Estuarine, Coastal and Shelf Science 133 (2013) 23–31

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

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

Land cover changes in tidal salt marshes of the Bahía Blanca estuary (Argentina) during the past 40 years

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ARTICLE INFO

Article history:
Received 5 June 2012
Accepted 24 July 2013
Available online 8 August 2013

Keywords:
South America
Argentina
Bahía Blanca estuary
sea level changes
coastal erosion
sedimentation
salt marshes

ABSTRACT

The present work describes land cover changes in the inner section of Canal Principal, in the Bahía Blanca estuary. The study area is subjected to a rising relative sea level, large interanual variations in rainfall, and recent changes human in land use. We used historical aerial photographs, high resolution satellite images, and GIS to quantify changes in land cover for the years 1967, 1996, and 2005. The replacement of *Sarcocornia perennis* marshes and halophytic shrub-like steppes by mudflats is a recurring pattern through the area. We estimated a total loss to mudflats of 33 and 6% of the area of marshes and steppes, respectively, and it may reflect increased erosion of relict Holocene coastal terraces in response to a rising sea level. Human activities have played a significant role in reshaping coastal landscape, particularly in the harbor area. Fifty percent of the area originally covered by shrub-like steppes and 33% of the *Sarcocornia perennis* marshes were replaced by human land uses. Major changes correspond to dredged spoil deposition and landfilling. One of the most striking changes observed is the increase of the area covered by *Spartina alterniflora* marshes. This type of replacement suggests the occurrence of depositional environments that, at least locally, allow bed elevation and vegetation growth. In the harbor area, an enhanced sedimentation may result from maintenance dredging. At the mouth of Maldonado channel, sediment deposition may occur during extraordinary heavy rainfall associated to El Niño.

1. Introduction

Long term changes in sea level are major driving forces in coastal wetlands evolution. A rising sea level produces a change in the ecological state of wetlands, expressed in a transgression upslope of the different zones within the coastal wetlands continuum (Brinson et al., 1995). While the eustatic (globally averaged) sea level has been rising from the Last Glacial Maximum (LGM) to the present, the relative height of the sea with respect to land can vary from place to place due to local tectonic and hydrographic effects. A falling or fluctuating relative sea level during the late Holocene characterized most coastlines of the southern hemisphere and eastern Asia (Pirazzoli, 1991). In the study area, several authors have identified a highstand about 6000 years ago when the relative sea level reached around 6 m above present (Isla, 1989; Violante and Parker, 2000; Cavallotto et al., 2004), and the late Holocene falling trend in relative sea level has resulted in wide low-lying coastal terraces, inherited from the former estuarine dynamics. According to the present rising trends in relative sea level, estimated from tidal gauge records (Fig. 1), this regressive landscape presently occupied by high marshes and supratidal steppes is undergoing increasing inundation.

Climate is a major control of coastal wetlands vegetation and also their development over time. In arid climates, especially, species composition and biomass of plant communities can respond to interannual and seasonal variations in rainfall (Callaway and Sabraw, 1994; Noe and Zedler, 2001). The Bahía Blanca coastal region is located between 38° S and 39° S, close to the so-called South American Arid Diagonal (Bruniard, 1982). The area presents a steep decrease in rainfall southward, and is characterized by large interannual variations in rainfall due to El Niño influence. In

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http://dx.doi.org/10.1016/j.ecss.2013.07.016

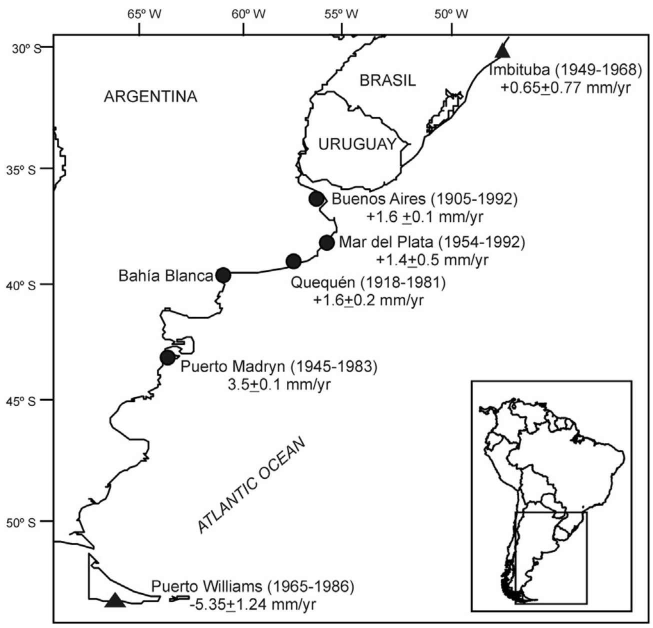

Fig. 1. Relative sea level trends derived from different tide gauge records along the temperate Atlantic coast of South America. Time periods considered in the estimations appear in parenthesis. Sources: Lanfredi et al., 1998 (circles) and Raicich, 2008 (triangles).

temperate salt marshes from Southern Brazil to the northern coasts of Argentina, *Spartina alterniflora* is the dominant species in the low marsh (Cagnoni and Faggi, 1993; Costa, 1997; Costa et al., 2003). The southernmost record of *Spartina alterniflora* 42°25′S (Bortolus et al., 2009). At greater latitudes, under the seasonally hypersaline conditions of the Patagonian coast, salt marshes are dominated by *Sarcocornia perennis*. In the study area, these two types of salt marshes overlap, and changes in the species distributions are expected to occur as a response to long term meteorological cycles like El Niño, that largely affect rainfall patterns.

