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Historical Change of Seagrasses in the Mississippi and Chandeleur Sounds

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HISTORICAL CHANGE OF SEAGRASSES IN THE MISSISSIPPI AND CHANDELEUR SOUNDS

by

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ABSTRACT

HISTORICAL CHANGE OF SEAGRASSES IN THE MISSISSIPPI AND CHANDELEUR SOUNDS

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Seagrasses are important coastal resources facing numerous stressors, and seagrass losses have been documented from local to global assessments. Under the broad theme of habitat loss and fragmentation, a study of historical change in total area and landscape configuration of seagrasses in the Mississippi and Chandeleur Sounds was conducted. Mapping data was collated from a multitude of previous projects from 1940 to 2011.

Comparisons of seagrass area among various studies that used different mapping methods can result in overestimation of area change and misleading conclusions of change over time. The vegetated seagrass area (VSA) data were generalized to a common resolution for further analysis. Spatial configuration of the seagrass landscape was examined through: (1) an exploratory spatial data analysis using seagrass patch size distribution and hot spot analysis, and (2) a core set of seagrass landscape FRAGSTATS metrics and a principal component analysis to identify major pattern. This study demonstrated a comprehensive analysis across spatial and temporal scales and used multiple landscape indices (from habitat, species composition, VSA, patch size distribution, to spatial configuration at patch- and landscape-levels) to provide insights on the pattern and dynamics of the seagrass landscape. A conceptual model of seagrass

landscape change based on two principal factors, overall landscape lushness and continuity, was proposed for the Mississippi Sound.

Overall the study area lost seagrasses with contracted spatial extent over the 71-year period, ostensibly due to loss or reduction of protective island barriers and reductions in water quality. The seagrass landscape in the Mississippi Sound exhibited signs of area loss and fragmentation as far back as the 1940-1950s. The landscape in the 1970s was characterized by loss of habitat, loss of seagrass species, the lowest VSA, a faster rate of decline and a higher loss in VSA than before 1970, a low proportion of large-sized patches and their low contribution to VSA, a reduced intensity of hot spots, and a high degree of fragmentation. Recovery of seagrass occurred during the 1980s and 1990s, with the landscape exhibiting characteristics of a more contiguous and more vegetated condition throughout the early 2000s.

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DEDICATION

I wish to show my deepest appreciation to my family and friends for their unceasing encouragement and support. A heartfelt thank is extended to my husband, Bao Le Hung, who has learnt how to handle me in stressful time toward the end of my graduate study.

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LIST OF ABBREVIATIONS

AL	Alabama
ANOVA	Analysis of Variance
AREA	Patch area
AREA_AM	Area-weighted mean patch area
CIRCE_AM	Area-weighted mean patch related circumscribing circle
CIRCLE	Patch related circumscribing circle
DIV	Landscape Division Index
ED	Edge Density
EDA	Exploratory Data Analysis
ENN	Patch Euclidean nearest neighbor distance
ENN_AM	Area-weighted mean patch Euclidean nearest neighbor distance
ESDA	Exploratory Spatial Data Analysis
ESRI	Environmental Systems Research Institute
FGDC	The Federal Geographic Data Committee
FRAC	Patch fractal dimension index
FRAC_AM	Area-weighted mean patch fractal dimension index
GIS	Geographic Information Systems
GNDNERR	The Grand Bay National Estuarine Research Reserve

GOM	Gulf of Mexico
GPS	Global Positioning System
GYRATE	Patch radius of gyration
GYRATE_AM	Area-weighted mean patch radius of gyration
LC	Large and complex patches
LISA	Local Indicators of Spatial Association
LPI	Largest Patch Index
LSI	Landscape Shape Index
MS DEQ	The Mississippi Department of Environmental Quality
MS	Mississippi
MSC	The Mississippi and Chandeleur Sounds
MSS	The Mississippi Sound
NCI	The northern Chandeleur Islands
NP	Number of patches
NWRC	The National Wetland Research Center
PAFRAC	Perimeter/ Area Fractal Dimension
PARA_AM	Area-weighted mean patch perimeter/area ratio
PC	Principal Component
PCA	Principal Component of Analysis
PD	Patch Density

PERIM	Patch perimeter
PSGH	Potential Seagrass Habitat
SA	Spatial Autocorrelation
SAV	Submerged Aquatic Vegetation
SS	Small and simple patches
TE	Total Edge
TIGER	Topologically Integrated Geographic Encoding and Referencing
USGS	The United States Geological Survey
UTM	Universal Transverse Mercator
VSA	Vegetated Seagrass Area
WGS	World Geodetic System

CHAPTER I – GENERAL INTRODUCTION

Seagrass Definition, Importance, and General Distribution

Seagrasses are submerged aquatic angiosperms that are found in shallow, coastal marine environments worldwide with the exception of Antarctica (Green and Short 2003). All seagrasses are monocotyledons, encompassing 12 genera in the 6 families of Zosteraceae, Cymodoceaceae, Posidoniaceae, Hydrocharitaceae, Ruppiaceae, and Zannichelliaceae (den Hartog and Kuo 2006). There are about 60 - 64 species of seagrasses worldwide (Hemminga and Duarte 2000) even though the exact number is debatable. Species found in the Mississippi and Chandeleur Sounds (MS-C) and the northern Gulf of Mexico (GOM) are classified biogeographically into the Caribbean seagrass flora and include *Halodule wrightii* Asch. (shoal grass), *Halophila baillonis* Asch. (clover grass), *Halophila decipiens* Ostenf. (paddle grass), *Halophila engelmannii* Asch. (star grass), *Halophila johnsonii* Eiseman (Johnson's seagrass), *Ruppia maritima* L. (widgeon grass), *Syringodium filiforme* Kütz. (manatee grass), and *Thalassia testudinum* Banks ex König (turtle grass) (Phillips and Meñez 1988). Establishment of seagrasses requires a marine environment, adequate rooting substrate, sufficient immersion in seawater, and good illumination (Hemminga and Duarte 2000). Most seagrass meadows are comprised of monospecific patches, especially in temperate regions, but can include a mixture of species, especially in subtropical and tropical regions (Hemminga and Duarte 2000). Seagrasses provide key ecosystem services via carbon and nutrient cycling, sediment stabilization, nursery habitat and refugia for invertebrates, finfish, and shellfish, and a food source for birds and marine endangered species (e.g., dugong *Dugong dugon*, manatees *Trichechus* spp., and green turtle

Chelonia mydas) via trophic transfers to adjacent habitats (Orth et al. 2006, Waycott et al. 2009). Along with coastal ecosystems in general, seagrasses are subjected to numerous stressors such as direct physical damage to seagrass habitats, nutrient and sediment pollution, the introduction of exotic species, and global climate change (Orth et al. 2006, Lirman et al. 2008).

Seagrass Decline

The loss of seagrasses has been documented from local and regional studies to worldwide assessments (Kemp et al. 1983, Orth and Moore 1983, Robblee et al. 1991, Duke and Kruczynski 1992, Thayer et al. 1994, Green and Short 2003, Waycott et al. 2009). Waycott et al. (2009) estimated that globally, seagrass decline accelerated from a pre-1940 median rate of 0.9% per year to 7% per year since 1990. In the United States, Kemp et al. (1983) and Orth and Moore (1983) reported that declines in submerged aquatic vegetation (SAV), including *Zostera marina* L. (eelgrass), in Chesapeake Bay started in the 1960s, and then accelerated during the 1970s to the present. Robblee et al. (1991) and Thayer et al. (1994) documented the die-off of *T. testudinum* and plant community changes in Florida Bay seagrass meadows during the late 1980s. In the northern GOM, Duke and Kruczynski (1992) reported 20-100% seagrass losses over the past 50 years. Fifteen years later, Handley et al. (2007) reported that loss of seagrasses in Texas, Louisiana, Mississippi (MS), Alabama (AL), and Florida was much greater than gains during 1940-2002. Although seagrass loss in many parts of the GOM has occurred since 1940, there are areas where seagrass loss has not been as dramatic or increases in cover have actually been recorded. Seagrasses in Tampa Bay, Florida had been recovered since 1980s, total area in 2016 was even higher than in the 1950s (Sherwood et al. 2017).

Lewis et al. (2008) reported a less severe decline in seagrass area in 1980-2003 than in the preceding 1960-1980 time period for the Pensacola Bay system, with some increases observed in Pensacola Bay and Santa Rosa Bay from 1992 to 2003. In Mobile Bay and adjacent waters, Vittor and Associates, Inc. (2004, 2005, and 2009) found substantially reduced seagrass area in 2002 compared to 1940, 1955, and 1966. According to their study, an increase in seagrass area between 2002 and 2008 (or 2009 in some of the quadrangles in the northeastern Mississippi Sound (MSS)) must be considered together with a change in species composition, from monospecific beds of *H. wrightii* in 2002 to mixed beds of *H. wrighitii* and *R. maritima* (an opportunistic species) in 2008-2009. Overall, this compendium of results suggests that a drastic decline in seagrasses at global, regional and local scales has occurred from the mid-20th century to the present.

Landscape Ecology and its Application to Seagrass Studies

A landscape can be defined in many ways but its definitions invariably include an area of land containing an interacting mosaic of patches or landscape elements (McGarigal 2014). All landscapes have a structure (pattern) that is hypothesized to influence its function (process). Landscape pattern is the combination of (1) composition, i.e., non-spatial aspects such as diversity and relative abundance, and (2) configuration, i.e., spatial aspects such as spatial character, arrangement, and context of landscape elements, consisting of both patchiness and gradients (McGarigal 2014, Wu 2012).

Landscape ecology, which emphasizes the relationship between spatial pattern and ecological processes across multiple scales (Turner et al. 2001a, Wu 2012), has a long history rooted in Central and Eastern Europe but has only received attention in the United States since about the early 1980s. Compared to the European school, the

American school pays more attention to natural systems, to theory, and to model development. Landscape ecology emerged and distinguished itself from other branches in ecology as a result of: (1) the growing awareness of broad-scale environmental issues, (2) the development of new scale-related concepts in ecology, and (3) technological advances of remote sensing, computer hardware and software (McGarigal 2014, Turner et al. 2001a). Fractal geometry, percolation theory, and self-organized criticality continue to provide theoretical stimuli for landscape ecology.

Though the development of landscape ecology closely links to terrestrial systems, the techniques are also applicable to the marine environment. Robbins and Bell (1994) were the first to propose a landscape approach in studying seagrass-dominated systems. Seagrass landscapes are generally simpler than their terrestrial counterparts in terms of species diversity and structural complexity. Nevertheless, they possess both temporal (e.g., bed expansion and contraction over several time scale) and spatial variation (e.g., strong patterns of depth zonation, variable bed shape and structure, a hierarchical arrangement of structure) over a multitude of spatial scales. Seagrass landscapes may include either floral components (e.g., seagrass, macroalgae, epiphytes) or faunal components (e.g., invertebrates and vertebrates living in seagrass meadows) or a combination of both. Only its floral element is of interest in this dissertation. In this dissertation a seagrass landscape will be defined as a heterogeneous land area containing an interacting mosaic of seagrass patches.

Study Area and Objectives

In the state of MS, the MSS is the primary body of water that supports seagrasses, whereas the Chandeleur Islands support the most important seagrass resource in

Louisiana (Poirrier 2007). The MSS is a lagoon formed by a chain of barrier islands that extends 130 km along the AL and MS coastlines. With a 3-m mean low tide depth and a small diurnal tidal range of 0.5 m (Kjerfve 1986), the MSS connects hydraulically with Lake Borgne, LA westward and Mobile Bay, AL eastward. Chandeleur Islands, the largest and oldest transgressive barrier island arc in the northern GOM, are a remnant of the Mississippi River's St. Bernard Delta (McBride et al. 1992). This originally 80-km long island chain has been undergoing major geomorphological alterations with the passage of tropical storms and hurricanes (Penland et al. 1981, Stone et al. 1997, Penland et al. 2005, Otvos and Carter 2008, 2013, Fearnley et al. 2009, Grzegorzewski et al. 2011). The chain consists of five individual islands but only the northern Chandeleur Island (main arc) is under consideration in this study. The shallow 40-km-wide Chandeleur Sound, a critical component of the lower Pontchartrain Basin, has a nominal depth of 3.5 m (Hart and Murray 1978), a diurnal mean tidal range of 0.43 m (Tate et al. 2002) with a maximum amplitude of about 0.8 m. Salinity varies from 20 to 35 parts per thousand in the MS-C (Eleuterius 1976, Kjerfve 1983, 1986, Michot et al. 1994, Lopez et al. 2010, Henkel et al. 2012, Moshogianis et al. 2013), and generally increases in an offshore direction from the MS mainland towards the MSS and Chandeleur Islands (Comyns et al. 2008).

A study of historical change over approximately 70 years for seagrasses in the MS-C was conducted under the broad theme of habitat loss and fragmentation. Seagrass mapping data were collated from a multitude of projects including field surveys, herbarium records, remote sensing imagery, and geographic information system (GIS) maps to provide information on seagrass change from 1940 to 2011. Further detailed analysis of these data utilizing theories and techniques borrowed from landscape ecology were used to quantify patterns and dynamics of the seagrass landscape on each of the barrier islands in the MS-C. Changes in the seagrass landscape can potentially alter or diminish important ecosystem services, affecting environmental and human well-being and prosperity, hence studying these changes can help us to better manage this seagrass resource and the condition of the coastal environment.

[1]This chapter was previously published as: Pham LT, Biber PD, Carter GA. 2014. Seagrasses in the Mississippi and Chandeleur Sounds and problems associated with decadal-scale change detection. Gulf of Mexico Science 32: 24-43.

CHAPTER II – SEAGRASS MAPPING IN THE MISSISSIPPI AND CHANDELEUR SOUNDS AND PROBLEMS ASSOCIATED WITH DECADAL-SCALE CHANGE DETECTION[1]

Introduction

Seagrass Mapping in the Mississippi Sound

Information and maps of MSS seagrasses are available from a multitude of previous studies, but do not provide common methods or mapping units, complicating comparisons of area change over time. Submerged flowering plants were reported in the MSS as early as the 1950s (Humm and Caylor 1955, Humm 1956). However, it was not until the 1970s that the first map of MSS seagrass was published (Eleuterius 1973) based on fieldwork conducted during the spring and summer of 1969, just prior to Hurricane Camille (August 17-18, 1969). In that study, a series of north to south transects were run across the MSS with stations that were ≤ 3.2 km (2 miles) apart. Seagrass extent was then mapped by manually connecting observation points and the general species composition of the seagrass beds was recorded. Later, Eleuterius included seagrasses in his 1978 vegetation maps of Horn and Petit Bois islands (Eleuterius 1979). During the 1990s, the United States Geological Survey – National Wetland Research Center (USGS-NWRC 1998b) published a map of SAV in MS by interpreting June 1992 natural-color aerial photography. This was part of the northeastern GOM seagrass mapping project; the classification system was based on percentage of ground cover of patches within the delineated area and consisted of five classes of seagrass cover, from very sparse patchy seagrass to dense continuous seagrass beds (Handley 2007). Seagrasses were also included when the USGS-NWRC (1999) mapped coastal MS habitats using March 1996 color-infrared aerial photography as the primary data source for identification of wetland and upland vegetation classes. Moncreiff et al. (1998) published the first report on changes in seagrass area and potential seagrass habitat ("PSGH" in their publication) in MS based on comparisons of mapping data from 1969 and 1992 (Eleuterius 1973 and USGS-NWRC 1998b, respectively). In 1992, PSGH was delineated from the 2-m depth contour applied to the USGS-NWRC map. Moncreiff et al. (1998) reported a decline in seagrass area from 5252 ha to 809 ha (84.6%) but a reduction of only 19.6% in PSGH from 1969 to 1992. In January 1999, natural-color aerial photography of the MS barrier islands was collected for a National Park Service project and then photo-interpreted by USGS-NWRC (2003) using protocols similar to the 1992 study. Moncreiff (2007a) combined these 1999 data with the previous 1998 report (Moncreiff et al. 1998) to update seagrass status and trends, and suggested a slow increase in seagrass area throughout the MSS between 1992 and 1999. The 2007 paper, however, did not include information on changes to PSGH.

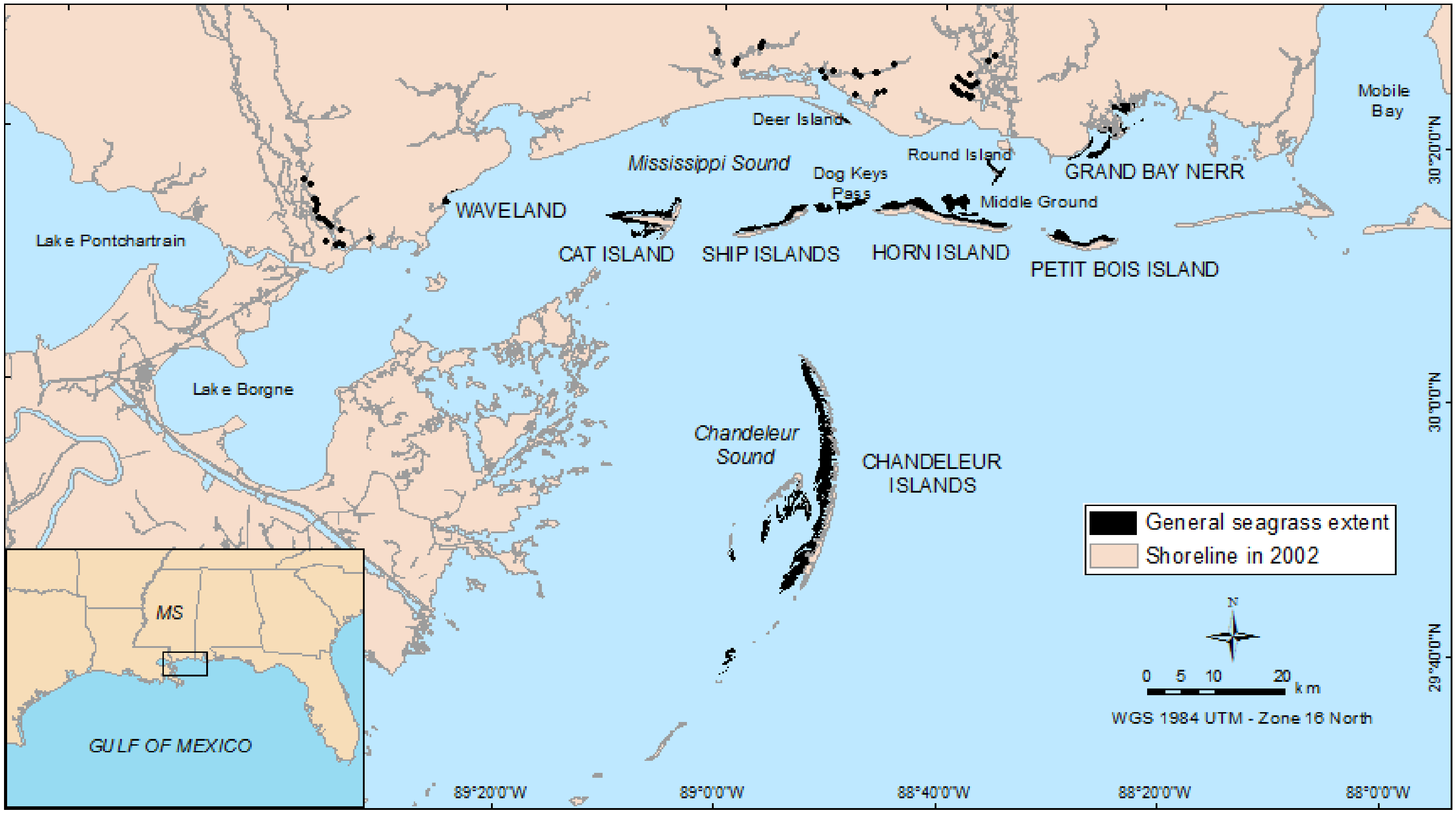

Figure 1. Study area.

Note. General seagrass extent (in black) for all years (1940-2011) was delineated by the outermost border of all historical occurrences. Black dots are point locations of submerged aquatic vegetation found along the Mississippi mainland.

Subsequently, the vegetated seagrass area (VSA) on the MS barrier islands spanning the 1940-2007 period was recalculated by Carter et al. (2011) from analysis of high-spatial-resolution (0.3 m to 2.4 m) aerial photographs or digital image data. Only historical imagery that had been acquired in the fall when water clarity appeared to be high, as evidenced by seafloor visibility, were selected for analysis. This increased the likelihood that individual seagrass patches would be delineated clearly in the imagery and that their spatial extent and foliar density would be near-maximum for the given year. For the more recent years since the 1980s in which multispectral or hyperspectral image data were acquired, only data from a single green band (500-600 nm wavelength) extracted from the full spectral data set were used in the analysis. Vegetated patches were classified based on a pixel edge solution that was modified from Urbanski (2006), with individual seagrass patches being mapped at spatial resolutions from 0.5 m to 2.4 m. The earliest seagrass maps by Carter et al. (2011) were developed from 1940 images of Horn and Petit Bois islands, 1963 images of Ship Island and a 2003 photomosaic of Cat Island. Comparing between results obtained from their recently generated data and those of earlier mapping products, Carter et al. (2011) suggested that conclusions regarding temporal change in seagrass coverage should be made with caution because survey objectives and methods, including spatial resolution, can vary appreciably among studies. Most recently a survey of the seagrasses on the five MS barrier islands was completed by Vittor and Associates, Inc. (2014) based on aerial imagery and complementary ground-truthing in 2010, but cannot be compared directly with maps by Carter et al. (2011) due to the very different minimal mapping units of 404 m^2 versus 1-2 m^2, respectively.

Field and remotely sensed data suitable for mapping seagrasses along the mainland coast of MS are scarce relative to those available for the MS barrier islands. Due to high water turbidity nearshore, data used for mapping were obtained primarily in the field as individual geo-location points, and polygon data are available only for a few years. The two main areas supporting seagrasses along the mainland coastline are Waveland (near Buccaneer State Park) in the western MSS and Grand Bay National Estuarine Research Reserve (GNDNERR) in the eastern MSS. Seagrasses in Waveland were previously mapped only twice (Eleuterius 1973, Moncreiff et al. 1998). GNDNERR seagrasses have been mapped more frequently and recently by the USGS-NWRC (1998b, 1999) and Sanchez-Rubio (2004) who updated the presence/absence as well as total area of the GNDNERR seagrass beds mapped previously by Eleuterius (1973) and Moncreiff et al. (1998). Sanchez-Rubio (2004) located seagrass beds by snorkeling in June 2002 based on a field survey grid of 5-second latitude and longitude (approximately 155 m) intervals, and subsequently area of the beds was estimated from aircraft video camera imagery collected on September 29, 2003. Later, May and other GNDNERR staff (C. A. May, personal communication) organized intensive fieldwork involving snorkeling, rake surveys, and visual identification to map seagrasses in the reserve during June-August 2005 and 2006. They collected global positioning system (GPS) points using a Trimble GeoXT unit (Trimble, Sunnyvale, CA) set to 1.5 m geospatial accuracy and later connected those points to form polygons representing the extent of the seagrass beds. Strange and May (2009) used 2009 aerial imagery to update the size and location of seagrass beds determined in the earlier 2005 survey. Most recently, some of the seagrass beds in Point aux Chenes Bay (northwestern GNDNERR) were mapped in July 2011 and

July 2012 using side-scan sonar, which was found to be an effective technique in these highly turbid and shallow waters (Hendon 2013).

Seagrass Mapping in the Chandeleur Sound

Most of the seagrass mapping studies in the Chandeleur Sound combined image interpretation with ground observations to verify the presence of seagrasses. Johnston and Handley (1990) mapped Chandeleur Sound seagrasses from aerial photography acquired in 1978, 1982, and 1987. Poirrier and Handley (2007), in their paper discussing status and trends of seagrasses on the islands, provided additional maps for April 1969, October 1969, April-June 1992, and November-December 1995. More recently, Bethel et al. (2006) mapped seagrasses on the northern part of the Chandeleur Islands using April 1999, November 2000, and November 2002 aerial photography. Bethel and Martinez (2008) studied the impacts of Hurricane Katrina (August 28-29, 2005) on seagrass area using imagery acquired in January and October 2005. Additional photography was obtained following the Deepwater Horizon Oil Spill (April – July 2010), and the number of seagrass maps completed by USGS for the Chandeleur Islands increased to 14 by February 2012 (L. R. Handley, personal communication). However only the 1992 map (USGS-NWRC 1998a), which used November 1992 color infrared aerial photography as the primary data source and January 1992 natural-color aerial photography as supplemental data for photo interpretation, is currently publically available. A common theme resulting from these maps is that erosion during hurricanes and lesser storms has periodically reduced the Chandeleur Island land and the associated seagrass area, with recovery of land area and seagrass cover occurring during the relatively calm intervals between storms.

Study Objectives

This chapter presents an updated and comprehensive evaluation of decadal-scale changes to the seagrass area found in the MS-C. Specific objectives were to: (1) combine all available data that were acquired in both Sounds from 1940 to 2011 to develop maps showing the general distribution of seagrasses over time, and (2) determine interannual and decadal-scale changes in seagrass area within the MS-C.

Materials and Methods

Study Area and Base Map

The study area encompassed coastal MS and the Chandeleur Islands in Louisiana (Figure 1). Study sites were placed into three groups representing an inshore to offshore gradient: Group 1 is the MS mainland coastline, specifically the GNDNERR and Waveland locations dominated by *Ruppia maritima*; Group 2 includes the MS barrier islands (Cat, West and East Ship, Horn, and Petit Bois islands) dominated by *Halodule wrightii*; and Group 3 comprises the northern Chandeleur Islands (NCI) dominated by *Thalassia testudinum* co-occurring with other seagrass species.

Seagrass species composition differs among the three study groups following an inshore to offshore gradient. *Ruppia maritima* occurs in soft muddy bottoms and can tolerate brackish to hypersaline waters. *Halodule wrightii*, *Halophila engelmannii*, *T. testudinum*, and *S. filiforme* grow on sandy or sand-mud substrates with high salinity and relatively clear water. Along the mainland in Group 1, *R. maritima* dominates at Waveland and coexists with some *H. wrightii* in GNDNERR. The MS barrier islands in Group 2 have mainly supported *H. wrightii* since 1978 (Eleuterius 1979), even though *T. testudinum*, *S. filiforme*, and *H. engelmannii* were documented there in 1969 and earlier

(Humm and Caylor 1955, Humm 1956, and Eleuterius 1973). *Syringodium filiforme* individuals were encountered on Horn and Petit Bois Islands in 1993 and 2005 but only in very small areas (Heck et al. 1994, Heck and Bryon 2006, Moncreiff 2007a). A small population of *T. testudinum* still persists on Horn Island in a brackish lagoon named Ranger Lagoon. All of the five seagrass species mentioned above occur in Group 3 within the Chandeleur Sound.

The base map of the region was developed using GIS layers for the coastal hydrographic area (USGS, US Environmental Protection Agency 1999) and shorelines (Mississippi Department of Environmental Quality (MS DEQ) 2004, Carter et al. 2011). The most recent shoreline available for the whole study area is from 2002, therefore, it was chosen for the base map. Other shorelines dated back to 1850 with gaps among years; in some years, data were only available for certain portions of the coast. Historical data that were utilized when displaying a specific seagrass layer for a single year include the shoreline closest in time to that specific seagrass layer. Topologically integrated geographic encoding and referencing (TIGER) census geographic data (U.S. Department of Commerce 2009), primarily roads, were used for image georectification for that same year. All GIS files were downloaded from the Mississippi Geospatial Clearinghouse Portal website (MS DEQ et al. 2003).

Seagrass Mapping Data

An overview of the data sources used in the development of the seagrass distribution maps is provided in Table 1. Paper maps for 1969 (Eleuterius 1973) and 1992 (Moncreiff et al. 1998) were scanned at a resolution of 600 dots per inch (dpi) and saved as TIFF image files. These digital files were then georeferenced in ArcGIS version 9.3

(Environmental Systems Research Institute – ESRI, Redlands, CA) using 1950 and 1993 shorelines, respectively. The georeferenced images were then georectified to World Geodetic System (WGS) 1984, Universal Transverse Mercator (UTM) zone 16 North projection. Seagrass areas in the georectified images were then manually delineated on the computer screen by tracing along the seagrass polygon edges using digitizing tablets.

Digital GIS files of USGS-NWRC's maps (1992, 1996, and 1999 for MSS; 1992 for the Chandeleur Islands) were used in this project, but note that these maps are also available as hard copies. Data were downloaded as ArcInfo interchange files (.E00), imported to ArcGIS using "Conversion Tools", and then transformed to the map coordinate system (WGS 1984 UTM 16N, or UTM 15N for Chandeleur Island data). The USGS-NWRC (1998b) map was the primary data for 1992, only the Waveland section of the paper map by Moncreiff et al. (1998) was ultimately used because that area was not covered in the former map. However, the map by Moncreiff et al. (1998) contains inaccuracies in both seagrass extent and geo-location at Waveland because they were hand-drawn based on the general and inaccurate location of seagrass patches indicated in the previous 1969 paper map by Eleuterius (1973). In the USGS-NWRC's 1996 habitat map, only the features identified as E2AB3L, which are "estuarine intertidal aquatic beds - rooted vascular" and are equivalent to our definition of seagrasses, were extracted. The 1996 seagrass data were discovered to be exactly identical to the 1992 map (USGS-NWRC 1998b) and, therefore, were excluded from the final maps. Other digital data for GNDNERR (Sanchez-Rubio 2004, Strange and May 2009), the MS barrier islands (Carter et al. 2011), and the Chandeleur Islands (Bethel et al. 2006, Bethel and Martinez

2008) were obtained in polygon vector and/or raster formats, and only map projection transformation was needed.

Table 1

Citations of Mapping Data Sources

Sites & Year	Pixel dimension (m)	Source	References
Group 1 – Mississippi mainland			
Waveland			
1969	NA	T	Eleuterius 1973
1992	NA	A	Moncreiff et al. 1998
1998	NA	A	this study
2004-2011	1	G	this study
Grand Bay GNDNERR			
1969	NA	T	Eleuterius 1973
1992	NA	A	Moncreiff et al. 1998
1992	NA	A,G	USGS-NWRC 1998
2003	NA	T	Sanchez-Rubio 2004
2005, 2006	NA	G	May (personal communication)
2009	NA	G	Strange and May 2009
Group 2 - Mississippi barrier islands			
Cat Island			
1969	NA	T	Eleuterius 1973
1992	NA	A	Moncreiff et al. 1998
1992, 1996, 1999	NA	A,G	USGS-NWRC 1998, 1999, 2003
2003, 2006, 2007	0.3, 1, 2	A	Carter et al. 2011
2010	1	A,G	Vittor and Associates, Inc. 2014
East and West Ship Islands			
1963	1.5	A	Carter et al. 2011
1969	NA	T	Eleuterius 1973
1975	0.5	A	Carter et al. 2011
1992	NA	A	Moncreiff et al. 1998
1992, 1996, 1999	NA	A,G	USGS-NWRC 1998, 1999, 2003
2003, 2006, 2007, 2008	0.3, 1, 1, 2.4	A	Carter et al. 2011
2010	1	A,G	Vittor and Associates, Inc. 2014

Table 1 (continued).

Sites & Year	Pixel dimension (m)	Source	References
Group 2 - Mississippi barrier islands (continued)			
Horn Island			
1940, 1952	1, 0.5	A	Carter et al. 2011
1969	NA	T	Eleuterius 1973
1971	2	A	Carter et al. 2011
1978	NA	T	Eleuterius 1979
1992	NA	A	Moncreiff et al. 1998
1992, 1996, 1999	NA	A,G	USGS-NWRC 1998, 1999, 2003
2003, 2006, 2007, 2008	1, 1, 2, 2.4	A	Carter et al. 2011
2010	1	A,G	Vittor and Associates, Inc. 2014
Petit Bois Island			
1940, 1952	1, 0.5	A	Carter et al. 2011
1969	NA	T	Eleuterius 1973
1978	NA	T	Eleuterius 1979
1985	1	A	Carter et al. 2011
1992	NA	A	Moncreiff et al. 1998
1992, 1996, 1999	NA	A,G	USGS-NWRC 1998, 1999, 2003
2003, 2006, 2007, 2008	0.5, 1, 2, 2.4	A	Carter et al. 2011
2010	1	A,G	Vittor and Associates, Inc. 2014
Group 3 - Chandeleur			
North Chandeleur Islands			
1992	NA	A,G	USGS-NWRC 1998
1999, 2000, 2002	2, 2, 2	A,G	Bethel et al. 2006
2005	2.4	A,G	Bethel and Martinez 2008

Note: Spatial resolution is in meter square but reported as pixel dimension (m), NA indicates a previously completed mapping product (paper or digital) with unknown pixel resolution. Source of data includes A, aerial imagery; G, GPS-based polygons; and T, transects with points. Sources of vector-point data are not listed here.

