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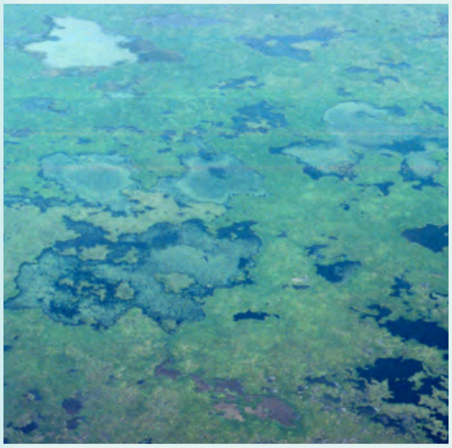


Scientific Investigations Report 2006-5287

U.S. Environmental Protection Agency 855-R-04-003

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Cover: Background, seagrass through surface of water (Tommy Michot, U.S. Geological Survey). Left, seagrass sampling (Tommy Michot, U.S. Geological Survey). Center, oblique aerial photograph of seagrass (Tommy Michot, U.S. Geological Survey). Right, manatees in seagrass habitat (Sirenia Project, U.S. Geological Survey).

Seagrass Status and Trends in the Northern Gulf of Mexico: 1940–2002

Edited by L. Handley, D. Altsman, and R. DeMay

Scientific Investigations Report 2006–5287

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Suggested citation:
Handley, L., Altsman, D., and DeMay, R., eds., 2007, Seagrass Status and Trends in the Northern Gulf of Mexico: 1940-2002: U.S. Geological Survey Scientific Investigations Report 2006-5287 and U.S. Environmental Protection Agency 855-R-04-003, 267 p.

Preface

The Gulf of Mexico Program (GMP) is a network of citizens dedicated to promoting the economic health of the region by managing and protecting the gulf's resources. Although administered by the U.S. Environmental Protection Agency (EPA), the GMP engages many organizations across the Gulf of Mexico coastal region to implement and lead restoration and conservation projects that are environmentally and economically sound. The GMP includes representatives from State and Federal agencies, nonprofit organizations, the scientific community, business and industry, and an organized citizens group. These members are appointed by the five State governors along the gulf coast. The GMP focuses on three ecological issues: (1) public health, (2) excess nutrient enrichment, and (3) habitat degradation and loss, including the introduction of nonindigenous species.

The GMP has long recognized that seagrasses, estuaries, and coastal wetlands are vital in providing food and shelter for plants and animals, improving water quality, sediment filtration, and flood and erosion control. In 1999, the GMP's Habitat Focus Team set a goal to restore, enhance, or protect 20,000 acres of important coastal habitats of the gulf by 2009. The Habitat Team, recognizing that seagrass beds are some of the most productive habitats in nearshore waters, set a goal to produce this summary report on gulfwide seagrass status and trends. The purpose of the summary report is to provide current baseline information on the status of seagrasses in the Gulf of Mexico.

To produce this report, the GMP's Habitat Team formed a subcommittee, consisting of over 30 gulf coast seagrass scientists and environmental managers. Committee members provided data on seagrass maps, seagrass status and trends, causes of change in seagrass acreage, monitoring activities, and restoration efforts important to seagrass areas along the gulf coast. The U.S. Geological Survey's (USGS) National Wetlands Research Center provided extensive support in the production of data, maps, writing, and editing for this summary.

Lastly, *Seagrass Habitat in the Northern Gulf of Mexico: Degradation, Conservation, and Restoration of a Valuable Resource*, written in lay terms and developed for the general public, legislators, and Gulf of Mexico stakeholders, is meant to be a companion to this summary report. For example, nomenclature of seagrasses in the summary report follows this outreach document. Additional information will be available on the USGS National Wetlands Research Center's Web site at ***http://www.nwrc.usgs.gov*** and the GMP Web site at ***http://www.epa.gov/gmpo/***.

Acknowledgments

The U.S. Geological Survey, National Wetlands Research Center, and the U.S. Environmental Protection Agency, Gulf of Mexico Program partnership would like to thank the following scientists and environmental managers for reviewing the Gulf of Mexico seagrass status and trends. Your dedication, time, and expertise are greatly appreciated.

Ron Boustany, U.S. Department of Agriculture, Natural Resources Conservation Service

Dr. Ken Dunton, University of Texas

Dr. Steve Jordan, U.S. Environmental Protection Agency

Dr. Michael Lewis, U.S. Environmental Protection Agency

Randy Roach, U.S. Fish and Wildlife Service

Frank Sargent, Florida Department of Environmental Protection, Southwest Florida Water Management District

Dr. Judy Stout, University of South Alabama

Dr. Sandy Wyllie-Echeverria, University of Washington

Dr. Judy Stout, University of South Alabama

Captain David Yeager, Mobile Bay National Estuary Program

Personnel from the USGS National Wetlands Research Center who played a key role in the production of this document included

Pete Bourgeois, cartography

Helena Schaefer, peer review coordination

Blaire Hutchison, data compilation

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Seagrass Status and Trends in the Northern Gulf of Mexico: 1940–2002

Edited by L. Handley,[1] D. Altsman,[2] and R. DeMay[3]

Abstract

Over the past century, seagrass habitats from the bays of Texas to the gulf shores of Florida have decreased. Seagrass beds, which are highly dependent on water quality and clarity for survival, are home to a multitude of aquatic plants and animals and a source of economic activity through commercial and recreational fishing and ecotourism. The U.S. Environmental Protection Agency's Gulf of Mexico Program (GMP) and its partners have made a commitment to restore, enhance, and protect this important ecosystem. As seagrass habitats decrease, the need for information on the causes and effects of seagrass loss, current mapping information, and education on the importance of seagrassess becomes greater. This report is the initial effort of the GMP's research and restoration plan for seagrasses. The purpose of this report is to provide scientists, managers, and citizens with valuable baseline information on the status and trends of seagrasses in coastal waters of the Gulf of Mexico. Within the northern Gulf of Mexico region, 14 individual estuarine systems where seagrasses occur, as well as statewide summaries for Texas, Louisiana, Mississippi, Alabama, and Florida, are examined in this study. Each estuarine system is detailed in vignettes that address current and historical extent and quality of seagrasses, seagrass mapping and monitoring, causes of status change, restoration and enhancement activities, background information for the entire study area as well as the subareas for study, and the methodology employed to analyze and document the historical trends and current status of seagrasses.

The systems, moving from west to east, include the Laguna Madre, Texas Coastal Bend region, and Galveston Bay in Texas; the Chandeleur Islands in Louisiana; the Mississippi Sound; and Perdido Bay, Pensacola/Escambia Bay, Choctawhatchee Bay, St. Andrew Bay, Florida's Big Bend region, Tampa Bay/St. Joseph Sound, Sarasota Bay, Greater Charlotte Harbor, and Florida Bay in Florida. (Mobile Bay is dealt with only in the statewide summary for Alabama.)

Introduction

The Gulf of Mexico provides a wide array of valuable natural resources to the nations that border its shores. As the value of the gulf coastal environment continues to be recognized, it becomes increasingly important to invest in the conservation of those resources. Reductions in both abundance and diversity of various organisms and habitats emphasize a critical need to protect these natural assets, many of which serve important ecological functions. In response to increasing trends in habitat degradation, several organizations and institutions have begun to act together with local residents to address these issues. One such effort, facilitated by the U.S. Environmental Protection Agency's (EPA) Gulf of Mexico Program, will integrate the efforts of a wide range of scientific experts from Federal, State, and local partners to increase the environmental quality of the habitats of the northern Gulf of Mexico.

In 1999, the Gulf of Mexico Program committed to restoring, enhancing, and protecting 20,000 acres of important coastal habitats within the northern Gulf of Mexico region by the year 2009. The northern Gulf of Mexico region is defined as those waters lying adjacent to the States of Texas, Louisiana, Mississippi, Alabama, and Florida. This region encompasses over 2,414 km (1,500 miles) of coastline and is home to more than 25 million residents. Marine ecosystems such as sandy beaches, salt marshes, mangroves, coral reefs, and seagrass beds combine to create important coastal habitats that allow this region to flourish. Modern land development practices in the northern gulf, however, threaten these aquatic habitats, risking the economic values that form the foundation of these coastal communities.

Seagrasses have been particularly impacted by the degradation of coastal waters in the northern Gulf of Mexico. Over the past century, seagrass beds, which are highly dependent on water quality for survival, have decreased from the bays of Texas to the shores of Florida.

[1] U.S. Geological Survey, National Wetlands Research Center.

[2] U.S. Environmental Protection Agency, Gulf of Mexico Program.

[3] Barataria-Terrebonne National Estuary Program.

Seagrasses are submerged flowering plants that grow in bays, lagoons, and shallow coastal waters. Because seagrasses require light for photosynthesis, water clarity is very important for their survival. They anchor themselves to the seafloor with strong root systems that allow them to withstand strong currents and waves. Seagrasses differ from terrestrial vegetation in that they must reproduce underwater and accomplish reproduction with filamentous pollen grains that can be transported by water currents.

Continuous expanses of seagrass beds provide valuable habitat by stabilizing coastal sediments, decreasing wave energy, and providing shelter for a variety of aquatic organisms. A multitude of plants and animals inhabit seagrass ecosystems, forming complex food webs that link a variety of species together. Large species, such as Florida manatees (*Trichechus manatus*), green sea turtles (*Chelonia mydas*), and bottlenose dolphins (*Tursiops truncatus*), are found in seagrass beds throughout the northern Gulf of Mexico region and use this habitat for feeding, either eating the seagrass directly or capturing smaller species drawn to the ecosystem.

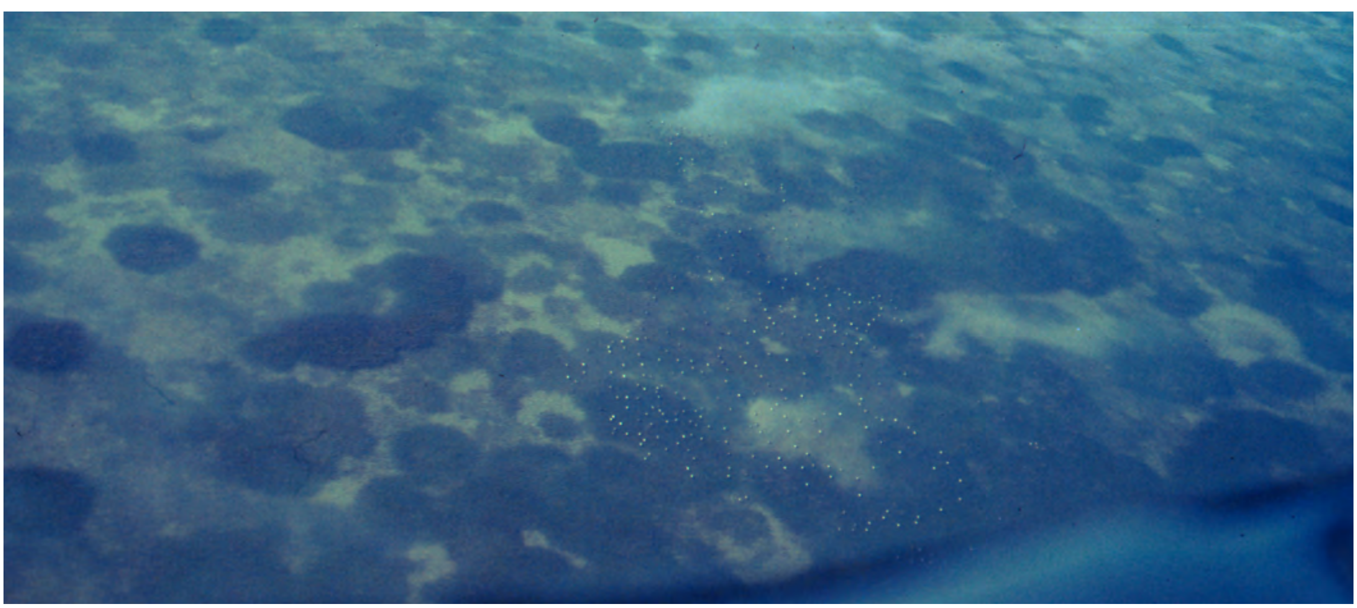
Finfish species that use seagrass habitat include drum (Sciaenidae), sea bass (Serranidae), porgy (Sparidae), grunt (Haemulidae), and snapper (Lutjanidae). Larger fish use seagrass habitat as nursery areas, while smaller species use the cover provided by seagrass as protection from predators.

Shellfish that populate seagrass habitat include the Caribbean spiny lobster (*Panulirus argus*), queen conch (*Strombus gigas*), West Indian sea star (*Oreaster reticulata*), pink shrimp (*Penaeus duorarum*), and eastern oyster (*Crassostrea virginica*). Most of these species live at the base of seagrass plants, while others, such as the spiny lobster, are drawn from nearby locations to feed at night.

Birds that use seagrass habitat include wading and diving birds, such as common merganser (*Mergus merganser*), common loon (*Gavia immer*), great cormorant (*Phalacrocorax carbo*), brown pelican (*Pelecanus occidentalis*), and redhead duck (*Aythya americana)*. Most of these birds use seagrass habitat as regular feeding grounds, consuming seagrass blades and rhizomes or fish swimming among the grasses. Other species temporarily inhabit the areas during their seasonal migrations along the coast.

There are 58 species of seagrasses found around the world, consisting of two main families: Potamogetonaceae and Hydrocharitaceae (Phillips and Menez, 1988; Kuo and McComb, 1989). Short and Coles (2001) stated, however, that recent work established that there were 60 seagrass species in 4 families and 12 genera. Six distinct species of seagrasses have been identified in the bays, lagoons, and shallow coastal waters of the northern gulf region. These species include paddle grass (*Halophila decipiens*), star grass (*Halophila engelmannii*), turtle grass (*Thalassia testudinum*), shoal grass (*Halodule wrightii*), manatee grass (*Syringodium filiforme*), and one freshwater species, wigeon grass (*Ruppia maritima)*, which is also capable of tolerating saline waters (Phillips and Menez, 1988; Fonseca, 1994). Wigeon grass and water celery (*Vallisneria americana*) are two salt-tolerant freshwater plants and have been included in the analyses of a few of the local areas along the Gulf of Mexico. The reason for their inclusion in these areas, which will be further identified in the State and local chapters, is that in early assessments, there was no differentiation between species, and differentiation in zones where fresh water and salt water mix was not possible from aerial photography (e.g., *Halodule* vs. *Ruppia*).

Seagrass habitats support important recreational and commercial fisheries which form the economic foundation of many northern Gulf of Mexico communities. Commercial fishing accounts for over $800 million in annual revenues gulfwide (National Oceanic and Atmospheric Administration, 1997). Recreational fishing is also a very important source of revenue for the northern gulf region, with both in-state and out-of-state anglers contributing to local economies. Species such as bonefish (*Albula vulpes*) and tarpon (*Megalops atlanticus*) support a multimillion dollar recreational fishery that delivers income to charter boats, marinas, hotels,



restaurants, and tackle shops. Tourism is a significant economic attribute of seagrass habitats, with direct benefits accruing from residents and nonresidents who take ecotourism trips to seagrass meadows. Nonresidents spend hundreds of millions of dollars annually at hotels, restaurants, and outdoor outfitting shops along the northern gulf coast. In addition, discarded shells of over 30 species of bivalves are actively collected by tourists and commercial retailers.

While there is no single estimate of the value of seagrasses in the northern Gulf of Mexico region, several attributes demonstrate the importance of the habitat to both the aquatic ecosystem and to the local economy. The State of Texas has attempted to quantify some of these attributes, finding that seagrass habitat produced an economic value of $9,000 to $28,000 per acre in commercial, recreational, and storm protection functions (Texas Parks and Wildlife Department, 1999). In Florida, the Department of Environmental Protection has estimated that each acre of seagrass has an economic value of approximately $20,500 per year, which translates into a statewide economic benefit of $55.4 billion annually (Florida Department of Environmental Protection, 2001).

Because seagrass habitat is so valuable, its degradation is a major concern for communities throughout the northern Gulf of Mexico. It is estimated that over the last 50 years, seagrass habitat losses ranged anywhere from 20% to 100% for most estuaries in the region. Seagrasses are threatened by increased loadings of nutrients into coastal waters, dredging activities, and shoreline development, as well as increased commercial and recreational boating and fishing practices. In 1992, the Gulf of Mexico Program's Habitat Degradation Committee produced a report on the status and trends of emergent and submerged vegetated habitats in the Gulf of Mexico (Duke and Kruczynski, 1992). At that time, the total seagrass coverage in the shallow estuarine and nearshore waters of the northern gulf was estimated to be 1,019,844 ha (2.52 million acres).

In 1999, the Gulf of Mexico Program's Habitat Focus Team and a panel of seagrass experts and researchers developed a comprehensive seagrass research and restoration program. The main objectives of this program are as follows:

- assess the areal and temporal extent and quality of seagrasses;
- determine trends in the extent and quality of seagrass habitats;
- identify factors that determine establishment and persistence of seagrasses; and
- identify the critical factors which determine natural structural and functional characteristics of seagrass habitats.

Many studies need to be conducted to address the objectives listed above. To better assess the areal and temporal extent of seagrasses, current seagrass acreage in the coastal waters must be quantified and mapped. Geological and historical seagrass coverage must also be determined to evaluate quantity and locations of seagrass decline. To help determine the quality of seagrass beds, indicators of seagrass health must be identified on appropriate spatial and temporal scales. Rapid assessment techniques and sample designs to routinely monitor seagrass beds also need to be developed. The water-quality parameters that will protect and preserve seagrasses need to be identified in order to evaluate the factors which determine seagrass establishment and persistence. Stressors (both natural and human-induced) need to be identified and quantified, as well as the cause and effect relationships between potential stressors and seagrasses. Water column and sediment characteristics required to establish seagrasses in areas needing to be restored should also be evaluated.

One of the initial activities proposed to support the research and restoration goals of the Gulf of Mexico Program was a comprehensive review of the current status and trends of seagrasses in the northern Gulf of Mexico. Fourteen estuarine systems experiencing various degrees of seagrass degradation were selected for the study. Scientists and environmental managers researching the seagrass communities in these systems have coordinated their sampling, monitoring, and reporting practices to develop a comprehensive review of historical seagrass coverage, current seagrass coverage, and significant threats to both short-term and long-term seagrass productivity in each of the systems. For this report, the majority of data produced used techniques described in the appendix. The resulting report includes information on the following:

- the Laguna Madre, Coastal Bend region, and Galveston Bay, three estuarine systems located along the Texas coast.
- the Chandeleur Islands, Mississippi Sound, and the Alabama coast in the central northern gulf region.
- Pensacola Bay, Perdido Bay, Choctawhatchee Bay, St. Andrew Bay, and Florida's Big Bend region along the northwest Florida coast.
- Tampa Bay and St. Joseph's Sound, Sarasota Bay, Charlotte Harbor, and Florida Bay along the southwest Florida coast.
- Statewide summaries of seagrass coverage of all five Gulf of Mexico Coast States.

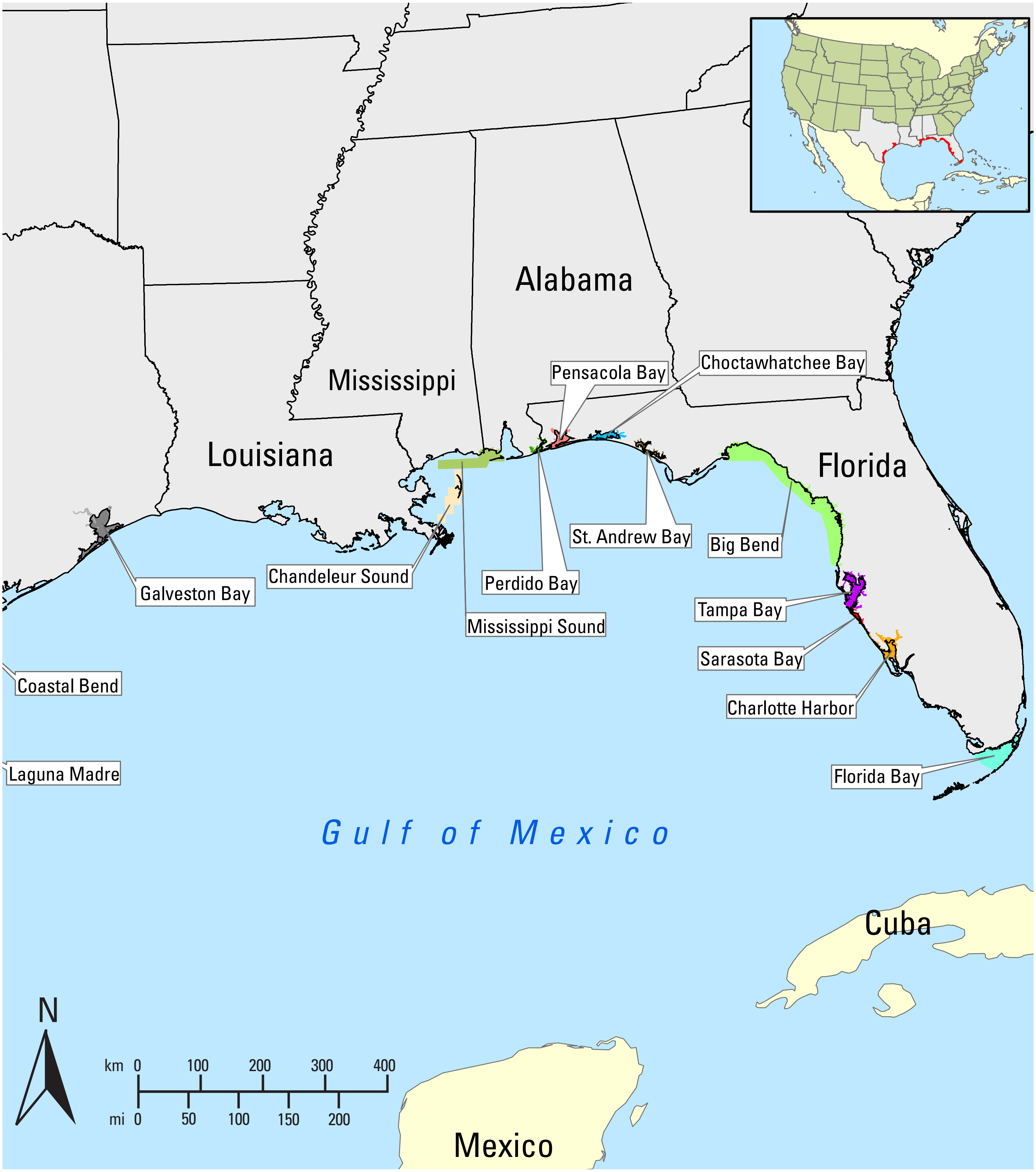
This report's local estuarine systems (see fig. 1) are described in 14 vignettes, each beginning with a cover map that depicts the watershed of each seagrass acreage area. The watershed map also depicts the streams, rivers, and bays that ultimately can influence the health and conservation of the seagrasses in that area. The vignettes primarily discuss the local seagrass acreage, causes and effects of acreage change, and restoration and monitoring ongoing in the area.

All 14 estuarine systems have experienced some declines in seagrass habitat. Laguna Madre has experienced between 10% and 20% loss in seagrass habitat since 1965. Mississippi Sound, home of the Grand Bay National Estuarine Research Reserve, has lost over 4,500 ha (11,120 acres) of seagrass habitat since 1969, leaving only about 750 ha (1,853 acres) total seagrass coverage. In Tampa Bay, Fla., over 6,000 ha (14,826 acres) of seagrass have been lost since the 1950s.

It is important to note that, as scientists learn more about the ecological value of these key habitats and continue efforts to educate and inform the public on the critical nature of seagrass systems, conservation efforts in these areas are yielding positive results. For instance, Florida Bay, home to species such as the manatee, bottlenose dolphin, and loggerhead sea turtle (*Caretta caretta*), gained 5,000 ha (12,355 acres) of seagrass habitat between 1987 and 1994. In addition, the Coastal Bend of Texas, part of EPA's National Estuary Program, has seen an increase of 2,168 ha (5,357 acres) since 1958.

This report presents the initial efforts of the Gulf of Mexico Program and its partners to implement a comprehensive research and restoration plan for seagrasses. The purpose of the review is to provide scientists, managers, and citizens with valuable baseline information on the current status of seagrasses in the coastal waters of the northern Gulf of Mexico. In the following sections of this document, detailed backgrounds on each of the respective geographic areas are presented. Each section includes the areal extent, geology, bathymetry, and key aquatic species inhabiting the area. Additionally, seagrasses species, their abundance and distribution, and methods used to evaluate the historical and current trends are presented. For each system, information on the causes of seagrass habitat degradation is offered, as well as potential restoration opportunities and techniques for improving seagrass health in that specific estuary.

Figure 1. Local estuarine systems described in this report.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Big Bend
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Mississippi Sound
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Sarasota Bay
Charlotte Harbor
Coastal Bend
Laguna Madre
Florida Bay
Gulf of Mexico
Cuba
Mexico
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km 0 100 200 300 400
mi 0 50 100 150 200

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Statewide Summary for Texas

By Warren Pulich, Jr.,[1] and Christopher Onuf[2]

Background

Seagrass-dominated communities (stands of coastal, saltwater-tolerant, submerged vascular vegetation) support higher biodiversity and production than any other biotic community along the Texas coast. These systems provide nursery areas for estuarine fish and wildlife; direct food sources for various fauna, including fishes, waterfowl, and sea turtles; major contributions of organic matter to coastal systems; key functions in nutrient cycling processes; and a stabilizing agent for coastal sedimentation and erosion control (McRoy and McMillan, 1977). Because of the high quality and limited extent of seagrass beds in Texas in the early 1990s (about 88,877 ha, or 219,612 acres), any impact to this important habitat raises concerns of resource managers, coastal scientists, and conservationists.

Seagrasses are very unevenly distributed along the Texas coast in the following systems: Laguna Madre, Texas Coastal Bend Region, and Galveston Bay (fig. 1). A suite of factors, such as precipitation and freshwater discharge, contributes to a sharply increasing gradient in seagrass coverage from northeast to southwest. Average annual precipitation decreases from 120 cm (47 inches) at Galveston Bay to little more than half that at the south end of Laguna Madre, while average annual pan evaporation increases from 180 cm to 200 cm (71 to 79 inches) over the same span of coastline. Relatively large river systems discharge into embayments of the upper Texas coast compared to the middle Texas coast, while the only continuous freshwater inflow into Laguna Madre is agricultural drainage and municipal discharges to Arroyo Colorado, an abandoned distributary of the Rio Grande. The Rio Grande itself discharges directly to the Gulf of Mexico 5 km (3 mi) south of Laguna Madre and has recently gained notoriety because of lack of freshwater inflows reaching the gulf. The consequence of greater precipitation and freshwater inflow to the upper Texas coast is that its bays are subject to greater freshening and receive heavier sediment and nutrient loadings than do the estuaries farther to the south, resulting in higher turbidity and smaller areas of near-marine salinities suitable for seagrass growth. The other essential ingredient for seagrass meadow development is shallow, protected water, most commonly provided by the lagoonal extensions away from the main stem of the bays, behind barrier islands and peninsulas. Thus, Laguna Madre is doubly advantageous as an environment supporting seagrass as it features both limited inflow and a higher proportion of protected shallow water than any other Texas bay, and seagrasses abound accordingly.

Over the past 30 yr or so, there has been a growing recognition of factors affecting seagrass productivity, species distribution, and susceptibility to human disturbance. The population of Texas coastal counties increased 75% between 1970 and 2000, from 2.9 million to 5.2 million, and is projected to increase another 45% by 2030, to 7.5 million (Skrabanek and others, 1985; Texas Water Development Board, 2001). The population is distributed very unevenly, with the upper coast accounting for 81% of the total in 2000, mostly around Galveston Bay; however, the counties of the lower coast, encompassing lower Laguna Madre, experienced the fastest proportionate growth between 1970 and 2000, increasing 129%. Increased commercial and recreational use of the bays, increased and altered inputs of nutrients and contaminants from the watershed, and associated burgeoning coastal populations are stressing coastal natural resources like seagrasses.

Table 1 shows trend analysis of seagrass beds indicating severe loss of grassbeds in several areas (Galveston Bay (Pulich and White, 1991) and lower Laguna Madre (Onuf,1994)) and minor loss as well as fluctuations in others (Redfish Bay (Pulich and others, 1997) and San Antonio Bay (Pulich, 1991)). In recent years, seagrasses have received attention as environmental indicators of estuarine water quality and ecosystem health because of their sensitivity to nutrient enrichment and incipient eutrophication (Dennison and others, 1993; Neckles, 1994; National Oceanic and Atmospheric Administration, Office of Ocean Resources Conservation Assessment, 1995). Various studies have documented significant impacts at regional scales from dredging (Pulich and White, 1991; Onuf, 1994), boating traffic (Sargent and others, 1995; Dunton and Schonberg, 2002), nutrient loading (Orth and Moore, 1983; Cambridge and McComb, 1984; Pulich and White, 1991; Short and Burdick, 1996; Tomasko and others, 1996), subsidence processes (Pulich and White, 1991), and altered freshwater inflow cycles (Eleuterius, 1987; Quammen and Onuf, 1993; Pulich and others, 1997).

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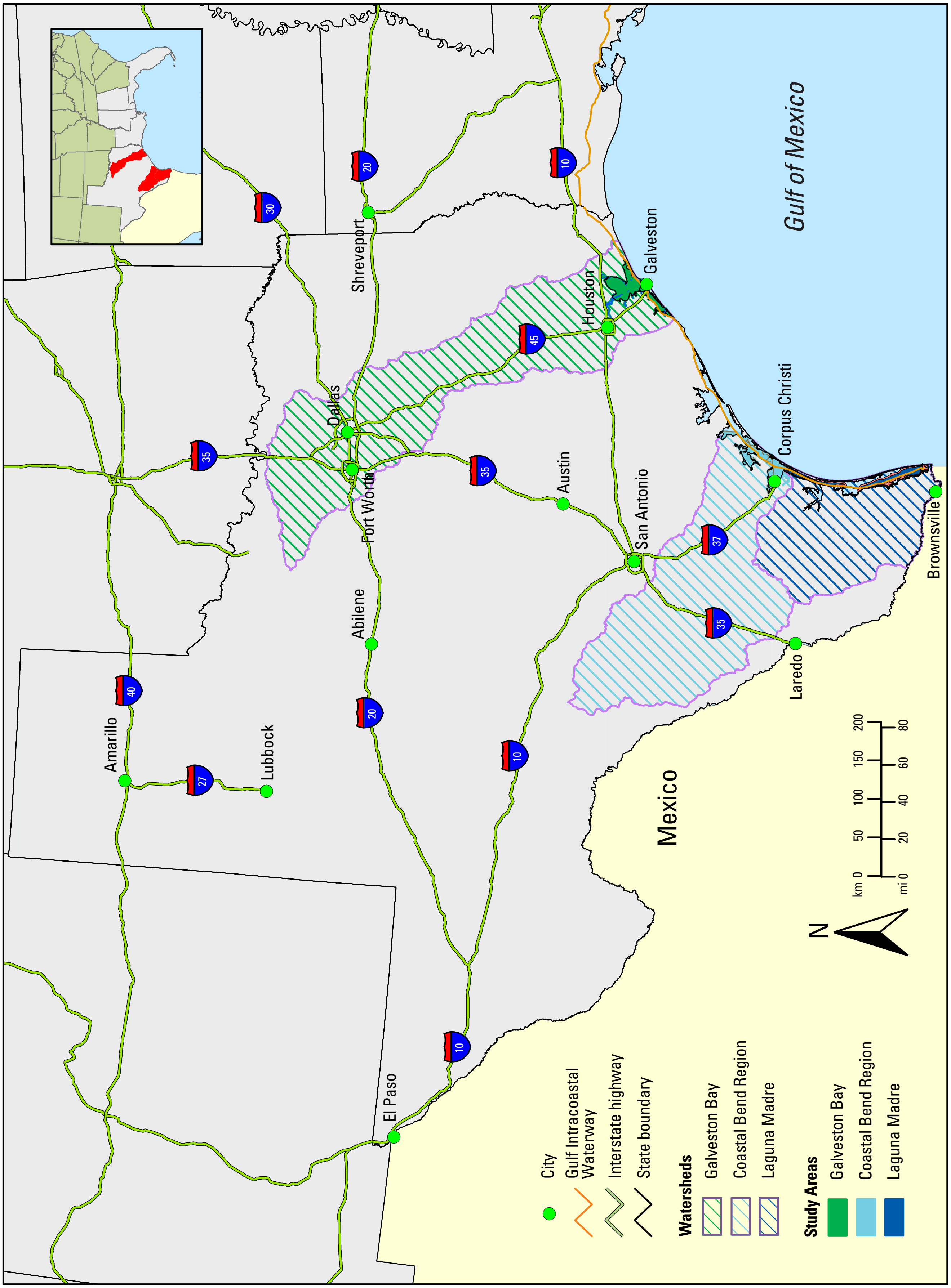

Figure 1. Watershed for the State of Texas.

Statewide Status and Trends Data

In the early 1990s, Galveston and Matagorda Bays on the upper Texas coast had 1,706 ha (4,215 acres) of seagrass meadows total, covering about 1% of bay bottom. The San Antonio, Aransas-Copano, and Corpus Christi Bay systems of the middle Texas coast had an order of magnitude greater of seagrass cover totaling 15,454 ha (38,186 acres) and 12% of bay bottom. Upper and lower Laguna Madre and Baffin Bay of the lower Texas coast supported greater than four times that amount (71,717 ha, or 177,210 acres) of seagrass meadow, covering 50% of bay bottom. In Laguna Madre proper, seagrasses truly define the ecosystem and carpet 65% of the bottom. Status and trends of these three most-investigated Texas coastal areas are presented in vignettes in this report. Summaries of these areas illustrating the range of problems affecting seagrasses coastwide are presented below. Seagrasses in areas not directly covered in the vignettes (i.e., Matagorda/Tres Palacios Bays and San Antonio/Espiritu Santo Bays) appear to be stable or possibly decreasing (i.e., San Antonio Bay system), although mapping data for these systems are rather sparse and rudimentary (Pulich, 1999). Meadows in these latter bay systems totaled 5,855 ha (14,468 acres), approximately 6.6% of the State total (see table 1).

Galveston Bay System

In the Galveston Bay system on the upper Texas coast, practically all seagrass beds disappeared from the main parts of Galveston Bay in the late 1970s. As of 1998, only about 210 ha (518 acres) of mostly shoal grass (*Halodule wrightii*) with some star grass (*Halophila engelmannii*) and 0.6 ha (1.6 acres) of turtle grass (*Thalassia testudinum*) remained in the secondary bay region of Christmas and Drum Bays (Pulich, 2001). Although 1956 is our earliest documented reference point, it is interesting to note that seagrasses were actually more abundant in the Galveston Bay system (even in East Bay and mid-Galveston Bay) during the early part of the 20th century based on anecdotal information. Turtle grass formerly occurred in West Bay, where total seagrass acreage (predominantly shoal grass) decreased from 570 ha (1,408 acres) in 1956, to 200 ha (494 acres) in 1965, to 50 ha (123 acres) in 1975, and finally to 0 in 1982 (Pulich and White, 1991). In the Galveston Bay system, the complete disappearance of shoal grass and turtle grass from the main bay system has been attributed to direct effects of dredge-and-fill operations, adjacent shoreline development, and land subsidence, along with episodic climatic events (i.e., hurricanes, freshwater inflow pulses). Indirect effects are suspected to involve nutrient/pollutant loading and spills or discharges from shoreline developments. With the reintroduction and recolonization of shoal grass and star grass in mid-West Bay near Galveston Island State Park in 1998, it appears that conditions have become favorable once again for restoration of seagrasses to the lower Galveston Bay system (Ikenson, 2001).

Texas Coastal Bend Region

For the Texas Coastal Bend (i.e., Corpus Christi) region, status of seagrass distribution indicates that this area has the third most extensive coverage of seagrasses (12.5%) on the Texas coast, exceeded only by the upper (28.5%) and lower (50.5%) Laguna Madre. Some historical information is available from the University of Texas Bureau of Economic Geology (UT-BEG) reports of Brown and others (1976) and White and others (1983) on submerged grassbed changes between the 1950s and 1979. Complete baywide status and trend analysis performed in 1994 provided a historical perspective for seagrass changes in the Texas Coastal Bend over the last 40 yr and confirms probable causes of trends (Pulich and others, 1997). The Coastal Bend Bays and Estuaries Program (CBBEP) commissioned this study as part of the data synthesis program for developing its Comprehensive Conservation and Management Plan (Coastal Bend Bays and Estuaries Program, 1998).

Results from this study suggested that different coastal processes have contributed to seagrass trends at localized sites. In the entire Corpus Christi/Redfish/Nueces Bays system, total seagrass bed acreage appeared to be fairly stable over a 40-yr timeframe, despite dynamic cycles and localized changes in seagrass bed distribution. Comparisons of 1958, 1975, and 1994 inventories for the Redfish Bay/Harbor Island complex revealed an increase of 819 ha (2,023 acres) between 1958 and 1975, but a decrease of 488 ha (1,205 acres) between 1975 and 1994, for a net increase of 330 ha (815 acres) in total area for this system between 1958 and 1994 (see table 1). Evidence of bed fragmentation or deterioration is reflected in the progressive increase in patchy grassbeds, while continuous beds apparently underwent conversion to patchy morphology; however, large increases in grassbeds (1,838 ha, or 4,540 acres, between 1958 and 1975; 519 ha, or 1,282 acres, between 1975 and 1994) occurred along Mustang Island over the same period.

Landscape analysis revealed “hot spots” of seagrass impact and loss in parts of the estuary. The Redfish Bay and Harbor Island systems may be at a stage where seagrass decline is escalating. Both progressive fragmentation of beds and seagrass loss have been noted. Although not definitive, accumulations of wrack, drift macroalgae, and epiphytes observed in Redfish Bay suggest possible water quality problems. Increase of shoreline development along the north Redfish Bay region may be contributing excess nutrients to this area. In addition, widespread physical damage to shallow water grassbeds was observed in the entire Redfish Bay/Harbor Island complex from motorboat propeller scarring (Dunton and Schonberg, 2002) and navigation channel impacts.

Table 1. Summary of total seagrass changes for Texas bay systems over four decades.

[Seagrass values are in hectares with acres in parentheses]

Bay systems	[1]Late 1950s or mid-1960s	[2]Mid-1970s	[3]1987 or early 1990s	[4]1998
		Galveston Bay system		
Galveston/ Christmas Bays	590[a] (1,457)	134[a] (331)	113[b] (279)	210[c] (519)
		Midcoast region		
Matagorda Bay system			1,099[b] (2,716)	
San Antonio Bay system		5,000[d] (12,350)	4,305[d] (10,638)	
		Coastal Bend region		
Aransas/Copano Bays			2,871[e] (7,094)	
Redfish Bay and Harbor Island	5,380[e] (13,293)	6,200[e] (15,320)	5,710[e] (14,109)	
Corpus Christi Bay system			2,568[e] (6,346)	
		Laguna Madre system		
Upper Laguna Madre	12,321[f] (30,445)	20,255[g] (50,050)	22,903[h] (56,593)	22,443[i] (55,456)
Lower Laguna Madre	59,153[f] (146,166)	46,558[g] (115,044)	46,624[h] (115,207)	46,174[i] (114,095)
Baffin Bay			2,200[j] (5,436)	

[1] Data for Galveston/Christmas Bays, Redfish Bay, and Harbor Island based on 1956/58 Tobin photography. Data for upper and lower Laguna Madre based on field surveys during mid-1960s.

[2] Data for Galveston/Christmas and Redfish Bay/Harbor Island based on 1975 (National Aeronautics and Space Administration Johnson Space Center (NASA-JSC) photography; San Antonio Bay based on 1974 NASA-JSC photography. Data for upper and lower Laguna Madre based on 1974–75 field surveys.

[3] Data for Christmas, Matagorda, and San Antonio Bay systems from 1987 NASA-Ames Research Center photography. Data for Aransas/Copano, Redfish, and Corpus Christi Bay systems based on 1994 TPWD photography. Data for upper and lower Laguna Madre based on 1988 field surveys. Data for Baffin Bay based on 1992 U.S. Fish and Wildlife Service National Wetlands Inventory photography.

[4] Data for Christmas Bay from 1998 Galveston Bay National Estuary Program photography. Data for upper and lower Laguna Madre from 1998 field surveys.

[a] From Pulich and White (1991).

[b] From Adair and others (1994).

[c] From Pulich (2001).

[d] From Pulich (1991).

[e] From Pulich and others (1997).

[f] Areas computed for this review from McMahan (1965–67). See Laguna Madre vignette.

[g] Areas computed for this review from Merkord (1978). See Laguna Madre vignette.

[h] Areas computed for this review from Quammen and Onuf (1993). See Laguna Madre vignette.

[i] Areas computed for this review. See Laguna Madre vignette.

[j] Areas computed for this review by Texas Parks and Wildlife Department, Coastal Studies Program, Austin, Tex. (unpub. data).

Laguna Madre System

Along the lower coast in the Laguna Madre, two largely independent dynamics have led to fundamental change in the seagrass community over the last 30 yr. First, hydrological alteration brought about by construction of the Gulf Intracoastal Waterway (GIWW) has resulted in moderation of the hypersaline conditions that set the Laguna Madre ecosystem apart from all others along the U.S. coastline. Salinity levels dropped from highs exceeding 100 ppt before construction of the GIWW to the low 50s ppt (after GIWW construction) in the upper Laguna Madre and from the mid-60s ppt to 50 ppt or less in the lower Laguna Madre. A successional process has ensued in lower Laguna Madre, with the euhaline (intolerant of salinity extremes) species manatee grass (*Syringodium filiforme*) increasing from 6,100 ha (15,073 acres) to 12,900 ha (31,875 acres) and turtle grass expanding in coverage from 440 ha (1,080 acres) to 11,100 ha (27,428 acres) between the mid-1960s and 1998 at the expense of shoal grass, a colonizing species that is also more tolerant of high salinities. In upper Laguna Madre, where hypersalinity had been even more extreme, cover of shoal grass increased from 12,300 ha (30,393 acres) to 22,900 ha (56,585 acres) for the first 20 yr of record; however, in the last 10 yr, manatee grass has begun to displace shoal grass. While manatee grass was not dominant anywhere in the upper laguna before 1988, it had become dominant over 1,450 ha (3,582 acres) by 1998.

Factors affecting the transmission of light to the bottom are the second set of forcing functions driving change to the Laguna Madre ecosystem. In lower Laguna Madre, increased light attenuation attributable to maintenance dredging practices for the GIWW is implicated in the loss of more than 10,000 ha (25,000 acres) of shoal grass meadow in deep areas. In upper Laguna Madre, an algal bloom of unprecedented persistence, the Texas brown tide, also reduced underwater light to the extent that 1,200 ha (2,965 acres) of seagrasses were lost, mostly shoal grass. One disturbing aspect of this perturbation is that recovery has not occurred in the 5 yr since the continuous bloom conditions.

The net result of these changes between the first whole system distributional survey in the mid-1960s and the most recent survey in 1998 (table 1) is an overall reduction in seagrass cover of 4%, a decrease of 36% in cover of shoal grass, and very large proportionate increases in the other species. The continuing loss of shoal grass is a special concern because it is almost the sole food of wintering redhead ducks (*Aythya americana*), and the Laguna Madre system of Texas and Tamaulipas serves as the wintering ground for more than 75% of the world population of redheads.

Causes of Change

Natural Process Changes

Natural processes include mostly climatic effects from drought, hurricanes, or freshwater inflows. Climatic events such as droughts, hurricanes, and periods of higher than normal rainfall can influence bay water levels and turbidities, which in turn affect the distribution of seagrass. The most extreme drought in recorded history, which occurred in Texas in the 1950s and climaxed in 1956 (Riggio and others, 1987), produced lower than average sea levels along the entire Texas coast. Following the 1950s drought was a period of abnormally high rainfall that was punctuated by aftermath rains associated with Hurricanes Beulah (1967) and Fern (1971). The period from 1968 to 1975 was much wetter than normal in south-central Texas. Bay and gulf water levels rose at an accelerated rate coastwide from the 1960s to 1975, as exemplified by tide gages at Galveston, Rockport, and Port Isabel, Tex., as well as in Louisiana, Mississippi, Alabama, and Florida (Swanson and Thurlow, 1973; Ramsey and Penland, 1989)

Human-induced Changes

Human-induced effects include (1) nutrient loading causing water-quality degradation and light attenuation from phytoplankton blooms, epiphyte growth, or macroalgae accumulation; (2) suspended sediments from dredging or boat traffic; and (3) direct physical disturbances, including dredged material deposition, removal by channelization and waterfront construction, boat propeller scarring, and effects of ship traffic. Based on recent events in the CBBEP study area and site-specific studies by Pulich (1985), Quammen and Onuf (1993), Dunton (1994, 1996), Onuf (1994), Pulich and others (1997), and Dunton and Schonberg (2002), seagrass distributions and trends are suspected of changing under the influence of natural estuarine conditions (e.g., freshwater inflow alterations and phytoplankton blooms) or contemporary human disturbances including dredging, channelization, and propeller scarring.

Some of the relative rise in sea level along the Texas gulf coast (including at Rockport) is attributed to subsidence associated with the pumping of ground water, or oil and gas, as opposed to eustatic sea-level rise (Ramsey and Penland, 1989). Average annual water levels recorded at the Rockport gage rose approximately 20 cm (8 inches) between 1964 and 1975 (Swanson and Thurlow, 1973). As the land surface subsided, seagrasses became more deeply submerged, leading to loss of plants in deeper waters. In the Galveston Bay system, the complete disappearance of shoal grass and turtle grass from the main bay system has been attributed to a combination of direct effects of dredge-and-fill operations, adjacent shoreline development, and land subsidence, along with episodic climatic events (i.e., hurricanes, freshwater inflow pulses) (Pulich and White, 1991). Indirect effects are suspected to involve nutrient/pollutant loading and spills or discharges from shoreline developments. The increases in seagrass cover occurring primarily on the bay side of Mustang Island in the Texas Coastal Bend region are probably a response to continuous submergence of wind-tidal flats because of subsidence and eustatic sea-level rise (Pulich and others, 1997).

Gaps in Data Coverage

Bay Systems Lacking Inventories

The middle Texas coast bay systems—including East Matagorda Bay, Matagorda Bay/Tres Palacios Bay, and San Antonio Bay/Espiritu Santo Bay—require more concerted, regular surveys and mapping coverage to ascertain seagrass dynamics. Only sketchy, rudimentary mapping data are currently available for these systems (Pulich, 1991; Adair and others, 1994).

Data Limitations and Future Needs

Some of the seagrass coverage values presented in this report (table 1) are not consistent with those of earlier sources, such as Pulich (1999). The compilation of seagrass coverage estimates presented in table 1 is based on reanalysis and digitization of maps from a variety of sources as cited in the table. In the process of preparing the old maps for digitization, some errors were found and corrected. Therefore, the coverage values can differ from those presented in earlier references and represent improvements on the earlier calculations.

Mapping data provide a reference baseline for comparing inventory results. It is critical to continue status and trends monitoring at the 1:24,000 scale to detect large changes in seagrass distributions caused by natural or human-induced environmental stresses. Mapping at 2–3 yr intervals represents the best strategy for routine monitoring of such seagrass landscape dynamics. Fragmentation patterns within grassbeds may be correlated with incipient stages of stress response (Robbins and Bell, 1994). Landscape patterns may indicate problems from the observed changes in bed morphology, before complete loss of seagrass occurs.

Poor availability of good color photography and reliable field data limits historical seagrass trend analysis in Texas to only about the last 25–30 yr. This perspective is important since long-term cycles may require more time to detect; however, because seagrass plants in Texas bays are mostly perennial species (star grass and wigeon grass being the exceptions), the population dynamics of established grassbeds are very stable compared to annual plants which must reestablish from seeds every year. Consequently, seagrass distributions can be reliably detected from clear, large-scale historical photography. Using such techniques, seagrass trends have been reasonably documented in the Galveston Bay, Texas Coastal Bend, and the Laguna Madre systems since the 1970s.

Results also indicate that resource managers must examine seagrass responses cautiously and on a case-by-case basis to identify environmental stressors causing changes. Net quantitative change in area for an entire bay seldom gives an accurate clue as to causes of seagrass changes. Rather, spatial location of changes as determined by geographic information system (GIS) analysis is necessary to infer relationships to environmental factors (Robbins, 1997). Generic stressors (e.g., water quality degradation, water level changes, and climatic conditions) may be suspected when effects are produced over wide bay areas; however, stressors such as mechanical or physical impacts often produce localized, site-specific effects. Monitoring programs for seagrasses must take into account the localized mechanisms by which stress responses are propagated.

Species composition of grassbeds is another key parameter for monitoring incipient stress effects. Replacement of a colonizing species, such as shoal grass, with a climax species, such as turtle grass, may represent normal succession in a grassbed over time. The opposite direction of succession, from turtle grass to shoal grass, is more likely to indicate some disturbance or stress to the grassbed. As a stage in the grassbed fragmentation process, this response could be expected to occur prior to complete loss of seagrass. Monitoring of such species succession patterns requires more detailed ground surveys coupled with remote sensing analysis.

In order to increase the resolution of site-specific data, remote sensing data should be collected at a 1:10,000 or larger scale. High-resolution photography should be interpreted by using groundtruthing data based on differential Global Positioning System parameter or techniques, which have a precision of +/- 1 m resolution. While this resolution of monitoring cannot be supported coastwide, target sites would be amenable to assessment at these finer scales of resolution. Landscape dynamic studies would provide a measure of variability over very small ground distances, yielding information on patch coalescence and spreading rates, or denudation rates of fragmenting grassbeds.