In addition to these physical sources of variation, human activities have also exerted profound changes in coastal marshes. Compared to Europe and North America, most coastal environments of South America have been little modified by human activities. However, there is increasing pressure concentrated in major coastal cities, where large projects of coastal engineering interfere with natural processes, and pollution is a growing problem (Veblen et al., 2007). In the specific case of Bahía Blanca, the estuary holds the largest system of deep water ports in Argentina. These harbors include oil transfer buoys, trading docks and the largest navy base in the country. Between 1989 and 1991, the main navigation channel was dredged to a depth of 13.5 m from a previous 10 m, what promoted the settlement of one of the most important petrochemical poles in the region (Zilio et al., 2013).

In this work, we describe salt marsh change dynamics in the Bahía Blanca estuary. We used historical aerial photographs, high resolution satellite images, and GIS to quantify changes in land cover in four sites along the main navigational channel, for the years 1967, 1996, and 2005. Through transition matrices and Kappa Indices of Agreement (KIA), we analyzed changes in size and position of the characteristic plant communities within the coastal zone, as well as major human modifications like dredge spoil deposition and lanfilling.

2. Material and methods

The Bahía Blanca estuary comprises a series of major channels running NW-SE and extensive tidal flats, marshes and islands dissected by numerous small tidal channels (Melo, 2004). The northern portion of the estuary is dominated by Canal Principal (Fig. 2), a funnel-shaped channel that has a total length of 61 km, and varies in width from 200 m at the head to about 3–4 km at the mouth (Piccolo and Perillo, 1997). Mean tidal range increases along the Canal Principal reaching nearly 4 m at the head, close to the study area (Perillo et al., 2004). Based on tidal records obtained in Puerto Quequén (290 km north-east from Bahía Blanca), Lanfredi et al. (1988) estimated that relative sea level in the area increases at a rate of 1.6 mm/yr.

We compared four areas along the inner zone of Canal Principal, which were identified as Site 1 (191.8 ha), Site 2 (1147.2 ha), Site 3 (822.4 ha), and Site 4 (1306.3 ha). These sites were selected to be representative of the environmental gradient along the inner estuary, from the less modified landscape, close to the head of Canal Principal (Sites 1 and 2), to the harbor area in the transition to the middle reach, more affected by anthropic transformations (Sites 3 and 4).

Land cover classification was based on previous work in the same area (Verettoni, 1961, 1974; Nebbia and Zalba, 2007; Parodi, 2007). These authors identified different plant communities through field surveys and multivariate ordination techniques. For this study, we defined four natural land cover types based on physiognomic features visually identified on aerial photographs (scale 1:20,000). These four categories and their corresponding plant associations are listed below along with one class referred to as "human land use."

1-*Spartina alterniflora* marshes are the only vegetation type appearing below the level of the mean high tide.

2-*Spartina densiflora* marshes occasionally appear as pure stands that form a transition zone between *Spartina alterniflora* and *Sarcocornia perennis* in places influenced by freshwater discharges.

3-*Sarcocornia perennis* marshes commonly appear slightly above the limits of the mean high tide.

4-Halophytic shrub-like steppes, dominated by the shrubs *Cyclolepis genistoides*, *Allenrolfea patagonica*, and *Atriplex undulata*, occur at elevations above the highest tide.

5-The human land use class includes landfills and dredge spoils deposits.

In order to maintain equal areas in every period, a sixth class named mudflats and channels was also considered.

2.1. Assessment of land cover changes

Panchromatic aerial photographs at scale 1:20,000 were available for years 1967 and 1996. The photographs were scanned at 600 dpi and then orthorectified and georeferenced to the National Cartographic System (POSGAR94), at a ground resolution of 1 × 1 m. We additionally used a set of four high resolution satellite images (Ikonos) acquired in 2005. All the aerial photographs were registered to the rectified Ikonos images using the Georeferencing tool in ArcGis 9.0 (ESRI), and the information was integrated to obtain three photo-mosaics for years 1967, 1996, and 2005.

Four polygons were digitized to delineate the areas of interest and the boundaries between land cover types were digitized on screen within each one of the areas, and for each year. The only exception was the area site 4 for year 2005, which could not be considered because the corresponding image was acquired during high tide. The resulting 11 land cover maps where then transformed to raster format (pixel size 1 × 1 m) to proceed with the analysis.

Land cover changes were evaluated through transition matrix analysis and Kappa indices of Agreement (KIA). Transition matrices describe the number of raster cells that fall within each possible combination of land cover types on the two dates involved.