Additional efforts were spent on creating new GIS data for Waveland, which is a historically understudied, yet important seagrass resource along the mainland coastline of MS. Three complementary approaches were used to fill this critical gap: (1) recent ground-truthing, (2) historical herbarium specimen geolocations, and (3) aerial photograph interpretation. From 2006 to 2012, the authors conducted ground-truthing on a yearly basis of any seagrasses present at this site. Days in November and December with extreme low tides were considered the best time to spot the sparse coverage and short *R. maritima* plants at Waveland. GPS coordinates of the approximate centroid of all seagrass patches were recorded. Additionally, if a patch was mappable (at least 0.5 m in diameter), a new polygon feature was created by walking along the patch edge with a Trimble GeoExplorer 2008 Series XT GPS unit. These point and polygon data were then transferred, differentially corrected to increase spatial accuracy of the raw data, and exported to shapefiles using Pathfinder Office ver. 4.0 (Trimble). The second complementary but less precise approach was plotting coordinates associated with specimens of *R. maritima* in the Gulf Coast Research Laboratory Herbarium (HGCRL) collected during the period from September 1974 to June 2001 prior to the annual ground-truthing visits. Our third method was to map seagrass patches from historical aerial photography. Out of the available aerial images containing the Waveland area, the image in February 1998 downloaded from the Earth Resources Observation Systems – Earth Explorer data center website (1999) is the only one acquired during low tide when water was clear, resulting in a high contrast between the relatively bright sand bottom and dark patches of vegetation. The image was geo-rectified using ground control points and TIGER transportation routes as outlined previously, and then digitized by manually outlining the seagrass patch boundaries. Field data notes and historical herbarium records were considered during the visual estimation of seagrass patches in this image.

Map Generation and Seagrass Area Calculations

A map of seagrass distribution in the study area was created in ArcGIS by drawing outline boundaries of all historical and recent seagrass extents and including points of occurrence where polygon data were not available (Figure 1). All data were projected to a common coordinate system, WGS 1984 UTM, zone 16 North for MS files and zone 15 North for Chandeleur Island files. GIS data layers were stored separately by year and site. Each GIS file was accompanied with an attribute table of collection/mapping date, collector's names, location, area and perimeter (for polygons). Metadata, if not already available, were created to the Federal Geographic Data Committee (FGDC) standard (FGDC and USGS 1999).

There are different definitions of seagrass area that have been used prior to this chapter, which can result in confusion and difficulties in calculating change over time. Here we define three terms used to describe different aspects of seagrass area in this study: (1) Seagrass extent is the area within the outline encompassing all seagrass on a map or in a field survey, such as was done in Eleuterius (1973) and Vittor and Associates, Inc. (2014), and includes potentially extensive non-vegetated areas between the seagrass patches. (2) Seagrass coverage is based on the area of polygons that each contain multiple seagrass patches (ranging from sparse to dense patch aggregation), such as in USGS-NWRC (1998b), and includes a small amount of non-vegetated area between the seagrass patches. (3) VSA is the sum of the area of all the seagrass patches. An individual seagrass patch is equivalent to an individual polygon as mapped by Carter et al. (2011); note that not all seagrass patches may be getting mapped (e.g., those with very sparse shoot density, or those that are in deeper water and cannot be distinguished readily from the background). It is clear that VSA is a subset of seagrass coverage, which in turn is a subset of seagrass extent, and that newer mapping methods have made the increase in spatial accuracy obtained by the VSA method possible. Finally, the term PSGH was defined by Moncreiff et al. (1998) as the area of everything shallower than the 2 m depth contour, and is generally a larger area than the seagrass extent. The “true” seagrass area is likely to be somewhat greater than the VSA but less than the area obtained from seagrass coverage calculations, and substantially lower than either seagrass extent or PSGH.

A revised estimate of seagrass area (Table 2) was obtained using the “Calculate geometry” function in ArcGIS even if it was reported previously (e.g., Eleuterius 1973, Moncreiff et al. 1998, Moncreiff 2007a) to minimize calculation errors that may have otherwise occured by comparing results obtained by different geometric formulae. Maps for 1969 (seagrass extent), 1992 (seagrass coverage), and 2006 (VSA) were chosen to represent changes in seagrass area, due in part to the different mapping methods (Figure 2 and 3), as in those years data are available for the majority of the study sites. An exception was the use of 2004 data for Waveland and 2005 data for the NCI due to the lack of 2006 maps at these two locations. Also, the 1969 data for the Chandeleur Islands mentioned by Poirrier and Handley (2007) is not publicly accessible, therefore is not included in this study. The three time points illustrate some of the potential error associated with comparing different mapping techniques directly (seagrass extent vs. seagrass coverage vs. VSA) from different investigators over the decades.

Change Analysis of Seagrasses on Horn Island

Seagrasses on Horn Island were studied in more detail to illustrate changes occurring to island geomorphology and the potential influence on seagrass area during the study period from 1940 to 2008. The island’s shoreline, if not already available, was digitized from aerial photography when access to the original images was available. Horn Island was divided into three equal sections based on the distance from the eastern tip to the western tip in a particular year. Seagrass extent was divided into thirds accordingly and seagrass area calculated for each third (eastern, middle, western) along with the percentage of the total seagrass area of the island in each year as shown in Figure 4.

Results

General Seagrass Distribution in the Mississippi and Chandeleur Sounds

Seagrasses can be found primarily on the MS barrier islands within the Gulf Islands National Seashore and along select portions of the MS mainland coast, particularly in GNDNERR and Waveland, the two mainland areas where seagrasses are protected. The NCI support seagrasses on their western shoreline (Chandeleur Sound), and the MS barrier islands have seagrasses on the northern side (MSS). In both Sounds, seagrasses are found only on the lee side of the islands where they are protected from the higher wave energy of the open GOM (Figure 1). Seagrasses mostly occur as beds of small vegetated patches in MSS in contrast to continuous meadows on the NCI.

Decadal-Scale Changes in Vegetated Seagrass Area

Over the years included in this study, each group of sites has experienced changes in VSA at various rates (Table 2, Figure 2 and 3). More data are available during 1990 – 2010 than in previous decades, a reflection of advances in geospatial technology (remote sensing and GIS), which promote increasingly accurate natural resource mapping. Horn and Petit Bois Islands have the most consistently available information, only data for either the 1970s or 1980s are missing (Table 2). Other sites do not have information prior to 1960, and this paper includes no mapping data prior to 1992 for the NCI (Table 2), even though earlier imagery does exist (Handley 1994, Poirrier and Handley 2007). Decadal-scale changes in seagrass area are demonstrated in Figure 2 and 3 using data from 1969, 1992, and 2005/2006.

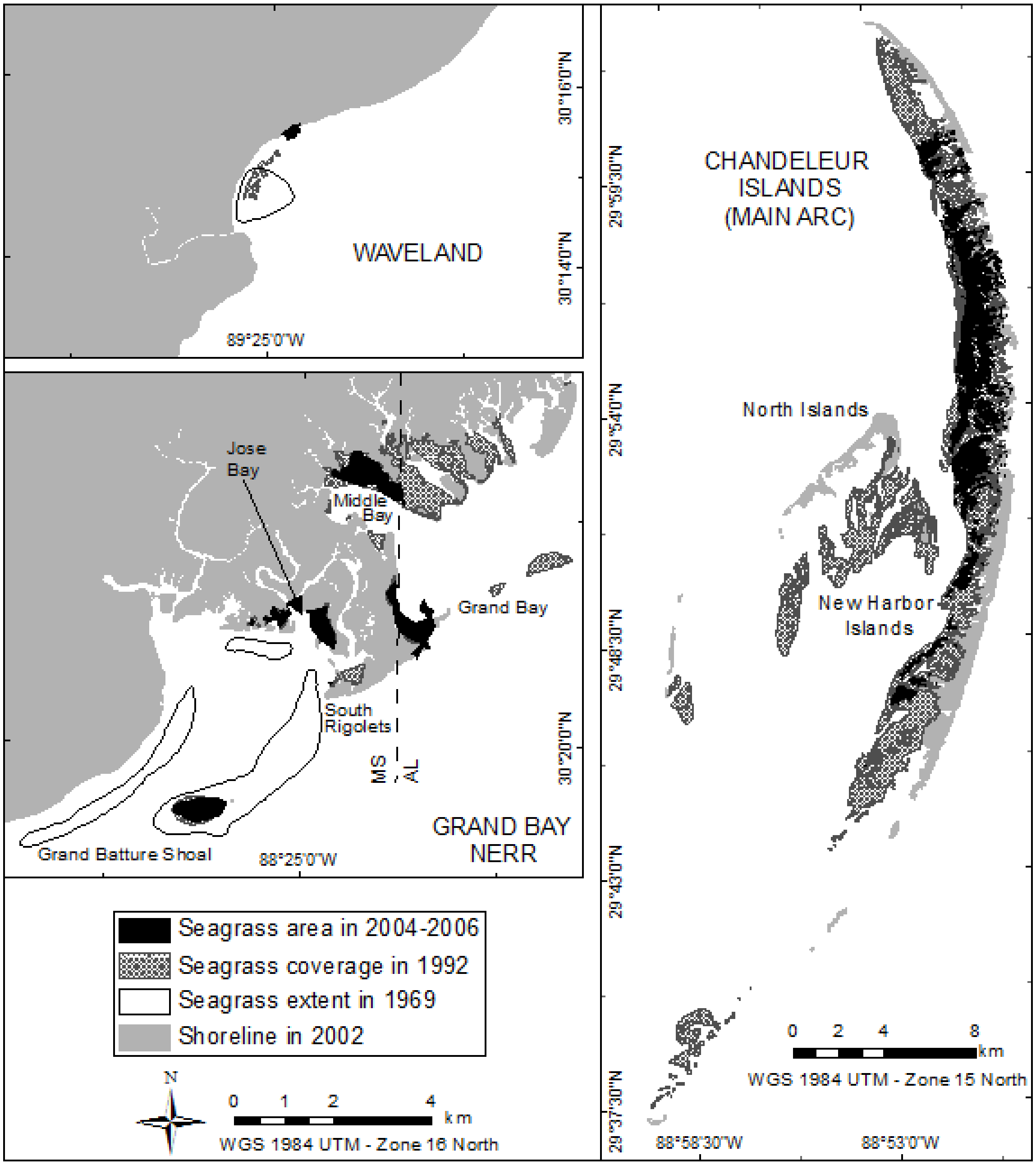

Figure 2. Change in seagrass along Mississippi mainland and on the Chandeleur Islands (Group 1 and 3) from 1969 to 2006.

Note. In the 2000s, seagrass maps were in 2004 for Waveland, in 2006 for GNDNERR, and in 2005 for Chandeleur Islands. The apparent location shift of seagrasses at Waveland is due to a mapping error in the early paper maps rather than a geolocation change. The Mississippi – Alabama state line is indicated as a dotted line in GNDNERR map. Data in 1969 was not available for the Chandeleur Islands. Data for the North Islands and New Harbor Islands, as well as the southern portions of the island arc are not available for 2005. The Chandeleur Islands are in half-scale to the Mississippi sites.

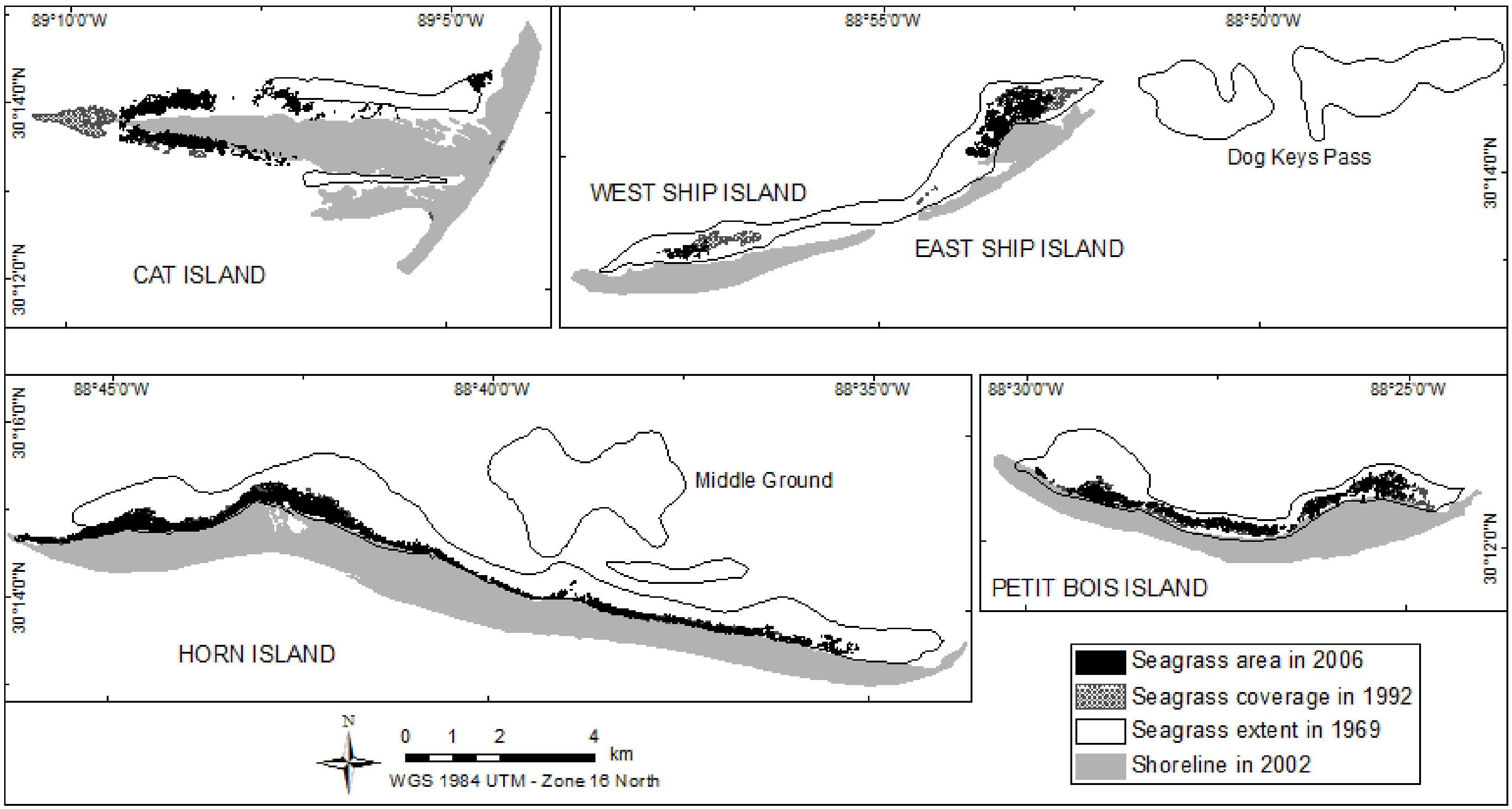

Figure 3. Change in seagrass in the Mississippi Sound barrier islands (Group 2) from 1969 to 2006.

Note. Dog Keys Pass and Middle Ground data were included only in the 1969 map but were not present in later map dates.

Table 2

Seagrass Area (ha) Calculated for Each Site by Year

Year	Wave -land	GND-NERR	Cat	West Ship	East Ship	Horn	Petit Bois	North Chandeleur
1940						76.7	54.1	
1952						45.7	15.2	
1963					30.4			
1969	**90.9**	**550.5**	**226.3**		**655.6**	**1,365.0**	**650.2**	
1971						19.2		
1975				1.8	1.6			
1978						**249.4**	**222.8**	
1985							17.6	
1992		*183.1*	*55.4*	*23.2*	*27.6*	*87.4*	*76.9*	*4994.5*
	(9.6)	*(122.3)*	*(58.3)*	*(26.2)*	*(45)*	*(134.3)*	*(95.0)*	
1996			*56.4*	*23.1*	*28.7*	*90.0*	*78.2*	
1998	27.9							
1999			*645.5*	*0.5*	*97.1*	*233.9*	*172.1*	1495.7
2000								1545.1
2002								1525.1
2003		*142.5*	21.8	0.02	16.5	50.7	8.0	
2004	+[a]							
2005		*214.5*						(Jan) 1,193.3 (Oct) 895.0
2006	0	*181.6*	25.5	0.9	15.5	82.0	18.9	
2007	+[a]		71.2	1.7	14.0	38.1	16.7	
2008	+[a]			3.2	17.9	18.6[b]	7.3	
2009	+[a]	*213.8*						
2010	+[a]		**693**	**50.6**	**105.6**	**394.1**	**218.9**	
2011	0.0009							

Note: Data from 1992 is calculated from two different sources based off the same original imagery: area calculated by USGS-NWRC (1998b) is on the top line while area calculated by Moncreiff et al. (1998) is listed below in parentheses. Values in bold are seagrass extent, values in italics are seagrass coverage, and all other values represent vegetated seagrass area. (a) Ground-truthed data at Waveland indicate presence of *Ruppia maritima* that was not mapped or too small for polygon mapping. (b) Area calculated for Horn Island in 2008 contains only 70% of the island and is missing the eastern third.

The difference between the seagrass area estimated in the 1969 map and other maps in this paper is very large (Table 2, Figure 2 and 3). The 1992 and 1999 maps created by USGS-NWRC (1998b, 2003) also show substantially larger calculated seagrass area compared to maps in other years. These differences are largely the result of different investigators using different mapping criteria. Seagrass area was calculated in 2005/2006 based on vegetated seagrass polygons, whereas in 1992 it was calculated from seagrass coverage, and in 1969 the data used measured seagrass extent.

Breaking down the temporal changes in seagrass area by study group yields some interesting findings. Seagrasses in Group 1 (MS mainland coast) had different dynamics between east and west. In the western part of the state, the *R. maritima* population located near Buccaneer State Park in Waveland was first documented in the late 1960s. The seagrass beds there were severely damaged by Hurricane Camille in 1969 and seagrass patches were not noticeable in the early 1970s (J. D. Caldwell, personal communication). There was no extensive seagrass survey done in this area during the 1970s and 1980s. The beds were documented again in the late 1990s; photos from this time show that seagrass patches at Waveland were comparable in abundance and size to those on the MS barrier islands in the 2000s. After being wiped out by Hurricane Katrina in 2005, this *R. maritima* population remained almost completely absent with very few or sometimes no plants found in the yearly field surveys from 2006 to 2013; only after 2014 were bigger patches of seagrasses observed again in Waveland, with recovery to pre-Katrina patch sizes occurring by December 2016. This population was decimated by Hurricanes Camille in 1969 and Katrina in 2005 and appears to take many years to recover, presumably from seeds or fragments originating from nearby freshwater populations. In the eastern end of the state is GNDNERR, where a considerable amount of *H. wrightii* and *R. maritima* has been consistently found at four discrete locations: Middle Bay, Grand Bay, South Rigolets, and Grand Batture shoal (Figure 2). Seagrass area in

GNDNERR showed an apparent decline from 1969 to 1992 (methodological differences may account for part of this decline), and then fluctuated between 150 - 200 ha until 2009. Overall, the eastern end of the MS mainland supported consistent seagrass coverage through the years, whereas *R. maritima* has fluctuated more dramatically in the western part of the state after strong hurricanes damaged the beds in 1969 and 2005.

Trends in seagrass area also differed among the five MS barrier islands in Group 2. Horn Island, as the largest island, has the largest seagrass area in this group while West Ship Island, the second smallest island, generally had the least seagrass depending on the year surveyed (Table 2). One of the biggest problems in determining temporal change in seagrass area for Group 2 is the large difference in area calculated by different investigators using different mapping techniques. Earlier publications (e.g., Eleuterius 1973, 1979, but also Vittor and Associates, Inc. 2014) generally calculated seagrass area as seagrass extent, thereby greatly inflating the number of hectares reported by including substantial areas of unvegetated sand bottom (Table 2, values in bold). As mapping technology evolved and greater spatial accuracy was possible, publications tended to report area calculated from seagrass coverage with polygons representing large numbers of discrete seagrass patches with similar density (e.g., USGS-NWRC 1998b, Bethel et al. 2006), which still results in some inflation of the number of hectares (Table 2, values in italics). Finally the approach adopted by Carter et al. (2011) to delineate each patch as a separate polygon allows for the most accurate calculation of VSA, but may underestimate the “true” area where patches fail to get mapped (e.g., very sparse shoot density or patches smaller than image pixel size). To reduce bias introduced with the different mapping techniques used, the interpretation of temporal change in Group 2 will focus on the VSA technique, as it is the most accurate and covers the largest time domain.

Cat Island had around 55 ha (calculated from seagrass coverage) of seagrass in the 1990s but that declined to around 22 ha in the early 2000s before increasing to 71.2 ha in 2007 (both calculated from VSA), tripling the 2003 value (Table 2). Seagrass patches also had become more numerous on the western end of Cat Island during this period, and had declined along the northeastern shoreline (Figure 3). Seagrasses on Ship Island declined dramatically after 1969 when Hurricane Camille cut through the island and divided it into East Ship and West Ship islands. In 1963 there were 30.4 ha of VSA on Ship Island, whereas by 1975, there were only 3.4 ha of seagrasses left on these two islands with almost equal amounts on each (Table 2). During the 1990s seagrasses had recovered to pre-Camille levels (based on seagrass coverage) but then declined again and it was not until the late 2000s that the VSA on West Ship Island came back to earlier levels. On East Ship Island, VSA fluctuated around 16 ha between 2003 – 2006 before reaching a peak of 17.9 ha in 2008. East Ship Island supports more seagrasses than West Ship Island, which has a Civil War-era fort (Fort Massachusetts) and regular tourist ferry service from March to October. At the beginning of the study period in 1940, Horn Island supported 76.7 ha of seagrasses. About 40% of this VSA was lost by 1952, and only 25% (19.2 ha) was left by 1971 (after Hurricane Camille). In the 2000s, Horn Island seagrasses had recovered with a higher VSA present in 2003 than in 1952. In 2006 (after Hurricane Katrina), VSA on Horn Island had the highest single observation (82 ha) in the 71-year study period, 7% higher than in 1940. The data show a decline after 2006, as in 2007 the amount of VSA had shrunk back to only 50% of that in 1940. Finally, Petit Bois

Island supported 54.1 ha of seagrasses in 1940. Only 30% of this area remained by 1952, followed by a slight increase to 17.6 ha VSA in 1985. In 2003, only 15% (8 ha) of the original 1940 VSA was found. In the 2000s, VSA on Petit Bois Island showed a similar trend as on Horn Island, which was an increase in the first half of the decade followed by a decline in the latter half. However, with 36% of the 1940 VSA existing in 2006, Petit Bois Island did not experience as dramatic a recovery as Horn Island did. In short, there was an overall decline in VSA on MS barrier islands from 1940 to 2008. In the western MS Sound, Cat Island experienced an increase in VSA from 2003 to 2007, while in the eastern MSS on Horn and Petit Bois islands there was an increase in VSA in the first part followed by a decrease in the latter part of the same decade. The dataset for the 2000s decade also demonstrates that large inter-annual fluctuations in VSA are possible on a given island. This complicates interpretation of decadal-scale change in Group 2, as it is not known whether the single year of data in earlier decades represents an average or a possible extreme value for VSA within that decade.

Among all study sites, the NCI (Group 3) had the greatest seagrass area, at nearly 5,000 ha in 1992, and also experienced much higher loss (> 80%) during the subsequent decades, although it is not clear how much bias associated with the different mapping techniques may have influenced this apparent loss (Table 2, Figure 2). Seagrass area on the NCI in 1992 was 10 times higher than at all MS sites combined when using the same mapping technique (seagrass coverage) for both locations. But in 2005-2006 there was only roughly twice as much when VSA is used instead as a common method to calculate area. In 1999 – 2002, the VSA on the NCI fluctuated around 1,500 ha. Comparison of VSA in January 2005, before Hurricane Katrina, with October 2005 showed a decline of

25% in a single year. The lack of consistent mapping techniques for the Chandeleur Sound complicates understanding of decadal-scale changes in seagrass area.

In summary, seagrasses have continued to be supported in all the locations that were first mapped in 1969, except for Waveland on the mainland, which is recovering. There was an overall decline in VSA from 1940 to 2011 in the MS-C region. The Chandeleur Islands, which are farthest offshore, are home to a much greater area and diversity of seagrasses when compared with the MS sites, but may have also experienced the largest decline due to protective barrier island erosion and loss.

Decadal-Scale Changes in Seagrass Using Horn Island as a Case Study

To demonstrate habitat change and the subsequent shift in location and total area of seagrass beds at a particular site, Horn Island was chosen as a case study. A set of seagrass maps on Horn Island spanning the study period from 1940 to 2008 is shown in Figure 4. The island underwent thinning in the north-south and shortening in the east-west direction over time. The 1940 seagrass area showed the farthest offshore distribution of vegetated patches in the north-south axis (however, data from 1969 are of insufficient spatial resolution to be considered very accurate), whereas the 2006-2008 seagrass area showed a shift onshore into shallower waters compared to the earlier data. There was land loss on the eastern end of the island at a much higher rate than land gain on the western end of the island. Horn Island is gradually moving westward and the shape of the eastern tip keeps changing. Seagrass distribution on Horn Island shrunk in both north-south and east-west directions mirroring the geomorphological changes of the island shoreline.

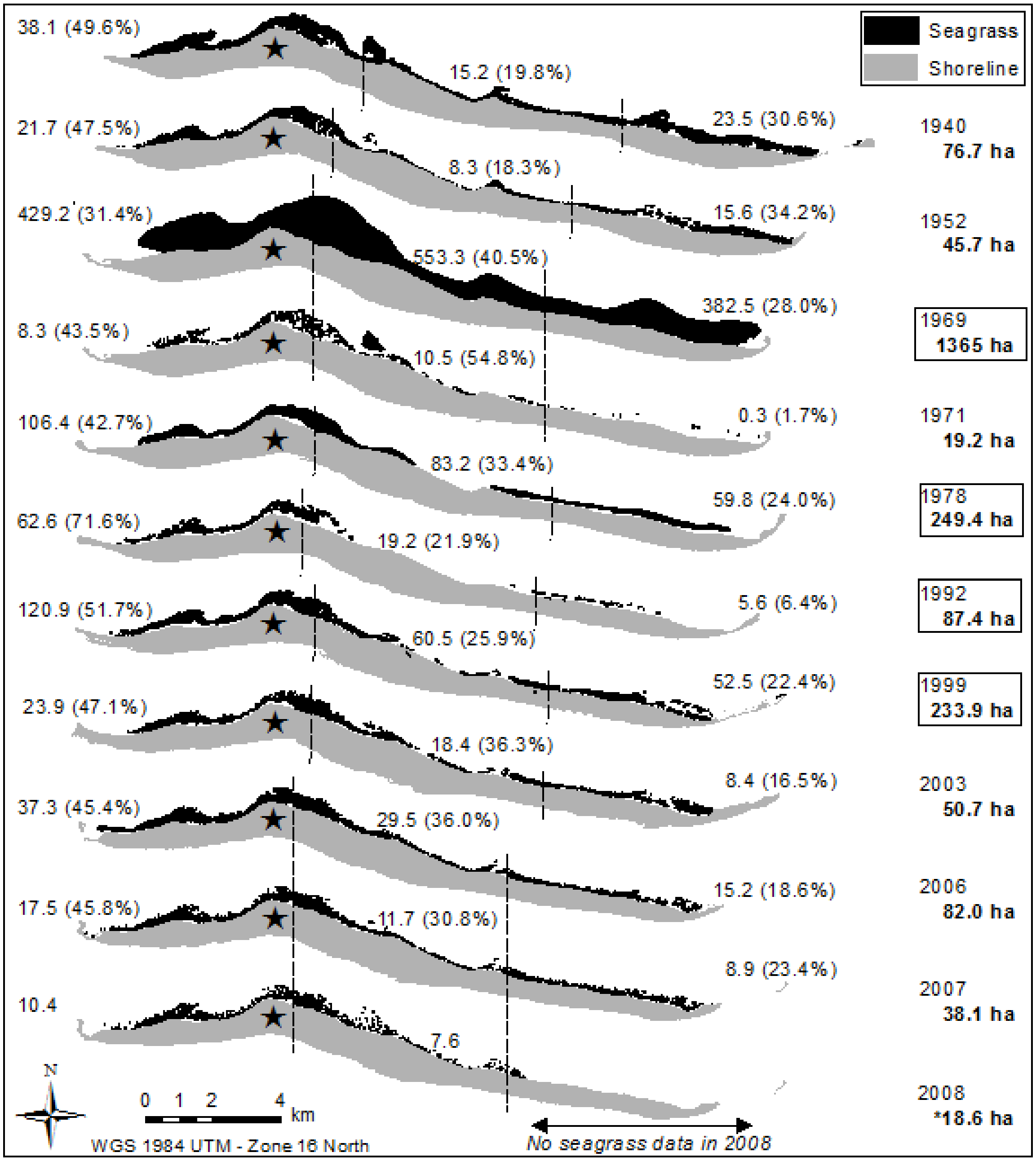

Figure 4. Time series maps showing changes in seagrass and island size of Horn Island from 1940 to 2008.

Note. The star indicates a common geo-reference in all years. Map dates and the associated total seagrass area (ha) are shown on the right. The four years (1969, 1978, 1992, and 1999) indicated in boxes are based on coarse polygon maps of seagrass extent/coverage and highlight the potential inflation of the total area calculated compared to VSA in other years. The island and adjacent seagrass habitat were divided at the dotted lines into three equal segments based on the distance from its eastern to western tips for that year. Seagrass area (ha) and percent of total area for each third is shown above that segment. The 2008 data cover less than 70% of the island length, as seagrass survey data was not collected from the eastern portion, therefore no percent of total area was calculated for this year.

Comparing among years using the VSA data only (1940, 1952, 1971, 2003 – 2008), maps show a decline from 76.7 ha in 1940 to 19.2 ha in 1971, followed by a recovery to a mean of 57 ha in the period of 2003-2007 (Figure 4). The year with the single greatest VSA was 2006 (82 ha), compared with 1,365 ha of seagrass extent in 1969 and 233.9 ha of seagrass coverage in 1999, the latter two values both inflated by including unknown amounts of bare sand bottom. From west to east, VSA on each segment of Horn Island underwent different rates of change. In the 1940s and 1950s, the accreting western third supported the largest seagrass area (21.7 - 38.1 ha), following by the eroding eastern third (15.6 - 23.5 ha), then the middle third (8.3 – 15.2 ha) supported the least seagrass. In 1971, post-Camille, the protected middle segment had the highest VSA (10.5 ha), the western part took second place (8.3 ha), and the eastern part had the lowest (0.3 ha). Since 2003, the highest VSA occurred on the accreting western third (17.5 – 37.3 ha) and the eroding eastern third had the lowest area (8.4 - 15.2 ha).

A correction for bias in area calculated by the different methods (seagrass extent vs. seagrass coverage vs. VSA) can be attempted by comparing the percent of the total seagrass in a given year found on each third of the island. Comparing percentage of total seagrass from maps for VSA (1940, 1952, 1971, 2003 – 2008) shows the accreting western third of the island has remained consistent over time with 43.5-49.6% of the total seagrass, whereas the eroding eastern third has dropped from > 30.6% in 1940-1952 to < 23.4% in 2003-2007, and the more stable midsection has increased from around 19% in 1940-1952 to around 33% in 2003-2007 (Figure 4). Dramatic departures from these proportions were found in 1971, after Hurricane Camille, with the stable mid-section having 54.8%, and the eroding eastern third supporting only 1.7% of the total seagrass for that year (Figure 4). Maps based on seagrass coverage, from the USGS-NWRC projects in the 1990s, suggest a higher proportion of the total seagrass was found on the accreting western third (51.7 – 71.6%) than on the eroding eastern third (6.4 – 22.4%).

In summary, the accreting western side of the island consistently had the largest amount or percentage of seagrass over the years, whereas the eastern side kept losing seagrasses as this end of the island eroded (Figure 4). Comparing the 1940 map with the data during the 2000s, the accreting western third of the island had the largest VSA with a stable percentage (45-50%) of the total, the stable middle third of the island showed the smallest decline in VSA and actually an increase in the percentage, while on the eroding eastern section of Horn Island VSA and percentage decreased over this time period.