Overview of Seagrass Restoration Efforts

Seagrass Conservation Plan and Resource Management Needs

Texas Parks and Wildlife Department (TPWD), U.S. Fish and Wildlife Service (USFWS), and the U.S. Geological Survey (USGS) National Wetlands Research Center (NWRC) have conducted and coordinated monitoring and ecological assessment of seagrass along the entire Texas coast. Over the past 20 yr, special studies for Galveston Bay National Estuary Program (Pulich and White, 1991), Coastal Waterfowl/Fisheries Habitat Assessment (Quammen and Onuf, 1993; Adair and others, 1994), Coastal Natural Resources Inventories, and Coastal Bend National Estuary Program (Pulich and others, 1997; Dunton and Schonberg, 2002) have documented status and trends of most Texas seagrass beds and their sensitivity to disturbance from environmental factors. In recent years, results compiled from these studies on seagrass distribution trends, environmental factors (such as sea-level changes, water-quality conditions, and adjacent watershed land use), and physical disturbance features (e.g., dredged-material disposal, navigation channels, and motorboat propeller scarring) have led to serious concerns for Texas seagrass conservation and resource management.

The TPWD has been spearheading a statewide program that aims to coordinate research, monitoring, and management activities focused specifically on Texas seagrass habitat. Other collaborators in this process are the Texas General Land Office (TGLO), Texas Commission on Environmental Quality (TCEQ), and the two State Estuary Programs, one in the Galveston Bay area and the other in the Corpus Christi Bay area. During fall 1996, TPWD organized and cosponsored the "Texas Seagrass Symposium" to prepare for developing the comprehensive *Seagrass Conservation Plan for Texas* (SCPT) dealing with strategies and recommendations for protecting coastal seagrass habitat. Some 100 persons attended the planning symposium to identify issues relevant to producing a plan for Texas. These efforts culminated in the writing and publication of the SCPT in early 1999 (Pulich, 1999).

The plan represents an effort to coordinate and compile information on all the seagrass conservation, management, and research programs in Texas. Seagrass issues were categorized into three topical areas for planning purposes: research, management, and public stewardship. The SCPT reviewed the status of information on these issues and identified gaps in critical programs that should be addressed to slow or reverse losses of seagrass. After a comprehensive list of needs was developed, high-priority recommendations were made for the establishment of an organized, coastwide seagrass monitoring program, an integrated and easily accessible seagrass database, and stakeholder-driven stewardship and restoration projects. The SCPT also recommended development of more effective restoration techniques and applied management programs. Since the TPWD often collaborates on seagrass management problems with State groups (including the TCEQ and TGLO), with Federal resource agencies (including National Marine Fisheries Service (NMFS), U.S. Army Corps of Engineers (USACE), USGS, and USFWS), and with State universities, these groups were considered key to coordinating research and management programs.

Overview of Monitoring, Restoration, and Enhancement Opportunities

Monitoring for Seagrass Ecosystem Health

The SCPT (Pulich, 1999) recommended that a coastwide Texas seagrass monitoring plan be a high priority. In August 2000, the "Texas Seagrass Monitoring Workshop" was held in Corpus Christi to organize a working group of resource managers and scientists who would contribute to the design and development of such a monitoring plan. Approximately 75 people participated in discussions, which resulted in proposals for monitoring goals and objectives, seagrass health indicators, seagrass ecosystem model and sampling design, and a monitoring data management system. The consensus from this meeting was that, in order to detect meaningful changes in seagrass ecosystem health, there was a dual need for regular, organized field surveys and high-resolution landscape monitoring. The group also decided that this monitoring program should be based on a rigorous statistical design that allows assessment of baywide seagrass conditions from stratified random subsamples. From aerial photography and spatial analysis techniques, results from subsamples could be scaled up and extrapolated to larger areas.

In early 2001, the "Seagrass Monitoring Steering Committee" was formed—consisting of TPWD, TCEQ, TGLO, USGS, the University of Texas Marine Science Institute, NMFS, CBBEP, and the U.S. Environmental Protection Agency members—to design a formal seagrass monitoring program. The primary objectives of such a monitoring program are (1) to determine seagrass status and trends coastwide, at suitable scales; (2) to survey environmental and plant criteria and conditions indicative of ecological health of seagrass beds; (3) to compile and store seagrass datasets in an official database allowing users convenient access to seagrass distribution maps and associated, environmental indicator data; and (4) to subsequently apply the seagrass indicator monitoring data to specific management and environmental assessment programs. The first phase will include selection of a seagrass ecosystem and landscape dynamics model on which to base a quantitative sampling design, identification and testing of proposed seagrass health indicators, and establishment of the certified, standardized database for storage and analysis of seagrass monitoring data. A proposed Web-based data distribution network will link standardized, quality-assured spatial datasets at various agencies and scientific institutes. The monitoring program guidance, field and landscape sampling designs, database management system, and other recommendations of the Seagrass Monitoring Steering Committee were recently released for review in a strategic planning document (Pulich and others, 2003).

For management objectives, a habitat landscape analysis perspective was considered extremely important in helping to assess causes of seagrass impacts. This type of analysis requires an effective data management model for integrating plant and environmental indicator data with habitat landscape data. The spatial data generated from monitoring critical areas by using remote sensing and site-specific sampling potentially lend themselves to GIS data models for habitat management applications (Robbins, 1997; Lathrop and others, 2001). Modeling of appropriate spatial data will allow managers to hypothesize about causes from spatial patterns or correlations; these hypotheses can in turn be tested by field-monitoring measurements that are more labor intensive.

The combination of GIS and landscape analysis has power in its ability to show where changes are occurring and the areal extent of changes. This technique has revealed “hot spots” of seagrass disturbance and concentrated loss in parts of the Corpus Christi Bay system (Pulich and others, 1997) and the Laguna Madre (Onuf and others, 2003). Evidence of impact to grassbeds in the entire Redfish Bay/Harbor Island complex from mechanical damage (for example, motorboat propeller scarring), was documented by Dunton and Schonberg (2002). Other studies have shown indications of species succession in progress (Onuf and others, 2003).

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Galveston Bay System

By Warren Pulich, Jr.[1]

Background

The Galveston Bay system of Texas ranks as the seventh largest estuary in the United States, with about 153,000 ha (378,063 acres) of open water and about 44,000 ha (108,724 acres) of estuarine salt and brackish marshes (White and others, 1993). Figure 1 shows the extent of the Galveston Bay system and its watershed. Its secondary fisheries production is well known, with the system annually accounting for a major portion of the total Texas commercial harvest. In the 1980s, annual production amounted to an average of 35% of the harvest of shrimp, 30% of the blue crab, and 60% of the oyster catch of the Texas coast (Auil-Marshalleck and others, 2002). The Galveston Bay system displays this outstanding resource value despite being in the heart of the Houston metropolitan area. As a result of its location, the estuary has been heavily impacted by industrial and municipal development, discharge of pollutants and wastewater effluents, channelization and dredging projects, subsidence, and alterations in bay-water circulation dynamics. In particular, this region is recognized as a worldwide center for the shipping and petrochemical-refining industries. The Ports of Houston and Galveston, which rank 2d and 12th, respectively, in U.S. shipping tonnage, have greatly altered this bay system because of the extensive supporting network of dredged ship channels and canals, as well as numerous port-related facilities.

The population of the Galveston Bay area is one of the densest in the United States, having grown from 3.2 million in 1980 to 3.9 million in 1990 and to about 4.8 million in 2000 (U.S. Census Bureau, *www.census.gov*). In 1989, the upland habitats around the bay system consisted of 25% urban lands, 26% croplands, 30% rangelands, 9% forest, and 10% shrublands (White and others, 1993).

For the 1956 period, Fisher and others (1972) mapped approximately 2,024 ha (5,000 acres) of salinity-tolerant, submerged vascular vegetation (i.e., true seagrasses and wigeon grass (*Ruppia maritima*)) in the Galveston Bay system. This mapping did not include the Trinity River delta where considerable oligohaline submerged vegetation (e.g., water celery (*Vallisneria americana*) and wigeon grass) also occurs. The loss of at least 95% of this submerged vegetation by 1975 (White and others, 1985; Pulich and White, 1991) represents an alarming reduction of valuable estuarine fisheries habitat and environmental degradation.

Scope of Area

The Galveston Bay system is part of the Trinity-San Jacinto estuary, formed by discharge of the San Jacinto and Trinity Rivers into the upper secondary bays (San Jacinto and Trinity Bays). The San Jacinto River contributes median annual freshwater inflows to the bays of about 616.5 m^3 (500,000 acre-ft) and the Trinity River 9,247.5 m^3 (7.5 million acre-ft). A third major source of inflow to the bay system comes from the Buffalo Bayou/Houston Ship Channel, which carries Houston storm drainage and inflows from smaller tributaries on the west side of Houston.

In general, distinctly different submerged vegetation occurs in three separate estuary segments (fig. 2). The Trinity Bay-upper Galveston Bay segment extends along the western shore of Galveston Bay from Baytown to Texas City and along the eastern shore of Trinity Bay from Anahuac to Smith Point. The West Galveston Bay segment (in the lower Galveston Bay region) is bordered by the mainland to the south of the Houston area and by the Galveston Island shoreline. It extends from the city of Galveston proper down the island to San Luis Pass. Christmas and Drum Bays in the lower Galveston Bay region make up a third segment.

Methodology Employed To Determine and Document Current Status

Pulich and White (1991) determined extant locations of submerged vegetation, covering a minimum of 0.05 ha (0.125 acre), for the Galveston Bay system based on November 1987 color-infrared aerial photographs flown by NASA-Ames (California) and at a scale of 1:65,000. The delineations were corroborated by groundtruthing surveys in summer and fall 1988. White and others (1993) evaluated similar high-altitude, color-infrared photography flown by NASA-Ames in December 1989. Submerged vegetation coverage for these years was photointerpreted from 23 cm x 23 cm (9 inches x 9 inches) positive transparencies and mapped onto U.S.

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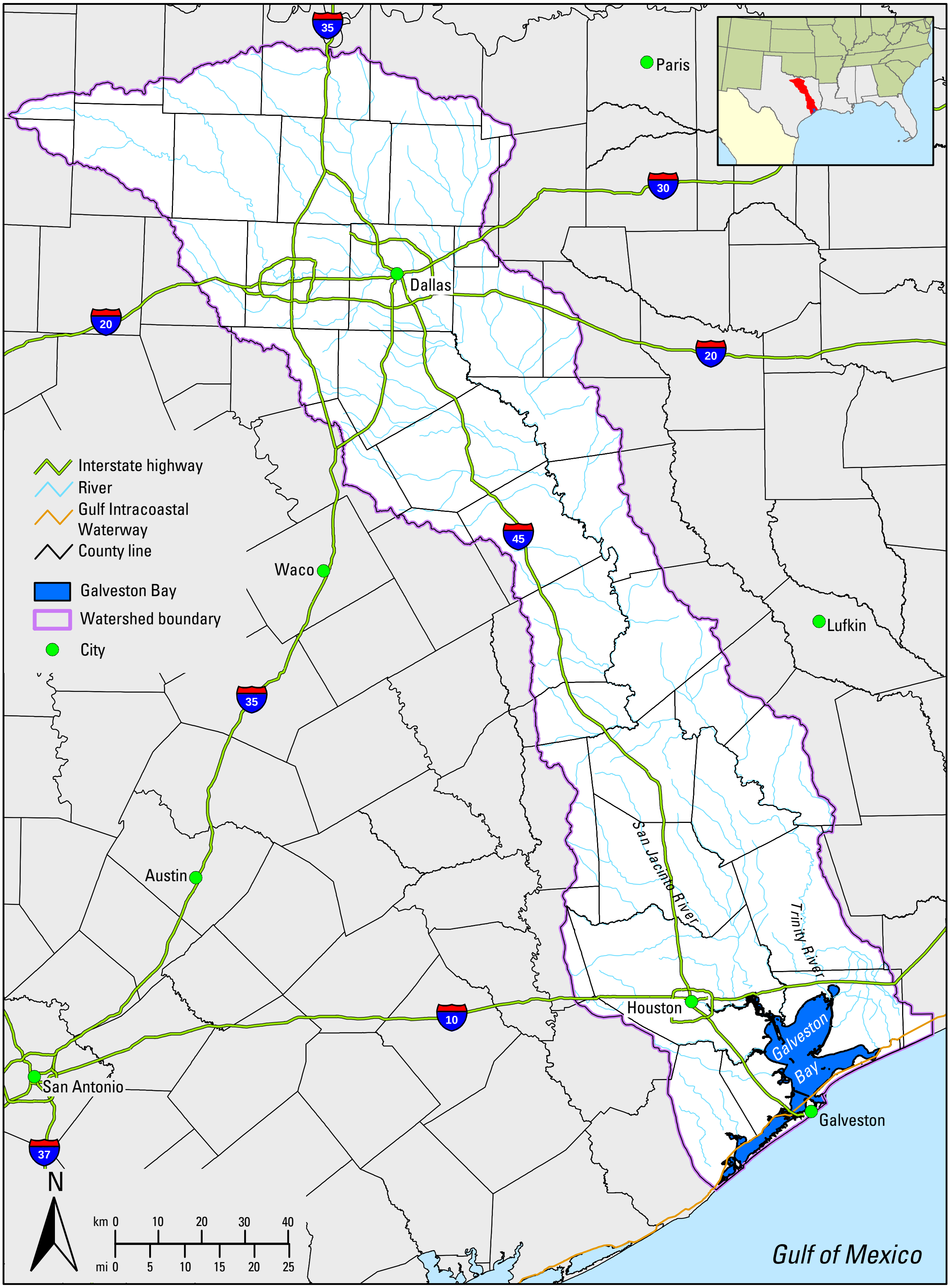

Figure 1. Watershed for Galveston Bay.

Geological Survey (USGS) quadrangle sheets at a scale of 1:24,000 by using zoom transfer scope techniques. Seagrass acreage statistics were determined by digitization with AutoCad™ software. Only generic seagrass distribution was delineated, and no classification of bed types was attempted. Disturbance features, such as residential developments, dredged channels, and boat marinas, were also mapped. The study conducted by Adair and others (1994) basically relied on line-transect field surveys during summer 1987 for qualitative assessment. Quantitative mapping methods (photointerpretation and zoom transfer methods) were not used to construct 1:24,000-scale maps.

Recent aerial photography (true color, Kodak SO358 film, flight scale 1:10,000) for Christmas and Drum Bays was taken on December 15, 1998, by Aerial Viewpoint, Inc., of Houston, Tex., under contract to the Galveston Bay Estuary Program (GBEP). Groundtruthing surveys supported by Global Positioning System (GPS) technology were performed in summer and fall 1999. Texas Parks and Wildlife Department (TPWD) staff in the Coastal Studies Program (Austin) scan-digitized the transparencies by using a flat-bed scanner and registered the raster images to 1:24,000-scale digital orthoquadrangles (DOQs). Seagrass polygons were then delineated with ArcView Image Analyst™ software by using "on-screen" digitization techniques, and datasets were entered into a geographic information system (GIS) (Pulich, 2001).

Methodology Employed To Analyze Historical Trends

The study by Pulich and White (1991) documented the chronology of submerged vegetation decline from the mid-1950s. Seagrass distribution at different time periods was quantitatively mapped, and the causes of trends were evaluated. The sources and scale of historical photography analyzed for the entire system were as follows: 1956, 1:24,000-scale black and white photomosaics from Edgar Tobin Aerial Surveys, San Antonio, Tex.; 1961, 1962, and 1965, 1:20,000-scale black and white photographs from the U.S. Coast and Geodetic Survey; 1975, 1:24,000-scale color-infrared photographs from National Aeronautics and Space Administration (NASA) Johnson Space Center (JSC), Houston, Tex.; and 1982, 1:24,000-scale color-infrared photographs flown by Texas General Land Office. Copies of all archived photography originally flown by the Federal or State agencies were obtained through Texas Natural Resources Information System, Austin, Tex. Except for 1982 photography, seagrass coverages from all other photomissions were photointerpreted and transferred by zoom transfer scope to USGS 1:24,000-scale quadrangle maps, and then acreage statistics were determined by digitization with AutoCad™ software. Former occurrences of seagrass in the bay area were corroborated from project reports of TPWD biologists at the laboratory in Seabrook (Pullen, 1961; West, 1973); from interviews with knowledgeable field biologists from agencies (pers. commun. with Robert Hofstetter and Charles Wilkes, TPWD; Roger Zimmerman, National Marine Fisheries Service) and universities (Sammy Ray, Texas A&M University-Galveston); from interviews with many local fishermen; and from review of other archived aerial photographs. Photography from 1982 was used only for visual observations.

Physical and hydrographic factors affecting distribution and growth of submerged rooted plants were analyzed in an attempt to identify processes contributing to seagrass impacts. Seagrass distribution was correlated chronologically with historical data on the following physical/hydrographic processes: shoreline erosion, relative sea-level rise associated with subsidence, hurricane and other climatic events, physical alterations related to channel dredging and onshore developments, and degradation in selected water quality parameters.

Status and Trends

The study conducted by Pulich and White (1991) examined all regions of Trinity Bay, upper and West Galveston Bays, and Christmas Bay. Environmental conditions at stressed sites where vegetation had disappeared were also compared with sites where plants persisted (i.e., east Trinity and Christmas Bays). Seagrass declines could be attributed to different processes in two different parts of Galveston Bay; upper Galveston and Trinity Bays showed significant contrast with lower Galveston Bay. In their parallel study on the status of seagrass in the Galveston Bay system, Adair and others (1994) reviewed and corroborated many of the Pulich and White (1991) observations. This later study by Adair and others (1994), however, proposed that environmental factors and the mechanism(s) causing submerged vegetation decline in West Bay were still unidentified although point-source pollution was probably involved. The acreage results presented below are from Pulich and White (1991).

Upper Galveston Bay and Trinity Bay

Figure 3*A* shows at least 86 ha (212 acres) in 1956 of submerged vegetation (wigeon grass, based on archived field reports), offshore of Clear Lake in the Seabrook area. Severe subsidence, erosion, and shoreline disturbance are likely to have placed wigeon grass along this western shoreline of upper Galveston Bay under considerable stress, which made these beds vulnerable to complete destruction by Hurricane Carla in fall 1961. Figure 3*B* shows this Seabrook shoreline in 1979, with dense development and devoid of submerged vegetation. Currently, fringe beds of wigeon grass still occur on the eastern shoreline of Trinity Bay in the proper season (60 ha, or 148 acres, were mapped here in 1987).

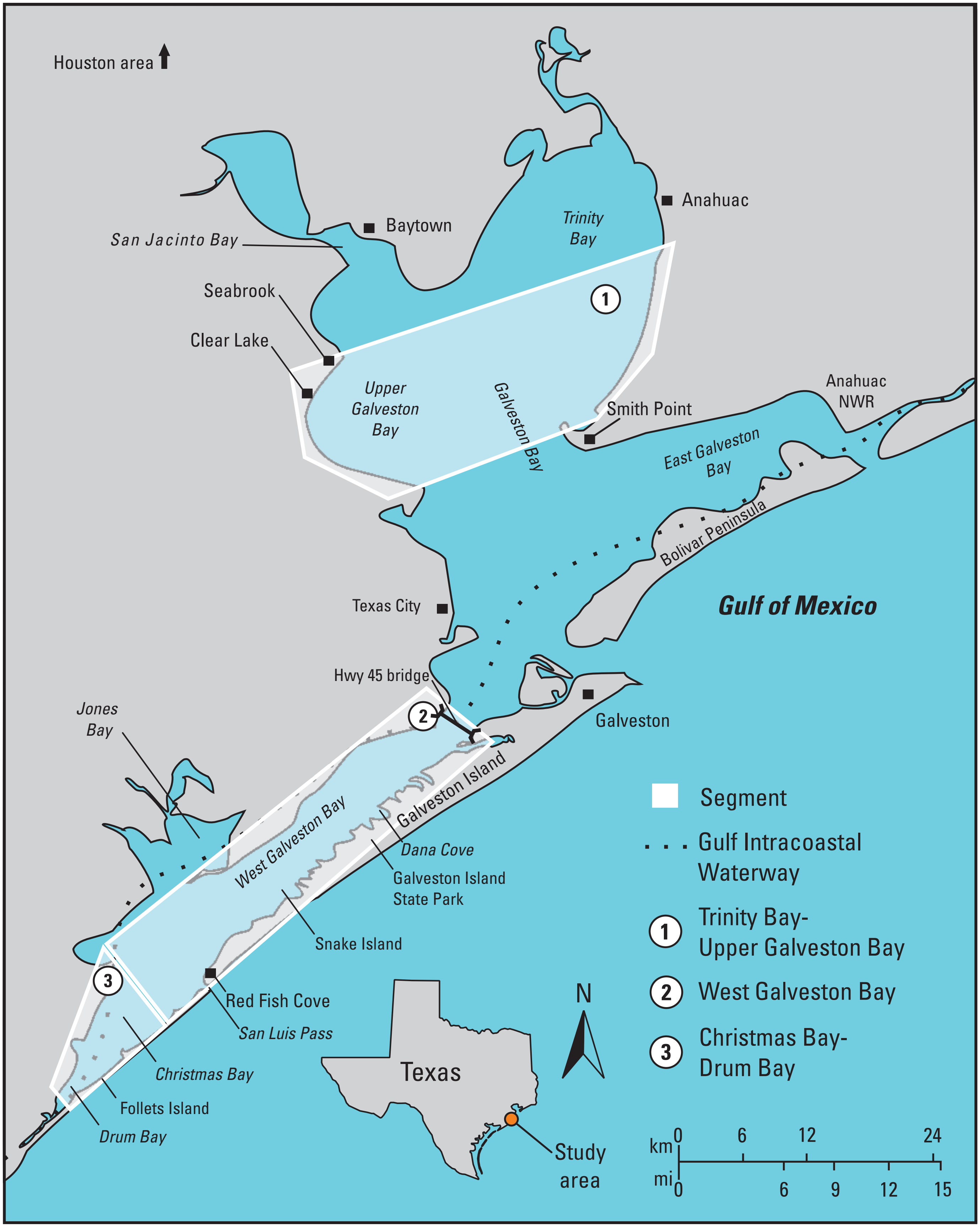

Figure 2. Scope of study area for the Galveston Bay system vignette.

West Galveston Bay

A different scenario for West Galveston Bay, consisting of typically polyhaline waters, emerges from the compilation of available data. True seagrasses, consisting of predominately shoal grass (*Halodule wrightii*), formerly occurred in this western part of the Galveston Bay system, both along the back side of Galveston Island and on the northern shoreline along the mainland. Based on historical photoanalysis, about 458 ha (1,132 acres) of shoal grass were estimated along the back side of Galveston Island during 1956 and 125 ha (308 acres) estimated during 1965 (Pulich and White, 1991). In 1975 photographs, only 37 ha (91 acres) of shoal grass beds were visible in the 3.6-km (2-mi) shoreline section just north of San Luis Pass. Wigeon grass had always been found mixed in with these shoal grass beds and continued to be found sporadically in some shoreline areas of Galveston Island after 1979, which is when shoal grass was last observed by fishermen in West Galveston Bay. All of these seagrass beds had disappeared completely by the early 1980s, as no trace was evident in July 1982 photographs (1:24,000 scale). Several small patches of turtle grass (*Thalassia testudinum*) were also located along the Galveston Island shoreline just north of San Luis Pass until the late 1970s, according to anecdotal reports.

Christmas Bay

Although true seagrasses disappeared from West Galveston Bay during the late 1970s, a remnant population survived in the secondary bay system of Christmas and Drum Bays, in the extreme southwest region adjacent to Galveston Island. Figure 4 shows seagrass distribution for Christmas and Drum Bays based on December 1998 color aerial photography and year 2000 ground surveys. Total coverage in Christmas and Drum Bays was mapped at 172 ha (424 acres) and consisted predominately of mixed shoal grass and star grass (*Halophila engelmannii*) (with wigeon grass often abundant, especially during spring). Shoal grass beds in Christmas Bay are also interspersed with about 20 patches of turtle grass along the south shore (totaling 0.65 ha, or 1.6 acres).

Mapping from 1975 and 1987 aerial photography (Pulich and White, 1991) revealed seagrass coverage of 93 ha (230 acres) and 77 ha (190 acres), respectively. White and others (1993) also examined the 1950s photography and estimated about 130 ha (321 acres) in Christmas Bay in 1956. These data seem to suggest that Christmas Bay grassbeds may have also declined during the 1960s through the 1980s period although not as precipitously as in West Galveston Bay. Based on the recent 1998 mapping results (172 ha, or 425 acres, of seagrass), this trend would appear to be reversing.

The size of the increase in seagrass acreage between 1988 and 1999 (as well as the decrease between 1975 and 1987) is probably misleading, however, because reexamination of the 1987 photography reveals that some seagrass was probably missed during this previous mapping because of turbid, high-tidal water and lack of resolution in the high-altitude 1987 photos. White and others (1993) were able to corroborate this idea by examination of 1989 color-infrared aerial photography. They remapped Christmas Bay seagrass from the 1989 photography and determined that about 156 ha (385 acres) of seagrass were present then. The 1989 photographs were taken at a lower tide (in December) than were the 1987 photographs (in November) 2 yr earlier, and seagrasses would have been more clearly visible.

Causes of Change

Upper Galveston-Trinity Bays

In mid-September 1961, Hurricane Carla physically destroyed the wigeon grass beds along the Seabrook shoreline late in the annual growth cycle (Pullen, 1961). As a result, increased erosion occurred during the ensuing winter and spring when the area is normally subjected to the full force of north and northeasterly winds associated with frontal passages. Increased nearshore water depth caused by the ongoing subsidence would have further reduced the amount of light reaching bay bottom and submerged vegetation, thus eliminating suitable wigeon grass habitat along the Seabrook shoreline as shown in a 1979 aerial photo (fig. 3*B*). This loss of habitat would account for the

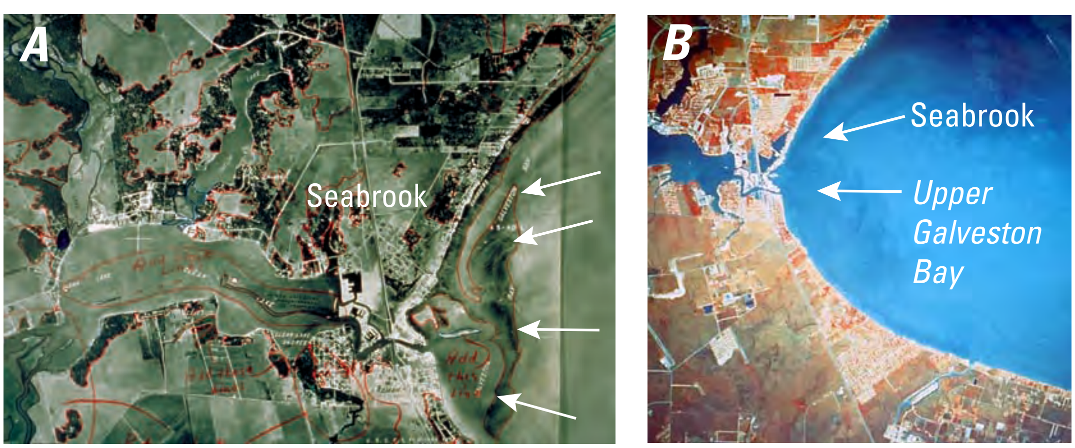

Figure 3. Aerial photographs of upper Galveston Bay along the Seabrook shoreline in 1956 (*A*) and 1979 (*B*). Arrows indicate where seagrass was present in 1956.

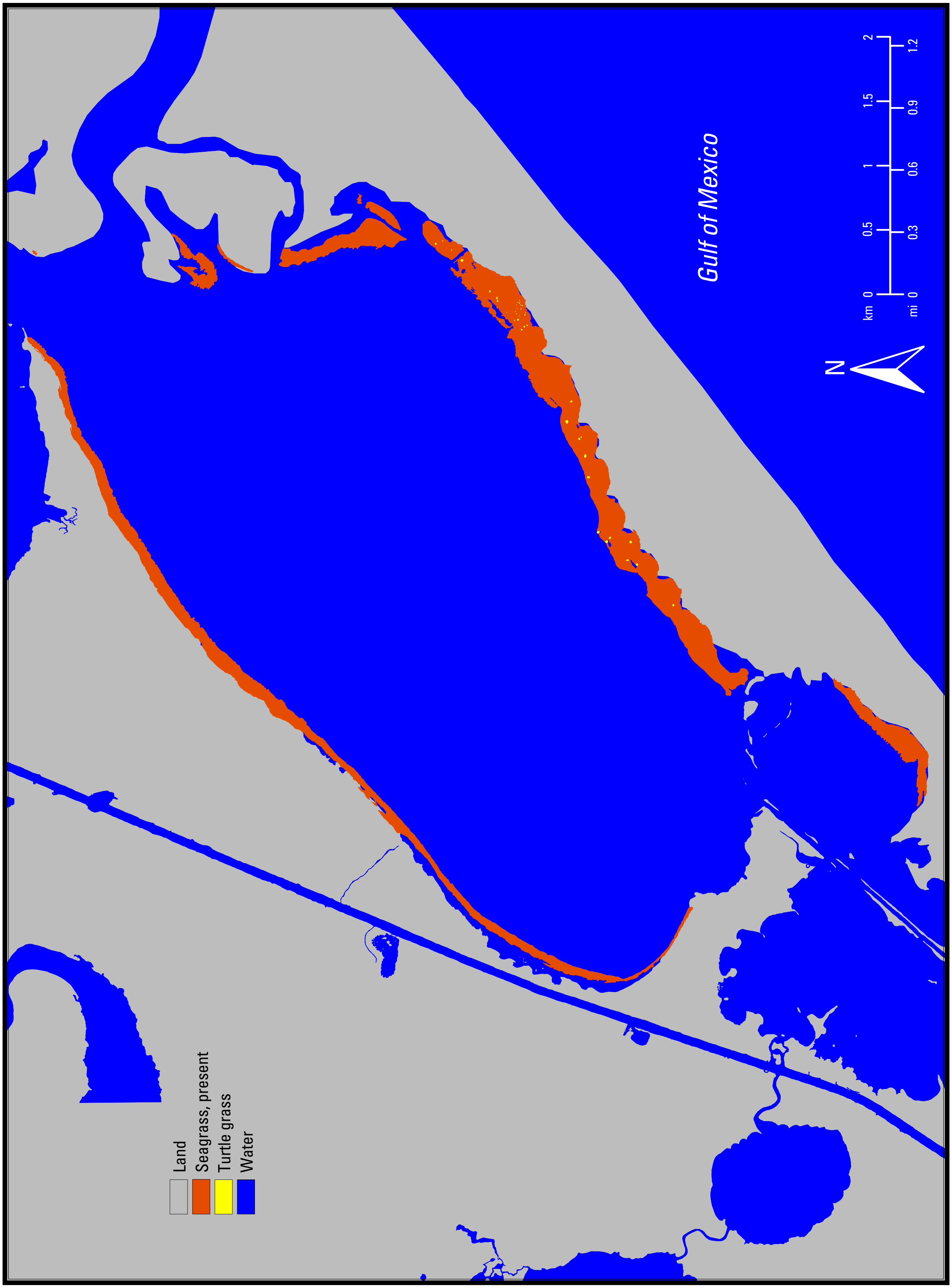

Figure 4. Distribution of seagrass in Christmas and Drum Bays, 1998.

complete disappearance of submerged vegetation in this bay area after 1961. A few areas along this shoreline that do have shallow depths are in exposed, high wave energy zones where wigeon grass was not found in the 1950s. This hypothesis is further supported in a comparison of the Seabrook shoreline area to the eastern shoreline of Trinity Bay. Areas on this opposite shoreline with suitable depths (i.e., unaffected by subsidence) that were protected during Hurricane Carla still support substantial wigeon grass beds (61 ha, or 150 acres, in 1987) during the proper season.

The upper bay near Seabrook, east of Houston, had also experienced major geomorphic modifications from land subsidence in the 1960s and 1970s (Swanson and Thurlow, 1973). Gabrysch (1984) calculated that groundwater withdrawal caused the Seabrook area to subside over 0.66 m (2.2 ft) between 1964 and 1973, or equivalent to a rate of about 6 cm (2.4 inches) per year. As a result, water depths near the Seabrook shore would have increased 30–60 cm (1–2 ft) between 1968 and 1977. Turbid freshwater inflows, after the drought of the late 1950s ended, added to the ongoing processes of subsidence, erosion, and shoreline disturbance and likely placed wigeon grass along this western shoreline of upper Galveston Bay under considerable stress, ultimately making it vulnerable to destruction by Hurricane Carla in 1961. This timeline agrees with observations of Eleuterius (1987), who reported a 33% reduction in seagrass beds in the Mississippi Sound as a result of erosion and sedimentation during Hurricane Camille and subsequent reductions in salinities in the aftermath of flooding.

The shoreline in the upper Galveston Bay area is predominantly an erosional one, and as a result, this shoreline has been artificially stabilized in many areas by bulkheads, riprap, and other erosion-control measures (Paine and Morton, 1986). In areas of rapid subsidence such as the Seabrook area, however, these stabilizing features also contribute to deeper water conditions near shore by inhibiting the natural development of a broad, shallow, and gently sloping, subaqueous, bay-margin profile that would likely develop along an unmodified, retreating shoreline. Figure 3*B* reveals the results of extensive bulkheading and associated urban development along the Seabrook shoreline in 1979, as well as the complete lack of seagrass (i.e., wigeon grass) along this western upper bay shore. Water depths of approximately 1 m (3.3 ft) were measured at shoreline bulkheads during surveys in the Seabrook area in 1988.

West Galveston Bay

Although Hurricane Carla slightly damaged West Galveston Bay shoal grass beds in 1961, the major decline of seagrass in this segment between 1956 and 1965 suggests a stronger correlation with increased residential and commercial waterfront development. During this same time period, many such shoreline projects occurred adjacent to West Galveston Bay, and these projects represented classic examples of "dredging and filling" of bay wetlands. Figure 5 shows the extent of such typical channel-front marina construction in 1989 along lower Galveston Island. Subsequent stress to seagrasses would have been caused by shoreline erosion, redistribution of dredged sediments, excessive nutrient loading from wastewater discharges, nonpoint-source runoff, and impact from recreational boats. Although there was only minor impact from subsidence to this area (Gabrysch, 1984), this process would also have contributed to slightly deeper water depth. In many respects, this explanation is consistent with the mechanism proposed for the decline of seagrass in Chesapeake Bay (Kemp and others, 1983); Cockburn Sound, Australia (Cambridge and McComb, 1984); and Tampa Bay, Fla. (Lewis and others, 1985). These cases document the serious damage to seagrasses from physical habitat disturbance and various agents that reduce water-column light penetration.

A major difference between Galveston Bay and these referenced bay systems may be the degree to which increased turbidity, subsidence, or eutrophic/polluted conditions have contributed to seagrass loss. Excessive nutrient or organic loading is believed to stress seagrass populations by stimulating growth of planktonic and epiphytic algae, as well as by causing premature senescence (Phillips and others, 1978; Kemp and others, 1983). Evidence of this stress was suggested by the observations that particulate chlorophyll made up a higher percentage of the suspended material in West Bay compared to Christmas Bay during years when seagrass was actually declining (Pulich and White, 1991). Heavy growths of epiphytes or phytoplankton inhibit seagrass photosynthesis by reducing the light available for absorption by plant leaves (Sand-Jensen, 1977; Dennison and others, 1993). In addition, phytoplankton blooms associated with storm or sewage runoff often cause anoxic events, which pose a lethal stress to seagrass because of the sulfide production during decomposition processes, especially in warm weather and calm water conditions (Nienhuis, 1983). Senescence, and then plant death, would quickly result if these highly toxic conditions occurred during historical low-dissolved oxygen

Figure 5. Lower Galveston Island north of San Luis Pass in 1989 showing marina and shorefront development (NASA-Ames color-infrared photograph, 1:63,000 scale).

events that often occurred in West Galveston Bay during the 1960s and 1970s.

While impacts from individual factors are difficult to separate and quantify from this analysis, the evidence is compelling that cumulative impacts from dredging, subsidence, shoreline activities, effluent discharges, and Hurricane Carla have been deleterious to seagrasses in the Galveston Bay system. Before reliable restoration of seagrass habitat in this bay system could occur, concerted efforts had to be made to control or eliminate the environmental stresses caused by humans.

Christmas Bay System

Although a higher salinity regime was suggested as a contributing factor favoring survival of Christmas Bay seagrass beds (Adair and others, 1994), actual data indicate that salinity in Christmas Bay is not significantly different from the lower third of West Galveston Bay, that is, from Snake Island to San Luis Pass (Pulich and White, 1991; Sheridan and others, 1998). This latter area is where turtle grass and shoal grass disappeared in the late 1970s. Actually, the seagrass dynamics appear consistent with the relative protection of Christmas Bay from human-caused factors (e.g., absence of major dredging and waterfront development in the bay, lack of point-source discharges, and generally low nonpoint-source runoff), as observed by Tomasko and others (1996). Recent seagrass increases in the lower Galveston and Christmas Bays region may reflect the renewed efforts to protect estuarine resources and water quality under State and Federal coastal zone programs.

Species Information

The Trinity Bay-upper Galveston Bay segment, a generally low-salinity area, has been historically characterized by the occurrence of mostly wigeon grass, with occasional other freshwater submerged vegetation. The West Galveston Bay segment, a moderate-salinity (mesohaline) area, historically has contained true seagrasses, predominately shoal grass, mixed with wigeon grass at times. Christmas and Drum Bays contain well-developed, polyhaline seagrass communities, including predominately shoal grass and star grass, and some turtle grass. This area represents the northernmost location for turtle grass on the Texas coast and is somewhat of a distribution anomaly because the closest known population is far to the south in Aransas Bay near Rockport.

Since the late 1950s, seagrass vegetation in upper Galveston and Trinity Bays (excluding the Trinity River delta) has consisted entirely of wigeon grass beds. There are anecdotal reports that shoal grass occurred there in the first half of the 1900s, but these reports are unconfirmed. In West Galveston Bay, shoal grass has historically been the predominant seagrass. In certain seasons, however, particularly in spring, localized grassbeds will contain some wigeon grass. As mentioned previously, a few small patches of turtle grass also occurred at the lower end of Galveston Island near San Luis Pass up until the late 1970s. These patches were interspersed in the large beds dominated by shoal grass.

Monitoring for Seagrass Health

Seagrass monitoring was not conducted while seagrasses were actually disappearing in the Galveston Bay system. As indicated above, studies since 1988 have been conducted after seagrass declines. These results have demonstrated the need for a proactive, regular monitoring program to assess seagrass health and to detect impacts before complete loss of seagrass habitat. Monitoring would also be important for documenting the success of restoration efforts. This documentation is a primary objective of the proposed coastwide Texas Seagrass Monitoring Program, which is currently under development (see Statewide Summary for Texas section) by State resource agencies (Pulich and others, 2003). As recommended by Neckles (1994), it is critical for such a program to develop good indicators of seagrass community health and then establish a statistically robust sampling scheme to measure these indicators.

Because of its sensitive, remnant seagrasses, the Christmas Bay system has been a priority conservation site for State resource management programs. In 1988, this area was designated an official State Coastal Preserve and State Scientific Area by the Texas General Land Office and TPWD, the two State agencies with primary coastal management authority for coastal wetlands. This designation allows for special monitoring programs and management regulations to provide extra protection to the seagrass community and other fisheries resources there. Water-quality protection and avoidance of wetland impacts are major legal safeguards provided by the Clean Water Act regulations (both section 404 and section 401). Monitoring of discharges of pollutants and wastewater, shoreline erosion control, and stringent mitigation measures for oil and gas exploration are being implemented for seagrass protection through various State and Federal programs (Galveston Bay National Estuary Program, 1995; Texas Parks and Wildlife Department, 1996).

Since its inception in 1988, the Galveston Bay National Estuary Program (GBEP) has worked to develop management plans and strategies focused on wetlands such as the pristine Christmas Bay system. The Christmas Bay Coastal Preserve was one of GBEP's first action-plan demonstration projects. Currently a "BayWatch" conservation plan is being developed to protect Christmas Bay from physical impacts (e.g., dredging or shoreline development) and to detect water-quality degradation, which will help to enhance seagrass beds.

Mapping and Monitoring Needs

The proposed Texas Seagrass Monitoring Program (Pulich and others, 2003) will combine a two-part approach of intensive probabilistic field sampling and landscape sampling from aerial photography. Aerial photography will be flown at 1:24,000 scale every 5–10 yr for status and trends assessment of the entire bay system, while annual photography at 1:9,600 or larger scale will be flown at sites where field sampling is occurring. The high-resolution photography will be especially important for monitoring seagrass habitat quality at target sites or documenting recovery of former seagrass areas.

Restoration and Enhancement Opportunities

State, Federal, and nonprofit agencies have been focusing on seagrass restoration in lower Galveston Bay. The GBEP identified submerged vegetation restoration as a major element of its Comprehensive Conservation and Management Plan (Galveston Bay National Estuary Program, 1995). A management goal was set to restore a total of 567 ha (about 1,400 acres) of seagrasses by the year 2004. Since 1994, U.S. Environmental Protection Agency (EPA), GBEP, U.S. Fish and Wildlife Service (USFWS), National Marine Fisheries Service (NMFS), and TPWD have provided substantial funding to reestablish true seagrasses in the lower part of West Galveston Bay where salinities and other water conditions are considered favorable.

The Near Coastal Waters Program of EPA Region 6 funded work in 1994 by Peter Sheridan (formerly at Galveston National Marine Fisheries Lab) to restore two sites on the back side of Galveston Island: one mid-island site at Snake Island Cove and a southern island site at Redfish Cove north of San Luis Pass (see Hammerstrom and others, 1998). In total, 1 ha (2.5 acres) was transplanted in late April and early May 1995 by using donor shoal grass plants from eastern and western Matagorda Bay. The Redfish Cove site became permanently established, and a 0.4-ha (1-acre) initial bed has expanded and now covers about 2 ha (5 acres). The Snake Island site survived about 1 yr and then failed rather abruptly over the summer period of the second year. Analysis of various factors and the dynamics of die-off suggest that primarily sediment factors may be involved and that water column light conditions, salinity, and eutrophic conditions seem to be ruled out (Sheridan and others, 1998).

Seagrass transplanting efforts have continued along this Galveston Island shoreline. In 1999, with funding from GBEP, Sheridan's group performed another restoration project as a follow-up to their 1994 work. This project examined the source of donor plant material as a factor in seagrass transplant success. Shoal grass donor plants were obtained from populations in Christmas Bay, East Matagorda Bay, Matagorda Bay, and the previous Redfish Cove site. Donor material was transplanted to areas in West Galveston Bay around the Redfish Cove-San Luis Pass area and has been monitored to determine effects of donor-plant source on transplanting survival. Although final results are not complete, there is some indication that local plants (i.e., from Christmas Bay) have surprisingly fared poorly. In October 2000, a third project was initiated by Sheridan at the Galveston Island State Park areas of Dana Cove and Carancahua Cove. This work, funded by the Natural Resource Damage Assessment (NRDA) Program, was in conjunction with a major wetland restoration project involving geotube engineering and marsh terracing techniques.

As part of the overall plan to stabilize eroding shoreline along the edge of Galveston Island, seagrass transplanting was conducted within the submerged cells created by marsh levee terracing. A newer transplanting technique was also compared and involved testing the planting boat/injection wheel method from Aquatic Subsurface Injection Systems, Inc. (ASIS), Tampa, Fla. This work, described in the Texas Coastal Bend vignette (this report), is detailed in a TPWD report by McEachron and others (2001). Seagrass donor material was obtained from the cooling pond of the Central Power and Light powerplant at Corpus Christi, Tex., and transplanted as either floating wrack or plugs in peat pots. Results to date indicate rather poor success for all planting techniques in the marsh terrace cells (P. Sheridan, 2003, oral commun.).

The USFWS has been very active in restoration efforts. The Clear Lake USFWS office funded applied research by Belaire Environmental Inc. from 1997 to 1999 to establish shoal grass in Snake Island Cove and Mentzell Bayou-Dana Cove areas at Galveston Island State Park. After 3 yr of effort during which some 6,735 m^2 (72,500 ft^2) of shoal grass was transplanted, the Snake Island site was reported to contain only 70 m^2 (750 ft^2) of shoal grass in 2000. Donor plants for all attempts were taken from Redfish Bay grassbeds near Aransas Pass, Tex., and planted as bare-root sprigs over various months from May through December. The USFWS, in collaboration with TPWD, has recently been instrumental in determining effectiveness of the ASIS planting/injection boat method from Tampa, Fla. Several extensive plots in Dana Cove and Snake Island Cove (over 0.8 ha, or 2 acres, total) were planted by ASIS with shoal grass sprigs from Corpus Christi in October 2000. These results showed very poor (2%–8%) success in 2002 from the injector boat method of transplanting.

Since 1998, when nontransplanted seagrasses first reappeared in the Dana Cove area, major seagrass recolonization has been underway near the Galveston Island State Park shoreline (Ikenson, 2001; John Huffman, USFWS, pers. commun., 2001). Star grass was the predominant species when patches were first noticed during reconnaissance for the Galveston Island State Park wetland restoration project there, with some shoal grass also present. Seagrass patches covered an estimated 4 ha (10 acres) the first year noted (fall 1998) and had spread to approximately 16 ha (40 acres) in December 2000. Annual color aerial photography has been taken for several years documenting the establishment and expansion dynamics of the grassbed. By December 2001, these patches had spread to more than 40.5 ha (100 acres) (estimated) and now contain considerable shoal grass.

The history of this recolonization event is under study, but there is circumstantial evidence that the source of the donor material is the upper Laguna Madre near Corpus Christi. This donor material can be traced to biologist Bryan Pridgeon with USFWS (formerly at the Clear Lake office), who brought several truckloads of shoal grass-star grass wrack from the Laguna Madre near the Corpus Christi Central Power and Light powerplant. This seagrass wrack material was transported to West Galveston Bay by truck and released into two cages in the State park waters near Dana Cove in December 1996. Only cursory follow-up monitoring was conducted to determine survival or establishment success over the next 2 yr, and it was not until fall 1998 that sufficient seagrass became established to be detected by field surveys. Since 1999, progressive seagrass expansion has been well documented from the annual aerial photographic surveys.

Based on the early progress of these “volunteer” beds in the Galveston Island State Park project, prospects are very good for successfully establishing new seagrass beds in additional projects in West Galveston Bay that are currently underway. The TPWD, USFWS, GBEP, the National Oceanic and Atmospheric Adminstration, the Texas General Land Office, and local industry and nonprofit partners are engaged in other restoration activities involving the creation of erosion barriers to protect existing marsh areas and restore marsh areas previously lost to subsidence and erosion. A recently completed project in Jumbile Cove and additional projects underway in Jumbile Cove and Delehide Cove in West Bay are anticipated to create hundreds of acres of shallow water habitat suitable for reestablishing seagrasses. The shallow, protected areas within the restoration sites are ideal for reestablishing seagrass beds, as the calm waters and established vegetation allow for precipitation of suspended solids and prevent resuspension of settled sediments, improving the clarity of water within the project sites.

The natural, unsupervised recolonization by seagrasses in Dana Cove is certainly evidence that conditions in this part of West Galveston Bay are now favorable to support major seagrass beds. The failure of some manipulated restoration projects in other parts of West Galveston Bay area (e.g., Sheridan, ASIS Inc.), however, sounds a note of caution to such efforts. These project results suggest that our scientific knowledge of seagrass restoration techniques may still be lacking and that we need a better theoretical basis of the controlling factors which determine success or failure of restoration projects.

Another element of the Galveston Bay restoration program involves establishment of a seagrass plant nursery. For a number of years, Jim Webb at Texas A&M University at Galveston has been developing and operating a seagrass and marsh plant nursery facility for applied research purposes. Through research and student projects, significant progress is being made in culturing shoal grass, wigeon grass, and turtle grass strains adapted to the Galveston Bay environment. These nursery-grown plants could eventually serve as the source (donor) material for large-scale applied restoration projects around the bay. Most of the donor material has been collected from Galveston and Christmas Bays and is being maintained in outdoor pond facilities on Pelican Island in Galveston.

In April 2002, the Galveston Bay Foundation (GBF) and GBEP organized and led citizen volunteers in collecting shoal grass plants from Cold Pass near Christmas Bay and then transplanting the sprigs to a 0.2-ha (0.5-acre) site in Redfish Cove (West Galveston Bay) as part of the annual GBEP “Marsh Mania” event. The main purpose of this outreach activity was to begin organizing and training citizen volunteers to work on seagrass and other wetland restoration projects. Through such educational events, an organized group of trained, interested citizens will become available for additional seagrass plantings which are being planned in the future. The GBEP and GBF will be able to call upon these experienced volunteers when needed. Though overall restored areas may be relatively small, the event promotes public awareness and stewardship of important and fragile seagrass habitats.

