BSMR coastal lagoon system.

Although LULCC assessments using remote sensing data are commonly conducted (Lam, 2008), these analyses have been limited to a single image per year in most cases, ignoring seasonal, atmospheric, or phenological changes. Therefore, some of the inter-annual changes that have been detected could be the result of different environmental conditions rather than true LULCC (Younes et al., 2017). Thus, in this study, the thematic map series and LULCC mapping were completed by integrating the information from multiple images per year (except for 1985) into the classification process based on availability to reduce the bias produced by spectral variations associated with landscape phenology when characterizing LULCC patterns.

Thirty Landsat Thematic Mapper (TM) and 12 Landsat Operational Land Imager (OLI) images (path/row 33/42 and 33/43) recorded between January 1985 and May 2018 (Table 1) were downloaded from the EarthExplorer website to mask the limits of the study area (Fig. 1). The resulting scenes, each with six spectral bands, were grouped by sets corresponding to the years 1985, 1994, 2002, and 2017. Then, the principal components of each set were obtained via an unstandardized forward t-mode Principal Component Analysis (Eastman, 2016). The images of the first four components of each set, which explained at least 95% of the total variation, were classified into eight LULC classes (Table 2) using a segmentation method (Blaschke, 2010; Jensen, 2015). Subsequently, the thematic maps were updated with the polygons corresponding to the shrimp farm and human settlement classes, which were digitalized based on a photointerpretation of the image collection in Google Earth Pro v. 7.3.3. that was supplemented with data from the Vegetation and Land Use Map Series I, III, IV, V, and VI (scale 1:250,000) for 1997, 2001, 2005, and 2016 of the National Geographic and Statistics Institute (INEGI) and information from the Aquaculture Production Units Inventory of the National Fisheries and Aquaculture Commission of Mexico (CONAPESCA).

Table 1
Landsat image collection (path/row 33/42 and 33/43) used in this study.

Platform/Sensor	Year*	Acquisition date	Spectral bands
Landsat 5/Thematic mapper (TM)	1985	85-01-16	1 - Blue (0.45–0.52 μm)
	1994	93-09-19, 93-10-21, 94-02-10, 94-02-26, 94-10-24, 94-11-25, 95-01-12	2 - Green (0.52–0.60 μm) 3 - Red (0.63–0.69 μm) 4 - NIR (0.76–0.90 μm)
	2002	01-10-25, 02-01-31, 02-05-07, 02-06-08, 02-11-14	5 - MWIR 1 (1.55–1.75 μm) 7 - MWIR (2.08–2.35 μm)
Landsat 8/Operational Land Imager (OLI)	2017	17-02-25, 17-03-29, 17-10-23, 17-11-24, 18-04-17	2 - Blue (0.45–0.52 μm) 3 - Green (0.52–0.60 μm) 4 - Red (0.63–0.69 μm) 5 - NIR (0.76–0.90 μm) 6 - MWIR 1 (1.55–1.75 μm) 7 - MWIR (2.08–2.35 μm)
Notes: NIR (near infrared), MWIR (mid infrared)			

Table 2
Land use/Land cover (LULC) classes used for the classification of Landsat satellite images.

Code	Class	Description
LE	Lagoons and estuaries	Subtidal coastal wetlands and deep-water habitats permanently flooded with tidal and ocean waters or permanently or temporarily diluted with fresh water from rivers and runoff. Includes bays, estuaries, and coastal lagoons.
SLT	Saltmarsh	Intertidal coastal wetlands. Plains temporarily flooded by tides and occasionally by freshwater runoff. Includes unconsolidated substrate wetlands with or without emergent vegetation.
MNG	Mangroves	Intertidal coastal wetlands. Plains temporarily flooded by tides or channels and basin permanently flooded with brackish water with the presence of shrubs and trees of three mangrove species: *Laguncularia racemosa*, *Rhizophora mangle*, and *Avicennia germinans*. There are also specimens of *Conocarpus erectus.*
SB	Sandy beach	Intertidal coastal wetlands. Exposed areas with unconsolidated substrate dominated by sands flooded by tides. With or without emergent vegetation.
RB	Reed bed	Altered intertidal coastal wetland. Saltmarsh invaded by emergent vegetation, mainly by individuals of the genus *Typha*, driven by excessive freshwater inputs.
SF	Shrimp farming	Artificial wetlands. Ponds dedicated to shrimp farming, generally with the semi-intensive method.
DP	Dams and ponds	Artificial wetlands. Freshwater storage dams and ponds.
TV	Terrestrial vegetation	Dry forest, thorn forest, open scrub forest, and xerophilous scrub covers
AGR	Agriculture	Land dedicated to agriculture, mainly irrigation. Induced grasslands and grazing areas for livestock are also included.
HS	Human settlements	Urban, suburban, and rural centers: Cities and towns.
Sources: Berlanga-Robles, Ruiz-Luna & de la Lanza (2008); Berlanga-Robles et al. (2011a).		

The accuracy of the thematic maps was assessed with error matrices produced with 2436–2630 reference points that were randomly selected from the Vegetation and Land Use Map Series I, III, IV, V, and VI and verified by photointerpretation of the image collection available in Google Earth and field data collected with a GPS receiver (4-m precision). Reference point selection and verification were determined independently of the classification process. Then, the producer and user accuracies, global accuracy, and Kappa concordance coefficient (K’) were estimated (Congalton & Green, 2008). Image processing and accuracy assessments were conducted with TerrSet v. 18.3 (Eastman, 2016).

2.3 LULCC and future projections

LULCC were detected via a post-classification comparison analysis (Mas et al., 2009; Eastman, 2016) of the following pairs of years: 1985–1994, 1994–2002, 2002–2017, and 1985–2017. For each comparison, the overall change proportion and K’ were estimated (Berlanga-Robles & Ruiz-Luna, 2007; Eastman, 2016). Lastly, the average annual rate of change of each LULC class was estimated (SEMARNAT, 2015) with Eq. (1):

$$r = \left[\left(\frac{S_2}{S_1} \right)^{\frac{1}{t}} \times 100 \right] - 100 \text{, Eq. (1)}$$

where r is the annual average change rate; S_1 and S_2 are the surfaces for the start and end times, respectively; and t is the time interval.

The LULC patterns were projected to 2027 and 2037 based on the changes detected between 1985 and 2017 with the tools available in the Land Change Modeler module of TerrSet v. 18.3. This was accomplished in a three-step process that included detecting LULCC (the post-classification comparison analysis described at the beginning of section 2.3), modelling the transition potential, and predicting LULCC (Eastman, 2016). The second step was carried out with the multi-layer perceptron method to create potential transition maps (Bagaria et al., 2021; Eastman, 2016; Pereira et al., 2020). In this study, six drivers of change were used: 1) closeness to roads and highways, 2) closeness to human settlements, 3) closeness to agricultural fields, 4) distance to the littoral shore, 5) elevation (Digital Elevation Model), and, 6) slope. The Digital Elevation Model data (15-m resolution) were downloaded from the Continuo de Elevaciones Mexicano (CEM) website (INEGI, 2013) and geometrically corrected to the spatial parameters of the LULC thematic maps.

