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ASSESSING VEGETATION CHANGE IN COASTAL LANDSCAPES OF THE NORTHERN GULF OF MEXICO

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Abstract: Multiple factors have caused rapid changes in coastal landscapes in the last half century. Coastal natural areas have been set aside to mitigate some of these changes for habitat preservation, among other goals. However, areas set aside for conservation are not exempt from these rapid changes. A major concern for coastal wetlands is the potential for habitat loss resulting from external land-use changes and sea-level rise, which essentially threaten these natural areas from all sides. In order to quantify these trends, we determined the types and rates of land-use/land-cover conversion in differing coastal sites in the U.S. along the northern Gulf of Mexico from the 1950s to the 1990s using existing National Wetlands Inventory (NWI) habitat data and Geographic Information Systems (GIS). All sites were located in protected areas and contained an intact marsh-to-forest transition. A buffer zone of ~2000 m around each site was also analyzed. Two sites, Mandalay National Wildlife Refuge (MNWR) and the Barataria Preserve Unit of the Jean Lafitte National Historical Park and Preserve (JLNHPP), were located on the Mississippi Deltaic Plain in Louisiana, while the other sites, Grand Bay National Estuarine Research Reserve (GBNERR) and Weeks Bay National Estuarine Research Reserve (WBNERR), were located on the Gulf Coastal Plain in Mississippi and Alabama, respectively. Results showed prevalent marsh loss across all sites in the study, although the rate and type of marsh conversion to other land-cover types varied between the Mississippi Delta sites and the Coastal Plain. In the Delta, marsh was converted to open water along shorelines and in internal patches, but the majority of marsh loss was attributed to scrub-shrub encroachment. In the Coastal Plain, marsh was lost more slowly overall, both along the shoreline and forest-marsh boundary. The main trend in the Coastal Plain was replacement of agricultural areas by forest. The buffers experienced an increase in anthropogenically-modified categories, except for a decrease in agricultural areas. Our study suggests that coastal transitions of the northern Gulf of Mexico have indeed experienced landward and seaward losses and that marsh areas are especially vulnerable. It appears that marshes are not keeping pace with the spatial shifts in the aquatic to terrestrial transition as sea level rises, although results in the Coastal Plain are less conclusive because major land-use changes dominate the trends.

Key Words: coastal wetlands, landscape ecology, National Wetlands Inventory, sea-level rise, vegetation change, wetland-upland transitions

INTRODUCTION

Coastal landscapes have undergone rapid changes in the last century. One major reason for this change has been human population growth and resulting land conversion, pollution, and hydrologic modifications (Day et al. 2000, Kennish 2001, Kennish 2002). Coastal areas are highly desirable for human settlement and are almost three times as densely populated as the global average, with 1.2 billion people living within 100 km of the shoreline (Small and Nicholls 2003). In addition to stresses imposed by local impacts, coastal wetlands, which are located on the downstream portion of drainages, are exposed to the cumulative impacts of urbanization, deforestation, and agriculture that occur on the entire watershed (Sklar and Browder 1998, Kennish 2002). Coastal wetlands are also threatened by sea-level rise, which is predicted to increase into the future (IPCC 2001). Sea-level rise, in combination with human influences and natural disturbances, is likely to produce pressure on an already highly vulnerable biological system, leading to a wide array of potential changes to wetland vegetation over time (Michener et al. 1997, Williams et al. 1999a, Williams et al. 1999b, Denslow and Battaglia 2002).

Sea-level rise is projected to be one of the most deleterious consequences of global warming (IPCC

Table 1. Published accretion and RSLR rates representing marshes of similar habitats in areas near our study sites in the Louisiana Deltaic Plain and the Gulf Coastal Plain.

Region	Accretion Rate (cm/yr)	RSLR (cm/yr)
Barataria Basin, LA	0.65 (Hatton et al. 1983)	9.2 (Hatton et al. 1983)
Terrebonne Basin, LA	0.90* (DeLaune et al. 1989)	1.09 (Penland and Ramsey 1990)
Northern Gulf Coastal Plain	0.56 (Callaway et al. 1997)	0.15 (Penland and Ramsey 1990)

* This rate may not accurately represent accretion rates at our Terrebonne study area (MNWR) due to geographic separation between the two sites.

2001). Although sea-level rise is not occurring in a uniform fashion across the world, the average annual rate of increase taken from all available long-term tidal gauge measurements indicates that eustatic sea level (overall mean global water level) has risen 1–2 mm/year in the last century (Douglas 1992, Michener et al. 1997, IPCC 2001, Kearney 2001, Leatherman 2001, Scavia et al. 2002), and the effects of sea-level rise have already been experienced in many coastal areas (Kearney 2001). Although coastal ecosystems are adapted to fluctuating sea levels over a relatively long time period, if sea level is rising faster than the rate of mineral sediment deposition and organic matter accumulation (Callaway et al. 1997, Reed 2002, Scavia et al. 2002), then land loss is inevitable. The rate of this land loss may lag behind and thus not closely track sea-level rise if a threshold must be reached before land loss occurs (Michener et al. 1997). In this case, historical changes should be investigated in order to document these critical sea levels.

Subsidence, which is land "sinking," exacerbates the impacts of sea-level rise, and the combination is known as relative sea-level rise (RSLR) (Penland and Ramsey 1990). Subsidence can be the result of natural geologic processes or hydrologic modifications (Baumann et al. 1984). Where RSLR is greater than vertical accretion, wetland systems are threatened with inundation, erosion, and saltwater intrusion (Baumann et al. 1984, Day et al. 1995, Martin et al. 2001). Rates of RSLR vary across the Gulf of Mexico, with the highest rates found in coastal Louisiana on the Mississippi Delta. The Mississippi Delta, which is experiencing tectonic downwarping, sediment compaction, and anthropogenic alterations such as levees and canals, is especially prone to RSLR (Penland and Ramsey 1990, Day et al. 1995, Visser et al. 1999), and the state of Louisiana lost over 1036 km^2 of land between 1978 and 1990, which is a rate of ~91 km^2/year (Barras et al. 1994). The rate of land loss in Louisiana is five times greater than the Gulf of Mexico average (Penland and Ramsey 1990). In contrast, the Gulf Coastal Plain is not influenced by the processes characteristic of a major delta system and, therefore, experiences much lower rates of RSLR, which vary from ~2 to 5 mm per year (Hanson and Maul 1993). In a study documenting accretion rates in the Gulf Coastal Plain, Callaway et al. (1997) found that the vertical accretion of the marsh exceeds subsidence and sea-level rise.