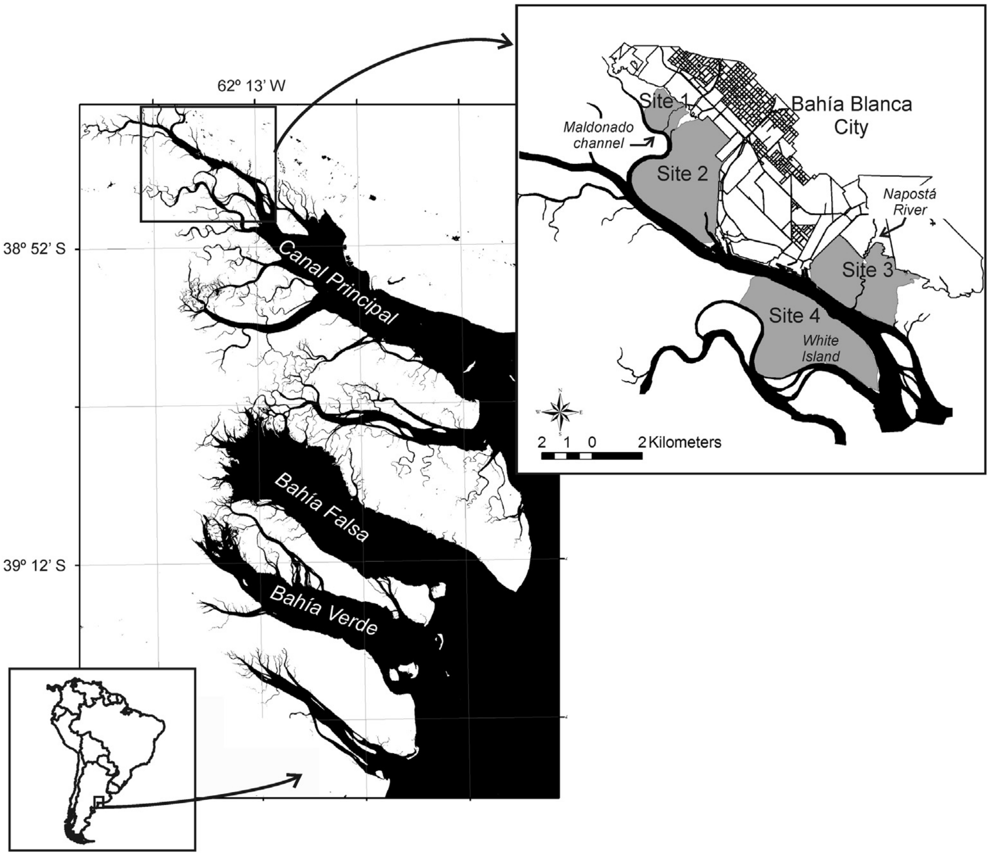

Fig. 2. Geographic extent of the Bahía Blanca estuary, Argentina, and location of the four study areas within the inner zone of Canal Principal in the insert.

Diagonal entries tabulate all cells that did not change their class from the first date to the second, and off-diagonal elements represent changes. The general KIA is a measure of the agreement of two maps and represents the magnitude of changes between them (Eastman et al., 1995; Congalton and Green, 1999). Kappa indices with values between 1 and 0.8 represent little change, values between 0.8 and 0.4 represent moderate change, and values less than 0.4 represent extensive changes (Landis and Koch, 1977).

The general KIA was calculated as:

$$\mathrm{KIA} = \left(N \sum_{i=1}^{k} x_{ii} - \sum_{i=1}^{k} x_{i+} x_{+i} \right) \Bigg/ \left(N^2 - \sum_{i=1}^{k} x_{i+} x_{+i} \right)$$

where k is the number of rows (land cover classes) in the matrix, x_{ii} is the number of raster cells in row i and column i (diagonal elements), $x_{\mathrm{i+}}$ and $x_{+\mathrm{i}}$ are the total marginal of row i and column i respectively, and N is the total number of raster cells. The frequency matrices and KIA calculations were performed using Idrisi32 software.

3. Results

According to KIA values, the four areas presented moderate changes, with Sites 3 and 4 as the areas showing the greatest landscape modifications (Table 1). The loss of *Sarcocornia perennis* marshes was a recurring pattern observed through the four sites under study (Table 2), with a total loss of 644.7 ha (65% of the original area in 1967). The area covered by halophytic steppes was also reduced, from the initial 375.5 ha considered in 1967 to 161.6 ha in 2005. Within these losses, we could identify two major types of replacements (Table 3). In the case of *Sarcocornia perennis* marshes, 331.2 ha were lost by erosion (33% of the original area) and replacement by human land uses accounts for a similar percentage loss (327.8 ha). The loss of halophytic shrub-like steppes was mainly due to the replacement of the original land cover by human land uses. About 49% of the original halophytic steppes area (182.9 ha) was replaced by landfills and dredge spoil deposits, and only 16% of these natural environments were lost to mudflats and channels.

While *Sarcocornia perennis* marshes and halophytic steppes diminished, *Spartina alterniflora* marshes extended their cover. In three of the four sites we observed an expansion of this land cover type, from the original 215.3 ha present in 1967 to 773,6 ha in 2005. Human land cover types derived from landfilling and dredging also increased, from 1.3 to 629.1 ha. Besides the general patterns described, a geographical gradient was also observed, characterized by the dominance of marsh erosion in the inner sites (Sites 1 and 2),

Table 1
General KIAs for Sites 1, 2, 3 and 4.

Site	Period	General KIA
Site 1	1967–1996	0.8399
	1996–2005	0.9334
	1967–2005	0.7951
Site 2	1967–1996	0.7527
	1996–2005	0.874
	1967–2005	0.7164
Site 3	1967–1996	0.6756
	1996–2005	0.8431
	1967–2005	0.5748
Site 4	1967–1996	0.6803

Table 2
Change in area (ha) for each land cover type, in each site, and total change for each land cover type, proportional to the original area. Positive numbers indicate gains, negative numbers indicate losses.