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Laguna Madre

By Christopher P. Onuf[1]

Background

Laguna Madre is a place of superlatives. It is separated from the Gulf of Mexico by Padre Island (fig. 1), the longest barrier island in the world. Excluding its tributary Baffin Bay (not treated in this account), Laguna Madre makes up about 20% of Texas' protected coastal waters while contributing 40%–51% of the State's commercial fish catch historically (Hedgpeth, 1947). The lagoon is even more important in support of the redhead duck, *Aythya americana.* Over 75% of the world population of redhead ducks overwinters on the Laguna Madre of Texas and on its sister lagoon, the Laguna Madre de Tamaulipas, just south of the delta of the Rio Grande (Weller, 1964). While in residence, the redhead ducks feed almost exclusively on the rhizomes of one seagrass species, shoal grass (*Halodule wrightii*). The lagoon's rich endowment of seagrasses provides critical habitat for fish and waterfowl. Seagrass meadows cover 65% of the bottom of Laguna Madre, and Laguna Madre accounts for more than 75% of seagrass cover along the entire Texas coast. The lagoon is also unusual for being one of only five hypersaline coastal ecosystems in the world (Javor, 1989).

Scope of Area

Laguna Madre is naturally divided into two segments (fig. 2) by a 20-km-long (12-mi-long) expanse of seldom-flooded sand and mudflats. Upper Laguna Madre extends 80 km (50 mi) northward from the flats to its terminus in the southeast corner of Corpus Christi Bay and ranges in width from 3 to 6 km (~2 to 4 mi). Lower Laguna Madre extends 95 km (59 mi) southward from the flats to within 5 km (3 mi) of the Mexican border and ranges in width from 3 to 12 km (~2 to 7 mi). Both segments have been included in systematic mapping efforts dating back to the 1960s.

A variety of circumstances act together to make the Laguna Madre hospitable environment for seagrasses. The average depth of the lagoon is 1 m (3.3 ft). Although the mapped watershed of the lagoon is more than 15 times the water area of the lagoon, the surrounding land is so flat and precipitation so low that most of the watershed does not actually contribute to the lagoon (Texas Department of Water Resources, 1983). Consequently, there is little inflow of nutrients or suspended particulates, and the waters of the lagoon are generally clear. The only exception is Arroyo Colorado, a distributary of the Rio Grande that is no longer connected to the river but serves as the major drain of the irrigated Lower Rio Grande Agricultural District and receives the treated effluents from most of the rapidly growing municipalities of the lower Rio Grande valley. Most of the shoreline is undeveloped. Padre Island National Seashore and The Nature Conservancy control most of the east side of Laguna Madre, while most of the west side is owned by large ranches, such as the King Ranch, and Laguna Atascosa National Wildlife Refuge. Only at the northern and southern ends is there considerable residential and marina development of the landward margin of the lagoon. Row crop agriculture occurs along intermittent creeks that drain into Baffin Bay.

Although only limited development of the shoreline has occurred, some modifications of the lagoon itself are exerting a strong influence on the Laguna Madre ecosystem. Prior to completion of the Gulf Intracoastal Waterway (GIWW) in 1949, a 20-km (12-mi) reach of emergent sand and mudflats had divided the lagoon into separate basins, which were then connected only under conditions of extreme high water. Now the GIWW provides a permanent water connection between upper Laguna Madre and lower Laguna Madre. This hydrological connection, together with currents produced by prevailing winds oriented along the long axis of the lagoon, improved circulation between basins and with the Gulf of Mexico and moderated the extent to which lagoon waters became hypersaline.

Methodology Employed To Determine and Document Current Status

Because of the particular importance of shoal grass for redhead ducks wintering on Laguna Madre, the delineation of seagrass meadows according to dominant seagrass species has been an essential element of seagrass mapping there. Accordingly, mapping efforts have built upon intensive field sampling along transects. Knowledge of current species

[1] U.S. Geological Survey, National Wetlands Research Center.

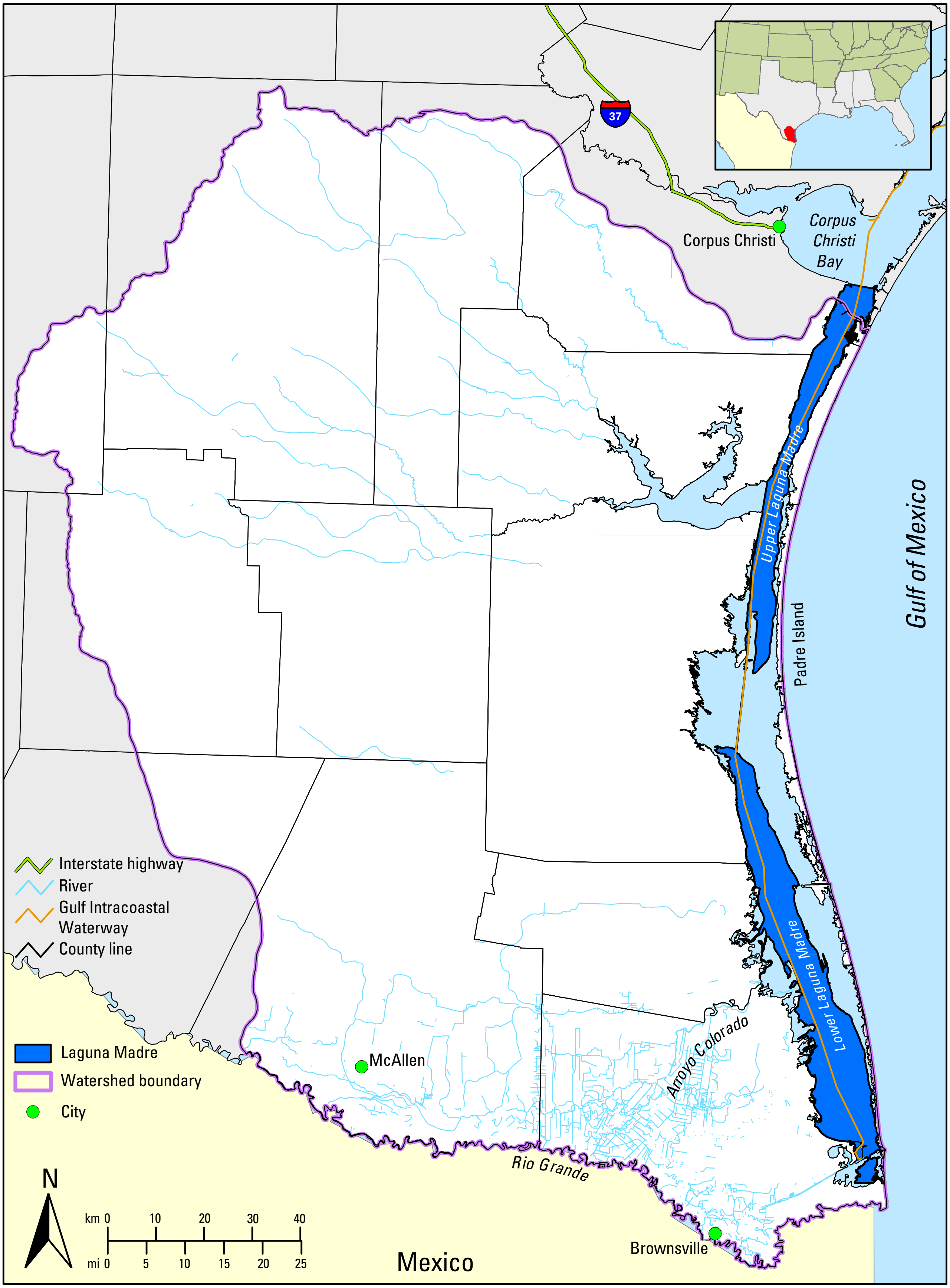

Figure 1. Watershed for Laguna Madre.

distribution is based on a survey conducted mostly in 1998, but sampling efforts continued into 2000 to resolve questions arising during production of the maps.

In lower Laguna Madre, sampling was along east-west transects at 2’ intervals of latitude from 26° 4’ N to 26° 48’ N. In upper Laguna Madre, transects generally conformed to those first used in a survey conducted in the mid-1970s (Merkord, 1978) and resampled in 1988 (Quammen and Onuf, 1993). Spacing was irregular, but most transects were oriented perpendicular to the long axis of the lagoon and ranged from 1’ to 5’ of latitude apart. In 1998, transects were added in the biggest gaps and eliminated where they were closer than 2’ of latitude to provide more uniform coverage. Along each transect, four 10-cm-diameter (4-inch-diameter) cores were collected at stations located 0.25’ longitude apart as determined by a Rockwell Precision Lightweight Global Positioning System (GPS) Receiver with Federal Precise Positioning Service. Species of seagrass present in each core and on the anchor were recorded in order of abundance, and the station was classified according to dominant species. Where cover type changed between stations, the boundary was established by sampling at successively smaller intervals until different species were dominant or the presence or absence of seagrass coverage occurred at stations less than 0.01’ (17 m) apart. Samples were not collected unless the GPS unit was operating with a positional accuracy of less than 10 m (33 ft).

The boundary coordinates of meadows and dominant species were plotted on 25 U.S. Geological Survey (USGS) 7.5’ topographic sheets representing the frequently submerged parts of the lagoon. National Aerial Photography Program imagery flown in 1995 was projected onto the topographic sheets to aid in interpolation of boundaries between transects. Maps were sent to the USGS National Wetlands Research Center in Lafayette, La., for digitization and geographic information system (GIS) analysis.

Methodology Employed To Analyze Historical Trends

Whole lagoon surveys of seagrass distribution have been conducted about every 10 yr since the mid-1960s. The 1960s maps were taken from Federal aid projects conducted by the Texas Parks and Wildlife Department (TPWD) (McMahan, 1965–67), based on transects across the lagoon at about 1.6-km (1-mi) intervals with stations 0.5 km (0.3 mi) apart. Visual estimates of abundance were made in clear water and from 10 samples taken with a posthole digger in turbid locations. The mid-1970s maps taken from Merkord (1978) were based on a less dense and less uniform set of transects with stations at 0.4-km (0.25-mi) intervals, coupled with aerial photointerpretation. Cover was estimated visually or from posthole digger samples. Quammen and Onuf (1993) reported the results of a survey conducted in 1988, following the sampling array of Merkord (1978), but relied entirely on three or four 10-cm-diameter (4-inch-diameter) core samples plus material brought up on the anchor to determine the occurrence of seagrasses at a location.

The source maps for the earlier surveys are small format and have been digitized and transferred to a common scale for this analysis of trends. The shore is of such low relief along much of Laguna Madre, particularly on the west side of Padre Island, that the landward limits are poorly defined. No attempt has been made to reconcile differences in shorelines between surveys. They either arise from differing interpretations of ambiguous features on aerial photographs or real differences in water levels between mapping periods. These discrepancies led to large uncertainty in the extent of bare areas in the lagoon. Almost no dredge islands were shown on the 1960s maps, but they appear on USGS quadrangles from the same period. Little change was seen in these features between the mid-1960s and the mid-1970s. Consequently, island polygons from the 1970s mapping were added to the 1960s map so that seagrass coverage estimates for the 1960s would not be artificially inflated. One area erroneously classified as “emergent and unvegetated submergent dredged material” on the 1970s map was redrawn to show an area of surviving seagrass.

No assessment of map accuracy was attempted. Boundaries in the 1960s map could be off by as much as 0.5 km (0.3 mi) because of the spacing of stations and by an unknown additional amount arising from errors in compass and dead-reckoning navigation. This uncertainty in seagrass boundaries, however, is mitigated by the facts that seagrass cover was continuous across the whole lagoon on many transects and there was a very limited occurrence of any species except shoal grass. The resolution of the mapping improved in successive surveys to 17 m (56 ft) on transects in 1998, but with greater uncertainty for interpolated boundaries between transects. North-south limits of seagrass cover were usually resolved to 0.01’ of latitude, but north-south limits between species were not resolved between transects in most cases. Because of these limits on the resolution of the maps, differences in seagrass coverage between surveys (decades) of less than 5 km^2 (2 mi^2) may not be indicative of change, unless the trend continues into subsequent surveys. As noted above, differences between surveys in what was considered “bare” bottom are even more uncertain.

Status and Trends

Lower Laguna Madre

The far right panel of figure 3 illustrates seagrass coverage in lower Laguna Madre in 1998. Seagrasses covered 67% of the permanently and regularly flooded portions of the basin. Deep areas (>1.3 m, or 4 ft) at the north and south ends of the basin accounted for most of the bare bottom there. Turtle grass (*Thalassia testudinum*) strongly dominated

Figure 2. Scope of area for the Laguna Madre seagrass vignette.

meadows at the south end, with very narrow fringes of shoal grass along some of the shore and at the deep margin of the meadow on the west side. In the shallow middle reach from 26º 15’ to 26º 30’ N, shoal grass dominated the shore zone, broadly (a 1–3 km, or 0.6–2 mi, band) on the Padre Island flats and more narrowly and variably along the more indented mainland shore. A continuous belt of manatee grass (*Syringodium filiforme*) dominated along the long axis of the lagoon, splitting around the deep, bare areas at the north and south ends. A 10-km (6-mi) slice of turtle grass meadow was sandwiched between the manatee grass and shoal grass bands on the Padre Island flats just north of 26º 15’. The manatee grass and shoal grass bands were continuous on the Padre Island flats to the north end of the lagoon. Along the steeper western shore, the shoal grass band was narrow and in some places absent. Manatee grass was limited to disjunct lobes on ridges of sand or beach rock projecting out from shore. A thin band of star grass (*Halophila engelmannii*) bordered most of the deep, bare area at the north end of the lagoon.

The distribution of seagrasses in lower Laguna Madre has been very dynamic over the past 30–35 yr (fig. 3). Seagrasses covered almost all of the entire bottom in 1965, but between 1965 and 1974, large tracts of deep bottom became bare and have remained bare to the present, with only small differences in configuration. The vegetated area of lower Laguna Madre decreased 12,595 ha (31,122 acres) (21%) from the mid-1960s to the mid-1970s, increased 66 ha (163 acres) (0.1%) to 1988, and then decreased by 450 ha (1,112 acres) (1.0%) to 1998 (table 1). Shifts in species composition have been even more far reaching, with shoal grass dominant over 89% of the seagrass meadow in 1965 progressively being replaced by manatee grass and turtle grass over the next three decades and only dominant over 46% of the meadow in 1998. Shoal grass decreased 19,930 ha (49,247 acres) (38%) from the mid-1960s to the mid-1970s, another 11,005 ha (27,193 acres) (34%) to 1988, and 476 ha (1,176 acres) (2%) to 1998 (table 1). In contrast, manatee grass increased 5,906 ha (14,594 acres) (95%) from the mid-1960s to the mid-1970s and 5,479 ha (13,539 acres) (45%) to 1988, before decreasing 4,712 ha (11,643 acres) (27%), and turtle grass increased 939 ha (2,320 acres) (215%) from the mid-1960s to the mid-1970s, 2,586 ha (6,390 acres) (187%) to 1988, and 7,171 ha (17,720 acres) (181%) to 1998.

Upper Laguna Madre

The far right panel of figure 4 shows seagrass coverage in upper Laguna Madre in 1998. Seagrasses covered 63% of the permanently and regularly flooded portions of the basin. Most of the bare areas coincided with the deepest parts of the lagoon, except the large expanse of bare bottom at the south end of the lagoon, in an area known locally as “Nine-Mile Hole,” and most of the bare fringe along the east side of the lagoon. Shoal grass accounted for 92% of the area covered by seagrasses, while manatee grass contributed 6%, star grass 1%, and wigeon grass (*Ruppia maritima*) 1%. Manatee grass was dominant in a large patch toward the north end, with a slender southward projection on the edge of a deep trough; however, small patches occurred in the surrounding shoal grass meadow. Star grass was confined to the deep fringe of meadows in two locations in the middle reach of upper Laguna Madre. Wigeon grass was dominant only on shallow flats in the south adjacent to the GIWW and to an undetermined extent westward.

In upper Laguna Madre, shoal grass cover increased through 1988, but then a central deep area and the south end went bare, and a large patch of manatee grass took over at the

Table 1. Bottom cover (in hectares) of lower Laguna Madre in different years.

Cover Type	Mid-1960s[1]	Mid-1970s[2]	1988[3]	1998
Bare	9,181	20,978	21,026	22,761
Shoal grass[4]	52,529	32,599	21,594	21,118
Manatee grass[5]	6,188	12,094	17,573	12,861
Turtle grass[6]	436	1,375	3,961	11,132
Star grass[7]		490	3,496	1,063
Total	68,334	67,536	67,650	68,935
Total vegetated	59,153	46,558	46,624	46,174

[1] Areas computed from digitized versions of maps from McMahan (1965-67).

[2] Areas computed from digitized versions of maps from Merkord (1978).

[3] Areas computed from digitized versions of maps from Quammen and Onuf (1993).

[4] *Halodule wrightii*.

[5] *Syringodium filiforme*.

[6] *Thalassia testudinum*.

[7] *Halophila engelmannii*.

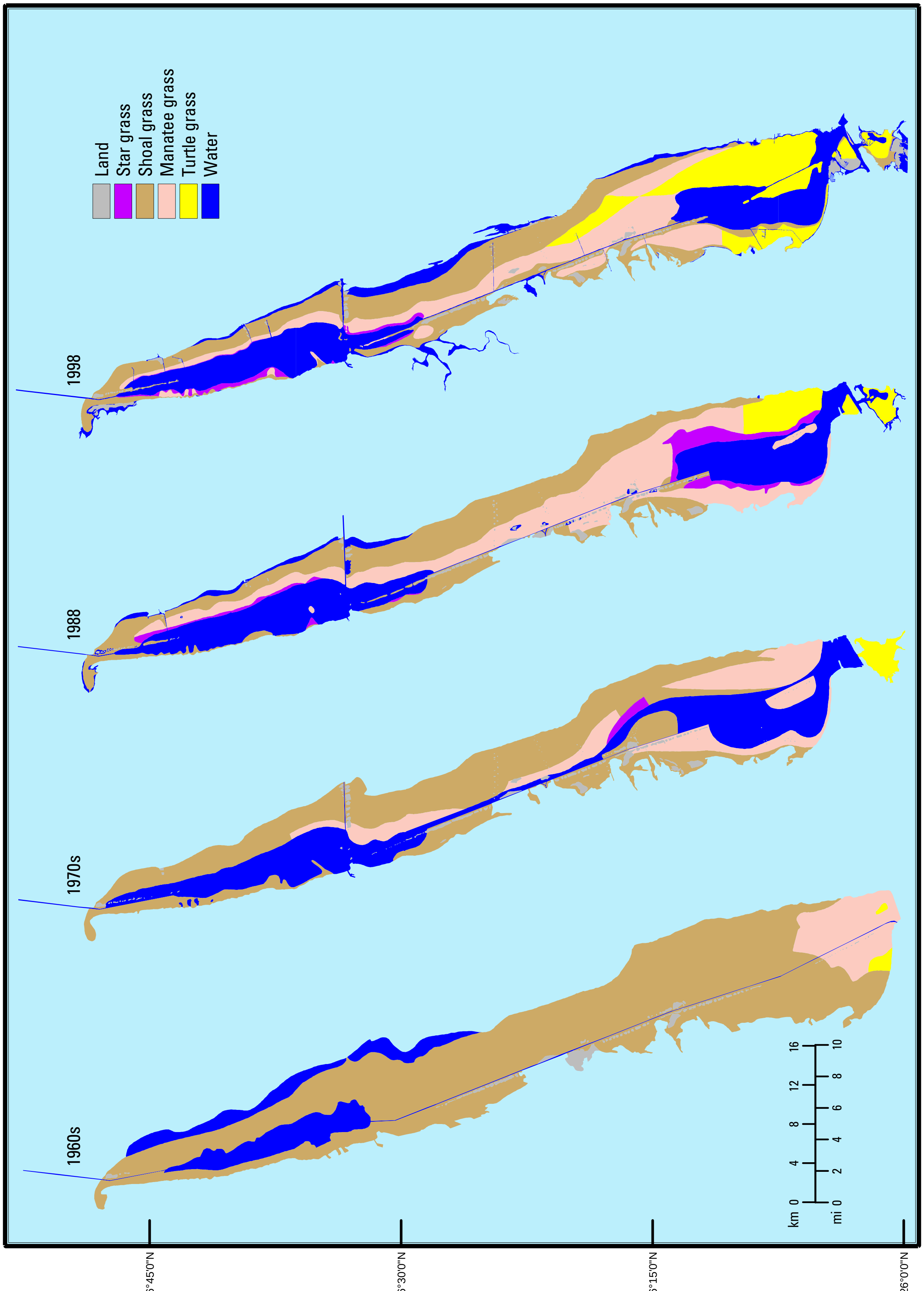

Figure 3. Distribution of seagrass in lower Laguna Madre, 1960s–1998.

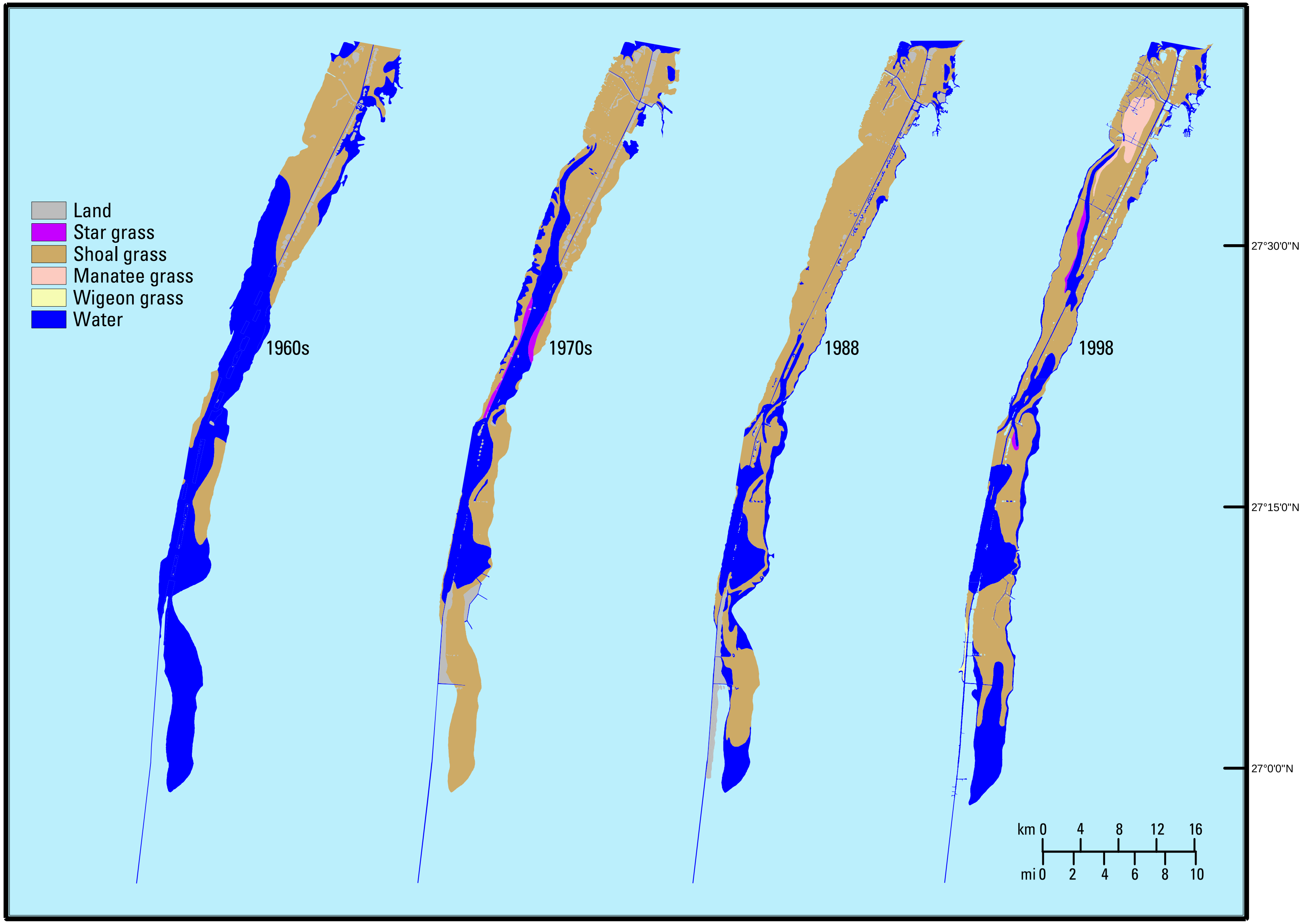

Figure 4. Distribution of seagrass in upper Laguna Madre, 1960s–1998.

north end by 1998 (fig. 4). In contrast to the lower Laguna Madre, the area of vegetated bottom in upper Laguna Madre increased 7,934 ha (19,605 acres) (64%) from the mid-1960s to the mid-1970s, increased again 2,648 ha (6,543 acres) (13%) to 1988, and then decreased 460 ha (1,137 acres) (2%) to 1998 (table 2). Shoal grass cover increased 7,323 ha (18,095 acres) (59%) from the mid-1960s to the mid-1970s, increased again 3,259 ha (8,053 acres) (17%) to 1988, and then decreased 2,350 ha (5,807 acres) (10%) by 1998. Manatee grass was not recorded in the first two surveys, covered an insufficient area to be mapped in 1988, but by 1998 dominated 1,452 ha (3,588 acres) of the bottom, accounting for 4% of the seagrass meadow.

Laguna Madre as a Whole

From the mid-1960s to 1998, the area of vegetated bottom decreased 2,856 ha (7,057 acres) (4%) for the lagoon as a whole. Cover of turtle grass increased from barely present to dominating 11% of lagoon bottom, and manatee grass first increased from 6% to 17% but then fell back to 14%. All of these increases were at the expense of shoal grass, which covered 64% of lagoon bottom in the earliest survey but diminished to 40% by 1998 (table 3).

Causes of Change

Lower Laguna Madre

Between 1965 and 1974, 118 km^2 (46 mi^2) of vegetated bottom became bare. Noting the alignment of new unvegetated bottom with the GIWW and adjacent naturally deep parts of the lagoon, Merkord (1978) proposed that reduced light resulting from turbidity caused by maintenance dredging of the waterway was likely responsible for the loss of seagrass. Intensive monitoring of the light regime 3 months before and 15 months after a maintenance dredging project in 1988 supported this interpretation. The study documented increased light attenuation to the end of the study in the region where seagrasses had been lost (Onuf, 1994). Although turbidity from dredging activity dissipates within hours of the cessation of dredging, the mounds of dredged material left behind are much more prone to resuspension. Laguna Madre is renowned for its winds, and frequent episodes of resuspension of the fine, unconsolidated deposits by waves and dispersion by currents assure propagation of dredging effects over long spans of space and time.

The GIWW is also involved in the major shifts in species composition that have occurred over the last 35 yr and that are still in progress. Prior to completion of the waterway in 1949,

Table 2. Bottom cover (in hectares) of upper Laguna Madre in different years.

Cover Type	Mid-1960s[1]	Mid-1970s[2]	1988[3]	1998
Bare	20,826	10,785	9,893	12,950
Shoal grass[4]	12,321	19,644	22,903	20,553
Manatee grass[5]	0	0	0	1,452
Star grass[6]	0	611	0	307
Wigeon grass[7]	0	0	0	132
Total	33,147	31,040	32,796	35,394
Total Vegetated	12,321	20,255	22,903	22,444

[1] Areas computed from digitized versions of maps from McMahan (1965–67).

[2] Areas computed from digitized versions of maps from Merkord (1978).

[3] Areas computed from digitized versions of maps from Quammen and Onuf (1993).

[4] *Halodule wrightii*.

[5] *Syringodium filiforme*.

[6] *Halophila engelmannii*.

[7] *Ruppia maritima*.

a 30-km (19-mi) reach of usually exposed sand and mudflats effectively isolated the two sections of the lagoon. Because of limited water exchange with the Gulf of Mexico, infrequent freshwater inflow, and average annual evaporation that was approximately twice the average annual precipitation, it was not uncommon to find salinities double that of the adjacent gulf in lower Laguna Madre and triple that of the gulf in the southern extremity of upper Laguna Madre. The 40-m-wide by 4-m-deep (125-ft-wide by 12-ft-deep) channel that was cut through the flats provided the first permanent water connection between the two segments of the lagoon. Prevailing winds along the long axis of the lagoon now enhance exchange within the lagoon and between the lagoon and the Gulf of Mexico, considerably moderating hypersaline conditions within the lagoon. Since completion of the waterway, salinities never rise much above 50 ppt.

Life history and competitive characteristics of the plants almost certainly account for the time-course of change evident in this trends analysis. Not only is shoal grass the only species that can tolerate salinities greater than 60 ppt, but it is also superior to manatee grass or turtle grass as a colonizer. Therefore, shoal grass was probably widespread before construction of the waterway, although there are only incidental reports from a few locations for that period. Given its strong colonizing abilities, it is not surprising that shoal grass overwhelmingly dominated lower Laguna Madre by the time of the first systematic survey of seagrass cover in the mid-1960s, while the other species were still limited to within 12 km of the natural gulf outlet at the extreme south end of the lagoon (fig. 3). Gradual displacement of shoal grass from the south by manatee grass, which in its turn was later displaced by turtle grass (fig. 3), is consistent with what we know of the relative colonizing and competitive abilities of these species under conditions of moderate salinity (Quammen and Onuf, 1993; Onuf, 1995).

Upper Laguna Madre

Upper Laguna Madre has experienced the same system shift in salinity regime as did lower Laguna Madre, yet its trajectory of seagrass change has been very different. While shoal grass cover increased by 85% through 1988 in upper Laguna Madre (fig. 4), it decreased by almost 60% in lower Laguna Madre (fig. 3). Manatee grass was not even seen in upper Laguna Madre until 1988 (Quammen and Onuf, 1993) and was not detected in a decennial sampling program until 1998 (fig. 4). Changes in upper Laguna Madre seem to lag those in lower Laguna Madre by 20 or 30 yr. The extreme hypersalinity of the southern section of upper Laguna Madre before completion of the GIWW probably accounts

Table 3. Bottom cover (in hectares) of Laguna Madre (whole lagoon) in different years.

Cover Type	Mid-1960s[1]	Mid-1970s[2]	1988[3]	1998
Bare	30,007	31,763	30,919	35,711
Shoal grass[4]	64,850	52,243	44,497	41,671
Manatee grass[5]	6,188	12,094	17,573	14,313
Turtle grass[6]	436	1,375	3,961	11,132
Star grass[7]	0	1,101	3,496	1,370
Wigeon grass[8]	0	0	0	132
Total	101,481	98,576	100,446	104,329
Total Vegetated	71,474	66,813	69,527	68,618

[1] Areas computed from digitized versions of maps from McMahan (1965-67).

[2] Areas computed from digitized versions of maps from Merkord (1978).

[3] Areas computed from digitized versions of maps from Quammen and Onuf (1993).

[4] *Halodule wrightii*.

[5] *Syringodium filiforme*.

[6] *Thalassia testudinum*.

[7] *Halophila engelmannii*.

[8] *Ruppia maritima*.

for the difference in responses between parts of the lagoon. Even shoal grass cannot tolerate salinities over 100 ppt. Consequently, it was probably absent from most of upper Laguna Madre at least through 1949, except in areas near Corpus Christi Bay. Because source populations were much farther away, colonization of shoal grass continued until 1988 in upper Laguna Madre, as opposed to already being maximal in lower Laguna Madre by 1965. Similarly, the establishment of manatee grass in upper Laguna Madre took even longer because, in 1965, source populations were even more remote, at the south end of lower Laguna Madre or across Corpus Christi Bay, near the closest gulf outlet to the north. Not surprisingly, establishment of manatee grass in upper Laguna Madre was long delayed compared to lower Laguna Madre, where it was present from the outset (Onuf and others, 2003).

Other recent changes in upper Laguna Madre have reversed the trend of increasing seagrass cover seen through 1988. Laguna Madre was renowned for its crystal clear water until June 1990 with the appearance of the "Texas brown tide." This phytoplankton bloom varied in intensity but was continually present from 1990 to 1997 and has flared up sporadically since. Over large areas, light at 1 m (3.3 ft) was reduced to half of what it had been before the brown tide, and shoal grass gradually died back in deep areas (fig. 4) (Onuf, 1996a). Although a suite of factors played a role in the initiation and unprecedented persistence of the brown tide (Stockwell and others, 1993; Buskey and others, 1997, 2001), nutrients regenerated from the gradual dieback of the seagrass meadow almost certainly were involved in sustaining the bloom, until steady state was reached between seagrass distribution and the brown-tide-influenced light regime (Onuf, 2000). A disturbing aspect of this perturbation is that as yet there is little sign of recovery. Apparently because of the loss of seagrass cover, the bottom is much more prone to sediment resuspension. Because new recruits have no reserves to tide them over during episodes of low light, establishment has not occurred.

The loss of shoal grass in relatively shallow water at the south end of upper Laguna Madre between 1988 and 1998 (fig. 4) is problematic. The area was not visited between surveys, and nothing is known of environmental conditions that might have precipitated the loss. Multiple shoal grass patches were seen in this area in 2003 (K.H. Dunton, University of Texas, written commun., 2004), indicating partial recovery to that time.

Monitoring for Seagrass Health

The status of seagrasses in Laguna Madre is determined from transect sampling. The dominant seagrass species and occurrence of other seagrasses and algae are recorded for each of the four 10-cm-diameter (4-inch-diameter) cores taken at each station, plus similar information for what is brought up on the anchor. A subset of cores is retained for biomass determination. From these data, detailed information on patchiness and condition of seagrasses can be extracted beyond the mapped distributional patterns. Biomass change presaged distributional change by a couple of years in assessments of the impact of the brown tide on seagrasses (Onuf, 1996a, 2000). Fixed, underwater, light-monitoring stations (Dunton, 1994; Onuf, 1996a) and continuous monitoring of chlorophyll and turbidity are also in progress and will provide critical information in the assessment of seagrass health near the monitoring stations. As yet, no systematic assessment of seagrass health has been performed; however, much higher biomass of macroalgae found in the middle reach of lower Laguna Madre than elsewhere in the lagoon in 1988 (Onuf, 1996b) may point to localized nutrient overenrichment.

Mapping and Monitoring Needs

In 1999 the "Seagrass Conservation Plan for Texas" was adopted by the TPWD, the Texas General Land Office (TGLO), and the Texas Commission on Environmental Quality (TCEQ), which are the State agencies charged with protection of seagrass habitats. One of the highest priority needs outlined in the plan was to develop a statewide seagrass mapping and monitoring plan. In July 2000 the TCEQ added seagrass as an "Aquatic Life Use" in the Texas Surface Water Quality Standards (*http://www.tnrcc.state.tx.us/permitting/waterperm/wqstand*, p. 16, 39). This designation is currently pending U.S. Environmental Protection Agency (EPA) approval. The adoption of seagrass as an aquatic life use only increases the need for a routine monitoring plan. Currently, State and Federal agencies as well as seagrass researchers are developing such a plan.

Unlike other Texas coastal estuaries, there have been seagrass distribution studies of the whole Laguna Madre system conducted approximately once a decade since the mid-1960s (McMahan, 1965–67; Merkord, 1978; Quammen and Onuf, 1993; this report). This decade-cyclic sampling used for determining the status and trends reported in this vignette has revealed many interesting changes in seagrass distribution in the Laguna Madre. These mapping efforts also prove the need for more intensive monitoring at targeted locations in order to better understand factors affecting the health and changes of seagrass beds. These targeted locations should include areas near development, such as in the town of South Padre Island or in the vicinity of potential large inputs of nutrients such as at the mouth of the Arroyo Colorado.

Restoration and Enhancement Opportunities

Seagrass restoration and enhancement opportunities in the Laguna Madre primarily focus around U.S. Army Corps of Engineers (USACE) programs such as the management of maintenance dredged material from the GIWW and mitigation for regulated permit actions such as oil and gas exploration, residential canal subdivisions, and dredging. Two other areas of focus include the reduction of nutrients into the Laguna Madre ecosystem and educating the public about boat propeller impacts (scarring) to seagrass meadows.

In 1988, seagrasses covered 730 km^2 (282 mi^2), or about three-quarters of the subtidal area of the Laguna Madre (Onuf, 1996b). Therefore, activities which are regulated under the USACE Section 10 (Rivers and Harbors Act of 1899) and Section 404 (Clean Water Act of 1976) programs have a high potential for impacting seagrasses. Sections 10 and 404 activities such as marinas, oil and gas exploration, pipelines, residential canal subdivisions, and channels are often planned in areas containing ecologically sensitive seagrass beds. As such, the State and Federal natural resource agencies (TPWD, TGLO, TCEQ, U.S. Fish and Wildlife Service (USFWS), EPA, USACE, and National Park Service) work closely with applicants in an effort to avoid and minimize seagrass impacts as much as possible. Compensatory mitigation (in the form of seagrass creation or restoration) is sought for justifiable and unavoidable impacts. Significant efforts have been made to avoid and/or minimize impacts over performing compensatory mitigation. Seagrass mitigation is very costly and often unsuccessful. Furthermore, it appears that seagrass mitigation sites take many years before they begin to function like natural seagrass beds (Montagna, 1993). There are very few areas that are currently available to be used as seagrass mitigation sites in the Laguna Madre. The resource agencies and academia are currently evaluating strategies to address the aforementioned issues and to identify the most beneficial way to approach seagrass mitigation associated with USACE permit actions.

There is evidence that maintenance dredging activities along the GIWW may have caused significant loss of seagrass in the Laguna Madre (Onuf, 1994). Concerns about disposal of maintenance dredged material in the Laguna Madre prompted a lawsuit in September 1994 by the National Audubon Society and other local environmental groups against the USACE. The lawsuit ultimately led to a settlement wherein the USACE agreed to develop a supplemental environmental impact statement (SEIS) for the Laguna Madre reach of the GIWW. The USACE and the Texas Department of Transportation (TxDOT is the non-Federal sponsor of the project) committed to identifying solutions that allowed continued navigation and addressed environmental concerns (Texas Department of Transportation, 2000). To this end the USACE and TxDOT developed an interagency coordination team for the Laguna Madre. This interagency team consists of members from the USACE, TxDOT, EPA, USFWS, National Marine Fisheries Service, TGLO, TCEQ, TPWD, and the Texas Water Development Board. The team's charter included objectives to identify the environmental concerns associated with the GIWW in the Laguna Madre and to develop scopes of work to address these environmental concerns. The team began work in February 1995 and continues to the present. Currently, the draft SEIS is being finalized. If past maintenance dredging activities have adversely affected seagrasses, it is hoped that, through the work of the team, solutions will be found and recolonization of affected areas will take place.

In the lower Laguna Madre, nutrient input from the Arroyo Colorado is becoming a concern. Onuf (1996b) found algal biomass to be three times greater near the Arroyo Colorado as compared to other areas of the Laguna Madre. The Arroyo Colorado is the only major drainage into the lower Laguna Madre. This drainage receives much of the municipal, industrial, and agricultural wastewater generated in the lower Rio Grande valley. Consequently, the tidal segment of the arroyo does not meet the dissolved oxygen standard for aquatic life use, and nutrient concentrations exceed screening levels. Currently, to address the water-quality problems in the arroyo, a total maximum daily load (TMDL) assessment is being conducted by the TCEQ. Once the TMDL is completed and implemented, reduction in nutrient loading might benefit lower Laguna Madre.

Recently, propeller scarring in seagrass beds by recreational boaters was found to be significant in some areas of the upper Laguna Madre (Dunton and Schonberg, 2002). Because propeller scarring has been implicated in seagrass bed fragmentation and loss (Zieman, 1976), the Corpus Christi Bay National Estuary Program funded informational signs currently being installed at many upper Laguna Madre boat ramps. These signs are 1.2 m wide and 2.5 m long (4 ft by 8 ft) and are designed to educate boaters about the importance of seagrasses as well as give boat operators information to help minimize damage from propellers. At the southern extremity of upper Laguna Madre, TPWD designated a mandatory no-motor zone in 2000, and adjacent to it, Padre Island National Seashore has designated a voluntary no-motor zone. In 2005, TPWD will evaluate the effectiveness of the no-motor zone and decide whether to continue the designation.

Acknowledgments

The author thanks Beau Hardegree, U.S. Fish and Wildlife Service, Ecological Services Division, Corpus Christi, Tex., for assistance in preparing the sections on mapping and monitoring needs and restoration and enhancement opportunities.

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Texas Coastal Bend

By Warren Pulich, Jr.[1]

Background

This vignette summarizes seagrass distribution data as of 1996 for the "Texas Coastal Bend" region near Corpus Christi (specifically the Nueces and Mission-Aransas estuaries). It also provides a review of trends in seagrass distribution over 40 yr for parts of the Nueces estuary and correlates these historical trends with natural and human-induced stressors that affect submerged vegetation.

Land use around the Texas Coastal Bend watershed (fig. 1) is dominated by southern Texas farmlands (row crops), pastures, and brushy rangeland. The population of the area (approximately 560,000 in 2000) is relatively sparse, despite having grown from 475,000 in 1980 (U.S. Census Bureau, *www.census.gov*). Urban development has been minor, with Corpus Christi the only city in the area with population above 20,000 (300,000 in 2000). The Port of Corpus Christi, which ranks sixth in the nation, and agriculture have been the mainstays of the local economy. Within the last 20 yr, however, accelerating resort, tourist, and retirement developments have been occurring along the barrier islands (Mustang and north Padre Islands), as well as in the Ingleside area and Blackjack Peninsula of the mainland north of Corpus Christi. This development pressure has put increasing stress on the submerged wetlands of the region (Coastal Bend Bays and Estuary Program, 1998).

Historically, waterborne shipping and hydrocarbon-extraction activities have been the major industries impacting the Texas Coastal Bend bays. More than any other feature, navigation channels that form extensive networks and are lined with dredged material have altered the Texas Coastal Bend bays, and these channels have locally affected seagrass habitats in the system. Approximately 110 km^2 (42.47 mi^2) of channels and dredged material were mapped in the early 1970s within the Corpus Christi area (Brown and others, 1976). Among the major channels are the Gulf Intracoastal Waterway (GIWW), Corpus Christi Pass and Aransas Pass ship channels, Lydia Ann Channel, La Quinta Channel, and Aransas Channel. Major boat harbors include the Inner Harbor and Turning Basin at the Port of Corpus Christi, Port Ingleside, and Conn Brown Harbor at Aransas Pass. Many small intrabay channels and canals were dredged across seagrass flats in Redfish Bay and Harbor Island, as well as the back side of Mustang Island, for access to shallow areas for oil and gas exploration. Dredged materials were dumped along the channels, forming upland mounds that locally blanketed seagrass habitats. Most of these channels were constructed before 1958, but dredging of the GIWW through the western Redfish Bay area along the mainland was only completed in 1959–60, and impacts from this project were documented in a trend analysis study by Pulich and others (1997). Numerous small industrial marinas and residential boat basins, built after 1960, occur along the shoreline along the GIWW.

Scope of Area

The Texas Coastal Bend area comprises two separate estuarine systems, the Nueces estuary system, formed by drainage from the Nueces River watershed, and the Mission-Aransas estuary system to the north, which receives inflow from the Mission and Aransas Rivers (fig. 1). These systems provide annual median freshwater inflows of 348,000 acre-ft (Nueces) and 318,000 acre-ft (Mission-Aransas) per year to the two estuaries. The area is a subtropical, semiarid region, with average annual rainfall of about 78 cm (30 inches) but with evaporation usually exceeding 180 cm (70 inches) per year for the Corpus Christi area.

The Nueces estuary system includes several distinct segments: north Corpus Christi Bay, Oso Bay, Nueces Bay, Redfish Bay, the Harbor Island complex, and the bay side of Mustang Island (fig. 2). The separate Mission-Aransas estuary system extends from near Rockport, along the bay side of San Jose Island, west to the Aransas and Mission River deltas, and north to the Aransas National Wildlife Refuge. This estuary includes Aransas Bay proper, Copano Bay, Port Bay, St. Charles Bay, and Mesquite Bay.

While current seagrass status was determined for both estuaries, only trends for the following separate segments of Nueces estuary were analyzed.

Redfish Bay

Redfish Bay lies north of Corpus Christi Bay proper and southwest of Aransas Bay and parallels the mainland

[1] River Systems Institute, Texas State University–San Marcos.

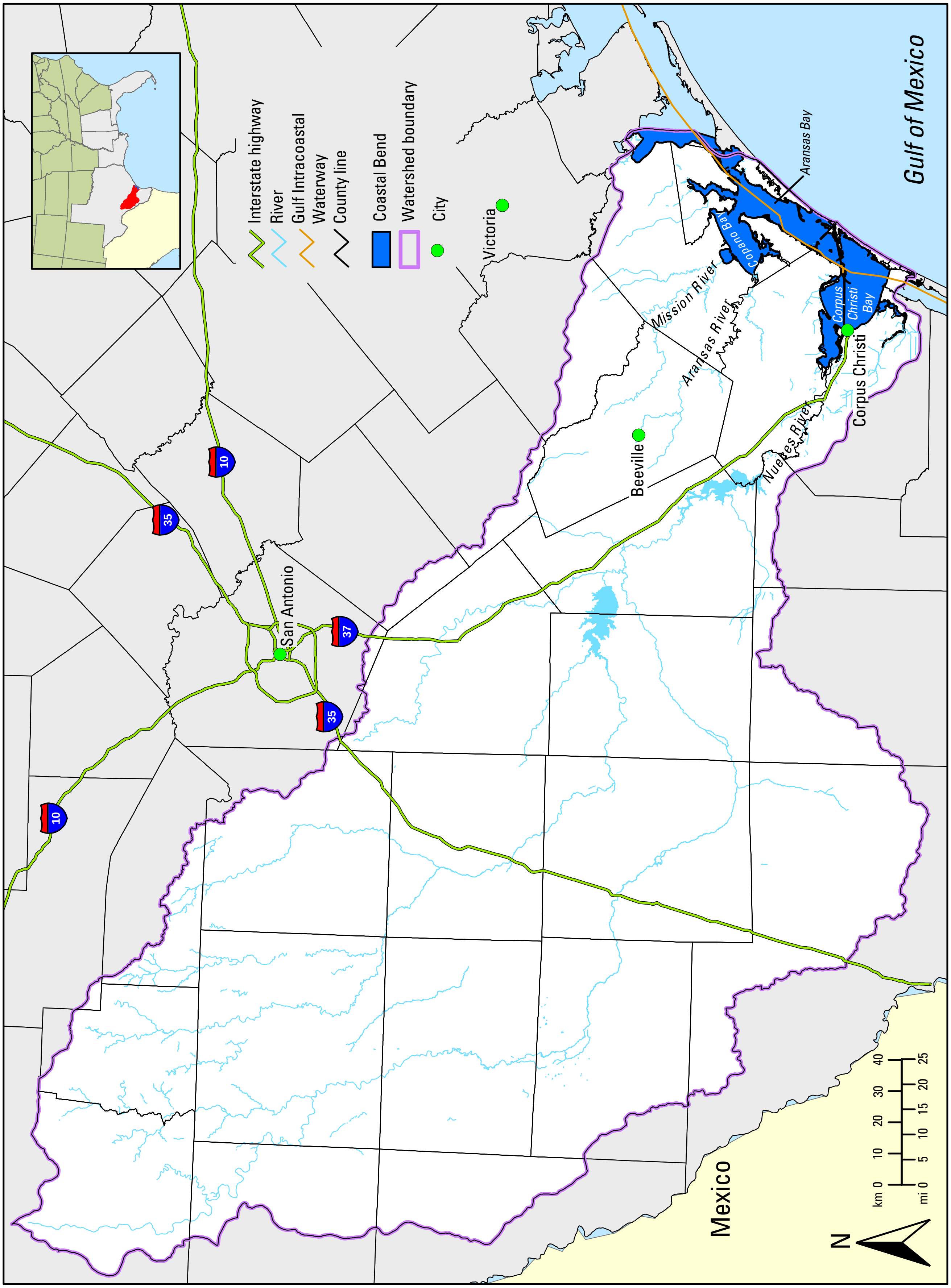


Figure 1. Watershed for Texas Coastal Bend region.

area between Ingleside and Rockport. The GIWW forms its western border, separating it from the mainland. This large, very shallow bay is protected from prevailing southeast winds by numerous saltmarsh islands, oyster reefs, and shoals and is well known for its extensive seagrass beds, particularly turtle grass (*Thalassia testudinum*) and manatee grass (*Syringodium filiforme*) beds. Salinities are typically polyhaline, and water clarity is moderately high.

Harbor Island

Harbor Island forms a flood-tidal delta complex that is large, triangular, and shallow and that is bordered and bisected by dredged channels and dredged-material disposal sites. It is bounded on the east by Lydia Ann Channel, which connects to the GIWW in Aransas Bay; on the west by Redfish Bay; on the north by Aransas Bay; and on the south by the Corpus Christi Channel, which separates it from Mustang Island. Bisecting this large bay-delta complex is the Aransas Channel. In addition, there are several smaller and tributary intrabay channels. This bay and tidal delta complex (approximately 4,047 ha, or 10,000 acres) represents the most extensive and northernmost estuarine tropical wetlands on the Texas coast and is composed of black mangrove (*Avicennia germinans*) and *Spartina* marsh, seagrass beds, and oyster reefs.

Mustang Island

This region of eastern Corpus Christi Bay, situated in the lee of Mustang Island, is well protected from prevailing southeasterly winds. In combination with the generally polyhaline to euhaline clear waters and sandy sediments, the habitat is conducive to supporting extensive seagrass beds and salt marsh.

Nueces Bay

Nueces Bay, a shallow, secondary bay in the upper estuary that receives direct inflow from the Nueces River, empties into the western portion of Corpus Christi Bay. Its salinity regime fluctuates from oligohaline to hypersaline levels, depending on inflows and evaporation. Along with its generally muddy waters, the salinity regime makes Nueces Bay a very dynamic habitat for seagrasses.