Overlapping thresholds of species produce characteristic zonation in composition and physiognomy of community distribution along the elevation gradient of coastal transitions. These coastal transitions grade from marsh (next to the water's edge) into shrub- and forest-dominated zones on the landward end of the transition. These general vegetation types, along with anthropogenic land-use categories, can be thought of as states (Brinson et al. 1995) that change through time in response to differing rates of RSLR and accretion. Land loss generally occurs on the seaward portion of the marsh, but there is potential for marsh species to migrate upslope, replacing shrub and forest species if conditions are conducive to establishment (Brinson et al. 1995).

The rate of RSLR in combination with vertical marsh accretion varies across our sites, which should influence establishment success and pattern of vegetation change. Available data indicate that RSLR far exceeds accretion in the Mississippi Delta. Conversely, the Coastal Plain has a positive net accretion in selected sites along the Gulf Coast (Callaway et al. 1997). Although long-term RSLR and accretion data specific to our sites were not available, we list some published rates from nearby sites that are ecologically comparable (Table 1). Based on this information, we predict that sites in the Mississippi Deltaic Plain are changing more rapidly than the non-Delta sites. In the Delta, we expect migration to be unable to keep pace with RSLR, causing vegetation to be replaced by open water. Although marsh would experience the most loss in this scenario, shrub and forested areas would also decrease but at a slower rate due to longer lifespans of constituent species. In the Coastal Plain, if marsh accretion does exceed RSLR, we expect contrasting trends. In this case, vegetation coverage would increase as open water decreased and species migrated seaward. Outside of the protected areas, we expect conversion from 'natural' vegetation to anthropogenically-modified vegetation and land-cover categories. Landscape changes in surrounding areas should increase habitat fragmentation, limit area for migrating species to establish, and further compress the aquatic – terrestrial transition in combination with RSLR (Titus 1998).

To test our predictions, we focused on vegetation structure at the landscape scale in wetland-upland transitions of protected coastal areas and the surrounding land-use changes that occurred during the latter half of the twentieth century. Although several studies have documented the land loss and habitat changes in the Lower Mississippi Alluvial Valley (Sasser et al. 1986, Day et al. 1995, Turner 1997, Visser et al.1999, Day et al. 2000), no studies to our knowledge have compared habitat changes in Delta sites with other sites along the Gulf of Mexico, especially those that include a marsh-to-forest transition. This transition zone is where one would expect visible changes to be occurring in the face of RSLR.

Our objectives in this study were as follows: 1) to quantify the amount, rate, and type of vegetation and land-use change in selected coastal wetland-upland transition zones in the Mississippi Delta and the Gulf Coastal Plain and 2) from these analyses, to compare trends and patterns occurring in this region. We used available National Wetland Inventory (NWI) habitat data and GIS technology to quantify these land-use/land-cover changes at a landscape level. Although NWI data provide a useful tool for examining wetlands and surrounding areas for multiple time periods, there are limitations inherent in the data, and caution must be used when interpreting results (Kentula et al. 2004).

METHODS

Site Description

Our study included four sites that met the following criteria: 1) the presence of intact landscape transitions from forested habitats to coastal marsh vegetation, 2) minimal anthropogenic impact in terms of hydrologic modifications (there is no active regulation of water levels at sites) and fragmentation caused by land-use change, and 3) availability of NWI data. All sites are state or federally protected. From west to east, the sites include the Mandalay National Wildlife Refuge (MNWR) managed by the U.S. Fish and Wildlife Service (USFWS) in Louisiana; the Barataria Preserve Unit of the Jean Lafitte National Historical Park and Preserve (JLNHPP) in Louisiana managed by the National Park Service; Grand Bay National Estuarine Research Reserve (GBNERR) in Mississippi managed by the Mississippi Department of Marine Resources; and Weeks Bay National Estuarine Research Reserve (WBNERR) in Alabama managed by the Alabama Department of Conservation (Figure 1). Sites range in size from 648 ha at WBNERR to 8,094 ha at JLNHPP.

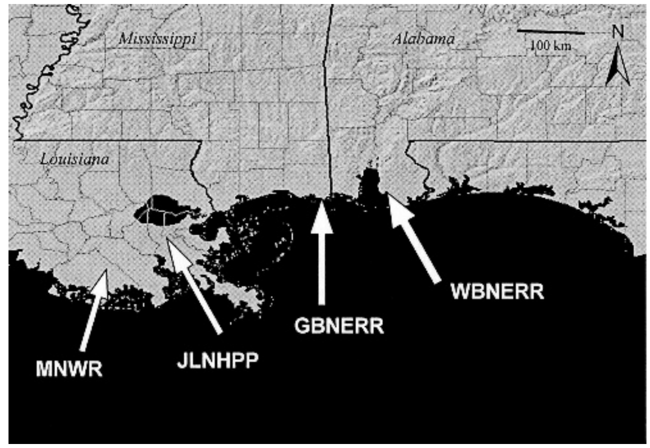


Figure 1. Map of the northern Gulf of Mexico coastline showing general locations of the four study sites. Mandalay National Wildlife Refuge (MNWR) and Jean Lafitte National Historic Park and Preserve (JLNHPP) are located on the Mississippi River Deltaic Plain. Grand Bay National Estuarine Research Reserve (GBNERR) and Weeks Bay National Estuarine Research Reserve (WBNERR) are on the Gulf Coastal Plain. Maximum distance between sites is ~350 km.

The species assemblages also vary across sites, but the general vegetation structure is similar along each elevation gradient (elevation change from water's edge to terrestrial forest ranges from 2–4 m). The Deltaic Plain sites, MNWR and JLNHPP, contain floating freshwater marsh, a forest-marsh ecotone, and a seasonally flooded bottomland hardwood forest. Common plant species that occur in each of these assemblages are *Panicum hemitomon* J.A. Schultes, *Sagittaria lancifolia* L., and *Leersia hexandra* Sw.; *Morella cerifera* (L.) Small, *Ilex cassine* L., and *Triadica sebifera* (L.) Small; *Acer rubrum* L., *Liquidambar styraciflua* L., *Quercus nigra* L., and *Taxodium distichum* (L.) L.C. Rich. Vegetation along the transition at JLNHPP also includes a narrow band of brackish marsh dominated by *Spartina patens* (Ait.) Muhl. and Schoenoplectus americanus (Pers.) Volk. ex Schinz & R. Keller. The Coastal Plain sites, GBNERR and WBNERR, contain salt marsh along the coast that grades into a low brackish marsh, freshwater high marsh, and forest-marsh ecotone. Common species in each

Table 2. Scale and emulsion of photography used to create NWI data.

