ASSESSMENT OF ACREAGE AND VEGETATION CHANGE IN FLORIDA'S BIG BEND TIDAL WETLANDS USING SATELLITE IMAGERY

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ABSTRACT

Fluctuations in sea level and impending development on the west coast of Florida have aroused concern for the relatively pristine tidal marshes of the Big Bend. Landsat Thematic Mapper (TM) images for 1986 and 1995 are processed and evaluated for signs of change. The images cover 250 km of Florida's Big Bend Gulf Coast, encompassing 160,000 acres of tidal marshes. Change is detected using the normalized difference vegetation index (NDVI) and land cover classification. The imagery shows negligible net loss or gain in the marsh over the 9-year period. However, regional changes in biomass are apparent and are due to natural disturbances such as low winter temperatures, fire, storm surge, and the conversion of forest to marsh. Within the marsh, the most prominent changes in NDVI and in land cover result from the recovery of mangroves from freezes, a decline of transitional upland vegetation, and susceptibility of the marsh edge and interior to variations in tidal flooding.

1.0 INTRODUCTION

The importance of wetlands has only recently been realized. Until the 1960s, wetlands were viewed as wastelands and the 'reclamation' of wetlands to more useful purposes was the driving force behind the dredge and fill of many of these sensitive and productive ecosystems (Tiner, 1984). Estuarine wetland losses have been particularly pronounced in the Gulf of Mexico, including substantial losses in Florida, representing a considerable loss to commercial and recreational fisheries (Frayer and Hefner, 1991). While tidal wetlands play a role in the trapping, transformation, and filtering of detritus, nutrients, sediment, and pollutants (Durako et al., 1985), a singular public benefit derived from intertidal wetlands is the buffering of storm surge, ameliorating the effects of flooding and erosion in coastal communities (Tiner, 1984). Growing public awareness and concern for the effect of sea-level rise in coastal areas has created a demand to know more about the responses of coastal environments to sea-level fluctuations (Wanless, 1989).

An effort has been made to document the extent of wetlands in the U.S. and to evaluate change in those wetlands over time. The National Wetlands Inventory (NWI) has produced information on trends within the U.S. from the 1780s to the 1980s (Dahl, 1990). The NOAA Coastal Change Analysis Program (C-CAP) is developing methods for acquiring current land cover information and characteristics in the coastal zone (Dobson et al., 1995). The program employs Landsat satellite imagery to describe coastal habitat change on a 1- to 5-year cycle and emphasizes changes in land cover categories as a means to detect change. Guidelines and protocol for image selection, processing, classification, and change detection are delineated in Jensen et al. (1993) and Dobson et al. (1995).

Both NWI and C-CAP compare changes in land cover categories to identify and quantify change, which carries the inherent difficulty in distinguishing true change from change attributable to misclassifications. An alternative is to examine changes in canopy and biomass, as represented by vegetation indices, which provide

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indicators of stress and vegetation change that are not identified in land cover classifications (Gross et al., 1990). The vegetation index approach provides a means of expanding the concept of change beyond land cover types and eliminates change as an artifact of misclassification.

This paper examines the Big Bend coast with two objectives. We document areal extent of estuarine marsh over a 9-year period, 1986 to 1995, using an unsupervised classification, and document change with a vegetation index from Landsat TM. The vegetation index, a surrogate measure of biomass, identifies patterns and trends in vegetative health and is discussed in the context of processes and disturbances. Field observations were conducted between 1992-1995 to verify the interpreted imagery.

2.0 STUDY AREA

The tidal marshes of Florida's Big Bend extend 250 km along the Gulf Coast from Wakulla County to Pasco County (Figure 1). The area is relatively pristine; however, rapid urbanization and intensive land use are encroaching from all directions. The region is on a boundary between climatic zones 8 and 9 (Cathey, 1990), with tropical species such as mangroves found in the southern counties. The shoreline has a low erosion rate, less than 1 m/yr. (Dolan et al., 1985). The extremely low-elevation gradient makes the adjacent uplands particularly susceptible to changes wrought by sea-level rise, which may cause migration of coastal habitats (Williams et al., in press; Kurz and Wagner, 1957). Documentation of habitats and habitat zonation can provide resource managers with information to establish appropriate land management strategies and provide data suitable for an ecosystem approach (Haddad and Joyce, 1997).