In the third step, the extent of future LULCC (2027 and 2037) was modeled with a Markov Chain (MC) empirically adjusted from the change trends summarized in the 1985–2017 change detection matrix (Berlanga-Robles & Ruiz Luna, 2011; Brown et al., 2004). Likewise, LULC thematic maps for 2027 and 2037, with the same categories as those included in the classified maps, were produced with a hard prediction model, which was based on a competitive land allocation model, similar to the multi-objective decision making process (Eastman, 2016).

2.4 Landscape metrics

The impacts of shrimp farming on the spatial structure of the coastal wetlands were assessed via landscape metrics. For this analysis, the landscape was limited to the potential wetland area (Fig. 1), defined by INEGI (2014) as sites which possess characteristics suitable for hosting wetlands in the absence of human intervention. The corresponding layer (1:250,000 scale) was downloaded from https://www.inegi.org.mx/temas/humedales/ #Descargas.

With Fragstat v. 4.2 (McGarical & Ene, 2013), eight metrics related to wetland complexity and connectivity were estimated at class and landscape levels (Table 3). A 4-cell neighborhood rule and a minimum patch size of 0.36 ha (4 pixels) and a mosaic landscape model were employed (Berlanga et al., 2011b). The metrics were calculated for the lagoon and estuary, saltmarsh, mangrove, sandy beach, and reed bed classes. The shrimp farm ponds and the linear infrastructure needed for their operation (e.g., canals and roads) were reclassified as internal background (a matrix containing the wetland patches was also included when calculating the total area of the landscape). The remaining LULC classes and areas outside the potential wetland area (Fig. 1) were reclassified as external background, with no data values (Berlanga-Robles, 2011b).

Table 3
Landscape metrics used to assess the impacts of shrimp farming on the spatial structure of coastal wetlands of the Bahia Santa Maria-La Reforma (BSMLR) coastal lagoon system.

Metric	Description
Landscape Area (A)	A equals the sum of the areas (m^2) of all patches of all classes and the area (m^2) of internal background, divided by 10,000 (to convert to hectares); that is, total landscape area
Percentage of Landscape (PLAND)	"PLAND equals the sum of the areas (m2) of all patches of the corresponding patch type, divided by total landscape area (m2), multiplied by 100 (to convert to a percentage); in other words, PLAND equals the percentage of the landscape comprised of the corresponding patch type"
Number of Patches (NP)	"NP equals the number of patches of the corresponding patch type (class)"
Large Patch Index (LPI)	"LPI equals the area (m2) of the largest patch of the corresponding patch type divided by total landscape area (m2), multiplied by 100 (to convert to a percentage); in other words, LPI equals the percentage of the landscape comprised by the largest patch"
Mean Patch Size (AREA_MN)	AREA_MN is the mean area of the patches in a class
Perimeter-Area Fractal Dimension (PAFRAC)	PAFRAC indicates the complexity of the patch shapes of a class and "approaches 1 for shapes with very simple perimeters such as squares, and approaches 2 for shapes with highly convoluted, plane-filling perimeters"
Landscape Division Index (DIVISION)	"DIVISION is based on the cumulative patch area distribution and is interpreted as the probability that two randomly chosen pixels in the landscape are not situated in the same patch of the corresponding patch type". It takes values between 0 and 1, "0 when the landscape consists of single patch and approaches 1 when the focal patch type consists of single, small patch one cell in the area"
Effective Mesh Size (MESH)	"MESH equals the sum of patch area squared, summed across all patches of the corresponding patch type, divided by the total landscape area". "Mesh is based on the cumulative patch area distribution and is interpreted as the size of the patches when the corresponding patch type is subdivided into S patches, where S is the value of the splitting index"
Source: McGarigal & Ene (2013).	

The road and channel networks for the 1994 LULC were edited from the vector data set of the topographic charts (scale 1: 250,000) published by INEGI in 1993. In addition, the networks for 2002 and 2017 were edited from the digital versions of the topographic charts (scale 1:50,000) published by INEGI in 2000 and 2015, respectively. The 2015 topographic chart was also used as a proxy for 2027 and 2037.

The monotonic trend of the metrics throughout the six years analyzed was evaluated with the Mann-Kendall test (Wagner et al., 2013) with the 'Kendall' package in R (McLeod, 2015).

3 Results

Four LULC thematic maps (1985, 1994, 2002, and 2017) and two future projected maps for 2027 and 2037 (Fig. 2) were produced, with overall accuracies for the LULC maps above 85% and Kappa coefficient (K’) values of 0.81–0.83 that reflected strong agreement between the reference and classified data (Congalton & Green, 2008). With the exception of the reed bed class, the other coastal wetland types were classified with producer and user accuracies above 79% and 70%, respectively (Table 4).

Table 4
Summary of the accuracy assessment of the Land use/Land cover (LULC) thematic maps of the Bahia Santa Maria-La Reforma (BSMLR) coastal lagoon system produced by the classification of Landsat images.

Landscape level								
N	1673		1908		1816		1746	
OA (%)	88		87		88		87	
K'	0.82		0.81		0.82		0.83	
Class level								
	1985		1994		2002		2017	
Class	PA(%)	UA(%)	PA(%)	UA(%)	PA(%)	UA(%)	PA(%)	UA(%)
Lagoons and estuaries	95	97	92	97	93	97	95	98
Saltmarshes	90	68	86	71	88	70	79	71
Mangroves	84	85	80	83	84	86	80	77
Sandy beaches	81	87	83	71	81	94	92	80
Reed beds	64	82	20	29	54	78	22	40
Shrimp farming	NA	NA	100	100	100	100	100	100
Dams and ponds	87	76	90	90	86	92	57	100
Terrestrial vegetation	80	78	75	75	81	74	89	75
Agriculture	91	94	91	93	90	95	88	95
Human settlements	100	100	83	83	88	88	82	84
Notes: N (number of test points), OA (overall accuracy), K’ (Kappa coefficient estimator), PA (producer accuracy), UA (user accuracy), NA (Not available).								

For all years, the agriculture class dominated the landscape and covered an average of 56% of the total study area. The natural wetland classes with the greatest average extensions were lagoons and estuaries (11%), saltmarshes (6%), and mangroves (4%), while the coverages of sandy beaches and reed beds were less than 1% each. Shrimp farming, which was not present in 1985, covered 0.6% of the study area in 1994. This value increased to almost 4% in 2017 and was projected to reach 5% by 2037 (Fig. 3).