Site	Year	Scale	Emulsion
MNWR	1955	1:20k	B&W
	1979	1:65k	CIR
	1988	1:65k	CIR
JLNHPP	1958	1:20k	B&W
	1978	1:65k	CIR
	1988	1:65k	CIR
	1998	1:40k	CIR
WBNERR	1955	1:20k	B&W
	1979	1:65k	CIR
	1988	1:65k	CIR
GBNERR	1955	1:20k	B&W
	1979	1:65k	CIR
	1988	1:65k	CIR

assemblage are *Spartina alterniflora* Loisel., *Juncus roemerianus* Scheele, *Panicum virgatum* L., *Cladium mariscus* (L.) Pohl ssp. *jamaicense* (Crantz) Kükenth., *Morella cerifera* (L.) Small, and *I. cassine*. At GBNERR, the ecotone grades into a pine savanna dominated by *Pinus* spp., whereas at WBNERR, it grades into a hydric seep forest dominated by *Nyssa biflora* Walt. and *Magnolia virginiana* L. and then a mesic hardwood forest dominated by *Magnolia grandiflora* L. and *Q. nigra*.

National Wetlands Inventory Data

National Wetlands Inventory (NWI) data were used to examine trends in vegetation and land use over time. Originally, NWI data were produced in response to growing concerns about the status of the nation's wetlands, and they are still used for research and management (Wilen and Frayer 1990, Cashin et al. 1992, Gaines et al. 2000). NWI data consist of polygons depicting outlines of wetland and upland categories. Hard copy maps (1:24,000) were prepared by manually delineating categories based on stereoscopic analysis of high altitude aerial photography. The scale and emulsion of the photography differed among sites/years (for details of the aerial photography refer to Table 2). Hard-copy maps were digitized, rectified, and georeferenced to a UTM coordinate system. Designated categories were verified by comparison to soil survey maps, ground-truthing, prior knowledge of the sites, and external critique from multiple agencies. The quality control steps employed in the creation of NWI data ensure a highly accurate product that provides information on wetland trends and detailed mapping of wetland areas (Tiner 1990). The quality of NWI data has been tested by field-checking in numerous studies and found to be over 90% accurate, with the non-forest categories being the most correctly identified categories (Stolt and Baker 1995, Kudray and Gale 2000).

We obtained digital NWI data from the National Wetlands Research Center (NWRC) located in Lafayette, Louisiana, USA. Wetland categories were based on the Cowardin et al. (1979) classification system, and uplands were classified using a custom classification system based on Anderson et al. (1976). The classification scheme can be found online at <http://atlas.lsu.edu/central/nwrc_nwi_legend.txt>. Although we did not include a systematic ground-truthing component, we did use vegetation data from recently established ground transects and limited aerial photography to help verify the classification.

Data Analyses

Digital NWI shapefiles were downloaded for each park, refuge, or reserve (hereafter, the area within the borders of the park, refuge, or reserve will be referred to as a site), plus a 2,000-m buffer extending beyond the boundary perimeter. The availability of NWI data varies among the sites; 1955, 1979, and 1988 were mapped for MNWR, GBNERR, and WBNERR and 1958, 1978, 1988, and 1998 for JLNHPP. One dataset, GBNERR 1978, was removed from the analysis due to the use of mixed-category classifications, which prevented comparison with all other data. Also, the area delineated on the 1955 NWI maps for MNWR included all of the refuge, but the northern extent of the mapping was the Gulf Intracoastal Waterway, which fell short of the 2,000-m buffer on the north side of the refuge. For consistency, the datasets for 1979 and 1988 were clipped to match this area. All data were imported into ArcMap 9.1 for analysis.

NWI polygons were aggregated into the following categories: forest, scrub-shrub, marsh, upland vegetation, non-vegetated, open water, modified vegetation, modified open water, agriculture, and urban. 'Forest,' 'scrub-shrub,' and 'marsh' are wetland vegetation categories. 'Upland vegetation' is any upland vegetation type (e.g., upland scrub-shrub and forest are not separated). Anthropogenically-altered categories of vegetation and open water are distinguished from other 'natural' or unmodified areas. For a detailed description of the aggregated NWI categories refer to Table 3.

Datasets containing the aggregated categories were divided into those for the site and those for the buffer. Within each site, we calculated the total area of each category, the total change in area for

Table 3. Description of classification system used to aggregate NWI categories and examples of NWI codes that would be classified within each aggregated category.

Aggregated Category	Classification rules	Example NWI codes and descriptions
Agriculture	Must contain Agriculture (A or DV2)	UA (Upland Agriculture)
		UDV2 (Upland Developed-Agriculture)
Forest	Must contain Forested (FO).	PFO (Palustrine Forested)
	Can be Estuarine (E), Riverine (R), Lacustine (L), or Palustrine (P). No special modifier.	E2FO2 (Estuarine Intertidal Forested-Needle-leaved deciduous)
Marsh	Must contain Emergent (EM).	E2EM1 (Estuarine Intertidal Emergent-Persistent)
	Can be E or P.	PEM (Palustrine Emergent)
	Does not contain special modifier.	
Modified Vegetation	Must contain Emergent (EM), Forested (FO), Scrub-shrub (SS), Upland Forested (UF), Upland Range (UR), or Upland Scrub-shrub (USS) AND contain one of the following special modifiers: b (beaver), d (partially drained/ditched), h (diked/impounded), s (spoil), or x (excavated). Can be E, R, L, or P.	E2EM1Ps (Estuarine Intertidal Emergent – Persistent Irregularly flooded - Spoil) PSS3Fx (Palustrine Scrub-shrub Broad-leaved evergreen Semi-permanently flooded – Excavated)
Non-Vegetated	Exposed areas free of vegetation. Can be E, P, or U.	E2RS (Estuarine Intertidal Rocky Shore)
		PUS (Palustrine Unconsolidated Shore)
Open Water	Must contain AB (Aquatic Bed), OW (Open Water), BB (Benthic Bed), or UB (Unconsolidated Bottom).	L1AB (Lacustrine Littoral Aquatic Bed)
	Can be E, R, L, or P.	E1OW (Estuarine Subtidal Open Water)
	No special modifier.	
Open Water Modified	Any open water category that contains one of the following special modifiers: d, h, o (oil/gas) or x.	POWx (Palustrine Open Water – Excavated)
	The majority of the Open Water Modified category is canals.	RABo (Riverine Aquatic Bed – Oil/Gas)
Scrub-shrub	Must contain SS.	E2SS (Estuarine Intertidal Scrub-shrub)
	Can be E or P.	PSS2 (Palustrine Scrub-shrub – Needle-leaved deciduous)
	No special modifier.	
Upland Vegetation	Must contain Upland (U).	UF (Upland Forested)
	No special modifier.	USS (Upland Scrub-shrub)
Urban	Must contain urban areas (U or DV1).	UDVI (Upland Developed – Urban)
		UU (Upland Urban)

each time period, and the rate of change. In addition, we analyzed patch dynamics by calculating the number of polygons (i.e., patches) per category and the average size of the patches for each year. In order to assess the surrounding land-use change, we calculated the combined area of all polygons for each vegetation/land-use category for each date in the buffer.