Discussion

Spatial Distribution of Seagrass in the Mississippi and Chandeleur Sounds

Principal factors responsible for the observed pattern of seagrass distribution in the MS-C include water depth, salinity, type of substrate, and protection from wind and waves. Because of low light levels in the turbid waters, seagrasses do not currently colonize depths greater than about 2 m in the MSS (Heck et al. 1994, 1996), but historically grew down to 6 m or more in the clearer water of the Chandeleur Sound (Eleuterius 1987). Seawall construction from Biloxi to Gulfport in the 1930s and artificial beach maintenance since then have resulted in continuously shifting sandy sediments that are unsuitable for seagrass establishment, accounting for the large gap in seagrass distribution evident along the mainland coastline between Waveland and GNDNERR. *Ruppia maritima* is found in muddy sediments in brackish waters only in two areas along the mainland: (1) Waveland, where part of the beach was intentionally not renourished after the 1980s to protect this SAV, and (2) GNDNERR, where saltmarshes dominate the shoreline of the eroding former delta of the Escatawpa River. Additional species of SAV occur in brackish to freshwater in the river deltas and bayous along the mainland (Figure 1) including *Vallisneria americana*, *Najas guadelupensis*, *Zannichellia palustris*, and *Potamegeton* spp. (Cho et al. 2010, 2012). The seagrass, *H. wrightii*, occurs on sand to sand mud bottom, and in relatively clear waters with a salinity of 20 ppt or more (Eleuterius 1987), and is most prevalent in the shallow waters along the MS barrier islands. However, *H. wrightii* can also be found in the southern parts of GNDNERR on the submerged shallow sand-banks of the former Grand Batture Islands, which eroded away due to lack of riverine sediment supply, storm damage, and constant wave energy from the prevailing southeasterly summer winds. The prevailing winds stirring up the fine sediments in the nearshore shallows is a major reason why the MS mainland coast does not support seagrasses, except for in protected bays, bayous, marshes, and ponds. No seagrasses are found on the high-wave-energy Gulf-facing southern beaches of West Ship, East Ship, Horn, and Petit Bois islands. The uniquely T-shaped Cat Island supports seagrasses on both the north and south shores as they are protected from the southeast summer winds by its north-south oriented sand spit at the east end of the island. The NCI also stretch in a north-south direction with seagrasses established on the protected western side, but SAVs are not found along the eastern beaches exposed to the open GOM. In the Chandeleur Sound to the lee (west) of the islands, substrate changes from sandy offshore to more muddy inshore, supporting all five species of seagrasses, with

T.testudinum, *S. filiforme,* and *H. wrightii* occurring more offshore, whereas *R. maritima* and *H. englemanni* are found in the nearshore shallows (Pham and Biber 2013).

Temporal Changes in Seagrass Area in the Mississippi and Chandeleur Sounds

There are several factors thought to be responsible for changes in seagrass area in the MSS over time. Eleuterius (1989) discussed hurricanes, salinity depression, low winter water temperatures, and sand bar movement as obvious causes of seagrass decline. Moncreiff (2007a, 2007b) indicated that the overall decline in water quality, from cumulative effects of human activities in the coastal environment, was also potentially responsible for the apparent ongoing reduction in seagrass area in the MSS between 1969 and the late 1990s. More recently available data suggest that loss of land area on the MS barrier islands and the NCI also has an important impact resulting in reductions to both PSGH and VSA, for example the case of Petit Bois Island presented in Carter et al. (2011).

Change of Species Composition

One of the main reasons cited for widespread loss of *T. testudinum* and *S. filiforme* along the MS barrier islands (Group 2) was severe salinity depression following increased rainfall and ten openings of the Bonnet Carré Spillway (January 28 – March 16, 1937; March 23 – May 18, 1945; February 10 – March 19, 1950; April 8 – June 21, 1973; April 14 – 26, 1975; April 17 – May 31, 1979; May 20 – June 23, 1983; March 17 – April 17, 1997; April 11 – May 8, 2008; and May 9 – June 20, 2011) designed to protect the city of New Orleans, LA, from river flooding. Even though overflow of freshwater from the Mississippi River via Lake Pontchartrain and Lake Borgne into the western MSS was a historically natural phenomenon, Eleuterius (1987) argued that the spillway acted as a concentrated point source of large volumes of freshwater discharge, which caused a more dramatic salinity perturbation than previous natural distributions of freshwater. Salinities in the MSS, especially westward, can remain depressed for months after the openings of the spillway (Moncreiff 2007a). The four halophilic seagrass species, *Halophila engelmannii, H. wrightii, S. filiforme*, and *T. testudinum*, which were found historically in the MSS (Humm 1956, Eleuterius 1973), require a relatively high salinity of at least 15-20 ppt for survival. Their demise in the MSS was highly correlated with the frequent openings of Bonnet Carré Spillway during 1973-1983 following damage caused by Hurricane Camille in 1969 (Eleuterius 1987, Moncreiff 2007a). Among these species, *H. engelmannii*, is the most sensitive to disturbance and has not been reported to occur in the MSS since 1972 (Eleuterius 1989). *Syringodium filiforme* and *T. testudinum* were once abundant in the MSS and have almost completely disappeared except for a few small patches on the north shore of Petit Bois Island and in Ranger Lagoon on Horn Island. *Halodule wrightii* is currently the dominant species remaining in the MSS, in large part as a result of its rapid growth and tolerance to a wide range of salinity (Eleuterius 1989).

Seagrass communities along the mainland (Group 1) and NCI (Group 3) did not experience similar species loss but change in species composition has also been documented. Cho et al. (2009) and Cho et al. (2017) reported a relative increase in the opportunistic *R. maritima* at the expense of *H. wrightii* at GNDNERR in 2006 and 2007 following Hurricane Katrina, although it does not appear that this was a permanent change. At Waveland, Eleuterius (1973) initially described an *H. wrightii* population but only *R. maritima* was recorded in subsequent years. It seems likely that salinities in these waters are actually too low (Eleuterius 1976) to support *H. wrightii* and that the earlier report of its presence at Waveland resulted from an error in species identification. Though less information is available, NCI seagrass meadows are also suspected to have undergone changes in the relative abundances of the five species due to salinity depression following high rainfall and the openings of Bonnet Carré Spillway.

Change of Seagrass Habitat and Seagrass Area

The seagrass area and suitable habitat have changed at all study sites but possibly for different reasons. In Group 1 at Waveland, the area where seagrass occurred has been excluded from beach renourishment since the 1980s. Importantly, the location shift of the seagrass polygons in the Waveland maps do not indicate a true change in the geolocation of suitable habitat and seagrass beds. Although the 1969 map indicates the general location where seagrasses were thought to have occurred by a single large polygon, the 1992 map shows multiple polygons to denote the patchy coverage. The smaller 1992 polygons also did not indicate the exact number or locations of patches (J. D. Caldwell, personal communication); rather they were meant to convey that seagrass was less abundant than in 1969. The 1998 Waveland seagrass map created in this study was correctly georeferenced and shows the true location of the seagrass patches, rather than the previously incorrect representations of location shown in the hand-drawn 1969 and 1992 maps (Eleuterius 1973, Moncreiff et al. 1998). Obviously, any determinations of VSA in Waveland from the hand-drawn 1969 and 1992 maps are inaccurate. Towards the east in GNDNERR, seagrasses at the South Rigolets and Grand Batture sites were once protected by the Grand Batture Islands. However, by 1980, all remnants of this island chain were gone (Otvos 2007) exposing the shoreline to increased wave erosion, which partly helps to explain the reduction in seagrass habitat from 1969 to 1992 in GNDNERR. Again, no accurate calculations of VSA can be made from these two mapping efforts due to the generalized nature of the polygons that were hand-drawn on paper maps.

The MS barrier islands in Group 2 have continued their historical trend of westward movement and continual land loss (Otvos and Carter 2008, 2013), which in turn has caused changes in PSGH and VSA. The three most important morpho-dynamic processes associated with barrier island translocation are (1) unequal lateral transfer of sand related to greater up-drift erosion compared to down-drift deposition, (2) barrier island narrowing resulting from erosion of both the GOM- and Sound-side shores, and (3) island segmentation related to storm breaching, as with the division of Ship Island into East and West Ship islands in 1969 (Morton 2007, 2008).

First, the dominant westward alongshore littoral sediment transport results in net erosion on eastern ends and net accretion on western ends of the MS barrier islands, with the exception of Cat Island. The erosion rate on the eastern ends exceeds the accretion rate on the western ends, leading to a progressive land loss and westward movement of the islands (Figure 4). When there is no longer adequate substrate supply to maintain the eastern tip, seagrass decline is rapid along this portion of the island. On the western ends, seagrass establishment and growth do not stabilize new sediment deposits quickly enough to match rates of seagrass loss on the eastern end. Thus, over time, there is a net loss not only of PSGH but also of VSA, as illustrated by the Horn Island case study. Continued habitat loss in the eastern segment, a relatively low rate of habitat gain on the western end, and the relatively stable middle part of Horn Island help to explain the observed decadal-scale changes in VSA (Figure 4). Human activities, such as dredging navigational channels to accommodate increasingly larger vessels, may disrupt alongshore sediment transport by trapping sand (Morton 2007, 2008). These channels require continuous maintenance-dredging. For example, Petit Bois Island continues to shorten in the east-west direction but has stopped migrating because westward littoral drift falls into the Pascagoula Ship Channel adjacent to its western end (Morton 2008). Not only that the dredging process would directly affect the seagrass bed through disturbance, sand burial, and reduced water clarity, but a decline in seagrass on Petit Bois Island since 1940 also corresponds with a decline in above-water island land area (Carter et al. 2011). Recently, the commercial operation of the expanded Panama Canal starting on June 26, 2016 allows New Panamax ships that are larger and more than double in cargo capacity. This has been creating demand for ports around the world to undertake renovations, which involves dredging and blasting, to accommodate New Panamax vessels. Mobile, AL has deepened its harbor and Port of Pascagoula will likely follow, which would soon show its effect on the seagrass bed in the Sounds.

Second, under the effects of summer tropical cyclones and winter cold fronts, MS barrier islands have been progressively narrowing as a result of long-term beach erosion on both the GOM and Sound sides. The apparent southward movement of seagrass beds, e.g., on Horn Island (Figure 4), is partly because of this north-south narrowing process. Third, storm impacts on shoreline fragmentation depend on the orientation and other geomorphological characteristics of each island. In the MSS, Ship Island is the most vulnerable to storm-driven land losses as topographic and bathymetric boundary conditions focus wave energy onto the island (Morton 2007, 2008). Ship Island was breached into East Ship Island and West Ship Island after Hurricane Camille in 1969, which explains the steep drop in VSA observed in 1975 (Table 2). This breach was substantially widened during Hurricane Katrina in 2005. Currently the U.S. Army Corps of Engineers plans to close this breach by pumping large volumes of sediment into the gap (Vittor and Associates, Inc. 2014). It remains to be seen whether seagrasses will re-establish once the rejoined island is stabilized.

In Group 3, the NCI form a low-profile island chain that undergoes breaching, thinning in an east-west direction, and erosion on the northern and southern tips; all of these geomorphic processes affect the area of seagrass habitat and the location of the seagrass beds. Tropical cyclone frequency is thought to dominate the long-term evolution of the Chandeleur Islands (Fearnley et al. 2009) with a high return frequency causing faster rates of land loss. While the GOM and Chandeleur Sound shorelines are both migrating landward, the GOM shoreline is migrating twice as fast as the Sound side, causing net deterioration of the island arc (McBride et al. 1992). The lack of new riverine sediments to help naturally renourish the islands is likely to be the cause for their ultimate demise in the future, much like has already occurred to the former Grand Batture Islands at GNDNERR, with the attendant loss of extensive seagrass meadows.

The effect of hurricanes on seagrass, however, may not necessarily be devastating. Cho et al. (2009) reported that both *R. maritima* and *H. wrightii* at GNDNERR were more abundant in 2006 than in any other years in the study period between 2005-2008; they suggested that physical disturbance by Hurricane Katrina in 2005 might have helped to expose the deep-buried seeds and promote their germination. Additionally, Anton et al. (2009) found no effect of Hurricane Katrina on *H. wrightii* leaf density and biomass at two weeks and again at one year after landfall at nearby Sandy Bay, AL. Also on the MS barrier islands, Eleuterius (1971) found that Hurricane Camille eroded and destroyed all the seagrass in the island passes, but the seagrass beds on the north side of the islands were not significantly disturbed and grew more robustly during the summer following the hurricane than in the year prior to the hurricane. He suspected that wind and waves disrupted the organic sediments and inland rains brought down more nutrients than normal, stimulating seagrass growth. Carter et al. (2011) found no apparent impact of Hurricanes Camille and Katrina on VSA. Even though Hurricane Katrina's extreme storm surge caused temporary flooding of the entire MS barrier islands that resulted in massive erosion and local accretion of terrestrial sediments (Fritz et al. 2007), it did not seem to devastate the seagrass beds (Heck and Byron 2006). The effect of hurricanes and tropical storms on seagrasses deserves further study in this region.

Potential Problems When Comparing Maps from Various Sources

There are numerous possibilities for misinterpretation when comparing among seagrass maps from different sources. For a given site, seagrass occurrence in one map, but not in the others, may be due to lack of survey information rather than any seagrass loss or gain. This is the case for Deer Island, Dog Keys Pass, Round Island, Middle Ground (included only in the 1969 map of the MS Sound), North Islands and New Harbor Islands (only in the 1992 map of the Chandeleur Islands). Since there were no comparable data available in the later maps, these sites were excluded from subsequent area calculations (Table 2). It is not clear whether the lack of seagrass in later maps is because those areas were just not surveyed, or whether there was indeed a complete loss of seagrasses. With respect to seagrass data for the GNDNERR, Eleuterius (1973) did not survey at the more eastern locations in Middle Bay and Grand Bay, but survey data for these two locations are included in the later maps. Depending on the investigator, GNDNERR seagrass maps may include only those areas that fall within the GNDNERR boundaries, or they may extend eastwards beyond the MS – AL state line. Thus, attention should be paid to discrepancies between political boundaries and actual seagrass range when comparing among maps to determine change over time. Seagrass coverage in northeastern Grand Bay was not included in our area determinations because of this surveying problem (Table 2).

A less obvious, but potentially major source of misinterpretation, is due to differences in mapping objectives and methods among studies. Eleuterius (1973, 1979) and Vittor and Associates, Inc. (2014) generated maps of seagrass extent, whereas Moncreiff et al. (1998) and USGS-NWRC (1998a, 1998b, 2003) generated maps of seagrass coverage; both approaches mapped general seagrass locations using relatively coarse-scale polygons that included unspecified large areas of unvegetated sand bottom. This approach may generally be more appropriate for estimating the extent of seagrass habitat rather than VSA and does not allow for the quantification of patch size and shape. More recent studies (e.g., Carter et al. 2011) have employed object-based mapping with manual pixel editing to accurately identify the boundaries of individual seagrass patches from vertical aerial image data with a horizontal resolution of 1 m or better and enable calculation of VSA. Such present-day methods, enabled by advances in computer processing power and software, are much more effective in quantifying the presence, shape, and area of the many small patches that often comprise a seagrass population. Direct comparisons of seagrass area among studies using the different techniques (seagrass extent vs. seagrass coverage vs. VSA) may yield conclusions of dramatic change over time (e.g., compare between 1969 and 1971 Horn Island data, Table 2), simply because of the various mapping methods employed. However, such conclusions may be primarily a consequence of improper comparisons between, for example, the seagrass extent compared to VSA on Horn Island in 1969 and 1971, respectively. The problem may be somewhat reduced with respect to the continuous seagrass meadows on the NCI versus the patchier seagrass beds on the MS islands. This issue has also been considered for seagrass mapping in Florida (Dixon and Perry 2003) and New South Wales, Australia (Meehan et al. 2005). Resampling from the original aerial imagery and a common definition of the mapping unit along with the study objective will be required to make maps at different scales and resolutions comparable. Also important is defining a threshold, such as an *a priori* minimum difference (e.g., 5-10%) between time points, to distinguish between real changes in VSA versus potential mapping errors (Meehan et al. 2005).

Yet another source of error comes from comparing data collected in different seasons, as leaf biomass and shoot density can vary substantially during the year. For instance, seagrasses on the barrier islands proliferate from May to October and are at their peak canopy density during the late summer and fall months. *Ruppia maritima* along the MS mainland coast exhibits bimodal peaks in density in late spring and late fall (Moncreiff 2007b). Late fall, with high leaf biomass and relatively high water clarity, is considered the best time of the year for mapping seagrasses in the MSS. For the purposes of decadal-scale change detection, it is important to compare maps made only within the same season so that changes in VSA among years are not confounded with seasonal biomass changes.

In summary, studies in the MSS indicate both seagrass loss and gain depending on location, but decline has remained the major overall trend consistent with earlier reports (Eleuterius 1979; Heck et al. 1996; Moncreiff et al. 1998; Moncreiff 2007a; Carter et al. 2011). The early 1970s was suggested as the beginning of seagrass decline in the MSS, largely due to prolonged low-salinity incursions from frequent Bonnet Carré Spillway openings. Eleuterius (1989) estimated that only about 30% of the vegetated area found in 1969 remained in 1989, despite similar amounts of PSGH. The 1992 estimate of seagrass coverage (809 ha) by Moncreiff et al. (1998) represented an 84.6% loss when compared to the prior 1969 estimates (5,252 ha) by Eleuterius (1973), but this is confounded by the inaccurate comparison of seagrass coverage in 1992 with seagrass extent in 1969. Moncreiff et al. (1998) reported a 19.6% reduction in PSGH based on a 2-m depth limit. Moncreiff (2007b) already noticed that the methodological differences in the two maps prepared by Eleuterius (1973) and by the USGS-NWRC (1998b) precluded any direct comparisons outside of the loss of species and recognition of a decline in seagrass habitat.

The compilation of various data in this study confirms the previously reported trend of seagrass decline (both in potential habitat and seagrass area), but the rate of change is not as high as reported previously based on inappropriate comparisons of maps created using different mapping methods. Change in seagrass area varies dramatically among the three study groups and even among islands within Group 2, where the best data are currently available. On the one hand, as shown in our case study of Horn Island, direct estimation of seagrass area change can be highly misleading if accepted *prima facie* without understanding the different mapping techniques used. Two examples of this problem are the incorrect statements in the literature, based on the study by Moncreiff et al. (1998), that “seagrasses in Mississippi suffered a decline of between 85 and 89 percent over 23 years” (Onuf et al. 2003), and that “the Mississippi Sound, Miss., has lost nearly all of its seagrasses – over 4,500 of 5,250 ha (11,120 of 12,973 acres) of seagrasses – since 1969” (Beck et al. 2007). On the other hand, development of new mapping techniques allows researchers to more thoroughly analyze patch fragmentation and obtain more accurate rates of change in VSA over time, which would be extremely helpful for better management of this declining natural resource. The available data are still limited because these very accurate maps often require carefully planned and expensive data acquisition, as well as time-consumed mapping efforts.

CHAPTER III - VEGETATED SEAGRASS AREA IN THE MISSISSIPPI-CHANDELEUR SOUND: CHANGE ANALYSIS ACROSS SCALES

Introduction

Seagrasses are an important natural resource that provide many ecosystem services, and thus have attracted much research attention, including coverage assessment and change monitoring over time. A number of methods have been used to map seagrass distribution and quantify its area, from transect surveys, to grid sampling, to remote sensing. With the integration of GIS, remote sensing has become popular in studies of natural resources due to its capability to capture a large spatial extent with reduced cost, and to overlay maps of multiple years for change detection. This gave birth to long-term trend studies of seagrasses in places such as Corpus Christi Bay, Texas (Pulich et al 1997), in Tampa Bay, Florida (Robbins and Bell 2000), in Denmark (Frederiksen et al. 2004a, b), in the GOM (Handley et al. 2007), and in the MSS (Carter et al. 2011).

It is essential, yet difficult to know if any change detected is significant or simply an artifact of image processing, mapping errors, and post-classification comparisons. Therefore, mapping data coming from various sources should be generalized to a similar level of thematic content and spatial scale before change detection (Petit and Lambin 2002). Without this prior equalization, any time-series analyses for change detection would be compromised by imprecision and inconsistencies across the various data sources. This holds true for all natural resource studies, but has been understated in the literature to date. Some seagrass studies have started to recognize the problem of different mapping methodology and scales when comparing data, such as Dixon and Perry (2003) in Florida, Meehan et al. (2005) in New South Wales, Australia, and Chapter II in this dissertation. Still an accepted methodology is lacking for generalizing seagrass maps derived from multiple sources to provide more confidence in natural resource change detection and management.

Scale, a widely recognized problem in ecology since the 1980s (Bailey 1985, Meentemeyer and Box 1987, Meentemeyer 1989, Turner et al. 1989a, Turner et al. 1989b, Wiens 1989, Levin 1992, O'Neill et al. 1996, Qi and Wu 1996, Meisel and Turner 1998, Wu and Qi 2000, Turner et al. 2001b, Wu 2004, Wu and Li 2006), especially needs to be considered with caution during generalizing and comparing maps. Scale refers to the spatial or temporal dimension of an object or a process, and is characterized by (1) grain - defined as the finest level of spatial resolution possible within a given data set, and (2) extent - defined as the size of the overall study area (Turner et al. 1989a, Wiens 1989, Turner et al. 2001b). While the previous chapter concerned itself more on thematic content (seagrass extent vs. seagrass coverage vs. VSA), this chapter focuses in more detail on the scale problem when attempting to detect change over time using VSA.

Various landscape metrics or indices are developed to measure landscape composition and configuration, most can be calculated in landscape analysis packages such as FRAGSTATS (McGarigal et al. 2012, McGarigal 2014). Total area (referred in this document as VSA) is just one among many FRAGSTATS metrics. The landscape metrics are highly scale-sensitive (Turner et al. 1989b, Wu 2004), hence mapping scale (grain and extent) affects the ability to detect and quantify landscape pattern.

This chapter aims to: (1) examine the effect of map grain size on VSA and selected landscape metrics as a basis to choose a common resolution to work with, (2) estimate annual change and rate of change in VSA adapting the approach from Waycott et al. (2009) to compare seagrass changes in the MS-C with the global estimates, and (3) assess changes in spatial distribution and extent of VSA after correcting for resampling errors.

Materials and Methods

Effect of Changing Grain Size on Vegetated Seagrass Area and Selected FRAGSTATS Metrics

The effect of changing grain size on VSA was studied by resampling seagrass data to different spatial resolutions and comparing the resulting calculated seagrass area. VSA data on the MS barrier islands in 1940-2008 (Carter et al. 2011) and on the NCI in 1999-2005 (Bethel et al. 2006, Bethel and Martinez 2008) were compared for this study. Seagrass patches in this data set were originally mapped at fine spatial resolutions of 0.3 – 2.4 m. Original seagrass vector files were first converted to raster in ArcGIS ver. 10.2 (ESRI, Redlands, CA) with progressively larger output cell sizes (changing grain size) while keeping the spatial extent constant. Each new raster file was converted directly from the original vector files rather than from the next lower raster resolution to avoid cumulative errors occurring through the aggregation process. Horn Island data were tested most exhaustively by resampling to output resolutions of 0.5, 1.0, 2.0, 2.5, 3.0, 4.0, 5.0, and 10.0 m pixel sizes. Data from other MS islands were resampled to resolutions of 1.0 m, 2.0 m, 2.4 m, and 2.5 m, while NCI data were resampled to resolutions of 2.0 m, 2.4 m, 2.5 m, and 3.0 m (NCI data files reached the computer storing and processing limit if resampled to a resolution of less than 2.0 m).

Sixteen FRAGSTATS metrics were chosen arbitrarily to examine the effect of grain size on landscape metrics, which includes: VSA (or TA), area-weighted mean patch area (AREA_AM), largest patch index (LPI), total edge (TE), edge density (ED), area-weighted mean patch radius of gyration (GYRATE_AM), perimeter/area fractal dimension (PAFRAC), area-weighted mean fractal dimension index (FRAC_AM), area-weighted mean related circumscribing circle (CIRCLE_AM), number of patches (NP), landscape division index (DIV), and area-weighted mean Euclidean nearest neighbor distance (ENN_AM). Definitions and formulas of these FRAGSTATS metrics can be obtained in McGarigal (2014). These 16 metrics were calculated for each of the resampled seagrass maps described above.

An analysis of variance (ANOVA) was used to test the effect of year and grain size (no interaction term) on VSA followed by a Tukey's post-hoc test. All statistical tests were performed in R ver. 3.1.1 for Windows (http://www.r-project.org). Differences in VSA and its relative percentage were calculated among all pairs of map dates for each island. The threshold of detectable VSA change was estimated as the percentage of VSA beyond which Tukey's post-hoc groups were identified as being significantly different.

Rate of Change of Vegetated Seagrass Area over Time

Spatial data resampled across all islands and years to a common 2.5 m grain size were then used in the calculation of annual rate of VSA change, interpolation of unobserved VSA in decades when there was no data, and estimation of net change in VSA over time. The following procedure was modified from Waycott et al. (2009). First, formulae that have been utilized to estimate annual rate of forest cover change (Puyravaud 2003) were borrowed for seagrass change calculation. Annual rates of total VSA change were calculated for each pair of map dates as follows:

$$r = \left[\frac{1}{t_2 - t_1} \ln\left(\frac{VSA_2}{VSA_1}\right)\right] \times 100 \quad (\%\ per\ year) \qquad \text{(Equation 1)}$$

where VSA_1 and VSA_2 are the total VSA (ha) at time t_1 and t_2, respectively. Estimated VSA of an unrecorded year within the study period was interpolated using the annual rate of total VSA change (r) (Equation 1). Another measurement, annual VSA change

$$R = \frac{VSA_2 - VSA_1}{t_2 - t_1} \quad (ha\ per\ year) \qquad \text{(Equation 2)}$$

was reported alongside r but not used in further statistical testing. At the decadal time scale, change in VSA on each MS barrier island was calculated as the difference between estimated VSA in a 10-year interval (1940-1950, 1950-1960, ..., 2000-2008). The resulting losses (negative change) and gains (positive change) were then summed separately to attain net change in VSA on all MS islands over the entire study period. Yearly net changes within the 2000s were also calculated in a similar way, but using 1-year intervals.

Changes in Spatial Distribution and Extent of Seagrass

Spatial data resampled to a common 2.5 m grain size were overlaid on pairs of map dates to visualize locations of VSA spatial changes. Comparisons tested were (1) the beginning and ending map dates for each island over the entire study period, as well as (2) within the 2000s decade, and finally (3) pairs of successive map dates (see Figure 5 with Petit Bois Island data as an example).

Overlaying paired seagrass layers in ArcGIS allowed visualization of locations where seagrass had increased, declined, or remained unchanged, and quantification of the area affected. Relative percentage of spatial change was calculated from these area measurements as follows:

$$\%Change_{relative} = \frac{Gain + Loss}{Gain + Loss + Unchanged} \times 100\% \quad \text{(Equation 3)}$$

Percent change caused by resampling a seagrass layer from its original grain size to 2.5 m was considered a displacement error ($error_{displace}$). Detectable percentage of change between compared map dates was corrected using percentage of correct placement (1- $error_{displace}$) in each overlay map as follows:

$$\%Change_{detectable} = \%Change_{relative} \times \left(1 - error_{displace1}\right) \times \left(1 - error_{displace2}\right)$$

(Equation 4)

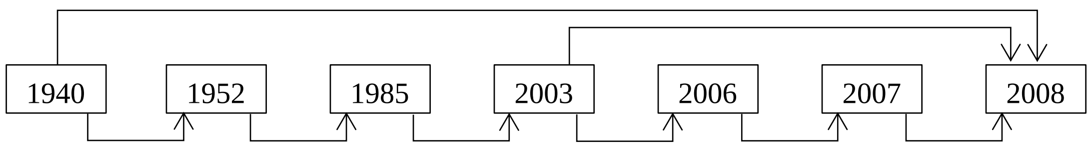

Figure 5. Design of change detection for Petit Bois Island seagrass data.

Note. Data for other islands were paired similarly.

Results

Effect of Changing Grain Size on Vegetated Seagrass Area and Selected FRAGSTATS Metrics

Each FRAGSTATS metric behaved differently across mapping resolutions (Figure 6). Resampling to a coarser resolution (2 m, 4 m…) caused more changes in the calculated metrics (overestimating VSA, underestimating NP and TE…) than resampling to a finer resolution did. Total edge, ED, FRAC_AM, and NP appeared to negatively correlate with resolutions while ENN_AM and PAFRAC displayed a positive correlation with increasing grain sizes. Other metrics (total area or VSA, AREA_AM, LPI, GYRATE_AM, CIRCLE_AM, and DIV) fluctuated but did not show any clear trend along changing resolutions. However, resampling to an even-numbered scalar (2, 4 …) of the original cell size inflated VSA while resampling to an odd-numbered scalar of the original cell sizes did not.

Table 3

Vegetated Seagrass Area Across Grain Sizes

	Cat ($p < 0.000$)	West Ship ($p < 0.000$)	East Ship ($p < 0.000$)	Horn ($p < 0.000$)	Petit Bois ($p < 0.000$)	Chandeleur ($p < 0.000$)
1940				83 ± 10.6^{a}	56.4 ± 3.7^{a}	
1952				49.3 ± 4.4^{b}	17.1 ± 2.3^{b}	
1963		11.5 ± 0.4^{a}	20.4 ± 1^{a}			
1971				19.4 ± 0.3^{d}		
1975		2 ± 0.2^{c}	1.7 ± 0.1^{e}			
1985					18.5 ± 1.3^{b}	
1999						1496.4 ± 1.6^{c}
2000						1519.5 ± 1.8^{b}
2002						1525.8 ± 1.4^{a}
2003	21.8 ± 0.02^{b}	0.02 ± 0.0002^{e}	16.5 ± 0.02^{c}	53.8 ± 4.8^{b}	8 ± 0.02^{c}	
2004						
2005						895.6 ± 0.9^{d}
2006	25.5 ± 0.03^{b}	0.9 ± 0.1^{d}	16.1 ± 1^{cd}	84.8 ± 4.5^{a}	20 ± 1.7^{b}	
2007	76.7 ± 8.5^{a}	1.8 ± 0.2^{c}	14.9 ± 1.5^{d}	38.8 ± 1.3^{c}	17.1 ± 0.6^{b}	
2008		3.2 ± 0.01^{b}	17.9 ± 0.01^{b}		7.3 ± 0.02^{c}	

Note: VSA (mean ± standard deviation) in hectares averaged across all grain sizes (original resolution and 1.0 - 2.5 m resampling resolutions). Tukey's post-hoc test superscripts are organized by column.

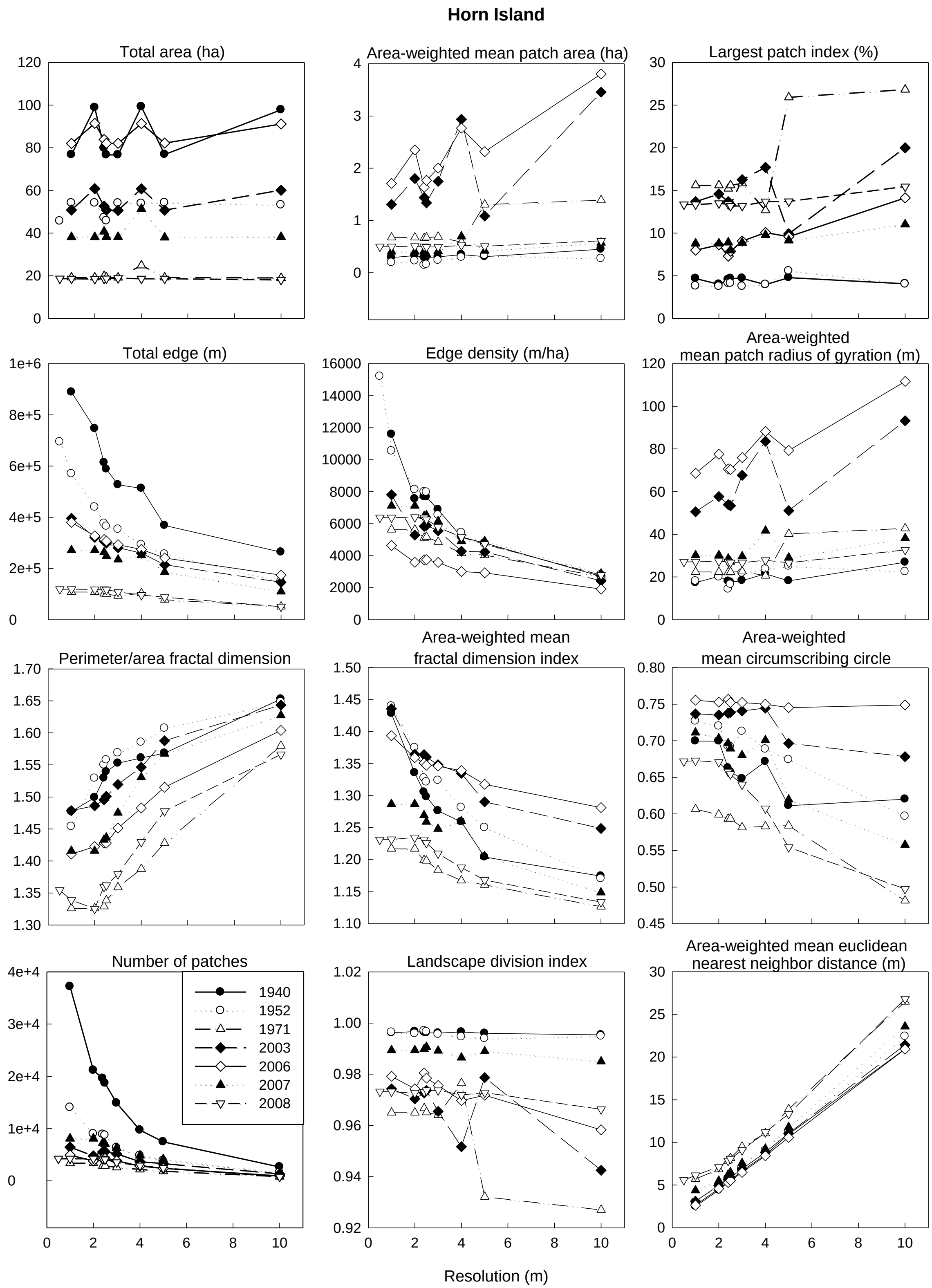

Figure 6. Effect of changing grain size on FRAGSTATS metrics on Horn Island

The VSA on each of the MS-C islands was statistically different ($p < 0.000$) among years, but not among grain sizes (Figure 7, Table 3). Mean VSA on Cat Island was not statistically different between 2003 and 2006 but then increased in 2007. Mean VSA on West Ship Island declined significantly from 1963 to 2003, before it increased in the late 2000s; the averaged VSA in 2008 was statistically greater than the mean value in 1975. East Ship Island showed no significant change in mean VSA overall from 1963 to 2008 even though there was a significant drop in 1975. The mean VSA on Horn Island was not statistically different between 1940 and 2006, however a significant trend of VSA decline from 1940-1971 was followed by a significant expansion from 2003-2006 before declining again in 2007. Meanwhile, Petit Bois Island suffered a significant seagrass loss of mean VSA from 1940-2008 and although there was some gain in 2006-2007, the 2000s were statistically the lowest point in that island's VSA record. On the NCI, the mean VSA was significantly reduced in 2005 compared to the stable period from 1999 to 2002. This overall trend in mean VSA on each island was consistent across different grain sizes and across a range of sampling window sizes.