Land cover type	Site 1 (ha)	Site 2 (ha)	Site 3 (ha)	Site 4 (ha)	(Total change/original area)
Sarcocornia perennis marshes	−35.2	−299.9	−85.8	−223.7	−0.65
Halophytic shrub-like steppes	–	−77.9	−64.3	−71.6	−0.57
Spartina alterniflora marshes	24.0	–	246.6	287.7	2.59
Spartina densiflora marshes	5.3	–	–	–	6.44
Human land use	–	152.4	183.3	292.2	502.3

and direct human transformations in the harbor area (Sites 3 and 4). The expansion of *Spartina alterniflora* marshes, however, did not show any clear geographical trend.

In Site 1, 60% of the original cover of *Sarcocornia perennis* marshes (31 ha) was lost to mudflats (Fig. 3), in a process that involves lowering surface elevation and loosing vegetation cover. As a counterpart, a total amount of 24.5 ha of formerly bare mudflats were replaced by marshes, in a process entailing sediment accretion, surface elevation, and colonization by plants. In this site, *Spartina alterniflora* was not present in 1967, with the exception of a single spot of a few square meters at the head of the creek. In 2005, *Spartina alterniflora* was the most common species forming marshes, covering 50% of the vegetated area (24.1 ha). Although marsh expansion was primarily associated to *Spartina alterniflora*, Site 1 was the only area considered here in which we identified a pure stand of *Spartina densiflora,* covering 6.1 ha by 2005.

In Site 2, a total area of 231.3 ha of *Sarcocornia perennis* marshes, 38% of their cover in 1967, transformed to mudflats (Fig. 4). A similar process of land loss occurred to 17.1 ha of halophytic shrub-like steppes (12% of their original cover). Additionally, 20.2 ha of steppes transformed to *Sarcocornia perennis* marshes, a replacement mediated by increased inundation and the loss of species intolerant to flooding. The human land use, in turn, expanded mainly over former *Sarcocornia perennis* marshes and halophytic steppes. Comparatively, 14% of the original area covered by *Sarcocornia perennis* marshes (93.7 ha) and 41% of the area covered by halophytic steppes (58.7 ha) were replaced by human land use between 1967 and 2005. The increase in area of the human land use occurred mainly before 1996, due to the expansion of a large landfill.

Landscape trends for Site 3 (Fig. 5) were similar to those observed in Sites 1 and 2. In this site, 49% of the original area

Table 3
Major types of replacement involved in the loss of *Sarcocornia perennis* marshes and halophytic shrub-like steppes.

Land cover type	Original area (ha)	Percentage of original area lost to mudflats and channels	Percentage of original area replaced by human uses
Sarcocornia perennis marshes	996.17	33.2	32.9
Halophytic shrub-like steppes	375.45	16.1	48.7

covered by *Sarcocornia perennis* marshes (52.9 ha) was lost to mudflats by erosion. Land loss by erosion also accounted for 20.9 ha of halophytic shrub-like steppes (13% of their original cover). *Spartina alterniflora* marshes expanded from 77.3 to 323.9 ha, being this change almost entirely explained (99%) by the colonization of former mudflats. Human land use expanded from 0.6 to 183.9 ha, mainly over mudflats (57%), but also over halophytic steppes (29%) and *Sarcocornia perennis* marshes (14%). In this case the human land use expanded through an extensive landfill covering most of the intertidal fringe, in contrast to the previous sites where human activities concentrated in places at higher elevations.

Given the available photographs, the only period considered in Site 4 was 1967–1996. The most striking change is the complete disappearance of *Sarcocornia perennis* marshes (208.2 ha) and halophytic shrubs (70.8 ha), the original vegetation types, which were replaced by human land use. After 1996, *Spartina alterniflora* marshes were the only natural land cover left, besides mudflats and channels (Fig. 6). According to the transition matrix, there was a threefold increase in the area covered by *Spartina alterniflora* marshes (from 137.9 to 425.7 ha). The supratidal portion exposed in the 2005 image showed that the area covered by human land use had not changed, and natural communities that had disappeared did not recover.

4. Discussion

The erosion of *Sarcocornia perennis* marshes and halophytic steppes was observed through the four sites under study, and may reflect a response to a rising sea level. According to our results, 331.17 ha of salt marshes were lost by erosion, equivalent to a loss of 0.87% of the original salt marsh area per year. Based on satellite images, Isacch et al. (2006) estimated that *Sarcocornia perennis* marshes in the Bahía Blanca estuary covered 20376 ha by 2003. Assuming the same rate of erosion over the entire region, the estimated land loss for the Bahía Blanca estuary, just by erosion of *Sarcocornia perennis* marshes, would have been about 267 ha/yr. Actual wetland loss is expected to be higher than this estimate, if we considered direct human interventions.

Wetland losses of this magnitude far exceed average estimates for the Eastern United States (Table 4), and resemble loss rates commonly reported in the Louisiana-Mississippi coastal zone. A well known example is the Modern (Birdsfoot) delta, within the Mississippi River Delta Complex, which has experienced rates of land loss between 570 and 1140 ha/yr since 1958 (Dunbar et al., 1992). In this case, the increased marsh inundation has been pointed as a major cause of vegetation diebacks (Mendelssohn et al., 1981; Webb et al., 1995), and resulted in some of the highest rates of wetlands loss for coastal Louisiana (Mendelssohn and Kuhn, 2003). Rates of relative sea level rise for the Mississippi River Delta range from 3.6 to 177 mm/year (Penland and Ramsey, 1990), largely exceeding the estimated rise in our study area. Moreover, we never observed vegetation diebacks in *Sarcocornia perennis* marshes at the Bahía Blanca estuary, suggesting that a slightly increased flooding duration was not the major cause of salt marsh loss.