RESULTS

Site Analysis

The loss of marsh was the most apparent change at the Delta sites, with much of this loss attributed to conversion of marsh to scrub-shrub. At MNWR, scrub-shrub, which was absent in 1955, covered 266 ha in 1988 (Figure 2a and Figure 3a). At JLNHPP, scrub-shrub went from 26 ha to 811 ha (Figure 2b and Figure 3b). The highest loss and gain rate overall in both Delta sites was the marsh and scrub-shrub, respectively; the rate was especially high during the 1978/79–1988 time period. Marsh area was also lost in the Coastal Plain sites but at a lower rate. The Delta and Coastal Plain sites were also distinguished by the trends in forested areas, which showed little change in the Delta but marked increases in the Coastal Plain sites. The increase in forest in the Coastal Plain was accompanied by a high loss rate of agriculture (Figure 2c,d, Figure 3c,d). GBNERR lost upland vegetation, whereas at WBNERR, the highest rate of vegetation change was an increase in upland vegetation in the 1979–1988 period. The two Coastal Plain sites also varied

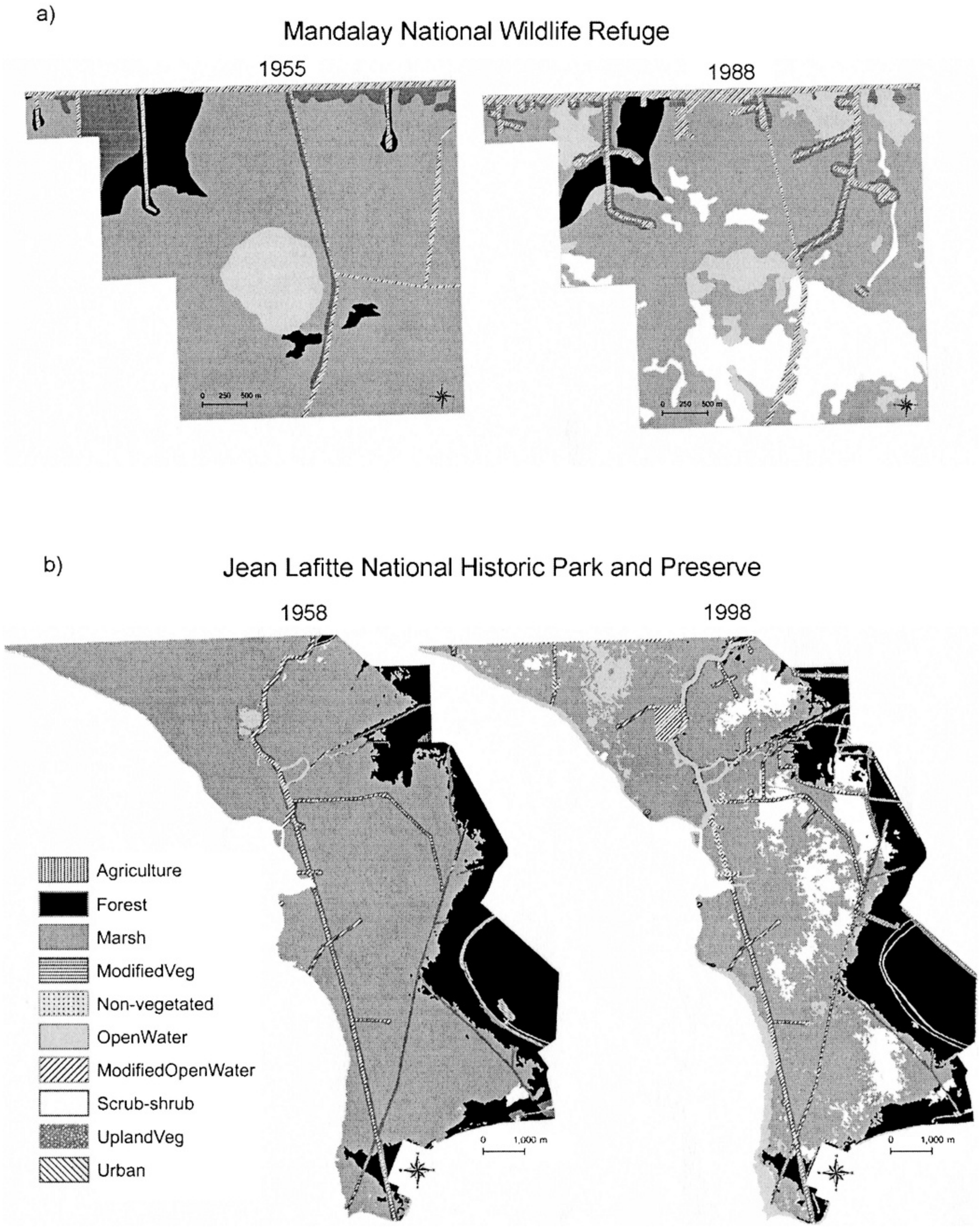
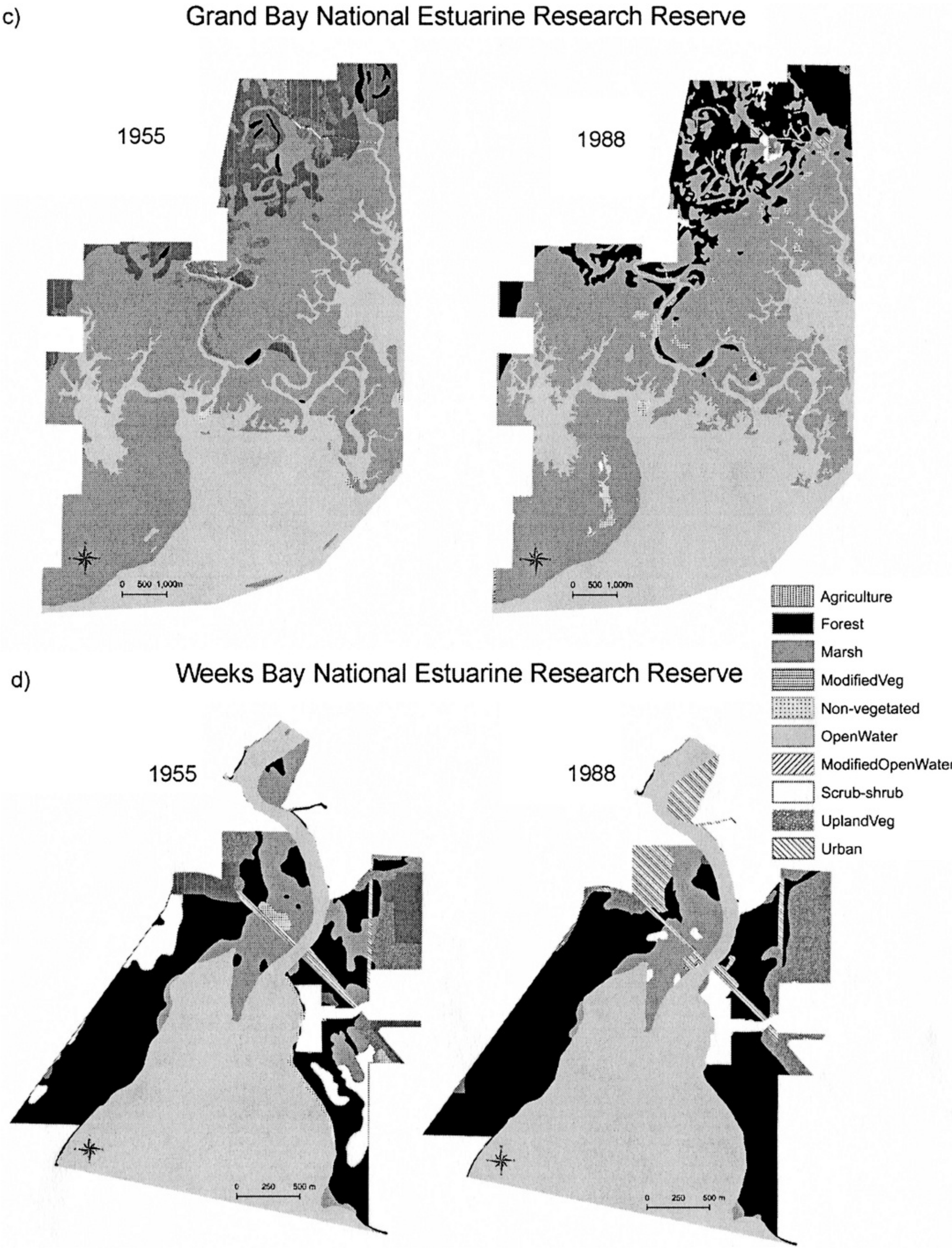