Methodology Employed To Determine and Document Current Status

A complete inventory of seagrass beds in the Texas Coastal Bend region, including for the first time the Mission-Aransas estuary, was performed for the Coastal Bend Bays and Estuaries Program (CBBEP, formerly the Corpus Christi Bay National Estuary Program) by Pulich and others (1997). Mapping was based on field surveys and photointerpretation of true color, aerial photography (1:24,000 scale) taken in November 1994. This inventory (Pulich and others, 1997) employed standard seagrass mapping protocols based on photointerpretation of 1:24,000-scale positive transparencies according to Ferguson and others (1993) and Dobson and others (1995). Seagrass distribution was determined from Kodak Aerocolor 2445 (color aerial film) photography flown on November 21, 1994, by aerial photography staff of the Texas Department of Transportation (TxDOT) in coordination with Texas Parks and Wildlife Department (TPWD), Coastal Studies Program. The mission was flown 2 days after a fall cold front when the weather was clear and winds calm. Tidal conditions were slightly higher than average annual water height. Large format (23 cm by 23 cm, or 9 inch by 9 inch) aerial film was exposed at an airplane altitude of 3,048 m (10,000 ft) to provide 1:24,000 scale.

Large format positive transparencies were photointerpreted for seagrass and other submerged features by using a backlit light table and 6-power magnifying lens onto 2 mil, transparent Mylar® sheets. The Mylar® overlays were scan digitized, and seagrass polygons were imported into ArcInfo geographic information system (GIS) software (ESRI, Redlands, Calif.). The digitized vector polygons were georeferenced to standard Universal Transverse Mercator map projection coordinate system by using Global Positioning System (GPS) based ground control points (at least eight control points per photo) and by using second-order rectification equations.

Classification Scheme

Mapping efforts resulted in discrimination of two seagrass bed types, continuous and patchy, based on morphology. Continuous beds represented extensive, homogeneous seagrass, with essentially 100% cover of plant shoots over 0.05-ha (0.12-acre) beds. Patchy beds were broken up into patches of plants interspersed with bare sediment patches. The size of these patches determined whether beds were mapped as patchy or continuous and in some cases whether an isolated patch of seagrass would be mapped at all. At 1:24,000 scale, the recognized standard for seagrass mapping, only features larger than about 0.05 ha (0.12 acre) in actual size can be accurately photointerpreted, and isolated patches smaller than 0.05 ha (0.12 acre) are omitted. Minimum mapping unit is also limited by water clarity in the aerial photography. Thus the definition of a patchy seagrass bed was a grouping of small patches of seagrass, between 0.05 and 0.1 ha (0.12–0.25 acre) in dimension, with equally small, open, bare sand areas separating them. Patchy beds often represented seagrass areas that are subject to impact from fetch, hydraulics, or mechanical disturbances such as dredged-material deposition, boating impacts, or shoreline development.

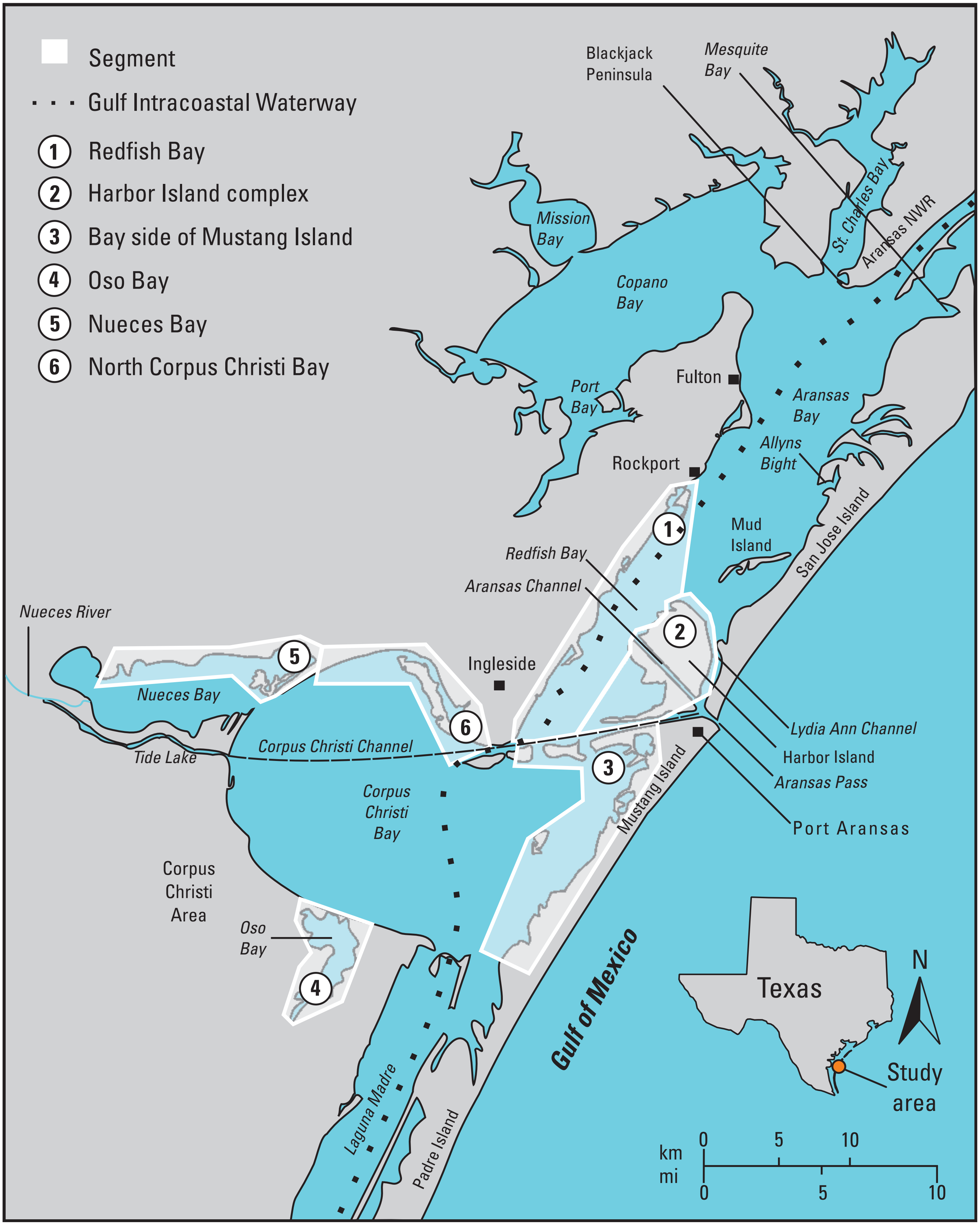


Figure 2. Scope of area for the Texas Coastal Bend seagrass vignette.

Attempts to separate seagrass beds into the density categories of sparse and dense produced variable results. Differences in water depth and clarity, as well as variable biomass between species, produced inaccuracies. Thus, although density categories were not used, ground surveys at extensive GPS stations allowed species distributions to be established. Percent frequency of species occurrence was calculated based on the total number of samples at GPS stations containing seagrass. From the spatial pattern of GPS points, the overall distribution of species was approximated from map coverages in each bay.

All groundtruthing surveys were conducted within 1.5 yr of acquiring the 1994 photography; thus, the trend analysis is considered accurate through the 1996 period. These surveys were often done from an airboat, which allowed access to very shallow grassflats. A GPS unit was used, which allowed real-time differential correction techniques to be applied. The seagrass distribution maps produced had a locational accuracy of ±3 m (10 ft).

Methodology Employed To Analyze Historical Trends

Seagrass trend analysis for the bay systems in the Texas Coastal Bend has received limited attention. Brown and others (1976) compiled the first quantitative wetland maps, including seagrass distribution, for this region from examination of 1950s black and white photography (scale 1:24,000) by Tobin Surveys, Inc. (San Antonio, Tex.). Biologists from TPWD also performed field surveys during the 1960s and 1970s of the San Antonio/Aransas/Corpus Christi/Laguna Madre areas, and McMahan (1965–67) and West (1971) produced a series of hand-drawn, small scale (>1:125,000) maps of these bays. Such mapping studies are very useful for general information on historical locations of seagrass although not reliable for quantitative evaluation of seagrass changes at 1:24,000 scale. A later mapping study by White and others (1978) at University of Texas-Bureau of Economic Geology provided data for the bay side of Mustang Island in the mid-1970s. Seagrass acreage was derived by quantitative photointerpretation of 1:24,000-scale color infrared photography acquired in 1974 by National Aeronautics and Space Administration (NASA) Johnson Space Center (JSC) (Houston, Tex.).

A complete study of seagrass bed trends for Corpus Christi and Redfish Bays, as well as for Harbor Island, in the Nueces estuary was performed for the CBBEP by Pulich and others (1997). Historical aerial photography at similar scales (at least 1:24,000 scale) was analyzed and compared for the late 1950s, 1975, and 1994 time periods. Historical changes and trends were established at 1:24,000 scale, including both spatial (geographic) locations and quantitative seagrass acreages lost or gained.

For the CBBEP study (Pulich and others, 1997), seagrass distribution maps from the three different time periods were produced by photointerpretation as described earlier. Historical photographic missions included 1956 and 1958 series black and white Tobin surveys (San Antonio, Tex.), scale 1:24,000; early October 1975 NASA-JSC (Houston) color-infrared series, scale 1:24,000; and early November 1989 NASA-Ames Research Center, California, color-infrared series, scale 1:63,000. Data for the recent (1994–96) time period and classification of seagrass features were previously described. For 1975 photos, a Bausch & Lomb stereo zoom transfer scope was used to transfer and register seagrass features to the appropriate 7.5 min U.S. Geological Survey (USGS) planimetric base map. For the 1950s photointerpretation, features were traced directly off of the rectified black and white prints which had been mosaicked, producing a USGS quadrangle-size sheet (50 by 58 cm (19.75 by 22.75 inches), similar to the 7.5 min planimetric base maps produced in 1975). Except for trend analysis in Nueces Bay, 1989 photography was used only as visual reference material.

After digitization, all digital map coverages for the 1950s, 1975, and 1994 time periods were entered into an ArcInfo GIS database, where trend analysis was performed by using GIS change analysis. Change maps were determined by postclassification change detection from the seagrass distribution maps representing the different time periods (Dobson and others, 1995). Digitized seagrass coverages were spatially correlated with a variety of ancillary spatial data on such parameters as channel dredging, spoil disposal areas, bathymetry, and locations of shoreline developments. Simple GIS overlay and buffer analysis techniques were used to correlate environmental features with seagrass distributions.

Trend analysis for the bayside area of Mustang Island (eastern shoreline of Corpus Christi Bay) was conducted by using different data sources from those used for Redfish Bay and Harbor Island. Numerical acreage values for the 1950s (as derived from 1956/58 Tobin photography) were taken from the study report by Brown and others (1976). Map data for 1974 were digitized and calculated from the printed georeferenced maps in White and others (1978), and the 1994 map data were from the study by Pulich and others (1997).

Trend analysis for Nueces Bay seagrass was conducted over different time periods. The only historical photography that could be located in which seagrass was visible was from the 1961 and 1989 periods. The 1961 photography was 1:20,000-scale black and white photographs taken by the U.S. Department of Agriculture Soil Conservation Service, and the 1989 photographs were high-altitude color-infrared from NASA-Ames (see previous paragraph). Seagrass coverage was photointerpreted from both sets of photographs and transferred to 1:24,000-scale USGS quadrangle maps of the Nueces Bay area by using zoom transfer techniques.

No historical trends were analyzed for the Mission-Aransas estuary.

Status and Trends

System Summary

Nueces Estuary System

In 1994, distribution of seagrass beds in the Nueces estuary system was as follows: Nueces Bay, 294 ha (726 acres); Oso Bay, 483 ha (1,193 acres); northshore Corpus Christi Bay, 290 ha (717 acres); bay side of Mustang Island, 1,503 ha (3,714 acres); Redfish Bay, 3,644 ha (9,004 acres); and the Harbor Island complex, 2,064 ha (5,100 acres) (fig. 3). The Nueces estuary system does not include the Laguna Madre north of John F. Kennedy Causeway. Total seagrass area, 8,278 ha (20,455 acres), was 9.3% of all Texas seagrasses. Continuous seagrass amounted to 4,194 ha (10,363 acres), while patchy beds made up 4,084 ha (10,092 acres).

Mission-Aransas Estuary

The 1994 distribution of seagrass beds for the Mission-Aransas estuary system (3,107 ha, or 7,677 acres total) was as follows: Mission/Copano Bays, 473 ha (1,169 acres); Port Bay, 724 ha (1,789 acres); St. Charles Bay, 348 ha (860 acres); Mesquite Bay, 236 ha (583 acres); and Aransas Bay proper comprising mostly Mud Island and bay side of San Jose Island, 1,326 ha (3,277 acres) (fig. 4). Continuous seagrass amounted to 1,691 ha (4,178 acres), while patchy beds made up 1,416 ha (3,499 acres).

Status Summary

In 1994, the Nueces and Mission-Aransas estuary systems of the Texas Coastal Bend contained a combined 11,385 ha (28,132 acres) of seagrass, of which 5,885 ha (14,542 acres) were continuous seagrass beds, while 5,500 ha (13,590 acres) were patchy (fragmented) seagrass beds. Both systems accounted for 12.8% of all Texas seagrass beds, by far the largest amount of seagrasses found in Texas bay systems outside of the Laguna Madre. Seagrass species found in the Texas Coastal Bend study area are turtle grass, shoal grass (*Halodule wrightii*), wigeon grass (*Ruppia maritima*), manatee grass, and star grass (*Halophila engelmannii*).

Primarily, seagrass distribution in the Texas Coastal Bend parallels the extent of mesohaline, shallow-water depth zones of less than 1.5 m (5 ft). Secondarily, it tends to follow the inflow and turbidity gradients in the bays. Seagrass is scarce in upper bays, where direct inflows are high and salinities are usually below 15 ppt, compared to areas in the lower estuary, where inflows are low, salinities are above 20 ppt, and depth is uniformly shallow. Not only must salinity be at appropriate levels for seagrass, but there must also be protection from physical disturbance factors, dredging, shoreline erosion, and the heavy wave action resulting from wind-induced fetch. This requirement for protection is critical in various parts of the Texas Coastal Bend. Seagrass beds on the lee side of the barrier islands (Mustang, Harbor, and San Jose Islands) show considerable expansion from this protective effect, whereas beds in open parts of Corpus Christi, Copano, and Aransas Bays tend to develop as fringe bands because of exposure stress. Beds in more developed areas (e.g., Redfish Bay) show a combination of stress from this impact, as well as from channelization and dredging.

Trends Summary

Redfish Bay

As mentioned previously, the Redfish Bay segment of Nueces estuary contains the most extensive, pristine seagrass beds on the Texas coast outside the Laguna Madre, with 1994 acreage of about 3,644 ha (9,000 acres) (fig. 3). Redfish Bay also contains the largest abundance of all five seagrass species outside the lower Laguna Madre. Historical distributions in this region for the 1950s and 1975 are presented in figures 5 and 6, respectively. Numerical results for Redfish Bay are presented in table 1. Figure 5 shows the 1950s coverage,

Table 1. Summary of total seagrass change for Mustang Island, Redfish Bay, and Harbor Island segments from the late 1950s to the mid-1970s and then to 1994.

Time period	Mustang Island	Redfish Bay	Harbor Island
Late 1950s	1,030 ha (2,545 acres)	4,180 ha (10,328 acres)	1,199 ha (2,962 acres)
Mid-1970s	1,128 ha (2,786 acres)	3,985 ha (9,847 acres)	2,211 ha (5,463 acres)
1950s–1970s net	+98 ha (+241 acres)	-195 ha (-482 acres)	+1,012 ha (+2,500 acres)
Percent change	+9.5%	-4.7%	+84.4%
Time period			
Mid-1970s	1,128 ha (2,786 acres)	3,985 ha (9,847 acres)	2,211 ha (5,463 acres)
1994	1,503 ha (3,713 acres)	3,644 ha (9,004 acres)	2,064 ha (5,100 acres)
1970s–1994 net	+375 ha (+927 acres)	-341 ha (-843 acres)	-147 ha (-363 acres)
Percent change	+33.3%	-8.6%	-6.7%

which represents seagrass distribution prior to dredging the GIWW along the mainland side of Redfish Bay. Between 1958 and 1975, total seagrass showed a slight decrease (4.7%) for the area. Between 1975 and 1994, seagrass coverage in Redfish Bay then decreased significantly (by 8.6%). Overall results indicate that a net loss of 536 ha (1,324 acres) (13%) of seagrass occurred between the 1950s and 1994 in Redfish Bay. Table 2 shows that this change was accompanied by substantial loss (48%) of continuous beds and a progressive increase (88%) in patchy (i.e., fragmented) seagrass beds.

Harbor Island

Historical seagrass distributions in the Harbor Island segment of the Nueces estuary system were determined for 1958 (fig. 5) and 1975 (fig. 6) (also see table 1). For the Harbor Island complex (north and south parts combined), there was a large increase in seagrass from the 1950s to 1975, especially of continuous seagrass beds (all shoal grass or wigeon grass). There followed a measurable decrease of about 6% (147 ha, or 363 acres) between 1975 and 1994 (fig. 6 and table 1). Overall results indicate that there was a net gain of 77% (866 ha, or 2,140 acres) seagrasses in the Harbor Island area that occurred between 1958 and 1994. This increase was mostly patchy beds (562 ha, or 1,389 acres) and lesser amounts of continuous beds (304 ha, or 751 acres) (table 2).

Mustang Island

On the bay side of Mustang Island, seagrass increased by 98 ha (241 acres) between 1958 and 1974, as reported by White and others (1978). That study concluded that, after the 1950s drought ended, higher bay water (sea level) greatly promoted the spread of seagrass onto bare tidal flats in the Harbor Island complex and on the bay side of Mustang Island. In the Pulich and others (1997) study, an additional 33% (375 ha, or 927 acres) increase in seagrass area was measured between 1974 and 1994, as shown in figures 3 and 6 and table 1. Gains in seagrass occurred not only along the upland margins of the wind-tidal flats in some areas but also along the bayward margins of the flats in deeper areas (fig. 3). In 1994, there were 917 ha (2,266 acres) of continuous seagrass beds and 586 ha (1,448 acres) of patchy seagrass beds in the Mustang Island segment.

Nueces Bay

Figure 3 shows the 1994 mapped seagrass distribution for Nueces Bay; numerical data for other time periods are reported here in the text. From 1961 photography, only 79 ha (195 acres) of seagrass (reported to be shoal grass by McMahan, TPWD, oral commun.) were mapped for Nueces Bay. After 1961, there was essentially a 100% loss of seagrass

Table 2. Changes in continuous and patchy seagrass beds in Redfish Bay and Harbor Island segments between late 1950s to mid-1970, and mid-1970s to 1994.

Time period	Redfish Bay		Harbor Island	
	Continuous	Patchy	Continuous	Patchy
Late 1950s	3,100 ha (7,660 acres)	1,080 ha (2,669 acres)	1,016 ha (2,511 acres)	182 ha (450 acres)
Mid-1970s	2,969 ha (7,337 acres)	1,016 ha (2,511 acres)	1,776 ha (4,389 acres)	436 ha (1,077 acres)
1950s–1970s net	-131 ha (-324 acres)	-64 ha (-158 acres)	+760 ha (+1,878 acres)	+254 ha (+628 acres)
Percent change	-4.2%	-5.9%	+74.8%	+139.6%
Mid-1970s	2,969 ha (7,337 acres)	1,016 ha (2,511 acres)	1,776 ha (4,389 acres)	436 ha (1,077 acres)
1994	1,669 ha (4,124 acres)	1,976 ha (4,883 acres)	1,320 ha (3,262 acres)	744 ha (1,838 acres)
1970s–1994 net	-1,300 ha (-3,212 acres)	+960 ha (+2,372 acres)	-456 ha (-1,127 acres)	+308 ha (+761 acres)
Percent change	-43.8%	+94.5%	-25.7%	+70.6%

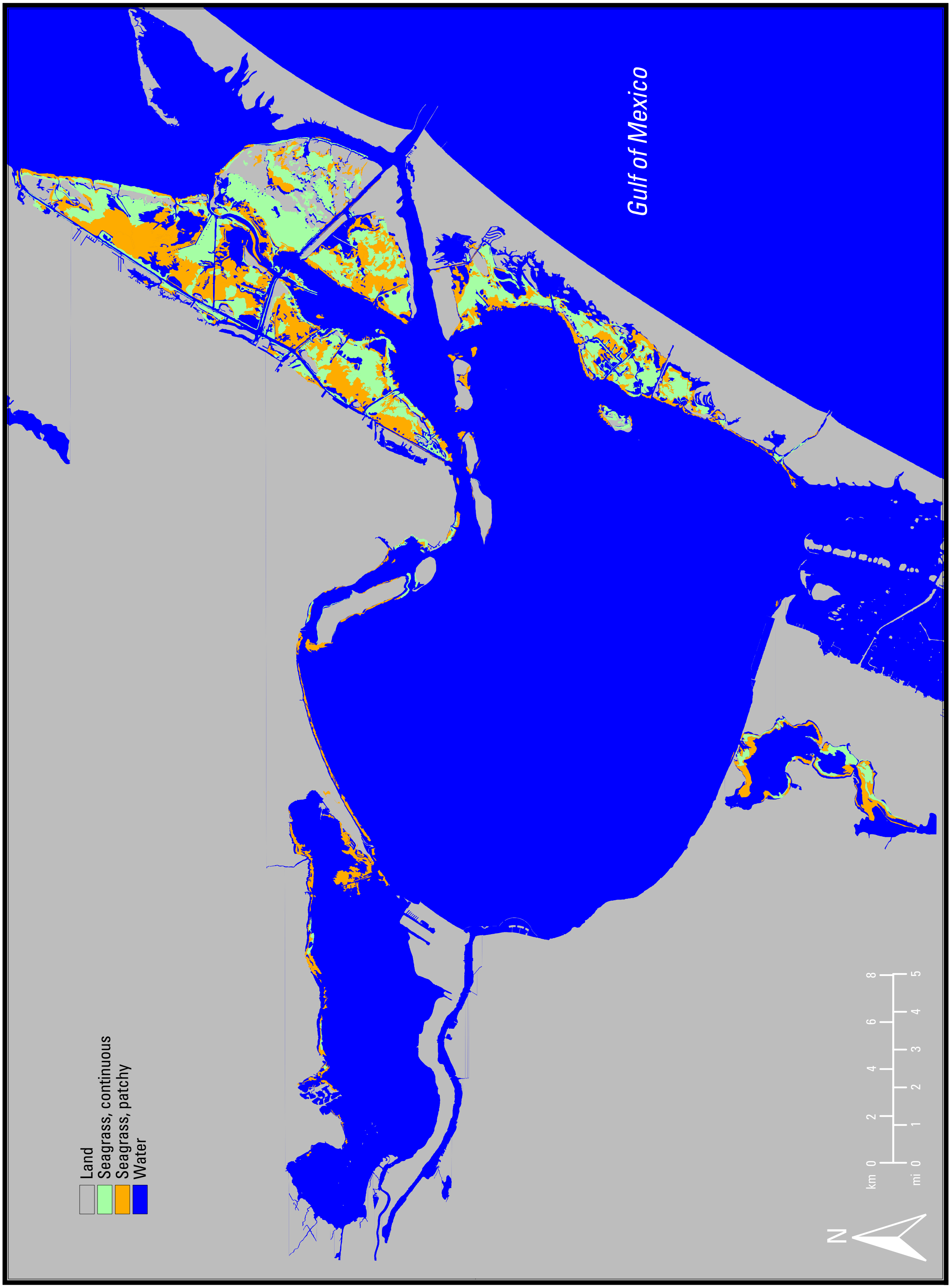


Figure 3. Distribution of seagrass in the Nueces estuary system, 1994.

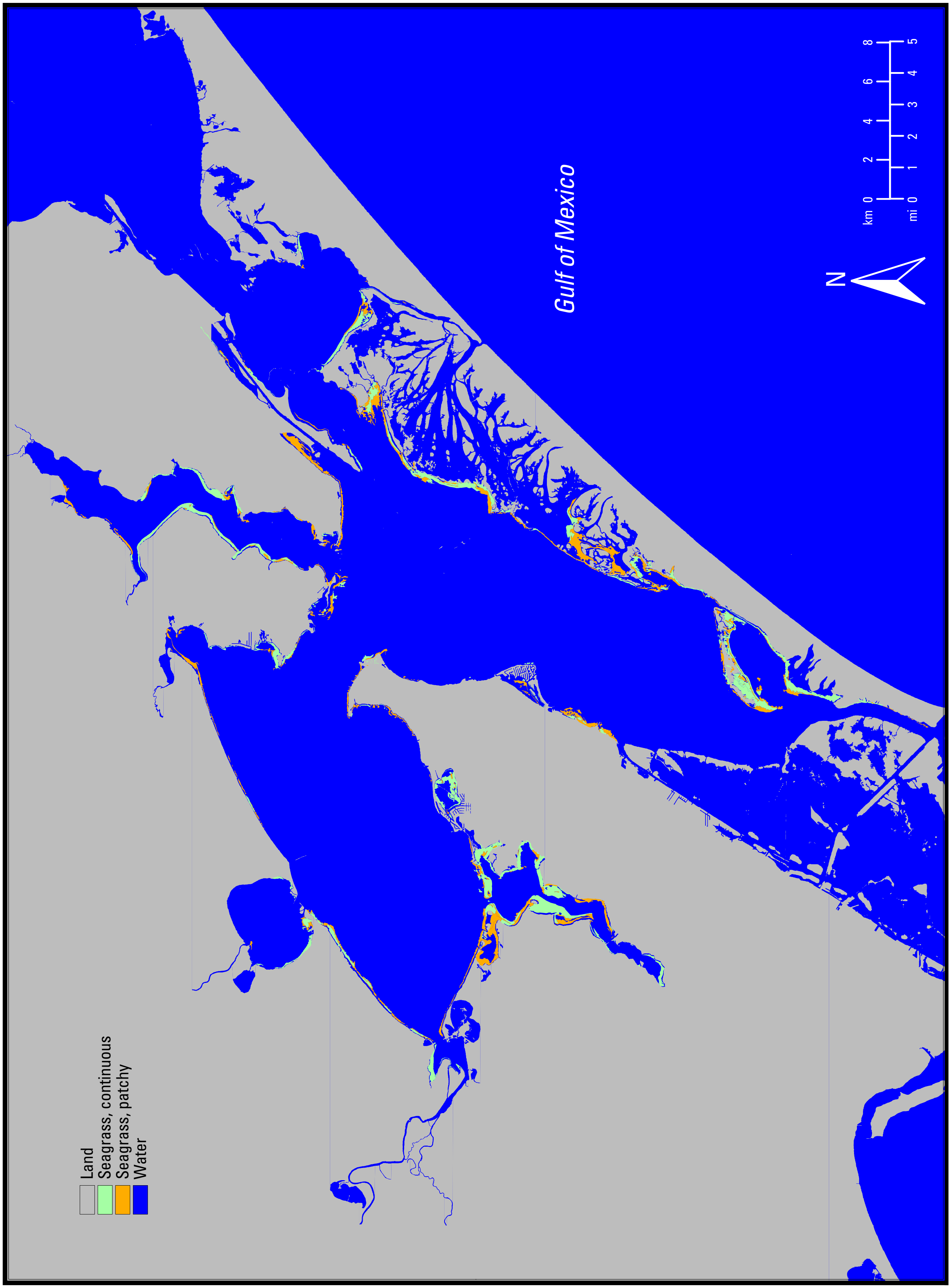


Figure 4. Distribution of seagrass in the Mission-Aransas estuary system, 1994.

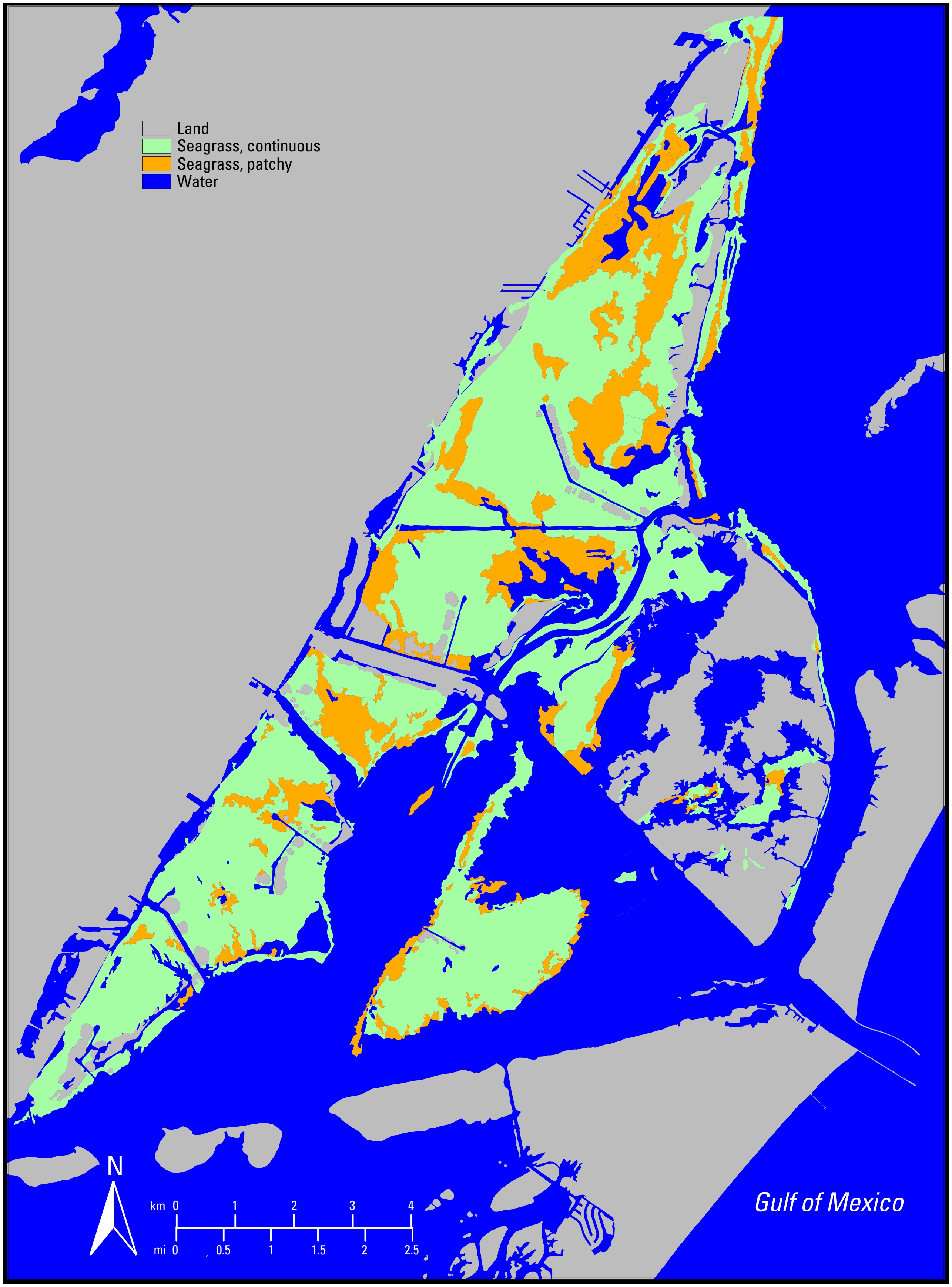


Figure 5. Distribution of seagrass in Redfish Bay and Harbor Island, late 1950s.

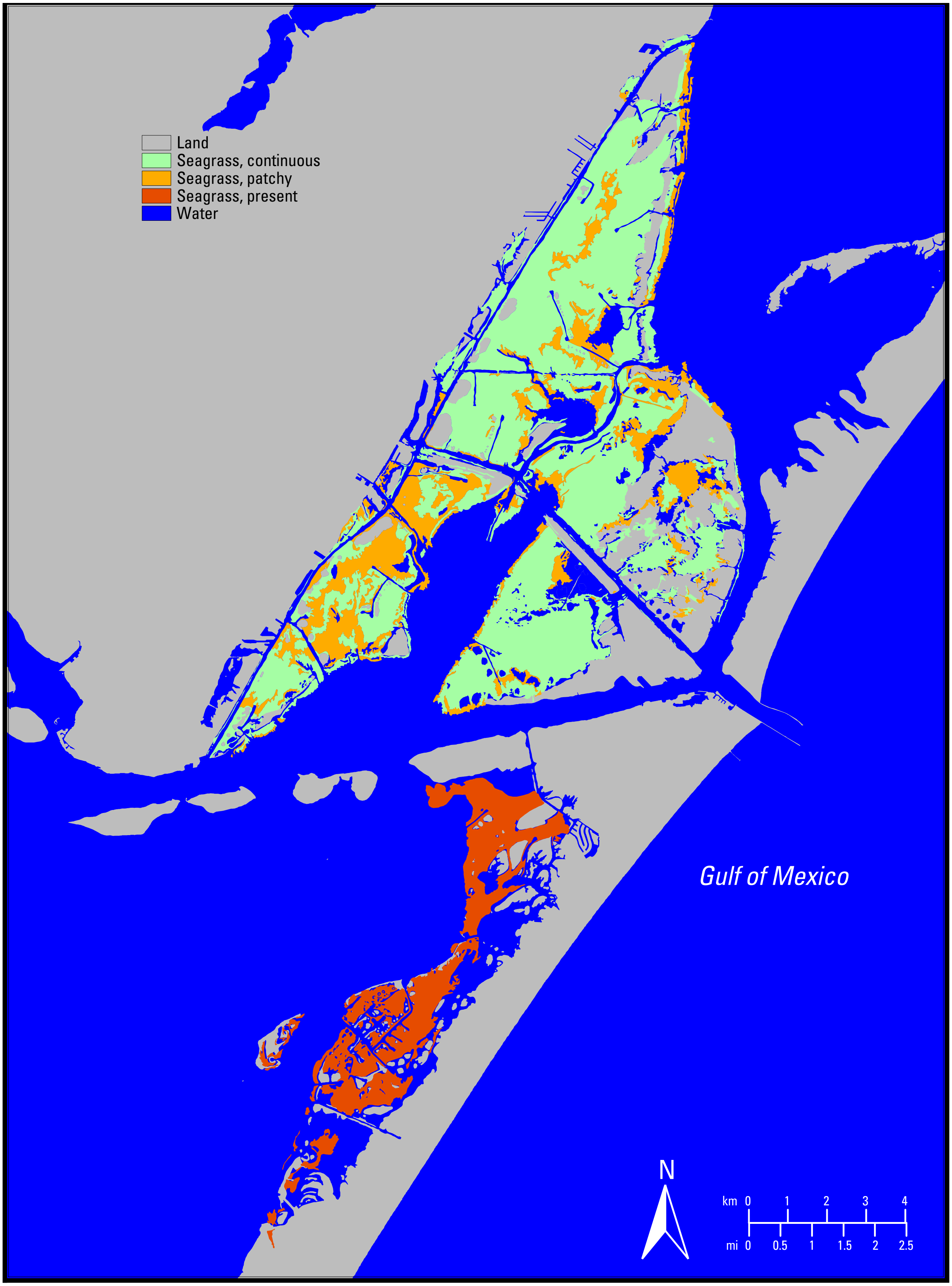


Figure 6. Distribution of seagrass in Redfish Bay and Harbor Island, mid-1970s.

beds by the late 1960s, coinciding with Hurricane Beulah in 1967 (McGowen, 1971). During the 1970s, only fringe shoreline patches of mostly wigeon grass were observed by TPWD biologists during sampling trips to Nueces Bay (Richard Harrington, TPWD, Corpus Christi, oral commun.). By the mid-1980s, major shoal grass beds had reappeared (Harrington, TPWD, oral commun.), reaching 200 ha (494 acres) by 1989 and 294 ha (726 acres) by 1994, as determined by this mapping study and representing an increase of 94 ha (232 acres) between 1989 and 1994. Percent changes in seagrass acreage were as follows: 1961 to about 1970, essentially 100% loss of shoal grass; 1970 to the early 1980s, small amounts of wigeon grass and no shoal grass observed; 1961 to 1989, 112% gain in seagrass compared to 1961; and 1989 to 1994, a 47% gain of shoal grass over 1989. These trend data indicate that Nueces Bay is an unusually dynamic seagrass area.

Texas Coastal Bend as a Whole

For the combined Redfish Bay, Harbor Island, and Mustang Island segments of the Nueces estuary system, net seagrass bed area may appear fairly stable over 40 yr, but this conclusion ignores the dynamic cycles in localized seagrass bed changes. Spatial analysis reveals that seagrass losses and gains occur simultaneously. Overall, a net increase occurred in total area for the system between 1958 and 1994 (net gain 802 ha, or 1,981 acres) (table 1). This gain was due primarily to the large expansion of seagrass into the Harbor Island complex between the late 1950s and 1975 (84% or 1,012 ha) and along Mustang Island (375 ha, or 926 acres) between 1974 and 1994. The simultaneous 13.3% decrease (536 ha, or 1,324 acres) and accompanying bed fragmentation in seagrass beds noted for Redfish Bay over the period from the late 1950s to 1994, however, suggested that seagrass conditions should be interpreted with caution for the entire system.

Causes of Change

Redfish Bay

Changes Related to Dredging and Channel Construction

Extensive, direct seagrass losses associated with dredged-material deposition occurred between 1958 and 1975 as shown by the network of dredged channels and disposal sites created in Redfish Bay. The majority of the losses were caused by seagrass burial principally related to construction of the GIWW through Redfish Bay and the resulting disposal of dredged material directly into seagrass areas; however, at some distance from disposal sites, seagrass beds were often converted to sparsely vegetated (patchy) beds. Odum (1963) reported a decrease in seagrass productivity and an imbalance of respiration over photosynthesis in Redfish Bay in summer 1959 that he attributed to dredging of the GIWW. Recovery was noted the following year when growth was exceptional, and he suggested that released nutrients may have stimulated growth (Odum, 1963).

Many smaller, intrabay channels had been dredged across seagrass areas in Redfish Bay prior to 1958. The initial impacts on seagrass beds from these channels and disposed dredged material have not been determined. Of interest is that between 1958 and 1975 there were gains in seagrass along the channels as seagrasses spread over the margins of these reworked, submerged dredged materials. Between 1975 and 1994, however, there were additional losses along some of these intrabay channels, apparently from maintenance dredging or boat traffic using the channels.

From GIS analysis (Pulich and others, 1997), 795 ha (1,964 acres) of seagrass were lost between 1958 and 1994 because of dredged-material deposition and channel impact zones. Concomitantly, 407 ha (1,006 acres) were gained, for a net loss of 388 ha (959 acres). Since most of this loss occurred in the Redfish Bay portion of the regional complex, it is interesting to compare this number to the total seagrass lost in Redfish Bay alone, which was 536 ha (1,324 acres). Thus it is evident that the 388 ha (959 acres) lost from channel dredging accounts for approximately three-fourths (72%) of the Redfish Bay total, which is a substantial impact.

Changes Related to Bathymetry

In the Redfish Bay area, Hurricane Carla (1961) apparently did not measurably affect seagrasses (see Pulich and others, 1997), substantiating that the main effect on the middle Texas coast from this storm was high tides and not erosion, wind, or fetch damage. This tidal effect is a great contrast to the destructive effect of Hurricane Carla on the seagrasses in Galveston Bay (Pulich and White, 1991).

Changes Related to Boat Propeller Scarring

Damage from boat traffic (i.e., propeller scarring) was noted extensively in a number of areas, especially where seagrass losses occurred. Most of the 1994–95 data came from groundtruthing and field surveys since propeller scars are features beyond the normal limit of resolution (0.05 ha or 0.12 acre) of 1:24,000-scale photographs. It was obvious that certain shallow areas (water depth less than 0.6 m, or 2 ft) were greatly affected by propeller disturbance. The affected areas tended to be the large, expansive areas in north Redfish Bay (near Hog Island and Estes Cove) and near Harbor Island (south side) where boaters attempted to take "short cuts" between favorite fishing areas in grassflats and residential developments or dredged channels. Comparison of 1994 and 1975 photography revealed obvious physical scarring of seagrass beds in this area of Redfish Bay over the 20-yr period. Since the dominant species in these seagrass beds is turtle grass, a sensitive, slow-growing climax species, this scarring may be more severe and long lasting than at other sites in the bay. Dunton and Schonberg (2002) quantitatively determined that scarring of seagrass beds within the boundaries of the CBBEP region ranged from 14.5% to 97.6%. The most severe scarring was found within north Redfish Bay in Nueces and Aransas Counties (91%–97.6%). Severe winter storms (cold fronts) and hurricanes in the Gulf of Mexico also exacerbate propeller scar damage to some seagrass beds as a result of high wave energy scouring the bottom.

Changes Related to Shoreline Development and Construction

The mainland (western) side of Redfish Bay is highly developed along the GIWW, whereas the Harbor Island (eastern) shoreline is essentially undisturbed. As has been observed in Australia (Cambridge and McComb, 1984), there is a high probability that impacts to seagrass may occur along the western margin from both industrial and residential activities. It is difficult, however, to directly quantify effects of this shorefront stress apart from GIWW dredging and channel impacts in the Redfish Bay area. Significant motorboat activity originating from these developments does in fact occur in this area and has contributed to large amounts of seagrass decline.

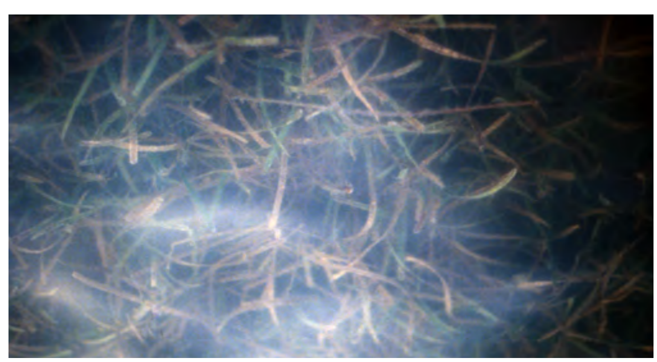
Changes Related to Light-attenuation Effects

From examination of 1996 National Pollutant Discharge Elimination System (NPDES) permit records from Texas Natural Resource Conservation Commission (TNRCC), it does not appear that direct wastewater discharges into seagrass beds regularly occur; however, nonpoint discharges from sites along the Ingleside-Aransas Pass-Rockport shoreline have the potential for contributing to higher nutrient loadings in that area. With the increase in shoreline marina developments along the GIWW from Aransas Pass to Rockport (about 517 ha, or 1,278 acres between 1975 and 1994), increased nonpoint-source runoff was predicted for this part of the bay (Pulich and others, 1997).

Because nutrient loadings may be hard to detect directly in the water column (see Dunton, 1996; Tomasko and others, 1996), nutrient concentrations in the water column may be poor indicators of incipient water-quality problems. Conversely, phytoplankton blooms, epiphytes, and macroalgae accumulations are considered good indicators of reduced water quality (National Oceanic and Atmospheric Administration, Office of Ocean Resources Conservation Assessment, 1995). Growth of these nuisance plants is stimulated by high dissolved nutrient levels, and the algae tend to shade and overgrow bottom-rooted seagrasses (Dennison and others, 1993). Recent reports from New England (Short and Burdick, 1996) and Florida (Tomasko and others, 1996) document the loss of seagrasses caused by dissolved nutrients leached from residential septic systems and carried into surrounding bay waters.

Changes Related to Macroalgae and Wrack Accumulation

Macroalgae mats (especially red and brown algae) in Redfish Bay were reported by Cowper (1978) to pose a potential light-shading stress to seagrasses. Epiphytes were also postulated by Pulich (1980) to reach potentially noxious levels for seagrass. Large accumulations of wrack and drift macroalgae were identified and mapped mainly from the western Redfish Bay system during the 1995 and 1996 field surveys. Often, rafts of red and green macroalgae appeared to be depositing in topographic depressions within the seagrass beds. Noxious, stagnant conditions produced over seagrass beds from dead, decomposing algae were also noted. The anoxia, accompanied by hydrogen sulfide, would be toxic to seagrass, in addition to the light limitation caused by shading from the algae plants. These conditions could also cause bacterial or fungal diseases to develop in the seagrass (Dennison and others, 1993; Short and Wyllie-Echeverria, 1996).



Localized hydrodynamics and circulation patterns in the western Redfish Bay area may contribute to nutrient and subsequent organic matter buildup. Extensive wrack and macroalgae deposits observed in the 1994 photography provide evidence that the north portion of Redfish Bay acts as a sink to trap material. Much of the western part of Redfish Bay may lie essentially out of the main circulation and tidal flow through the Corpus Christi or Aransas Bay system. Because of longer residence time for bay waters in that region, materials discharged into Redfish Bay waters, including dissolved nutrients, would tend to accumulate there. Studies by Tomasko and Lapointe (1991) in Florida suggest that this mechanism would lead to increased growth of macroalgae from the nutrients.

Harbor Island

Changes Related to Dredging and Channel Construction

On Harbor Island, the most extensive losses caused by dredging activities also occurred between 1958 and 1975 (Pulich and others, 1997). Losses occurred in the southwest corner of the island complex as a result of channels dredged for hydrocarbon exploration and from disposal of dredged material excavated from the Corpus Christi Ship Channel. Overall, these losses were relatively small compared to those along the GIWW. Losses of seagrass along the eastern side of Talley and Traylor Islands (Aransas Bay) may have been in part the result of open water discharge of dredged material on the western side of the GIWW to Lydia Ann Channel or in association with local intrabay channels. Barren nearshore areas that were more densely vegetated in 1975 may have been buried by discharged or reworked dredged material by 1994. Between 1975 and 1994, a slight decrease in seagrass area (approximately 148 ha, or 366 acres) was measured; the decrease was attributed to loss of mostly turtle grass beds along channels.

Changes Related to Bathymetry

Between 1958 and 1975, there was a net increase of more than 1,000 ha (2,471 acres) in shoal grass and wigeon grass beds from expansion onto shallow wind-tidal delta flats of north Harbor Island (Brown and others, 1976; White and others, 1983). This seagrass bed expansion is attributed by Pulich and others (1997) to an increase in water depth over the flats because of an accelerated rate of relative sea-level rise that followed the severe drought of the mid-1950s (Swanson and Thurlow, 1973; Ramsey and Penland, 1989). This seagrass expansion was complete by 1975.

Mustang Island

Changes Related to Dredging and Channel Construction

Direct changes in seagrass because of dredging operations on the bay side of Mustang Island have been minimal since the mid-1950s. Extensive changes occurred between 1938 and 1956, however, when several channels related to oil and gas exploration were dredged across seagrass beds and wind-tidal flats east of Shamrock Island (White and others, 1978).

Changes Related to Bathymetry

Seagrass beds also expanded over wind-tidal flats on Mustang Island as a result of rising relative sea level after the 1950s drought. Seagrasses expanded over the broad flats on the bay side of Mustang and North Padre Islands, increasing from approximately 1,030 ha (2,545 acres) in 1958 to 2,700 ha (6,672 acres) in 1974 (White and others, 1978). The trend set during this period from the late 1950s to 1974 continued from 1974 to 1994, with seagrass cover increasing 20.6% from 2,375 ha (5,868 acres) in 1974 to 2,870 ha (7,092 acres) in 1994. These increases in seagrass area from the late 1950s to the mid-1970s correlate positively with the increase in water levels for this time period as recorded at the Rockport tide gage (Swanson and Thurlow, 1973; Ramsey and Penland, 1989). Thus, seagrass expansion can be attributed to rising water levels during the 1970s, the protected physiography along the leeward side of Mustang Island, and the noticeable lack of residential and waterfront development along this protected bayside shoreline.

Nueces Bay

Changes Related to Dredging and Channel Construction

Direct losses in Nueces Bay seagrass are not documented as being a result of channel dredging or dredged-material disposal; however, oyster shell dredging, which actively occurred in the open bay prior to 1970 but was stopped in 1972, probably impacted the seagrass beds in the 1950s and 1960s.

Changes Related to Sedimentation

A major hurricane effect seems to explain the disappearance of shoal grass beds from Nueces Bay after Hurricane Beulah during the late 1960s. Seagrass dynamics appear most related to heavy runoff/sediment deposition from Gum Hollow Creek caused by Beulah's torrential, and record-level, rainfall in 1967 (McGowen, 1971). Most of the runoff from the extensive adjacent farmlands on the north side of Nueces Bay flows through this creek drainage into Nueces Bay. The extreme sediment deposition from this hurricane event created a large fan delta of fine clay and silt sediments in the bay (McGowen, 1971), which buried the existing seagrass beds and prevented reestablishment of shoal grass for about 15 yr.

Species Information

Generally, turbid, low-salinity water regimes appear to be responsible for stressing seagrasses in upper estuary bays (e.g., Nueces, Copano, and San Antonio Bays). The cyclical occurrence of shoal grass and wigeon grass in Nueces Bay (this vignette) seems to reflect these salinity and turbidity fluctuations. The resurgence of seagrass in Oso Bay over the last 20 yr also correlates with the growth requirements of shoal grass for clear, polyhaline (>18 ppt) marine waters which are discharged from the cooling pond system at the Central Power and Light Company powerplant in south Corpus Christi.