The enhanced erosion of soft sediments is one of the distinct physical impacts of a rising sea level (Nicholls and Leatherman, 1995). The land cover maps presented here suggest that lateral erosion at the edge of *Sarcocornia perennis* marshes, as well as headward erosion of tidal channels, would be major mechanisms of shoreline retreat and salt marsh loss. The coastal platform presently occupied by *Sarcocornia perennis* marshes derives from marine deposits that formed during the latest Holocene transgressive stage, under a slightly higher relative sea level (González, 1989). Under the current conditions, these low lying coastal landforms

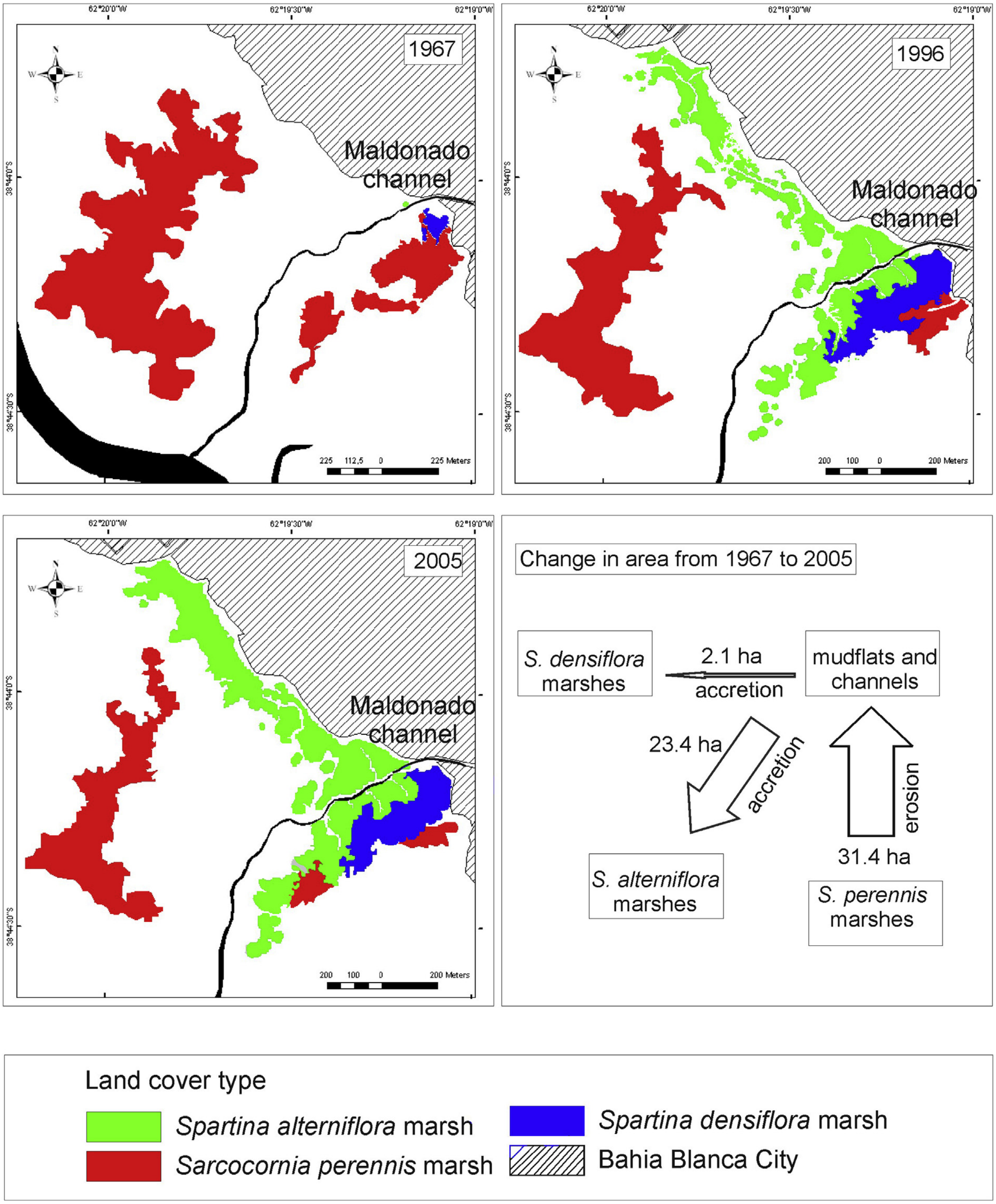

Fig. 3. Land cover maps for Site 1, at the mouth of Maldonado Channel, derived from historical aerial photographs (a, b), and from satellite images (c). Main land cover replacements between 1967 and 2005 are summarized in d, according to the area involved and the direction of change. The width of the arrow is proportional to the percent area involved in that replacement.

became relict and their erosion may be accelerated by the recent rise in sea level.

Human activities have also played a significant role in reshaping the coastal landscape, particularly in the harbor area. Fifty percent of the original area covered by shrub-like steppes, and 33% of the *Sarcocornia perennis* marshes were replaced by human land uses. In the case of Site 4, the island was completely transformed by a massive spoil dredge deposit. In Site 3, land fillings where performed to site a thermoelectric power plant and the largest industrial settlement in the area, directly connected to Ingeniero White Port.