Figure 2. Land-use/land-cover maps based on aggregated NWI categories for the first and last dates available for a) MNWR, b) JLNHPP, c) GBNERR, and d) WBNERR. Note: these maps do not include buffer areas surrounding each reserve (the outline of each image represents the site boundary).



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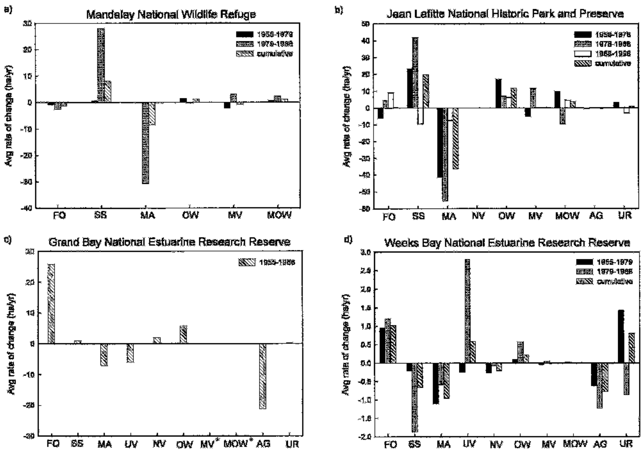

Figure 3. The rate of change (ha/year) of each category for each time period included in the analysis: a) MNWR, b) JLNHPP, c) GBNERR, and d) WBNERR. Rates were used to standardize inconsistencies in the time between study periods and differences among sites. The last bar for each category represents a cumulative rate for the entire study period. Only two dates for GBNERR were analyzed, and therefore, one bar represents 1955–1988. *Modified vegetation (MV) and Modified Open Water (MOW) at GBNERR, which are not visible on the graph, changed 0.206 and 0.089 ha from 1955 to 1988, respectively.

in scrub-shrub trends, with GBNERR gaining as WBNERR lost scrub-shrub. Open water and modified open water increased across all sites.

Patch Dynamics

The number of total patches increased from the beginning to end of the study period for all sites except WBNERR, and the average size of the patches decreased overall for all sites. In the Delta, the most apparent result was the large decrease in average patch size of the marsh (−85% at MNWR and −70% at JLNHPP) and corresponding increase in number of marsh patches. Marsh at JLNHPP increased from 306 to 825 patches and MNWR increased from 27 to 100 patches. Open water and modified vegetation increased in the Delta (Figure 4a,b). Scrub-shrub increased in the number of patches and average patch size.

Fewer trends in patch dynamics were evident for the Coastal Plain sites. Total number of patches increased from 140 to 295 at GBNERR and decreased from 89 to 70 patches at WBNERR. Forest, marsh, and non-vegetated patches increased in number substantially at GBNERR but decreased at WBNERR (Figure 4c,d). Scrub-shrub patches increased in number at both sites and increased in average patch size at GBNERR but decreased at WBNERR. Average mean size of forested areas increased slightly at both sites.

Buffer Analysis

For the analysis of the buffer area surrounding the sites, trends in 'natural' vegetation categories varied. Forested areas decreased in the Delta sites (especially at JLNHPP) and increased more than any other category in both Coastal Plain sites (Figure 5). Scrub-shrub increased substantially at MNWR (rate increasing from 1979–88) and GBNERR (rate increase 1979–88 but decreasing 1988–98). Marsh decreased at all sites except GBNERR, which showed no clear trend. Upland vegetation fluctuated slightly in the Coastal Plain sites and at JLNHPP.

Some trends were evident regardless of geographic location. All buffers showed increases of open water. The most extreme case was MNWR, which increased from 0.1% to 6.2% of the buffer covered by open water from 1955 to 1988. All sites also showed slight increases in modified open water and variable increases in modified vegetation. JLNHPP experienced the most change in modified vegetation (3.7% in 1958 to 19.2% in 1998). Urban areas increased except MNWR, which had no urban areas in its vicinity. Overall, 'modified' categories (including Agriculture, Modified Vegetation, Modified Open Water, and Urban) increased in all buffers at the expense of the 'natural' vegetation categories, with two exceptions. Our results show that agriculture at JLNHPP and GBNERR decreased.