Seven change detection matrices were produced, four for past LULCC and three for future projected changes (Supplementary Material). The values of K’ ranged from 0.82 to 0.99 and were associated with change percentages of 12% and 0.4%, respectively. From 1985 to 2017, the K’ value was 0.70, indicating changes in about 19% of the study area, whereas the predicted change from 2017 to 2037 was marginal (K’ = 0.97). From 1985 to 2017, the changes were principally associated with the negative average annual change rates recorded for the saltmarsh and terrestrial vegetation classes and the positive rates of the shrimp farming and agriculture classes (Table 5).

Table 5
Land use/Land cover change (LULCC) indicators in the Bahia Santa Maria-La Reforma (BSMLR) coastal lagoon system.

Start year	**1985**	**1994**	**2002**	**2017**	**2027**	**1985**	**2017**
End year	1994	2002	2017	2027	2037	2017	2037
Range (years)	9	8	15	10	10	32	20
Landscape level							
K'	0.82	0.86	0.82	0.97	0.99	0.70	0.97
CP (%)	11.8	8.6	11.6	1.7	0.4	19.4	2.1
Class level							
Class	r (%)						
Lagoons and estuaries	0.1	0.2	-0.1	0.0	0.0	0.0	0.0
Saltmarshes	-1.4	-1.7	-3.7	-4.3	-1.3	-2.5	-1.1
Mangroves	0.7	-0.9	0.9	0.0	0.0	0.4	0.0
Sandy beaches	0.1	-0.5	-0.4	0.0	0.0	-0.3	0.0
Reed beds	-0.1	0.7	-0.6	-1.7	-1.0	-0.2	-0.5
Shrimp farming	NA	15.9	4.0	3.3	0.3	36.4*	0.7
Dams and ponds	-8.8	-2.7	-1.2	0.0	0.0	-3.8	0.0
Terrestrial vegetation	-3.3	0.0	0.3	0.0	0.0	-0.8	0.0
Agriculture	1.3	0.0	0.0	0.1	0.0	0.4	0.0
Human settlements	0.5	0.0	3.7	0.0	0.0	1.9	0.0
Notes: K’ (Kappa coefficient estimator), CP (Change percentage), r (annual average change rate), NA (Not available), *estimated from 1994 to 2017.							

In all the analyzed periods, saltmarshes showed negative rates of change (Table 5), indicating that the loss of this wetland class will continue in the medium and long term. From 1985 to 2017, saltmarshes lost 50% of their surface. The main drivers of the reduction in saltmarshes were shrimp farming and agriculture, which were responsible for 57% and 31% of this loss, respectively. A similar pattern of change was predicted for the period of 2017–2037, and a 43% loss of these wetlands was projected over the next 16 years. In addition, from 1985 to 2017, ~ 28,000 ha of terrestrial vegetation was lost due to agriculture (Fig. 4).

The highest average annual growth for shrimp farming was observed between 1994 and 2002. The average annual growth from 2017 to 2037 was of 0.7% (Table 5). Shrimp farming has been and will continue to be one of the main drivers of saltmarsh loss. Approximately 94% of the area occupied by shrimp farming in 2017 was reclaimed from saltmarsh. Likewise, 97% of the new land estimated for shrimp farming in 2037 (8749 ha) could occupy land classified as saltmarsh in the 2017 classification, with the remaining 3% occupying reed beds.

The potential wetland polygon measured 154,337 ha, although 11.6% of this area was later reclassified as external background; therefore, the initial (wetland) landscape area (A) was 136,428 ha. A monotonic decreasing trend ($p = 0.009$) was observed that was mainly due to the agriculture frontier invading the wetland zone. On the other hand, the number of patches in the wetland landscape did not show any significant trend ($p = 0.452$). The Large Patch Index registered values between 43% and 46% at the landscape level, indicating that almost half of the landscape area was concentrated in a single patch (Table 6). At the landscape level, the mean patch size decreased between 1985 and 2017 but increased in 2027 and 2037, dispelling any possible trend ($p = 0.452$).

Table 6
Landscape metrics calculated at landscape class and level to assess the impacts of the shrimp farming on the wetland landscape of the Bahia Santa Maria-La Reforma (BSMLR) coastal lagoon system.

	Year						
Landscape							
Metric	1985	1994	2002	2017	2027	2037	Mann-Kendall*
A	136428	135350	134807	130361	128845	127560	tau = -1.000 (p = 0.00853)
NP	1028	1084	1276	1189	968	904	tau = -0.333 (p = 0.45237)
LPI	42.7	43.4	44.2	45.1	45.7	46.1	tau = 1.000 (p = 0.00853)
AREA_MN	132.7	122.3	97.5	93.8	105.7	111.3	tau = -0.333 (p = 0.45237)
PAFRAC	1.36	1.36	1.39	1.37	1.37	1.35	tau = -0.200 (p = 0.70711)
DIVISION	0.785	0.790	0.789	0.790	0.787	0.783	tau = -0.333 (p = 0.45237)
MESH	29390	28464	28437	27432	27440	27727	tau = -0.600 (p = 0.13285)
Lagoons and estuaries							
PLAND	43.1	43.7	44.5	45.5	46.0	46.5	tau = 1.000 (p = 0.00853)
NP	17	19	18	23	23	23	tau = 0.745 (p = 0.06997)
LPI	42.7	43.4	44.2	45.1	45.7	46.1	tau = 1.000 (p = 0.00853)
AREA_MN	3456.2	3110.9	3333.0	2578.7	2578.7	2578.8	tau = -0.552 (p = 0.18060)
PAFRAC	1.48	1.44	1.50	1.43	1.43	1.43	tau = -0.690 (p = 0.08517)
DIVISION	0.818	0.812	0.805	0.796	0.791	0.787	tau = -1.000 (p = 0.00853)
MESH	24840	25487	26358	26567	26880	27152	tau = 1.000 (p = 0.00853)
Saltmarshes							
PLAND	38.0	34.3	28.9	18.8	12.1	10.9	tau = -1.000 (p = 0.00853)