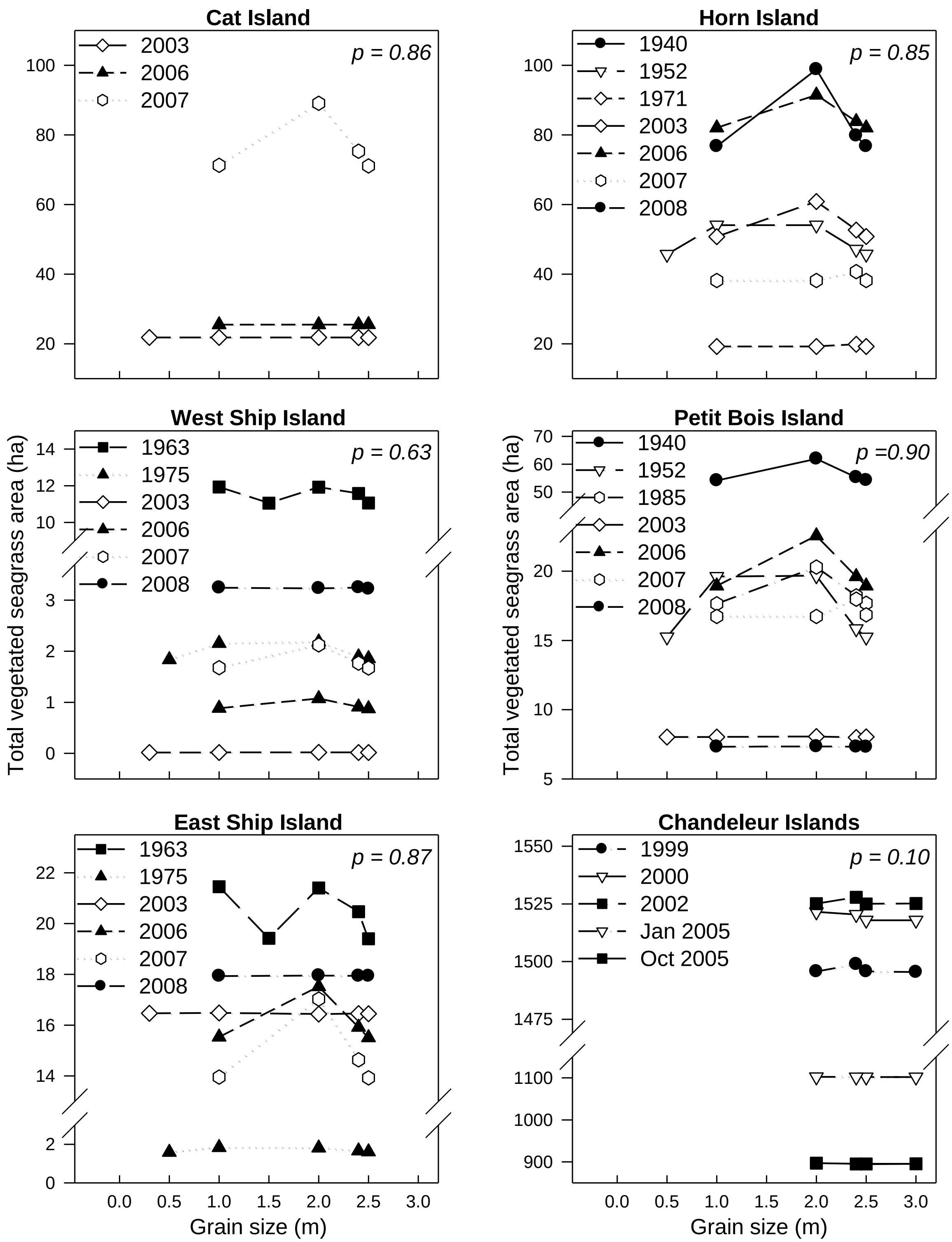

Figure 7. Effect of changing grain size on vegetated seagrass area.

Note. Horn Island data in 2008 covered only 70% of the island and were not included in the analysis. No significant effect of resolution on total VSA was found on any islands (all p > 0.05).

Table 4

Effect of Resampling (from Original Resolutions to 2.5 m) on FRAGSTAT Metrics

	Year	Res	TA	AR	LPI	TE	ED	GY	PAF	FR	CIR	NP	DIV	EN
Cat	2003	0.3	0			-	-							
	2006	1	0	+	0	-	-	+	0	-	-	-	0	+
	2007	2	-	0	--	-	0	+	0	0	-	-	0	+
West Ship	1963	1.5	0	+	0	-	-	+	0	-	-	-	0	+
	1975	0.5	0	+	+	-	-	+	+	-	-	-	0	++
	2003	0.3	0	++	+	--	--	+	+	-	--	-	0	+++
	2006	1	0	+	+	-	-	+	+	-	-	-	0	+
	2007	1	0	+	0	-	-	+	+	-	-	-	0	+
	2008	2.4	0	0	+	0	0	0	0	0	0	0	0	0
East Ship	1963	1.5	0	+	-	-	-	+	0	-	-	-	0	+
	1975	0.5	0	+	0	-	-	+	0	-	-	-	0	+
	2003	0.3	0	0	0	-	-	0	0	0	0	0	0	0
	2006	1	0	+	0	-	-	0	0	-	-	-	0	+
	2007	1	0	+	-	-	-	+	+	-	-	-	0	++
	2008	2.4	0	0	0	0	0	0	0	0	0	0	0	0
Horn	1940	1	0	++	0	-	-	++	0	-	-	-	0	++
	1952	0.5	0			-	-							
	1971	2	0	+	0	-	-	+	0	0	0	-	0	+
	2003	1	0	+	0	-	-	+	0	-	-	-	0	++
	2006	1	0	+	0	-	-	+	0	-	-	-	0	++
	2007	2	0	+	-	-	-	+	0	0	0	-	0	+
	2008	2.4	0	0	0	0	0	0	0	0	0	0	0	0
Petit Bois	1940	1	0	+++	0	-	-	++	+	-	-	--	0	++
	1952	0.5	0	+++	+	--	--	++	+	-	-	--	0	+++
	1985	1	0	+	0	-	-	0	0	-	-	-	0	+
	2003	0.5	0	+	+	-	-	0	0	-	-	-	0	++
	2006	1	0	+	++	-	-	+	0	-	-	-	0	+
	2007	2	0	+	++	-	-	+	0	0	0	-	0	+
	2008	2.4	0	0	-	0	0	0	0	0	0	0	0	0
NCI	1999	2	0	-	0	-	-	-	0	-	-	+	0	-
	2000	2	0	0	0	0	0	0	0	0	0	0	0	-
	2002	2	0	-	0	-	-	-	0	-	-	+	0	0
	2005J	2.4	0											
	2005O	2.4	0											

Note. Res = original resolution (m) of input data. AR = AREA_MN, GY = GYRATE_MN, PAF = PAFRAC, FR = FRAC_MN, CIR = CIRCLE_MN, EN = ENN_MN, the rest of abbreviations for metrics are as usual. NCI = Northern Chandeleur Islands, 2005J = January 2005, 2005O = October 2005. Plus signs indicate positive bias when converting from original resolution to 2.5 m: (+) 5-50%, (++) 50-100%, (+++) ≥ 100%. Minus signs indicate negative bias: (-) 5-50%, (--) 50-100%, (---) ≥ 100%. Zero (0) indicates no or little bias (0-5% of differences). Empty cells are unavailable data due to computer memory limit.

Resampling all data to 2.5 m resolution appeared to preserve FRAGSTATS metrics well. This conversion resulted in little change in VSA (less than 3% change reported in most seagrass layers), PAFRAC, and DIV (Table 4). Mean seagrass patch area, radius of gyration, and nearest neighbor distance for MS barrier islands had a positive bias (occasionally being quite large) while the values for the NCI showed a negative bias. The more different the original resolutions were from the targeted resolution of 2.5 m, the larger the reduction in TE was. Edge density also showed negative bias, except for Cat Island in 2007. Fractal index and related circumscribing circle suffered a negative bias in conversion. The number of seagrass patches on MS barrier islands reduced when resampling but the opposite happened with NCI data.

Rate of Change of Vegetated Seagrass Area over Time

From 1940 to 2008, VSA decreased on each MS barrier island and on all islands combined, with the exception of Cat Island (Table 5, Figure 8). The estimated net loss in the MSS represented 63.1% of the original VSA of 161.3 ha in 1940 down to 59.5 ha in 2008 (excluding Cat Island, Figure 8a). Total VSA for all five barrier islands combined in 2008 (59.5 + 71.1 = 130.6 ha, including Cat Island) was equal to the amount of seagrasses in 1940 on Horn and Petit Bois Islands alone (130.8 ha, Figure 8a, d). The annual rate of VSA decline (r) on the MS islands from 1940 to 2008 was 1.5% per year (Table 5), equivalent to an annual change (R) of approximately 1.5 ha per year. The rate of VSA loss per island was less than 3% per year (median of 2% per year and a mean of 1.8% per year). From 1963 to 2008, VSA decline on West Ship Island was five times higher (7.8 ha vs. 1.5 ha) and occurred at a rate fifteen times higher (2.7% per year vs. 0.18% per year) than the loss on the adjacent East Ship Island (Table 5). Horn and Petit

Bois islands each lost approximately 46 ha of seagrasses from 1940 to 2008 (Figure 8b) but the rate of decline was more than double that on the latter, also smaller, island (1.3% per year and 2.9% per year, respectively). West Ship and Petit Bois islands lost VSA at almost 3% per year, more than double the loss rate on Horn Island, and more than 15 times that of East Ship Island (Table 5).

Table 5

Overall Changes in Vegetated Seagrass Area and Rate of Change

	Entire study period			Within the 2000s		
	ΔVSA (ha)	*r* *(per yr)*	Time period ($t_1 \rightarrow t_2$)	ΔVSA (ha)	*r* *(per yr)*	Time period ($t_1 \rightarrow t_2$)
Mississippi Sound (exclude Cat)						
Overall	-101.8	-1.5%	1940-2008	-8.9	-1.7%	2000-2008
Median	-26.7	-2.0%	1940-2008	0.7	0.7%	2000-2008
Cat	na	na		49.3	29.6%	2003-2007
West Ship	-7.8	-2.7%	1963-2008	3.2	59.3%	2000-2008
East Ship	-1.5	-0.2%	1963-2008	5.1	4.2%	2000-2008
Horn	-45.6	-1.3%	1940-2008	-15.3	-5.0%	2000-2008
Petit Bois	-46.9	-2.9%	1940-2008	-1.9	-2.8%	2000-2008
Chandeleur	na	na		-600.6	-8.6%	1999-2005

Note. ΔVSA represents change in total vegetated seagrass area (ha), *r* represents annual rate of total vegetated seagrass area change (per year). Positive values indicate area gain while negative values indicate area loss.

At the decadal time scale, each MS island experienced a different trajectory of VSA change (Figure 8b). West Ship Island had the highest annual rate of total area loss at -2.7% per year (or -0.05 ha per year) from 1963 to 2000s, about ten times as fast as the decrease on East Ship Island at -0.2% per year (or -0.0046 ha per year, Table 5). East Ship Island lost VSA from 1963 to 1980 but then recovered slowly back to the value in 1963 (Figure 8b); the rate of recovery from 1980 to 2008 of 4.2% per year (or 0.08 ha per year) was lower than the previous rate of loss, however (Table 3). In contrast, after 1975, West Ship Island still continued losing VSA until the 2000s as East Ship Island underwent recovery (Figure 8b). Like East Ship Island, West Ship also gained VSA within the 2000s (Figure 8e), but at an extremely high rate of 59.3% per year (or 0.4 ha per year, Table 5) although this accounted for a net gain of only 3.2 ha. This probably reflects a high growth rate when the population is near to minimum point. Horn Island lost VSA at an annual rate of -4.5% per year (or -0.04 ha per year) in 1940-1970, regained VSA in the subsequent three decades at a lower rate of 2-3% per year, and then lost VSA again in the 2000s at a higher rate of -5% per year. On Petit Bois Island, VSA decreased at a high annual rate of -10.6% per year (or -0.10 ha per year) from 1940 to 1952, recovered slightly from 1952 to 1985, and then continued decreasing at -2% to -4% per year, with the lowest observation in 2008. Annual rates of change were generally smaller than 5% per year (Table 5), with a few extreme exceptions, notably at Cat Island and West Ship Island during the 2000s. From west to east the lowest amounts of VSA calculated on each island were in 2000 (West Ship), 1980 (East Ship), 1970 (Horn), and 2008 (Petit Bois) while the four islands combined had the least amount of VSA in the decade of 1970-1980 (Figure 8a, b). Overall, the MS barrier islands lost more seagrass and at a much higher rate in the first part of the study period, from 1940 to mid-1970s, than in the later years (Figure 8c). Prior to the 1970s, seagrass loss dominated on the MS islands and an increase in VSA was not recorded at any of the sites before 1960 (Figure 8c). The trend reversed when seagrass recovery was enhanced and exceeded losses from 1960 to 2000, but declines have occurred again in the last decade (Figure 8c, f).

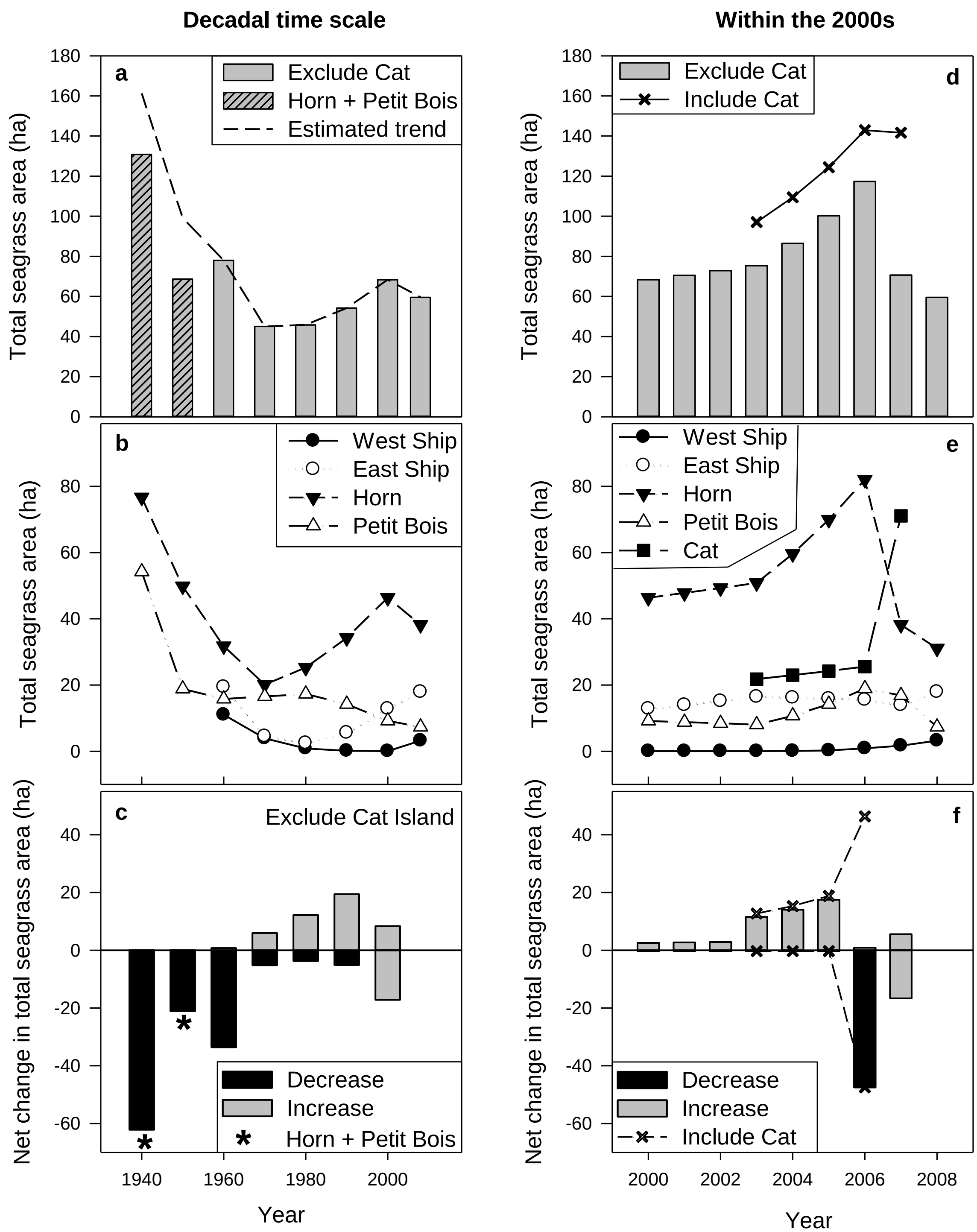

Figure 8. Interpolated vegetated seagrass area and estimated net change on the Mississippi barrier islands across and within decades.

Seagrass loss remained the predominant trend in the MS-Cs in the 2000s, but positive changes were reported at several islands (Figure 8d, e). Within the 2000s, the MS islands combined (excluding Cat Island) lost 8.9 ha of seagrass in total (13% of the 2000 amount) at a rate of 1.7% per year (or 1.1 ha per year) (Table 5). The rate of VSA change on each island in the 2000s was generally higher than the rate for the whole study period (Table 5). On all the islands combined, VSA increased from 2000 to 2006, then decreased again in the two following years (Figure 8d) with the VSA loss being higher in 2006-2007 than in 2007-2008 (Figure 8f). Seagrass gain dominated on the three western islands in the MSS, but not on the two eastern ones nor on the NCI in the 2000s (Figure 8f). All MS islands gained seagrasses in 2003-2006 (with exception of East Ship Island where it remained stable), while from 2003 to 2007 only Horn Island lost seagrasses (Figure 8e).

Total VSA on the NCI increased from 1999 to 2002 then declined in 2005 at an accelerated rate. The NCI suffered the highest VSA loss and also the highest annual rate of loss recorded within the 2000s (Table 5). The seasonal change (pre- and post-Katrina) in 2005 was higher than any yearly changes observed in the 2000s on the NCI.

To sum up, seagrass loss was the overall trend in the entire MSS and on each island in the study period (Figure 8c, f). Yearly VSA change within the 2000s was as large as the observed decadal VSA change, but in turn may have been smaller than a single seasonal change exemplified by the NCI in 2005, albeit strongly influenced by heavy losses resulting from Hurricane Katrina. The annual rate of VSA change was slightly higher within the 2000s than over the entire period from 1940 to 2008 (Table 5, Figure 8).

Changes in Spatial Distribution and Extent of Seagrasses

Table 6

Percentage of Spatial Change on Overlay Maps

Island	Year	%Change$_{relative}$	%Change$_{detectable}$
Cat	2003-2006	93.8%	44.8%
	2006-2007	92.9%	38.3%
	2003-2007	90.9%	35.8%
West Ship	2003-2006	99.9%	19.0%
	2006-2007	97.7%	35.9%
	2007-2008	82.8%	32.9%
	2003-2007	99.9%	17.9%
	2003-2008	99.9%	20.5%
East Ship	2003-2006	87.7%	47.4%
	2006-2007	86.0%	43.8%
	2007-2008	73.0%	34.9%
	2003-2007	89.2%	40.7%
	2003-2008	88.9%	45.0%
Horn	2003-2006	83.8%	45.9%
	2006-2007	81.0%	43.8%
	2007-2008	85.5%	40.4%
	2003-2007	89.4%	41.8%
Petit Bois	2003-2006	95.9%	39.7%
	2006-2007	91.1%	42.6%
	2007-2008	88.7%	38.5%
	2003-2007	95.4%	44.6%
	2003-2008	97.6%	37.4%
Chandeleur	1999-2000	57.9%	54.3%
	2000-2002	62.2%	58.4%
	2002-Jan2005	57.0%	53.3%
	Jan-Oct2005	60.4%	56.6%
	1999-Oct2005	74.7%	69.3%
	2002-Oct2005	67.6%	62.8%
Mississippi (2000s)		*90.1 ± 6.6%*	*38.6 ± 7.6%*
Chandeleur (2000s)		*63.3 ± 6.2%*	*59.1 ± 5.5%*

Table 6 (continued).

Island	Year	*%Change$_{relative}$*	*%Change$_{detectable}$*
West Ship	1963-1975	98.6%	37.0%
	1975-2003	100.0%	16.5%
	1963-2008	97.3%	45.3%
East Ship	1963-1975	99.5%	37.1%
	1975-2003	99.5%	40.7%
	1963-2008	96.4%	44.5%
Horn	1940-1952	90.8%	29.1%
	1952-1971	97.5%	40.1%
	1971-2003	97.1%	49.4%
	1940-2007	94.7%	37.1%
Petit Bois	1940-1952	93.6%	30.0%
	1952-1985	96.7%	30.3%
	1985-2003	97.1%	43.0%
	1940-2008	97.7%	47.9%
Mississippi (decadal)		*96.9 ± 2.4%*	*37.7 ± 8.6%*

Changes in seagrass spatial distribution were different from VSA trends, yet both results indicate the dynamic nature of the system studied. Relative location changes and displacement errors were much higher on the MS Islands than on the NCI (20-70% versus 3-4%, respectively). However, detectable spatial change was smaller in the former group after correction (Table 6). These values were not affected by time intervals; differences between observations from two different decades were not necessarily larger than those between two years within the same decade (See Appendix A for maps of pairs of successive observations made for each island).

Spatial distribution of VSA change throughout the study period was different on each island (Figure 9). Seagrass shrank back to the center on each of the two Ship islands, migrated westward on Horn Island (higher eastern loss than western gain), and were lost at the eastern end of Petit Bois Island with no gain on the western end. The seagrass landscape also flattened in the north-south direction on West Ship, Horn, and Petit Bois islands (Figure 9). These observations indicate a generalized trend for contraction of the seagrass extent over the study period.

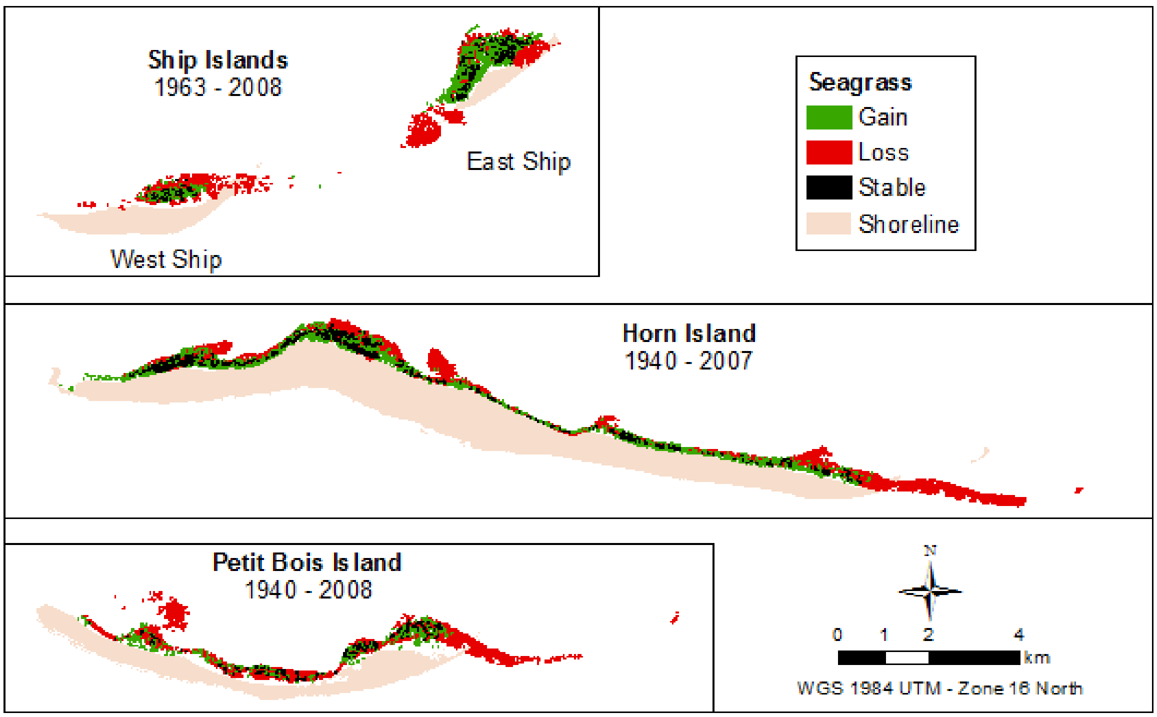

Figure 9. Spatial distribution of vegetated seagrass area change in the Mississippi Sound throughout the study period.

Seagrass increase on Cat Island from 2003 to 2007 far exceeded any loss or stable areas during the same time period (Figure 10). A majority of this gain happened in just one year (2006-2007) (Appendix Figure A1) and might be the result of thick algal mats or detritus accumulations rather than a true seagrass growth. The most stable seagrass during the 2000s was located just south of the western tip.

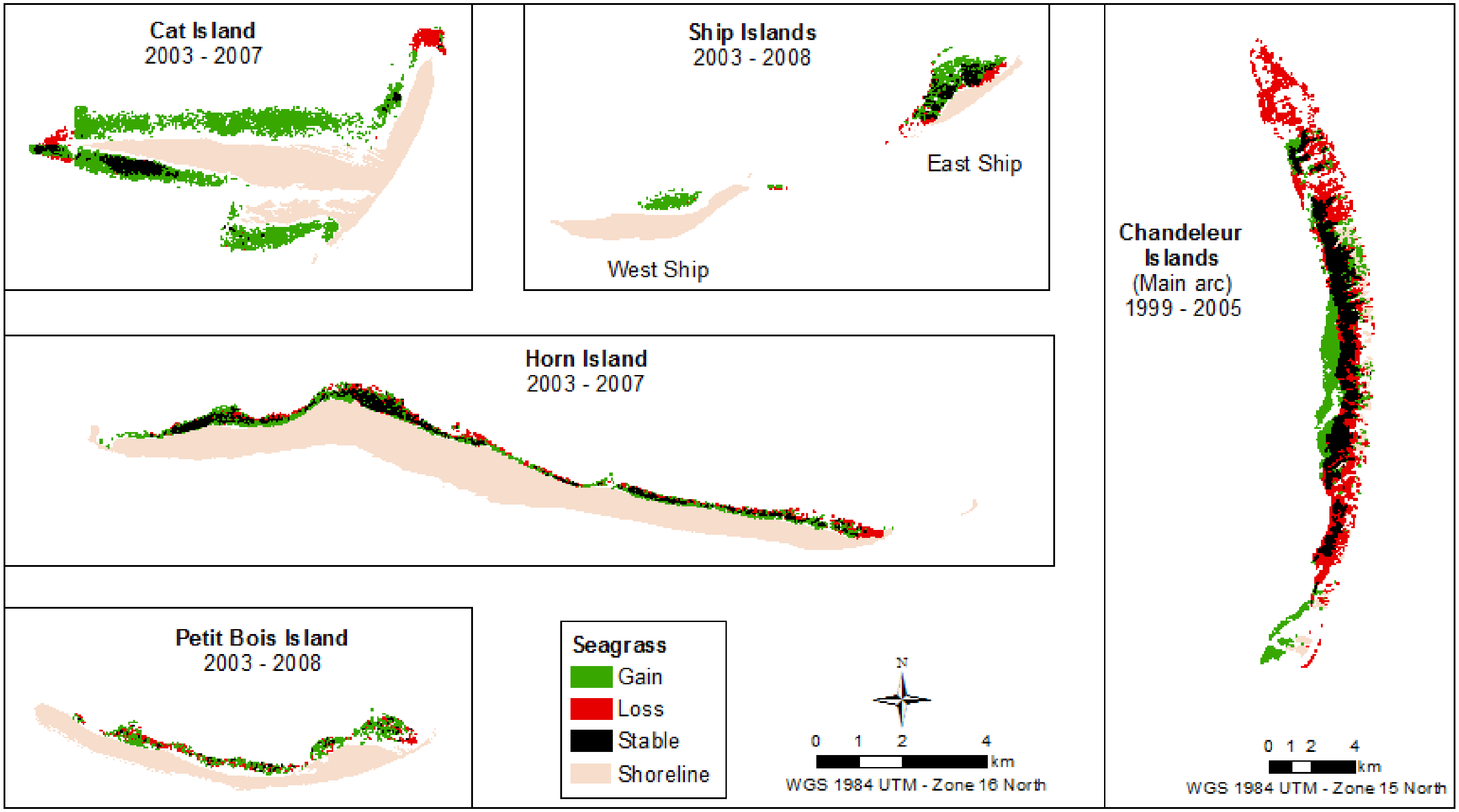

Figure 10. Spatial distribution of vegetated seagrass area change in the Mississippi-Chandeleur Sound within the 2000s.

Note. Shoreline is from the later date in each comparison. Chandeleur Islands are in half scale to the Mississippi islands.

The dramatic loss of seagrass on the two Ship islands from 1963-1975 resulted in the seagrass landscape becoming a shorter and narrower band closer to the shoreline (Appendix Figure A2). By 2003 seagrass already exhibited signs of recovery on East Ship Island as a wider band, but continued to decline on West Ship Island. Later in the 2000s, seagrass reoccurred in the center of West Ship Island with increasing stability (Figure 10). Meanwhile on East Ship Island, VSA fluctuated around 15-18 ha throughout 2003-2008 with seagrass patch disappearance and reappearance in all three cardinal directions except south, where the island is located.

On Horn Island, seagrass loss from 1940 to 2007 mostly occurred at the eastern end and the offshore shoals to the north, while a slow increase tended to occur on the western end and closer to shore (Figure 9). There were 63.5 ha of new vegetation from 2003 to 2006 along Horn Island owing to a gain close to the shoreline and a newly established area at the western tip (Appendix Figure A3). Approximately the same amount of vegetation was subsequently lost in 2006-2007 throughout the island even though the overall seagrass extent appears stable. The seagrass on the western and central part of Horn Island remained fairly stable in 2003-2007 (Figure 10), which accounted for the approximately 19 ha of unchanged seagrass each year; this is also the highest amount of stable seagrass area found on any of the MS barrier islands.

On Petit Bois Island, a major part of seagrass loss from 1940 to 2008 occurred on the eastern end and the northern side of the island and the distribution became more flattened in the 2000s (Figure 9 and 10). Seagrass expansion that had occurred throughout the length of the island in 2003-2007 was not stable and was lost again by 2008 (Appendix Figure A4).

