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Historical changes in seagrass coverage on the Mississippi barrier islands, northern Gulf of Mexico, determined from vertical aerial imagery (1940–2007)

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Vertical aerial image data were used with an edge-detection procedure and visual image interpretation to determine yearly to decadal changes in seagrass (predominantly *Halodule wrightii* Ascherson) coverage on the Mississippi barrier islands. On Horn Island, seagrass coverage declined from 77 ha in 1940 to 19 ha in 1971, but returned to its 1940 value by 2006. Coverage on Petit Bois declined from 54 ha in 1940 to 8–19 ha from 1952 through 2007. On East Ship, seagrass coverage remained at 16–19 ha from 1963 to 2007. On West Ship, coverage dropped to zero in 2003, but by 2007 it had increased to slightly exceed its 1975 value of 18 ha. On Cat Island, coverage increased from 22 ha in 2003 to 71 ha in 2007. There was no apparent negative impact of Hurricane Camille or Hurricane Katrina on seagrass coverage, which could vary annually by a factor of two or more.

Keywords: seagrass; remote sensing; barrier island; coastal system

1. Introduction

Seagrass beds provide essential habitat for a wide variety of aquatic species (Hughes *et al*. 2009), stabilize and enrich sediments, dissipate turbulence, reduce current flow, cycle nutrients, improve water clarity (Hemminga and Duarte 2000) and sequester carbon (Duarte *et al*. 2005). Worldwide, seagrass beds are in decline (Hemminga and Duarte 2000, Green and Short 2003) and the rate of loss is accelerating (Waycott *et al*. 2009). These declines have been attributed to a variety of indirect and direct causes including reduction in water clarity, alteration of sediment migration via dredging, direct destruction from coastal engineering, boating and commercial fishing (Green and Short 2003, Orth *et al*. 2006, Waycott *et al*. 2009). Mapping seagrass beds at high spatial resolution over time is important for distinguishing the effects of anthropogenic impacts and major natural disturbances from variation in seagrass coverage in the absence of perturbation (Kirkman 1996, Kendrick *et al*. 2000, Pasqualini *et al*. 2001, Cunha *et al*. 2005, Dekker *et al*. 2005).

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Gulf of Mexico seagrass beds represent more than 50% of total seagrass coverage in the US (Green and Short 2003). In the Mississippi Sound (Figure 1), most seagrass beds are located along the northern shores of the Mississippi barrier islands (Moncreiff 2007), a portion of the barrier island chain which extends along the US Atlantic and Gulf of Mexico coasts (Pilkey 2003). From west to east, the Mississippi islands include Cat, West and East Ship, Horn and Petit Bois (Figure 1). Since 1971, each of these islands except Cat has been largely protected, including a 1.6 km marine habitat buffer zone, under stewardship of the Gulf Islands National Seashore, US National Park Service. However, prior to 1995, the island seagrass beds were subjected to disturbance by shrimp trawling. Furthermore, each of the islands has been impacted by numerous hurricanes, including severe impacts by Hurricane Camille (1969) and Hurricane Katrina (2005), and decreased steadily in land area since they were first mapped accurately ca. 1850 (Morton 2008, Otvos and Carter 2008).

Comparisons of earlier seagrass surveys (Eleuterius and Miller 1976) with more recent studies (Moncreiff *et al.* 1998, Foster 2005, Peneva *et al.* 2008) have raised concern that seagrass coverage on the Mississippi barriers may be declining. Eleuterius and Miller (1976) reported that in 1969, seagrass habitat on Horn Island totaled 2253 ha. Moncreiff *et al.* (1998) reported a 1992 seagrass coverage of 215 ha, while Foster (2005) and Peneva *et al.* (2008) reported 2003 coverage of approximately 100 ha. However, the extent to which this apparent decline might be explained by methodological differences among studies versus actual change in