Seagrass species distributions were determined from the 1995–96 field surveys for the Redfish Bay area, Harbor Island complex, Aransas Bay/Copano Bay area, and Mustang Island area (Pulich and others, 1997). Based on the groundtruthing surveys, frequency of species occurrence was calculated (table 3) for these individual bay regions. The relative percent occurrence reveals the dominance of shoal grass (63%–90% of all samples) in the system and the scarcity of manatee grass (2.0%–7.6%) and star grass (0.4%–6.0%). Turtle grass appeared much more frequently in Redfish Bay (61%) and the Harbor Island area (24% of samples) than in either Mustang Island or Aransas Bay (15.5% average). The low-salinity-tolerant wigeon grass is found frequently in all shoal grass beds within the Texas Coastal Bend area. Wigeon grass, being primarily an annual species, is very seasonal, occurring most abundantly in spring and fall; however, in some backbay and upper bay areas, wigeon grass is also frequent in summer (Port, Mission, and Nueces Bays; San Jose Island and Harbor Island areas), when salinities are low.

Shoal grass is the most abundant species throughout the Texas Coastal Bend area, with extensive beds even in mesohaline, upper bay segments. Shoal grass is the dominant species in all bay segments except Redfish Bay. Along with wigeon grass, these two are the only species found in Nueces, Copano, Port, and St. Charles Bays. Small amounts of star grass are found in all bay systems except Nueces, Copano, Port, and St. Charles Bays.

Turtle grass and manatee grass were locally dominant only in the Redfish Bay area, in the Harbor Island complex, along Mustang Island in the "East Flats" area, and around Mud Island and Allyns Bight in Aransas Bay proper. These two tropical species frequently occur together and appear most abundant in polyhaline areas, primarily the lower bay outlets to the gulf at Port Aransas. Except for the relict population of turtle grass still located in Christmas Bay (near West Galveston Bay), no other populations of turtle grass or manatee grass are presently known to occur farther north on the Texas coast than this southeastern shore of Aransas Bay.

Monitoring for Seagrass Health

Seagrass monitoring work should be conducted while seagrasses are actually disappearing. Consequently, monitoring is a primary objective of the proposed coastwide Texas Seagrass Monitoring Program (Pulich and others, 2003), which is currently under development by State resource agencies (see Statewide Summary for Texas, this report). As indicated previously, studies in Texas Coastal Bend bays since 1990 have been conducted after seagrass declines. These results demonstrate the need for a proactive, regular

Table 3. Frequency of occurrence (percent of samples) for five seagrass species in the Texas Coastal Bend, 1995–96. Total percentage in each segment is more than 100% because of mixed species assemblages in samples.

Segments	Shoal grass[1]	Wigeon grass[2]	Turtle grass[3]	Manatee grass[4]	Star grass[5]
Redfish Bay	62.7	7.1	61.2	7.6	4.6
Harbor Island	84.6	24.9	24.2	3.0	0.5
Mustang Island	86.0	15.0	15.8	1.9	0.4
Aransas/Copano Bay system	90.0	9.2	15.3	2.8	6.0

[1] *Halodule wrightii*. [2] *Ruppia maritima*. [3] *Thalassia testudinum*. [4] *Syringodium filiforme*. [5] *Halophila engelmannii*.

monitoring program to assess seagrass health and to detect impacts prior to fragmentation or loss of seagrass habitat. In addition, monitoring would be important in documenting the success of restoration efforts.

Mapping and Monitoring Needs

For the proposed Texas Seagrass Monitoring Program (see Pulich and others, 2003), it is critical to develop good indicators of seagrass community health and then to establish a statistically robust sampling scheme to measure these indicators. The Texas monitoring program plans to use a two-part approach of intensive, probabilistic-based field sampling combined with landscape sampling from aerial photography. Intensive field sampling will be focused on detecting water-quality degradation and establishing water-quality criteria (standards) for these seagrass systems. Aerial photography will be flown at 1:24,000 scale every 5–10 yr for status and trends assessment of seagrass distribution in an entire bay system, and annual photographs at 1:9,600 or larger scale will be taken at targeted sites where impacts are suspected from specific stressors. The high-resolution photography will be especially important for monitoring seagrass patch dynamics at these “target sites” or documenting restoration of former seagass areas.

Restoration and Enhancement Opportunities

Propeller Scar Restoration

Because of the serious fragmentation of seagrass beds noted from monitoring studies (Pulich and others, 1997; Dunton and Schonberg, 2002), TPWD recently started an active program to prevent and restore motorboat damage to turtle grass beds. This program has primarily targeted turtle grass beds because of their scarcity as climax habitat and their slow recovery from such damage. While restoration and recovery of propeller scar areas in turtle grass have been studied at some length in Florida seagrass beds (Dawes and others, 1997; Kenworthy and others, 2000), no studies had been previously undertaken in Texas waters. Since conditions between Florida and Texas seagrass beds are expected to be different, TPWD sought to document the applicability of such restoration techniques for Texas turtle grass beds.

With funding available from the National Oceanic and Atmospheric Administration (NOAA) Gulf of Mexico Sustainable Fisheries Program, a 1999–2001 propeller scar restoration project was conducted to test restoration techniques (see McEachron and others, 2001). Initial work was undertaken in the Redfish Bay area because of its proximity to urban areas and heavy use by boaters. Restoration techniques included filling propeller scar trenches with sediment (clean sand), removal of seaweed and seagrass wrack/litter, and injection of nutrients (fertilizer) and growth hormone mixtures into bare sediments of propeller scars or bare, “blowout” areas. A limited amount of work was performed to enhance recovery by manipulated transplanting of bare-root seagrass sprigs into representative propeller scars or bare areas. Regular monitoring of treated and untreated scars over a 2-yr period was designed to determine effects of underwater light attenuation, water-column conditions, wrack accumulation, and sediment chemistry on plant production. Seagrass recovery in scars was compared to adjacent undisturbed native seagrass beds.

Shoal grass transplantings and nutrient/growth hormone additions were performed by using a special pontoon boat and injector wheel system developed by ASIS Inc. (Aquatic Subsurface Injection Systems) of Ruskin, Fla. (McEachron and others, 2001). Turtle grass propagules with meristems were also hand planted. Results indicated problems with using the special ASIS boat and injector wheel to transplant shoal grass sprigs. Problems were related to sediment type (too soft a substrate), the wheel sprigging mechanism itself, and the method of bundling the donor sprigs into planting units. Other results indicated that the addition of nutrients, growth hormone, or root stimulator solutions used in this restoration effort did not aid in recolonization of hand planted turtle grass in Redfish Bay. Until these methods are improved for Texas sites, hand planting of shoal grass plugs would be recommended (McEachron and others, 2001).

Other restoration efforts demonstrated that the addition of sand into propeller scars contained in a geotube (“sand sock”) may promote seagrass expansion, but this technique needs further investigation (McEachron and others, 2001). The geotube technique stabilizes the added sand, but sediment grain size within the geotube appears to be critical. If fine grain sediments are used, recolonization and lateral expansion of native seagrasses across the scar may be enhanced.

Texas Parks and Wildlife Department continues to recommend that techniques to restore turtle grass beds be developed. While the collection of turtle grass propagules is destructive to donor beds, whose survival is then impaired, work with seedlings and nurseries may hold promise. The use of seedlings and nursery propagation could make turtle grass replanting feasible. Shoal grass appears to often recover extensively by recolonization, and this species also acts as a natural colonizer to stabilize propeller scars in turtle grass beds.

Designation of State Scientific Areas

As explained in “Statewide Summary for Texas” (this report), TPWD, Texas General Land Office (TGLO), and TNRCC developed the Seagrass Conservation Plan for Texas (SCPT), which identified research issues, management and policy issues, and education/public outreach needed to conserve Texas seagrass beds (Pulich, 1999). Because research thus far indicates that efforts to restore turtle grass propeller scars are unpredictable, inefficient, and expensive on a large scale, TPWD has concluded that an effective, practical solution in some cases is protective management of seagrass beds. Using its management authority to protect coastal fisheries habitat, TPWD therefore designated Redfish Bay in 2000 as an official State Scientific Area (Texas Parks and Wildlife Department, 2001). Under this jurisdiction, the establishment of no-motor zones was proposed in shallow turtle grass beds to protect them from propeller scarring and allow natural recovery over a sufficient time period (3–5 yr).

Public participation and outreach activities were also identified as critical in the process of conserving seagrasses. A citizens advisory group (Seagrass Task Force) was organized to help develop strategies to protect seagrasses in the “Redfish Bay Scientific Area.” The Seagrass Task Force included stakeholders and bay user groups including local governments, private citizens, business owners, and organized boater/fishing groups. They recognized the need for educating boaters about the ecological importance of seagrass beds and for providing navigation aids in shallow seagrass waters. This task force determined that propeller scarring was occurring primarily for two reasons: (1) boat operators were unfamiliar with the bay and did not know the limitations of their boats, and (2) boaters familiar with the area were using shallow grass flats as shortcuts to travel across the bay instead of using deeper channels. The result in both cases was unintended propeller scarring.

Signage and Boater Education Activities

To aid in boater navigation around seagrass beds, the Seagrass Task Force recommended that signage be developed to mark and restrict boating channels in severely scarred seagrass areas. With volunteer help and support from the task force, three popular but sensitive seagrass areas of Redfish Bay were marked as voluntary no-motor zones, and signs were erected in the water to warn boaters and to allow natural recovery of damaged grass beds. They also recommended that large displays be developed and placed at bay area boat ramps/marinas to provide information to boaters about seagrasses, the effects of propeller scarring, and ways to prevent it. These marina displays contained a photomap of Redfish Bay that indicated the location of the voluntary no-motor zones. In an effort to protect scarred turtle grass beds, 135 navigation signs were placed in the waters, and 11 displays were posted at marinas. As a result of this volunteer outreach project, sign postings now help protect approximately 1,385 ha (3,422 acres) of seagrass beds in Redfish Bay in the form of voluntary no-motor zones.

Seagrass Restoration Through Beneficial Use Projects

In the Corpus Christi area, beneficial use and disposal of dredged material from ship channel and GIWW dredging have been proposed as a method of creating and restoring seagrass beds, along with marshlands, bird islands, oyster reefs, and other bay habitats. The Port of Corpus Christi Authority and the U.S. Army Corps of Engineers established the Regulatory Agency Coordination Team in 2000 which also included U.S. Fish and Wildlife Service, U.S. Environmental Protection Agency, NOAA’s National Marine Fisheries Service, TPWD, and TGLO. This group will evaluate such possible beneficial uses of dredged material and incorporate them into a dredged materials placement plan for the Corpus Christi Ship Channel-Channel Improvement Project (CCSC-CIP). This project will generate 114.7 million m^3 (150 million yd^3) of new sediment requiring disposal in Corpus Christi Bay proper, and substantial amounts would be available for open-water, beneficial uses, if disposal problems can be adequately solved by planning and engineering techniques.

The beneficial uses approach is committed to having a net positive environmental effect over the 50-yr life of such projects. Thus, the dredged materials management plan must address environmental issues and problems up front. Protection of existing seagrass in the Redfish Bay area, the selection of candidate disposal sites, and the creation of habitat sites are particularly critical issues. Other potential disadvantages include the type of sediment being dredged and turbidity from the fine particles. Sandy sediments, with minimal amounts of clays, would allow for controlled disposal and subsequent establishment of seagrasses. Seagrass protection would also require construction of breakwaters or underwater berms to reduce wave energy. Some dredged material would be used to fill geotextile tubes for breakwaters to reduce shoreline or underwater erosion.

Coastal Bend Bays and Estuaries Program Outreach Strategies

The CBBEP, since its inception as a National Estuary Program in 1992, has worked to develop education plans and outreach projects focused on seagrass habitats. The publication and distribution of Bay Fishing and Bay User Guide Maps (2002) are two CBBEP outreach projects that provide locational information to the public for protecting seagrass areas. Currently CBBEP is planning to participate in the Texas Seagrass Monitoring Program by funding seagrass monitoring projects in the Texas Coastal Bend, and this work will complement other existing coastwide management programs.

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Statewide Summary for Louisiana

By Michael A. Poirrier[1]

Background

Although wigeon grass (*Ruppia maritima*) is common all along coastal Louisiana, true seagrass meadows containing turtle grass (*Thalassia testudinum*), manatee grass (*Syringodium filiforme*), shoal grass (*Halodule wrightii*), and star grass (*Halophila englemannii*) currently occur only east of the Mississippi River near the Chandeleur Islands (fig. 1). The bays west of the Chandeleur Island chain provide clear, high-salinity, and low-nutrient waters appropriate for seagrass growth. Conversely, barrier island bays west of the Mississippi River are characterized by high turbidity, low salinity, and high nutrient levels, providing marginal seagrass habitat. Montz (1977) reported seagrass beds in the backbays of barrier islands west of the Mississippi River. In studies conducted in 1974 and 1975, he found that shoal grass and wigeon grass were abundant and suggested that other species might be present. Based on helicopter observations by Suzanne Hawes in January 1976, submerged vegetation coverage was estimated to be 53 ha (130 acres) north of East Timbalier Island, 121 ha (300 acres) north of Timbalier Island, and 111 ha (275 acres) north of Isles Dernieres (Montz, 1977). No submerged vegetation, including wigeon grass, however, was found during boat surveys of Timbalier Island conducted in October 2003 and seaplane surveys of Timbalier Island and Isles Dernieres conducted in July 2004 by Carol Franze and Michael Poirrier.

Although total species composition is unknown, the disappearance of 285 ha (705 acres) of submerged vegetation represents a significant loss of habitat and biodiversity. Loss of submerged vegetation west of the Mississippi River may be due in part to the natural deterioration of deltaic marshes and shores; however, these natural processes have been exacerbated by the activities of humans such as dredging of navigation canals; land reclamation; flood control; subsurface withdrawal of oil, gas, and water; and ironically in some instances, restoration. For instance, deposition of dredged material as a method to restore western barrier islands causes increased turbidity and may have been a significant cause of seagrass decline.

Seagrasses help support the geologic integrity of barrier islands by stabilizing sediments and the biologic integrity by providing essential invertebrate, fish, and waterfowl habitat. Seagrass meadows in Chandeleur Sound provide habitat that does not occur elsewhere in Louisiana. Seagrasses enhance biodiversity and provide refuge for unique populations of commercial, rare, and endangered species. The islands also serve as the first line of defense against coastal land loss from hurricanes for southeastern Louisiana and the highly populated New Orleans area. They are also an intrinsic part of the lower Pontchartrain Basin and an essential component of the Lake Pontchartrain estuarine system. The loss of Chandeleur seagrasses would cause further degradation of and hamper restoration efforts in Lake Pontchartrain and the lower Pontchartrain Basin.

Statewide Status and Trends

At present, Louisiana seagrasses are limited to shoals west of the Chandeleur Islands. Unfortunately, the only statement that can be made about East Timbalier and Timbalier Islands and Isles Dernieres seagrasses is that whatever species were present no longer occur there. Therefore, the discussion of Louisiana seagrasses including statewide status and trends, causes of change, data gaps and monitoring, restoration, and enhancement activities is restricted to the Chandeleur Islands.

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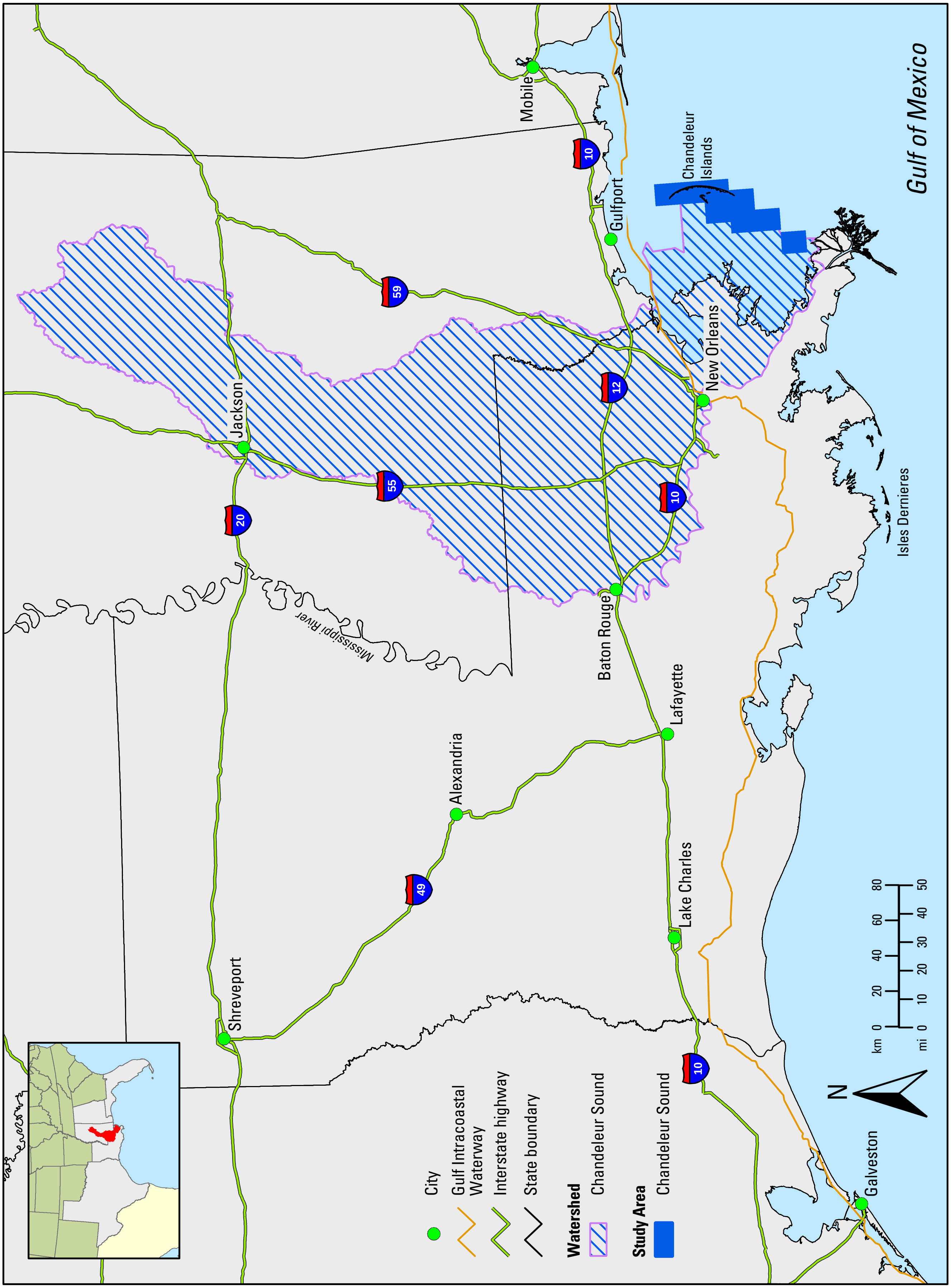

Figure 1. Watershed for the State of Louisiana.

Chandeleur Islands

By Michael A. Poirrier[1] and Lawrence R. Handley[2]

Background

The Chandeleur Islands (fig. 1) are the barrier remains of the Mississippi River's St. Bernard Delta. Coastal geological processes constantly alter this 72-km (45-mi) barrier island arc trend, and major geomorphological changes occur with the passage of tropical storms and hurricanes (Williams and others, 1997). These remote islands can be accessed only by boat or seaplane. With the exception of light tower navigation aids at the northern end (Hewes Point), they lack permanent structures. The islands were designated a national wildlife refuge in 1904 and became part of the National Wilderness Preservation System in 1975. This preservation has prevented direct, site-specific stressors from development that are found in many other populated areas. Seagrass meadows occur on shallow shoals in protected waters behind the northern islands. The distribution and abundance of Chandeleur seagrasses are almost entirely controlled by geological processes related to storms and barrier island dynamics. The fate of Louisiana seagrasses, therefore, is dependent upon the fate of this fragile island chain.

Scope of Area

The Chandeleur Island Chain consists of Chandeleur Island to the north followed by Curlew, Grand Gosier, and Breton Islands to the southwest and Freemason, North, and New Harbor Islands to the west (fig. 2). The northern end of the Chandeleur chain is 35 km (22 mi) south of Biloxi, Miss.; the southern end, Breton Island, is 25 km (16 mi) northeast of Venice, La. Chandeleur, Freemason, North, New Harbor, and a northern portion of Curlew Islands are in St. Bernard Parish, and the southwestern islands, a portion of Curlew, Grand Gosier, and Breton Islands are in Plaquemines Parish. Significant seagrass meadows are limited to Curlew Island and islands in the chain north of Curlew.

Methodology Employed To Determine and Document Current Status

Natural color, 1:24,000-scale aerial photography was acquired. The mapping protocol consisted of stereoscopic photointerpretation, cartographic transfer, and digitization in accordance with strict mapping standards and conventions. Other important aspects of the protocol included the development of a classification system, groundtruthing, quality control, and peer review. Land, water, and areas where seagrasses were present were included on the maps.

The information derived from the photography was subsequently transferred by using a zoom transfer scope onto a stable medium overlaying U.S. Geological Survey (USGS) 1:24,000-scale quadrangle base maps. The primary data sources were 1:24,000-scale natural color aerial photography flown by National Aeronautics and Space Administration. In those cases where the data were inadequate or incomplete, contemporary supplemental data were acquired from other sources and used to complete the photographic coverage.

The groundtruthing phase included the participation of field staff from the USGS National Wetlands Research Center. Draft maps were sent out to these agencies and staff for review and comments. All comments received were incorporated into the final maps prepared and delivered.

Studies of seagrass species occurrence, distribution, and abundance were conducted on field trips during 1999, 2000, and 2001 in conjunction with studies of Hurricane Georges' impact and recovery (Poirrier and Franze, 2001; Franze, 2002).

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[2] U.S. Geological Survey, National Wetlands Research Center.

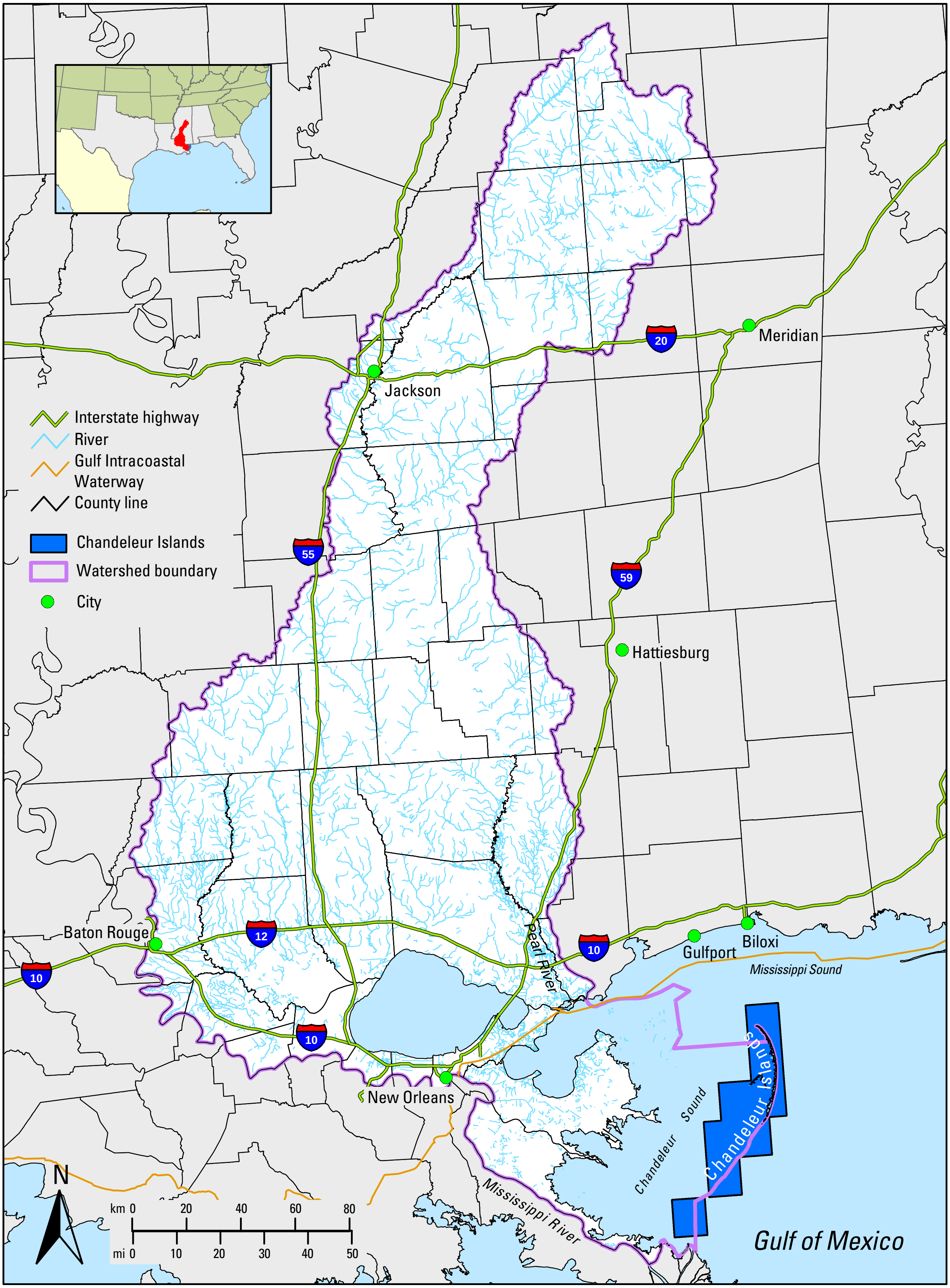

Figure 1. Watershed for Chandeleur Islands.

Methodology Employed To Analyze Historical Trends

Historical seagrass trends were analyzed by comparing changes in total areal coverage of habitat with seagrass present along a time sequence. Comparisons were made among data sums of seagrass coverage for April 1969 (fig. 3), October 1969 (fig. 4), April–June 1992 (fig. 5), and November–December 1995 (fig. 6). Maps of seagrass distribution for 1969, 1992, and 1995 were studied to determine the location of major changes of coverage. Methodology used for preparing maps can be found in appendix 1.

Status and Trends

Areal coverage of seagrasses (table 1) decreased by 1,708 ha (4,221 acres) (26.8%) between April and October 1969, increased by 1,868 ha (4,615 acres) (28.6%) from October 1969 to April–June 1992, and decreased by 2,025 ha (5,004 acres) (31%) from April–June 1992 to November–December 1995. It should be noted that April 1969 and October 1969 data were prepared to assess the changes associated with Hurricane Camille, which affected the islands in August 1969. Comparing October 1969 data with the 1992 data demonstrates how much the seagrass beds recovered after the hurricane.

Table 1. Seagrass areal coverage for the Chandeleur Sound shoals, Louisiana, from April and October 1969 and from April–June 1992 and November–December 1995.

Date	Hectares	Acres
April 1969	6,377	15,758
October 1969	4,669	11,537
April–June 1992	6,537	16,152
November–December 1995	4,512	11,149

Causes of Change

The northern Chandeleur Islands provide a relatively pristine seagrass habitat with few direct, site-specific, human-induced stressors (figs. 3, 4, 5, and 6). The distance of the northern islands—including Chandeleur, Freemason, North, New Harbor, and Curlew—of more than 35 km (22 mi) from Biloxi, Miss., and 60 km (37 mi) from the mouth of the Pearl River and the passes of Lake Pontchartrain in Louisiana protect them from pollution sources and other stressors. Nutrients and suspended solids from coastal discharges are probably assimilated in the Biloxi marsh system of Louisiana and the waters of Mississippi and Chandeleur Sounds and do not appear to have a major impact on the northern islands.

The southern Chandeleur chain, including Breton and Gosier Islands, are 15 km (9 mi) to 30 km (19 mi) from major passes of the Mississippi River. This chain does not support significant seagrass meadows (figs. 2, 3, and 4). Fresh water, plant nutrients, and turbidity from the Mississippi River may adversely affect seagrass development in the southern island chain. The lower Mississippi River-Gulf Outlet Channel is located between Breton and Gosier Islands. Increased turbidity from maintenance dredging may also adversely affect seagrasses. In addition, the southern islands are smaller and narrower and have a greater rate of retreat. Seagrasses are probably also limited by the constantly changing morphology of the southern islands.

Although the northern islands are subject to global environmental changes, the principal ecological drivers are natural coastal processes related to barrier island dynamics, abandoned river deltas, and damage from tropical storms and hurricanes (Williams and others, 1997). Some recovery through sediment deposition occurs after storms; however, land area of the Chandeleur Islands decreased from 3,462 ha (8,554 acres) to 1,215 ha (3,003 acres) (65%) between 1855 and 1999, with a 40% decrease (from 2,029 ha to 1,215 ha, or from 5,013 acres to 3,003 acres) from the passage of Hurricane Georges in 1998 (Penland and others, 2001). Other factors such as low water events, winter storms associated with frontal passages, wasting disease, coastal eutrophication, and damage from motorboat propellers may contribute to seagrass change. These factors do not, however, occur at spatial or temporal scales that would explain the rapid changes presented above, and if unabated, the human stressors would cause a steady decline not present in the previously mentioned data. The best explanation for these short-term, but major, decreases and increases is loss during storms and recovery after storms. The distribution and abundance of seagrass species are affected by storms, including winter storms, tropical storms, and hurricanes.

The 1,708-ha (4,221-acre) seagrass loss (26.8%) between April 1969 (fig. 3) and October 1969 (fig. 4) was the result of Hurricane Camille, which affected the islands in August 1969. The 1,868-ha (4,615-acre) increase (40%) from 1969 and 1992 (figs. 3 and 5) was probably caused by recovery from the effects of Camille. The islands were also impacted by hurricanes and other storms during this 13-yr period, however, and other cycles of increases and decreases may have occurred between 1969 and 1992 (figs. 3–5).

The limited data presented above are not sufficient to document the cause of the 2,025-ha (5,004-acre) decrease (31%) between April–June 1992 (fig. 5) and November–December 1995 (fig. 6). Seagrass meadows decreased in the western bays of Chandeleur and Curlew Islands and in the shoals near Freemason, North, and New Harbor Islands (figs. 5 and 6). Retreat of the Chandeleur Island meadow occurred at the northern and southern ends, in shallow water on the east side, and in deep water on the west side of shoals behind

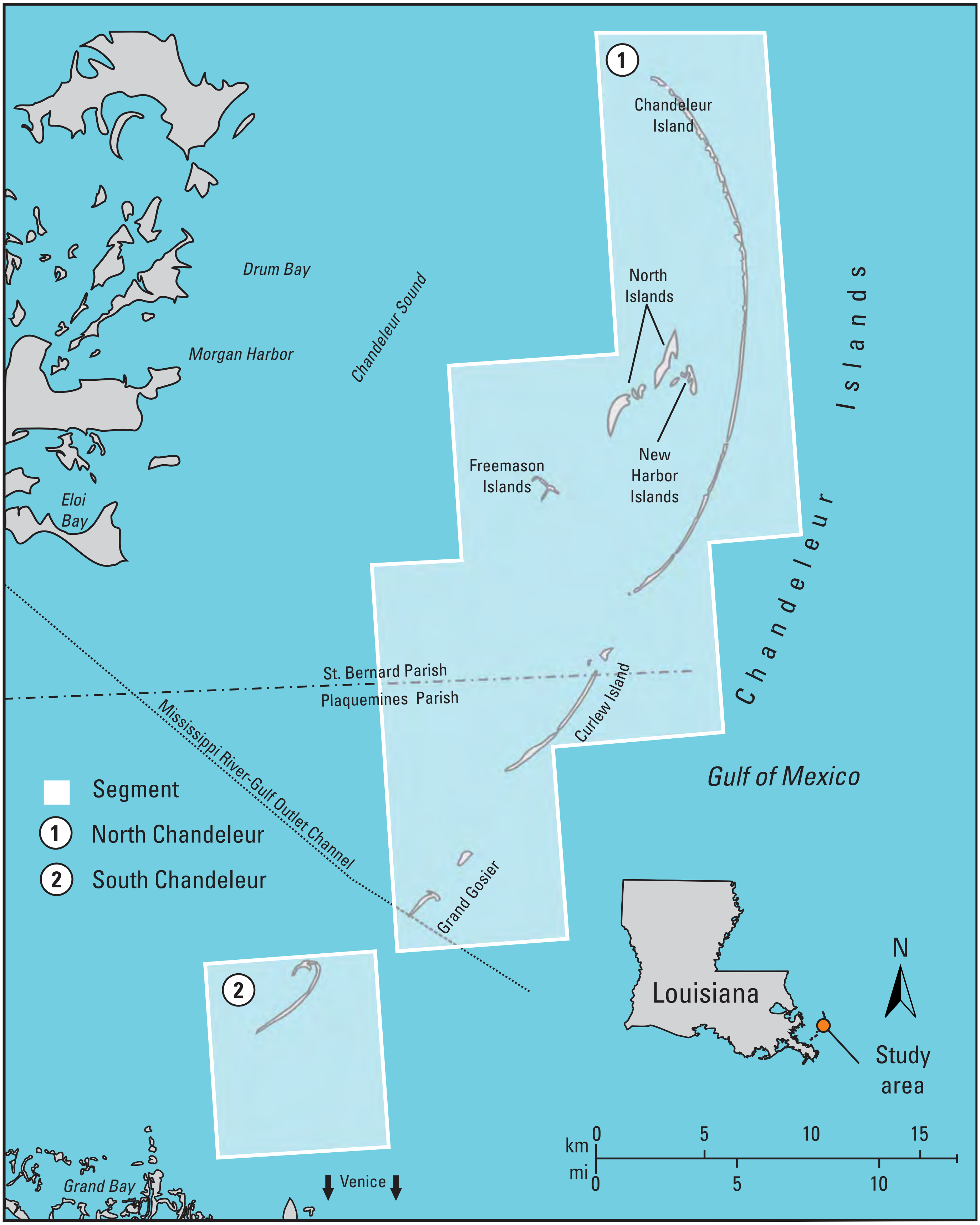

Figure 2. Scope of area for the Chandeleur Island seagrass vignette.

the island. There were no major changes in human-induced stressors at this remote site that could explain this widespread reduction. These changes may have been caused by erosion and turbidity from storms and hurricane and by seasonal differences in distribution and abundance.

Although there were no direct hurricane landings near the Chandeleur chain between 1992 and 1995, Hurricane Andrew hit the Louisiana coast in August 1992, and the 1995 hurricane season had three hurricanes, Allison, Erin, and Opal, in the eastern gulf, all of which affected seagrasses around the Chandeleurs to some extent. Opal was the most severe storm and was located south of Louisiana as it intensified and moved north across the Gulf of Mexico, but it shifted eastward before landfall in eastern Florida. Recovery from hurricane damage and subsequent meadow stability are also affected by severe winter storms and low water associated with frontal passages (Franze, 2002) and could have contributed to loss or affected the rate of recovery. Seasonal variation in seagrass coverage may have contributed to the differences between April–June 1992 and November–December 1995. Seagrass distribution in shallow water is limited by wave energy and low water events and in deep water by available light. Increased stress in these zones during winter may have decreased coverage of species that colonize shallow water in spring and summer and decrease during winter. The contribution of season to this decline, however, is probably minimal because coverage is generally high through November and early December, with major decreases occurring in late December, January, and February. The primary cause of this decline was probably hurricanes, particularly Hurricane Opal in October 1995.

Species Information

Turtle grass (*Thalassia testudinum*), manatee grass (*Syringodium filiforme*), shoal grass (*Halodule wrightii*), star grass (*Halophila engelmannii*), and wigeon grass (*Ruppia maritima*) have been reported by Hoese and Valentine (1972), Poirrier and Franze (2001), and Franze (2002). Data on the damage and recovery of Chandeleur Island seagrasses after the passage of Hurricane Georges in 1998 are presented in Moncreiff and others (1999), Poirrier and Franze (2001), and Franze (2002). Hurricane Georges caused numerous channel cuts through Chandeleur Island, and sediment deposition formed washover features. Seagrass damage was caused by erosion of meadows by channel flow and burial by washover features. When surveyed in 1999 (Franze, 2002), sites protected from hurricane damage were dominated by dense (100% coverage) turtle grass meadows. Exposed sites away from washover features were composed of turtle grass and manatee grass with some shoal grass. Star grass was present, but rare.

In the first year after Hurricane Georges, wigeon grass was associated with shoreline marshes, and shoal grass was present in shallow shoal areas. The channel cuts and sediment washover fans lacked vegetation. By 2001, colonization of the washover features by shoal grass and wigeon grass was widespread, but turtle grass and manatee grass colonization was patchy and appeared to be from buried root and shoot fragments (Franze, 2002). This pattern of disruption of mature turtle grass and manatee grass meadows by unvegetated washover features and gradual colonization of these features by shoal grass and wigeon grass observed after Hurricane Georges may be a regular pattern that occurs after major storms and hurricanes. There has been no change in species composition over time, but the relative abundance of wigeon grass and shoal grass may increase during recovery from major storms.

Monitoring for Seagrass Health

There is no established monitoring program to periodically evaluate the condition of the Chandeleur Islands seagrasses. Hoese and Valentine (1972) provided a species list along with general observations and comments on the effects of the passage of Hurricane Camille in 1969. Handley (1995) estimated the seagrass meadows to be about 6,000 ha (14,826 acres) based on 1989 aerial photographs. Michot and Chadwick (1994) investigated seagrasses as potential food for ducks. Poirrier and Franze (2001) and Franze (2002) studied damage and early stages of recovery from Hurricane Georges and the efficacy of transplanting to enhance recovery. Transplanting efforts were funded in part by the U.S. Environmental Protection Agency's Gulf of Mexico Program. Because the islands have few direct, human-induced stressors and because seagrass population dynamics are driven by storms and geological processes, the need to monitor seagrass health could be questioned. The relative health of Chandeleur seagrasses, however, is unknown and can be assessed only through monitoring. Healthy systems also need to be monitored because data on the productivity and population dynamics of meadows, not influenced by multiple human stressors, would be valuable in managing stressed meadows.

Mapping and Monitoring Needs

Future mapping and monitoring of Chandeleur seagrasses are needed to document storm effects and to determine the rate and mechanisms of natural recovery. Chandeleur Island seagrasses were severely damaged by Hurricane Georges in 1998, and additional studies are needed because recovery is a slow process. Barrier island landscape changes caused by storms provide a natural laboratory to observe the development of seagrass meadows on nonvegetated habitat. These efforts would provide information on colonization, species succession, and interactions between coastal geological processes and seagrass population dynamics. Seagrasses may play an important role in maintaining the

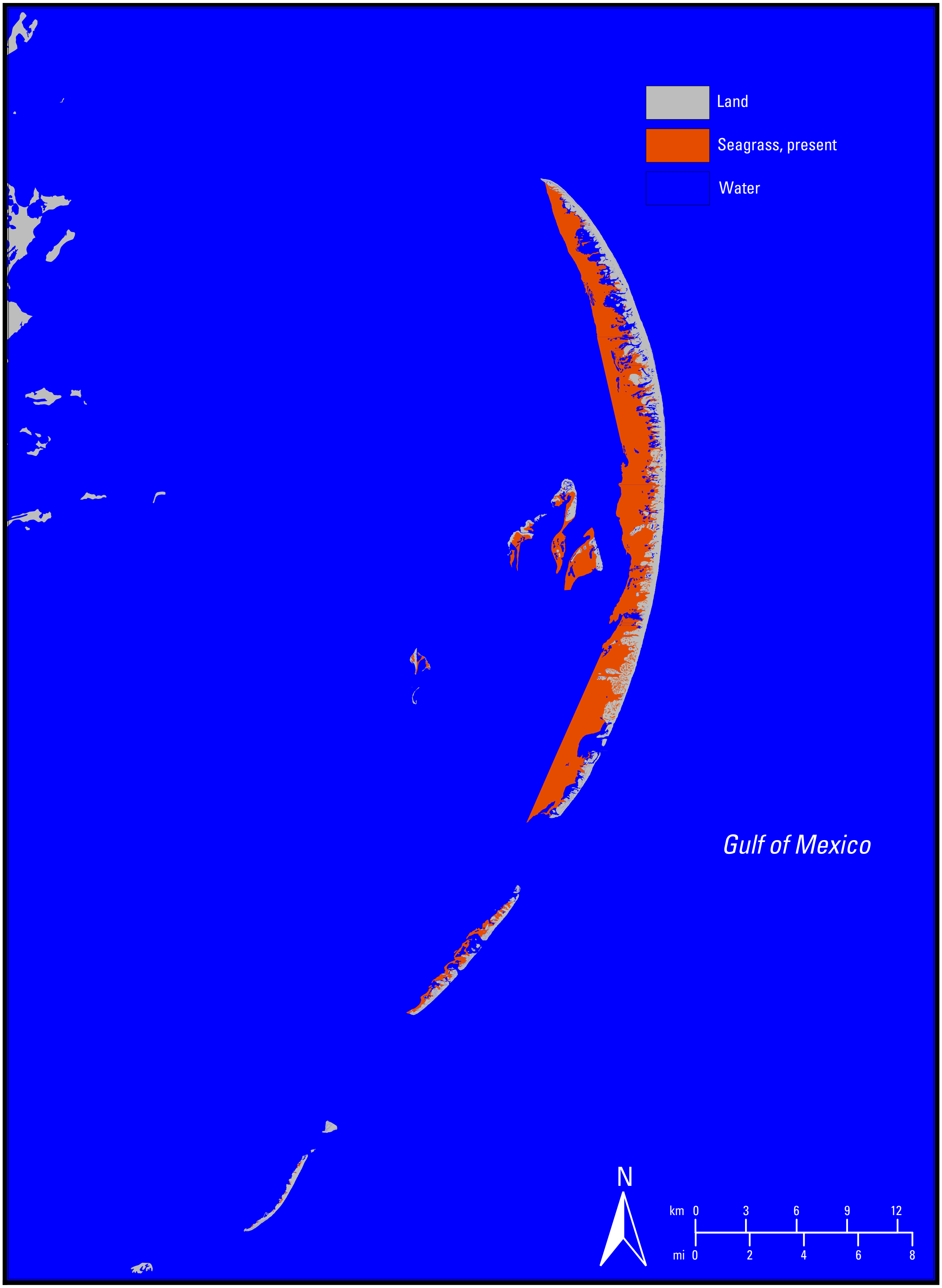

Figure 3. Distribution of seagrasses around the Chandeleur Islands, April 1969.

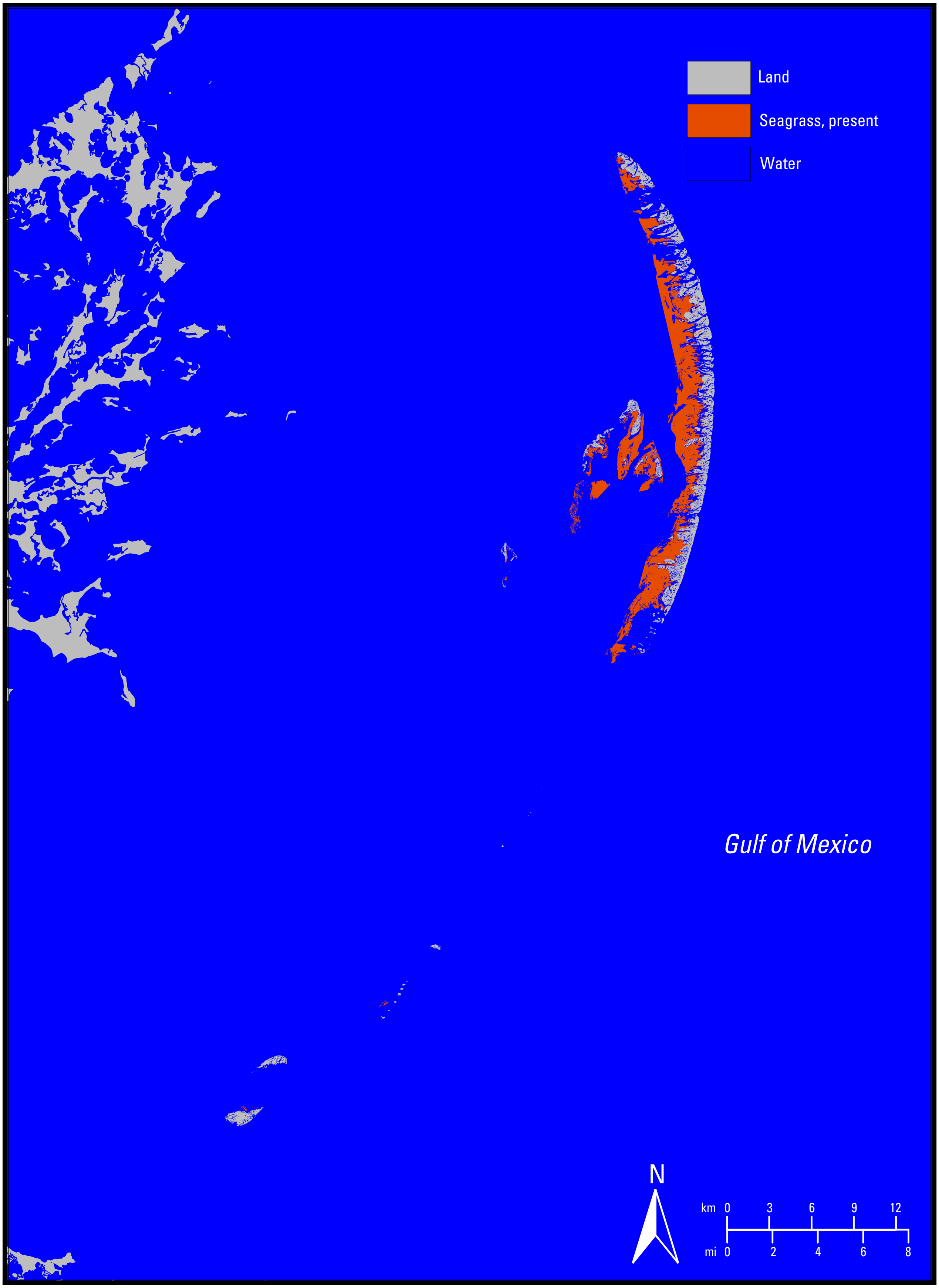

Figure 4. Distribution of seagrasses around the Chandeleur Islands, October 1969.

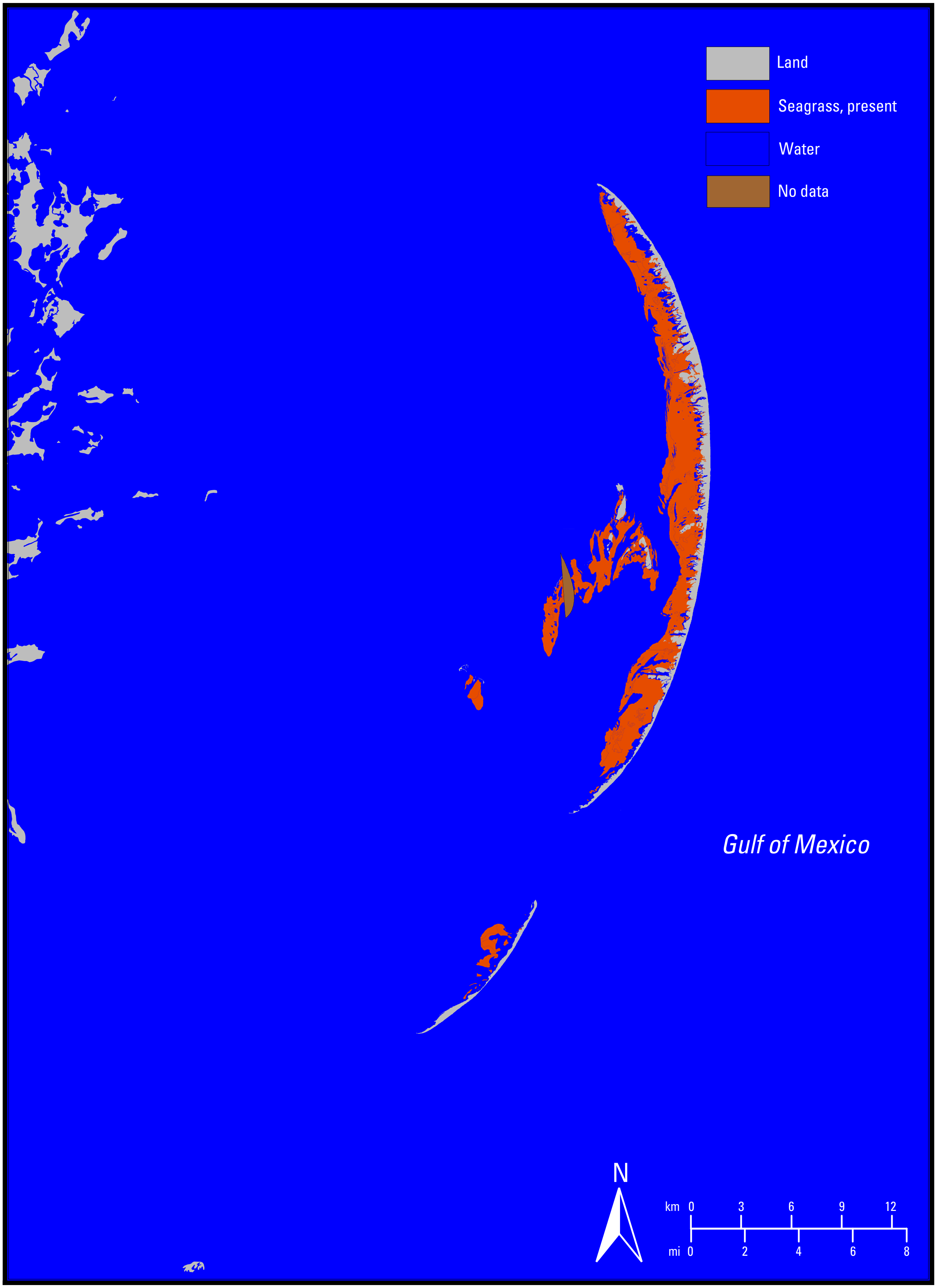

Figure 5. Distribution of seagrasses around the Chandeleur Islands, 1992.