In 1999-2005 the NCI gained seagrass on the western and southern sides while losing seagrass on the northern and eastern sides (Figure 10). The disappearance of the northern end of the emergent barrier islands and decline of the eastern extent contributed to the large amount of seagrass loss on the NCI from 1999 to 2005, despite newly-grown seagrass towards the western side of the island. While the northern end of the seagrass boundary kept shortening over time, the southern end went through a more complex trajectory of increase, migration, and disappearance (Appendix Figure A5). The most stable seagrasses were located in the middle of the island arc as continuous large meadows, mostly dominated by *Thalassia testudinum*.

In summary, contraction of seagrass distribution was observed on all islands within the MS-C (Figure 9 and 10). Each island had a unique trajectory of location changes with sometimes large fluctuations between map dates regardless of time interval.

Discussion

The Scale Problem

This study found no significant effect of grain size on VSA however demonstrated that not all scales were equally good and that conclusions made at only one scale can leave out important details. This agrees with the popular recognition that there is not a single appropriate scale for all ecological problems, but not all scales are equal either. A phenomenon of interest may occur over a certain range of scales; hence it is better to understand how a system changes across scales than to insist on a correct scale

(Levin 1992). Likewise, grain size was tested in this study and found a statistically significant effect of time on VSA, and a difference threshold of 15% among group means based on conservative post-hoc grouping.

On the other hand, resampling to a common scale is essential for change detection. A common grain size ensures that all data go through a resampling process (and hence would be slightly different from their original states) while maximizing the retained level of spatial detail. Another benefit found through this conversion step was noise reduction, meaning that seagrass polygons smaller than a certain size were removed from further analyses. In this study, a common grain size of 2.5 m was found to be a reasonable fine scale that preserved landscape metrics fairly well. Any seagrass patches smaller than 3.125 m^2 (half of a single pixel area of 6.25 m^2 = 2.5 m x 2.5 m) were most likely not included in the mapping output. However, patches larger than 3.125 m^2 account for more than 50% of a pixel area and have a higher likelihood of being retained through the resampling process. Field mapping data (Biber 2014) confirmed the existence of seagrass patches with one dimension as small as 1.8 m (smallest patch recorded in the field was 1.9 m x 2.1 m = 3.13 m^2, $\sqrt{3.13\ m^2} \approx 1.8$ m). In this study, the landscape extent (in hectares) was very large compared to the mapping unit (6.25 m^2) suitable to the ecological process of interest (dynamics of VSA). This grain size to sampling extent difference ensured that the process was studied at an appropriate scale, following the recommendation of O'Neill et al. (1996). It is worth emphasizing that ultimately choosing 2.5 m as the common grain size to represent this data set after looking at a range of scales by no means equals its status as the single best grain size for all processes in this landscape.

A few issues were found that required special attention in generalization of GIS maps from various sources. First, caution should be exercised when attempting to resample highly patchy data at an even-numbered scalar of the original pixel size due to the higher chance of overestimating total area in certain map configurations. Second, the resampling effect should be accounted for when comparing data, since the magnitude of change was different for each data layer and in its effect on data ranking and conclusions. Third, a comprehensive look at the data at multiple scales is strongly recommended before any final conclusion is made.

Rate of Change of Vegetated Seagrass Area over Time

The overall VSA loss and rate of change throughout the MS-C confirmed to the findings of previous studies. Handley et al. (2007) reported that loss of seagrasses in Texas, Louisiana, MS, AL, and Florida was much greater than gains during 1940-2002. In Mobile Bay and adjacent waters, Vittor and Associates, Inc. (2004, 2005, and 2009) found substantially reduced seagrass area in 2002 compared to 1940, 1955, and 1966. The rate of VSA loss (-1.5% per year) on the four MS islands (excluding Cat Island) in 1940-2008 was equal to the mean rate of global seagrass decline from 1879 to 2006, and was higher than the estimated pre-1940 global loss rate of -0.9% per year (Waycott et al. 2009).

The decade when seagrass loss started to accelerate was not the same for all locations studied. The decline of seagrass in MS was rapid since the 1940-1950, and total

VSA on all the MS barrier islands combined was lowest in the 1970-1980s. Though the 1940 to 1950 interpolated VSA reduction may be somewhat overestimated due to images taken in the early growing season (April 1952), the trend is supported by further loss occurring in the next two decades. The faster rate of decline and higher loss of VSA in the MSS prior to 1970 mirrors a more severe decline in seagrass area reported during 1960-1980 for the Pensacola Bay system (Lewis et al. 2008). These two systems, however, contradict the faster loss rate and larger net seagrass loss occurring after 1970 reported for the globe (Waycott et al. 2009) and in Chesapeake Bay (Kemp et al. 1983, Orth and Moore 1983). The post-1980 decline in the rate and the net change of VSA in the MSS was less than the median global rate of -5% per year and the global estimation of -37 km2 per site per decade. As for the rate of increase in VSA, the MSS experienced the same acceleration from the 1970s to the 1990s as global seagrass meadows, although at a smaller magnitude.

Within the 2000s, seagrass loss was not as dramatic as prior decades and increases in VSA were recorded on the three islands in western MSS. Similar increases were reported in the northeastern MSS from 2002 to 2008/2009 (Vittor and Associates, Inc. 2009) and in Pensacola Bay and Santa Rosa Bay from 1992 to 2003 (Lewis et al. 2008). A possible explanation for some of this VSA gain and the higher annual rate of change in the MSS within the 2000s compared to the decadal rate of change prior decades may be the change in species composition mentioned in Chapter II. It was around the 1970-1980s when the monospecific beds of the opportunistic *Halodule wrightii* replaced slower growing beds dominated by *Thalassia testudinum* and *Syringodium filiforme*. In contrast to seagrass recovery observed in the MSS, the NCI exhibited a large decline in 2005 right after the passing of Hurricane Katrina, which biased the calculated rate of change. The NCI data from 1999 to 2005 represent a period of increased storminess (Hurricane Georges on 28 September 1998, Hurricane Ivan on September 16, 2004, and Hurricane Katrina on August 29, 2005) that impacted seagrass extent. Seagrass loss on the NCI within 2005 was likely highly impacted by hurricane Katrina, rather than purely by seasonal fluctuations due to seagrass growth.

Change in Spatial Distribution and Extent of Seagrass

In comparing overlaying pairs of seagrass layers, it is important to appropriately calibrate the raw output before data visualization is to be applied. For data collected initially at fine spatial resolutions, thousands of miniscule changes can add up to a big total error. If not corrected by spatial displacement error, this may overestimate the percentage of spatial change and obscure any important change patterns visualized in the data. Fortunately, the spatial distribution pattern of VSA changes was little affected, even when the percentage of spatial change calculated may have been unusually high at some time points (e.g., Cat Island and Horn Island 2006 – 2007).

Some concerns about the data should be noted. The increase on Cat Island from 2006 to 2007 may be due to map classification errors rather than a real change in seagrass cover. Field surveys in June 2014 (Center for Plant Restoration – Gulf Coast Research Laboratory) found extensive algal mats at Cat Island, previously also noted by Eleuterius (1973). Also questionable is the huge VSA loss on Horn Island from 2006 to 2007. In both situations, there is some doubt as to whether the outspread of algae and detritus might have made seagrass beds appear denser and larger, and might have prevented good separation between algae and true seagrass in the remotely sensed data. The lack of ground-truthed data precludes any assessment on the magnitude of these potential confounding effects. The VSA trend may not have changed, but the magnitude of change may have been overestimated. These concerns help to support the value of overlaid maps, since spatial problems like this cannot be addressed when only looking at VSA.

The results also revealed a strong dependency of seagrass distribution on the migration and geomorphology of barrier islands. Overall there was shrinkage in spatial distribution of VSA on all islands (except Cat Island due to limited data), but the timing and reasons were different across islands. Some responsible factors were Camille Cut on Ship Island, westward movement on Horn Island, and eastern shrinkage accompanied by western channel dredging on Petit Bois Island. A similar geomorphological effect occurred in Ria Formosa (Portugal) where Cunha et al. (2005) quantified losses of seagrass habitat due to movement of the protective barrier islands.

In conclusion, the seagrass landscape in the MS-C is an extremely dynamic system with a high degree of fluctuation in both VSA and spatial extent. By combining both grain size and sampling extent, a significant change over time and an ongoing net loss of VSA in both the MSS and the NCI since 1940 (the beginning of data record) was discovered. The annual rate of change varied by island, but on average was < 3% per year over the entire study period. The within-decade net change and annual rate of change in VSA could be equal to or greater than the decadal change. Spatial extent contraction was observed on a majority of the islands, but the timing was not consistent across islands. It is therefore difficult to catch the critical time point when a catastrophic loss or a sustainable improvement occurred in VSA. Additionally, the VSA trajectory and seagrass distribution do not reflect changes in species composition and landscape patch configuration. This implies that area metrics by themselves may not be enough for holistic seagrass assessment over time.

CHAPTER IV – SEAGRASS IN THE MISSISSIPPI AND CHANDELEUR SOUNDS: AN EXPLORATORY SPATIAL DATA ANALYSIS AT THE PATCH-LEVEL SCALE

Introduction

Exploratory Spatial Data Analysis

In statistical analysis, exploratory data analysis (EDA) emphasizes data display and the use of simple indicators to elicit patterns and suggest hypotheses (Good 1983, Anselin 1999). However, none of the traditional EDA tools deal with spatial data. Certain EDA techniques, such as box plots and scatterplot matrices, could be used in studies that combined GIS and spatial analysis, but they ignore spatial characteristics of the data. There are two important spatial effects as a result of locational information, they are (1) spatial dependence (spatial autocorrelation (SA)) and (2) spatial heterogeneity (non-stationarity) (Anselin and Getis 1992). The first conflicts with the assumption of independent observations in statistics. The second is related to spatial differentiation or a lack of spatial uniformity of the effects of spatial dependence and/or of the relationships between the variables under study; this represents a complex realization of the nature of the variables under study and the effects of size, shape, and configuration of spatial units. Both can affect the validity of standard statistical techniques and a special set of spatial statistical methods are needed (Anselin 1994).

Methods of EDA that account for the spatial aspects of the data are called exploratory spatial data analysis (ESDA). ESDA is a group of techniques aiming to describe and visualize spatial distributions, identify spatial cluster and outliers, and suggest different spatial regimes and non-stationarity (Anselin 1994). One of the most essential concepts to ESDA is spatial autocorrelation. ESDA techniques include visualizing spatial distribution, visualizing and assessing SA in both geostatistical and lattice perspectives.

Spatial Autocorrelation

The first law of geography states that "everything is related to everything else, but near things are more related than distant things" (Tobler 1970). Spatial autocorrelation refers to the pattern in which observations at adjacent geographic locations have the tendency to resemble each other more than by chance alone (Sokal et al. 1998, Fortin et al. 2002). There exist both global SA and local SA.

Global SA is computed over all or many geo-referenced points in the sample. Global analyses include: nearest neighbor, Ripley's K, join-count, Moran' I and Geary's c, semivariogram, Mantel test, trend surface analysis, kriging, spline, and Voronoi polygons. Two of the most important assumptions of these global indicators are (1) spatial stationarity requiring a normal distribution of data with a constant mean and constant variance over the entire study area, and (2) isotropy implying the same intensity of pattern in all directions. These assumptions are hardly satisfied by GIS data. Rather than emphasis on global SA, ESDA should focus more on local patterns of spatial association, local non-stationarity and heterogeneity (Anselin 1994, Anselin 1996).

Local SA is the dependence of the value of a variable at any one location upon neighboring values of that variable (Sokal et al. 1998). Local indicators of spatial association (LISA) is a group of statistics (local Moran' I, local Geary's c) that satisfies: (1) "the LISA for each observation gives an indication of the extent of significant spatial clustering of similar values around that observation, and (2) the sum of LISAs for all observations is proportional to a global indicator of spatial association" (Anselin 1995).

Getis and Ord (1992) and Ord and Getis (1995) introduced the G-statistics family, which is especially useful to identify spatial clusters of high or low values when global statistics fail to detect significant pockets of clustering. The only difference between G_i and G_i^* is the inclusion of the location itself in the computation of the latter but not of the former. Getis and Ord (1992) defined

$$G_i^*(d) = \sum_j w_{ij}\,(d) y_j / \sum_j y_j \qquad \text{(Equation 5)}$$

where w_{ij} is a binary matrix with w_{ij}= 1 when i and j are within a distance *d* from each other and zero otherwise (see Ord and Getis (1995) for the revised formula). Though the G-statistics do not meet the second requirement of LISAs defined by Anselin (1995), they are often included together as a group of local spatial statistics.

These local SA statistics move the focus from the density of points (first order) to the information contained in all the inter-point distances (second order). The LISA statistics may be used to indicate local pockets of non-stationarity (or hot spots) or to identify "outliers" (Anselin 1995). While local Moran's I and local Geary's C compare whether the value for each observation is similar to those that neighbor it, G_i and G_i^* compare local averages to global averages (Anselin 1995).

Hot Spot Analysis Using Local Indicators of Spatial Association

There is growing interest in detection of clusters to identify "hot spots" without any preconceptions about their locations (Ord and Getis 1995). The hot spot analysis using LISA tests if there are statistically significant clusters of high values and clusters of low values in the data set (Nelson and Boots 2008). This allows the detection of distinct regions, regardless of spatial scale, without the assumption of global methods (Barrell and Grant 2013).

Since many, if not most, spatial biological datasets are globally autocorrelated, tests of individual local SA statistics will be too liberal, for example, coefficients appear to be significant when they are not. Hence a global SA analysis should be performed before any significance tests of local SA statistics (Anselin 1995, Sokal et al. 1998). In case the studied variables are highly significantly globally spatial autocorrelated, it might be worthwhile to compute and examine magnitudes of individual local autocorrelation coefficients for data exploration purpose.

Landscape Metrics at Patch-Level Scale

While the above-mentioned spatial statistics estimate the spatial structure of the values of a sampled variable, landscape metrics characterize the properties of a patch or mosaic of patches (Fortin 1999). Within FRAGSTATS (McGarigal et al. 2012), the indices are divided into different major groups including: (1) area and edge metrics, (2) shape metrics, (3) core area metrics, (4) aggregation metrics, and (5) diversity metrics. Also, the metrics can be computed at three different levels: for every patch (patch-level metrics), every patch type/class (class-level metrics) in the landscape, and for the entire patch mosaic (landscape-level metrics). This chapter focuses on patch properties, in other words, metrics at the patch-level scale.

Study Objectives

This chapter aims at exploring spatial attributes of the seagrass landscape in the MS-C using techniques from exploratory data analysis. The specific goals are: (1) to examine seagrass patch orientation, (2) to examine patch size distribution of seagrasses, (3) to identify statistically significant local spatial clusters of different seagrass patch-level metrics simultaneously and changes of hot and cold spots over time.

Materials and Methods

Measuring Seagrass Patch Indices Using FRAGSTATS

Seagrass data resampled to 2.5-m grain size in Chapter III were used in this chapter. Since the seagrass data were binary (seagrass presence or absence), FRAGSTATS statistics can be computed for each patch in the landscape and for the landscape as a whole (but no class level) and mostly be used to explain landscape configuration (size, shape, and spatial arrangement of patches). In FRAGSTATS ver. 4.2 (McGarigal et al. 2012), patch-level metrics were calculated from the seagrass raster file following an 8-neighboring cell rule, which meant a seagrass patch was defined as a group of cells adjacent to each other in at least one of the four ordinal or of the four diagonal directions. Therefore, seagrass patches in the hostile sand matrix were always separated by at least one full pixel width (2.5m) from adjacent patches.

The input raster had an exterior background and no border. The exterior background denoted seagrass absence, was given negative values (-999), was not included in the total landscape area, and did not affect the calculation of any metrics. Seagrass presence was given a value of 1. Total VSA equals to the sum of all seagrass cells (positive values), all the background values are ignored. To calculate edge length, all background/boundary interfaces are counted as edge (landscape boundary and background edges are the same in this case). The FRAGSTATS outputs were accompanied with patch ID files that linked each patch with its corresponding metrics. This output was converted back to polygon shape files through a series of iterations in ArcGIS ModelBuilder in ArcGIS ver 10.2 (ESRI, Redlands, CA) which includes projecting data, converting from raster to polygon, selecting only seagrass polygon (positive values), and connecting polygons that belong to the same patch. The final ArcGIS dataset included the seagrass vector files in which each patch has a unique patch identification number and associated patch metrics computed by FRAGSTATS.

Six patch metrics were calculated: area (AREA) (m^2), perimeter (PERIM) (m), radius of gyration (GYRATE) (m), related circumscribing circle (CIRCLE), fractal dimension index (FRAC), and Euclidean nearest neighboring distance (ENN) (m). While measurement of AREA and PERIM is straight forward, the other three metrics require further details. Radius of gyration can be considered as “the average distance an organism can move within a patch before encountering the patch boundary from a starting point” (McGarigal 2014). The radius of gyration of patch *ij* ($GYRATE_{ij}$) is calculated as follow:

$$GYRATE_{ij} = \sum_{r=1}^{z} \frac{h_{ijr}}{z} \quad (m) \qquad \text{(Equation 6)}$$

where h_{ijr} is the distance (m) between cell *ijr* [located within patch *ij*] and the centroid of patch *ij* (the average location), and z is the number of cells in patch *ij*. GYRATE increases as the patch increases in extent (McGarigal 2014).

In the group of shape metrics, CIRCLE and FRAC were chosen as they are not influenced by patch size. Related circumscribing circle is based on the ratio of patch area to the area of the smallest circumscribing circle, as follows:

$$CIRCLE = 1 - \frac{a_{ij}}{a_{ij}^{s}} \qquad \text{(Equation 7)}$$

in which a_{ij} is area (m^2) of patch *ij* and $a_{ij}^{s}$is area (m^2) of smallest circumscribing circle around patch *ij* (McGarigal 2014). Fractal dimension index reflects shape complexity across patch sizes and is calculated as:

$$FRAC = \frac{2 \ln(0.25 p_{ij})}{\ln a_{ij}} \qquad \text{(Equation 8)}$$

in which a_{ij} is area (m^2) and p_{ij} is perimeter (m) of patch *ij* (McGarigal 2014).

Finally, ENN measures the distance between cell center of the two closest cells in the two considering patches (McGarigal 2014).

Seagrass Patch Size Distribution

To evaluate the relative importance of different patch sizes in the landscape, seagrass patches on each island in certain years were categorized by size as follows: (1) small-sized (S) patches with areas of 6 - 25 m^2 (equivalent to 1 - 4 pixels), (2) medium-sized (M) patches with areas of 25 – 1,000 m^2, (3) large-sized (L) patches with areas of 1,000 – 1,000,000 m^2 (0.1 – 100 ha). Visualization of the distribution of patch sizes (S, M, and L) for each island in the MS-C was reported for the most recent year available. As many of the patches on the NCI were larger than 10 ha, additional size categories for 10 – 100 ha and 100 – 1000 ha were created specifically for that visualization only. The percentages that each patch size category (S, M, L) contributed to the total area (VSA) and total number of patches (NP) in the landscape were tabulated for each island and year.

Hot Spot Analysis and Changes in Patch Clusters

Hot spot analysis workflow was automated using ArcGIS ModelBuilder (Figure 11) in ArcGIS ver 10.2 (ESRI, Redlands, CA). A spatial weights matrix is defined as an NxN table representing the spatial relationship among features in a given data set, where N is the number of features in the data set. A spatial weights matrix file was constructed in ArcGIS (using “Generate Spatial Weights Matrix” tool under “Spatial Statistics Tools” in Arc Toolbox) using the inverse distance rule, the previously determined distance threshold of 500 m, a minimum of 10 neighbors, and row standardization. For each island on each map date, the workflow was repeated for each of the six patch-level metrics (AREA, PERIM, GYRATE, CIRCLE, FRAC, ENN). Hot spots and cold spots were detected using the LISA, particularly Getis-Ord G_i^* in this study. A hot spot is a statistically significant spatial cluster of high values, quantified by highly positive Z-scores (G_i^* Z-score > 1.96) and a small p-value ($p < 0.05$). A cold spot is a statistically significant spatial cluster of low values, quantified by low negative Z-scores (G_i^* Z-score < -1.96) and a small p-value ($p < 0.05$). This method is adapted from hot spot analysis toolset and tutorial by ESRI (2009a, 2009b, and 2009c) and Chainey (2010).

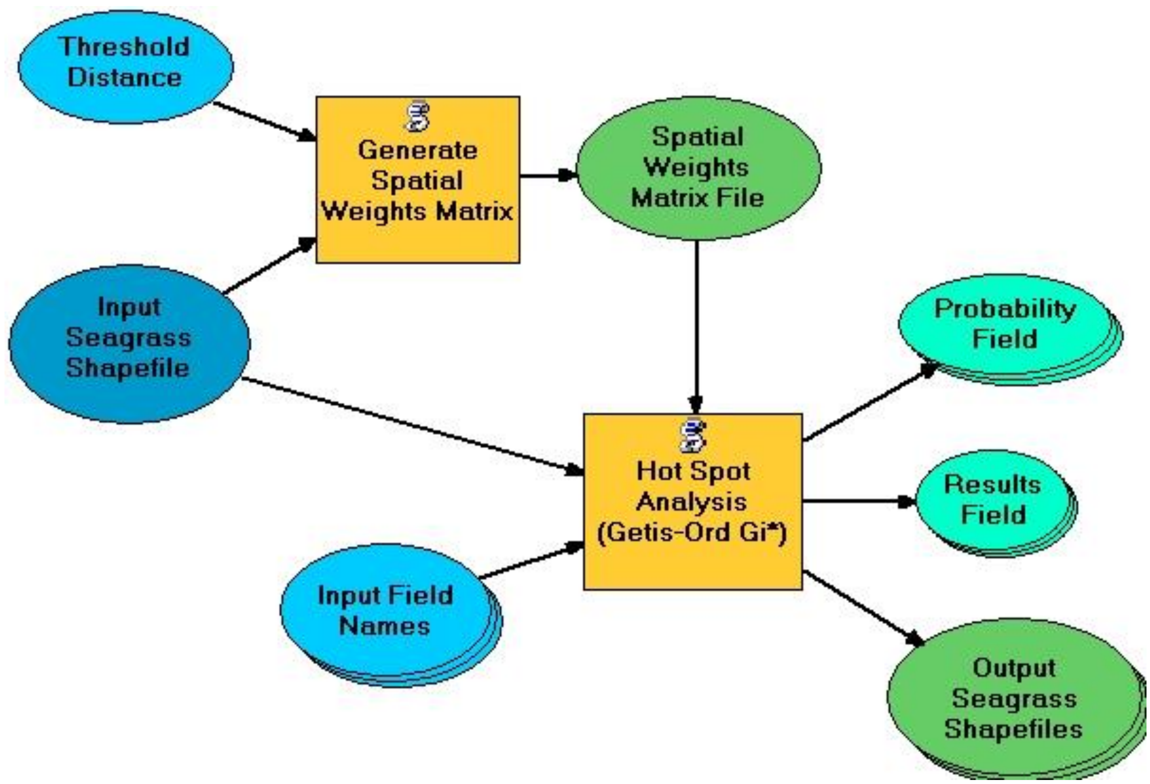

Figure 11. Hot spot analysis workflow using ArcGIS ModelBuilder.

Note. The dark blue oval represents a group of input raster files. Light blue ovals indicate additional criteria necessary to run geoprocessing tools. Orange rectangular shapes represent ArcGIS spatial statistics tools. Green ovals are output data after each step. Aquamarine ovals are fields associated with output data.

Results

Seagrass Patch Size Distribution

Seagrass landscapes on all islands in the MS-C were slightly skewed toward small and medium patches; the degree of skewness appeared to be higher on the MS islands than on NCI (Table 7). Seagrasses on the NCI existed in continuous beds extending more than several hundred hectares, while there was no seagrass patch larger than 10 ha found on any of the MS islands and no patch bigger than 1 ha on West Ship Island (Figure 12, Table 7). However, the large portion of small patches equal to or less than 25 m^2 (40.3 – 82.8%) in the MSS only contributed a modest portion of total vegetated area (3.9 – 50.2%, except West Ship Island in 2003 with 100% small patches), whereas the smaller proportion of large patches greater than 1000 m^2 (1.2 – 16.6%) accounted for a considerable proportion (20.7 – 89.4%) of VSA (Table 7). On the NCI, the patch size distribution was less unbalanced with a higher percentage of large patches, which in turn accounted for more than 98% of total VSA across all years (Table 7).

On Cat Island in 2007 there was a substantially higher number of patches (22977) compared to the two previous years (3383 and 4088 respectively) and the percentage of small-sized patches increased compared to the prior years (Table 7). A decrease of 7.2% in the percentage of large-sized patches in 2007 led to about a 20% reduction in their contribution to VSA, suggesting a disproportionate effect of large patches to VSA.

On West Ship Island, a 15.4% decrease in percentage of large patches from 1963 to 1975 led to a reduction of over 60% of this patch size to VSA contribution (Table 7). Since its worst condition in 2003, when there were no large patches, there are signs of

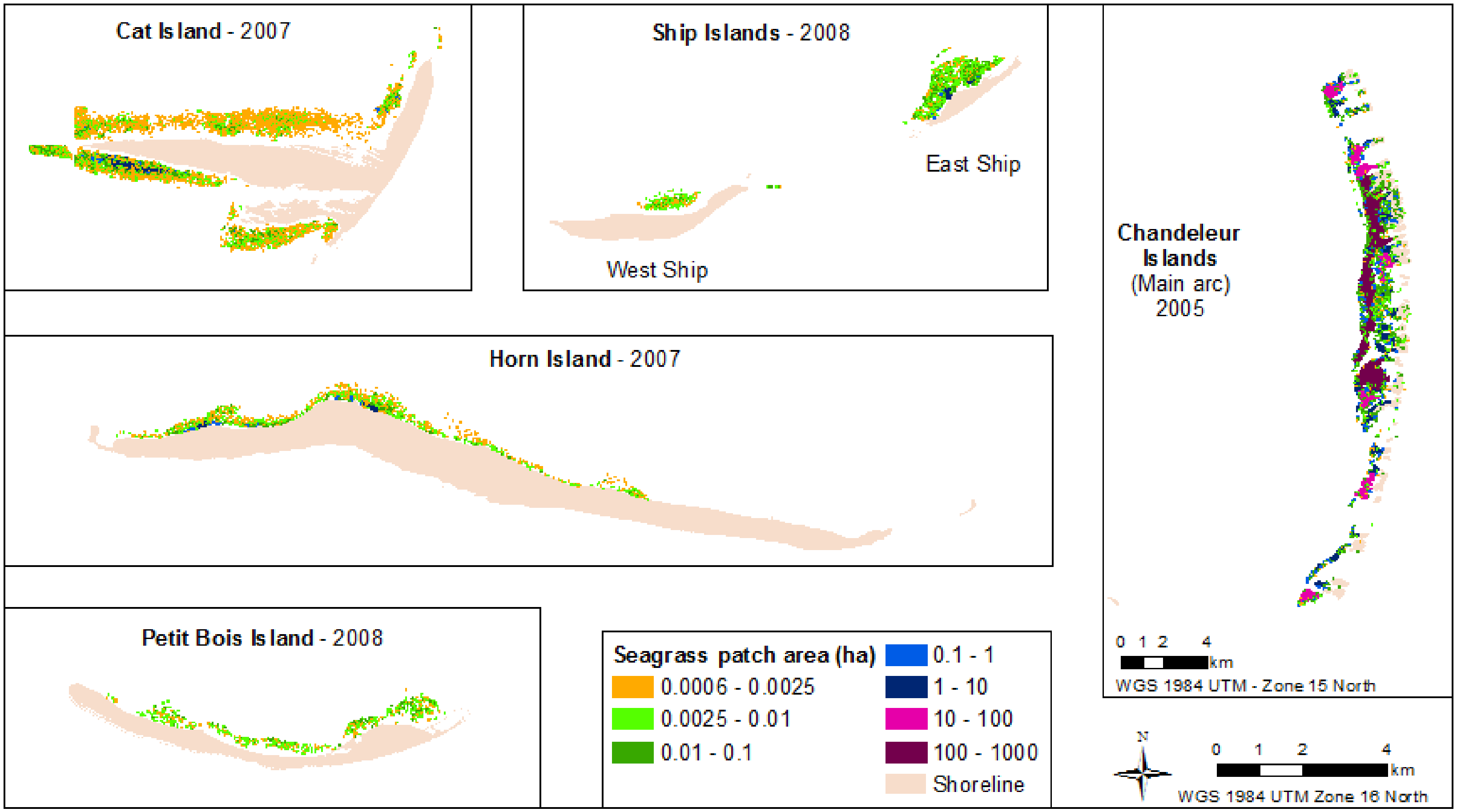

Figure 12. Status of seagrasses in the Mississippi-Chandeleur Sound categorized by patch size.

Note. Patches larger than 10 ha were mapped on the Chandeleur Islands but not found on any of the Mississippi islands.

Table 7

Percentage of Vegetated Seagrass Area and Patch Number Categorized by Patch Size (S, M, L)

	Year	VSA (ha)	S (%)	M (%)	L (%)	NP	S (%)	M (%)	L (%)
Cat	2003	21.8	13.7	17.7	68.6	3383	71.4	23.1	5.5
	2006*	25.6	12.9	21.8	65.4	4088	64.0	25.3	10.7
	2007	71.2	28.6	23.3	48.1	22977	82.0	14.6	3.5
West Ship	1963	11.1	9.1	22.5	68.4	1486	53.7	29.7	16.6
	1975	1.9	50.2	41.6	8.2	936	80.9	18.0	1.2
	2003	0.02	100	0	0	24	100	0	0
	2006	0.9	34.6	44.7	20.7	318	72.0	23.9	4.1
	2007	1.7	30.9	56.3	12.8	616	68.0	29.6	2.4
	2008	3.2	12.3	57.2	30.5	670	40.6	50.6	8.8
East Ship	1963	19.4	16.2	27.9	56.0	3958	64.9	25.6	9.5
	1975	1.6	40.0	31.8	28.2	687	81.5	16.3	2.2
	2003	16.5	9.4	17.2	73.4	2014	63.0	26.9	10.1
	2006	15.5	7.4	16.0	76.7	1567	57.8	29.6	12.6
	2007	13.9	10.6	23.7	65.8	2141	58.9	28.3	12.8
	2008	18.0	7.4	29.3	63.3	2203	40.3	44.0	15.7
Horn	1940	76.7	19.2	19.0	61.8	18671	78.6	14.8	6.6
	1952	45.8	14.3	22.8	62.9	8702	67.7	22.1	10.2
	1971	19.2	9.9	25.6	64.5	2864	57.4	30.5	12.1
	2003	50.9	8.7	16.4	75.0	5578	60.4	27.6	12.0
	2006	82.1	3.9	6.7	89.4	4297	60.3	23.9	15.8
	2007	38.2	15.2	25.3	59.6	7068	65.1	26.6	8.3
	2008*	18.7	18.3	29.9	51.8	4080	68.1	27.3	4.6
Petit Bois	1940	54.2	6.7	9.4	83.9	5119	68.1	18.4	13.5
	1952	15.2	42.4	36.9	20.7	7640	82.8	14.8	2.5
	1985	17.7	8.7	23.8	67.6	2212	47.6	34.6	17.8
	2003	8.1	21.9	37.4	40.7	2066	64.4	27.9	7.7
	2006	19.0	10.8	24.2	65.0	2798	55.9	29.6	14.4
	2007	16.9	14.0	34.9	51.1	3226	53.8	34.7	11.5
	2008	7.3	19.8	51.9	28.3	1892	54.8	38.5	6.7
Chandeleur	1999	1495.8	0.2	1.6	98.2	8031	21.4	53.9	24.6
	2000	1517.2	0	0.2	99.8	1758	0.4	28.0	71.6
	2002	1525.1	0.3	1.0	98.7	7614	37.8	40.1	22.2
	2005J	1101.8	0.2	0.8	99.1	4649	32.3	31.7	36.0
	2005O	895.2	0.3	1.0	98.7	5123	34.5	30.8	34.7

Note: VSA = Vegetated seagrass area, NP = number of patches, S = small, M = medium, L = large patches, 2005J = January 2005, 2005O = October 2005. Data of Cat Island in 2006 and Horn Island in 2008 did not cover the whole island.

recovery in the late 2000s with a decline in the relative importance of small patches, and an increase in the contribution of medium (57.2%) and of large patches (30.5%) to VSA.

On East Ship Island, 1975 had the lowest percentage of large-sized patches contributing to both NP and VSA. By 2008, like many of the other islands, the landscape had returned to a state that had more numerous and larger patches, which contributed 63.3% to VSA (Table 7).

On Horn Island, the highest NP occurred in 1940 (18671) and this declined through the 1950s and 1960s with a low point in 1971 (2864), followed by recovery of NP to around 4000 in the mid 2000s. Patches larger than 0.1 ha made up 15.8% of the number and 89.4% of the VSA contribution in 2006, indicating a growth and merging process of patches since 2003. This post-Katrina (2003-2006) rapid patch growth was not stable, however, and vegetation die-off in the 2007 quickly resulted in seagrass loss (lower VSA) and fragmentation in the overall 2003-2007 period (Table 7) evident in an increase in NP and decrease in the relative contribution of large-sized patches.