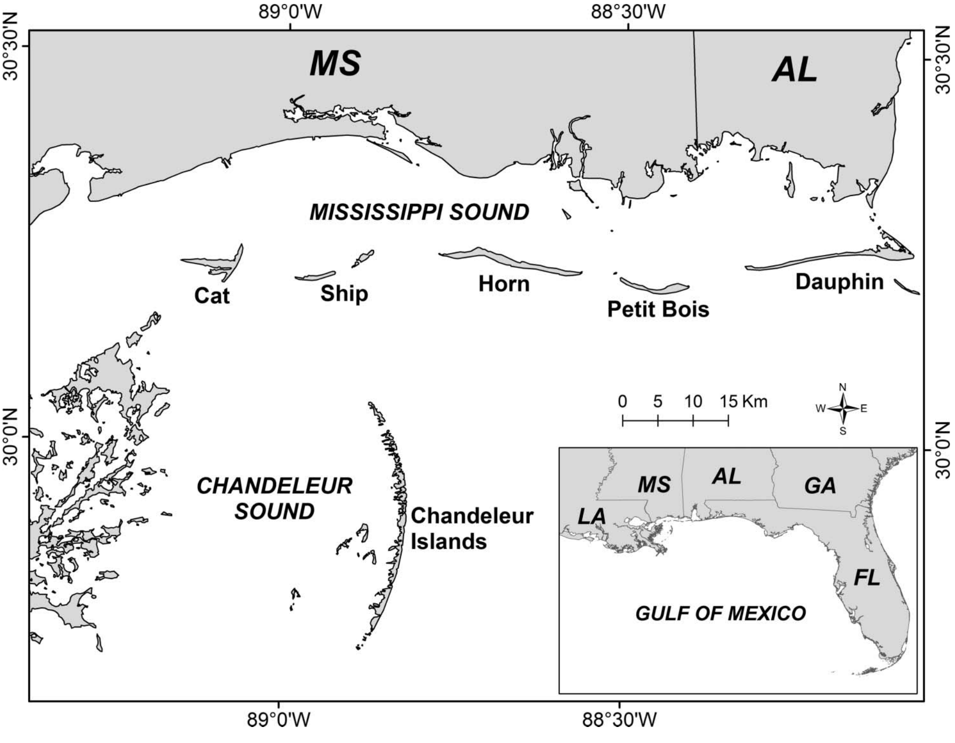

Figure 1. Location of the Mississippi barrier islands, northern Gulf of Mexico.

seagrass coverage has been unknown. Thus, the objective of this study was to analyse remotely-sensed data to quantitatively determine seagrass (predominantly *Halodule wrightii* Ascherson) coverage throughout a period of several decades for each of the Mississippi islands.

2. Methods

Analyses were limited to vertical (camera view angle approximately perpendicular to the ground) aerial image data. All but one dataset was acquired within the period of 6 October–16 November, when seagrass canopies in the Mississippi Sound are typically well-developed prior to late-autumn and winter decline, and water turbidity is relatively low owing to seasonally reduced rainfall. This allowed for the identification of seagrass through the water column. The earliest dates of available imagery which met these criteria were 1940 for Horn and Petit Bois Islands, 1963 for West and East Ship Islands and 2003 for Cat Island (Table 1). The only exception to the use of autumn data was the inclusion of 7 April 1952 imagery of Horn and Petit Bois. These images indicated seagrass beds clearly and served to fill a data void in the 1950s. Image frames acquired by film photography (panchromatic black-and-white in 1940, 1952 and 1963; red (R), green (G) plus blue (B) natural colour (NC) in 1971 and 2003; and R, G plus near-infrared (NIR) colour infrared (CIR) in 1975 and 1985) were scanned (digitized) individually, then georeferenced and projected using 2004 QuickBird imagery (Digital Globe, Inc., Longmont, CO). The QuickBird data previously had been orthorectified to 2004 LIDAR data (Joint Airborne Lidar Bathymetry Technical Center of Expertise, or JALBTCX, Kiln, MS) using ENVI® v. 4.3 (ITT Visual Information Systems, Boulder, CO, USA) (average root mean square error = 1.1 m). The most recent image data were acquired in 2006 and 2007 by a digital multispectral camera system (model DT4100, Duncan Tech, Auburn, CA) and a hyperspectral imaging system (CASI 1500, ITRES Research Ltd., Calgary, Alberta, Canada; data provided by JALBTCX), respectively. The multispectral data also were georeferenced to the QuickBird data, whereas the CASI data

Table 1. Characteristics of the image data used to determine seagrass areal coverage.