	Year						
NP	531	587	704	753	552	497	tau = -0.067 (p = 1.00000)
LPI	12.8	9.8	9.5	4.8	1.4	1.5	tau = -0.867 (p = 0.02417)
AREA_MN	97.6	79.0	55.4	32.6	28.3	28.1	tau = -1.000 (p = 0.00853)
PAFRAC	1.36	1.37	1.40	1.38	1.36	1.33	tau = -0.333 (p = 0.45237)
DIVISION	0.969	0.981	0.987	0.997	0.999	0.999	tau = 0.966 (p = 0.01290)
MESH	4232	2585	1759	385	74	83	tau = -0.867 (p = 0.02417)
Mangroves							
PLAND	14.2	15.2	14.2	16.6	16.8	17.0	tau = 0.828 (p = 0.03538)
NP	313	322	378	352	352	349	tau = 0.447 (p = 0.31406)
LPI	3.2	3.3	2.9	3.5	3.5	3.5	tau = 0.645 (p = 0.11900)
AREA_MN	61.7	63.8	50.5	61.6	61.6	62.3	tau = 0.733 (p = 0.06029)
PAFRAC	1.32	1.32	1.35	1.36	1.36	1.35	tau = -0.298 (p = 0.54582)
DIVISION	0.998	0.998	0.998	0.997	0.997	0.997	tau = 0.596 (p = 0.15871)
MESH	240	303	237	397	402	408	tau = 0.894 (p = 0.02677)
Sandy Beaches							
PLAND	3.6	3.6	3.5	3.4	3.4	3.4	tau = -0.856 (p = 0.03982)
NP	41	27	30	28	28	28	tau = -0.298 (p = 0.54582)
LPI	2.3	2.5	2.4	2.4	2.4	2.4	tau = -0.577 (p = 0.24157)
AREA_MN	118.3	180.7	155.9	156.7	156.7	156.8	tau = 0.276 (p = 0.56609)

	Year						
PAFRAC	1.48	1.49	1.47	1.53	1.53	1.53	tau = 0.298 (p = 0.54582)
DIVISION	0.999	0.999	0.999	0.999	0.999	0.999	tau = -0.149 (p = 0.84043)
MESH	77	86	82	77	78	79	tau = 0.745 (p = 0.06997)
Reed beds							
PLAND	1.2	1.3	1.2	1.2	1.0	1.0	tau = -0.745 (p = 0.06997)
NP	126	129	146	33	13	7	tau = -0.600 (p = 0.13285)
LPI	0.3	0.3	0.2	0.5	0.5	0.5	tau = -0.645 (p = 0.11900)
AREA_MN	13.5	13.2	11.1	48.3	102.9	174.8	tau = 0.467 (p = 0.25966)
PAFRAC	1.34	1.28	1.37	1.37	1.35	NA	tau = 0.600 (p = 0.13285)
DIVISION	1	1	1	1	1	1	tau = 0.733 (p = 0.06029)
MESH	2	2	1	6	6	6	tau = 0.600 (p = 0.22067)
Notes: * Tau statistic with two-tailed p-value. A (Landscape Area), PLAND (Percentage of Landscape), NP (Number of Patches), LPI (Large Patch Index), AREA_MN (Mean Patch Size), PAFRAC (Perimeter-Area Fractal Dimension), DIVISION (Landscape Division Index), MESH (Effective Mesh Size).							

In agreement with the results of the change detection analysis, the impacts of shrimp farming on the spatial structure of coastal wetlands were more evident in the saltmarsh class. In addition to a loss of area, patch size was also reduced, increasing the fragmentation of this wetland type. In 1985, saltmarshes covered 38% of the wetland landscape, and the largest saltmarsh patch covered almost 13% of the area. However, by 2037, it is likely that saltmarshes will only cover 11% of the wetland landscape, with the largest patch covering only 1.5% of the area.

As a result of the reduction in saltmarsh area, the representativeness of the wetland classes of lagoons and estuaries and mangroves increased, as their areas remained relatively constant over time. In addition, the spatial metrics of DIVISION (Landscape Division Index) and MESH (Effective Mesh Size), which were used to assess landscape connectivity, showed significant monotonic positive and negative trends, respectively, as they are negatively correlated (McGarigal & Enem, 2013), indicating an increase in the fragmentation of the wetland landscape, particularly for saltmarshes (Table 6).

4 Discussion

The landscape of the BSMR is dominated by agriculture practices but maintains a notable proportion of coastal wetlands. The current spatial heterogeneity patterns of the landscape are due to natural and anthropogenic drivers operating at different spatiotemporal scales (Bertolo et al., 2012; Roy & Tomar, 2001). In the case of the BSMR, the coast is of primary origin and rapid sediment accumulation during its formation favored the creation of a broad coastal plain in which Vertisols predominated. Since the 1930s, the agrarian reforms of the Mexican government stimulated agriculture development with the creation of hydraulic infrastructure (Berlanga-Robles & Ruiz-Luna, 2011). As a result, agriculture in the BSMR is currently the prevailing land use, and this trend will probably continue for the next 20 years, even though the extent of agricultural land has remained relatively constant since the mid-1990s.

As detected in this and other studies in the region (Gurrola, 2000; García, 2005), the expansion of agriculture has come at the expense of natural terrestrial vegetation (e.g., dry forests and xerophytic shrubs) and saltmarshes. In addition, notable changes in natural covers are due to shrimp farming, with facilities being built primarily on saltmarshes with low mangrove deforestation rates.

Saltmarshes offer advantages when constructing shrimp ponds, such as flatness (slope < 5°), soils with fine textures and low infiltration rates (e.g., Solonchak and Vertisols), a minimal presence of trees, and the proximity to lagoons and estuaries that simplifies water supply and drainage operations. Moreover, the legislation to protect and regulate the use of saltmarshes in Mexico is ambiguous and only specifies some restrictions when saltmarshes are near mangroves, which further facilitates the creation of shrimp farms in these habitats (Berlanga-Robles et al., 2011a; 201b).

The same geological process that allowed the coastal plain to advance in central and northernSinaloa (including BSMR) also allowed for large areas of intertidal wetlands to form as saltmarshes. In Sinaloa, there are between 115,300–159,400 ha of saltmarshes (Berlanga-Robles & Ruiz-Luna, 2011; Hernández-Guzmán et al., 2021), representing 42–63% of the total saltmarsh cover in Mexico (272,527 ha; Mcowen et al. 2017). As such, the saltmarsh cover of the BSMR coastal lagoon system represents around 10% of that reported for all Mexico in 2017. However, by 2037, the BSMR saltmarsh cover could be reduced to 6%.

To date, most of this reduction has been due to shrimp farming growth, which was responsible for the loss of around 19,000 ha from 1985 to 2017. Shrimp farming is also expected to result in additional losses in cover of ~ 8500 ha by 2037. Moreover, this activity has substantially impacted the connectivity of the entire BSMR wetland complex, which is probably a consequence of the linear infrastructure (e.g., roads and channels) required for shrimp farming rather than of the ponds themselves (Berlanga-Robles et al. 2011b).

These changes also alter the capacity of wetlands to deliver ecosystem services and highlight the interdependence of humans and nature and the benefits that ecosystems provide to societies

(Lebreton et al., 2019; Zorrilla-Miras et al., 2014). The monetary value of ecosystems has been increasingly used to support sustainable socio-ecosystem management policies, but for saltmarshes, these assessments may underestimate their value due to a lack of understanding of their ecological functions and the ecosystem services that they offer.