Petit Bois Island experienced seagrass loss (decrease in VSA) and fragmentation (increase in NP and decrease in the relative importance of large-sized patches) from 1940 to 1952 (Table 7). A similar seagrass loss and fragmentation process was observed from 1985 to 2003 (decline in VSA and relative contribution of patches larger than 0.1 ha). One year after Hurricane Katrina in 2005, the Petit Bois Island seagrass landscape also had higher VSA, lower NP, and a higher relative contribution of large-sized patches compared both to two years pre-Katrina and to 1952. This indicates overall growth and coalescence process in 1952 – 2006. In the two following years, area loss and fragmentation dominated the landscape, similar to what was observed on nearby Horn Island.

Hot Spot Analysis and Changes in Patch Clusters

The G_i^* statistics was able to identify the location of pockets of high/low spatial association (Figure 13), some clusters even seemed to persist over the years. Clusters of metrics in the same group (area and edge metrics - AREA, PERIM, and GYRATE; shape metrics – CIRCLE and FRAC) tended to behave similarly. Often the hot spots identified tended to co-occur where larger patches were found and where more stable seagrass areas were previously identified in Chapter III.

On Cat Island in 2007, clusters of large and complex (LC) patches were located near tips of the island, while the northern shore was characterized by clusters of small and simple (SS) patches (Figure 13). In 2008, patches with larger extent aggregated at the center of West Ship Island, while on East Ship Island, LC patches clustered closer inshore, followed by a spatially random mixture of small- and large-extent patches with an aggregation of SS patches offshore. The seagrass landscape on Horn Island in 2007 was mainly characterized by a random distribution of patch extent, however, there was a higher degree of clustering within the western third of Horn Island with aggregation of LC patches around the wider part of the island and aggregation of SS patches near the island's western tip (Figure 13). There were also clusters of SS patches at the thinnest part of the island (around the Ranger Pier). On Petit Bois Island in 2008, LC patches were aggregated at the thickest part of the island close to its eastern tip, while SS patches clustered along portions of the rest of the island. Meanwhile on the NCI in 2005, LC patches clustered toward the center of the island chain.

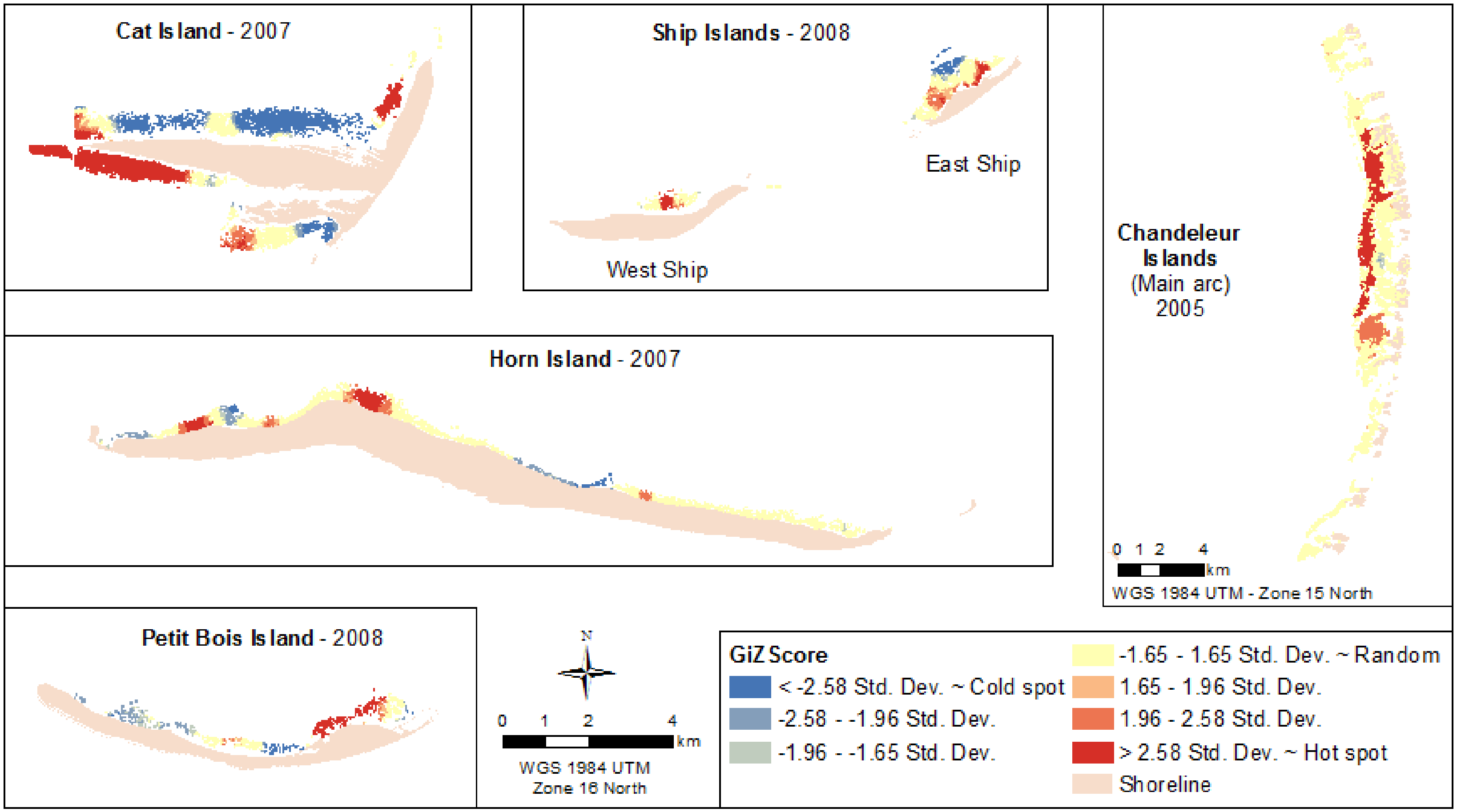

Figure 13. Hot-spot analysis on the most current data in the Mississippi-Chandeleur Sound.

Note. The reddish color indicates aggregation of large and complex patches or a hot spot. The blueish color indicates aggregation of small and simple patches or a cold spot. Regions of randomly distributed patch extent are in yellow.

Discussion

Seagrass Patch Size Distribution

The skewness towards smaller patch sizes in the MSS shows agreement with other seagrass landscape studies on different seagrass species (Olesen and Sand-Jensen 1994, Vidondo et al. 1997). This positively skewed patch size distribution was documented as the result of the rapid turnover rate of small patches (Vidondo et al. 1997, Frederiksen et al. 2004a). This important finding suggests that relying on VSA alone, as was done in the prior Chapters, misses key features of the landscape configuration and may fail to detect ecologically relevant attributes that can only be determined by exploring spatial attributes of the landscape patches in more detail.

Similarly, changes in cumulative percent of VSA and NP could provide evidence of seagrass fragmentation in the MS-C. For example, the highest number of patches larger than 1 ha and the highest relative contribution of large-sized patches on Horn Island in 2006 indicated a growth and merging process of patches. This post-Katrina (2003-2006) rapid growth appeared to be unstable, as a decrease in the NP and in the relative importance of large-sized patches in 2007 suggests a quick vegetation/detritus disappearance. There is a high possibility that this is not true seagrass loss as the rate of change reported here was much higher than what we observed in laboratory conditions.

Spatial Distribution of Patch Clusters

The similar local pattern detected for metrics in the same group is not surprising, as these metrics measure similar aspects of patches, therefore, are often strongly correlated. Also predictable was the reverse pattern of ENN compared to other metrics, hot spots of other metrics would generally be cold spot of ENN and vice-versa. While hot and cold spots were often found near the tips of Cat, Horn, and Petit Bois Island, local pockets tended to be found in the center of both Ship islands. This might be the effect of island shape, which results in the concentration of seagrasses themselves in the middle of the two Ship islands but hardly any grass at the island tips.

The enhanced hot spot near the western tip on Cat Island in the 2000s might be due to protection from wind and wave. The intensified cold spot on the northern shore could be the result of more seagrass growing at this location at the early unstable stage. The inhomogeneous look of Cat Island in 2007 is partly due to the outgrowth of small-sized patches, affecting the local means of all patch metrics. The intense hot spot found on the NCI might be the artifact of the large size of the polygons delineated and the difficulty of separating the lush meadow into smaller polygons. This, on the other hand, is not necessarily far from the true nature of spatial association of seagrass on the island chain. These large polygons in 1999 then broke apart in 2002, leading to diminishing hot spots in the later years. This can be considered as evidence of fragmentation. Interestingly, though the eastern end of Petit Bois Island was shortening, it still supported hot spots of all metrics. The center of the island supported local association before 2000. Unfortunately, data available in this dissertation is not enough to explain the local aggregation of seagrasses on Horn and Petit Bois islands. The intense hot spot found on the Chandeleur Islands might be the artifact of the large size of the polygons delineated and the difficulty of separating the lush meadow into smaller polygons. This, on the other hand, is not necessarily far from the true nature of spatial association of seagrass on the island chain. These large polygons in 1999 then broke apart in 2002, leading to diminishing hot spots in the later years. This can be considered as evidence of fragmentation.

The spatial pattern of hot and cold spots seemed to be stable within the 2000s across all islands but was generally less so among earlier decades. This might suggest an underlying inherent persistent spatial configuration within these seagrass landscapes over multiple years. For instance, the general location of seagrass hot and cold spot clusters remained similar in consecutive years (2006-2007-2008), but was found to be more different after a big storm event (e.g., from 2003 to 2006 after Hurricane Katrina) or over longer periods (e.g., decadal time).

A hot spot could indicate areas for protection, as they are dominated by larger and more complex patch configurations, while cold spots might be dynamic areas that are more susceptible to change due to the small and simple nature of the patches. However, it is important to consider that hot and cold spots in the maps (Figures 12, Appendix D Figure D1 to D5) are relative to the mean of patch values on a particular island in a particular year. As such, a hot spot in one year does not necessarily have a higher density than a neutral/random spot in another year. Analysis using a common global reference scale for hot and cold spot identification may have resulted in slightly different visualizations and interpretations of spatial patterning. Similarly, using a different index of LISA (e.g., Moran's I, Geary's c) may have resulted in maps that identified the location of hot and cold spots slightly differently than was found using Getis-Ord Gi*. Finally, using a different suite of FRAGSTATS metrics to calculate the spatial association would also have affected the outcomes and visualizations for the different islands. The resulting visualizations may not be as suitable for the NCI due to the more lush and continuous nature of seagrass meadow compared to the MS islands. Also, these results are influenced by calculating the hot spots from the six somewhat arbitrarily chosen FRAGSTATS metrics (AREA, PERIM, GYRATE, CIRCLE, FRAC, ENN) using the annual local means and using the Getis-Ord Gi* statistic. Future studies should more comprehensively explore these combinations and generate a range of output visualizations that can be compared to determine consensus locations of hot spots that are more method independent.

CHAPTER V – SEAGRASS LANDSCAPE PATTERN ANALYSIS IN THE MISSISSIPPI SOUND

Introduction

Landscape Pattern Analysis and FRAGSTATS

Landscape pattern analysis is the quantification of landscape patterns and their spatio-temporal dynamics (McGarigal et al. 2012). Landscape dynamics, in turn, refers to variation of a landscape over time, as a system of interacting components, structures, and processes (Tongway and Ludwig 2012). There are more than 50 landscape metrics that can be calculated through FRAGSTATS (McGarigal et al. 2012, McGarigal 2014) to quantify landscape composition and configuration. Each spatial metric has its own strengths and limitations, and the choice of metrics will affect the results of pattern analysis (Hargis et al. 1998, Li and Wu 2004). Metrics can be inherently redundant, i.e., representing the same information or empirically redundant, i.e., metrics measuring different aspects of landscape pattern can become statistically correlated and important information can be derived from this (McGarigal et al. 2012). For example, Riitters et al. (1995) performed a factor analysis on 55 metrics for 85 land-use land-cover maps and suggested that landscape pattern and structure can be represented by only six metrics: average perimeter-area ratio, contagion, standardized patch shape, patch perimeter-area scaling, number of attribute classes, and large-patch density-area scaling. That study highlighted the importance of the need to conduct a search for a core set of metrics to that best quantify landscape pattern.

Landscape Fragmentation

Landscape fragmentation is the "breaking up of vegetation or other land cover types into smaller patches that impedes the flows of organisms, energy, and material across a landscape" (Wu 2012). A closely-related term, but more habitat-explicit, is habitat fragmentation, which is "the discontinuity, resulting from a given set of mechanisms, in the spatial distribution of resources and conditions present in an area at a given scale that affects occupancy, reproduction, or survival in a particular species" (Franklin et al. 2002). Landscape or habitat fragmentation can be both a state as well as a process, and should be distinguished clearly from the process of habitat loss and changes in habitat quality (Franklin et al. 2002). Different criteria have been used to characterize the state of fragmentation, such as "progressively smaller, geometrically more complex (initially, but not necessarily ultimately), and more isolated habitat fragments" (McGarigal and McComb 1999), or the combination of both degree of subdivision and aggregation of habitat patches, and complexity of fragment shapes (Bennett and Saunders 2010). For seagrass habitats, Sleeman et al. (2005) suggested that landscape division and area-weighted mean perimeter to area ratio were most useful to detect differences in seagrass fragmentation.

Study Objectives

For this chapter the island model was chosen to represent the seagrass patch mosaic landscape occurring on the barrier islands of the MSS (Group 2 as defined in Chapter II) in the time period of 1940-2008. The study goals are: (1) to quantify a representative subset of FRAGSTATS metrics that best capture seagrass landscape pattern in the Sound, (2) to use multivariate ordination on the entire dataset to visualize and elucidate inherent landscape patterns, and (3) to examine landscape pattern and dynamics and evidence of fragmentation in this seagrass landscape from 1940 to 2008 in general and after specific hurricanes in particular.

Materials and Methods

Table 8

Computed Subset of 16 FRAGSTATS Configuration Metrics

Group	Landscape-level metrics	Acronym
Area and edge metrics		
	Total area (ha) (equivalent to VSA in previous chapters)	TA
	Largest patch index (percent)	LPI
	Total edge (m)	TE
	Edge density (m/ha)	ED
	Area-weighted mean patch area (ha)	AREA_AM
	Area-weighted mean radius of gyration (m)	GYRATE_AM
Shape metrics		
	Area-weighted mean shape index	SHAPE_AM
	Area-weighted fractal index	FRAC_AM
	Area-weighted mean perimeter-area ratio	PARA_AM
	Area-weighted related circumscribing circle	CIRCLE_AM
	Area-weighted contiguity index	CONTIG_AM
	Perimeter-area fractal dimension	PAFRAC
Aggregation metrics		
	Number of patches	NP
	Patch density (number of patches per ha)	PD
	Landscape shape index (dispersion)	LSI
	Area-weighted mean Euclidean nearest neighbor distance (m) (isolation)	ENN_AM

Note: See Appendix C for values calculated for each of the metrics. Mathematical definitions can be found in McGarigal et al. (2012).

Choosing Landscape-Level Metrics for Seagrass Landscape Pattern Analysis

Seagrass data resampled to 2.5-m grain size raster format were used in this chapter. In FRAGSTATS ver. 4.2 (McGarigal et al. 2012), landscape-level metrics were calculated from the seagrass raster file following an 8-neighboring cell rule, which meant a seagrass patch was defined as a group of cells adjacent to each other in at least one of the four ordinal or diagonal directions. Therefore, seagrass patches in the hostile sand matrix were always separated by at least one full pixel width (2.5m) from adjacent patches.

In this initial analysis, a preliminary subset of FRAGSTATS indices was chosen to satisfy the following criteria. First, due to the binary nature of the seagrass landscape in this study, only indices in the three following groups were calculated: area and edge metrics, shape metrics, and aggregation metrics. There were not enough classes and further information to apply core area metrics, contrast metrics, and diversity metrics. Second, density metrics that promote comparison among different landscapes were preferred to absolute metrics; for example, PD was chosen over TE. Third, only the area-weighted mean was kept among those statistics that summarize central distribution characteristics (e.g., mean, area-weighted mean, median, range, standard deviation, coefficient of variation) of the patch properties for the entire landscape. Area-weighted mean (a landscape-centric perspective) was chosen over the mean (a patch-centric perspective) due to its "reflection of the average conditions of a pixel chosen at random or the conditions that an animal dropped at random on the landscape would experience" (McGarigal et al. 2012). Further, none of the other patch-level statistics follow the normal distribution (data not shown here), making the interpretation of mean and standard deviation potentially misleading. Lastly, inherently redundant metrics were avoided. For example, mean patch size and PD are both functions of total area and number of patches, therefore inherently redundant. Total edge and edge density are completely redundant when comparing landscapes of identical size (McGarigal 2014). As seagrass landscapes in this study contain no background, AREA_AM, division index, effective mesh size, and splitting index are closely related and become redundant. Following these criteria, the preliminary subset of landscape-level metrics contained 16 metrics from three groups (area and edge metrics, shape metrics, and aggregation metrics) (Table 8). To further examine potentially empirically redundant metrics, Pearson correlations (*r*) among this subset of 16 chosen landscape metrics in Table 8 were calculated. For each group of highly correlated metrics ($r > 0.9$), only one was arbitrarily retained in the final subset of metrics for further quantification of landscape pattern over time. Note that West Ship Island in 2003 (outlier) and Horn Island in 2008 (incomplete map) were not included in this data set.

Principal Component Analysis (PCA) of Seagrass Landscape Metrics

Values of the final nine metrics were normalized using z-transformation (subtract mean and divide by standard deviation) so that the following multivariate ordination procedures were not unduly influenced by high-magnitude variables within the correlation matrix. A PCA using varimax rotation (an orthogonal technique where the factors are rotated 90^{o} to each other and thereby minimizing the number of variables that have high loadings on each factor (Yong and Pearce 2013)) was performed on the normalized data. Kaiser's criterion (keeping eigenvalues greater than 1.0) (Kaiser 1960) and the Cattell's scree test (plotting eigenvalues in descending order against component number and determining the inflexion point) (Cattell 1966) were used to determine how many principal components (PCs) to retain. The PCs or factor scores were then calculated for each combination of map date and FRAGSTATS metrics using the loadings for that factor. The factor score was used to plot the relative positions of map dates by island along the axis corresponding to that principle component. Seagrass landscape patterns and dynamics over time were then interpreted based on a biplot of the first two PCs with the aid of the patch size distribution and the maps generated in the previous chapters. The method was adapted from Riitters et al. (1995).

Results

Landscape-Level Metrics for Seagrass Landscape Pattern Analysis

Seven metrics were further removed from the preliminary subset of 16 chosen FRAGSTATS metrics due to high correlation ($r > 0.9$, Table 9 and 10). Total Area (TA) was found to be highly correlated ($r = 0.92$) with TE. Number of Patches (NP) was highly correlated with TE ($r = 0.94$), and LSI ($r = 0.93$). Patch Density was found to be highly correlated with five other metrics: ED ($r = 0.93$), ENN_AM ($r = 0.93$), and three shape metrics PARA_AM ($r = 0.93$), CIRCLE_AM ($r = -0.94$), and CONTIG_AM ($r = -0.92$). Total Edge was highly correlated with LSI ($r = 0.96$); whereas ED was perfectly correlated with the two shape metrics PARA_AM ($r = 1.00$), and CONTIG_AM ($r = -1.00$). Finally, the three area-weighted metrics AREA_AM, GYRATE_AM, and CIRCLE_AM were highly correlated to each other as well as SHAPE_AM ($r = 0.92$) and CONTIG_AM (-1.00). The final subset of FRAGSTATS metrics chosen to quantify seagrass landscape in the MSS included nine metrics, three in each group identified in Table 8: Area and Edge metrics - TA, LPI, GYRATE_AM; Shape metrics – PAFRAC, FRAC_AM, CIRCLE_AM; and Aggregation metrics – PD, LSI, ENN_AM (Table 10). Results for each of the nine metrics calculated on the five MS barrier islands for each year of mapping data are presented in Appendix C (Table A1, A2, and A3).

Table 9

Pearson Correlation Coefficient among Calculated FRAGSTATS Metrics

	Area and Edge Metrics						Shape Metrics						Aggregation Metrics			
	TA	LPI	TE	ED	AREA _AM	GYRATE _AM	SHAPE _AM	FRAC _AM	PARA _AM	CIRCLE _AM	CONTIG _AM	PA-FRAC	NP	PD	LSI	ENN _AM
TA	1.00	0.14	0.92	-0.32	0.54	0.64	0.56	0.66	-0.32	0.61	0.37	0.51	0.76	-0.36	0.83	-0.68
LPI		1.00	0.07	-0.43	0.77	0.71	0.76	0.62	-0.43	0.45	0.45	0.29	0.00	-0.40	0.05	-0.40
TE			1.00	-0.04	0.36	0.43	0.47	0.58	-0.04	0.46	0.10	0.61	0.94	-0.14	0.96	-0.62
ED				1.00	-0.51	-0.60	-0.42	-0.47	1.00	-0.67	-1.00	0.15	0.18	0.94	0.06	0.49
AREA_AM					1.00	0.97	0.92	0.75	-0.51	0.64	0.54	0.36	0.19	-0.47	0.28	-0.55
GYRATE_AM						1.00	0.92	0.83	-0.60	0.76	0.63	0.40	0.22	-0.58	0.34	-0.66
SHAPE_AM							1.00	0.88	-0.42	0.73	0.46	0.56	0.32	-0.48	0.43	-0.67
FRAC_AM								1.00	-0.47	0.90	0.53	0.76	0.38	-0.60	0.58	-0.87
PARA_AM									1.00	-0.67	-1.00	0.15	0.18	0.94	0.06	0.49
CIRCLE_AM										1.00	0.71	0.57	0.21	-0.80	0.46	-0.82
CONTIG_AM											1.00	-0.08	-0.13	-0.95	0.00	-0.55
PAFRAC												1.00	0.54	-0.07	0.73	-0.67
NP													1.00	0.09	0.94	-0.47
PD														1.00	-0.09	0.62
LSI															1.00	-0.66
ENN_AM																1.00

Note: West Ship Island in 2003 and Horn Island in 2008 were not included in the data set. Only one metric was retained per highly correlated group (|*r*| > 0.9, are highlighted in dark grey).

Final selected metrics are in bold. Metrics are represented in the same order as in Table 8.

Table 10

Groups of Highly Correlated Metrics and Representative Metrics Used in PCA

Group	Group members	Representative
1	TA	TA
2	LPI	LPI
3	AREA_AM – GYRATE_AM – SHAPE_AM	GYRATE_AM
4	FRAC_AM	FRAC_AM
5	CIRCLE_AM	CIRCLE_AM
6	PAFRAC	PAFRAC
7	ED – PARA_AM – CONTIG_AM – PD	PD
8	TE – LSI – NP	LSI
9	ENN_AM	ENN_AM

Principal Component Analysis of Seagrass Landscape Metrics

The nine eigenvalues of the normalized correlation matrix explained 100 percent of the variation in the original dataset (nine FRAGSTATS metrics by 22 map dates). The first two eigenvalues or PCs explained 80% of the variance in the data set (Table 11) and were retained for further data interpretation and visualization.

Table 11

Eigenvalues (Scaled) of Principal Component Analysis

	Principal Component								
	1	2	3	4	5	6	7	8	9
Standard deviation	2.33	1.23	0.90	0.72	0.46	0.36	0.18	0.13	0.11
Proportion of Variance	0.63	0.17	0.09	0.06	0.02	0.01	0.00	0.00	0.00
Cumulative Proportion	0.63	0.80	0.89	0.95	0.98	0.99	1.00	1.00	1.00

Table 12

Output of Principal Component Analysis

	PC1	PC2	h2	u2	com
TA	0.82	0.28	0.75	0.245	1.2
LPI	0.01	0.78	0.61	0.386	1
GYRATE_AM	0.37	0.82	0.81	0.192	1.4
FRAC_AM	0.65	0.73	0.95	0.047	2
CIRCLE_AM	0.51	0.78	0.87	0.128	1.7
PAFRAC	0.83	0.18	0.72	0.278	1.1
PD	-0.07	-0.84	0.71	0.287	1
LSI	0.96	-0.01	0.93	0.074	1
ENN_AM	-0.7	-0.59	0.84	0.16	1.9

Note. The table included loadings for the first two principal components after varimax rotation (PC1 and PC2), the communalities – merely the sum of squared factor loadings for that FRAGSTATS metric (h2), the uniqueness (u2 = 1 - h2), and the complexity of the component loadings for that FRAGSTATS metric (com). Loadings higher than 0.5 are highlighted in grey.

The first principal component (PC1) increased with increasing factor loadings due to dispersion (LSI = 0.96), area gain (TA = 0.82), shape complexity (PAFRAC = 0.83), patch complexity (FRAC_AM = 0.65), patch elongation (CIRCLE_AM = 0.51) and patch extent (GYRATE_AM = 0.37), and with decreasing isolation (ENN_AM = -0.7) (Table 12). This PC1 can be interpreted as a measure of the overall lushness of the landscape. A landscape with high values of PC1 would have vegetated patches that tend to be aggregated and abundant with convoluted shape, while a landscape with low values of PC1 would tend to be characterized by patches that are isolated with compact and simplified shape.

The second principal component (PC2) increased with increasing factor loadings due to dominance of the largest patch in the landscape (LPI = 0.78), with increasing patch extent (GYRATE_AM = 0.82), patch complexity (FRAC_AM = 0.73), patch elongation

(CIRCLE_AM = 0.78), and total area (TA = 0.28), and with decreasing subdivision (PD = -0.84) and isolation (ENN_AM = -0.59). This PC2 can be interpreted as representing the overall continuity and homogeneity of a landscape. A landscape with high values of PC2 would have vegetated patches that tend to be larger in size and more clustered, while a landscape with low values of PC2 would tend to be characterized by patches that are smaller and more spread out.

Maps with positive values of both PCs (upper right quadrat of Figure 14) would tend to be dominated by patches that are larger in area and more coalescent while maps with negative values of both PCs (lower left quadrat of Figure 14) would tend to be characterized by patches that are smaller in area and have become more isolated from each other. The positive values of PC1 and negative value of PC2 (lower right quadrat of Figure 14) could indicate sprouting of new patches and out-spreading of seagrass across the landscape while negative values of PC1 and positive values of PC2 (upper left quadrat of Figure 14) might suggest loss of patches and declining seagrass over the landscape.

Generally speaking, the smaller the temporal difference between two map dates for the same island, the closer they appeared on the bi-plot (Figure 14). Each deviation from the cluster seemed to link with a catastrophic event (see discussion on hurricane impacts). Except for Cat Island, seagrass maps of each island seemed to cluster and can be separated from other islands using the two PCs (Figure 14).

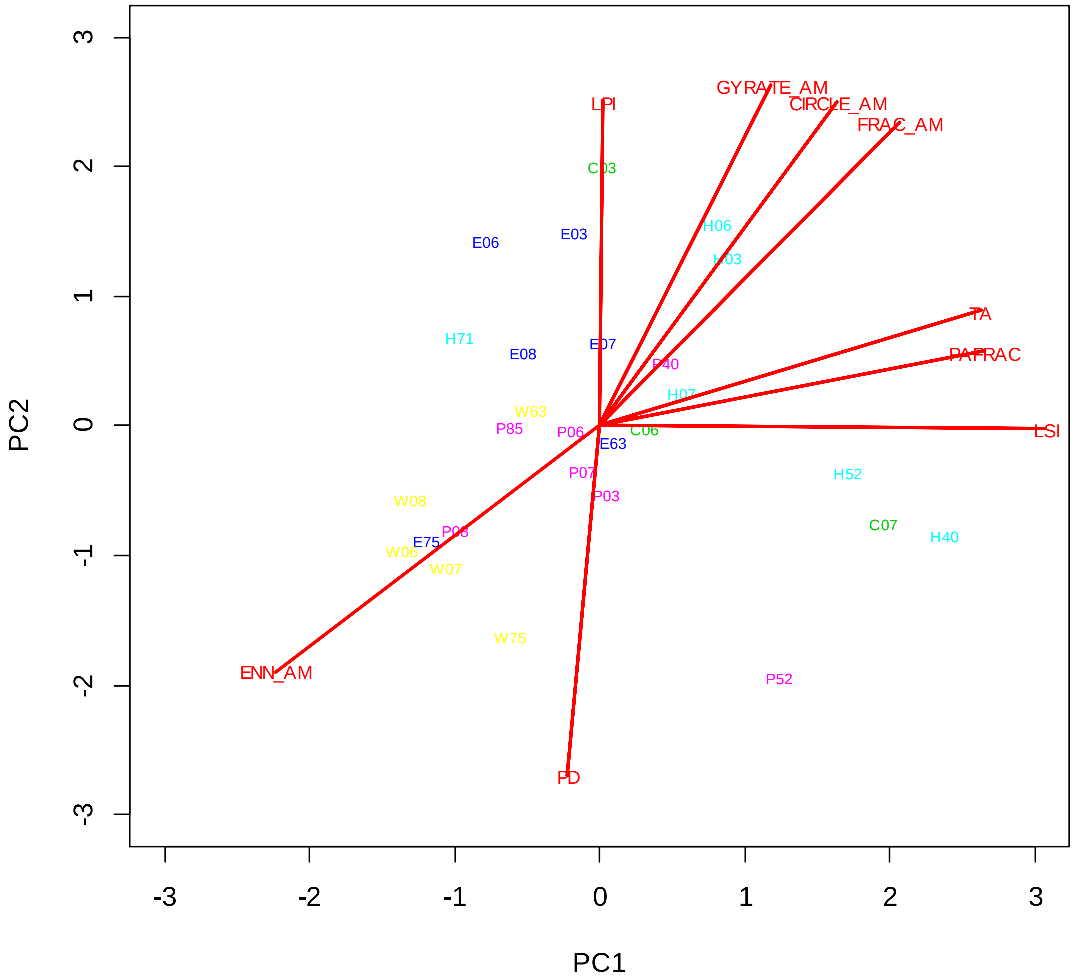

Figure 14. Bi-plot from Principal Component Analysis.

Note. Each data point represents an individual map year and is color-coded by island (Cat Island – green, West Ship – yellow, East Ship – blue, Horn Island – cyan, Petit Bois – magenta). The red arrows indicated FRAGSTATS metrics used in the analysis.

Cat Island in 2000s and Horn Island (except in 1971) had positive values of PC1 and might be considered more lush than the other islands. Cat Island seagrasses seemed to change from high continuity levels in 2003 to a more expanding yet disconnected landscape in 2007. East Ship Island had negative values on PC1 and positive values on PC2, suggesting a lower spatial expansion than on Cat and Horn islands. East Ship Island seagrasses looked isolated in 1975 but probably increased in continuity in the 2000s even though lushness was probably not as high as in 1963. In contrast, West Ship Island seagrasses seemed to be the most impacted with negative values of both PCs, suggesting an isolated and highly patchy seagrass landscape. It would be reasonable to say that seagrass on West Ship Island appeared isolated in all years except for in 1963. Horn Island seagrass patches appeared highly dispersed in the 1940s and 50s, then the seagrass landscape declined in 1971 before recovering and becoming more coalescent in the 2000s. Finally, seagrass on Petit Bois Island also exhibited declining values on PC1 and generally negative scores on PC2 and would be likely the second most isolated and patchy landscape after West Ship (Figure 14). From 1940 to 1952, although the expansiveness of seagrass on Petit Bois Island seemed to increase, the landscape continuity might have declined. The seagrass landscape on this island appeared most isolated in 2008.

Discussion

Landscape Pattern Analysis using FRAGSTATS Metrics

Seagrass landscape metrics, unsurprisingly, were sensitive to scale; it can be increasing, decreasing, or fluctuating with no clear trend. The set of FRAGSTATS metrics chosen to quantify landscape pattern and dynamics in this study was not the same group as those of Riitters et al. (1995) and Sleeman et al. (2005). Part of the reasons for this discrepancy may be the fine-resolution data in this study compared to data in the first paper, and the patchier nature of seagrass landscape in the MSS compared to the seagrass landscape in the second study. Small-sized patches were not removed from the analysis since the resampling process had already removed error-prone pixels occurring in the original resolution; those pixels retained were presumed to be a more realistic presentation of the landscape though this resampling could have made the data somewhat more noisy (see Chapter III). The pattern of vegetation growth presented by these small patches reflects the response of the vegetation to the hydrodynamic conditions of the area, dominated by westerly flow and sediment transport, and the process of recovery from ongoing disturbance in the form of sand bar movement and long-term westward migration of the islands themselves.