Besides generalized marsh erosion and direct human transformations in the harbor area, one of the most striking changes observed was the expansion of *Spartina alterniflora* marshes, as well

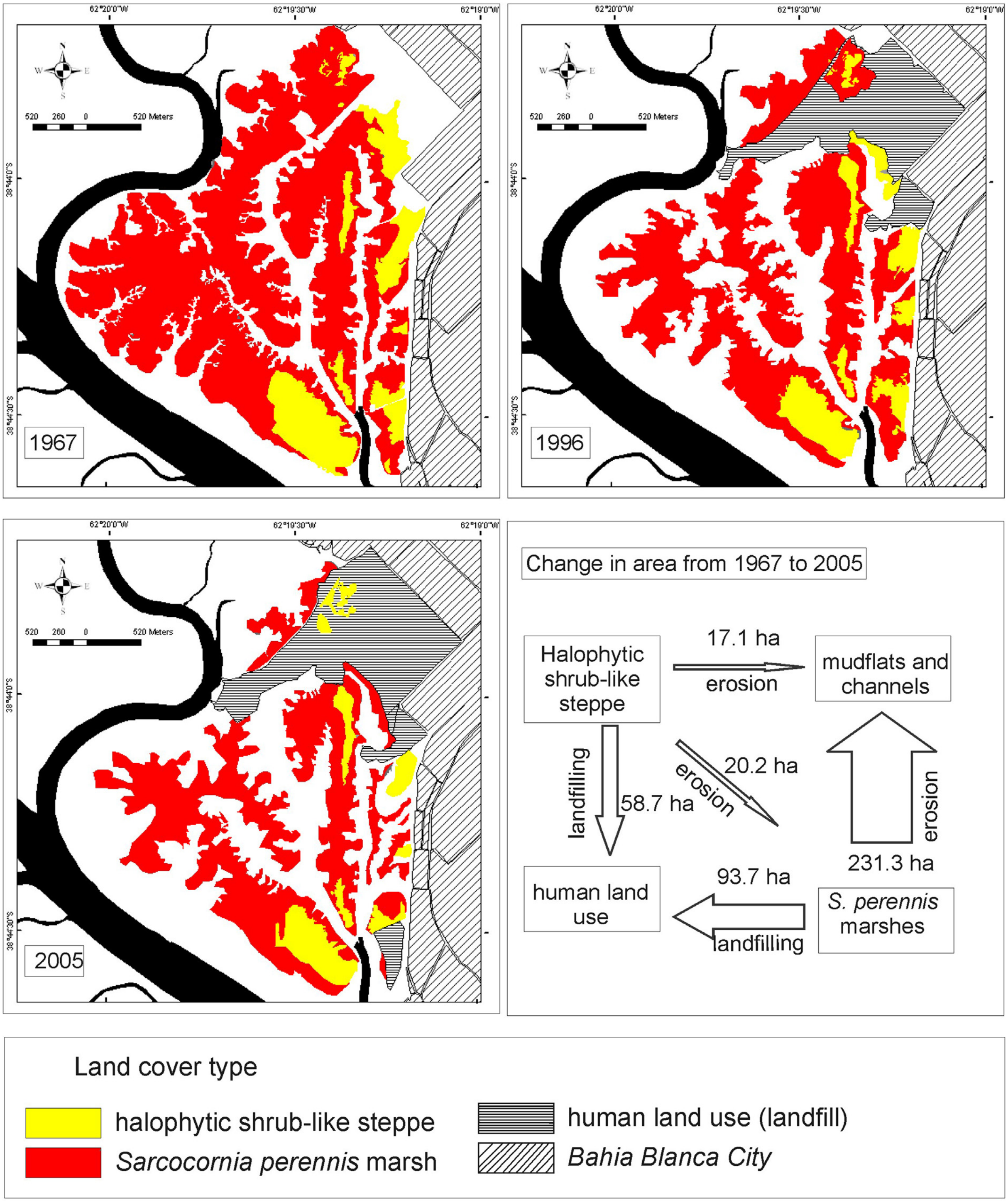

Fig. 4. Land cover maps for Site 2 derived from historical aerial photographs (a, b), and from satellite images (c). Main land cover replacements between 1967 and 2005 are summarized (d) according to the area involved and the direction of change. The width of the arrow is proportional to the percent area involved.

as the appearance of new marshes on former bare mudflats. Range expansions of *Spartina alterniflora* are common in places where this species has been introduced. Along the east coast of China, its coverage increased from approximately 260 ha by 1985 (Chung, 1989) to more than 112000 ha by 2000 (An et al., 2007). In Willapa Bay, in the Pacific Coast of North America, *Spartina alterniflora* was introduced in the 19th century. By 1990, marshes covered approximately 1100 ha, and the species would be spreading at a rate of about 20% per year (Feist and Simenstad, 2000). In the Bahía Blanca estuary, however, *Spartina alterniflora* is within its native range of distribution, and the expansion of these marshes suggests a change in environmental conditions.