In addition, identifying the interdependence between humans and nature is complex in ecosystems that are highly connected to other nearby ecosystems (e.g., saltmarshes). Likewise, valuations of ecosystem services require the segmentation of ecosystem components, and thus any services associated with the emergent properties of a coastal ecosystem complex cannot be assessed (Lebreton et al., 2019). Despite these limitations, a negative balance can be identified when saltmarshes (natural wetlands) are converted to shrimp farms (artificial wetlands).

In 2007, shrimp farming in Mexico produced 111,787 t from ~ 72,500 ha of shrimp ponds (Berlanga-Robles et al., 2011a), which was valued at 421,206,200 (2007 USD; FAO, 2022). This amounts to 5,810 (2007 USD)/ha/year, which is 39% less than the value estimated for saltmarsh ecosystem services by Camacho-Valdez et al. (2014) of 9,544 (2007 USD) per ha/year. Therefore, from the start of shrimp farming in the BSMR region until 2017, a loss of 70,946,000 (2007 USD) due to the conversion of saltmarshes to shrimp farms has accrued. From 2017 to 2037, this loss is expected to grow to 31,739,000 (2007 USD). These losses are based on the direct elimination of saltmarshes and do not consider either other indirect effects due to their disappearance, such as fragmentation, or emergent effects that depend on the entire coastal system.

Activities that increase the provisioning of services of high economic value at the expense of regulating services, such as saltmarsh-to-shrimp farm conversions, are major drivers of LULCC worldwide (Zorrila-Miras et al., 2014). Moreover, while many saltmarsh ecosystem services come in the form of collective goods, the economic benefits of shrimp farming are mainly limited to farm owners, with narrow benefits spreading to local communities in the form of taxes or temporary jobs.

According to the official statistics for the municipality of Angostura, which includes most of the BSMR wetland system, the social impacts of shrimp farming are difficult to assess, especially considering recent methodological changes to measure poverty and well-being in Mexico. Even so, the number of people in poverty in Angostura decreased from 48% in 2010 to 26% in 2020, although this reduction was accompanied by a decrease in population size of 14.5% from 52,438 to 44,814 inhabitants (CONEVAL, 2022). Moreover, a notable rise in the number of people employed in the fishing and aquaculture industry can be observed from 1998 (2,124) to 2003 (3,596), although this number fell once again in 2008 (2,156; INEGI, 2015). At the national level, around 19,000 workers were employed by 955 shrimp farming operations in 2018 (INEGI, 2021).

The frequency and duration of tidal flooding, which primarily define the structure and ecological functions of saltmarshes, are also affected by changes in saltmarsh cover (Meijer et al., 2021; Roman & Burdick, 2012; Schutte et al., 2019). When saltmarshes are restricted, their biophysical processes are affected, resulting in disturbances to vegetation patterns, fish and avian communities, biogeochemical cycling, the links among physical and biological factors, and the marine-coastal-terrestrial environmental gradient (Meijer et al., 2021; Roman & Burdick, 2012; Schutte et al., 2019). The interruption of tidal flows in saltmarshes threatens various biological and biogeochemical processes, such as the vertical and horizontal migrations of benthic microalgae, and benthic-pelagic coupling that supports high primary productivity and sustains food webs (Kuwae et al., 2021; Lebreton et al., 2019). Coastal wetlands, such as saltmarshes and mangroves, may also be responsible for ~ 50% of the carbon that is sequestered each year in the oceans (Adam, 2019). However, changes in the natural hydrology of wetlands due to fragmentation alter the balance between methane emissions and CO_2 uptake, reducing the efficiency of carbon sequestration (Mitsch, 2016). In addition, the direct impacts of fragmentation in saltmarshes may lead to the release of carbon into the atmosphere that had been previously sequestered for hundreds of years (Adam, 2019).

The saltmarshes and other coastal wetlands in BSMR host critical wintering habitats for migratory waterbirds and shorebirds, supporting nearly one-third of all wintering shorebirds in the North American region of the Pacific Flyway (Engilis et al., 1998; Kramer & Migoya, 1989; Morrison et al., 2009). Consequently, these coastal wetlands host notable portions of the global populations of several shorebird species.

Given their importance, these coastal wetlands have been designated as priority sites for conservation in several national and international bird conservation initiatives (Enríquez-Andrade et al., 2005). Some of these distinctions offer various levels of legal protection in national and international frameworks (Boere & Piersma, 2012). However, the degradation of coastal wetlands affects habitat availability for migratory shorebird species. As a result, shorebird populations are declining due to the limited amount of available space and the loss of habitat quality in coastal non-breeding areas (Piersma et al., 2017; Rosenberg et al., 2019), which also impacts the food chain and associated biogeochemical cycles.

Besides shrimp farming, which is clearly one of the main drivers of landscape fragmentation and the loss of saltmarshes in the BSMR, other threats to wetlands in the study area are present and include sewage discharge, which increases nutrient concentrations and the eutrophication of the lagoon system (Gurrola et al., 2016). This threat is particularly severe in the southern region of the BSMR coastal lagoon system due to the discharge of polluted water from industrial, agricultural, livestock, and urban sources in the neighboring municipalities (Leyva et al., 2020). In semi-intensive shrimp farming systems, such as those of the BSMR coastal lagoon system, the discharge of nitrogen and phosphorus per harvest cycle has been estimated to be 45 and 12.9 kg/ha, respectively (Páez-Osuna, 2001). Thus, according to this and the extent of the shrimp farming estimated in this study, nutrient discharge from this activity increased from 0 t in 1985 to 1834 t (N) and 526 t (P) in 2017. Moreover, these values may increase to 2621 t (N) and 751 t (P) by 2037, considering an average of two shrimp production cycles each year. Moreover, shrimp farming contributes 10.2% and 3.3% of the nitrogen and phosphorus that are annually discharged into the Gulf of California, while agriculture contributes 67.2% and 62.5%, respectively (Miranda et al., 2009).

This study highlights the importance of coastal wetlands, particularly that of saltmarshes, in the BSMR coastal lagoon system. At a regional level, this study also highlights how these systems have been impacted by LULCC, especially those due to agriculture and shrimp farming. Thus, instruments are needed to regulate the territorial expansion of these activities while current management efforts should be strengthened (Mcowen et al., 2017). Economic activities that impact the natural systems and biodiversity of the BSMR coastal lagoon system have benefited local communities; however, shrimp production has been mainly designed to satisfy external markets, leading to the overexploitation of natural resources. In particular, shrimp farming has affected the biodiversity in the region, and the resulting land use changes have caused notable wetland losses, particularly those of saltmarshes.

As a result of the associated problems in the BSMR coastal lagoon system, stakeholders, agencies, and non-governmental civil organizations have worked to mitigate the impacts of shrimp farming through restoration, conservation, and environmental education efforts. Currently, governance has strengthened through the formation of committees to design and implement strategies that mitigate the environmental and social impacts caused by the different economic activities conducted in wetlands. In addition, participatory action schemes in communities are available, although it is limited to the islands of the system.