Date	Source	Islands covered	Image type	GSD (m)
27 October 1940	USGS	Horn, Petit Bois	Scanned BW	1
7 April 1952	USGS	Horn, Petit Bois	Scanned BW	1
10 October 1963	USCGS	Ship	Scanned BW	1.5
10 November 1971	USGS	Horn	Scanned NC	2
21 October 1975	USGS	Ship	Scanned CIR	0.5
6 October 1985	USGS	Petit Bois	Scanned CIR	1
15 October 2003	USM/GCGC	Cat, Ship, Horn, Petit Bois	Scanned NC	1
29 October 2006	USM/GCGC	Cat, Ship, Horn, Petit Bois	Multispectral (520–590 nm)	1
10–16 November 2007	USACE JALBTCX	Cat, Ship, Horn, Petit Bois	Hyperspectral (570 ± 9 nm)	1–2

Note: Date of image acquisition (flyover) is given in day/month/year. Abbreviations: BW, black-and-white; CIR, colour-infrared; GCGC, Gulf Coast Geospatial Center; GSD, ground spatial distance; NC, natural colour; USACE, US Army Corps of Engineers; USCGS, US Coast and Geodetic Survey; USGS, US Geological Survey; USM, University of Southern Mississippi.

had been georectified earlier by the JALBTCX based on geopositioning from their onboard LIDAR (SHOALS) system. Each island image set was mosaicked as necessary using brightness balancing to minimize across-track illumination gradient effects.

Only green-band data from NC, CIR, digital multispectral (520–590 nm band) or hyperspectral data (570 ± 9.4 nm band) were selected for use in seagrass mapping and aerial coverage estimation. The use of green-band data was based on earlier results from the same waters (Horn Island) which demonstrated that seagrass spectral reflectance, determined from airborne imaging spectrometer data, differed most from background reflectance in the 570–600 nm range (Peneva *et al.* 2008). Seagrass beds were readily visible in the earlier panchromatic or more recent green-band black-and-white images, contrasting against white sand bottom (Figure 2). A

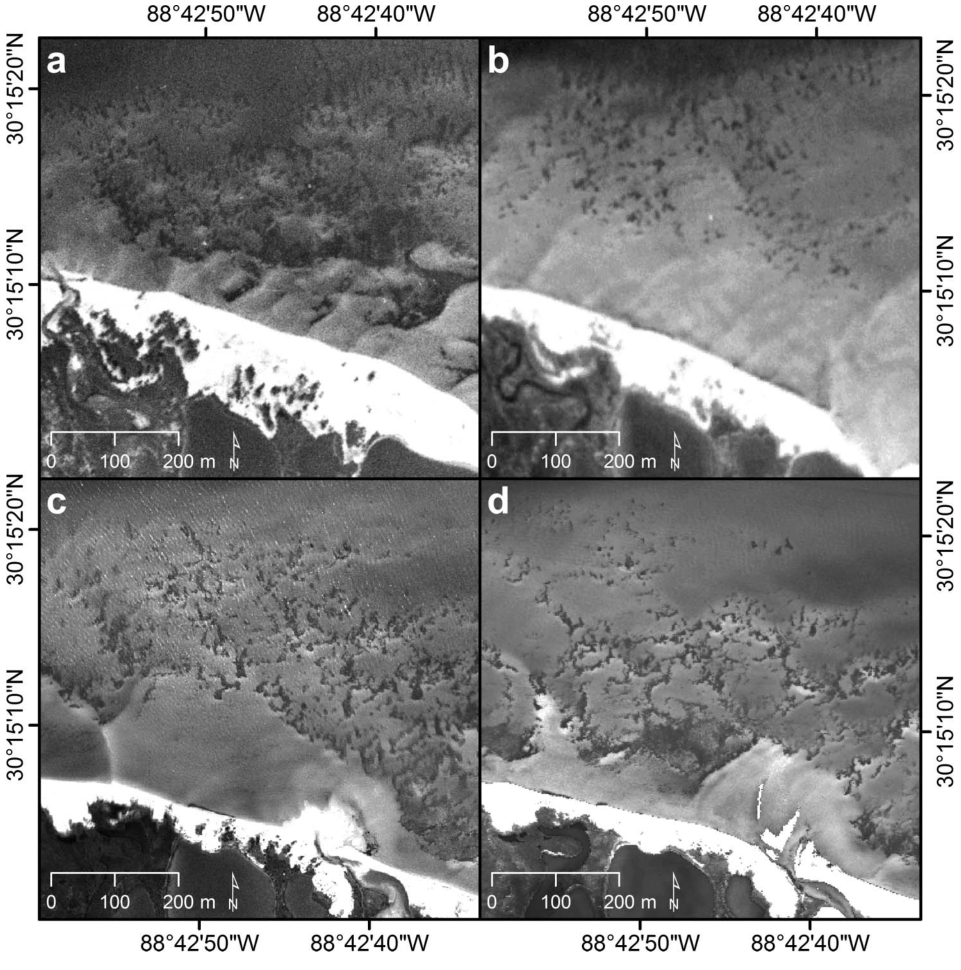

Figure 2. Vertical aerial imagery acquired over a portion of Horn Island, Mississippi, USA, in (a) 1940, (b) 1952, (c) 2003 and (d) 2007 (see Table 1). Gray-scale image data from a variety of sources were used in the study, including digitized panchromatic black-and-white (BW) film photographs (e.g. 1940, 1952), green-band data from digitized natural colour (NC) film photographs (e.g. 2003) and data from the hyperspectral CASI band centered at 570 nm wavelength (2007).