Differences between Delta and Coastal Plain sites are illustrated by the dominance of certain states in the buffers. The main distinction was the dominance of agricultural areas in the Coastal Plain sites. There was a small amount of agriculture at JLNHPP in 1958, but none by 1978 (Figure 2b) and no agricultural areas at MNWR (Figure 2a). Although agriculture occupied a considerable portion of the total area of Coastal Plain sites in 1955, the trends differ between them. At GBNERR, agricultural land decreased greatly during the study period (39.2% to 0.01%), making it no longer prominent on the landscape, whereas at WBNERR, the area of agricultural use increased slightly.

DISCUSSION

Site Dynamics

The results show variation in land-cover composition and change among sites, but some general trends suggestive of RSLR are also apparent. As

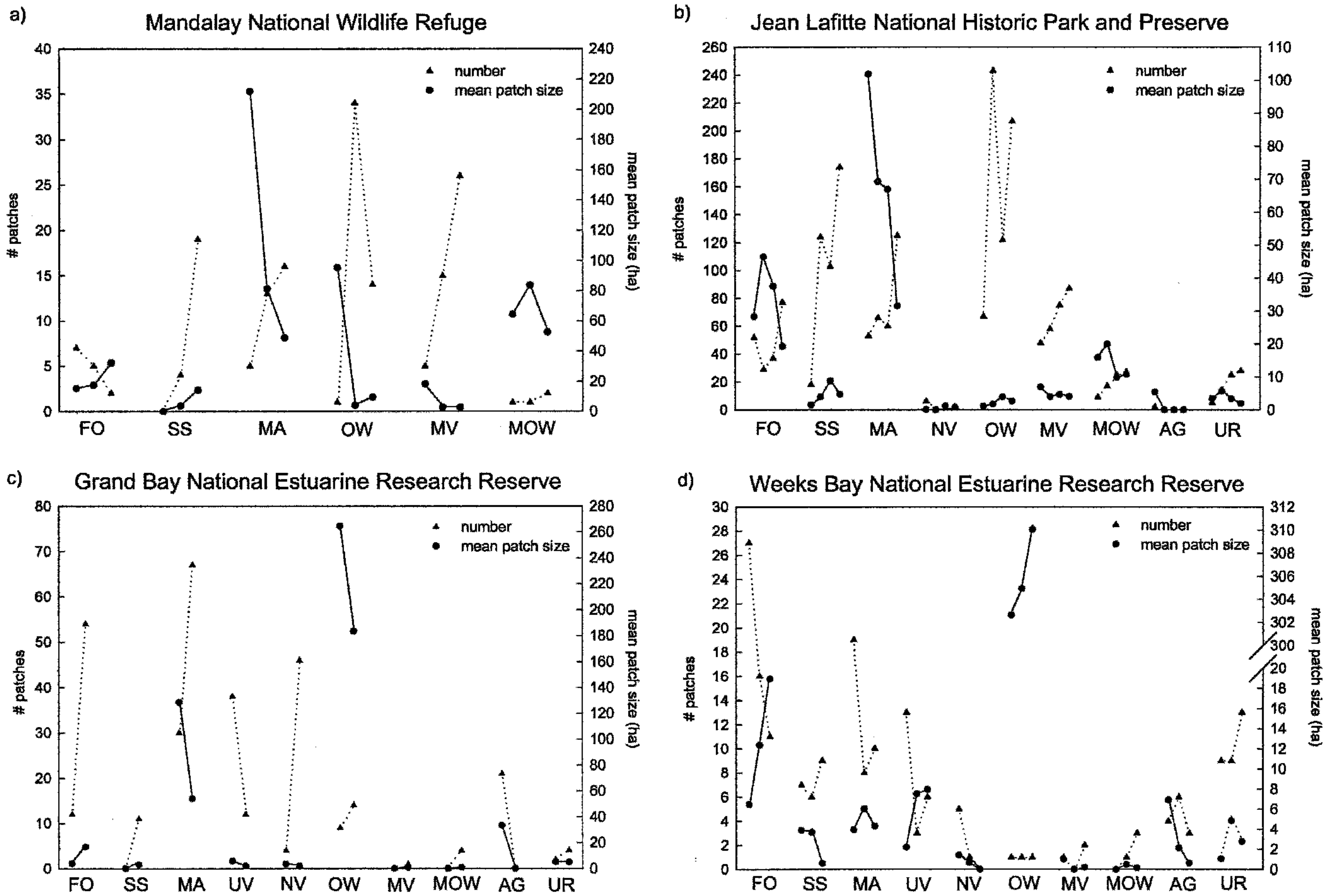

Figure 4. Patch dynamics for each category/site, which includes the total number of patches found (left y axis) and the mean patch size in hectares (right y axis). Within each category, the following years are represented: a) MNWR 1955, 1979, and 1988; b) JLNHPP 1958, 1978, 1988, and 1998; c) GBNERR 1955 and 1988; and d) WBNERR 1955, 1978, and 1988. For each year, patch number (▲) and mean patch size (•) are in order chronologically left to right. Each time series within a category is connected by a dotted line (number of patches) and a solid line (mean patch size).

predicted, both Delta sites lost marsh area and gained open water overall, although not all marsh loss could be attributed to RSLR. The clearest examples of marsh conversion to open water in the Delta were the appearance of new canals and expansion of existing canals during the study period. These results are supported by Turner (1997), who found canal-dredging and associated impacts to be the dominant forces behind conversion of marsh to open water. Canals further degrade marshes by serving as a conduit for salt-water intrusion at rates that prevent even establishment of salt-tolerant vegetation (Sasser 1986).

In addition to direct hydrologic modifications, some evidence of erosion was apparent at JLNHPP along the shore of Lake Salvador, and open water appeared in interior portions of the marsh at both Delta sites (Figure 2). The open water patches contributed to marsh fragmentation, as reflected in the patch analysis. Similarly, Barras et al. (1994) found that most marsh loss from 1978–1990 in the Terrebonne and Barataria Basins occurred within interior marshes. These interior marshes experience less vertical accretion than streamside or lakeside marshes (Hatton et al. 1983). However, floating marshes complicate the relationship among land-loss, accretion, and subsidence. Floating marshes (or *flotant*) may be able to offset the effects of subsidence temporarily by rising along with water levels. Conversely, because the vegetation is in contact with below-surface water (Swarzenski et al. 1991), it is potentially more exposed to salinity changes. The mechanism responsible for the observed changes in conversion of marsh to open water likely involves a combination of multiple factors (Day et al. 2000).