At the international level, programs have been established to promote the active involvement of local people in environmental protection efforts (Artigas et al., 2014), but it is still necessary to foster interactions among stakeholders to encourage engagement in participatory processes whose outcomes must be monitored and supported with legal backing. Current and future management efforts in the BSMR coastal lagoon system and similar regions should also consider adopting integrated coastal management programs that rely on dynamic decision-making to ensure the sustainable use, development, and protection of coastal and marine areas and their resources (Cisin-Sain & Kenecht, 1998).

Given the changes in the coastal landscape of the BSMR region, restoration strategies that ensure the coexistence of natural and artificial wetlands are needed. While a recovery of the saltmarshes is unlikely, coastal wetland conservation and land-use planning in the BSMR region may prevent further LULCC in this important wetland system. Likewise, interventions are required that restore hydrological flows and that manage tidal activity and freshwater runoff to safeguard the ecological functioning of the remaining saltmarshes and extended wetland network and their ecosystem services. According to Berlanga et al. (2011b), these interventions should be made over the linear infrastructure (e.g., roads and canals) rather than the shrimp farms themselves. In addition, the regulation of economic activities at the regional level is necessary, as the BSMR coastal wetlands are located in the lower portion of an anthropized basin with more than 60% of its surface affected by LULCC (García, 2005). Management proposals must also consider pollution and wastewater from productive activities and human settlements. In doing so, nutrient loads and eutrophication may be reduced.

5 Conclusions

Four LULC thematic maps (1985, 1994, 2002, and 2017) with overall accuracy values > 80% and two projected maps (2027 and 2037) were produced. Agriculture was the dominant LULC class in the BSMR coastal lagoon system, although saltmarshes appear as the most critical wetland type in the area. Shrimp farming, which was nonexistent in 1985, represented 4% of the total landscape in 2017. By 2037, this value is expected to increase to 5%. Overall, saltmarshes showed negative trends due to the expansion of shrimp farming. Given the importance of wetlands and their ecosystem services, this study highlights LULCC due to economic activities and the need to improve management strategies to protect the wetlands of the BSMR coastal lagoon system. To this end, new information relating to wetlands and the services they deliver is needed to fill knowledge gaps while supporting environmental education programs aimed at both users and decision-makers. An economic valuation of the BSMR wetlands is also recommended, as this will help to define the relationships between people and the environment while highlighting the costs of losing these important ecosystems (TEEB-Foundation 2010).

Given that ecosystem services depend on ecosystem integrity, increases in the pressures and stresses that wetlands face can affect the well-being of local populations. Hence, it is essential to further our understanding of wetlands and the services they deliver, but as discussed by Cervantes-Escobar & Camacho-Valdez (2021), this is not an easy task, as users are able to more easily identify ecosystem services related to foods, materials, medicines, and the like over those that may be classified as regulatory, cultural, or supportive. Therefore, surveys, such as the one proposed by the aforementioned authors, are needed to strengthen the commitments of local peoples to care for and protect ecosystems in order to continue enjoying the benefits that they provide.

Declarations

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Declaration of Interest

The authors declare no conflict of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript; or in the decision to publish the results.

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Figures

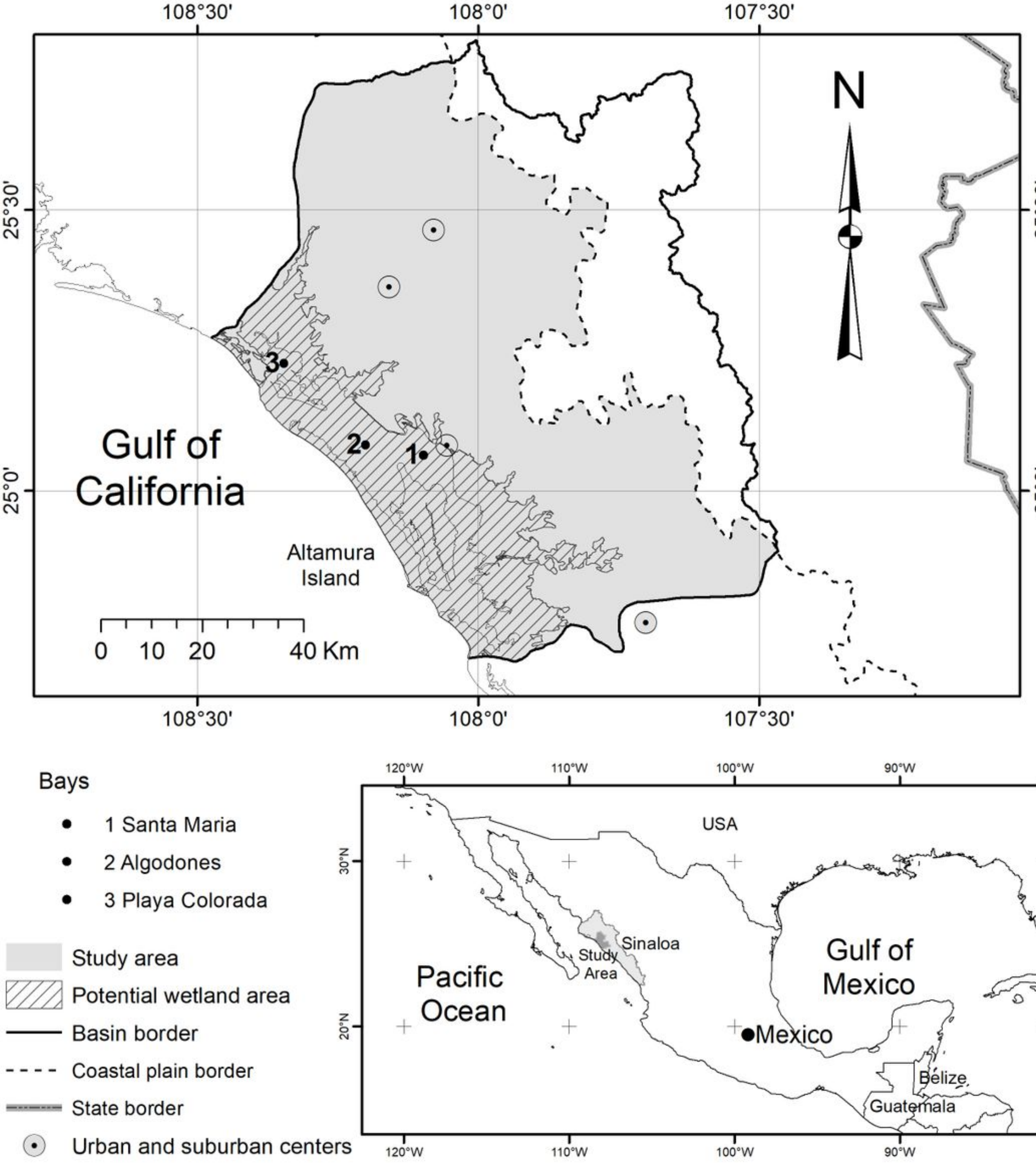

Figure 1

Study area. The Bahia Santa Maria-La Reforma (BSMLR) coastal lagoon system located in northwestern Mexico in the state of Sinaloa.