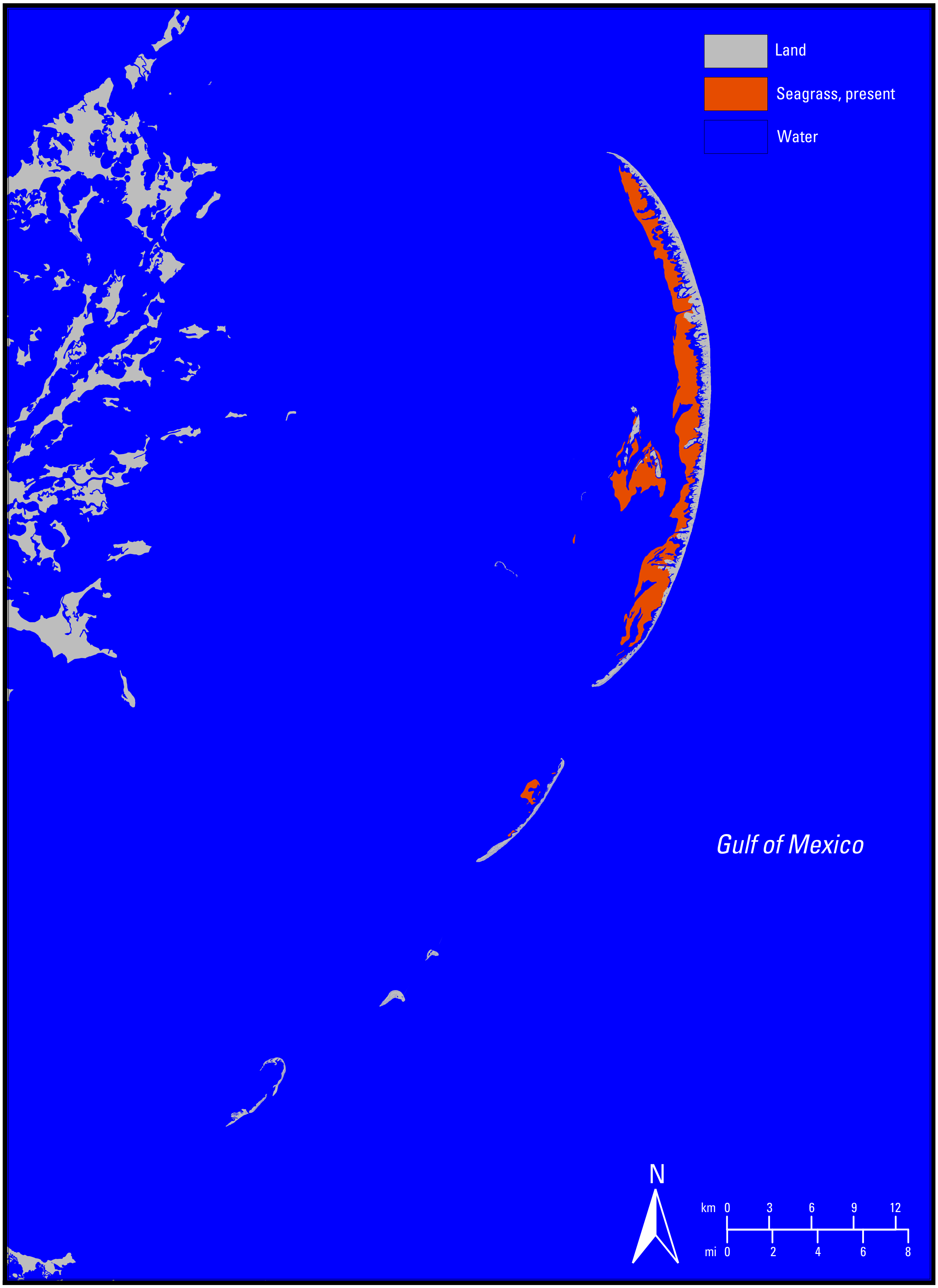

Figure 6. Distribution of seagrasses around the Chandeleur Islands, 1995.

integrity of the Chandeleur Islands and reducing the rate of island regression. Monitoring aspects of basic seagrass biology such as seasonal patterns, biomass, flowering, and epiphyte growth at a reference site relatively free from human-induced stressors would provide a standard for assessing the health of seagrasses at stressed sites in the Gulf of Mexico. Chandeleur seagrasses need to be mapped regularly to document damage and recovery from storms, certainly every 5 yr.

Restoration and Enhancement Opportunities

Changes in seagrass coverage are mainly driven by the long-term regression of the Chandeleur Island chain because of the deterioration of the St. Bernard Delta and direct seagrass loss from erosion and deposition from the passage of tropical storms and hurricanes. Additionally, the island chain has experienced increased damages to seagrasses over the past decades associated with boating. The persistence of Chandeleur seagrasses is dependent upon the calm, shallow bays provided by islands in the chain; furthermore, seagrasses may be important in stabilizing shoals on the bay side of the islands. These stable shoals provide a platform for the slow shoreward movement of islands. Loss of these shoals would hasten regression and island loss and result in loss of seagrass habitat.

Recovery from hurricane damage appears to be a slow process (Franze, 2002). Barrier island regression and community disruption by hurricanes are natural processes and appear to drive the Chandeleur seagrass community structure, which is the only true seagrass community in Louisiana. Outside stressors, however, could interrupt community resiliency and cause long-term seagrass and island loss. Considering the importance of these islands to geological and biological integrity of coastal Louisiana, it is prudent to obtain a better understanding of processes involved in seagrass recovery after hurricane damage to obtain effective ways to enhance recovery. Small-scale enhancement and restoration projects may not have a major effect on maintaining this large coastal system, but they do provide a knowledge base for effective large-scale projects if they are needed in the future. Louisiana does not have a seagrass monitoring or maintenance plan. Because of the importance of seagrasses to the integrity of the Chandeleur Island chain, seagrass monitoring, management, and restoration plans need to be developed and included in efforts to restore coastal Louisiana.

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Statewide Summary for Mississippi

By Cynthia A. Moncreiff[1]

Background

Although the coastline of Mississippi spans only 113 linear kilometers (70 mi), the estuaries within its borders constitute a much larger area, roughly 594 km (369 mi) (fig. 1). The primary body of water within the State's boundaries that supports seagrasses is Mississippi Sound, which covers 175,412 ha (433,443 acres) at mean low tide (Christmas, 1973). This body of water is immediately bounded by the coast of Mississippi to the north; Mobile Bay, Ala., to the east; a series of barrier islands that make up most of the Gulf Islands National Seashore to the south; and Lake Borgne, La., to the west (fig. 1).

Mississippi Sound is fed from the north by eight coastal mainland watersheds and drainage systems and from the south by tidal exchange with the Gulf of Mexico (through a series of five barrier island–bounded passes). From west to east, the mainland drainages include Lake Borgne (La.), the Pearl River (La. and Miss.), the Jourdan River (Miss., part of the St. Louis Bay complex), the Wolf River (Miss., part of the St. Louis Bay complex), the Tchoutacabouffa River (Miss., part of Biloxi Bay), the Pascagoula River (Miss., one of the only remaining undammed river systems in the continental United States), and Mobile Bay (Ala.). These systems have been combined into three estuarine drainage areas by the National Oceanic and Atmospheric Administration's (NOAA) National Ocean Service Special Projects Office: the western portion of the Mississippi Sound, the eastern Mississippi Sound, and Mobile Bay, Ala. (National Oceanic and Atmospheric Administration, 2003). The western portion of Mississippi Sound receives an average of 455.8 m^3/s (15,741 ft^3/s) of fresh water from a coastal watershed of 948,976 ha (3,664 mi^2) and a total watershed area of 2,726,493 ha (10,527 mi^2). Eastern Mississippi Sound drains a coastal area totaling 463,351 ha (1,789 mi^2) and a total watershed area of 2,500,645 ha (9,655 mi^2), with an average freshwater inflow of 426.6 m^3/s (15,065 ft^3/s). Waters from Mobile Bay, Ala., which receives an average of 1957.4 m^3/s (69,118 ft^3/s), also flow into Mississippi Sound; the watershed of the bay totals 11,441,843 ha (44,177 mi^2); 1,150,737 ha (4,443 mi^2) of this region is encompassed in the coastal watershed National Oceanic and Atmospheric Administration, 2003). Based on this information, total discharge of fresh water into Mississippi Sound averages 882.4 m^3/s (30,806 ft^3/s), excluding inflow from Mobile Bay, Ala.

Areas that support seagrasses within Mississippi's coastal waters include the Gulf Islands National Seashore (GINS), specifically Ship, Horn, and Petit Bois Islands, and Cat Island, which was partially purchased as an addition to the GINS. Two additional areas along the immediate coast, one at the margins of the Grand Bay National Estuarine Research Reserve at the eastern boundary of the State and the other at the western edge adjacent to Buccaneer State Park, complete the list of estuarine and marine areas within the State that support seagrasses. All of these areas fall within the boundaries of a single water body, the Mississippi Sound.

Statewide Status and Trends

Recent estimates of seagrass coverage based on 1992 aerial imagery (NBS, 1992; Moncreiff and others, 1998) indicate that only 3% of the bottom of the Mississippi Sound supports seagrass, with a total of 809 ha (1,999 acres) of seagrass, despite the sound having an average depth of 2 m (6 ft) and approximately 6,000 ha (14,826 acres) that is believed to be capable of supporting seagrasses. This 1992 estimate of seagrass coverage represents a substantial loss in cover when compared to previous 1969 estimates of 5,254 ha (12,983 acres) of seagrasses.

Causes of Change Statewide

The primary vector for the continued disappearance of seagrasses is thought to be an overall decline in water quality. The primary vector for the historical disappearance of seagrasses is thought to be a combination of physical disturbances associated with tropical weather systems, depressed local salinities associated with flood events, and an overall decline in water quality, which may have a deleterious effect on certain species of seagrasses.

[1]Botanical Consulting and Analyses, Ocean Springs, Miss.

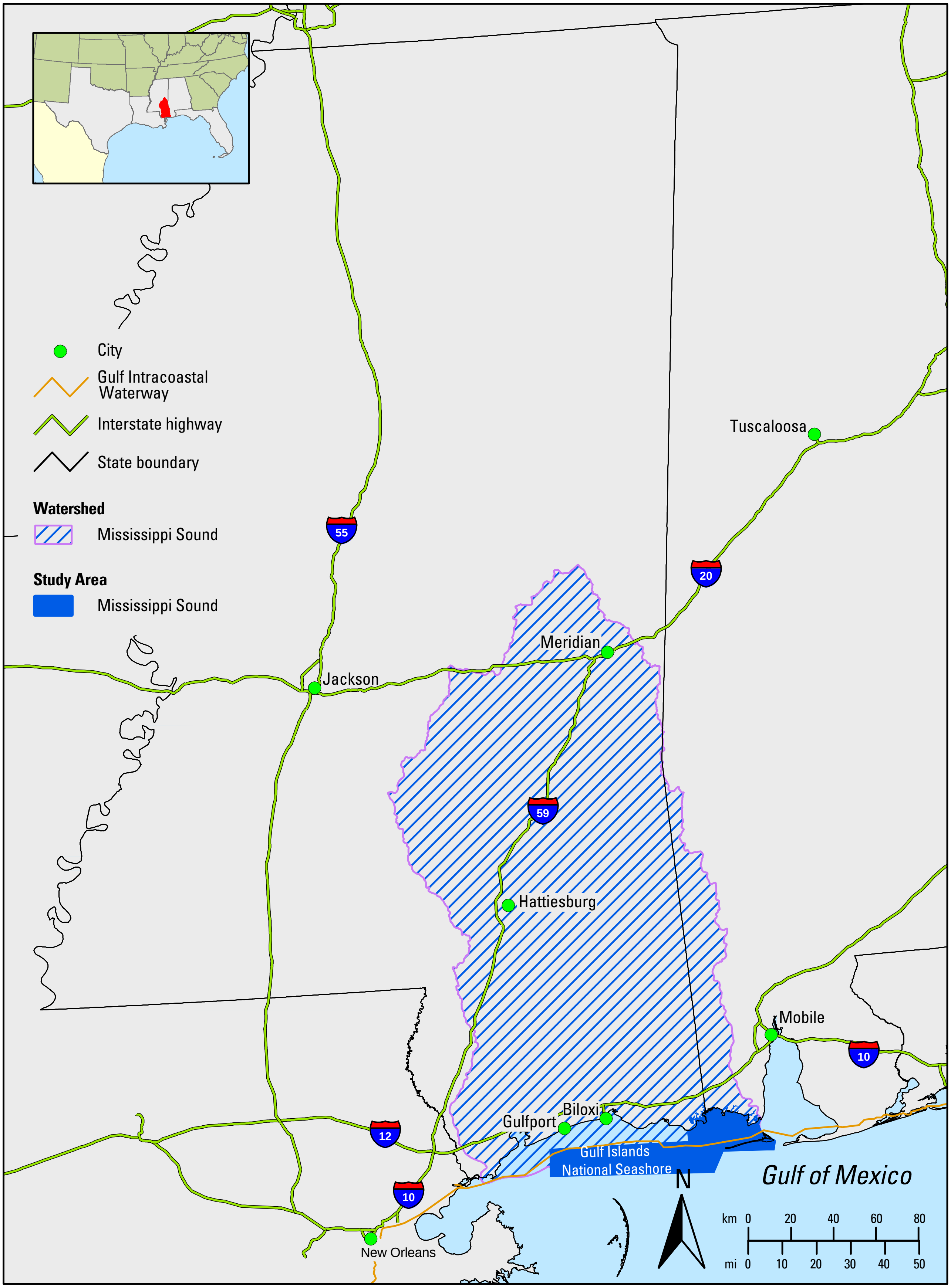

Figure. 1. Watershed for the State of Mississippi.

Gaps in Data Coverage

Regions within the State that lack detailed seagrass data where we believe seagrasses to exist are primarily within mainland coastal bayous that contain ephemeral beds of wigeon grass (*Ruppia maritima*) and the documented and extensive beds of wigeon grass that occur along the coastline in Hancock County at the western edge of the State. These beds have been observed to die back during summer and fall, exhibiting bimodal peaks in density in the late spring and in early winter. As a result, they may not have been detected during the selected index period for mapping.

Information on seagrass distribution in the coastal waters of Mississippi is limited. Seagrass was first mapped in 1967–69 as a component of the Cooperative Gulf of Mexico Estuarine Inventory and Study (Eleuterius, 1973); no estimates or measures of seagrass density were made, and the original data were destroyed by Hurricane Camille in August of 1969. Our most recent historical maps of seagrass beds and potential seagrass habitat (Moncreiff and others, 1998) are based on 1992 aerial imagery and maps prepared by the National Biological Service, now the U.S. Geological Survey (USGS) National Wetlands Research Center (NWRC). Aerial photographs taken in 1999 for the Gulf Islands National Seashore are in the process of being photointerpreted by contractors and NWRC.

Differences in the types and classes of seagrass cover reported by Eleuterius (1973) and in the information provided by the NWRC precludes any direct comparisons or estimates of change outside of the loss of species and changes in seagrass acreage. Future mapping efforts at more frequent intervals to address the issue of the development of synoptic datasets and the use of a similar classification system for each mapping effort to produce comparable data for monitoring of seagrass change is needed.

Overview of Seagrass Restoration Efforts

Activities initiated to address seagrass loss include a Federal ban on trawling within a 1.6 km (1 mi) distance of the shoreline of the GINS. Along the mainland, sand beach restoration, erosion control, and creation is limited to areas that do not support seagrasses. In addition, beach maintenance and creation is prohibited in areas that are within 0.8 km (0.5 mi) of existing seagrass beds. Mississippi's comprehensive coastal management plan complies with Federal regulations regarding seagrasses and seagrass habitat.

A community-based pilot seagrass restoration project, funded by the Gulf Restoration Network, is planned for the near future within the boundaries of the GINS. Viable turtlegrass (*Thalassia testudinum*) plants will be collected following any major storm events from wrack lines in the Perdido Key area of northwest Florida and then replanted along the north shore of Horn Island in the vicinity where turtle grass was last documented to have occurred in coastal Mississippi waters.

Overview of Monitoring, Restoration, and Enhancement Opportunities

Programs that target the monitoring, restoration, and enhancement of seagrasses within the State are currently limited to grant-supported efforts. Mitigation associated with the development of coastal areas for use by the casino industry may provide a venue for other means of restoring or enhancing seagrasses and seagrass habitat.




The Mississippi Department of Marine Resources is also charged with seagrass mapping and monitoring as one of its responsibilities. Future mapping may well be undertaken by this agency as a component of its marine resource monitoring and geographic information system based mapping efforts.

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Mississippi Sound and the Gulf Islands

By Cynthia A. Moncreiff[1]

Background

Seagrasses in Mississippi Sound were likely first documented by H.J. Humm (1956), though there are earlier descriptions of marine angiosperms associated with the barrier islands of Louisiana and Mississippi (Loyd and Tracy, 1901). Prior to Humm's work, it was believed that seagrasses, with the exception of wigeon grass (*Ruppia maritima*), occurred only very rarely between Bay County, Fla., and Aransas County, Tex. (Thorne, 1954). Humm (1956) described extensive beds of seagrasses along the northern margins of Mississippi's barrier islands, dominated by turtle grass (*Thalassia testudinum*), and indicated that turtle grass was the dominant seagrass in Mississippi Sound. He also documented the presence of manatee grass (*Syringodium filiforme*), shoal grass (*Halodule wrightii*), and star grass (*Halophila engelmannii*), in addition to the presence of previously reported beds of wigeon grass.

The location of most of Mississippi's seagrasses along the north shorelines of the State's barrier islands may be a key factor in the recruitment and survival of commercially harvested penaeid shrimp and blue crabs (*Callinectes sapidus*) and of many noncommercial species of shellfish and finfish. Seagrasses are often the closest structures to the tidal passes between the barrier islands. Studies of local food webs indicate that seagrass epiphytes are critical components of the base of the food web in the vicinity of the barrier islands (Moncreiff and Sullivan, 2001). Sampling within the seagrasses indicates that popular game fish species, such as "speckled trout" (spotted seatrout, *Cynoscion nebulosus*) and small sharks, also rely on this habitat for shelter and food.

Seagrasses were first mapped throughout the Mississippi Sound as a component of the Cooperative Gulf of Mexico Estuarine Inventory and Study, Mississippi, by L.N. Eleuterius between 1967 and 1969 (Christmas, 1973). All five previously mentioned seagrass species were again present and relatively abundant, though the beds of turtle grass described by Humm (1956) at Ship Island had disappeared as of March 1969 (Eleuterius, 1973); this period may have been the beginning of seagrass loss in Mississippi Sound. Stressors that might have caused seagrasses to become diminished or to disappear from Mississippi coastal regions likely resulted from the cumulative effects of human activities in the coastal marine environment. These activities include historical commercial uses and present-day recreational uses of seagrass habitat in addition to a number of other factors which may directly or indirectly impact seagrasses. Development may be a major factor, as it often results in higher sediment loads, introductions of contaminants, and elevated nutrient levels, which all can contribute to a loss of water quality, thus affecting seagrass communities (see watershed of area in fig. 1).

Land use and land-use changes in the eight watersheds feeding into the Mississippi Sound which may have an effect on seagrass resources include (1) a shift from the historical focus on agriculture and forestry for the paper and lumber industries to urban development related to the casino industry and (2) a shift in the State's focus to port development, plastics, and chemicals as regional industries. Land use in the region consists of silviculture and agriculture (horticultural crops and row crops) and urban development (residential and commercial) associated with a total of 11 water-based casino complexes located in Harrison and Hancock Counties. Industrial development includes commercial shipping, shipbuilding, phosphate rock refinement for phosphate fertilizer, and three electric power generating complexes; these industries are primarily located in Harrison and Jackson Counties. As of 2001, the three coastal Mississippi counties that make up most of the northern border of the sound had a total population of 366,263 within a land area of 462,315 ha (1,142,365 acres); this density constituted 12.8% of the State's population at 0.79 persons per ha (0.32 persons per acre), over three times the population density for the entire State. In addition, the region has experienced a 21.8% increase in population over the past decade, which is over twice that observed for the entire State (U.S. Census Bureau, 2002).

Weather patterns in the northern Gulf of Mexico may contribute to local patterns in the distribution of seagrasses. Mean annual precipitation is 154 cm (61 inches), resulting from an average of 75.7 days with thunderstorms (Office of Nuclear Waste Isolation, 1983); runoff associated with this rainfall contributes to the relatively high turbidity observed in coastal Mississippi waters. Hydrological changes over time within Mississippi Sound that may have a potential effect on seagrass resources involve the maintenance of two channels for port access, the Pascagoula and Gulfport ship channels, both of which have been deepened for access. The former disrupts longshore sediment transport, and the latter acts as

[1] Botanical Consulting and Analyses, Ocean Springs, Miss.

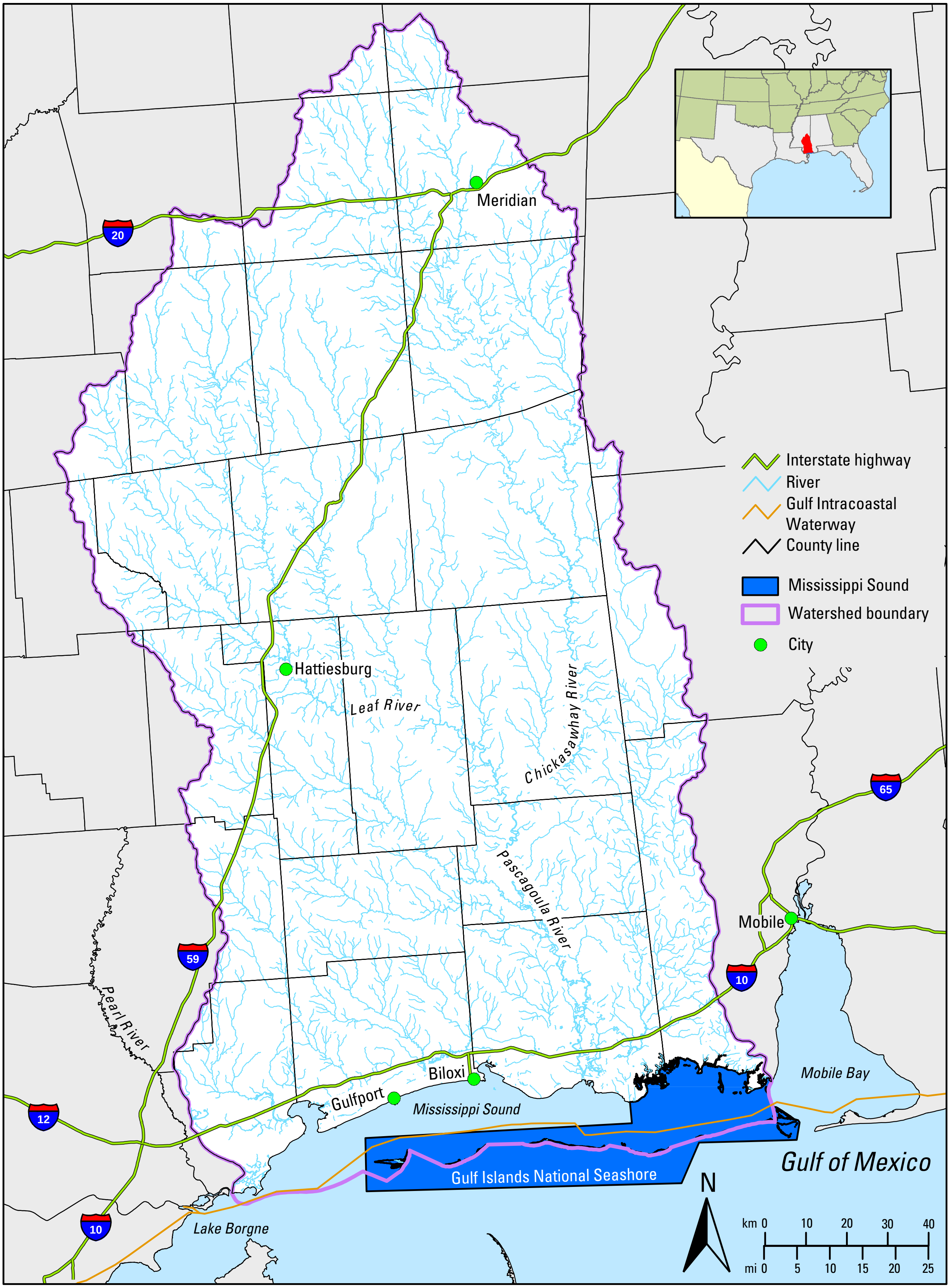

Figure 1. Watershed for Mississippi Sound.

a hydrological barrier in the observed patterns of surface circulation of water. An additional constructed feature which at times contributes tremendous amounts of fresh water to Mississippi Sound is the Bonnet Carre Spillway, a flood-control structure on the Mississippi River that is designed to protect New Orleans, La., from spring floods. Water from the Mississippi River passes over this structure into Lakes Pontchartrain and Borgne in Louisiana and then directly into Mississippi Sound. Salinities in Mississippi Sound can remain depressed for months after the spillway has been employed for flood relief and have been documented at or near zero over the western two-thirds of the sound for 2 to 3 months following opening of the spillway (H. Perry, oral commun.). Tropical storms and hurricanes are an additional feature of regional climate that can also directly affect seagrasses. Species distributions noted along the north shore of Horn Island by Sullivan in 1977 (Sullivan, 1979) suggested that the passage of Hurricane Frederick in 1979, combined with openings of the Bonnet Carre Spillway in 1979 and 1981, may have contributed to the demise of the more halophytic seagrasses in the sound.

The part of Mississippi Sound that supports the majority of the region's seagrass area lies within the boundaries of the Gulf Islands National Seashore. The barrier islands off of the Mississippi coast were incorporated into the Gulf Islands National Seashore by Congress in January 1971; Cat Island was added to the Gulf Islands National Seashore on March 28, 2002.

Important yet essentially unanswered questions about Mississippi Sound's seagrasses center around the following points, shared by seagrass ecosystems on a global scale: (1) current status of seagrasses in terms of areal coverage, species composition, standing crop, productivity, and other characteristics; (2) trends in seagrass coverage, seagrass phenology, and biology; (3) water-quality conditions needed to maintain seagrasses; and (4) conditions required to promote recovery and expansion of seagrasses.

Scope of Area

Bounded by the coast of Mississippi to the north, by Mobile Bay, Ala., to the east, by Lake Borgne, La., to the west, and by a series of barrier islands to the south that make up most of the Gulf Islands National Seashore (fig. 2), Mississippi Sound has a long history of coastal water transport by Native Americans and a series of European explorers, including Bienville and D'Iberville, who established a colony on its shores in 1699. Geologically, the entire area is similar, with most of the seagrasses occurring along the barrier islands, which are slowly migrating to the west (Otvos, 1981). Tidal amplitudes are low; average astronomical tidal range is 0.6 m (1.96 ft). The effects of wind on local hydrodynamics generally overwhelm the effects of tides and tend to dictate local water depth and surface level fluctuations. The climate is categorized as semitropical or subtropical, winds are typically from the south-southeast at 10.4 kph (6.5 mph), and October tends to be the driest month of the year (Office of Nuclear Waste Isolation, 1983). Rivers draining to the region tend to carry high sediment loads. Mississippi Sound also receives waters from the Pascagoula River, one of the last undammed rivers in the continental United States.

Methodology Employed To Determine and Document Current Status

The most current mapping study of seagrass coverage for the Mississippi Sound and the Gulf Islands area was conducted more than 10 yr ago by the U.S. Geological Survey (USGS) National Wetlands Research Center (NWRC) by using natural-color aerial photography taken in 1992 at a 1:24,000 scale as part of the northeastern Gulf of Mexico seagrass mapping project. In cases where the data were inadequate or incomplete, contemporary supplemental data were acquired from other sources (Moncreiff and others, 1998; Moncreiff, personal observation) and used to complete the photographic coverage.

Natural color, 1:24,000-scale aerial photography was acquired in June 1992; however, the primary data source was natural color, 1:24,000-scale aerial photography flown by NASA-Stennis in fall 1992. This photography was used in map generation and included stereoscopic photointerpretation, cartographic transfer, and digitization in accordance with strict mapping standards and conventions. Other important aspects of the protocol included the development of a classification system (described below), groundtruthing, quality control, and peer review. Seagrass distribution information derived from the photography was transferred by using a zoom transfer scope onto a stable medium overlaying USGS 1:24,000-scale quadrangle base maps.

The seagrass classification system used in the 1992 dataset consisted of two classes of open water—RIV (riverine, fresh water) and EST (estuarine or marine open water)—and five classes of seagrass habitats. One class of continuous seagrass (CSG) for which no density distinction was made and four classes of patchy seagrass were based on percent ground cover of patches in 5% increments: PSG1 (0%–10%, very sparse), PSG2 (15%–40%, sparse), PSG3 (45%–70%, moderate), and PSG4 (75%–95%, dense). All seagrass cover in Mississippi waters was very sparse to sparse, falling within the PSG1 and PSG2 classes.

The groundtruthing phase included the participation of field staff from Gulf Islands National Seashore, Dauphin Island Sea Lab, and Mississippi State University. Draft maps were sent out to the aforementioned agencies and staff for review and comments. All comments received were incorporated into the final maps.

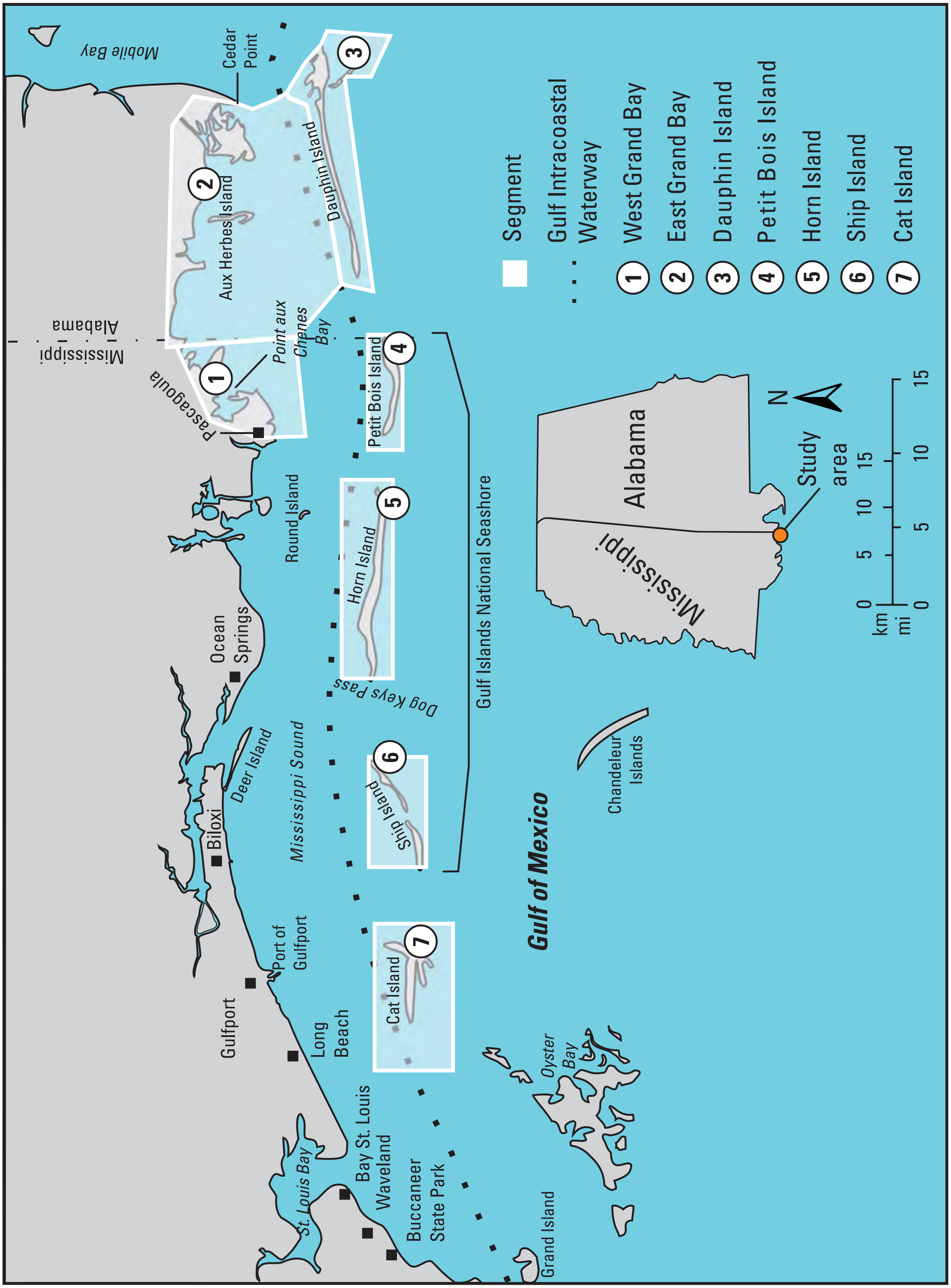

Figure 2. Scope of area for the Mississippi Sound seagrass vignette.

Methodology Employed To Analyze Historical Trends

Only one study of trends in seagrass cover over time has been conducted for Mississippi Sound. The results of this study, summarized in a report to the Mississippi Department of Marine Resources (Moncreiff and others, 1998), are limited by the lack of consistency in current and prior seagrass mapping efforts but are considered here because they constitute the sole effort in this area at this point in time. Hectares of seagrass coverage in Mississippi Sound were calculated from the Eleuterius (1973) 1967−69 dataset as well as from the data in the NWRC preliminary maps (U.S. Geological Survey, 1992) of seagrass distributions in Mississippi Sound. The area of potential seagrass habitat was also determined by using the 2-m (6.6-ft) depth contour as a criterion.

Distribution data for submerged aquatic vegetation in Mississippi Sound from Eleuterius (1973) and groundtruthed data from the 1992 NWRC aerial imagery study (described in the previous section) were plotted onto hydrographic survey maps. Potential seagrass habitat in Mississippi Sound was also identified and mapped by using a 2-m (6.6-ft) critical depth limit established by Heck and others (1994, 1996) in a previous seagrass project documenting distribution and abundance of seagrass along the Gulf Islands National Seashore. The 2-m (6.6-ft) depth contours used were taken from National Oceanic and Atmospheric Administration navigational charts (National Oceanic and Atmospheric Administration, 1996a, b).

Status and Trends

Information on seagrass distribution in Mississippi Sound is limited. Our most recent “historical” maps of seagrass beds and potential seagrass habitat are based on 1992 aerial imagery and maps prepared by NWRC (fig. 3). Aerial photographs taken in 1999 for the Gulf Islands National Seashore were photointerpreted by staff at NWRC for this report. Seagrass distributions from a 1969 Gulf of Mexico estuarine inventory were used as a source of historical documentation, while data from the 1992 NWRC aerial imagery study were used to provide more recent distribution patterns. Potential seagrass habitat was identified by using a 2-m (6.6-ft) critical depth limit which was previously established by Heck and others (1994, 1996) and described in a National Park Service seagrass monitoring project report. Hectares of seagrass in Mississippi Sound were calculated from the Eleuterius (1973) 1967–69 dataset as well as for the data from the NWRC maps (U.S. Geological Survey, 1992) of seagrass distributions in Mississippi Sound (fig. 4).

Table 1 provides the estimated historical, present, and potential areal extent of seagrass in Mississippi Sound. In 1969, about 5,254 ha (12,982 acres) of seagrasses were documented, and as of 1992, only 600 of those (1,482 acres) were vegetated. An additional 209 ha (516 acres) of seagrass were also documented in 1992 in areas that were excluded from Eleuterius’ (1973) work, for a total of 809 ha (1,999 acres) of seagrass in 1992. Imagery from 1999 suggests that seagrass coverage was slowly increasing throughout the sound, with an apparent dramatic increase in areal extent of seagrasses at Cat Island; however, 2,023 ha (4,999 acres) surrounding Cat Island were covered with several species of attached macroalgae during the 1969 survey and were not included in the seagrass cover estimates. In total, 6,024 ha (14,885 acres) of coastal Mississippi water bottoms have been identified as suitable habitat for seagrasses.

Information from the Eleuterius (1973) studies indicated that 67.6% of potential seagrass habitat was vegetated, in comparison to only 13.4% in 1992. Physical loss of seagrass habitat is assumed for areas where 1969 coverage exceeded current estimates of seagrass habitat; this total is estimated to be 19.6%. Loss of previously vegetated areas which still fall within the regions delimited as potential seagrass habitat, even when corrected for physical loss of habitat, totals 54.2%. Seagrass communities in Mississippi Sound need to be routinely mapped to determine if this roughly 50% loss of vegetated habitat is in response to natural or human-caused effects.

Causes of Change

The primary reason for the disappearance of seagrasses is thought to be an overall decline in water quality (Orth and Moore, 1983; Robblee and others, 1991). On a historical time scale, disappearance of seagrasses resulted from a combination of physical disturbances associated with tropical weather systems, depressed salinities associated with flood events, and an overall decline in water quality resulting in increased turbidity (Eleuterius, 1989), which has a deleterious effect on most if not all of the species of seagrasses that occur in the northern Gulf of Mexico (Thayer and others, 1994; Zieman and others, 1994).

In Mississippi Sound, seagrasses appear to be threatened by the cumulative effects of both natural events and human activities in the coastal marine environment. For example, extended periods of depressed salinity (Montague and Ley, 1993) and physical disturbances such as tropical storms and hurricanes can contribute. Physical loss of habitat and decreased light availability combined with declining water quality are the most visible features which directly affect seagrass communities. Some observed changes in seagrass distributions appear to be directly linked to physical loss of habitat; areas of seagrass habitat loss coincide with areas where rapid coastal erosion and massive long-term movement of sand have been well documented (Otvos, 1981; Oivanki, 1994). Loss of vegetated areas corresponds with potential loss in water clarity over time, caused by (1) human influences, (2)

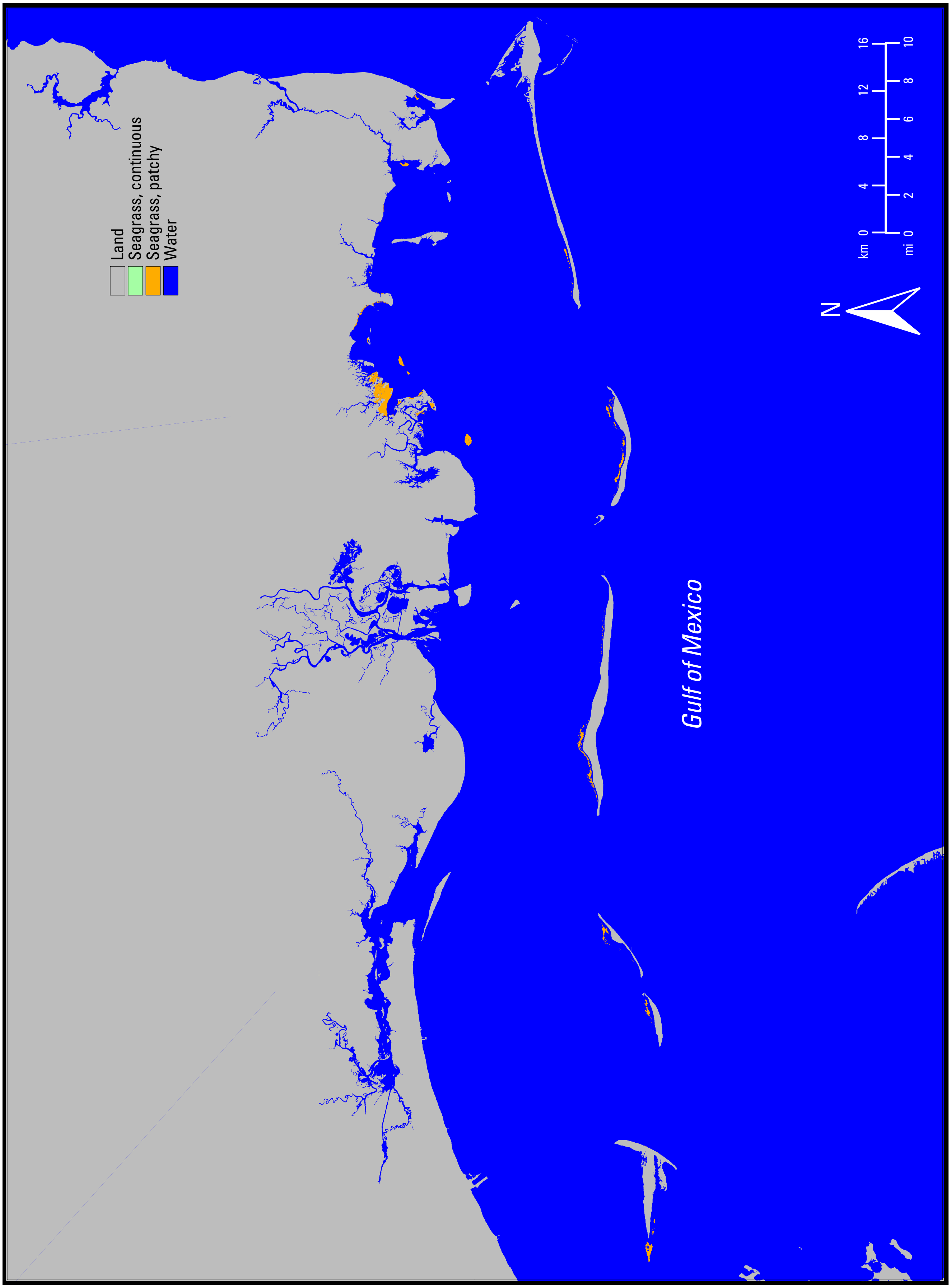

Figure 3. Distribution of seagrasses in Mississippi Sound, 1992.

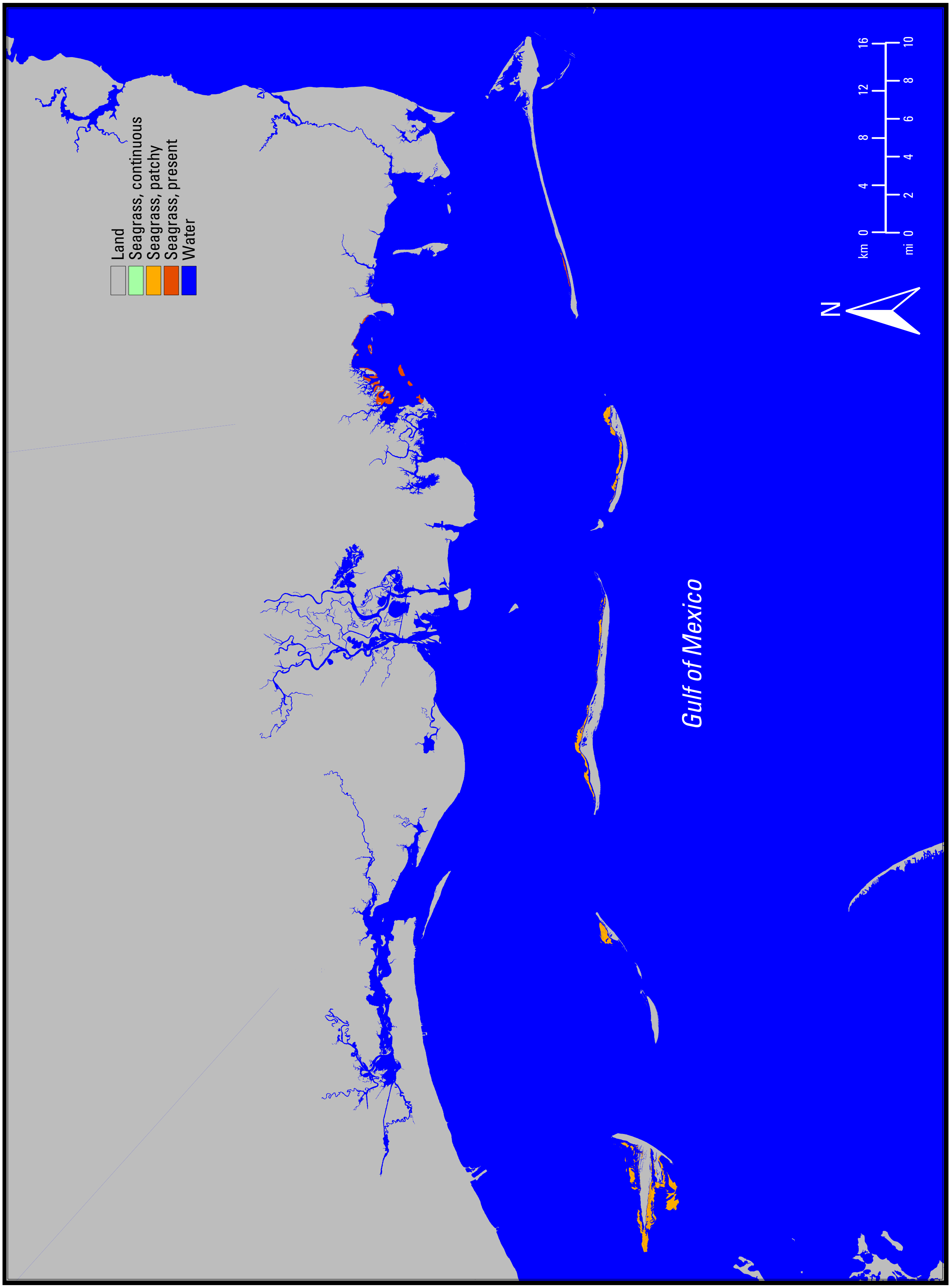

Figure 4. Distribution of seagrasses in Mississippi Sound, 1999–2001/2002.

cyclic shifts in precipitation patterns which would affect both salinity and turbidity, or (3) a combination of these factors. Some of the observed differences in coverage are also likely a result of mapping techniques; the information collected between 1967 and 1969 by Eleuterius (1973) is less precise than the detailed mapping efforts that are possible with current aerial imagery such as the photographs used to prepare the 1992 seagrass distribution maps for Mississippi Sound.

Seagrass Health

The central issue of concern in Mississippi Sound with regards to seagrass health is the genetic variation that remains in extant populations of shoal grass and wigeon grass following the dramatic losses of cover and of species between 1977 and 1987. Observations on the genetic status and implied health of populations of shoal grass are described in *2002 Annual Investigator's Report* to the National Park Service and will be published elsewhere in the near future. Studies of the phenology and genetic diversity of wigeon grass are in progress.

Critical points excerpted from the report to the National Park Service, which summarizes information collected for the thesis research of Opel (2002), are that in theory, because of the predominance of vegetative growth, shoal grass beds should be clonal in composition, and local populations of shoal grass may have little to no genetic variation caused by environmental and geographic constraints on their growth and reproduction. To test hypotheses related to the genetic composition of shoal grass, samples were collected from four populations in the northern Gulf of Mexico, including the Chandeleur Islands, La.; Petit Bois Island and central Horn Island, Miss.; and Perdido Key, Fla. DNA was extracted from these samples and used as a template for genetic analysis; genetic variation was detected at all levels within single seagrass beds, within populations, and among the four populations. Shoal grass in the northern Gulf of Mexico has a moderate amount of genetic diversity when compared to other submerged, marine angiosperms, all of which use hydrophilous pollination and have predominance towards vegetative growth. Data also indicate that shoal grass populations from Petit Bois Island may have a genetic component associated with phenotype; specifically, blade and shoot lengths may have a genetic component, as opposed to simply growing in response to environmental conditions.

The information generated from this thesis project (Opel, 2002) indicates that seagrass conservation techniques, such as transplanting, need to be done with utmost care to maintain the current ecology of the northern Gulf of Mexico. For example, certain individual genotypes exist in shoal grass populations within Mississippi Sound that have survived the reduction in seagrass distribution over the last 30 yr. If "foreign" genotypes

Table 1. Estimated areas of seagrasses in Mississippi Sound during 1969, 1992, and 1999, plus a bathymetry-based estimate of potential seagrass habitat (PSGH).

[Area is given in hectares and acres]

Location	1969	1992	Difference 1969–92	1999	Difference 1992–99	PSGH
Buccaneer State Park	83 ha 205 acres	22 ha 54 acres	61 ha 151 acres	No data	22 ha 54 acres	128 ha 316 acres
Cat Island	242 ha 598 acres	68 ha 168 acres	174 ha 430 acres	645 ha 1,594 acres	577 ha 1,426 acres	2,075 ha 5,127 acres
Ship Islands	621 ha 1,534 acres	103 ha 254 acres	518 ha 1,280 acres	98 ha 242 acres	5 ha 12 acres	648 ha 1,601 acres
Dog Keys Pass	841 ha 2,078 acres	0 ha 0 acres	841 ha 2,078 acres	No data	0 ha 0 acres	465 ha 1,149 acres
Horn Island	2,253 ha 5,567 acres	215 ha 531 acres	2,038 ha 5,036 acres	234 ha 578 acres	19 ha 47 acres	1,760 ha 4,349 acres
Petit Bois Island	684 ha 1,690 acres	147 ha 363 acres	537 ha 1,327 acres	172 ha 425 acres	25 ha 62 acres	732 ha 1,809 acres
Point-aux-Chenes Bay	528 ha 1,305 acres	45 ha 111 acres	483 ha 1,194 acres	No data	45 ha 111 acres	216 ha 534 acre
Totals	5,252 ha 12,977 acres	600 ha * 1,483 acres	4,652 ha 11,496 acres	1,149 ha 2,839 acres	693 ha 1,712 acres	6,024 ha 14,885 acres

* Areas at the northeastern margin of Mississippi Sound were omitted from the Eleuterius (1973) survey; this region supported 209 ha (516 acres) of vegetation in 1992 for a total of 809 ha (1,998 acres) of seagrasses.

are introduced, these transplants may not survive the factors that led to the seagrass decline historically observed. If these new plants do survive, they may dilute the genes and genotypes that were initially adapted to that location. Therefore, transplanting from one population to another may have no real benefit as it may be inefficient and costly in terms of money and of disruption to the gene pool at each population. Genetic information suggests that the examined shoal grass populations would benefit most from use of local plant material in conservation and restoration efforts.