description of how apparent bottom reflectance varies with depth in these waters was also provided earlier (Peneva *et al.* 2008).

The seagrass classification method (Figure 3) was based loosely on image segmentation methodology using a pixel edge solution as described earlier (Urbanski 2006), but simplified and adapted for ENVI®. For imagery of 1 m resolution, a low-pass convolution filter was applied to the data using a kernel size of 29 × 29 pixels. Kernel size was adjusted for other pixel resolutions to approximate a moving window near 30 × 30 m (i.e. 15 × 15 pixel kernel size for 2 m resolution imagery, etc.). Multiple trials used other kernel sizes, but the 30 m × 30 m size yielded the best result for extracting boundaries of the majority of seagrass beds as a consequence of the predominant seagrass patch size. Band math was applied using

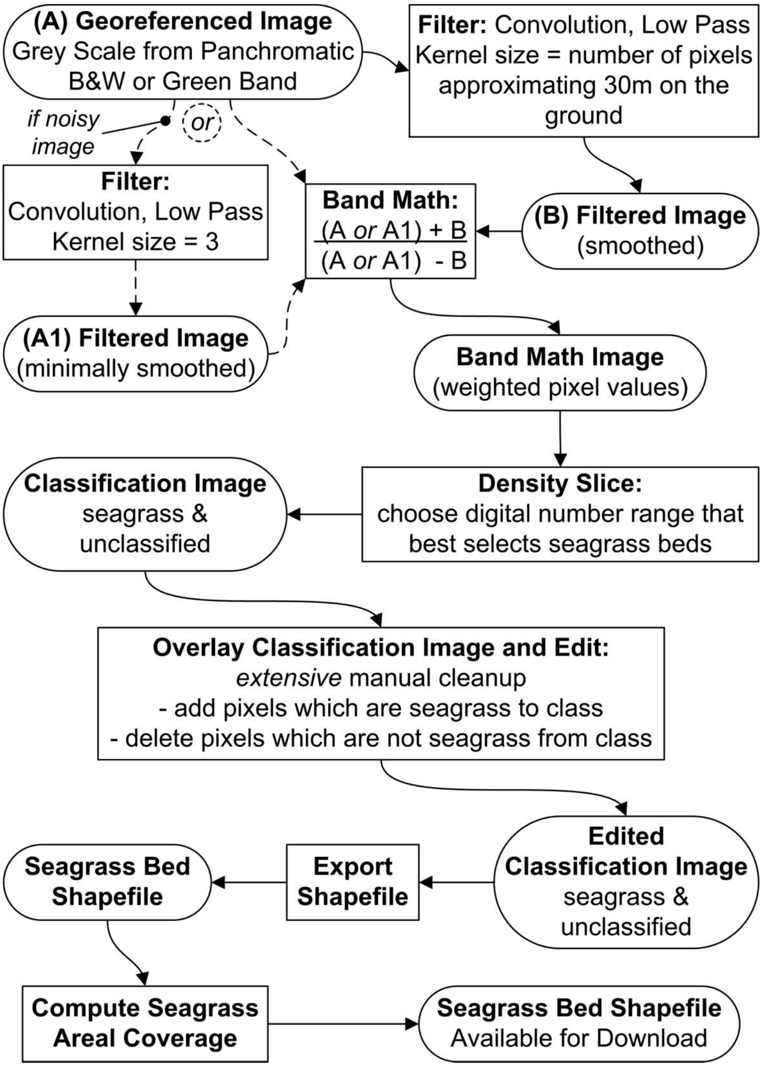

Figure 3. Summary of the analytical procedure used to produce seagrass map files and determine seagrass areal coverage.

the expression: (original image data plus low-pass filtered data) divided by (original image data minus low-pass filtered data). In cases where the original digital image data were noisy (grainy), a low-pass-filtered image derived from applying a 3 × 3 pixel kernel was used in place of the original image.