The Big Bend tidal marshes form a belt typically 3 to 4 km wide along the Gulf Coast of Florida. The area has a semi-diurnal tide with a range of about 1 m. The marsh occupies a karstified limestone shelf with a veneer of sediment 0 - 3 m. This section of the coast is directly exposed to the Gulf of Mexico and is considered to be a low or no-energy coastline. Although it lacks the barrier islands common elsewhere along the Gulf, the outer shelf is wide and shallow, thus dampening the effect of tides and currents (Stout, 1984). Average coastal salinities, at 20 to 25 ppt, are below open-marine salinities (30 to 35 ppt) because of the significant freshwater flow from the Floridan Aquifer (Orlando et al., 1993). The coast has a relatively low sea-level rise, an average of 1.5 mm/yr. since 1939, with a 2-cm increase in tide range since early 1960s (Stumpf and Haines, in press).

3.0 METHODS

The Landsat TM used in this study includes seven bands, six of reflected light and one of thermal infrared. Each image has a width of about 180 km. Two images are required to cover the Big Bend coast. Our analysis is based on images from April 1986 and April 1995, with complementary winter images employed in the classifications (Table 1). Water levels for the imagery fall within the full tidal range (Figure 2).

Table 1. Landsat TM images and water levels.

Landsat coordinates path/row	Location	Date	Water level at Cedar Key
17/40	Tallahassee, FL	01/23/85	0.40
17/40	Tallahassee, FL	04/25/86	1.05
17/40	Tallahassee, FL	11/29/93	0.71
17/40	Tallahassee, FL	04/09/95	1.46
18/39	Cedar Key, FL	01/16/85	1.18
18/39	Cedar Key, FL	04/25/86	1.29
18/39	Cedar Key, FL	01/12/95	1.26
18/39	Cedar Key, FL	04/02/95	0.54

The images were rectified to UTM (Universal Transverse Mercator) Zone 17 in WGS 1984 (World Geodetic System 1984) coordinate system, which is identical to NAD 83 (North American Datum 1983) at this scale. Rectification was based on ground control points obtained by GPS (Global Positioning System) units with a stated accuracy of $\leq$ 10 m. Approximately two dozen points were collected for each image, half for control in the rectification; the rest were used for accuracy validation. The final images meet national map accuracy standards for 1:25,000 (90% within 20 m), and co-registration between images at $\pm$ 1 pixel.

The digital counts in the image were transformed to reflectance using the calibration that came with the files and the equations of Price (1987) and Markham and Barker (1985)

$$R(\lambda) = \frac{\pi L(\lambda)}{E_0(\lambda)(1/r^2)\cos(\theta_0)} \qquad \mathbf{1}$$

where λ is the band; the radiance, L, is determined by

$$L = G * N + BIAS \qquad \mathbf{2}$$

E_0 is the solar constant, r is the normalized earth-sun distance, θ_0 is the solar zenith angle at the image center, N is the digital count, G is the calibration slope, and BIAS is the calibration offset for zero radiance.

An atmospheric correction was performed using subtraction from the bands. Because this region has black-water lakes and rivers, water can be found that has negligible reflectance in all bands. A dark-object subtraction was used, with the reflectance of the darkest water being the value subtracted. The correction decreases with wavelength (Chavez, 1989).

The normalized difference vegetation index (NDVI) was used to quantify the green biomass. This is defined as

$$NDVI = \frac{R(4) - R(3)}{R(4) + R(3)} \quad \textbf{3}$$

where 4 and 3 are near-infrared and red bands, respectively.

Unsupervised classification was performed using an ISODATA clustering algorithm on bands 2, 3, 4, and 5 from both seasons simultaneously. An unsupervised classification is preferred because it projects no preset criteria on the distinctions between land cover categories. A simple mask of the estuarine-influenced area is created using the contrast at the upland boundary with wetness on one side and high biomass on the other side. This is a regionally specific approach to help distinguish the two zones, but we feel variations on this tactic may be applied in other regions with reasonable success. Upland areas include forest, scrub, palustrine forest and emergents, grassland, and developed/barren areas. Estuarine emergent includes the salt barrens and all emergent communities in the intertidal zone. Estuarine scrub includes both the mangrove fringe and the marsh-interior scrub transition zone. The full classification follows the Level II class separation within the C-CAP protocol (Dobson et al., 1995). Class validation was conducted with a field laptop and GPS for the 1995 imagery, and with aerial photography and NWI maps for the 1986 imagery.

Changes in NDVI values were calculated on a pixel-by-pixel basis for each pair of images, 1986-1995 north and 1986-1995 south. Differences exceeding 0.1 NDVI are considered significant for change analysis based on reflectance values per count in bands 3 and 4. The vegetation index provides a continuous data set of derived values representing relative biomass. The difference in NDVI presents an unbounded and unmodified determination of the location and spatial distribution of vegetative change.