Expansions of *Spartina alterniflora* marshes have been reported elsewhere in association with dredging and spoil deposits (Streever, 2000; Proffitt et al., 2003), and dredged sediment additions are commonly used to stimulate plant production in degraded marshes (DeLaune et al., 1990; Pezeshki et al., 1992; Ford et al., 1999). Sediment deposition may reduce flooding, nutrient

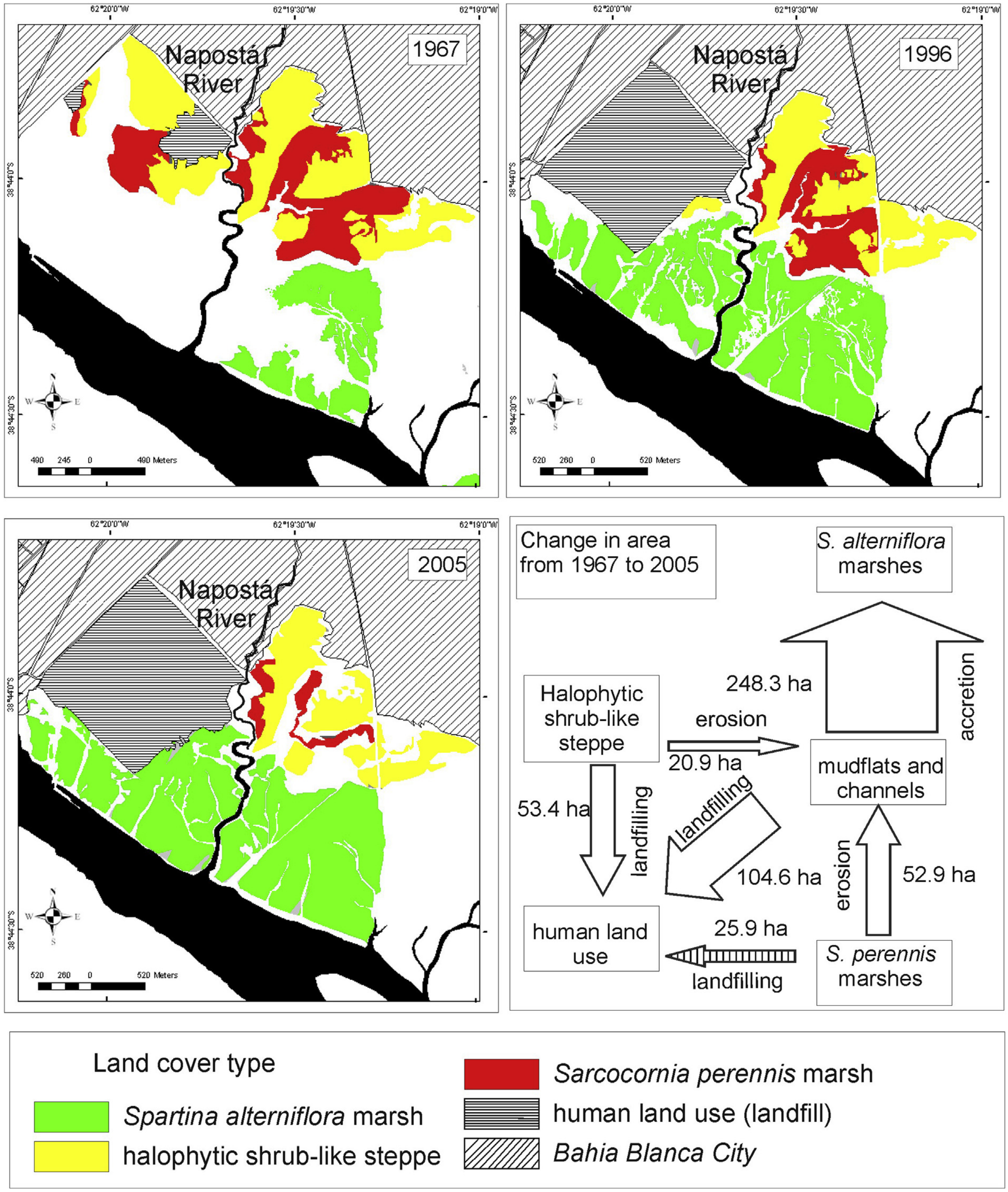

Fig. 5. Land cover maps for Site 3, at the mouth of Napostá River, derived from historical aerial photographs (a, b), and from satellite images (c). Main land cover replacements between 1967 and 2005 are summarized (d) according to the area involved and the direction of change. The width of the arrow is proportional to the percent area involved.

deficiency, and interstitial sulfide stresses. Driving mechanism for marsh expansion would then be related to these improved substrate conditions (Mendelssohn and Kuhn, 2003). In our study area, an enhanced sedimentation may result from maintenance dredging of navigation channels, and could have facilitated marsh expansion in Site 3 and 4.

Marsh appearance and expansion in Site 1, however, was not related to dredging. In this case, *Spartina alterniflora* appeared at the mouth of Maldonado channel, out of the area affected by harbor activities. Working in the Tijuana estuary, California, Ward et al. (2003) found a similar expansion in marshes of *Spartina foliosa*, a native species on the West coast of North America. Colonization of new clones was related to extraordinary events of sediment deposition and significant decreases in soil salinity after heavy rainfall. In our case, Maldonado channel was artificially connected to Napostá River upstream Bahía Blanca city during the late 1950's. This connection, designed to divert river overflow during storms, could facilitate sediment deposition and lower soil salinity at