A density slice was applied to the band-math output image, and a histogram stretch was applied to best contrast the seagrass beds against white sand bottom. The data range was then output to a class image (seagrass and unclassified). These automated procedures were accomplished relatively quickly for each dataset. However, the classified image required extensive manual editing to remove pixels that were erroneously classified as seagrass, and to add omitted seagrass pixels to the seagrass class. Furthermore, only the edges of large beds were differentiated by the automated procedure. The interiors of large beds were added manually. Thus, manual editing based on visual image interpretation constituted approximately 80% of total processing time. Upon completion of these manual corrections, a shapefile was created and seagrass areas were computed. Total seagrass areal coverage for an island in a given year was determined by the sum of areas within polygons encompassing each seagrass bed seen in the image.

3. Results

On Horn, the largest Mississippi island, seagrass coverage declined from 77 ha in 1940 to 19 ha in 1971 (Table 2 and Figure 4). Coverage had returned to its 1952 value of 46 ha by 2003 and to its 1940 value by 2006, but declined again in 2007, to somewhat below the 1952 value. Seagrass coverage on Petit Bois, the most rapidly-migrating island (Morton 2008, Otvos and Carter 2008), declined from 54 ha in 1940 to an approximately stable 8–19 ha from 1952 through 2007. On the smallest island, East Ship, seagrass coverage remained nearly constant at 16–19 ha from 1963 to 2007 despite dramatic reductions in its land area during Hurricanes Camille (1969) and Katrina (2005) (Morton 2008, Otvos and Carter 2008). On West Ship, coverage dropped to zero in 2003 but increased by 2007 to slightly exceed its 1975 value of 18 ha. Data for Cat Island extended only from 2003 to 2007, when coverage more than tripled from 22 ha to 71 ha. The seagrass maps, as shape files, may be downloaded from the USM/GCGC web portal at http://www.gcgcusm.org/sup/geocarto/TGEI-2011-0040. It should be emphasized that the 1952 data were acquired in spring (7 April), whereas all other image data used in the study were

Table 2. Seagrass coverage on the Mississippi barrier islands, 1940–2007. Sampling error was estimated to be ± 5% of the given area value.

Date	Cat	West Ship	East Ship	Horn	Petit Bois
27 October 1940	–	–	–	76.7	54.1
7 April 1952	–	–	–	45.7	15.3
10 October 1963	–	11.1	19.4	–	–
10 November 1971	–	–	–	19.3	–
21 October 1975	–	1.8	1.7	–	–
6 October 1985	–	–	–	–	17.7
15 October 2003	21.8	0.0	16.5	50.8	8.0
29 October 2006	25.5	0.9	15.5	82.0	18.9
10–16 November 2007	71.3	1.9	13.7	38.1	16.8

Note: Date of image acquisition (flyover) is given in day/month/year.

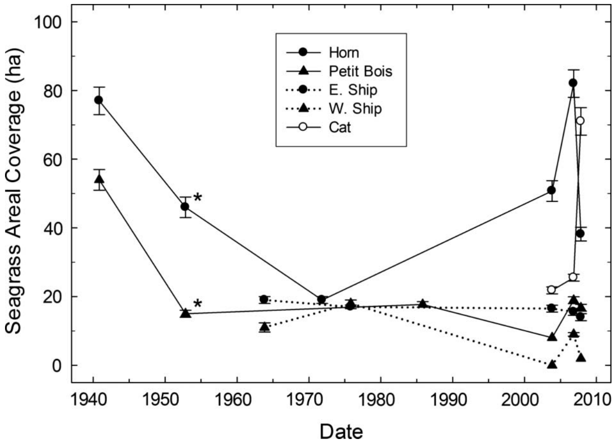

Figure 4. Seagrass coverage on the Mississippi barrier islands, 1940–2007. Sampling error (bars) was estimated to be ±5% of the given area value. Asterisks denote that the 1952 data were acquired in spring (7 April). All other image data used in the study were acquired in autumn (6 October to 16 November).

acquired in autumn (6 October–16 November) (Table 1). Thus, it is possible that seagrass areal coverage for 1952 reported herein is less than it might have been during autumn of that year.