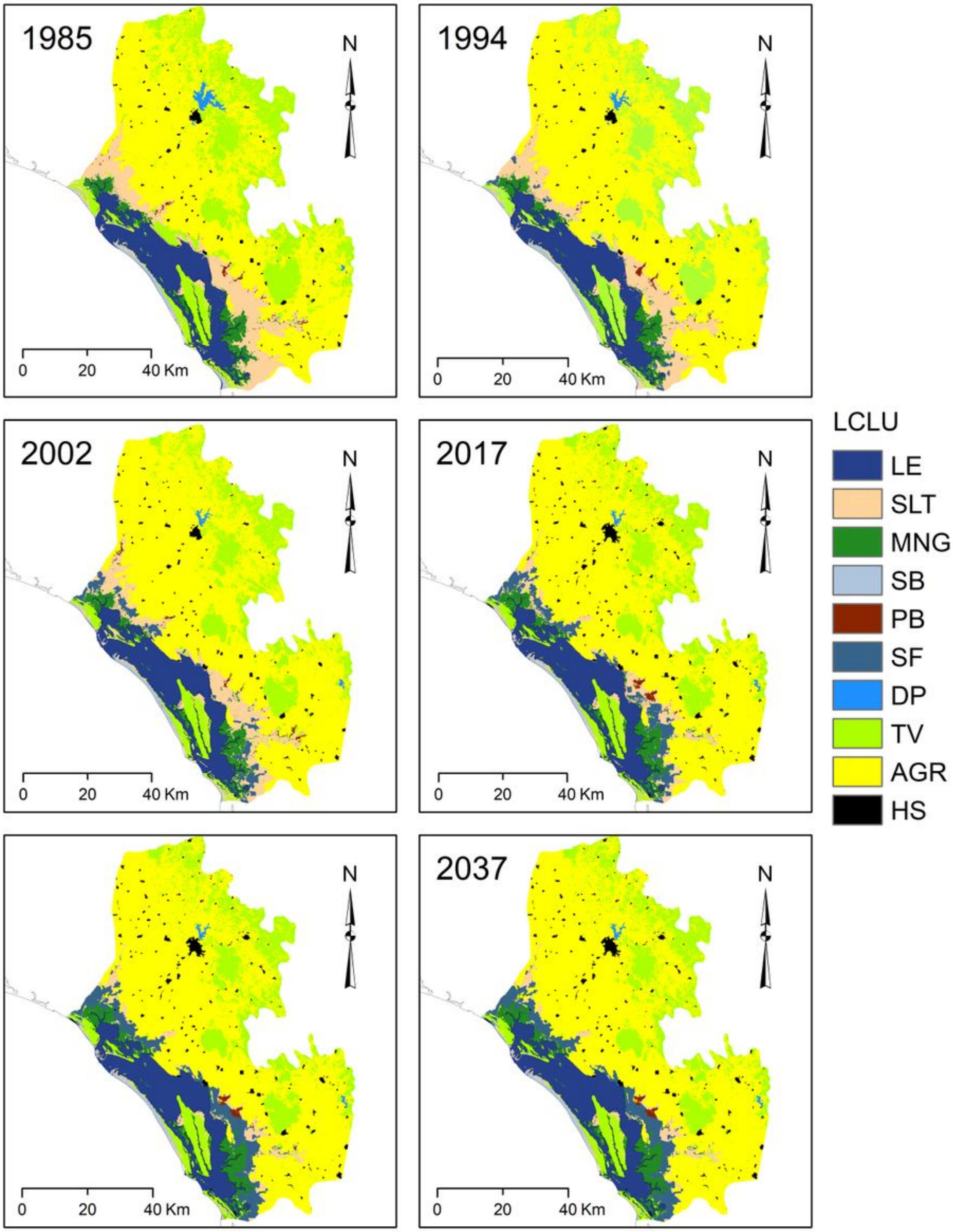

Figure 2

Land use/Land cover (LULC) thematic maps of the Bahia Santa Maria-La Reforma (BSMLR) coastal lagoon system. The maps from 1985 to 2017 were produced by the classification of Landsat images, and the maps of 2027 and 2037 were predicted with a three-step process. LE (lagoons and estuaries), SLT (saltmarshes), MNG (mangroves), SB (sandy beaches), RB (reed beds), SF (shrimp farming), DP (dams and ponds), TV (terrestrial vegetation), AGR (agriculture), HS (human settlements).