Species Information

Extant seagrass populations off of the coast of Mississippi consist almost exclusively of shoal grass. Historically, populations of shoal grass, star grass, wigeon grass, manatee grass, and turtle grass were present and abundant along the northern shores of the Mississippi barrier islands (Humm, 1956; Eleuterius, 1973; Eleuterius and Miller, 1976; Sullivan, 1979). Overall, Mississippi has lost half of the seagrass area that was present in the 1967–68 time frame and has virtually lost all but one of its marine seagrass species (Heck and others, 1994; Moncreiff and others, 1998). Only shoal grass exists in any type of measurable area throughout the southern portion of Mississippi Sound. Wigeon grass occurs in isolated but well-developed patches along the immediate coastline and as a rare component in shoal grass beds along the barrier islands in Mississippi Sound (Heck and others, 1996; Moncreiff and others, 1998). Peak development of wigeon grass may be bimodal in this subtropical region; it may not coincide with time frames that are optimal for development and mapping efforts for other seagrasses. Studies of the phenology of this species in coastal Mississippi waters are in progress to better document the time frames in which wigeon grass is most abundant for accurate mapping of this essential aquatic habitat and protected resource.

Well-established populations of shoal grass, star grass, wigeon grass, manatee grass, and turtle grass exist along the western shorelines and in the small internal bayou systems of the Chandeleur Islands. Though technically part of southeastern Louisiana, these islands begin 35 km (22 mi) south of Biloxi, Miss., and are likely a source of vegetative propagules and possibly of seeds supplementing or repopulating seagrass beds in some areas of the Mississippi coast and even Grand Bay, Ala. (north of Dauphin Island) (Stout, 2004, personal commun.).

Seagrasses and potential seagrass habitat in Mississippi Sound lie mainly along the northern shorelines of Ship, Horn, and Petit Bois Islands. Round and Cat Islands have had well-developed grass beds in the past (Eleuterius, 1973); no seagrasses occur at Round Island at present. Cat Island seagrass populations occur at the north and west tips of this T-shaped island, as well as in protected areas along its southwest shoreline. Wigeon grass occurs along the mainland shoreline in Hancock County between Waveland and Clermont Harbor, in the Grand Bay National Estuarine Research Reserve south of the marshes in eastern Jackson County, and along the Mississippi-Alabama border.

Monitoring, Restoration, and Enhancement Opportunities

State, Federal, and academic programs that target the monitoring, restoration, and enhancement of the seagrasses in Mississippi Sound are limited to funding and grant-supported efforts outside of the assessment that is described in this document. It is hoped, however, that future seagrass monitoring will be undertaken by the State's Department of Marine Resources as a component of its marine resource monitoring and mapping efforts based on geographic information systems.

A community-based pilot seagrass restoration project, funded by the Gulf Restoration Network, will be initiated within the boundaries of the Gulf Islands National Seashore. Originally, lead investigator for the project was Robin K. McCall (University of Southern Mississippi Gulf Coast Research Laboratory). McCall planned to collect viable turtle grass shoots following any major storm events from wrack lines in the Perdido Key area of northwest Florida, hold them for a short period of time to ensure viability, and then plant the shoots along the north-central shore of Horn Island. The location selected is at an appropriate depth and is located where a sample of turtle grass was collected in June of 1993, in association with sampling for invertebrates conducted by R. Heard and J. McLelland, both of the Gulf Coast Research Laboratory. The project currently has Jerry McLelland as lead investigator, and monitoring of seagrass establishment and growth is ongoing.

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Statewide Summary for Alabama

By Diana J. Sturm,[1] Judy Stout,[2] and Tim Thibaut[3]

Background

Natural Environment

The gulf coast of Alabama extends from the Mississippi State line eastward to the Florida State line, a distance of only 74 km (46 mi) (Alabama Coastal Area Board, 1980). The coastline, however, including the estuaries and inlets, covers a greater distance of 977 km (607 mi) (National Oceanic and Atmospheric Administration, 1997). Two large drainage basins empty into the northern Gulf of Mexico within coastal Alabama: the Perdido River basin and the Mobile River basin (fig. 1). The Perdido basin encompasses 3,238 km^2 (1,250 mi^2) located in Florida and Alabama (Special Grand Jury, 1999). The Mobile basin is the sixth largest drainage area in the United States and is the fourth largest river basin in terms of flow volume. The 111,370-km^2 (43,000-mi^2) Mobile basin encompasses parts of Tennessee, Georgia, Mississippi, and Alabama (Isphording and Flowers, 1990; Johnson and others, 2002) (fig. 1).

The coastal lowlands of Alabama, with gently undulating to flat topography, basically follow the shoreline along the Gulf of Mexico and Mobile, Perdido, and Bon Secour Bays (Sapp and Emplaincourt, 1975). The ecological environments and geomorphology consist of features such as wetlands (i.e., tidal marsh), two large peninsulas, a delta, lagoons, islands, and bays. The presence of a saline and/or fresh, high water table gives rise to the abundance of various wetland habitat types that are found within Alabama's coastal area. The largest bays of coastal Alabama include Mobile Bay, Perdido Bay, and Bon Secour Bay with the largest of these, Mobile Bay, being formed within a submerged river valley (Chermock, 1974). Some of Perdido Bay is in Florida, and there is a further discussion of seagrasses there in Perdido Bay in the Florida section of this report. Further, the Mississippi Sound estuary extends from southwestern Mobile Bay, behind the offshore barrier islands, crossing the State line and bordering the entire southern Mobile County and Mississippi coastlines (fig. 2); consequently, there is another discussion of seagrasses in Mississippi Sound in the Mississippi section of this report.

Alabama has two main peninsulas: (1) Fort Morgan Peninsula at the mouth of Mobile Bay, and (2) Perdido Key at the mouth of Perdido Bay. The Fort Morgan Peninsula is an elongated stretch of land extending westward from south Baldwin County and bounded by Bon Secour/Mobile Bay to the north and west and by the Gulf of Mexico to the south. Perdido Key is located at the southeastern extent of Baldwin County and is bounded to the north by Old River, to the west by Perdido Pass, and to the south by the Gulf of Mexico. Alabama has at least 10 coastal islands, and of these, Dauphin Island, defining the western mouth of Mobile Bay, is the largest.

The confluence of the Tombigbee and Alabama Rivers in north Mobile and Baldwin Counties forms the Mobile River. The Mobile, Tensaw, and Blakeley Rivers flow southward to Mobile Bay through the Mobile-Tensaw Delta. The alluvial-deltaic plain is located at the terminus of Mobile Bay, northward along the Mobile-Baldwin County line. Topographically, the Mobile-Tensaw Delta is flat and generally below 6 m (20 ft) in elevation. Additionally, other major coastal tributaries include Dog River and East Fowl River on the western side of Mobile Bay; the Blakeley, Fish, Magnolia, and Bon Secour Rivers on the eastern side of the Bay; West Fowl and Escatawpa Rivers discharging into Mississippi Sound; and the Perdido and Blackwater Rivers at the northern end of Perdido Bay.

Mean annual precipitation in the Mobile and the Perdido basins ranged from 135.6 cm/yr (53.4 inches) in Montgomery, Ala., to 162.6 cm/yr (64 inches) in Mobile, Ala., over the time period 1961–90 (National Oceanic and Atmospheric Administration, 1997). The Alabama area of the central gulf coast has one of the highest frequencies of hurricane landfalls in the United States. Tropical storms can vary the annual precipitation considerably (Hurricane Danny yielded 94 cm (37 inches) in 24 h in July 1997), causing tides and water movement that can redistribute bay bottom sediments and reconfigure shorelines (Schroeder and others, 1990). Conversely, droughts can reduce the precipitation dramatically, as in the case in 2000 when Mobile received an average of only 116.2 cm (45.7 inches) of precipitation (Coastal Weather Research Center, 2003).

With an average 1,756 m^3 (62,004 ft^3) per second of flow, upstream riverine inputs significantly influence water quality

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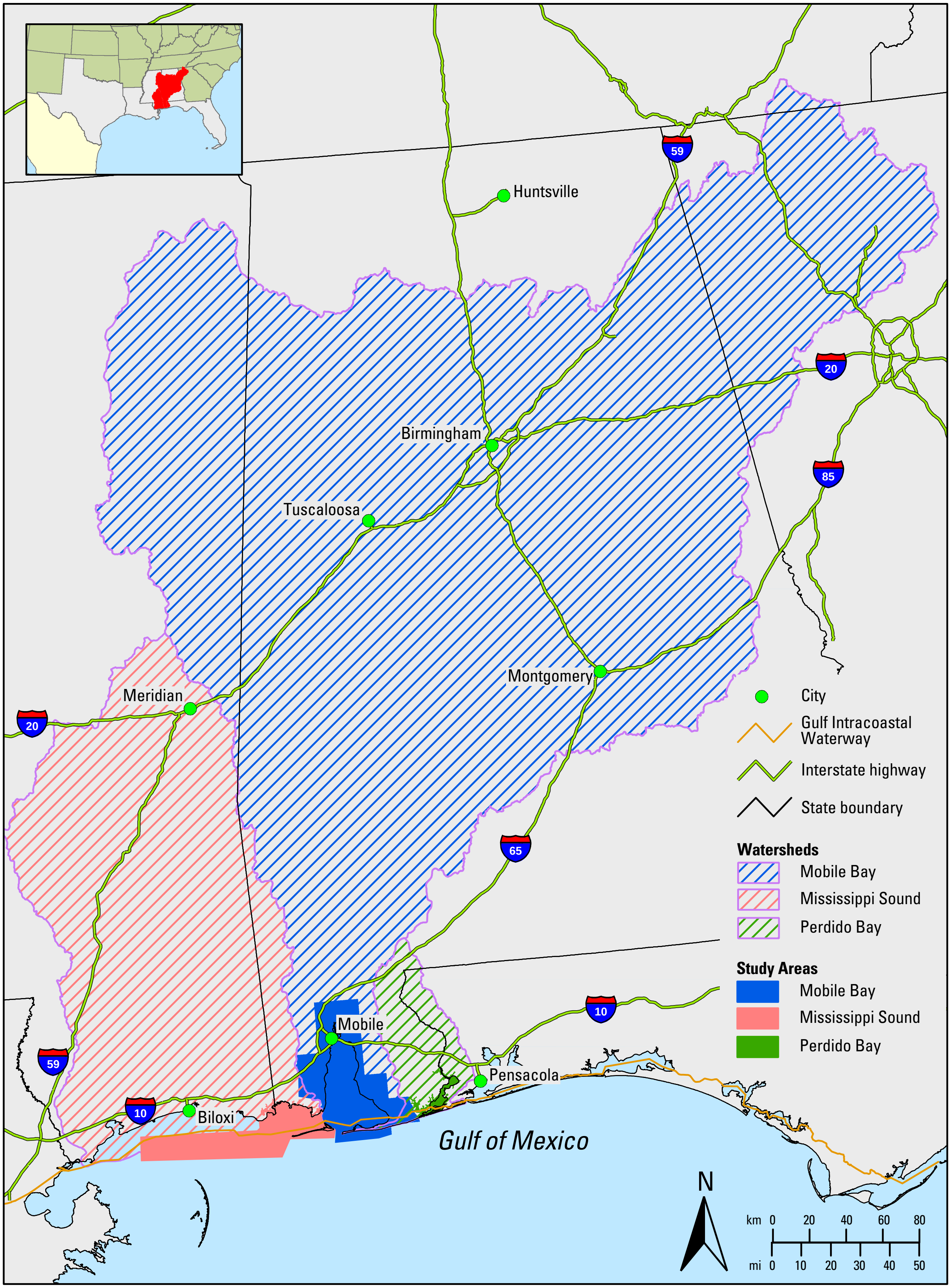

Figure 1. Watershed for coastal Alabama.

in Mobile Bay. Low upstream flows typically occur during late summer and early fall months when certain water-quality parameters (e.g., nutrient enrichment, salinity, dissolved oxygen levels) are most critical. During this time, coastal rainfall amounts and flow inputs from local sources may be more significant to the system (Baya and others, 1998).

Coastal waters of Alabama are primarily under the influence of a daily astronomical tide with a mean range of only 0.4 m (1.3 ft), a maximum spring tide range of 0.8 m (2.6 ft), and a minimum neap tide range of less than 0.1 m (0.3 ft). The seasonal cycle of tides is low in winter and early summer and high in spring and late summer through fall. Winds and storms can significantly affect tide levels. The most recent estimate for sea-level change along the northern gulf coast is more than 1.8 mm (0.07 inches) a year or more than 0.18 m (0.59 ft) per century. Recorded salinity ranges are 0–35 ppt in the lower bay and 0–24 ppt in the upper bay. Vertical salinity differences as high as 10 to 15 ppt and 20 to 30 ppt have been observed in the northern and southern ends of the bay, respectively (Schroeder and others, 1990).

The unconsolidated alluvial sand, gravelly sands, and clays found along the Alabama coast, when combined with varying amounts of precipitation, cause dramatic effects on the turbidity of the shallow receiving waters in Mississippi Sound and Perdido and Mobile basins. An estimated 4.85 million Mg (5.35 million tons) of sediment annually enters the Mobile Bay estuary, with 33% deposited in the delta and 52% in the bay (the remainder being transported to the Gulf of Mexico and Mississippi Sound) (Isphording and others, 1996). During periods of high precipitation, visibility may be reduced to 15 cm (6 inches) or less (Sturm, personal observation, 2003). Conversely, during periods of drought, the turbidity can reach 5 nephelometric turbidity units (NTU) or more (Weeks Bay National Estuarine Research Reserve, 2003). Five NTUs is equivalent to 6.3 mg (0.0002 oz) total suspended solids/L (Lake Access, 2003). Jackson (2004) found that light attenuation averaged 142.5 cm (56 inches) on quiescent days while averaging only 55.0 cm (22 inches) on days when weather fronts passed through the area, thus increasing the quantity of suspended sediments and seston material in the water column. Lastly, winds and storms cause substantial increases in turbidity when they are greater than 24 kph (15 mph) since Mobile Bay is a shallow, wind-driven system.

Context of Human Influences

Alabama coastal systems have been subjected to increasing pressure from a variety of proliferating activities including commercial and recreational fishing, silviculture, oil and gas extraction and refining, shipping and navigation channel excavation, industrial construction and waste discharges, residential development, municipal waste treatment discharges and accidental spills, and nonpoint source runoff. Surveys in 1998 by the Mobile and Baldwin County offices of the Natural Resources Conservation Service (file reports, Mobile and Bay Minette offices, respectively) indicated that 45% of Mobile County and 32% of Baldwin County were developed (urban, agriculture, or pasture). Of the remainder, which was classified as “forested,” an unreported portion was not natural habitat but managed for silviculture.

The Mobile metropolitan area (Mobile and Baldwin Counties) had a combined 2002 population of 551,578 (*http://cber.cba.ua.edu/edata/est_prj/alpop20002025.prn*, accessed May 18, 2004), a growth since 1990 of 51% for Baldwin County and 5.6% for Mobile County. Projections of additional growth from 2002 to 2025 of 63.6% and 10.9% for each county, respectively, result in an anticipated coastal population of 691,989 in 2025.

Current discharges of secondarily treated sanitary waste average 1.58 m^3/s (36,000,000 gallons per day) (Mobile Metropolitan Area Statistical Abstracts), but many areas are not served by waste treatment facilities; numbers of permits issued in 2003 for individual septic tanks were 1,619 for Mobile and 1,027 for Baldwin County (Mobile Bay Convention and Visitors Bureau, MBCVB Database, *http://www.southalabama.edu/mcob/cber/DB-port.htm*, accessed May 17, 2004). Aging sanitary waste collection systems and undersized treatment facilities have resulted in numerous large spills of raw sewerage in the last decade.

The Port of Mobile handled 35,587,125.4 Mg (39,227,431 tons) of cargo in 2003, with the principal coastal shipping route being the dredged and regularly maintained Mobile Ship Channel traversing Mobile Bay from north to south. Shallower draft shipping (barges) uses the depth-maintained Gulf Intracoastal Waterway through Mississippi Sound, crossing lower Mobile Bay and connecting to Perdido Bay by an artificial, excavated canal across Baldwin County. Coastal seafood landings were over 10,433 Mg (11,500 tons) in 2002, primarily delivered through dredged navigation channels into Bayou La Batre (off of Mississippi Sound) and the Bon Secour River in southeastern Mobile Bay (MBCVB Database, *http://www.southalabama.edu/mcob/cber/DB-port.htm*, accessed May 17, 2004).

In 1997, Pickel and Douglass (1998) used historical aerial photographs (1955, 1974, and 1985), surveys, and video (1997) to investigate the impact of shoreline armoring around Mobile Bay.

Approximately 46,756 m (153,400 ft) of the shoreline was armored in 1997, leaving 110,368 m (362,100 ft) still unarmored. Armoring increased from 8% of the shoreline in 1955 to 30% in 1997, and the period of greatest increase was 1974–85. There was a complete loss of intertidal habitat in front of bulkheaded properties, and it was predicted that this hard stabilization of bay shorelines would lead to a decrease in intertidal habitat by converting horizontal beach recession into vertical erosion in the estuarine fringes (Pickel and Douglass, 1998).

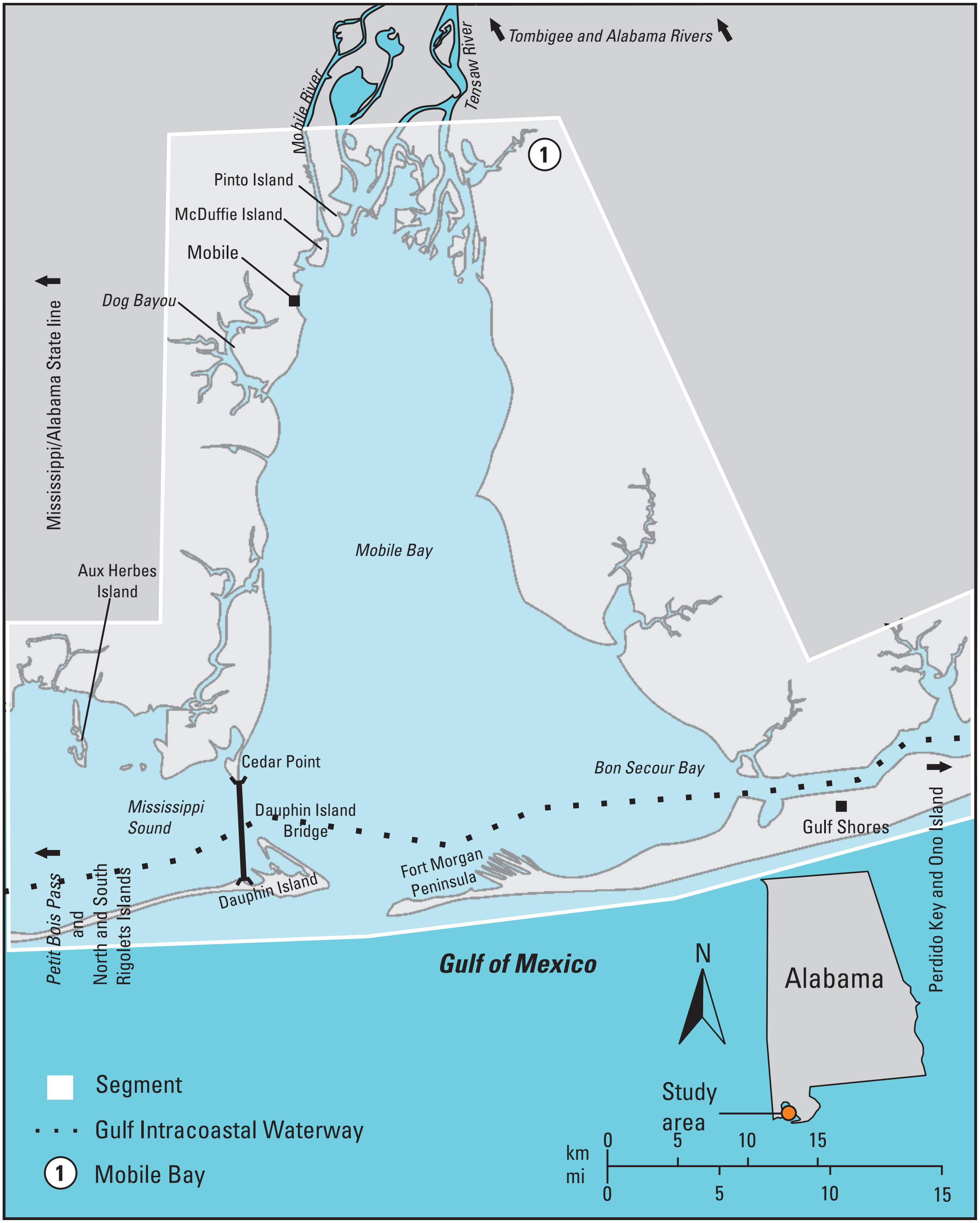

Figure 2. Scope of area for the Alabama summary.

Scope of Area

This report addresses seagrasses occurring throughout coastal Alabama, extending from the Mississippi State line eastward to the Florida State line and including all estuaries and drainage basins, as described below (fig. 2). Efforts have been made to include only those species considered seagrasses; however, because earlier reports included area totals for geographic areas with both freshwater aquatics and seagrasses, it was not always possible to separate the species (particulary in the lower delta and upper bay).

Only the portion of Mississippi Sound located in Alabama is included in this chapter. The Alabama portion of Mississippi Sound is bordered on the west by North and South Rigolets Islands and Petit Bois Pass west of Dauphin Island and by the Dauphin Island bridge on the east.

Only seagrasses in the portion of Perdido Bay located in Alabama are included in this chapter. The eastern border of coastal Alabama is defined along the coast by the State line splitting Perdido Bay down the center, snaking around the east end of Ono Island and crossing Perdido Key near its westward terminus.

The northern boundary of true seagrasses in coastal Alabama is defined by the salinity regime present during any given year. During years with normal rainfall, Interstate 10 and the Hwy 90/98 Causeway define the northernmost reaches of wigeon grass (*Ruppia maritima*). During drought years, however, wigeon grass may extend above the causeway into beds of submerged aquatic vegetation (SAV), which are almost exclusively freshwater aquatics.

Historical Documentation

Historical trends for seagrasses in Alabama have not been calculated. There have been few SAV surveys completed for coastal Alabama. Earliest studies by Baldwin (1957), Lueth (1963), and Beshears (1982) were limited surveys of distributions of SAV in the lower delta for wildlife and waterfowl management planning. Maps based on these data were not published. The Alabama Department of Conservation and Natural Resources conducted detailed surveys from 1987 to 1994, limited to SAV within the delta and the northern margins of Mobile Bay (Powell, 1979; Zolczynski and Eubanks, 1990; Zolczynski and Shearer, 1997). Other than occasional wigeon grass in the upper bay, seagrass beds were exclusively made up of submerged aquatic species and not seagrasses. Stout and Lelong (1981) used black and white photography, and Stout and others (1982) color-infrared aerial photography, combined with extensive ground surveys, to provide the first quantitative mapping of SAV area throughout coastal Alabama. Both SAV and seagrass species were included. Results were presented on hardcopy U.S. Geological Survey (USGS) 7.5’ quadrangle maps but not digitized.

Statewide Status and Trends

Historical Surveys

No formalized studies of seagrass trends in coastal Alabama have been undertaken as of the writing of this report. Efforts are currently underway by the Mobile Bay National Estuary Program (MBNEP) to collect all available information to develop a historical estimate of SAV along the Alabama coast and to analyze trends for time periods where data are adequate.

Stout and Lelong (1981) found that wigeon grass was the most ubiquitous seagrass species in the Alabama coastal area, excluding the delta, though not the most abundant, occurring near bay mouths and tributary rivers and in brackish waters of Mississippi Sound. Shoal grass (*Halodule wrightii*) was found to occur only in southern Perdido Bay and along the northern shore of the western end of Dauphin Island. Overall, wigeon grass covered 123 ha (305 acres) and shoal grass covered 266 ha (656 acres) in the 1980 survey of the coastal area (Stout and Lelong, 1981). Minor occurrences of turtle grass (*Thalassia testudinum*) were found previously among extensive beds of shoal grass in southern Perdido Bay (Stout and Lelong, 1981; Lelong, 1988).

Overall, areal coverage of SAV in coastal Alabama has decreased over time, apparently because of dredging and filling operations and from increased turbidity caused by shoreline development (Stout and Lelong, 1981). Based on comparison with early surveys, Borom (1979) concluded that SAV in the Mobile Bay and delta had not only declined but had changed in species density, diversity, and distribution. Baldwin (1957) reported that extensive SAV once grew along the northeastern shore of Mobile Bay, particularly beds of water celery (*Vallisneria americana*). Seagrass bed coverage along the eastern shore was much reduced by the late 1960s and almost completely gone in the 1970s (Borom, 1975). Stout and Lelong (1981) also noted that community diversity and species composition of SAV had declined, based on comparison with prior surveys, with single species beds more common during their survey than in the past.

Based on anecdotal evidence compiled from former residents and scientists, water celery and wigeon grass beds once were extensive along both the eastern and western shores of Mobile Bay but had disappeared from many of those areas by the time of Stout and Lelong’s 1980 study. Since then, wigeon grass has become reestablished in some of these areas, particularly along the southwestern shoreline of the bay. Stout and Lelong (1981) found that wigeon grass accounted for 11% of all submerged vegetation in their study area and covered approximately 121 ha (300 acres), mostly in pure stands. The spatial extent of shoal grass had also declined from past occurrences, particularly along the shores of Mobile Bay and in lower Perdido Bay (Stout and Lelong, 1981). Crance (1971) reported shoal grass in northern Mississippi Sound, but this seagrass had disappeared from this area by 1980, replaced by expansive wigeon grass beds (Stout and Lelong, 1981).

From 1940 to 1987, changes in the upper and middle parts of Perdido Bay consisted mainly of shifts in the locations of small meadows of shoal grass and wigeon grass, with only minor changes in density (Handley, 1995). In the lower bay, some shifting of locations and changes in density occurred, and coverage of seagrasses declined from 486 ha (1,201 acres) in 1940–41 to 251 ha (619 acres) in 1987 (Handley, 1995).

Results of area estimates from previous studies vary and include freshwater SAV as well as seagrass species (Vittor and Associates, 2004) (table 1). Differences may also be due to variations in geographic coverage of each study and different methods of estimation.

Table 1. Distribution of submerged aquatic vegetation in coastal Alabama from previous studies.

Source	Area	Hectares	Acres
Baldwin (1957)	Mobile Bay	2,024	5,000
	Delta	3,036	7,500
	Total	5,060	12,500
Stout and Lelong (1981)	Nondelta	1,118	2,763
Stout and others (1982)	Delta	1,496	3,696

Methodology Employed To Determine and Document Current Status

The most recent assessment of seagrass status in coastal Alabama was conducted in 2002 (Vittor and Associates, 2004) for the MBNEP. This mapping effort followed protocols established by the National Oceanic and Atmospheric Administration (NOAA) Coastal Change Analysis Program (C-CAP) (1995; *http://www.csc.noaa.gov/crs/lca/pdf/protocol.pdf*). Digital orthophotographs were created from true color aerial photography acquired July 19, 20, 22, and 31, 2002. Images were rectified by using a digital elevation model surface produced by aerotriangulation from an airborne geographic positioning system and inertial measurement unit data. Areas on the images with apparent SAV signatures were subsequently visited in the field with prints of the aerial photography used for locating the sites. The field effort included verifying SAV signatures on the aerial photographs and species identification at 295 locations for quality control and accuracy checks. Field location data were acquired at 31% of mapped polygons in freshwater portions of the study area and at 23% of mapped polygons overall. Aerial photographs were observed in the ArcView geographic information system (GIS), and SAV habitat polygons were digitally delineated on a computer screen display (fig. 3). Polygon coverage of SAV beds was created in ArcView version 3.2.

Table 2. Seagrass coverage in coastal Alabama by species, 2002 (Vittor and Associates, 2004).

Species	Hectares	Acres
Shoal grass (*Halodule wrightii*)	424	1,048
Turtle grass (*Thalassia testudinum*)	0.02	0.05
Wigeon grass (*Ruppia maritima*) (alone)	58	142
Wigeon grass (mixed with other freshwater species)	203	501
Total seagrass	684	1,691

Current Status and Changes

Vittor and Associates (2004) found shoal grass to be the dominant seagrass species in coastal Alabama during 2002. Overall, 684 ha (1,690 acres) of seagrass habitat were mapped (table 2). Greatest coverage by all species occurred in the delta and Mobile Bay, and the lowest acreage was in Perdido Bay (table 2).

Seagrasses were limited in distribution to the southern portion of the study area. Shoal grass made up most of the acreage, particularly in Mississippi Sound (332 ha, or 819 acres) and southern Perdido Bay (121 ha, or 300 acres), including Florida waters (table 3). In addition, relatively

Table 3. Geographic distribution of all submerged aquatic vegetation species in coastal Alabama, 2002 (Vittor and Associates, 2004).

Area	Hectares	Acres
Mobile Bay and Delta (freshwater submerged aquatic vegetation and wigeon grass (*Ruppia maritima*))	2,264	5,593
Mississippi Sound (seagrasses)	331	819
Perdido Bay (seagrasses)	95	234

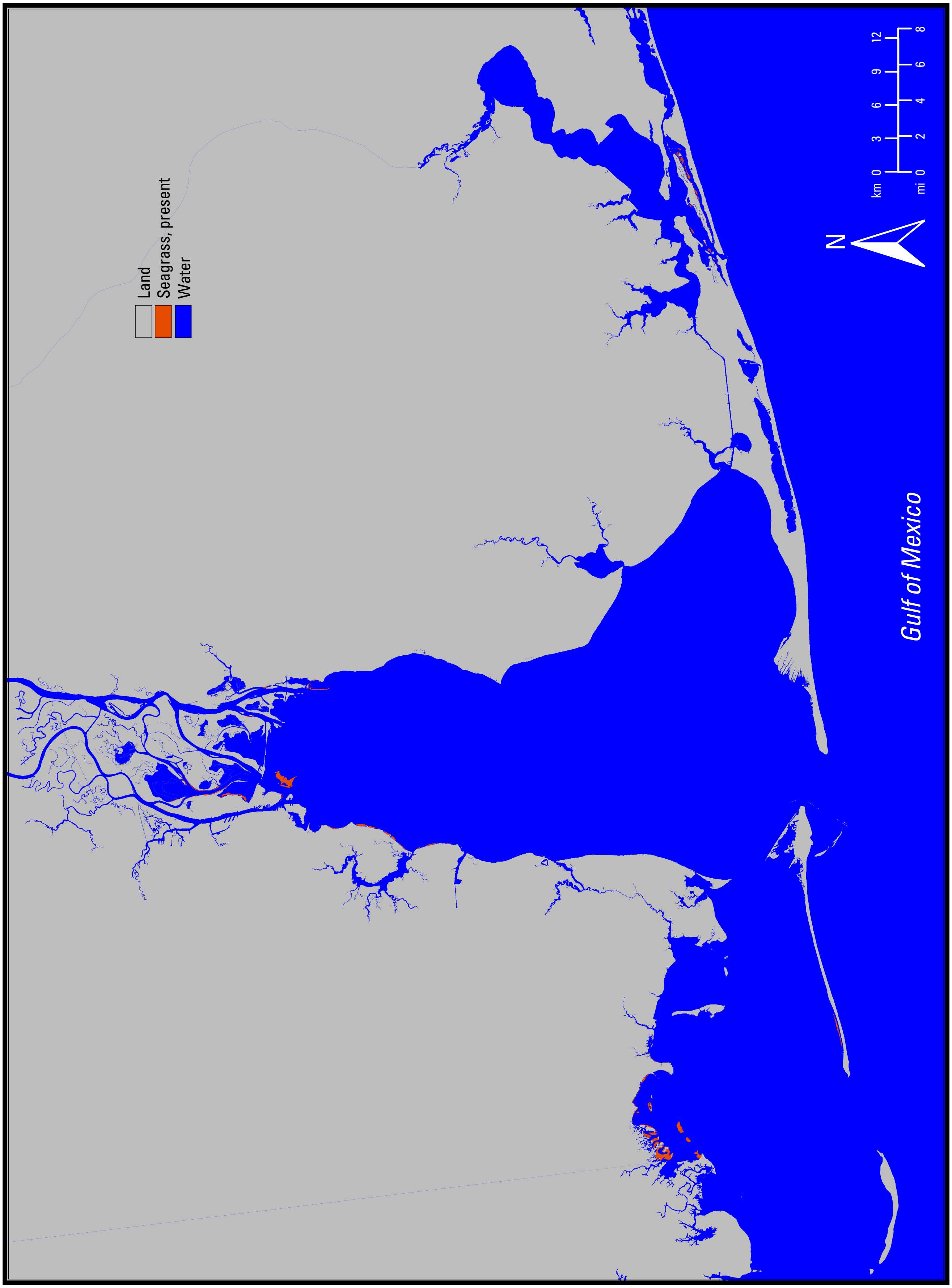

Figure 3. Distribution of seagrass along coastal Alabama, 2002.

small patches occurred along the northern shoreline of the western end of Dauphin Island and in Baldwin County in Little Lagoon and southwestern Perdido Bay tributaries. Shoal grass along the northern shoreline of Dauphin Island occurred in small, isolated patches that were included in a single polygon classified as scattered SAV. In Little Lagoon, several small areas of shoal grass were identified and mapped, as well as a bed of turtle grass. Turtle grass was previously reported in the Perdido Key area mixed with shoal grass (Stout and Lelong, 1981; Lelong, 1988). Lelong (1988) observed turtle grass intermixed with shoal grass from the Old River in Baldwin County but did not collect specimens. In subsequent conversations with Lelong (oral commun., 2003), he stated that, given the species rarity in the State, no collections were made. A sample of turtle grass collected from Little Lagoon during the 2002 survey represents the first vouchered specimen of the species from Alabama waters.

Much of the overall area of wigeon grass occurred in beds along the western shore of Mobile Bay (Vittor and Associates, 2004). Some beds were relatively large, particularly along the northwestern shores of the bay. Pure stands of wigeon grass primarily occurred south of Pinto Pass, where the Mobile River empties into Mobile Bay. Wigeon grass north of the Hwy 90/98 Causeway occurred in beds mixed with water celery, Eurasian watermilfoil (*Myriophyllum spicatum*), water stargrass (*Heteranthera dubia*), and other freshwater species.

The 2002 distribution of wigeon grass (Vittor and Associates, 2004) differed from the last major survey of the study area. Crance (1971) reported shoal grass in northern Mississippi Sound, but by 1980 it had been replaced by expansive areas of wigeon grass (Stout and Lelong, 1981). Wigeon grass also occurred in Little Lagoon and upper portions of Perdido Bay (Stout and Lelong, 1981) but did not occur in these areas in 2002. Instead, Mississippi Sound and Little Lagoon supported shoal grass beds at the time of the 2002 survey. Shoal grass distribution and area in the western portion of the study area had increased significantly since Stout and Lelong (1981) provided the last comprehensive survey of the project area. The study area had below-average precipitation for at least 2 yr prior to the 2002 mapping survey of Vittor and Associates. Lower-than-average precipitation may have influenced seagrass colonization in the study area relative to prior surveys, particularly in northern Mississippi Sound. If future average levels of precipitation approach or exceed normal amounts, patterns of wigeon grass occurrence relative to shoal grass distribution may revert to patterns documented in prior surveys (e.g., Stout and Lelong, 1981) of the study area.

Causes of Change

Change in seagrass area has occurred in coastal Alabama, but as of this date, there has been no investigation into the specific causes of change along coastal Alabama. Some general, probable causes for decreased seagrass coverage include tropical storm activity and turbidity in coastal waters. Tropical storms are frequent along the northern Gulf of Mexico coast. At least one tropical storm or hurricane will strike the Alabama coast every 2 yr. These storms vary dramatically in strength and resulting damage caused to the shorelines. In addition, land development is increasing in the Mobile Basin watershed and along the entire Alabama coast. Repeated, significantly large spills of raw and incompletely treated sewerage from municipal treatment facilities have probably affected seagrass beds in adjacent waters. Land development and agricultural activities within the drainage basins can cause increased amounts of sediments and nutrients to enter the waterways, thereby increasing turbidity of the water and reducing the amount of sunlight available to plants. Historical and continuing dredging to maintain navigation channels introduces periodic events of increased turbidity resulting in short-term reduction in light availability, which may affect seagrass health and/or distribution. In addition, changes in hydrology related to construction of a dredged material island and depth alterations may have indirect impacts on seagrass bed viability.

Seagrass Health

As of this date, no investigation has been made into the health of collective seagrass beds or individual species in coastal Alabama.

Species Information

Alabama’s brackish and marine seagrass beds are composed primarily of wigeon grass and shoal grass, with minor and sporadic occurrences of turtle grass. Wigeon grass is a euryhaline species that is not considered to be a true marine seagrass because of its intolerance for full seawater. Shoal grass and wigeon grass are pioneer species (den Hartog, 1970). In areas of early colonization, these species often occur in monotypic stands, unless or until later successional species, such as turtle grass, invade vegetated areas.

Although it does not tolerate full seawater, wigeon grass tolerates the greatest range of salinity compared with other SAV species, and because of its high rates of seed production and dispersal (Silberhorn and others, 1996), it is able to rapidly colonize suitable substrata. Wigeon grass can undergo prominent year-to-year changes in abundance and distribution that are due to salinity fluctuations (Verhoeven, 1975).

Shoal grass is a colonizer of disturbed areas where species such as turtle grass cannot grow. Of the marine seagrasses, shoal grass can withstand the widest range of temperatures and salinities, an ability which contributes to its colonizing ability. Shoal grass can tolerate salinities as low as 5 ppt (McMahon, 1968). As salinities increase in summer and fall or during drought years, shoal grass increases in abundance and may replace wigeon grass (Stutes, 2000; Heck and others, 2001). In general, shoal grass and wigeon grass are inferior competitors with other seagrasses and tend to occur in areas that are not suitable for other species (den Hartog, 1970). During years of drought, wigeon grass has been found in the delta north of Interstate 10 and the Hwy 90/98 Causeway and along the southeastern shore of the bay.

Monitoring, Restoration, and Enhancement Opportunities

There is currently no established monitoring program in Alabama for SAV. The 2002 survey (Vittor and Associates, 2004) was the first comprehensive coastal Alabama survey using standardized methods for image acquisition, rectification, mapping, and digitization. Digital comparisons to earlier comprehensive surveys (Stout and Lelong, 1981; Stout and others, 1982) have not been performed but are planned for 2004–05. As funding allows, future surveys will be planned to occur every 5 yr (David Yeager, MBNEP, oral commun.). Monitoring in coastal Alabama is coincidentally biased towards years with low precipitation. For example, if the most recent survey had been scheduled for 2003, the aerial photography would not have been flown because of the high precipitation and resulting high turbidity.

Mobile Bay has been a designated a U.S. Environmental Protection Agency National Estuary since 1995. The MBNEP Comprehensive Conservation and Management Plan (CCMP) contains a series of action items designed to monitor, protect, restore, and enhance SAV (Mobile Bay National Estuary Program, 2002): the SAV is one of five priority habitats identified and is addressed by action items to “protect or restore SAV habitats within the Mobile Bay estuary” with the goal to “maintain existing native SAVs at 2001 levels and increase acreage by 3% … by 2006.” Tasks completed to date include the 2002 mapping project, implementation of regulations restricting shrimp trawling in or near seagrass beds, and a pilot volunteer SAV gardening program to produce transplant materials, including wigeon grass, for restoration projects. Additional CCMP action items with indirect, anticipated positive effects on seagrasses include improving monitoring of living resources, reducing nutrient loading into coastal waters, restoring shorelines, management plans for nuisance species, comprehensive land-use planning, and reduction of shoreline erosion.

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Statewide Summary for Florida

By Paul R. Carlson, Jr., and Kevin Madley[1]

Background

The Gulf of Mexico coastline within the State of Florida (also known as the Florida gulf coast) extends approximately 1,000 km (over 600 mi) from the Alabama State line to the Dry Tortugas. Along this coast, Florida State waters and adjacent Federal waters include the two largest contiguous seagrass beds in the continental United States: the Florida Keys and the Florida Big Bend regions. The exact sizes of these two beds have not been determined because it is difficult to map deepwater seagrass beds dominated by paddle grass (*Halophila decipiens*); however, Sargent and others (1995) estimated that Florida State waters contained approximately 1,076,500 ha (2,660,000 acres) of seagrass, of which 55% (587,600 ha, or 1,451,900 acres) occurred in the Florida Keys and Florida Bay. An additional 334,600 ha (826,800 acres, 31% of statewide total seagrass area) occurred in the Big Bend region. The remaining seagrass area, 154,300 ha (381,200 acres), was distributed in estuaries and lagoons throughout the State. If seagrasses in adjacent Federal waters, including deepwater *Halophila* beds, are included, however, the total seagrass area in State and Federal waters is more than 1.2 million ha (3 million acres).

Florida's extensive estuarine and nearshore seagrass beds have developed as the result of the unique and stable geological history, climate, and circulation patterns along the Florida peninsula since the last ice age. The broad, shallow, and nearly flat continental shelf along the Florida gulf coast is larger (150,000 km^2, or 57,915 mi^2) than the land area of the Florida peninsula (139,700 km^2, or 54,000 mi^2). The shallow slope of the west Florida shelf, less than 20 cm/km (1 ft/mi), results in several million hectares of shallow bottom where sufficient light reaches the bottom for seagrass to survive. Nevertheless, Florida's seagrass resources are at risk, as human impacts over the past 100 yr have caused significant seagrass losses in all of the estuaries described in the following vignettes. At present, coastal development, nutrient loads caused by humans, and hydrological modifications threaten estuarine and nearshore seagrass beds along the entire Florida gulf coast (fig. 1).

[1] Florida Fish and Wildlife Conservation Commission.

Species Information

Although the climatic gradient results in significant variation in seagrass productivity, the same six seagrass species are found along the entire length of the Florida gulf coast (Phillips, 1960; Zieman and Zieman, 1989). Turtle grass (*Thalassia testudinum*), shoal grass (*Halodule wrightii*), and manatee grass (*Syringodium filiforme*) are the most abundant species in estuarine and nearshore waters of the Florida gulf coast (Phillips, 1960). Star grass (*Halophila engelmanii*) is locally abundant in turbid estuarine environments, and paddle grass, although diminutive, covers large areas of the west Florida shelf at depths from 9 to more than 30 m (30 to over 100 ft) (Continental Shelf Associates Inc., 1989). Because of its broad salinity and temperature tolerances, wigeon grass (*Ruppia maritima*) is also widely distributed in Florida estuaries. Water celery (*Vallisneria americana*) is a freshwater aquatic plant which occurs in the oligohaline reaches of many Florida tidal rivers, but it is not considered a seagrass because of its very low salinity tolerance.

Because seagrasses are very sensitive to water column transparency, their depth, distribution, and survival are primarily determined by water clarity. In areas with extremely clear water (the offshore areas of Big Bend and the Florida Keys, for example), seagrasses grow to depths greater than 20 m (65 ft) (Iverson and Bittaker, 1986). In turbid waters found in many Florida estuaries, however, seagrass beds are sometimes limited to depths less than 1 m (3.3 ft). If water column transparency at a particular site declines over time, seagrasses in deeper water die. As noted by Tomasko and Greening (Tampa Bay vignette), Tampa Bay seagrass cover declined 46% between 1950 and 1982, primarily as the result of declines in water column transparency resulting from human-caused nutrient loading, phytoplankton blooms, and resuspended sediment.

The climate along the Florida gulf coast ranges from temperate continental in the Florida panhandle to oceanic subtropical in the Florida Keys. Although Pensacola has a mean monthly minimum air temperature of 6°C (43°F) in January, the January mean sea-surface temperature is 12.5°C (approximately 58°F). At the very southern tip of the State, mean monthly minimum air temperature at Key West in January is 16°C (61°F), and the February mean sea-surface temperature is 23°C (73°F). Mean maximum air temperatures

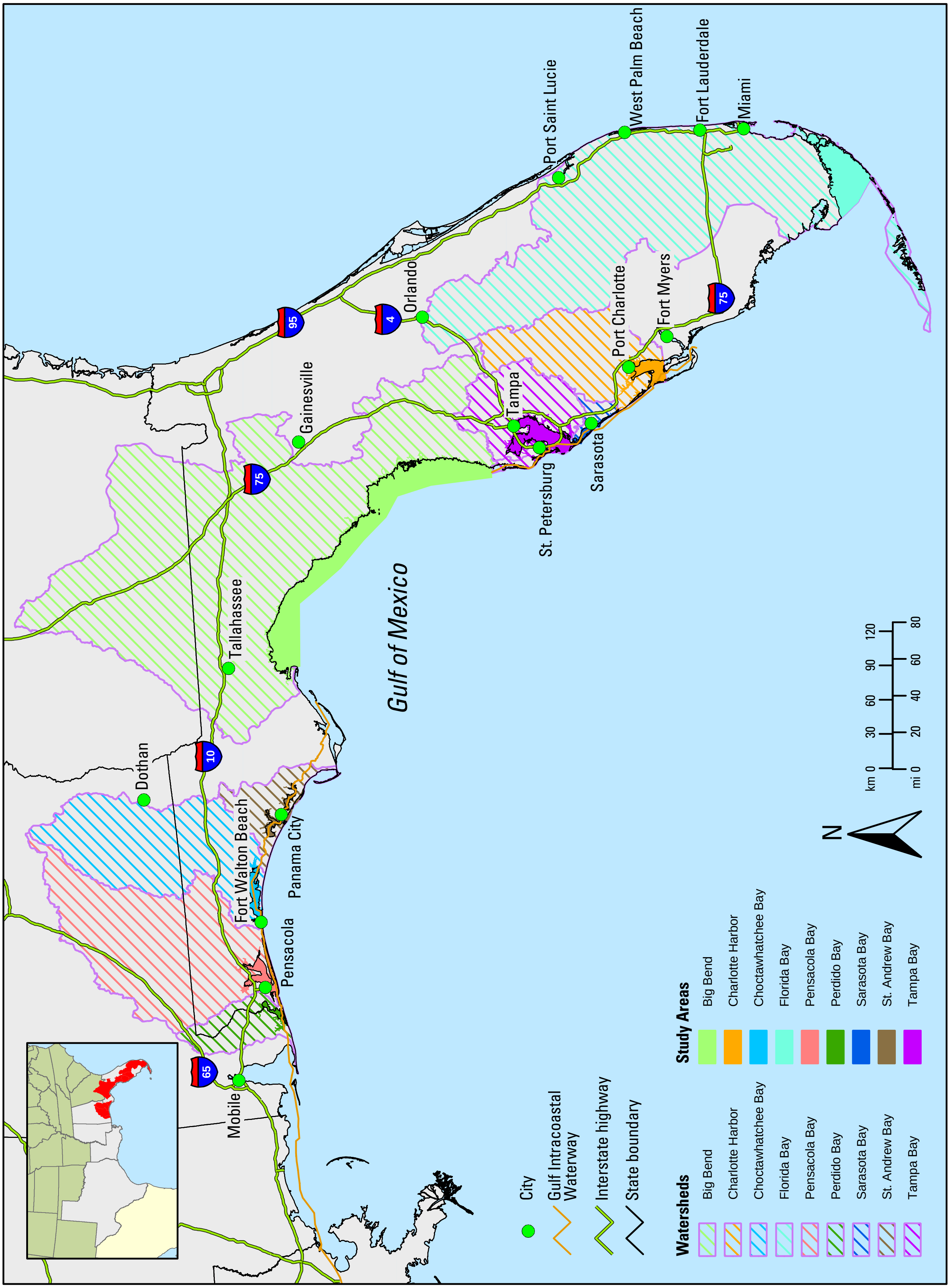

Figure 1. Watershed for the Gulf of Mexico coast of Florida.

for Pensacola (32°C, or 90°F) and Key West (33°C, or 91°F) are very similar, and monthly mean sea-surface temperatures for both cities are 30°C (86°F). The timing and volume of rainfall also vary along the Florida gulf coast. The Florida panhandle and north Florida receive considerably more winter rainfall than the southwest Florida and Florida Keys regions. The entire Florida gulf coast also receives a significant amount of rainfall during the wet season (June through October). Annual rainfall ranges from 160 cm (63 inches) at Pensacola, to 120 cm (47 inches) at Cedar Keys, to 130 cm (51 inches) at Saint Petersburg and Naples, and to 100 cm (39 inches) at Key West. Rainfall, in general, and El Niño events, in particular, have a very strong effect on seagrasses along the Florida gulf coast. For example, during the strong El Niño of 1997–98, there was heavy rain throughout Georgia and Florida. As a result, monthly mean discharge of the Suwannee River at Wilcox, Fla., in March 1998 was 1,160 m^3/s (40,960 ft^3/s), the highest monthly mean discharge for the month of March measured during the 60-yr period of record for the gage. Thick phytoplankton blooms occurred throughout winter, spring, and early summer 1998 in west Florida estuaries and the adjacent shelf, causing declines in seagrass density and health at several sites along the west coast of Florida (Carlson and others, 2003).