The behavior of each FRAGSTATS metric might appear erratic on its own, however, by studying the pattern in the data from all metrics combined, a more ordered pattern was revealed. The ordered pattern was revealed by the clustering of data points from the same island and smaller differences between nearby map dates (except for extreme events). Therefore, the combination of FRAGSTATS metrics provided a better synthesis of the landscape than could be found when analyzing just one metric alone, say VSA in Chapter III.

First, a landscape having an equal seagrass amount in two years can have a totally different degree of fragmentation and quality. For instance, Horn Island had the same amount of VSA in 1940 and 2006 but appeared less fragmented in 2006. However, the landscape in 1940 would have been much more connected for organisms of the same species with higher dispersal distance due to the higher overall geometric complexity and outspread of patch configuration.

Second, a combination of metrics makes the detection of a critical change in the landscape possible. As an example, the seagrass loss on Horn Island in 1940-1952 was a quantity change but further loss in 1971 made the landscape significantly more fragmented and indicated a quality change. This catastrophic shift was not able to be detected with information on total area only.

Third, patch-level indices and maps can provide helpful details to understand the behavior of landscape-level metrics. In particular, it may be confusing when seagrass areal gain co-occurs with reduced overall shape complexity, such as in the period of 2003-2006 on Horn Island or on East Ship Island in 2003-2008. The reason for this was the expansion of the seagrass boundary, which can be seen easily from overlay maps of loss and gain (see Chapter III). Confusion also happened when Horn Island lost a significant amount of seagrasses in 1971 but patch extent was higher than the two previous map dates. Without maps showing seagrass contraction (see Chapter III) and the decline in percentage of small-sized patches and their contribution to total VSA (see Chapter IV), it would be impossible to determine that numerous small patches disappeared and that the remaining larger patches became reduced in size and covered a much smaller portion of the island.

All of these examples demonstrate the necessity of a combination of total area, landscape metrics (both at patch- and landscape-levels), and spatial distribution of resources to form a comprehensive understanding of landscape pattern and dynamics.

Seagrass Landscape Pattern and Dynamics in the Mississippi Sound

The seagrass loss and fragmentation process was shown to be different among islands and at various temporal scales. However, as an overall trend across all islands, the seagrass landscape in the MSS exhibited early signs of area loss and fragmentation as far back as the 1940-1950s. The 1970s was then definitely the worst time for seagrasses in the MSS with loss of habitat and loss of seagrass species like *Thalassia testudinum* (Chapter II), the lowest total area combined across all islands, a faster rate of decline and a higher loss in total area than before 1970 (Chapter III), the low proportion of large-sized patches and their low contribution to total area, the reduced intensity of hot spot (Chapter IV), as well as the highest degree of fragmentation (this chapter). Recovery of seagrass occurred during the 1980s and 90s when there is less available data, with the landscape exhibiting characteristics of a more contiguous and more vegetated condition throughout the early 2000s.

The scarcity of studies quantifying seagrass fragmentation limits comparison between the MSS and other seagrass systems. The only comparable pair of data is between Petit Bois Island in 1985-2006 and Gold Coast Broadwater, Australia in 1987-2005 (Cuttriss et al. 2013). While overall seagrass loss or fragmentation was not evident on the former, the latter experienced increased landscape fragmentation but overall areal gain in seagrass coverage.

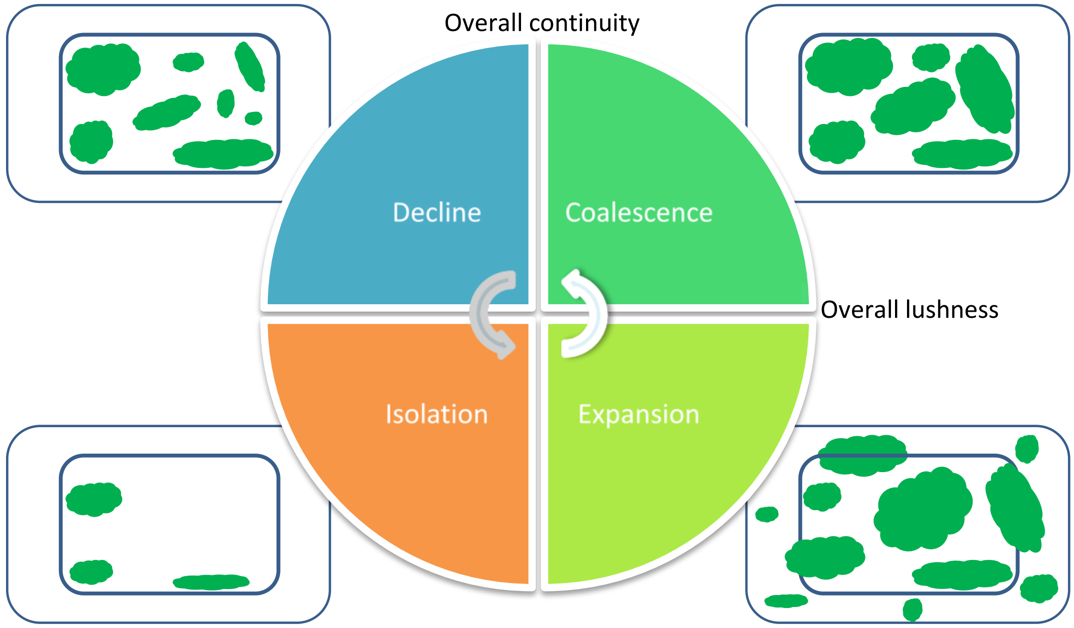

Figure 15. Conceptual model of seagrass landscape change in the Mississippi Sound.

A conceptual model of seagrass landscape change based on the two PCs, overall landscape lushness and continuity, is proposed for the MSS region (Figure 15). Seagrass loss would generally come with the loss of small patches, the contraction and disaggregation of large patches, resulting in a patch configuration of smaller size and extent, lower geometric complexity, lower connectivity, and higher isolation. The degree of fragmentation can be characterized by a certain combination of changes in landscape metrics, and these combinations occur in a certain order. When landscape metrics acting together were simplified into two PCs, the fragmentation process can be roughly divided into four stages: expansion, coalescence, decline, and isolation (Figure 15). When a seagrass landscape is in the expansion phase, it can be characterized by high overall expansion but low continuity due to the existence of many small, sporadic patches of new growth. If the conditions become unfavorable, the landscape boundary will contract and small patches may disappear but bigger patches still remain. Within the newly contracted boundary, the landscape is in the coalescence phase characterized by high overall lushness and continuity. If unfavorable conditions continue, patches may get both contracted and broken off, leaving a landscape with low lushness but still with high continuity. Finally, with continued decline the landscape becomes more isolated with fewer and more remote patches of smaller sizes, characterizing by both low lushness and low continuity. When the area loss becomes substantial the landscape boundary contracts again, and at this time the landscape could be considered to be returning to the expansion phase within the newly contracted boundary, and a new cycle could repeat if conditions kept worsening. This transition would be analogous to quantity changes leading to a change in quality. On the other hand, the trend can reverse if favorable conditions return. An increase in seagrass total area would generally occur first with new patches, and/or enlarged and coalesced existing patches within the current boundary, followed later by the appearance of small patches of new growth beyond the current boundary that make overall expansion higher but overall continuity lower. This suggests that the seagrass landscape in the MSS, though dynamic, might not be as unpredictable as it was originally thought when considering the landscape simultaneously from many different attribute metrics.

Effects of Hurricanes on the Seagrass Landscape

An application of this newly proposed conceptual model, combined with the unique long-term landscape data set, provided an opportunity to study different temporal scales of hurricane effects: immediate short-term effects (less than a year), medium-term effects (1-3 years), and long-term effects (3 – 10 years). Hurricane information was gathered during the period of record and the seagrass landscapes before and after each hurricane, where available, were compared to examine the direction of changes (Table 13).

Table 13

List of Seagrass Map Dates and Associated Hurricanes in the Mississippi-Chandeleur Sound in 1940-2008

Island – Year (seagrass data)	Hurricane – Year – Category	Lag time	Compari--son date
Short-term (less than a year)			
Chandeleur – Nov 2002	Lili – Oct 2002 – 1	1 month	2000
Chandeleur – Oct 2005	Cindy – Jul 2005 - 1 and Katrina – Aug 2005 - 3	2-3 months	2002, Jan 2005
Petit Bois – Oct 1985	Elena – Sept 1985 - 3	1 month	NA
Medium-term (1-3 years)			
Ship Island - 1963	Ethel – Sept 1960 - 1	3 years	NA
Horn Island - 1971	Camille – Aug 1969 - 5	2 years	NA
Chandeleur - 1999	Danny – Jul 1997 - 1	2 years	NA
5 Mississippi islands – 2006, 2007, 2008	Cindy – Jul 2005 - 1 and Katrina – Aug 2005 - 5	1, 2, 3 years	2003
Long-term (3-10 years)			
Horn, Petit Bois - 1952	NA – Sept 1947 - 3	5 years	1940
East Ship, West Ship - 1975	Camille – Aug 1969 - 5	6 years	1963
5 Mississippi islands - 2003	Georges – Sept 1998 - 2	5 years	NA

Short-term landscape effects could only be studied on the NCI and generally little to no impacts were evident at the landscape scale in those locations where seagrasses remained after the storm. However, large impacts to the overall extent of the seagrass landscape were observed, in particular after Hurricane Katrina in August 2005, when the NCI lost 25% of the VSA (see Chapter II), largely due to the extensive loss of the emergent islands themselves. Further, the apparent seagrass gain one year post-Katrina on the MS barrier islands (see Chapter II and III) may be due to sediment nutrients stirred up from the bottom during the storm and fostering seagrass or algae growth, similar to the more robust seagrass beds observed one year after Hurricane Camille (Eleuterius 1971). The lowest VSA within the study period recorded in 1971 could be the effect of quick die-off following the previous robust growth or a reflection of the ongoing decline of the seagrass landscape since the 1940s (Chapter III), or both.

The period of 2003-2008 served as a case study of the medium-term effects of a large hurricane on seagrasses in the MSS. The post-hurricane period of 2006-2008 was calm with no major storms and the closest opening of the Bonnet-Carré Spillway occurred in July 2008. A comparison between the seagrass landscape in 2003 (two years pre-Katrina) and the landscape after Hurricane Katrina revealed no medium-term damage, as previously reported in a field survey by Heck and Byron (2006). One year after the hurricane (2006) there were more seagrasses in the whole MSS; on all islands, except East Ship Island, seagrass patches appeared more connected (higher PC2 values, Figure 14) than in 2003. Two years after the hurricane (2007), there was evidence of seagrass loss and reduced connectedness on East Ship and Horn islands compared to 2003, but not on the other three islands. In contrast, from 2006 to 2007, the landscape gained seagrasses on Cat and West Ship islands, but seagrass loss and connectivity reduction were observed on the other three islands. Three years post-hurricane (2008), only Petit Bois Island had less seagrass than two years pre-hurricane, but the seagrass landscape exhibited less connectivity across the islands. Overall there was no apparent medium-term damage of Hurricane Katrina to the seagrass landscape observed in the MSS.

There is some limited evidence that long-term decline in a seagrass landscape may be linked to hurricane impacts through indirect effects on the barrier islands themselves. For instance, from 1940 to 1952, Horn Island declined in overall lushness (decline in PC1) and on Petit Bois Island landscape connectivity dropped drastically (decline in PC2), possibly as a result of impacts from an unnamed category 3 hurricane in September 1947 (Figure 14, Table 13); however, this may be confounded with less-than-maximum canopy in the 1952 imagery collected early in the growing season in April. Similarly, there was a substantial drop in landscape connectivity on West Ship Island from 1963 to 1975 and during the same period both overall lushness and connectivity declined on East Ship Island, most likely a long-term response to the damage caused by Hurricane Camille in August 1969 that resulted in the island being split into two. After Ship Island was cut into East and West Ship islands during Hurricane Camille, the severe damage on the seagrass landscape was still observed six years after the event. Inadequate pre-hurricane data prevented further examination of potential long-term effects of Hurricane Georges in 1998 on the MS islands.

Overall, the direct effects of hurricanes on the seagrass landscape tended to be short- or medium-term and depended on recovery duration after storm passage, while the indirect effects can be longer-term. A hurricane, if not leaving too much damage on the shoreline, was found to cause direct loss of VSA within a few months after the event, yet to promote fast growth in the summer that followed. In the following one or two years, seagrass can stay the same, continue to grow, or die off, probably depending on the magnitude of increase in the previous year and whether it had reached the carrying capacity. This shows resilience of the seagrass landscape after a big stress event. However, if a hurricane damaged the shoreline, the effect on seagrass beds will be long-term and more severe. It is important to note that the passage of a hurricane not only shows up on VSA but also is reflected in the spatial pattern of the seagrass landscape, especially landscape connectivity. In many cases, overall connectivity tends to decline, which can be explained either due to the damaging effect on large patches or sprouting of new, small patches during subsequent recovery.

CHAPTER VI – SUMMARY

Seagrass Status and Changes in the Mississippi-Chandeleur Sound

In the MS-C, seagrasses were found in three main groups representing an inshore to offshore gradient: along the MS mainland coastline, specifically the GNDNERR and Waveland locations dominated by *Ruppia maritima* (Group 1), on the northern shoreline of the MS barrier islands (Cat, West and East Ship, Horn, and Petit Bois islands, plus the Sound side of Cat Island) dominated by *Halodule wrightii* (Group 2), and on the western shoreline of the NCI dominated by *Thalassia testudinum* co-occurring with other seagrass species (Group 3). At the regional scale, seagrasses in the MS-C had remained in these locations from 1940 to 2011.

Changes in Species Composition

Seagrass species historically found in the MS-C include *Halodule wrightii* Asch. (shoal grass), *Halophila engelmannii* Asch. (star grass), *Ruppia maritima* L. (widgeon grass), *Syringodium filiforme* Kütz. (manatee grass), and *Thalassia testudinum* Banks ex König (turtle grass). While all five reported seagrass species still can be found on the NCI, three of them (*H. engelmannii, S. filiforme*, *T. testudinum*) disappeared in MS during the early 1970s. Seagrass communities in the mainland (Group 1) and NCI (Group 3) did not experience species loss but change in relative abundance of seagrass species has also been documented, with an increase in the opportunistic and widely tolerant *R. maritima* observed in the past decade.

Changes in Seagrass Habitat

Suitable habitat for seagrasses in the MS-C has changed at all study sites. In Group 1, the seagrass habitat in Waveland diminished due to beach renourishment, whereas the loss of the Grand Batture Islands exposed GNDNERR shorelines to increased wave erosion and resulted in habitat loss. In Groups 2 and 3, progressive land loss and westward movement, island narrowing, and segmentation of the islands themselves has led to the shortening and thinning of habitat in both north-south and east-west directions.

Changes in Vegetated Seagrass Area

The seagrass landscape in the MS-C is an extremely dynamic system with a high degree of fluctuation in both VSA and spatial location of seagrass. From 1940 to 2008, total VSA decreased on each MS island and on all islands combined (excluding Cat Island); the estimated net loss represented 63.1% of total VSA present in 1940. Rate of total VSA loss on the four MS islands (except Cat Island) was 1.5% per year (or an annual change of 1.5 ha per year) in 1940-2008, similar to the mean global seagrass decline in 1879-2006, and higher than the pre-1940 global loss rate of 0.9% per year.

The decadal time when seagrass loss started and accelerated was not the same for all locations in the MSS. From west to east, the lowest points of VSA on each island were in 2000 (West Ship), 1980 (East Ship), 1970 (Horn), and 2008 (Petit Bois), while the four islands combined had the least amount of seagrass in the decade of 1970-1980. The faster rate of decline and higher loss of VSA on MS islands before 1970 contradicted the faster global loss rate and larger global net seagrass loss after 1970. The post-1980 decline rate and net change in the MSS was not as high as the median global rate of 5% per year and the global estimation of -37 km^2 per site per decade. As for rate of increase, the MSS experienced the same acceleration from the 1970s to the 1990s as the global seagrass meadows but at a smaller magnitude. Overall the MS barrier islands lost more seagrass at a much higher rate in the first part of the study period, from 1940 to mid-1970s, than in the later years. Prior to the 1970s, seagrass loss dominated on the MS islands and an increase in VSA was not recorded at any sites before 1960. The trend reversed when seagrass recovery enhanced and overruled loss from 1960 to 2000, but declines have occurred again in the 2000s.

Within the 2000s, seagrass loss was not as dramatic and increases in cover were recorded in the three islands in western MSS, which may be partly due to change in species composition. The within-decade net change and annual rate of change in VSA could be equal to or greater than the decadal change.

Changes in Spatial Distribution and Extent of Seagrass

Contraction of seagrass distribution was observed on all islands within the MS-C. Each island had its unique trajectory of location changes with large fluctuations between map dates regardless of time intervals. Seagrasses shrank back to the center on the two Ship islands, migrated westward on Horn Island (higher eastern loss than western gain), and were lost at the eastern end of Petit Bois Island with no gain at the western end. The seagrass landscape flattened in the north-south direction on West Ship, Horn, and Petit Bois islands. These observations indicate a generalized trend for retraction in the seagrass extent over the study period.

Seagrass Patch Size Distribution

Seagrass landscapes on all islands in the MS-C were skewed toward small patches; the degree of skewness was higher on the MS islands than on the NCI. Seagrasses on the NCI existed in continuous beds stretching more than several hundred hectares while there was no seagrass patch larger than 10 ha found on any of the MS islands and no patch bigger than 1 ha on West Ship Island. However, the large portion of small patches less than 25 m^2 in the MSS only contributed a modest portion of total area, whereas the smaller proportion of large patches greater than 1000 m^2 accounted for a considerable proportion of total area. On the NCI, the patch size distribution was less unbalanced with a higher percentage of large patches, which in turn accounted for more than 98% of total area across all years. This positively skewed patch size distribution in the MSS shows agreement with other seagrass landscape studies on different seagrass species and was documented as the result of rapid turnover rate of small patches.

Changes in cumulative percent of VSA and NP could provide evidence of seagrass fragmentation in the MS-C. For example, the highest number of patches larger than 1 ha and the highest relative contribution of large-sized patches on Horn Island in 2006 potentially indicated a growth and merging process of patches. This post-Katrina (2003-2006) fast growth appeared to be unstable, evident as a decrease in number of patches and in the relative importance of large-sized patches in 2007. This could be a part of a seagrass process but could also relate to algae or detritus abundance after the storms.

Spatial Distribution of Patch Clusters

The spatial distribution of seagrass patches along each island in the MS-C was not totally random with regard to patch metrics, local pockets of association were found on all islands. The spatial pattern of hot and cold spots was quite stable within the 2000s but was generally less so among earlier decades. This might suggest an underlying inherent stable spatial configuration within these seagrass landscapes over multiple years. For instance, the general location of seagrass clusters remained similar in consecutive years (2006-2007-2008), but was found to be more variable after a big storm event (e.g., from 2003 to 2006 after Hurricane Katrina) or over longer periods (e.g., decadal time).

Changes in Spatial Pattern at Landscape-Scale Level

The seagrass loss and fragmentation process was shown to be different among islands and at various temporal scales. West Ship and Petit Bois islands experienced seagrass loss and fragmentation both in the overall study period and at a decadal scale, while there was seagrass loss but not very clear evidence of landscape fragmentation on East Ship and Horn islands.

However, as an overall trend across all islands, the seagrass landscape in the MSS exhibited early signs of area loss and fragmentation as far back as the 1940-1950s. The 1970s was then definitely the lowest point of the MSS seagrass landscape within the study period, with loss of habitat and loss of seagrass species like *Thalassia testudinum* (Chapter II), the lowest total area combined across all islands, a faster rate of decline and a higher loss in total area than before 1970 (Chapter III), the low proportion of large-sized patches and their low contribution to total area, the reduced intensity of hot spots

(Chapter IV), as well as the highest degree of fragmentation (Chapter V). Recovery of seagrass occurred during the 1980s and 90s when there is less available data, with the landscape exhibiting characteristics of a more contiguous and more vegetated condition throughout the early 2000s.

Overall, changes were observed on all attributes of the seagrass landscape (from habitat, species composition, total area, patch size distribution, to spatial configuration at patch- and landscape-levels) at all spatial scales and in all temporal scales (year-to-year within the 2000s, decade-to-decade, and over the study period 1940-2011). This study suggests that seagrass landscapes in the MS-C, though dynamic, might not be as unpredictable as it was originally thought when considered simultaneously from many different angles.

Conceptual Model of Seagrass Landscape Change

A conceptual model of seagrass landscape change based on the two PCs, overall landscape lushness and continuity, is proposed for the MSS region (Figure 15). Seagrass loss would generally come with the loss of small patches, the contraction and disaggregation of large patches, resulting in a patch configuration of smaller size and extent, lower geometric complexity, lower connectivity, and higher isolation. The degree of fragmentation can be characterized by a certain combination of changes in landscape metrics, and these combinations occur in a certain order. When landscape metrics acting together were simplified into PCs, the fragmentation process can be roughly divided into four stages: expansion, coalescence, decline, and isolation (Figure 15). When a seagrass landscape is in the expansion phase, it can be characterized by high overall expansion but low continuity due to the existence of many small, sporadic patches of new growth. If the conditions become unfavorable, the landscape boundary will contract and small patches may disappear but bigger patches remain. Within the newly contracted boundary, the landscape is in the coalescence phase characterized by high overall lushness and continuity. If unfavorable conditions continue, patches may get both contracted and broken off, leaving a landscape with low lushness but still with high continuity. Finally, with continued decline the landscape becomes more isolated with fewer and more remote patches of smaller sizes, characterizing by both low lushness and low continuity. When the area loss becomes substantial the boundary contracts again, and at this time the landscape could consider to be returning to the expansion phase within the newly contracted boundary, and a new cycle could repeat if conditions keep worsening. This transition would be analogous to quantity changes leading to a change in quality. On the other hand, the trend can reverse if favorable conditions return. An increase in seagrass total area would generally occur first with new patches, and/or enlarged and coalesced existing patches within the current boundary, followed later by the appearance of small patches of new growth beyond the current boundary that make overall expansion higher but overall continuity lower.

Potential Drivers of Changes

Principal factors responsible for the observed pattern of seagrass distribution in the MS-C include water depth, salinity, type of substrate, and protection from wind and waves. Potential drivers of change in seagrass habitat, total area, and spatial configuration in the Sounds are island migration, salinity depression due to frequent openings of Bonnet Carré Spillway, hurricanes, and direct human impacts.

There was a strong dependency of seagrass distribution on the migration and geomorphology of barrier islands. The MS barrier islands experienced an ongoing westward movement and continual land loss (barrier island narrowing and segmentation). The responsible factors for an overall shrinkage in spatial extent of VSA were Camille cut on Ship Island, westward movement on Horn Island, and eastern shrinkage accompanied by western channel dredging on Petit Bois Island. Seagrass distribution on Horn Island shrunk in both north-south and east-west directions mirroring the geomorphological changes of the island shoreline. The reduction in seagrass habitat in 1969-1992 at GNDNERR can be partly explained by losing protection from the Grand Batture Islands by 1980. Fluctuations found in the Waveland in the early 1970s and the late 2000s were due to effects of Hurricane Camille in 1969 and Hurricane Katrina in 2005. The Chandeleur Islands underwent breaching, thinning in an east-west direction, and erosion on both the northern and southern tips, especially after storms and hurricanes.

The disappearance of seagrass species, other than *H. wrightii*, on the MS islands was highly correlated with low salinity depression following high rainfall and the frequent openings of Bonnet Carré Spillway during 1973-1983 following damage caused by Hurricane Camille in 1969. In contrast, the Chandeleur Islands retained its seagrass diversity but the relative abundance of the five species has changed due to similar reasons (Darnell et al. 2017).

Hurricanes could affect seagrass in the Sound both directly and indirectly. The direct effects of hurricanes on a seagrass landscape tended to be short- or medium-term and depended on recovery duration after storm passage, while the indirect effects can be longer-term. A hurricane, if not leaving too much damage on the shoreline, was found to cause direct loss of VSA within a few months after the event, yet to promote fast growth in the summer that followed. In the following one or two years, seagrass can stay the same, continue to grow, or die off, probably depending on the magnitude of increase in the previous year and whether it had reached the carrying capacity. This shows resilience of the seagrass landscape after a big stress event. However, if a hurricane damaged the shoreline, the effect on seagrass beds will be long-term and more severe. It is important to note that the passage of a hurricane not only shows up on VSA but is also reflected in the spatial pattern of the seagrass landscape, especially landscape connectivity. In many cases, overall connectivity tends to decline, which can be explained either due to the damaging effect on large patches or sprouting of new, small patches during subsequent recovery.

Lastly, direct human impact might play an important role in the changes of seagrass landscape in the MSS. Boating activities, including recreational, trawling, fishing, and dredging for ship channel expansion, might all contribute to the changes in seagrass.

Implications for Natural Resource Mapping and Change Analysis

Comparing Mapping Data from Multiple Sources

An important finding with wider implications for natural resource mapping is that direct comparisons of seagrass area among various studies that used different mapping methods can result in large errors and misleading conclusions. Earlier publications calculated seagrass extent and greatly inflated the number of hectares reported, while with the development of mapping technology, later publications tended to report seagrass coverage or VSA area at a finer scale and more accurately than the inflated number calculated in earlier publications. Direct comparisons of seagrass area among studies using different techniques may yield an erroneous conclusion of dramatic change over time simply because of the mapping methods employed. Differences in mapping objectives and methods among studies can complicate comparisons among years. Further comparison requires a common resolution, a common definition of the mapping unit and study object, and a threshold to distinguish between real changes in VSA and mapping errors. In this particular study, although the data seemed abundant at first, only data from Carter et al. (2011) allowed further detailed analysis in both total area change and spatial analysis.

Landscape Pattern Analysis across Scales and of Multiple Indices

The study demonstrates a comprehensive analysis across spatial and temporal scales and of multiple indices to provide insights on the pattern and dynamics of the seagrasses landscape. Landscape metrics are sensitive to scale and the behavior of each FRAGSTATS metric might appear erratic on its own. However, the combination of

FRAGSTATS metrics provided a better synthesis of the landscape than could be found when analyzing just one metric alone. For example, relying on total area alone misses key features of the landscape configuration and may fail to detect ecologically relevant attributes that can only be determined by exploring spatial attributes of the landscape patches in more details. Finer details at the patch-level and spatial distribution maps can aid the pattern analysis at the landscape level by explaining in part the underlying process. All of these demonstrate the necessity of a combination of total area, landscape metrics, and spatial distribution maps to a comprehensive understanding of landscape pattern and dynamics.

No individual scale or index can capture all spatial patterns and dynamics. Data generalization (common thematic content and common scale) is essential for change detection; note that resampling effects should be accounted for when comparing data. A comprehensive analysis across scales and of multiple indices is mandatory to describe and detect change in landscape configuration. This study highlights the various complexities inherent in habitat mapping, which in turn is essential for natural resource management and conservation.

There does not exist a specific scale or a set of metrics that works for all landscapes. The choice of scale depends upon the process of interest, whilst the choice of metrics can be arbitrary or can result from a thorough process based on multivariate statistics. Using a different suite of FRAGSTATS metrics to study the spatial pattern and dynamics would have affected the outcomes for the different landscapes. Future studies should more comprehensively explore these combinations and generate a range of output that is more method independent and can be compared across landscapes.

APPENDIX A – Spatial Distribution of Vegetated Seagrass Area Change

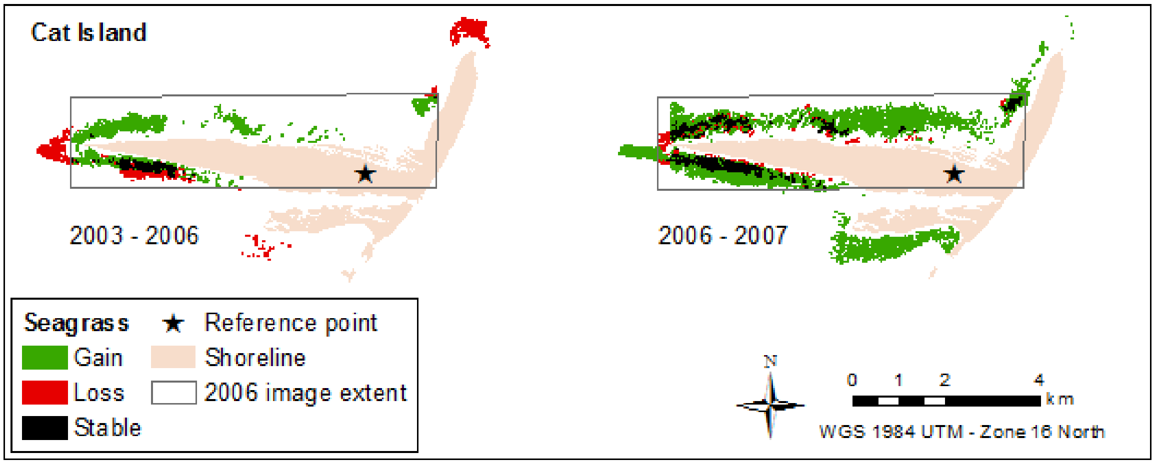

Figure A1. Spatial distribution of vegetated seagrass area change on Cat Island.

Note. The shoreline is of the later date in each comparison. No data prior to 2003 for Cat Island. Star indicates a common georeferenced point among all years.

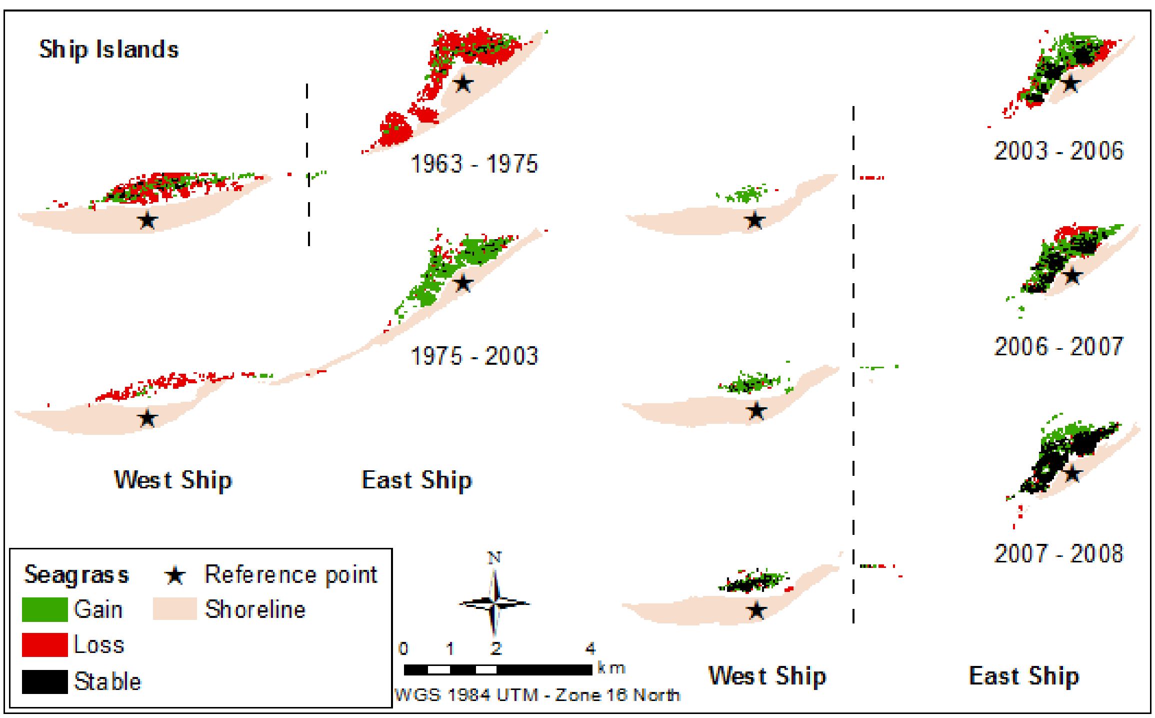

Figure A2. Spatial distribution of vegetated seagrass area change on the two Ship Islands.

Note. The shoreline is of the later date in each comparison. Dashed lines indicate separation between East and West Ship islands. Star indicates a common georeferenced point among all years.