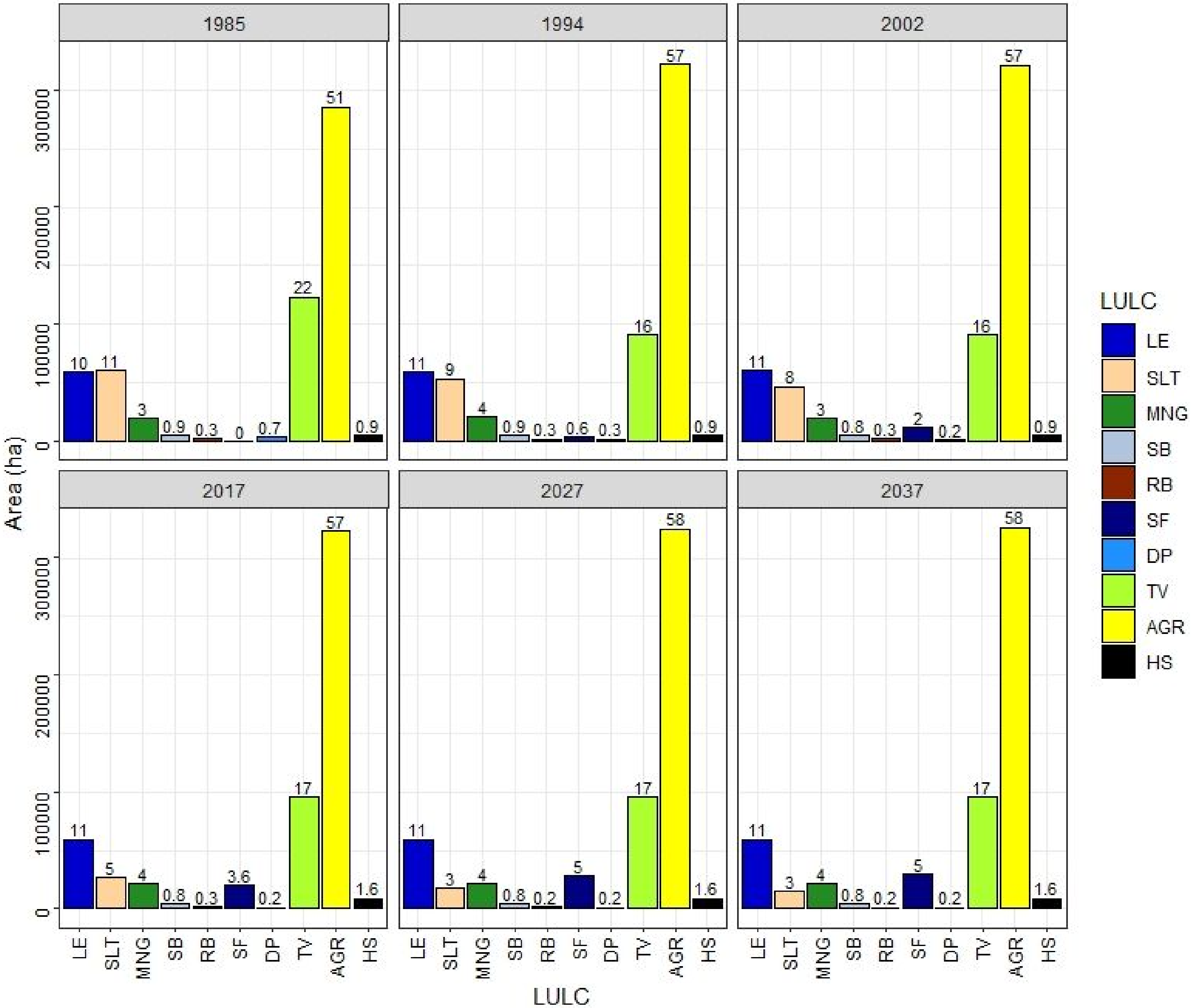

Figure 3

Land use/Land cover (LULC) distribution in the Bahia Santa Maria-La Reforma (BSMLR) coastal lagoon system. The number on the bars indicates the area percentage. LE (lagoons and estuaries), SLT (saltmarshes), MNG (mangroves), SB (sandy beaches), RB (reed beds), SF (shrimp farming), DP (dams and ponds), TV (terrestria vegetation), AGR (agriculture), HS (human settlements).

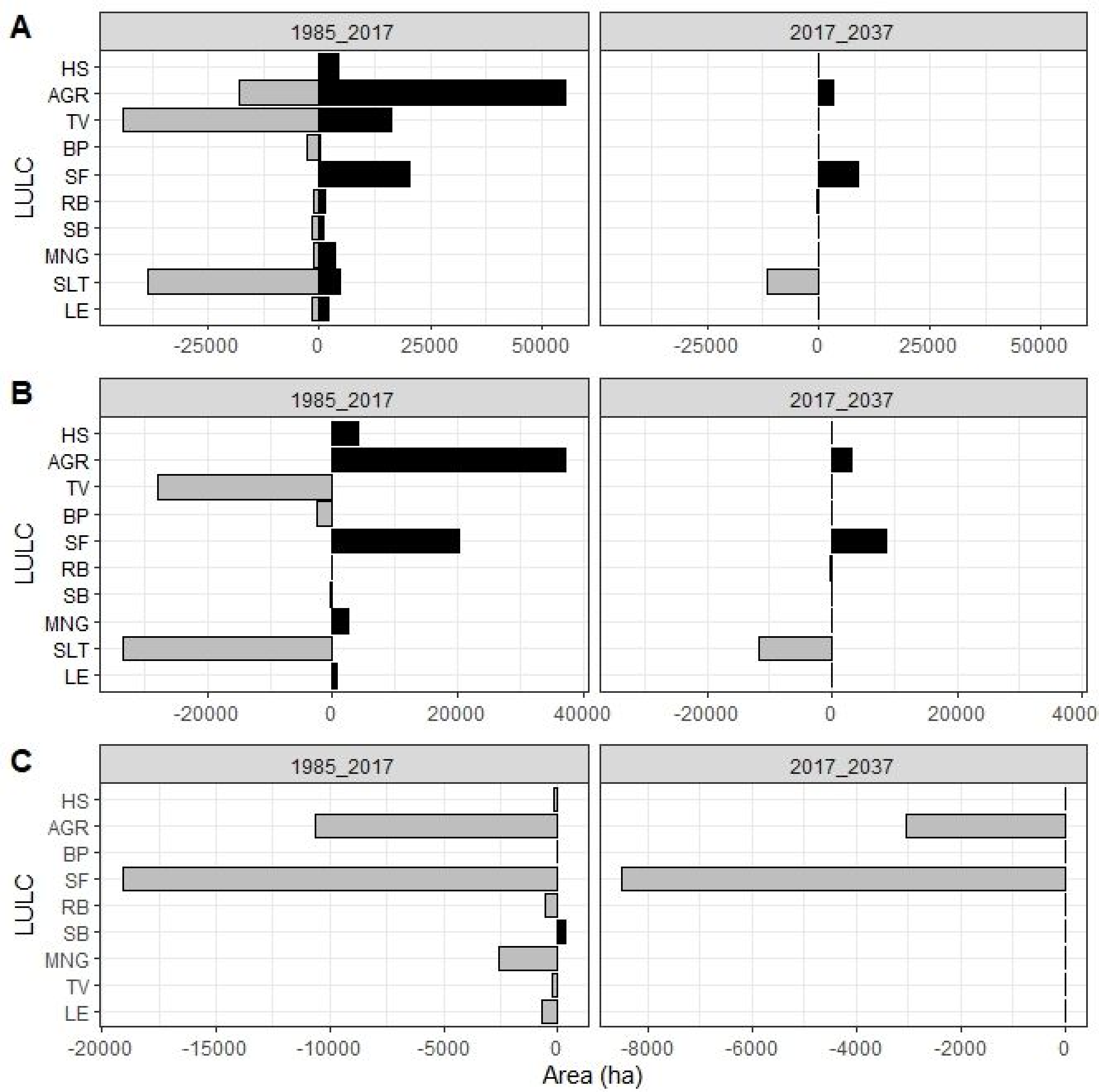

Figure 4

Observed (1985 to 2017) and predicted (2017 to 2037) Land use/Land cover change (LULCC) in the Bahia Santa Maria-La Reforma (BSMLR) coastal lagoon system. A) Losses (gray bars) and gains (black bars) by class. B) Net change by class. C) Contribution to net change experienced by saltmarshes. LE (lagoons and estuaries), SLT (saltmarshes), MNG (mangroves), SB (sandy beaches), RB (reed beds), SF (shrimp farming), DP (dams and ponds), TV (terrestria vegetation), AGR (agriculture), HS (human settlements).

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