The classifications and NDVI change are calculated for the full images, but the analysis of change is focused specifically on the intertidal area and on a 2-km wide upland buffer, to emphasize vegetation change in the immediate vicinity of the intertidal zone. All areas subject to tidal flushing during the normal tidal regime, up to and including the transitional upland edge, are included as intertidal categories in the analysis, which allows documentation of changes at critical vegetation zone boundaries (Kurz and Wagner, 1957). Counties on the Big Bend marsh coast included in this analysis are: Wakulla, Jefferson, and Taylor in the north, and Dixie, Levy, Citrus, Hernando and Pasco in the south (Figure 1).

Although the USDA climatic zone boundary (Figure 1) occurs just south of Cedar Key, mangroves have historically been documented on USGS topographic maps as far north as the Steinhatchee River. A warmer, tropical environment prevails south of the river and under prolonged, favorable conditions, mangrove stands may thrive. North of the river, mangrove stands are virtually nonexistent. Because the evidence for climatic zone 9 continues northward in the coastal marshes, the analysis has been divided between two regions, north and south of the Steinhatchee River (Figure 1).

4.0 RESULTS

4.1 ACREAGE

Classification of two Landsat TM images for the Big Bend Gulf Coast produces a tidal-marsh class within the 85% accuracy standard (Table 2). All classes have an allowable error that meets the 95% confidence interval for the expected map accuracy of 85% (Fitzpatrick-Lins, 1981). The error range for the intertidal category is shown in Figure 3. This is well-within the range displayed by other sources.

Table 2. Intertidal marsh area from various sources, Wakulla to Pasco County

Source	Intertidal marsh (acres)	North/South Counties (acres)
NWI (Lewis et al., 1982)	163,000	48,250/115,250
NWI (Field et al., 1991)	141,100	44,200/96,900
1986 Landsat TM classification	155,000	50,000/105,000
1995 Landsat TM classification	155,000	48,000/107,000

4.2 VEGETATION INDEX

Change in NDVI is documented in Table 3. The vegetation index produced for each year is compared pixel by pixel and all differences > 0.1 NDVI are considered an indication of change in vegetative characteristics. Changes in the index show a general increase in vegetation in 1995 across both the north and south regions (Table 3). A closer examination shows distinct patterns (Figure 4).

Decreases in marsh NDVI in the north region occur in linear and clumped patterns at the gulf edge and at the inland boundary of the marsh. The decrease in marsh NDVI at the gulf edge is due in part to higher water levels in 1995, which partially obscure the low marsh, *Spartina alterniflora*. A prominent NDVI increase occurs at the salt barrens near St. Marks (Figure 1). The increase in biomass in the salt barrens in 1995 follows a particularly wet season in which upland areas were extensively flooded for prolonged periods. Salt barrens occur only centimeters above MHHW (Raabe et al., 1996), receiving little flushing by tidal waters while accumulating salts (Stout, 1984; Kurz and Wagner, 1957). The recent increased precipitation events may have flushed the salt barrens sufficiently to allow for fresh growth of many salt-tolerant species already present in the salt barrens. Other NDVI increases occur in linear patterns, suggesting recovery from earlier storm deposits.

Generally, the decrease in marsh biomass in the south is similar to that in the north at ~15% and occurs primarily at the marsh interior, sporadically at the gulf edge, and at freshwater sources near the rivers, including the Suwannee River tidal delta. The southern counties exhibit an increase in marsh biomass, covering > 20% of the intertidal area. The increase is found primarily at the gulf and creek edges due in part to lower water levels in 1995, recovery from 1985 storm deposits, and from mangrove recovery. Most of the vegetation index increase occurs within the water and estuarine emergent land cover zones. Changes in NDVI within the 2-km buffer are provided to help identify adjoining land cover changes.