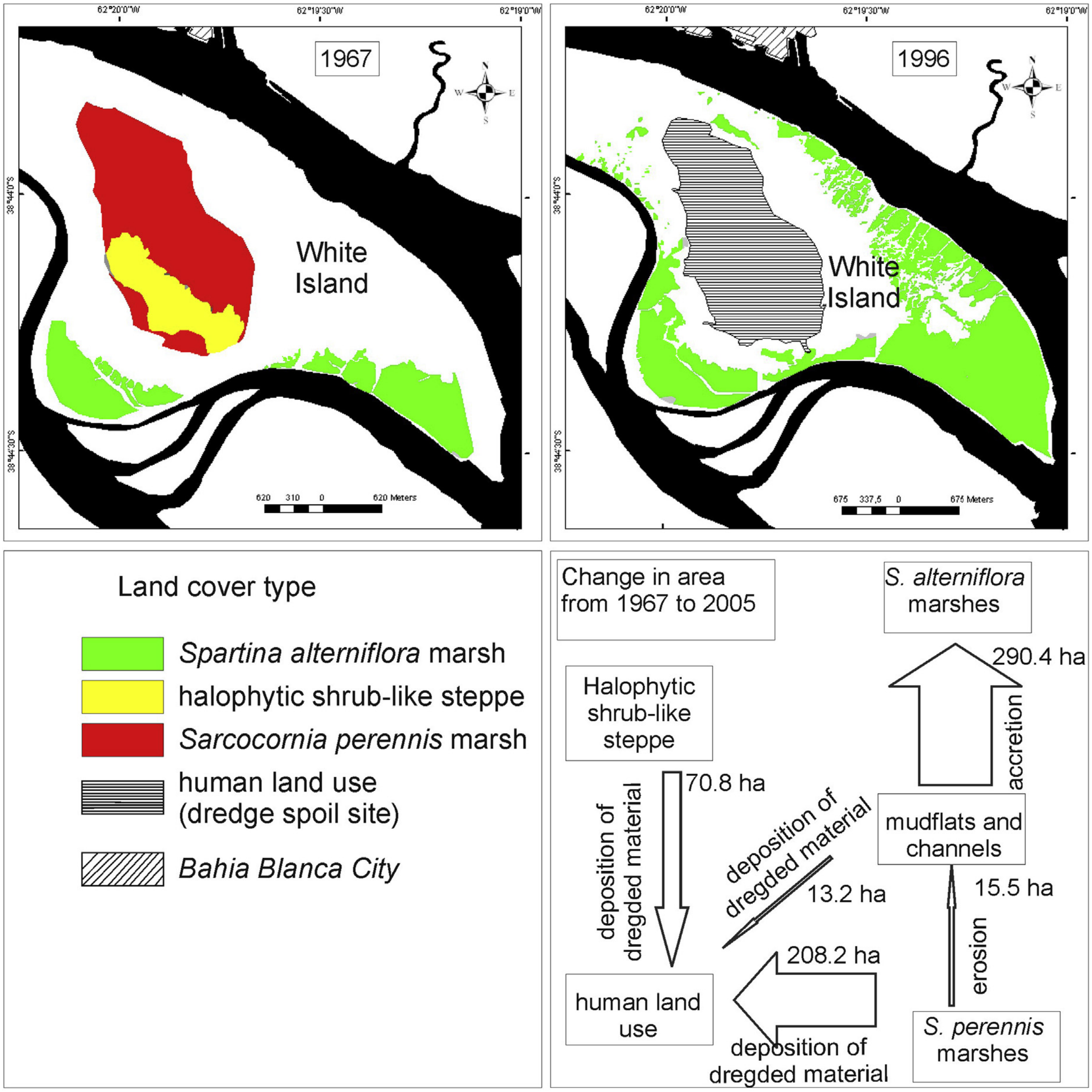

Fig. 6. Land cover maps for Site 4, in White Island, derived from historical aerial photographs (a, b), and from satellite images (c). Main land cover replacements between 1967 and 2005 are summarized (d) according to the area involved and the direction of change. The width of the arrow is proportional to the percent area involved.

the mouth of the channel during exceptional events of discharge, caused by extraordinary heavy rainfall during El Niño periods.

The present work describes salt marsh change at the Bahía Blanca estuary, a highly dynamic coastal environment in the South Western Atlantic. Through the quantification of recent land cover changes we were able to identify two major types of replacement, opposite in terms of the sedimentary process associated. While *Sarcocornia perennis* marshes are eroding, *Spartina alterniflora* marshes are expanding their distribution, in a process that involves an increase in surface elevation and mudflat colonization by plants. The observed marsh loss and the erosion of soft sediments are in agreement with the local rising trends in relative sea level. However, the eroded marshes may supply mud to the tidal sediment budget, and may contribute to the sediment deposition that takes place in the expanding *Spartina alterniflora* marshes. Although our work is geographically restricted, we provide a description of spatial and temporal patterns of salt marsh response for an area largely underrepresented in literature. Further work is needed to evaluate the potential to extrapolate the observations to a broader scale.

Table 4
Losses of coastal wetlands estimated for the United States.

	Years	% Lost/yr	Source
Louisiana-Mississippi	1955–1978	0.86	Turner, 1990
New England	1770–1999	0.19	Bromberg and Bertness, 2005
New England and Atlantic New York	1886–1976	0.60	Gosselink and Baumann, 1980
Atlantic	1974–1983	0.14	Tiner, 1991
Great Lakes, Atlantic, and Gulf of Mexico	1998–2004	0.18	Stedman and Dahl, 2008
Bahía Blanca estuary	1967–2005	0.87	This work

Acknowledgments

We dedicate this paper to the memory of our co-author and friend, Dr Mark M. Brinson, who sadly passed away in 2011. We are deeply grateful to their colleagues Linda Blum and Robert Christian, who made the completion of this manuscript possible. We also thank Dr. Ivan Valiela and the anonymous reviewer **for their insightful and constructive comments**. This research was supported by CONICET through PIP059, ANPCyT through PICT2071, and external award from the Fulbright Program.

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