Although some instances of marsh conversion to open water were obvious, in all cases, the amount of marsh lost was greater than the amount of open water gained, suggesting that other processes are operating at the Delta sites. The majority of marsh lost was converted to scrub-shrub. The increase in

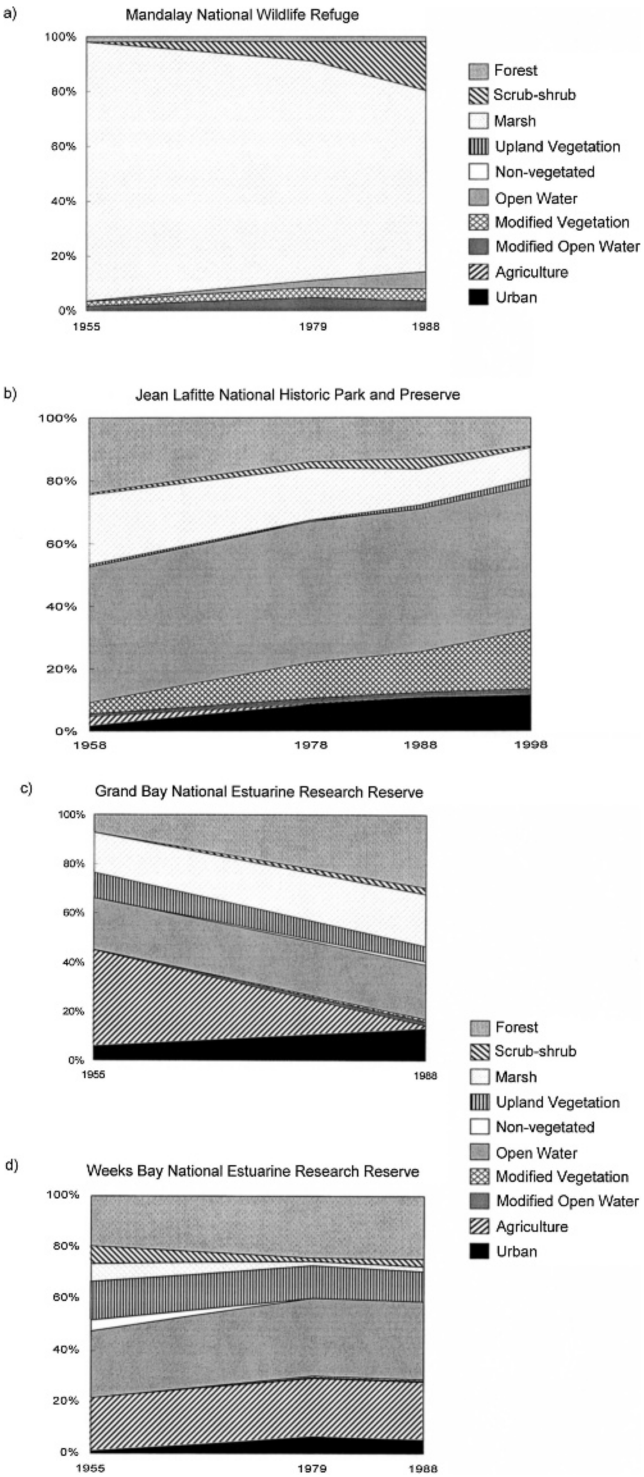

Figure 5. Area graphs showing the percentage of total area occupied by each category and change over time for the buffer zone surrounding each site: a) MNWR buffer, b) JLNHPP buffer, c) GBNERR buffer, and d) WBNERR buffer. The x-axis contains years included in the analysis, based on availability of National Wetlands Inventory data.

scrub-shrub was counter to our prediction that scrub-shrub would decrease in response to increased flooding and salinity associated with RSLR. Not only did scrub-shrub increase in the amount of land area covered, it also increased in both the number and average size of patches, indicating a spread of growing patches. These patches seemed to appear in the internal portions of the marsh at both sites. Accordingly, the marsh decreased in area and average patch size, but the number of patches increased, indicating marsh fragmentation. The rate of shrub expansion was highest at both sites in the 1978/79–1988 period (corresponding to the greatest rate of marsh loss) and then decreased at JLNHPP from 1988–1998. Data were not available after 1988 for MNWR, so it is unknown if there are trends that paralleled those of JLNHPP trends. However, periods of scrub-shrub expansion/contraction do not always coincide, even in areas of close proximity. Williamson et al. (1984) documented a decline in *Morella cerifera*, the dominant woody species in the *flotant*, from 1971 to 1980 at Salvador Wildlife Management Area, located on the opposite bank of Lake Salvador from JLNHPP. It is apparent from these studies that shrub populations in *flotant* marsh are dynamic, but the drivers and ecological consequences of these cycles are not well-understood.

The increase of woody species could be attributed to the absence of fire. Fire suppression could encourage seaward encroachment of woody species (Brinson et al. 1995). Although one would expect fire suppression to be counteracted by RSLR, the situation is complicated by the floating marshes, which can respond to some changes in water level by floating atop the water column. Their hydrology differs from attached marshes in that they have reduced surface flows (Swarzenski et al. 1991), increasing the chances of successful woody species encroachment. Another possible explanation for the shrub increase is the expansion of the exotic species Chinese tallow (*Triadica sebifera* (L.) Small), which comprises a large portion of the scrub-shrub patches in the marsh today (Battaglia, unpublished data). The longevity of scrub-shrub patches in *flotant* marshes is uncertain, however, because as woody species grow and expand into areas of herbaceous floating marsh that were previously supporting marsh vegetation, their weight lowers the floating mat. In cases where the mat is submersed for extended periods, this could eventually lead to shrub mortality (Williamson et al. 1984).

Trends in forest cover varied between Delta sites, increasing at JLNHPP and decreasing at MNWR. The rate of forest change in the Delta was lower than marsh and scrub-shrub, but this is to be expected, as trees have a longer turnover rate than herbaceous species or shrubs (Brinson et al. 1995). The loss of forested areas at MNWR occurred near the Gulf Intracoastal Waterway (Figure 2c), which could be a result of the increased exposure to flooding and/or salinity surges.

In contrast to our predictions, some of the marsh was replaced by open water in the Coastal Plain sites. However, the rate of open water gain was much lower in the Coastal Plain sites than in the Delta. The major obvious difference in these sites was the lack of a canal-network within the marsh. Most conversions of marsh to open water happened along shorelines and not in interior portions of marsh (Figure 2c), which could be storm-related erosion and/or associated sea-level rise.

The marsh decline in the Coastal Plain can also be attributed to replacement by forest, especially at WBNERR. There, the areas of marsh replaced by forest were smaller isolated patches surrounded by forest in 1955 (Figure 2d). This contributed to the decrease of marsh patches over the study period and corresponding amalgamation of marsh patches. These areas delineated as marsh could possibly have been abandoned agricultural areas in an early successional stage. There also appeared to be some forest encroachment near the forest-marsh boundary at GBNERR.