Table 3. Percentage of area with > 0.1 NDVI decrease or increase

	Percent of Area with NDVI Difference > 0.1			
Region	North of Steinhatchee River		South of Steinhatchee River	
Intertidal Area	~ 57,000 acres		~ 148,000 acres	
1995 Category	NDVI decrease %	NDVI increase %	NDVI decrease %	NDVI increase %
Water	6.5	0	4.5	7
Marsh	6.5	12	6.5	11
Estuarine Scrub	1	4	1	5
Tidal Upland	0	0.5	2	2.5
Total	14	16.5	14	25.5

2 km Upland Buffer	~ 95,800 acres		~ 157,900 acres	
1995 Category	NDVI decrease %	NDVI increase %	NDVI decrease %	NDVI increase %
Upland	6	10	10	11

5.0 DISCUSSION

5.1 ACREAGE

Estimates of tidal-marsh within the same technical approach are closer than between land cover classification techniques (Figure 2). NWI estimates are 163,000 acres (Lewis et al., 1985) and 141,000 (Field et al., 1991). The origin of error and inconsistencies between estimates include technical, environmental, and data source differences. The variability is indicative of the difficulty in obtaining precise and reliable estimates of marsh acreage and change detection using inconsistent data sources, processing, and areal-calculation techniques.

Environmental variables that will affect areal estimates include water level at time of photography or satellite overpass, recent precipitation and freeze events, season, and other changes in vegetation quality. Technical variations include differences between boundaries set for the estuarine environment, class inclusion such as sawgrass meadows and salt barrens, the classification of transition zones, and the classification of partially colonized substrate such as limestone and oyster bars. Source differences include resolution and spectral quality of aerial photography and satellite imagery, misclassification of habitats with variable signatures, aerial estimation techniques, and other biases inherent in each approach (Nichols, 1994; Federal Geographic Data Committee, 1992).

Whether the new estimates represent actual gains or losses in marsh area is not ascertained within the current analysis. However, given the consistent approach to image processing and classification, it will be possible to identify losses and gains in marsh area using the satellite imagery. Three additional analyses will be used to help refine and quantify changes identified in field observations: regional analysis over smaller areas, a classification/ NDVI matrix, and gradient analysis.

5.2 WATER LEVEL

Varying water levels at the time of data acquisition may affect the spectral response of low-lying marshes. A comparison of image overlap at the Steinhatchee River between April 2 and 9, 1995, indicates a 13% loss of marsh acreage with an increase of 0.92 m in water level, or a loss of ~1.4% tidal marsh per decimeter increase. Jensen et al. (1993) estimated water level to change wetland area determination by 1-2% per 0.1 m in South Carolina between years. Our results are consistent with their findings.

Our estimates of marsh area exhibit less variation than between previous estimates and do not show the full extent of water-level influence for two reasons. First, the majority of the estuarine intertidal environment on the Big Bend coast is high marsh, which is identifiable at most water levels. The only areas subject to confusion within variable water levels are the narrow tidal creeks and the low-marsh fringe at the gulf and creek edges. Secondly, classifications in this analysis employed two-season imagery, thus eliminating some portion of the water-level effect (Figure 2). The 1986 south region shows the least variability in water level between the winter and spring imagery, compromising our ability to detect the low-marsh environment for 1986 south.

An issue related to water level is ponding and actual changes in “wet” marsh areas from year to year. Interior ponding was apparent in the 1986 north imagery under low-water conditions, but these features were absent in the 1995 image with higher water levels. Approximately 3,500 acres of creek interiors and creek edges, flooded in 1986, were later exposed at the higher water levels in 1995. Most of the area is along the interior reaches of the tidal creeks. A feasible explanation is the obstruction of tidal creeks with storm debris, as in 1985, and the subsequent impoundment of water until another high-water event, as in 1993, restores tidal flow to the area. A similar number of acres in the emergent zone were flooded in 1995, but the area is primarily at the gulf edge as would be expected with the slight increase in water level. Fluctuations such as this will tend to mask actual alterations within the intertidal zone.

5.3 DISTRIBUTION AND PATTERNS OF NDVI CHANGE

Three constraining elements of a classification include water level, the scale of the imagery, and the scale of the changes occurring in the environment. At the 30-m resolution of the Landsat TM imagery, changes within the tidal boundary may be difficult to identify. A 1-2 m gain or loss at the shoreline will be lost in the coarseness of the imagery. However, we suspect that general alterations to the health and productivity of the marsh may be identified with the vegetation index and may serve as effective indicators of the changes taking place.

Although the quantification of acreage has not proved to be a significant source of information for change detection at the current level, the distribution and pattern of NDVI change suggests there are regions and specific sites prone to disturbance with differing rates and types of recovery and succession. These may serve as signals to zones that should be monitored and the type of monitoring that should be conducted.