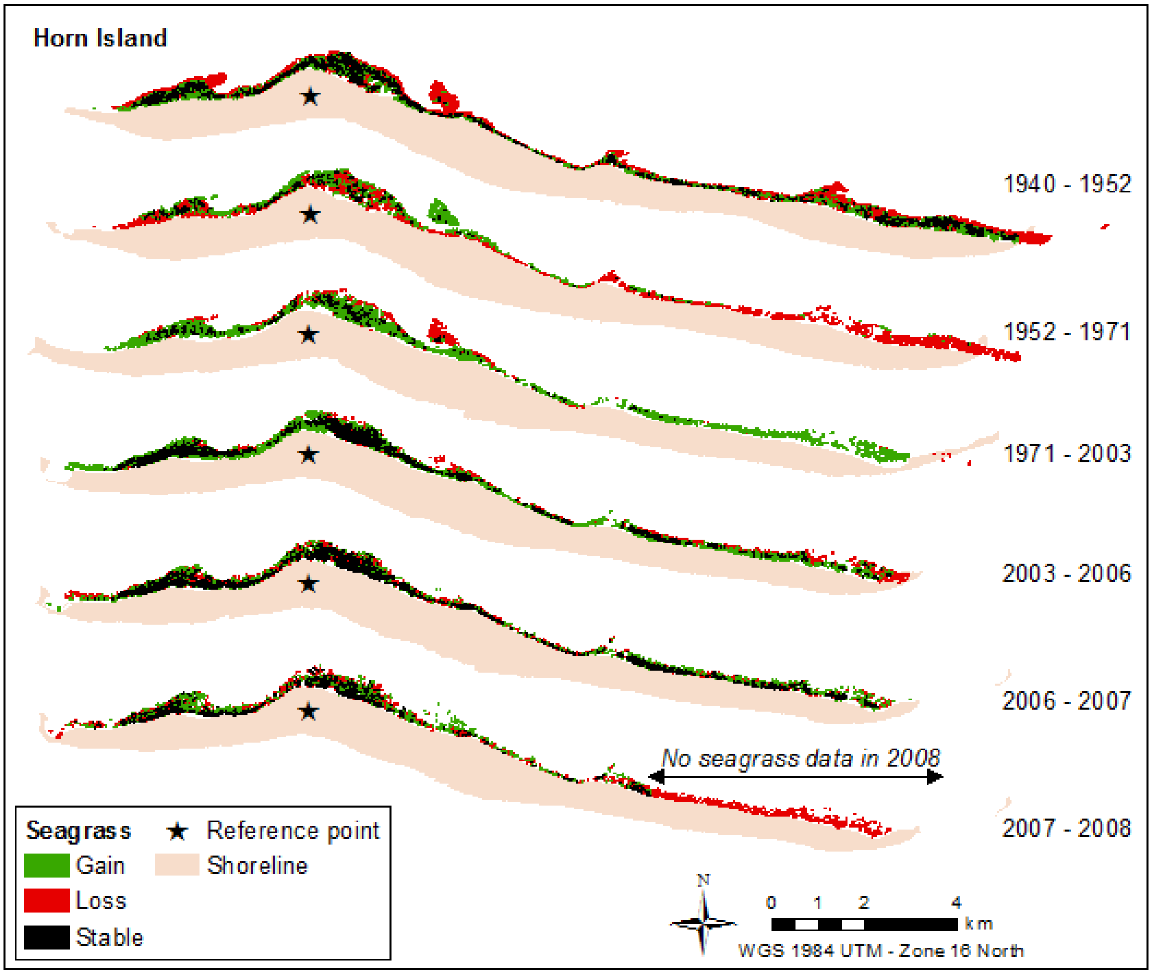

Figure A3. Spatial distribution of vegetated seagrass area change on Horn Island.

Note. The shoreline is of the later date in each comparison. Star indicates a common georeferenced point among all years.

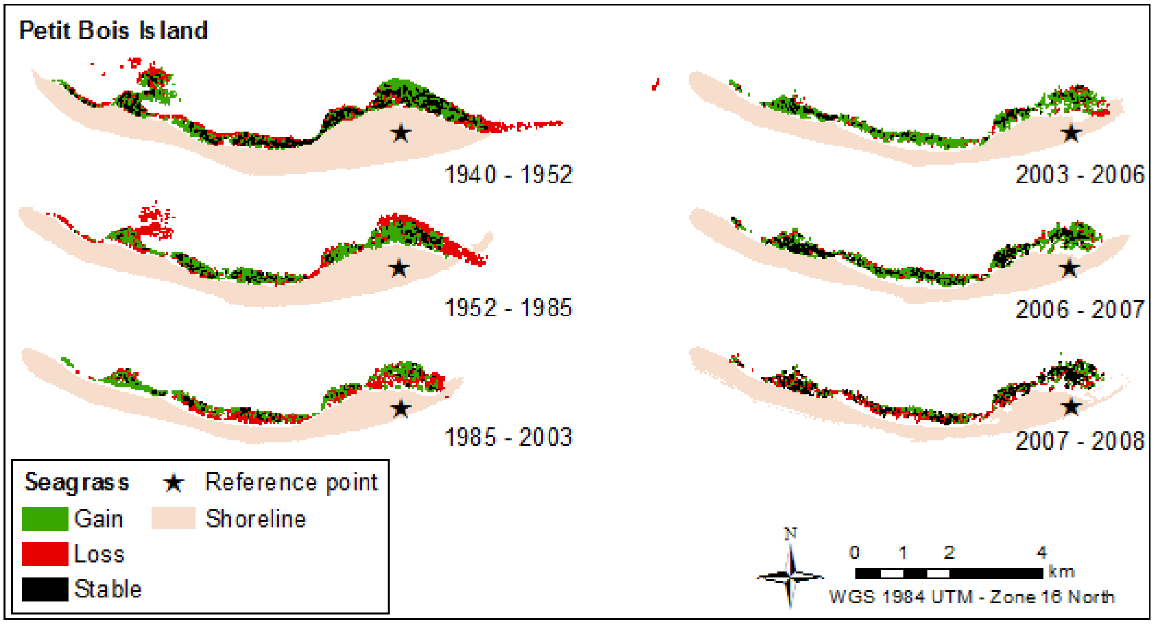

Figure A4. Spatial distribution of vegetated seagrass area change on Horn Island.

Note. The shoreline is of the later date in each comparison. Star indicates a common georeferenced point among all years.

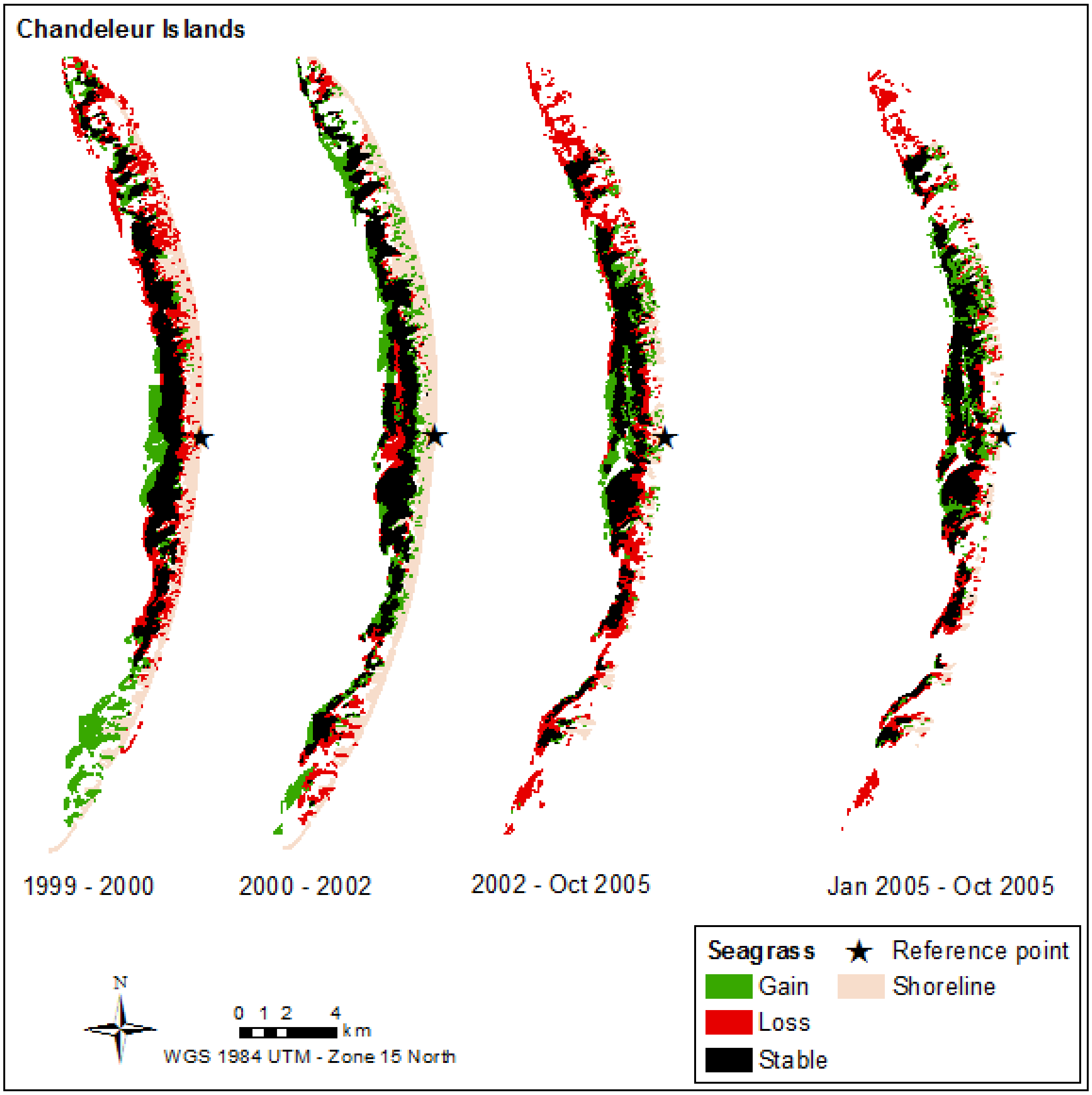

Figure A5. Spatial distribution of vegetated seagrass area change on Chandeleur Islands.

Note. The shoreline in 2005 was highly fragmented and is difficult to see in this map. Star indicates a common georeferenced point among all years.

APPENDIX B – Spatial Distribution of Hot Spots of Vegetated Seagrass Area Change

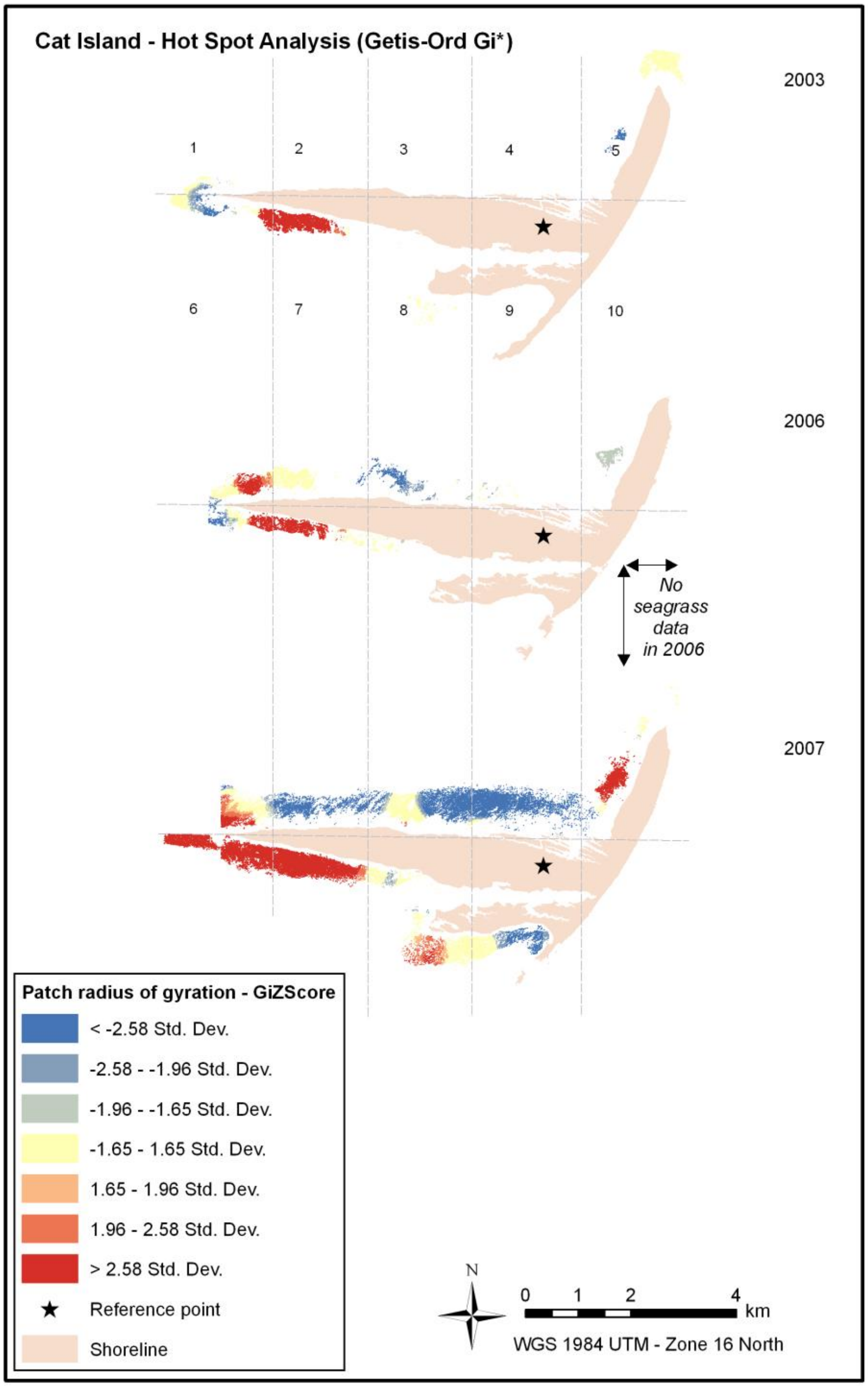

Figure A6. Hot spot analysis of seagrass landscape on Cat Island.

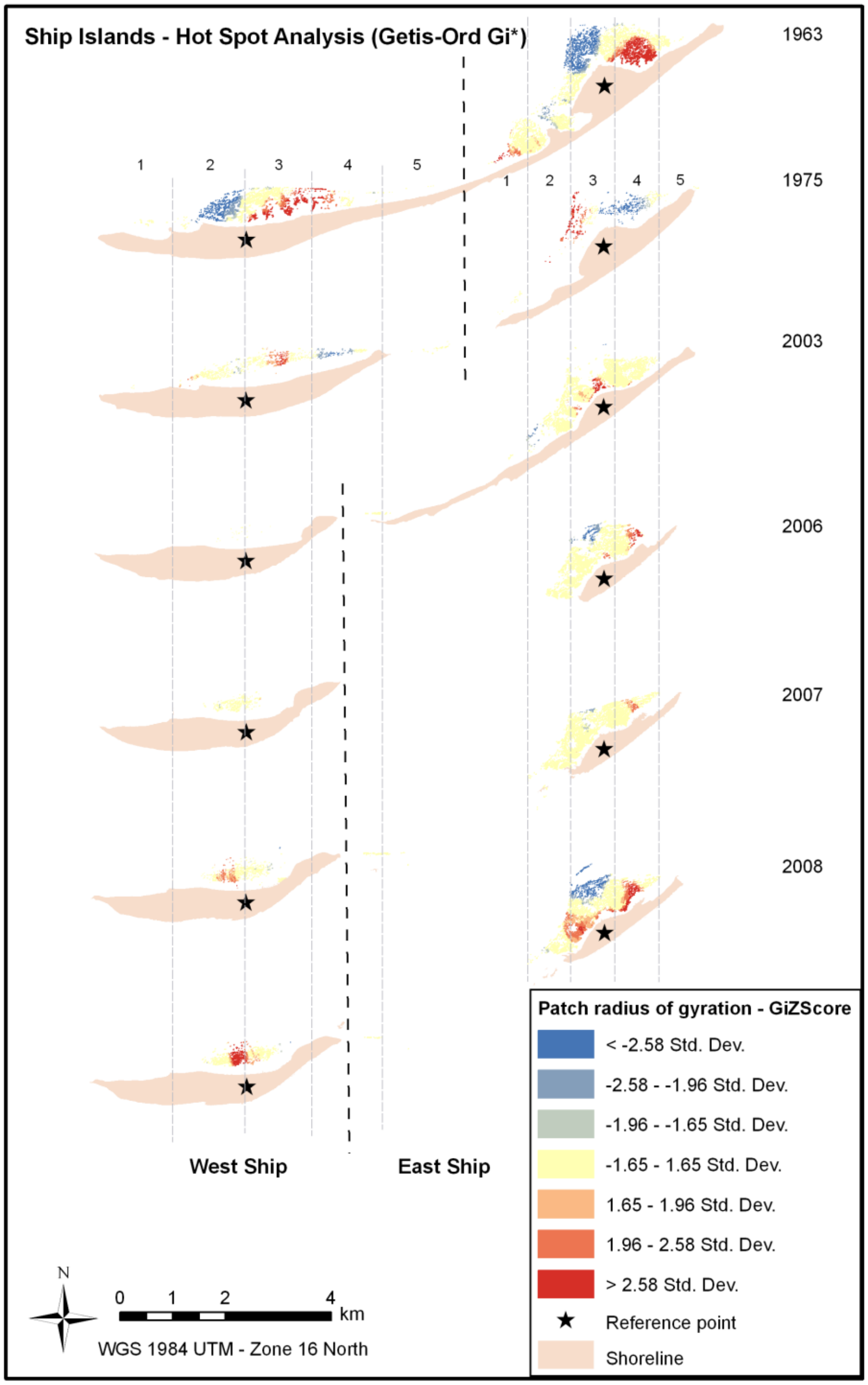

Figure A7. Hot spot analysis of seagrass landscape on Ship islands.

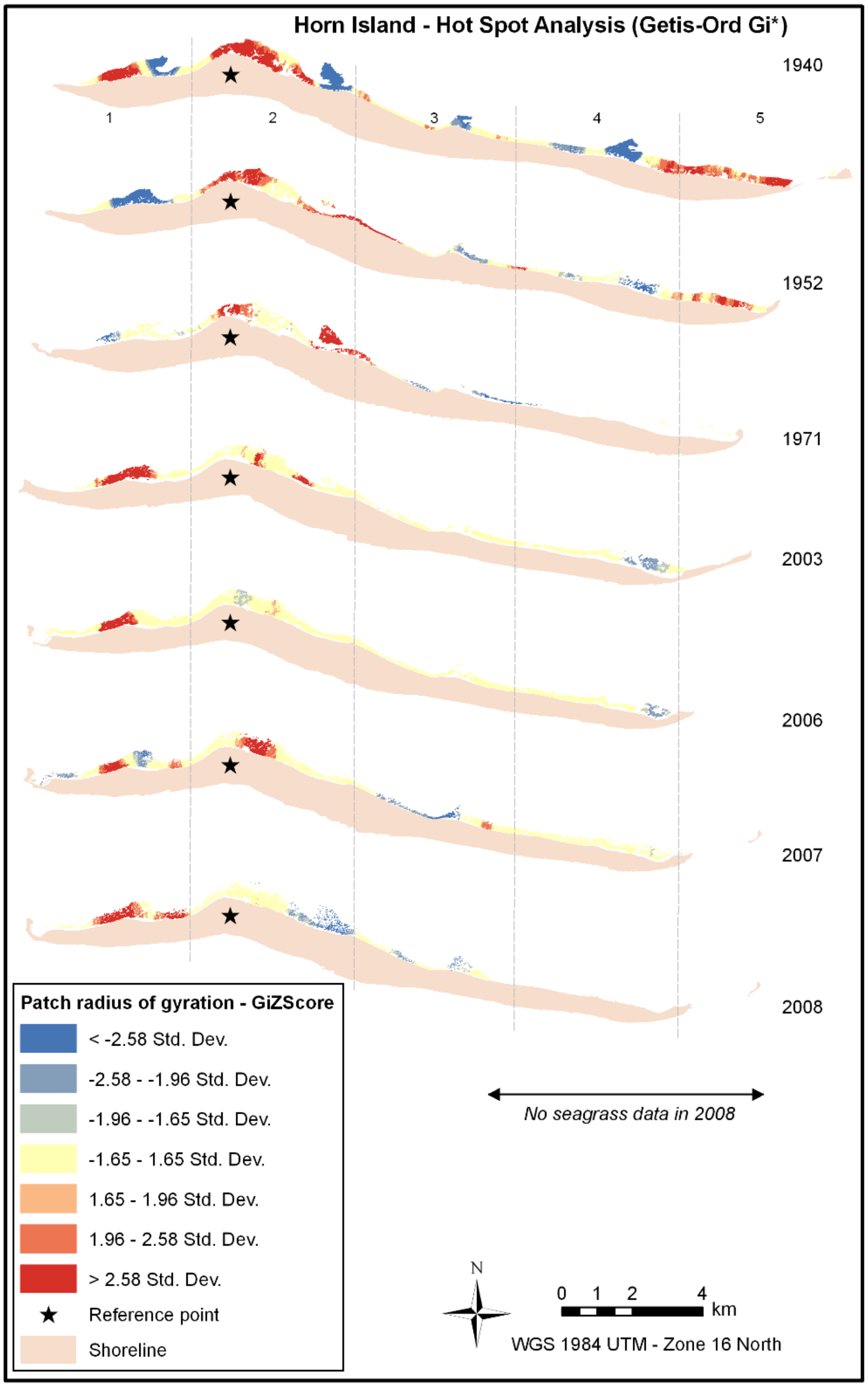

Figure A8. Hot spot analysis of seagrass landscape on Horn Island.

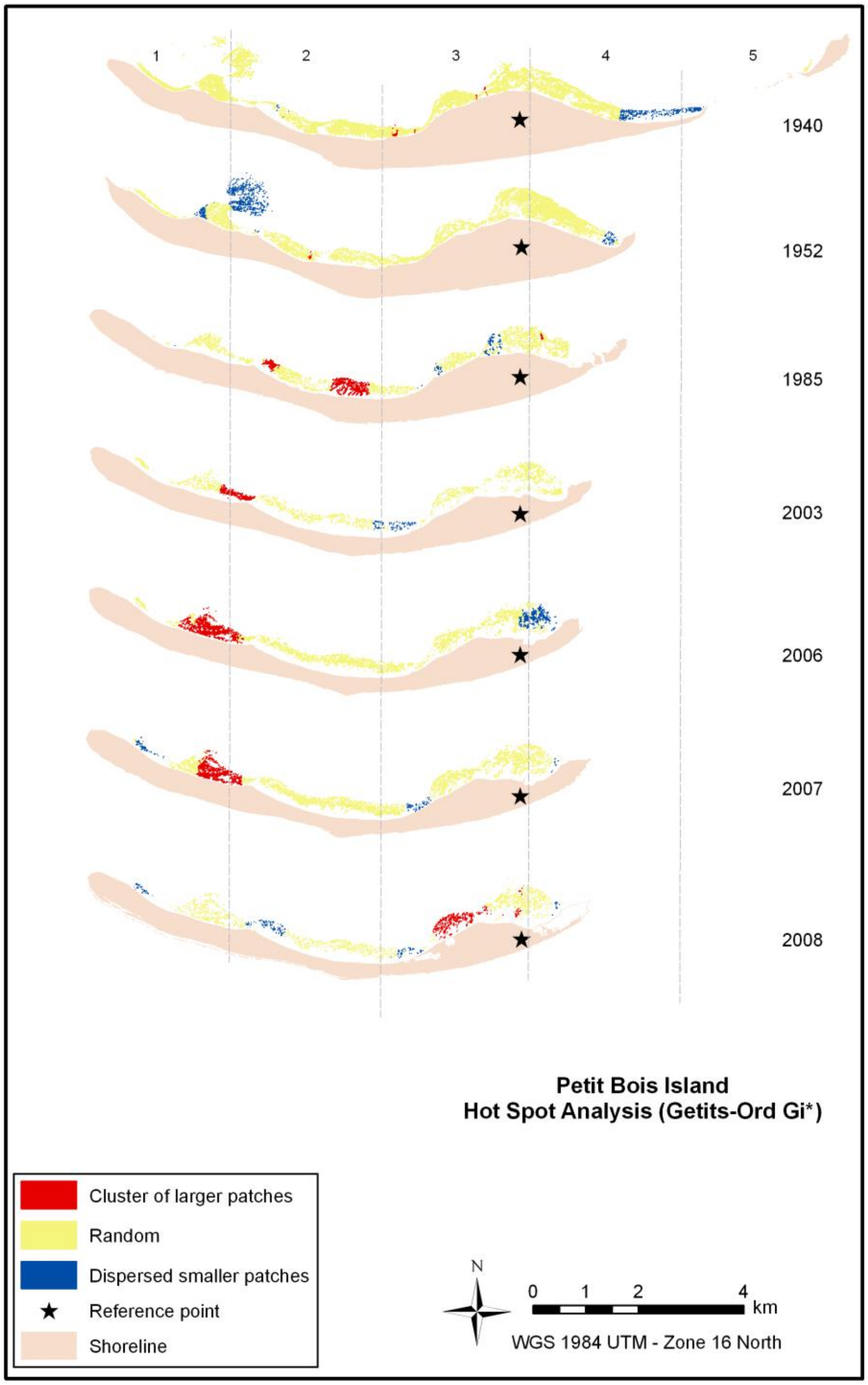

Figure A9. Hot spot analysis of seagrass landscape on Petit Bois Island.

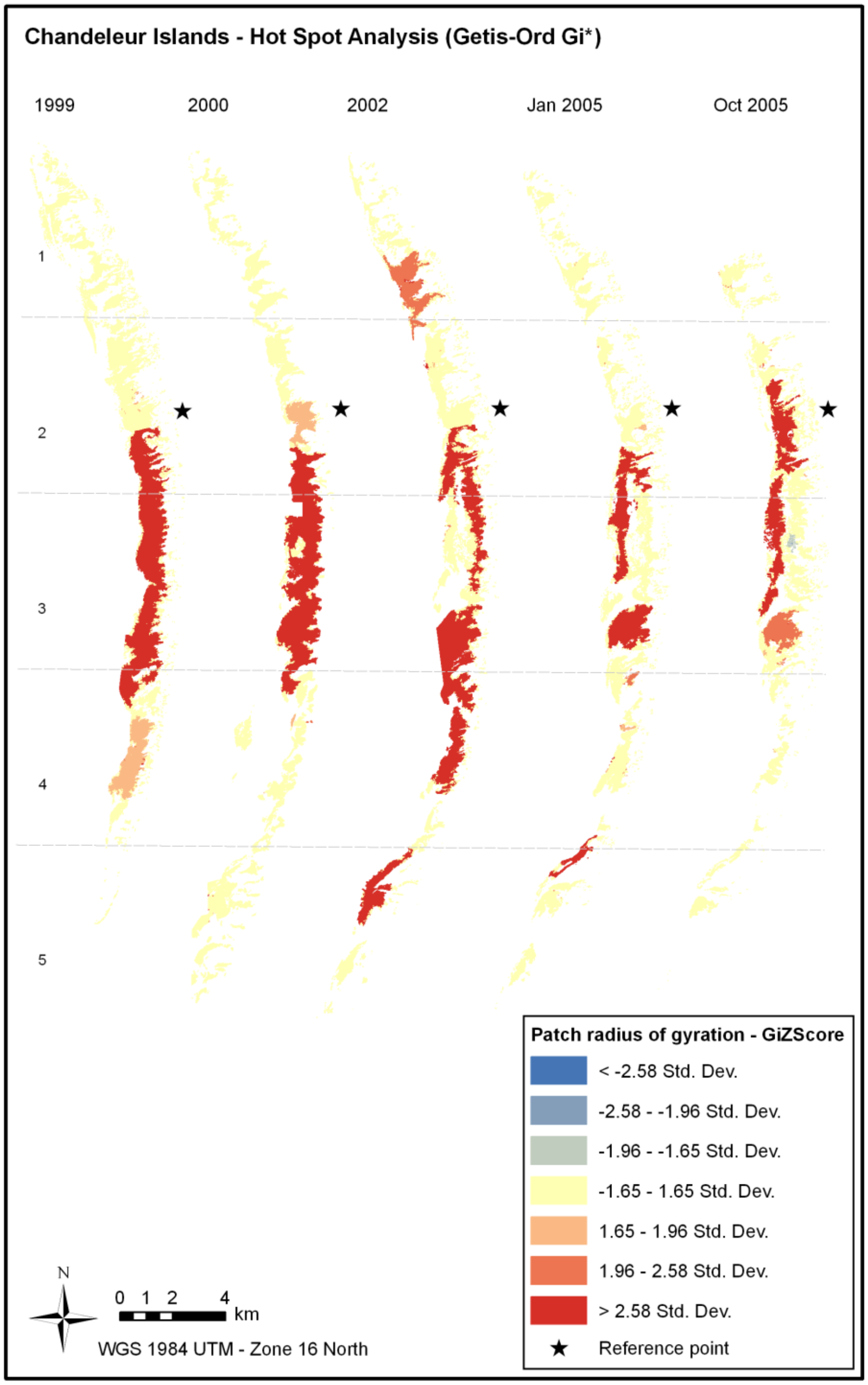

Figure A10. Hot spot analysis of seagrass landscape on Chandeleur Islands.

APPENDIX C – Calculated FRAGSTATS Metrics

Table A1.

Seagrass Landscape-Level Metrics – Area and Edge

Island	Year	TA	LPI	GYRATE_AM
Cat				
	2003	21.8	24.1	56.6
	2006	25.5	2.9	14.1
	2007	71.1	8.0	31.9
West Ship				
	1963	11.1	2.7	10.2
	1975	1.9	1.1	2.7
	2003	0.0	7.4	1.3
	2006	0.9	2.6	3.2
	2007	1.7	1.3	3.2
	2008	3.2	1.0	4.0
East Ship				
	1963	19.4	5.5	11.5
	1975	1.6	10.3	4.8
	2003	16.4	18.6	38.0
	2006	15.5	16.1	32.1
	2007	13.9	9.7	19.5
	2008	17.9	9.1	20.1
Horn				
	1940	76.7	4.7	17.6
	1952	45.7	4.1	16.5
	1971	19.2	15.6	22.2
	2003	50.8	13.3	53.4
	2006	82.0	7.9	70.3
	2007	38.2	8.0	27.2
	2008*	18.6	13.2	26.7
Petit Bois				
	1940	54.2	4.0	27.0
	1952	15.2	0.4	3.6
	1985	17.7	2.3	9.7
	2003	8.0	1.8	5.8
	2006	18.9	2.7	10.6
	2007	16.8	2.1	7.3
	2008	7.3	0.8	4.2

Table A2.

Seagrasss Landscape-Level Metrics – Shape

Island	Year	PAFRAC	FRAC_AM	CIRCLE_AM
Cat				
	2003	1.51	1.37	0.72
	2006	1.43	1.26	0.66
	2007	1.46	1.28	0.62
West Ship				
	1963	1.40	1.23	0.67
	1975	1.38	1.13	0.52
	2003	1.57	1.03	0.12
	2006	1.28	1.12	0.51
	2007	1.33	1.13	0.53
	2008	1.29	1.12	0.54
East Ship				
	1963	1.43	1.23	0.65
	1975	1.37	1.15	0.52
	2003	1.49	1.35	0.68
	2006	1.38	1.28	0.67
	2007	1.47	1.30	0.68
	2008	1.38	1.23	0.64
Horn				
	1940	1.54	1.30	0.66
	1952	1.56	1.32	0.69
	1971	1.34	1.20	0.59
	2003	1.50	1.36	0.74
	2006	1.43	1.35	0.75
	2007	1.44	1.26	0.69
	2008*	1.36	1.23	0.65
Petit Bois				
	1940	1.39	1.28	0.69
	1952	1.54	1.20	0.55
	1985	1.34	1.19	0.63
	2003	1.47	1.22	0.65
	2006	1.39	1.23	0.64
	2007	1.40	1.20	0.64
	2008	1.32	1.14	0.59

Table A3.

Seagrass Landscape-Level Metrics – Aggregation

Island	Year	LSI	PD/1000	ENN_AM
Cat				
	2003	72.0	15.5	5.9
	2006	78.9	16.0	6.4
	2007	173.9	32.3	6.3
West Ship				
	1963	49.0	13.4	8.5
	1975	34.4	50.4	11.2
	2003	4.7	142.2	58.2
	2006	19.3	36.4	10.3
	2007	27.8	36.9	11.1
	2008	29.7	20.8	9.6
East Ship				
	1963	76.8	20.4	7.3
	1975	27.7	42.6	12.7
	2003	58.2	12.2	6.1
	2006	45.7	10.1	6.4
	2007	63.4	15.4	6.3
	2008	57.1	12.3	7.1
Horn				
	1940	167.7	24.4	5.6
	1952	134.6	19.0	6.0
	1971	56.5	14.9	8.2
	2003	104.9	11.0	5.9
	2006	84.6	5.2	5.5
	2007	100.6	18.5	6.5
	2008*	67.4	21.9	8.1
Petit Bois				
	1940	83.2	9.4	6.0
	1952	104.8	50.1	7.3
	1985	56.2	12.5	8.2
	2003	58.5	25.7	8.4
	2006	65.1	14.8	7.2
	2007	69.7	19.2	7.8
	2008	50.0	25.9	12.4

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