Our predictions concerning woody vegetation dynamics in the Coastal Plain were partially supported, yet the results in their entirety were not consistent with expected responses to changes in sea level. The scrub-shrub category showed no clear trend, increasing slightly at GBNERR but decreasing at WBNERR. The most dramatic change in the Coastal Plain sites was the conversion of former agricultural areas to forest. At WBNERR, there were two main agricultural areas in 1955, which were replaced by forest and upland vegetation by 1988 (Figure 2d). The reversion of land from agricultural areas to forest also contributed to the overall decrease in the number of patches at WBNERR, as fields surrounded by forest reverted back to forest.

Buffer Analysis

The buffer analysis revealed that, in some cases, land-use changes occurring outside of the protected areas were similar to land use/land cover inside sites. RSLR is one example that would impact all low-lying areas within a certain proximity to the shoreline, regardless of their land-use designation. In this case, open water increased at all sites, which could be a result of RSLR. MNWR presented the most extreme case of open water spread. This dramatic increase of open water can be attributed mostly to widening of the Gulf Intracoastal Waterway, which experienced high boat traffic with resulting wave action and erosion of surrounding marsh areas (Louisiana Department of Natural Resources 2004) during the study period. The shoreline of the Intracoastal Waterway at MNWR lost on average ~4.0 m/year from 1944–1983 (May and Britsch 1987).

The MNWR buffer area was unique in the group of study sites in that the buffer was largely dominated by marsh, was lacking urban areas and agriculture, and is in a relatively remote location. Without the influence of anthropogenic land-use changes in the buffer, there were more similarities between the processes operating inside and outside the refuge borders, such as the replacement of marsh by scrub-shrub. However, the analysis at MNWR was limited by the exclusion of the area north of the Gulf Intracoastal Waterway, which has some development. Thus, developed land-use categories were probably underestimated in our analysis of this site.

In the Coastal Plain sites, trends in the buffers were dictated largely by agricultural patterns, whereas the buffer at JLNHPP did contain limited agriculture in 1958; it was never extensive during the study period, and all fields were abandoned by 1988. The agricultural trends of the Coastal Plain buffers were similar to those found within the sites, although agriculture was even more pervasive outside of the protected areas. At the GBNERR buffer, agriculture, which made up a large portion of the buffer in the 1950s, was largely absent by 1988 and replaced by forest. Abandonment of agricultural fields may have been the result of inadequate yields related to high salinity. There have been no active attempts to restore agricultural areas to forest, and we consider the appearance of forested areas to be the result of natural succession. At the WBNERR buffer, agriculture increased slightly during the study period and remained a considerable portion of the buffer in 1988. Similar results were found in a rural area on the North Carolina Coastal Plain, where ~95% of the loss in wetland area could be attributed to land conversion for forestry and agriculture, with urbanization, road construction, and rural development accounting for smaller percentages (Cashin et al. 1992).

Not surprisingly, all buffers that are not located in protected areas experienced an increase in urbanization, which increased from 4–10% in area during the study period at the expense of 'natural' categories. The exception was MNWR. The greatest increase of urban area was at JLNHPP. The expansion of residential neighborhoods from the west bank of New Orleans contributed to the increase in urban development in the northeast portion of the buffer. New levees built to protect these urban areas led to the conversion of surrounding natural vegetation categories to modified vegetation, which has undergone hydrologic modification and is therefore more vulnerable (Kennish 2002). In the JLNHPP buffer, modified vegetation covered 0.037% of the land area in 1955 and 19.2% in 1998. The GBNERR and WBNERR buffers had slight increases in urban expansion, but due to their location in relatively rural areas, they are not experiencing urban growth as quickly as the JLNHPP buffer.

Limitations of NWI Data

Although NWI data can be used to determine trends in wetland and upland categories in this study, the availability of NWI data (both hard-copy maps and digital data) is somewhat limited. The production of NWI maps is a multi-stepped process, requiring manual polygon delineation by trained photointerpreters, digitization and rectification, and then a number of quality-control checks. Therefore, funding and resource constraints allowed for only selected years and areas to be mapped. The years that are available for one area may not be the same as another, so the temporal comparison among multiple sites is restricted. In this study, data from only one site (JLNHPP) were available for the 1990s, although others are currently being created (L. Handley, pers. comm.).

There are also inherent limitations when using aerial photography for NWI data creation. The photointerpreter's ability to delineate habitat types accurately differs depending on the scale and emulsion of the original photography. Earlier black and white photography tends to be more difficult to interpret than color infrared photography taken after 1980 as part of the National High-Altitude Photography Program (Tiner 1990). However, even with more recent photography, some aspects of photointrepretation remain challenging. Most inaccuracies in NWI habitat delineation are a result of features undetectable in the photography, which has led to misclassification of temporarily flooded wetlands (see Tiner 1997, Kudray and Gale 2000). However, in this study, all flooding regimes are aggregated into one category, and we are instead looking at the structure of vegetation, which is more easily distinguishable in the aerial photography.

In addition, the methods used to create the NWI data used in this study prevent detailed spatial analysis. Multiple years of NWI data were not registered or tied to the same basemap, and therefore, maps cannot be overlaid to account for conversion of one category to another quantitatively. This limits the applicability of NWI data for producing change matrices in order to determine directional migration of vegetation or shoreline changes over time. In order to remedy this deficiency, all years of original aerial photography and polygons for each site would need to be corrected geometrically based on one image.

Conclusions

This study has documented many changes in land use/vegetation cover for protected areas in coastal transitions representing two contrasting systems. It appears that the marsh is changing the most in terms of the amount and rate of conversion to other states. In the Delta, some of the marsh is being converted to open water, supporting prior predictions of RSLR impacts. The conversion of estuarine marsh to open water and scrub-shrub in the Delta is representative of national trends from the 1950s to the 1970s (Wilen and Frayer 1990), the 1970s to the 1980s (Dahl and Johnson 1991), and the 1980s to 1990s (Dahl 2000). Scrub-shrub and water patches appear to be emerging in interior pockets of marsh, although the amount of scrub-shrub is decreasing at JLNHPP during the 1990s. Spatial analysis of multiple dates is warranted. Future spatially explicit analyses incorporating environmental parameters, such as salinity levels and elevation, could shed light on aspects of ecological changes in coastal systems (see Moorhead and Brinson 1995).

In addition to fragmentation of the marsh by water and shrubs within the protected areas, the surrounding land is increasingly being modified by human activities. Escalating urbanization and associated hydrologic modifications increasingly isolate protected areas from natural vegetation and potential migration sites. The boundaries of these protected areas are mostly static, unlike the land-cover changes occurring internally and externally, rendering these areas highly vulnerable to changes such as sea-level rise. Finally, our results highlight the need for an improved understanding of the underlying drivers and mechanisms of the observed land-cover change.

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