Field reconnaissance suggests that NDVI decreases of interior marsh biomass occur in areas subject to storm deposits, fire, and at the upland-to-marsh transition zone. Kurz and Wagner (1957) documented the upland-to-marsh successional phase in the St. Marks area, and Williams et al. (in press) show a similar decline in relict forest islands at Waccasassa Bay. A typical site will exhibit several mature sabal palms with an understory of black needlerush, sawgrass and salt-tolerant scrub. Coastal hammocks occupying scattered limestone highs throughout the marsh have been exposed to increased tidal flushing with rising sea level. The trend in morbidity, as documented during a five-year study by Williams et al. (in press), is attributable to tree damage from severe storms and a gradual decline in health and regeneration due to rising sea level. Decreases in marsh NDVI at the marsh interior may also be related to the loss of marsh following the deposition of debris during the 1993 No-Name Storm. Field observations record incomplete recovery of these areas as well as indications of burns.

Vegetation index change at the gulf edge is also attributable to water level-fluctuations. However, if water levels were the determining factor, we should see a larger portion of tidal wetlands in the north image with high biomass in 1986. Instead, we see a shift in the areas flooded between 1986 and 1995 with varying changes in biomass at the gulf edge. A portion of the increased biomass documented in 1995 may be attributable to the colonization by cordgrass of newly redistributed sediments along the gulf edge as observed in field reconnaissance. Although the slumping may signify a loss of the high marsh, the subsequent colonization suggests that the marsh edge will show an expansion seaward in the following years.

Some regions of increased NDVI correspond to past areas of mangroves. Topographic maps and aerial photography from the 1950s show extensive mangroves in this area. In the 1980s a series of freezes occurred (1983, 1985, 1989) that resulted in high mangrove morbidity. The 1985 freeze occurred only months before the 1986 satellite overpass, subsequently eliminating all signs of estuarine scrub along the gulf in that image.

Field reconnaissance from 1993 to 1995 shows healthy regrowth of the mangroves, up to 3 m in height in Pasco County and ~ 1 m high at Ozello and in Levy County. The increase in NDVI of the estuarine scrub class, however, is barely noticeable (Figure 4). The increase of NDVI in the estuarine scrub category is no different from that observed in the north and cannot be attributable to mangroves alone. Mangroves occur at the same elevation at the gulf edge as cordgrass. Consequently most of the area in the 1986 imagery was identified as intertidal emergent or water, and in 1995 imagery, a majority is also interpreted as intertidal emergent with a low canopy cover of estuarine scrub. Thus, although the vegetation index shows a “greening” along the gulf in the southern counties, many mangrove stands may continue to be classified as tidal emergent until sufficiently large stands are developed.

Analysis of the upland buffer zone indicates areas of NDVI increase exceed those with a decrease only in the northern counties. Most of this change occurs in regions where the timber industry is extensive. Land-use activities within the upland buffer are primarily pulp wood production with scattered, small-scale farming operations. Apart from the regrowth in timber production areas, the upland buffer presents a balance between biomass loss and gain.

6.0 CONCLUDING REMARKS

A classification of imagery provides only one perspective on land cover change. Land cover categories are an intellectual construct and as such contain artificial boundaries by placing the data into discrete categories. A measure of biomass, NDVI, allows us to examine variations in vegetative health and productivity without the confounding elements of a thematic coverage. The vegetation index presents a continuous data set showing relative biomass. The difference between the 1986 NDVI and the 1995 NDVI is distinctive, highlighting trends in vegetation change along the coastline. Some changes occur on cyclic short-term recovery cycles; these changes are representative of the natural variability and trends in the intertidal zone.

Landsat TM imagery provides information on vegetation change at the scale of 30 m. It cannot, however, distinguish the subtle changes in species composition or the variable cover of marsh and upland within the transition zone. Thus, current image interpretation only suggests that changes in biomass are associated with hammock decline or mangrove morbidity. Future analysis will secure more precise data on shoreline change, marsh acreage, and environmental variations by examining the coast on a regional basis and by gradient analysis.

Current debates over the validity of utilizing satellite imagery for wetland inventory and monitoring focus on issues of spatial and spectral resolution (Federal Geographic Data Committee, 1992). It is generally agreed that a combination of bands including Landsat TM band 5 can identify many wetlands and changes in wetlands. However, accurate wetland classifications are not achievable using satellite imagery alone (Federal Geographic Data Committee, 1992). We have presented here the preliminary results of unsupervised classification and vegetation index change in the context of natural processes on the Florida Big Bend Gulf Coast. The classification at this point is generalized to give an overview of the region. We have also delineated several components of the estuarine intertidal zone that can be evaluated in more depth. It is our belief that consistent and refined image processing, regionally specific constraints, and a ground-truthed familiarity with the region and the natural processes at work will permit accurate and valid interpretations of coastal wetlands and change detection using satellite imagery.

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