4. Discussion and conclusions

Comparison of 2006 versus 2007 data for Horn and Cat indicated that seagrass coverage can vary annually by at least a factor of two. There was no difference in 2006 versus 2007 data quality which might have explained these dramatic changes. Thus, more historical data would be needed to develop strong conclusions regarding the primary factors which have influenced this variation. For example, protection from the destructive impacts of trawling might be expected to promote an increase in seagrass coverage (Watling and Norse 1998, Dellapenna *et al.* 2006). However, recreational boat operations continue in the shallow waters surrounding each island, and can negatively impact seagrass plants and habitats (Short and Wyllie-Echeverria 1996, Kirsch *et al.* 2005, Larkum *et al.* 2006). Even small motor boats can negatively impact zooplankton, potentially altering food-web structure and function (Bishop 2008, Bickel *et al.* 2011). Similar indirect effects have been reported where over-fishing in kelp forests and coral reefs negatively impacted primary producers (Jackson *et al.* 2001, Heck and Valentine 2006). Fishing the higher trophic levels that feed on primary consumers, or direct harvesting of detritivores (mullet and shrimp), can result in increased proliferation of algae, either as epiphytes growing on seagrass blades, or drifting macroalgal mats (Hughes *et al.* 2004). Both types of algal overgrowth reduce light availability, resulting in reduced seagrass health and abundance (Hauxwell *et al.* 1998, 2001). Furthermore, nearby dredging for channel maintenance or expansion would be expected to negatively affect seagrass, via light-limitation stress caused by increased turbidity (Badalamenti *et al.* 2006, Erftemeijer and Lewis 2006).

Much of the seagrass habitat in the US is found along lagoonal sides of barrier islands (Biber, unpublished), yet few studies have linked changes in seagrass extent with island loss or migration. However, protection from waves (Koch 2001, Koch *et al.* 2006) and currents (Fonseca 1996, Fonseca and Bell 1998, Fonseca *et al.* 2007) is important to the long-term survival of seagrass. Seagrass decline on Petit Bois in particular may be at least in part a result of its rapid westward migration (Morton 2008, Otvos and Carter 2008) and nearly 40% reduction in land area from 1940 to 2007.

There was no discernable negative impact of Hurricane Camille (category 5, August 1969), or Hurricane Katrina (category 3–4, August 2005) on seagrass coverage (Figure 4). Eleuterius (1971) reported that Hurricane Camille denuded about 30% of Mississippi Sound seagrass habitats, most of which were along the northern shores of the barrier islands. However, the author noted that in 1970, most denuded areas showed signs of reestablishment, while areas that had been more protected from the storm exhibited more vigorous growth. Heck and Byron (2006) reported that Hurricane Katrina did not damage most seagrass beds on the Mississippi barrier islands. Similarly, hurricane impacts to seagrass in the Caribbean showed relatively minor impacts despite a 258 km h^{-1} wind-field (Michot *et al.* 2002). Thus, present results support the conclusion that seagrass habitats are remarkably resilient to even severe tropical storms.

In the present study, seagrass patches were digitized at a resolution of 1–2 m ground spatial distance (GSD), allowing an assessment of individual seagrass beds. Owing to technological limitations, some previous seagrass mapping efforts in the Mississippi Sound relied on a relatively coarse definition of seagrass coverage, referred to as potential seagrass habitat (PSH), to encompass all areas where seagrass would likely occur. PSH boundaries closely followed the 1.3 m depth contour, largely because the turbid Sound waters limit light penetration and restrict seagrasses to shallow depths (Moncreiff *et al.* 1998, Moncreiff 2007). Not surprisingly, the area of PSH is much greater than the area of vegetated seafloor. Thus, differences in mapping methods must be considered carefully when comparing seagrass coverages among studies. In the present study, the application of a consistent mapping method to best-available imagery resulted in a very different interpretation of seagrass change over the last approximately seven decades than in previous studies. Also, more detailed change analysis using landscape statistics (McGarigal and Marks 1995) can be used to better understand not only change in total area, but also how configuration of seagrass patches may have been affected over time. For example, a decrease in mean patch size results in more edge habitat at the expense of interior habitat. Previous research has highlighted the importance of such fragmentation to seagrass landscape structure (Uhrin and Holmquist 2003) and function (Irlandi 1996, 1997), and how this affects faunal composition and abundance of commercially important fisheries (Irlandi *et al.* 1999, Johnson and Heck 2006). Seagrass maps produced in this study (http://www.gcgcusm.org/sup/geocarto/TGEI-2011-0040) can now be used in future research to better understand the temporal and spatial dynamics of change in seagrass coverage on the Mississippi islands.

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