Scope of Area

Previous estimates of seagrass cover for the entire State of Florida (Sargent and others, 1995) divided the State into five regions: Panhandle, Big Bend, Gulf Peninsula, South Florida, and Atlantic Peninsula. For this document, we have divided the 24 Florida counties along the Gulf of Mexico among the Panhandle, Big Bend, Gulf Peninsula, and South Florida regions (fig. 2). To provide an areal estimate for the south Florida coast, four subregions have been extracted from the region described by Sargent and others (1995): (1) Ten Thousand Islands, which includes Collier County and the mainland portion Monroe County; (2) Florida Bay, including parts of Dade and Monroe Counties; (3) the gulfside Florida Keys, which extend from Long Key to the Tortugas; and (4) the southwest Florida shelf, which is a roughly rectangular region bounded by Cape Romano on the north side, the Dry Tortugas on the west, and the Florida Keys on the south (fig. 3). Vignettes have been prepared for Perdido Bay, Pensacola Bay, Choctawhatchee Bay, St. Andrew Bay, the Florida Big Bend region, Tampa Bay and St. Joseph Sound, Sarasota Bay, Charlotte Harbor, and Florida Bay. Areas outside the vignette regions—St. Joe Bay, Ten Thousand Islands, and gulfside Florida Keys—are described only in this statewide summary. The estuarine or nearshore systems described in the vignettes range from moderately healthy and stable (Big Bend) to severely degraded but improving (Tampa Bay). In most systems, the principal human threats are nutrient loading and degraded water quality, which decrease water column transparency and light available for seagrasses. In Florida Bay, however, hydrologic modification of The Everglades is probably the principal human impact on seagrass distribution and community structure.

Methods Used To Determine Current Status of Seagrasses in the State of Florida

Seagrass cover estimates for the State of Florida have been based on photointerpretation of aerial photography. Some seagrass aerial photography was flown at 1:48,000 scale, but most of the photography used for trend analysis has been flown at 1:24,000 scale. As part of a study to determine the impact of propeller scarring in seagrass beds, Sargent and others (1995) made the first effort to summarize seagrass cover data for the entire State of Florida using photography flown between 1982 and 1990 (table 1). To update areal estimates, Madley and others (2003) constructed new statewide seagrass maps using data produced from aerial photography acquired from 1987 through 1999 (table 1). Note that seagrass area estimates are reported to the nearest acre or hectare in accompanying tables; however, we have rounded estimates in the following text to the nearest 100 acres or hectares.

Status and Trends

The most recent estimate of seagrass cover which has been mapped in Florida State waters (Madley and others, 2003) is approximately 910,980 ha (2,251,000 acres; table 2). This estimate is limited to State jurisdictional boundaries on both the Atlantic and Gulf of Mexico coasts, but considerable amounts of seagrass lie outside State waters in the gulf and are excluded from this estimate. As noted earlier, inclusion of deepwater paddle grass beds in the Big Bend region and the southwest Florida shelf could raise the total area of seagrass in Florida gulf coast estuaries and on the west Florida shelf to over 3 million ha (7.4 million acres).

Of the statewide total seagrass area estimate, approximately 74% (678,681 ha, or 1.7 million acres) occurs in the Gulf of Mexico between the Dry Tortugas and the Florida/Alabama State line and in adjacent estuaries. Along the Gulf of Mexico coast, the Big Bend region has the largest total seagrass area of 207,206 ha (612,000 acres), followed by the Florida keys, including the Marquesas and Dry Tortugas, with 220,156 ha (544,000 acres). Florida Bay contains 147,715 ha (365,000 acres) of seagrass, and the gulf peninsula region, stretching from Cape Sable to Anclote Key, contains 43,343 ha (107,100 acres). The Panhandle region contains 17,483 ha (43,200 acres) of seagrass (table 2).

Recent estimates of Florida statewide seagrass cover range from 910,575 to 918,669 ha (2.25 million to 2.27

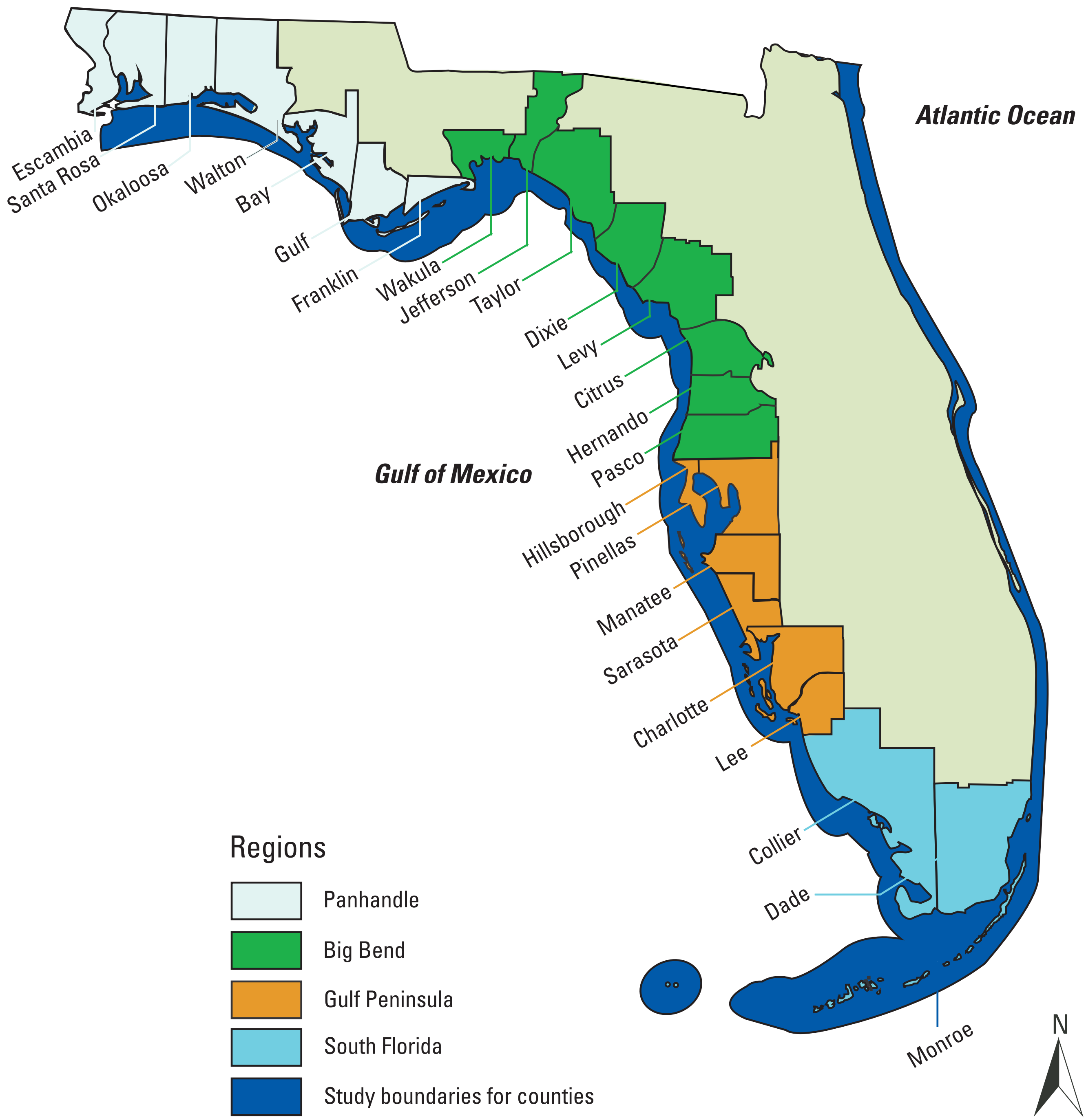

Figure 2. Florida counties along the Gulf of Mexico coast.

Table 1. Seagrass aerial photography datasets used for Florida statewide seagrass cover estimates by Sargent and others (1995) and Madley and others (2003).

[EPA = U.S. Environmental Protection Agency, USFWS = U.S. Fish and Wildlife Service, USGS = U.S. Geological Survey, MMS = Minerals Management Service, SWFWMD = Southwest Florida Water Management District, FDOT = Florida Department of Transportation, FDEP = Florida Department of Environmental Protection, SFWMD = South Florida Water Management District, FWRI = Fish and Wildlife Research Institute, NOAA = National Oceanic and Atmospheric Administration]

	1995 Data			2002 Data		
County	**Year**	**Scale**	**Source(s)**	**Year**	**Scale**	**Source(s)**
			Panhandle			
Escambia	1982–85	1:24,000	EPA/USFWS	1992	1:24,000	USGS
Santa Rosa	1982–85	1:24,000	EPA/USFWS	1992	1:24,000	USGS
Okaloosa	1982–85	1:24,000	EPS/USFWS	1992	1:24,000	USGS
Walton	1982–85	1:24,000	EPA/USFWS	1992	1:24,000	USGS
Bay	1982–85	1:24,000	EPA/USFWS	1992	1:24,000	USGS
Gulf	1982–85	1:24,000	EPA/USFWS	1992	1:24,000	USGS
Franklin	1982–85	1:24,000	EPA/USFWS	1992	1:24,000	USGS
			Big Bend			
Wakulla	1982–85	1:24,000	EPA/USFWS	1992	1:24,000	USGS
Jefferson	1983	1:40,000	MMS	1992	1:24,000	USGS
Taylor	1983	1:40,000	MMS	1992	1:24,000	USGS
Dixie	1983	1:40,000	MMS	1992	1:24,000	USGS
Levy	1983	1:40,000	MMS	1992	1:24,000	USGS
Citrus	1983	1:40,000	MMS	1992	1:24,000	USGS
Hernando	1983	1:40,000	MMS	1992	1:24,000	USGS
Pasco	1983	1:40,000	MMS	1992	1:24,000	USGS
			Gulf Peninsula			
Pinellas	1992	1:24,000	SWFWMD	1999	1:24,000	SWFWMD
Hillsborough	1992	1:24,000	SWFWMD	1999	1:24,000	SWFWMD
Manatee	1992	1:24,000	SWFWMD	1999	1:24,000	SWFWMD
Sarasota	1992	1:24,000	SWFWMD	1999	1:24,000	SWFWMD
Charlotte, north	1992	1:24,000	SWFWMD	1999	1:24,000	SWFWMD
Charlotte, south	1982, 1987	1:24,000	FDOT, FDEP	1999	1:24,000	SFWMD
Lee	1982, 1987	1:24,000	FDOT, FDEP	1999	1:24,000	SFWMD
			South Florida			
Collier	1982, 1987	1:40,000	MMS, FWRI	1987	1:24,000	SFWMD
Monroe	1982, 1987	1:40,000	MMS, FWRI	1987	1:24,000	SFWMD
Dade- Florida Bay	1982–86	1:40,000	MMS	1992/1994	1:24,000	NOAA, FWRI
Monroe-Fla. Keys	1982, 1987	1:40,000	MMS, FWRI			

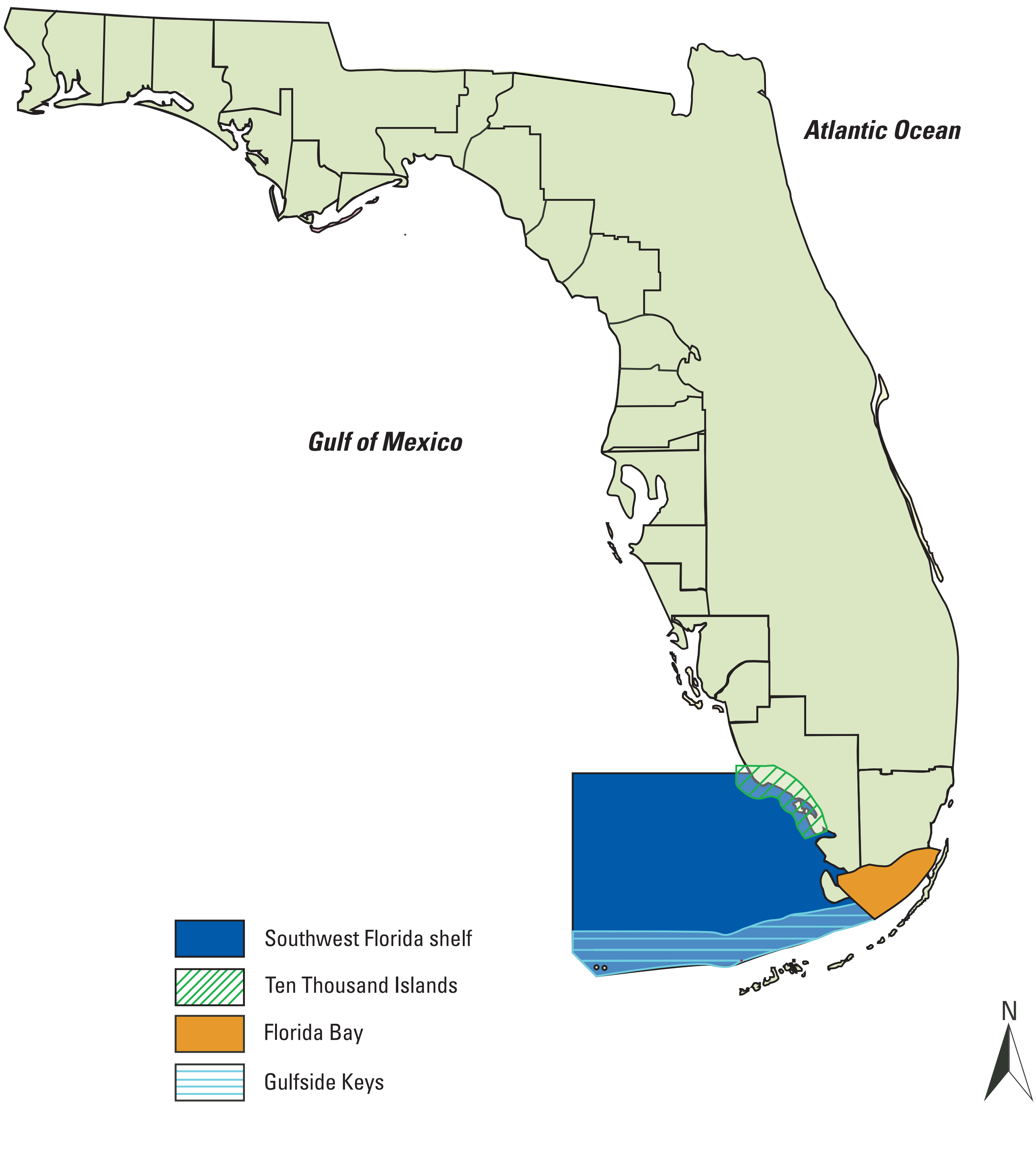

Figure 3. Four subregions of the south Florida coast.

Table 2. Recent and current estimates of seagrass cover in Florida State waters.

[FWRI = Fish and Wildlife Research Institute]

FWRI Seagrass	Continuous Seagrass		Patchy Seagrass		Total Seagrass	
Data File	Hectares	Acres	Hectares	Acres	Hectares	Acres
Statewide estimates						
Madley and others, 2003	402,563	994,733	510,476	1,261,387	913,039	2,256,120
Sargent, 2000	399,916	988,192	520,046	1,285,034	919,962	2,273,226
Sargent and others, 1995	N/A	N/A	N/A	N/A	1,075,783	2,658,260
Regional estimates (Madley and others, 2003)						
Panhandle	7,191	17,768	10,283	25,409	17,474	43,178
Big Bend	28,508	70,443	219,090	541,372	247,598	611,815
Gulf Peninsula	29,974	74,066	13,349	32,985	43,323	107,051
Atlantic Peninsula	27,413	67,737	2,357	5,823	29,769	73,560
South Florida*	309,478	764,719	265,398	655,798	574,875	1,420,517
* Includes significant seagrass area on Atlantic side of Florida Peninsula in Dade County. Complete seagrass surveys of Broward County have not been performed, so that area is represented with a zero value in the calculations for these projects.						
Florida gulf coast (within State jurisdictional boundaries) (Madley and others, 2003)						
1. Panhandle	7,191	17,768	10,283	25,409	17,474	43,178
2. Big Bend	28,508	70,443	219,090	541,372	247,598	611,815
3. Gulf Peninsula	29,974	74,066	13,349	32,985	43,323	107,051
4. Ten Thousand Islands**	0	0	3,085	7,623	3,085	7,623
5. Florida Bay	109,550	270,698	38,063	94,054	147,613	364,752
6. Gulfside Keys	90,730	224,193	129,658	320,384	220,388	544,579
Gulf coast total	265,953	657,168	413,528	1,021,827	679,481	1,678,998
Florida gulf coast (area of benthic habitat less than 18 m (60 ft) deep, including Federal waters and deepwater *Halophila* beds This is a measure of potential seagrass habitat based on depth, not actual mapped seagrass) (Madley and others, 2003)						
7. Big Bend					1,415,028	3,496,534
8. Southwest Florida shelf					1,433,127	3,541,254
Gulf coast total (equals 1, 3, 4, 5, 6, 7, 8 because 2 is within 7)					3,279,524	8,103,704

** Ten Thousand Islands seagrass was originally reported as presence/absence values. For this analysis we categorized all seagrass of the Ten Thousand Islands as patchy.

million acres). Although the 2003 estimate of statewide seagrass cover is 15% lower than the 1995 estimate of Sargent and others (about 1 million ha, or 2.6 million acres), the difference results in part from differences in the spatial extent of aerial photography and mapping coverage. The spatial extent of the geographic information system (GIS) data used by Sargent and others (1995) for the Big Bend region, in particular, was greater than that of the 1992 GIS data used in more recent estimates. Differences in habitat classification schemes and accuracy of alternate methods also contributed to the differences in seagrass area estimates. For example, the Minerals Management Service (MMS) 1982–87 photography for south Florida was flown at 1:40,000 scale and mapped at a coarser level of accuracy than the 1992–94 National Oceanic and Atmospheric Administration (NOAA) photography. Therefore, attempts to interpret trends in seagrass gains or loss on a statewide scale must acknowledge the variation among datasets being compared.

Several of the vignettes in this document focus on smaller areas of Florida where consistent mapping scales and classification schemes make trend analysis more reliable. In 7 of the 11 estuaries described in the following vignettes, significant seagrass loss has occurred. No measurable change in seagrass cover occurred in northern Charlotte Harbor, and seagrass cover in Tampa Bay increased 24% between 1982 and 1996, after declining 46% between 1950 and 1982. Seagrass cover in Sarasota Bay increased by 19% between 1988 and 1996 after declining by 30% between the 1950s and 1988. Therefore, significant seagrass loss has occurred in 10 of the 11 systems. In the two estuaries where seagrass cover has increased since the 1980s, recent gains have not offset historical losses.

Causes of Change

Several human activities have caused direct or indirect losses of seagrasses in Florida, and many of the Florida vignettes describe similar scenarios of seagrass loss. Significant amounts of seagrass were lost in many Florida estuaries as the result of dredging operations in the 1950s and 1960s. Propeller scarring impacts by commercial and recreational boats have also increased dramatically in the past decade (Sargent and others, 1995) as the total number of boats registered in Florida has risen to over 900,000. Seagrass losses caused by thermal effluent from powerplants have been documented in Biscayne Bay and St. Joseph Sound. Herbicides, metals, and hydrocarbons might also have caused seagrass loss in some Florida estuaries, acting alone or along with other stressors, but their distribution and impact on seagrasses in Florida have received little attention. Paper mills, known to discharge dioxins and organic compounds with high biological oxygen demand, nutrients, and color, have also caused significant loss of seagrass in several west Florida estuaries. For example, Livingston and others (1998) estimated that paper mill effluent has caused the loss of seagrasses over a large area at the mouth of the Fenholloway River. The greatest single cause of seagrass loss to date in Florida, in the Gulf of Mexico, and throughout the world, however, has been water-quality degradation. Along the west coast of Florida, the principal cause of water-quality degradation has been eutrophication resulting from domestic, agricultural, and industrial wastes. Several vignettes address the role of nutrients in seagrass losses as well as nutrient load reductions and corresponding recovery of seagrasses. Despite concerted efforts to improve water quality and restore lost seagrasses in Florida, losses might occur in the future as the result of groundwater contamination, freshwater removal from estuarine tributaries, and a shift in the dominance of human-caused nutrient impacts from point sources to less manageable nonpoint sources.

Dredge and fill operations for navigation and development have been strictly controlled in Florida since the early 1970s, but prior dredging caused significant direct and indirect losses of seagrasses. Dredging for navigation purposes along the Florida gulf coast began in the late 1890s, when the Federal Government authorized construction of channels from Tampa Bay to Sarasota and from Sarasota to Venice (Alperin, 1983). By 1936, the Gulf Intracoastal Waterway (GIWW) was complete between the Apalachicola River and New Orleans. The GIWW segment which extends 245 km (152 mi) from the Caloosahatchee River to the Anclote River was dredged between 1960 and 1967. Congress also authorized, but never funded, a dredged channel in Florida between St. Marks and the Anclote River to connect the two GIWW segments. Navigation channels were dredged in Tampa Bay in the early 1900s, and Lewis and Estevez (1988) determined that 34 km^2 (13 mi^2) of bay bottom in Tampa Bay had been dredged or filled for residential and navigational purposes. Navigation channels dredged through shallow seagrass beds caused the immediate loss of the seagrass removed from the channel, burial of adjacent seagrass by deposition of dredged material, as well as continuing losses due to turbidity created by resuspended fine sediments, and turbulent scour caused by the wakes of large ships. Several dredging projects created finger-fill residential developments in Tampa Bay and Sarasota Bay. In 1924, D.P. Davis dredged Hillsborough Bay, a subestuary of Tampa Bay, to create residential developments which still bear his name. Dredging for residential development continued into the 1950s and 1960s, and Taylor and Saloman (1968) estimated that residential dredging in Boca Ciega Bay alone filled in approximately 1,400 ha (3,460 acres) of mangrove and seagrass habitat.

Propeller scar impacts are an increasing problem for shallow seagrass beds in Florida. Propeller scars typically require several years to recover (Zieman, 1976; Dawes and others, 1997), and many scars become scoured by currents or wave action and fail to recover (Precht and Gelber, 2003). Prior to 1995, when the Florida State Constitution was amended to prohibit gill net fishing in Florida State waters, many propeller scars were created by commercial fishing boats circling schools of fish. Since 1995, however, the number of commercial fishing boats has declined dramatically, and the number of recreational boats has increased to almost 1 million, making recreational boaters responsible for most propeller scars. Sargent and others (1995) made the first comprehensive inventory of propeller scarring of seagrass beds in Florida. They determined that approximately 70,000 ha (173,000 acres) of Florida seagrass beds had some degree of propeller scarring. Moderate and severe propeller scarring was most prevalent in the Florida Keys, Charlotte Harbor, and Tampa Bay. A demonstration project at Fort Desoto County Park in Pinellas County found that additional signage (above and beyond navigation markers), speed restriction zones, and motor exclusion zones reduced the number and severity of propeller scars (Stowers and others, 2002).

Although dredging and propeller scarring have had serious impacts on seagrasses, degraded water quality has been responsible for most of the declines in the distribution and abundance of Florida seagrasses and submerged aquatic vegetation, echoing trends throughout the Gulf of Mexico (Lewis and others, 1982; Pulich and White, 1991), the United States (Orth and Moore, 1983), and the world (Cambridge and others, 1986) over the past 50 yr. Seagrasses are particularly vulnerable to declines in water column transparency resulting from human activities because all seagrasses require light for photosynthesis and because they possess nonphotosynthetic belowground tissues (roots and rhizomes) which require photosynthetically produced oxygen to survive. Reduced water column transparency, in turn, results from a number of natural and human factors. Turbidity can result from suspended sediment particles or phytoplankton cells, and any particles in the water column, living or dead, reduce light penetration (Gallegos, 1994; McPherson and Miller, 1994; Gallegos and Kenworthy, 1996). Suspended sediment can result from wind and wave action, boat wakes, dredging, and river runoff. In some gulf estuaries, dissolved organic matter also reduces light penetration through the water column (McPherson and Miller, 1994; Livingston and others, 1998). The principal cause of light attenuation in nearshore and estuarine waters of the Florida gulf coast, however, is phytoplankton blooms driven by nutrient runoff from urban, residential, agricultural, and industrial sources.

Seagrasses in many estuaries have also demonstrated the capability to recolonize and spread in areas where water quality has improved, and Tampa Bay provides a model for seagrass recovery in response to water-quality management (see Tampa Bay vignette). Lewis and others (1999) estimated that seagrass area in Tampa Bay declined from over 30,000 ha (74,000 acres) in the late 1800s, to 16,500 ha (41,000 acres) in 1950, and to 8,700 ha (21,500 acres) in 1982. Lowest seagrass cover coincided with high nutrient and chlorophyll concentrations in Tampa Bay. With implementation of advanced wastewater treatment at the City of Tampa sewage treatment plant and the subsequent development of a public and private nutrient management consortium, seagrass cover increased steadily in Tampa Bay and Hillsborough Bay between 1982 and 1996. For the entire Tampa Bay, Tomasko and Greening (Tampa Bay vignette) reported a 24% increase in seagrass cover during this period. Seagrass cover in Sarasota Bay also increased by 19% during the same period as the result of nutrient load reductions (Sarasota Bay vignette).

Although increases in seagrass cover have occurred in response to improved water quality, current trends in the volume, form, and spatial delivery pattern of Tampa Bay nutrient loads raise questions about the recovery potential of seagrasses in Tampa Bay and other Florida estuaries. In the mid-1970s, when Tampa Bay water quality was poorest, point sources accounted for 68% of the total nitrogen load of 8,437 Mg/yr (9,300 ton/yr) entering the bay (Zarbock and others, 1994). By the year 2010, even though total nitrogen loads are forecast to drop to 4,264 Mg/yr (4,700 tons/yr), nonpoint source nitrogen is projected to make up 84% of the total nitrogen load. Increasing dominance of nonpoint source nutrient inputs is cause for concern because the diffuse spatial pattern of nonpoint loads makes them difficult and expensive to control. Furthermore, the rate of seagrass recovery in Tampa Bay has slowed recently, suggesting that more stringent conditions might be required for seagrass reestablishment than for maintenance of existing beds.

The 1997–98 El Niño episode demonstrated the vulnerability of Florida gulf coast seagrasses to nonpoint nutrient loading. As noted earlier, Tampa received more than 50 cm (20 inches) of “extra” rainfall between December 1997 and April 1998 (Ross and others, 1998). Baywide seagrass cover declined by 7.7% from 10,900 ha (26,900 acres) in 1996 to 10,060 ha (24,900 acres) in 1999. One subbasin, Old Tampa Bay, lost 24% of its seagrass cover between 1996 and 1999. Sarasota Bay seagrass cover declined by approximately 10% during the same period, demonstrating the regional scale impact of El Niño rainfall and runoff. No data are available for seagrass losses in the Big Bend region during El Niño episodes, but SeaWiFS satellite imagery from the NOAA for spring 1998 showed extensive phytoplankton blooms along the Florida gulf coast, and seagrass cover and health declined at monitoring sites in the Big Bend and gulf peninsula regions (Carlson and others, 2003). Poe and others (2003) estimated that average-annual nonpoint source nitrogen loads for Tampa Bay were 3,151 tons N/yr during the 1995–98 El Niño period compared with 1,723 tons N/yr in 1992–94, driven by runoff values of 2,083 million cubic meters in 1995–98 and 1,161 million cubic meters in 1992–94. Increases in nitrogen loading driven by El Niño rainfall were most pronounced in Old Tampa Bay and Hillsborough Bay, the two Tampa Bay segments with the greatest rates of seagrass loss. High rainfall in 2003 has also been implicated as a factor slowing seagrass recovery in the Feather Sound portion of Old Tampa Bay (Greening and others, 2004).

These data suggest that estuarine and nearshore seagrass cover for the Florida gulf coast might be strongly affected by rainfall and associated nonpoint source nutrient loads. Controlling nonpoint nutrients is generally more expensive and difficult than controlling point sources, and different strategies will be required for different regions of the Florida gulf coast. Urbanized estuaries, such as Tampa Bay and Sarasota Bay, have limited space and capacity to intercept and treat storm water. By comparison, effective control of nonpoint source nutrients in the Big Bend region requires management of land use in the Suwannee River watershed, making up almost 25,900 km^2 (10,000 mi^2) in Florida and Georgia.

Because human-caused nutrient loads to estuaries and the nearshore Gulf of Mexico increase as coastal populations grow, population density, urban development patterns, and land-use changes are the most important factors affecting seagrass distribution and survival along the west coast of Florida. In 2001, the human population of Florida was estimated to be 16.4 million, of which more than 12 million lived in coastal counties (Florida Office of Economic and Demographic Research, 2004). Eight coastal counties of southwest Florida (Monroe, Collier, Lee, Charlotte, Sarasota, Manatee, Pinellas, and Hillsborough) contained 3.4 million people, or 21% of the State total. Seven coastal counties of the Panhandle (Escambia, Santa Rosa, Okaloosa, Walton, Bay, Gulf, and Franklin) contained 835,000 people, and three counties along the “Springs Coast” (Citrus, Hernando, and Pasco) contained 601,000 people. In contrast, five counties of the Big Bend region (Wakulla, Jefferson, Taylor, Dixie, and Levy) contained only 108,000 people in 2001. Not surprisingly, coastal counties with the largest human populations, such as Pinellas, Hillsborough, Manatee, and Sarasota Counties, have experienced the greatest amounts of seagrass loss.

As populations continue to grow in Florida coastal counties, especially those in the Gulf Peninsula, Panhandle, and Big Bend regions, the potential for seagrass loss will also increase. Pinellas County, the most densely settled county in Florida, has limited space for additional development, so the population is forecast to increase by only 6% between 2000 and 2010. Predicted rates of population growth for Collier, Lee, Charlotte, Pasco, Hernando, Citrus, and Levy Counties, however, range from 17% to 20% during the same period. As population growth, urbanization, and land-use changes occur in coastal counties, additional seagrass losses will occur unless concerted efforts are made to control point and nonpoint source nutrient loads. Other contaminants, such as herbicides, metals, and hydrocarbons, that enter nearshore waters from point sources and nonpoint source runoff might also affect seagrass growth and survival.

Mapping and Monitoring Problems and Information Needs

Data Problems

Data problems associated with seagrass mapping are not unique to the State of Florida. Typical problems include lack of standardization in habitat classification systems, scale of photography and mapping, absence of frequent and synoptic coverage, use of different projections for benthic habitat mapping and GIS data, and seasonal differences in seagrass cover (see discussion of phenology below). Uniform classification systems for seagrass mapping are currently being adopted by the Florida Fish and Wildlife Conservation Commission (FWC) and other State, Federal, and local agencies; however, many historical seagrass data have used differing classification systems. The NOAA Coastal Services Center has provided specifications and guidance for seagrass aerial photography and mapping, and those standards are often used as guidelines for projects nationwide (Finkbeiner and others, 2001).

Rescue and preservation of seagrass aerial photography are also important data problems and needs for the State of Florida. Although water management districts have carried out seagrass aerial photography along much of the Florida gulf coast since the late 1980s, portions of the coastline are still not flown on a regular basis, availability of earlier photography is spotty, and original photographs are being lost because no systematic program exists to locate, rescue, archive, and catalog historical aerial photography. For example, the earliest seagrass aerial photography reported in table 1 was flown in 1982 by NOAA and by MMS. Although the FWC’s Fish and Wildlife Research Institute (FWRI) has archived some MMS photography, efforts to locate the original negatives have been unsuccessful, emphasizing the need for data rescue.

Other significant data problems affecting seagrass resource management are errors or biases associated with seagrass photointerpretation and mapping which sometimes overestimate, and sometimes underestimate, seagrass cover. For example, with current manual photointerpretation techniques, seagrass beds in aerial photographs are classified as patchy or continuous, and minimum mapping unit (MMU) sizes for 1:24,000 aerial photography are generally set at

0.1 ha (0.25 acres) or larger, ignoring smaller seagrass beds. Seagrass cover can also be overestimated because patchy beds include substrate with seagrass cover as low as 10%. From a resource management standpoint, the most serious concern is that potentially significant declines in seagrass cover could occur undetected within polygons classified as patchy seagrass, compromising our ability to recognize and correct water-quality problems in a timely manner.

Digital, multispectral imagery and supervised software classification techniques provide opportunities to improve estimates of seagrass area and density by decreasing MMU size and increasing the number of seagrass cover classes. Costs for digital imagery acquisition and turnaround time for digital imagery processing and interpretation are also expected to decrease. These factors will enhance our ability to visualize seagrass changes and take management action in a timely manner. Despite improvements in seagrass mapping techniques, however, monitoring with field work is, and will continue to be, a necessary adjunct to aerial photography for seagrass assessment and management.

Seagrass phenology—seasonal changes in seagrass growth and senescence—is an important yet often overlooked data problem for seagrass aerial photography. Seagrasses along the Florida gulf coast grow during spring and summer, generally reaching peak biomass in early fall. When cold fronts begin to cool water temperatures in October and November, seagrass cover begins to decline. Between December and March, cold fronts and lunar tidal forces generate very low tides during daytime hours, which frequently causes shallow seagrasses to desiccate and defoliate, rendering them invisible to aerial photography at 1:24,000 scale. The decline and loss of seagrass cover occur earlier in fall and persist longer in spring in north Florida than in south Florida and the Florida Keys, creating potential problems within datasets and in comparing data from different times. Along the west coast of Florida, seagrass aerial photographs taken in December and January might show significantly less shallow seagrass cover than would photographs taken in November, especially for shoal grass (*Halodule wrightii*). Because, over the past 14 yr, Tampa Bay aerial photography documenting seagrasses has been flown once in October, twice in December, three times in January, once in February, and once in April, areal estimates of shallow seagrass might vary considerably.

Phenology is often overlooked or considered a secondary criterion because of overriding concerns about cloud cover and water clarity. Ideal conditions for seagrass aerial photography occur infrequently because they include sun angle, cloud cover, wind, and water clarity criteria. During summer in Florida, cloud cover is generally too great. For much of the Florida peninsula, runoff and associated phytoplankton and turbidity create unacceptable water clarity for seagrass aerial photography into late fall and early winter. As a result, many seagrass aerial photography projects are currently flown in late fall and winter to take advantage of low cloud cover, low tides, and better water clarity.

Spring months (March and April in south Florida and April and May in north Florida), however, have more cloud-free days than do December, January, and February. In typical years, water clarity is excellent in spring, and seagrasses have begun to grow again, creating a spring window of opportunity for aerial photography. The St. Johns River Water Management District (SJWMD), one of the first agencies in the State of Florida to develop a comprehensive seagrass mapping and monitoring program, flies seagrass aerial photography in spring (Virnstein and Morris, 1996), although the Southwest Florida Water Management District (SWFWMD) flies aerial photography between October and January for much of the west-central Florida coast. To obtain the best possible seagrass photography, the fall “window” should be shortened to avoid winter defoliation, and a spring “window” should be considered. Because photointerpretation costs are much greater than aerial photography costs alone, resource management agencies might consider acquiring photography in spring and fall and then using the best set or portions of each set to create the most complete, accurate assessment of seagrass cover. At a minimum, an effort should be made to determine the amount of seagrass cover “missing” in winter photography.

Because several factors (seagrass phenology, sun angle, cloud cover, haze, wind, and water clarity) affect the quality and interpretability of seagrass imagery, several sets of Florida seagrass imagery have been partly or totally unusable. The cost and effort of mobilizing aircraft and staff for aerial photography projects sometimes result in photography under less than optimal conditions. Large areas often require several days for complete photography, and changing conditions can affect imagery quality. To avoid costly mistakes acquiring imagery of poor quality, there are two relatively new resources available for seagrass aerial imagery planning: MODIS satellite imagery and real-time cloud cover data. MODIS imagery is downloaded daily by the University of South Florida Institute for Marine Remote Sensing, and the center maintains a Web site with a library of current and recent MODIS imagery for the Florida peninsula (*http://modis.marine.usf.edu*). Turbidity from resuspended sediments, as well as color resulting from dissolved organic matter and phytoplankton blooms, is readily apparent in MODIS imagery. Large seagrass beds are also visible. The daily frequency of the imagery allows rapid and frequent assessment of water clarity, cloud cover, and approaching weather systems. Real-time cloud cover maps are available from several National Weather Service Web sites, enabling users to hourly select cloud-free areas for aerial photography.

Another important information gap is the area of the west Florida shelf covered by deepwater paddle grass and star grass. Paddle grass meadows are not easily mapped by aerial photography because they frequently occur in water depths greater than 10 m (33 ft) and because the plants are small. Continental Shelf Associates, Inc. (1989) found that paddle grass covered 1.2 million ha (3 million acres), or 31% of their study area, on the southwest Florida continental shelf.

Paddle grass occurred to depths of 37 m (122 ft) with greatest biomass at depths from 21 to 27 m (70 to 90 ft). Continental Shelf Associates, Inc. (1985) also reported extensive beds of paddle grass in the Florida Big Bend region. In table 2, we have made a conservative estimate of potential paddle grass cover on the west Florida shelf, and the inclusion of paddle grass beds raises our estimate of seagrass area to about 3 million ha (7.4 million acres). The accurate assessment of area of paddle grass beds on the west Florida shelf is an important data gap to fill because of the potential trophic importance of these communities and their vulnerability to nutrification of the nearshore Gulf of Mexico. Because aerial photography is not useful for mapping paddle grass beds, other mapping techniques (acoustics and video transects, for example) should be used.

Monitoring Needs

Monitoring of seagrasses in the Big Bend region is one of the most urgent needs for the State of Florida because increasing nutrient loads to nearshore waters of the Florida gulf coast jeopardize extensive seagrass resources (Mattson and others, Florida Big Bend vignette, this report). In some springs of the Suwannee River drainage basin, groundwater nitrate concentrations have risen thirtyfold in the past 40 yr (Katz and others, 1997). SeaWiFS chlorophyll imagery in spring 1998 showed extensive and persistent phytoplankton blooms in the northeastern Gulf of Mexico resulting from El Niño runoff and increasing nutrient loads in west Florida rivers. Given the large size of offshore seagrass beds and the shallow slope of the west Florida shelf, a 50-cm (20-inch) decrease in seagrass compensation depth resulting from small increases in turbidity and/or phytoplankton biomass would eliminate thousands of acres of seagrass. In the absence of good mapping and monitoring for offshore seagrasses, we would be unaware of the losses. With funding from the U.S. Environmental Protection Agency Gulf of Mexico Program, FWC and Suwannee River Water Management District are developing a seagrass monitoring program for the Big Bend region, and FWC is also working with NOAA on *Halophila* beds of the southwest Florida shelf. Additional effort and funding will be required, however, to design and implement a seagrass monitoring program for the west Florida shelf.

Seagrass assessment programs incorporating aerial photography and mapping as well as fixed-site or probabilistic monitoring are evolving throughout Florida, and FWRI is developing an integrated and statewide seagrass mapping and monitoring program. The primary goals of this program are seagrass mapping and monitoring on a schedule that balances cost and the need for timely information for management decisions. Mapping and monitoring are complementary tools because they operate on different spatial and temporal scales. For example, in an integrated program such as that developed by the SJWMD (Virnstein and Morris, 1996), seagrass cover in the Indian River Lagoon is mapped every 2 or 3 yr from aerial photography. Groundtruthed data for aerial photographs and more frequent and detailed information on seagrass species composition, canopy height, and density are collected every summer from 80 fixed transects along the lagoon. Costs for aerial photography and photointerpretation of the Indian River Lagoon are approximately $428/acre, and costs for annual monitoring of 80 seagrass transects are approximately $60,000/yr (Virnstein, personal commun.). Similar integrated mapping and monitoring programs have been developed for Tampa Bay, Sarasota Bay, and Charlotte Harbor. Better integration of mapping and monitoring data is needed, and the merits of probabilistic sampling versus fixed transects should be evaluated. In the short term, however, we hope to collect data from all seagrass mapping and monitoring programs in Florida, producing annual monitoring reports and mapping reports every 6 yr.

For nearshore seagrass beds in the Big Bend shelf, southwest Florida shelf, and the gulfside Florida Keys, the sheer size of the areas involved and water depths will require mapping and monitoring programs of a different scale and frequency. Much of the initial offshore survey work was done to provide background information for potential oil and gas exploration and extraction. Lacking that economic incentive, we need to find other sources of funding to map, monitor, and manage these resources.

Assessment, Protection, Recovery, and Restoration Opportunities

Several agencies participate in seagrass monitoring and management along the Florida gulf coast, including the FWC, Florida Department of Environmental Protection (FDEP), Northwest Florida Water Management District, Suwannee River Water Management District, SWFWMD, and South Florida Water Management District. Tampa Bay and Charlotte Harbor have active estuary programs, and there are two national estuarine research reserves along the Florida gulf coast: Apalachicola Bay and Rookery Bay. Several counties also have environmental monitoring programs. These agencies focus on seagrass protection by coordinating mapping and monitoring activities, by providing information on seagrass management and restoration to managers, and by making efforts to protect and improve water quality.

The FDEP also has jurisdiction over submerged lands in State waters and administers several aquatic preserves along the Florida gulf coast. There are three national wildlife refuges along the Florida gulf coast (Lower Suwannee, Chassahawitzka, and St. Marks) and one national park (Everglades). The combined jurisdiction of State aquatic preserves, national wildlife refuges, and Everglades National Park provides statutory protection for much of the nearshore seagrass beds in the Big Bend and Ten Thousand Islands regions.

Protection of existing seagrasses and the maintenance or restoration of water quality are the most cost-effective seagrass management tools available because seagrass restoration plantings are extremely expensive, and the success of restoration plantings varies considerably. Fonseca and others (1998) reported that published costs for seagrass plantings ranged from $25,000/ha to $50,000/ha. They suggested that total costs were likely closer to $500,000/ha (approximately $200,000/acre), excluding costs for donor material collection or purchase. If that cost estimate is combined with the estimate by Lewis and others (1999) of seagrass loss in Tampa Bay between 1950 and 1982 (over 8,000 ha, or 20,000 acres), the cost of restoring the lost seagrass would be over $4 billion. Tomasko (2003) reported that, between 1982 and 1996, less than 1% of the seagrass recovery that occurred in Tampa Bay resulted from seagrass restoration plantings. The remaining 99% of seagrass recovery during that time period (approximately 2,130 ha, or 5,262 acres) resulted from natural expansion and recolonization of seagrasses within the bay in response to improved water clarity. The improved water clarity, in turn, resulted from a public-private partnership to manage nitrogen loading to Tampa Bay (Lewis and others, 1999).

Given the potential for natural recovery in response to water quality management, the best role for restoration plantings might be for mitigating unavoidable losses and for restoration of direct impacts such as vessel groundings and propeller scars. Even in that context, however, additional information is needed on development of reliable sources and methods to cultivate seagrass transplant material for restoration, mechanical and hand planting techniques, and techniques to stimulate regrowth of seagrass in propeller scars. For example, current restoration and mitigation programs rely on donor material from intact seagrass beds or on salvage material from beds which are being destroyed. Donor bed impacts are unavoidable in the former case. Alternatives to using donor seagrass material from healthy seagrass beds include micropropagation and seagrass nurseries. At present, complete micropropagation techniques have been developed and field tested for wigeon grass. Some success has been achieved with laboratory culture of shoal grass but not with turtle grass or manatee grass. Seagrass Recovery, Inc. in Ruskin, Fla., has successfully propagated shoal grass in brackish pond culture, a process that considerably reduces donor bed impacts.

In addition to problems with obtaining seagrass transplants, restoration plantings are also very labor intensive. Mechanical seagrass transplanting techniques are currently being tested and might provide a cost-effective method for restoration plantings. Propeller scar restoration techniques which are currently being tested by FWC staff and industry partners are sediment tubes and sediment amendments which might stimulate seagrass growth into propeller scars. Propeller scar recovery is particularly important because of the increasing number of recreational boats and the long recovery time (over 7 yr) of propeller scars measured in Tampa Bay (Dawes and others, 1997).

Florida seagrass scientists and resource managers have invested considerable effort in collecting and sharing information on seagrass biology, ecology, and management. The Tampa Bay Estuary Program has sponsored two conferences related to seagrass management in the past 3 yr: “Seagrass Management: It’s Not Just Nutrients” was held in St. Petersburg, Fla., in August 2000, followed by “Submerged Aquatic Habitat Restoration in Estuaries: Issues, Options, and Priorities,” held in Sarasota, Fla., in March 2003. Print copies of proceedings for both conferences are available from the Tampa Bay Estuary Program, and electronic copies of the presentations are posted on the Tampa Bay Digital Library Web site created by the U.S. Geological Survey (USGS) (*http://gulfsci.usgs.gov/library/*). The Tampa Bay Estuary Program and the U.S. Environmental Protection Agency have also collaborated to update Zieman and Zieman’s (1989) report *The Ecology of the Seagrass Meadows of the West Coast of Florida: A Community Profile*. The updated report by Dawes, Phillips, and Morrison (2004) titled *Seagrass Communities of the Gulf Coast of Florida: Status and Ecology* is available as a digital document from the Tampa Bay Estuary Program Web site (*http://www.tbeptech.org/TechPubs/t0304.pdf*).

The FWC and other resource management agencies also provide online access to seagrass GIS data. Statewide GIS data are available from the FWRI Marine Resources GIS Internet Map Server (*http://ocean.floridamarine.org/mrgis/viewer.htm*). The SWFWMD also offers downloadable GIS coverages for their biennial seagrass surveys in the Tampa Bay, Sarasota Bay, Lemon Bay, and Charlotte Harbor regions (*http://www.swfwmd.state.fl.us/data/gis/libraries/swim.htm*). Maps of Tampa Bay seagrass cover based on SWFWMD aerial photography and GIS data are available from the Tampa Bay Estuary Program (*http://www.tbeptech.org/html/sg_maps.html*). Metadata for FWRI and SWFWMD GIS data that is compliant with guidelines by the Federal Geographic Data Committee can be downloaded from their respective Web sites.

In a partnership, the FWRI, USGS, Tampa Bay Estuary Program, and the SWFWMD have created a Web site that provides digital imagery, digital nautical charts, seagrass data, and other relevant GIS data for the Tampa Bay area. This Web site (*http://ocean.floridamarine.org/tbep*) contains rectified and georeferenced time series aerial photography that allows users to view and overlay aerial photographs from 1928 through 2002. A set of easy-to-use but powerful tools allows any user to assess temporal changes in seagrass cover and construction throughout the Tampa Bay region. It is hoped that this Web site will serve as a model, providing intuitive access to complex data for a variety of users, and FWRI is making a concerted effort to rescue, preserve, digitize, and distribute historical seagrass aerial photography, surveys, and monitoring data from around the State.

Three relevant reports and databases have been developed by FWC's Florida Marine Research Institute. The Florida Seagrass Conservation Information System (*www.floridamarine.org/seagrass*) provides descriptive and contact information for a number of seagrass-related projects in Florida, including education, protection, restoration, mapping/monitoring, and research. Narrative project summaries are available. Users can query the online database to identify projects according to several key fields of interest, including project type, location, and principal investigator. Search results are arranged in print-friendly reports, and new projects can be submitted for inclusion in the database by completing online forms. Because this is a dynamic database with continuous update ability, it is hoped that acceptance and use of this system will promote up-to-date information exchange among seagrass scientists and managers.

The Florida Seagrass Manager's Toolkit is also available on the FWC Web site (*http://www.floridamarine.org/features/view_article.asp?id=23202*). The toolkit provides practical information for resource managers and other professionals directly involved in seagrass management, as well as for interested citizens. The ecological importance of Florida's seagrass habitats and the need for effective management are summarized. A basic problem-solving model that can be used to develop appropriate management responses is presented, and the importance of spatial and temporal scale in seagrass management and monitoring is addressed. Tools available for mapping and monitoring, protection and restoration, and replanting and repair of damage are described, and emerging issues of potential future interest to seagrass managers are discussed.

Finally, FWC has developed a framework for a statewide seagrass management program. Titled *Conserving Florida's Seagrass Resources: Developing a Coordinated Statewide Management Program*, the report is also available at *http://www.floridamarine.org/features/view_article.asp?id=23185*. This report is intended to serve as a nontechnical planning document that provides a conceptual framework for development of a coordinated statewide seagrass management initiative while also recognizing supporting and building upon the accomplishments of local, community-based programs. Effective local seagrass management programs are currently underway in several areas of Florida. In addition, a number of Federal and State agencies provide programs for the protection, mapping, and monitoring of seagrass habitats within their jurisdictions. This report recommends a series of steps that could be taken to initiate a coordinated, cooperative, multiagency program. A primary purpose of the statewide program should be to provide increased support for, as well as greater statewide consistency in the implementation of, the various components of seagrass management.

One key component of this plan is adoption of measurable goals for seagrass protection and recovery and a program to monitor seagrass status and trends throughout the State. With impetus from the U.S. Commission on Ocean Policy Ocean Initiative, the Florida Ocean Initiative has begun development of a statewide seagrass mapping and monitoring program. The Florida Department of Environmental Protection and Fish and Wildlife Conservation Commission are partnering in this effort. It is hoped that, by building on current seagrass mapping efforts by water management districts within the State of Florida and by adopting standardized monitoring protocols, onground assessments will be performed annually and complete statewide seagrass mapping will be performed every 6 yr.

Acknowledgments

We thank Jesse Lewis and Jim Krolick for GIS analysis of seagrass data. Bill Sargent and David Crewz provided information on FWC Web publications and restoration technique development, respectively. Laura Yarbro and Penny Hall provided valuable editorial comments on early drafts.

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Perdido Bay

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Background

Perdido River and Perdido Bay form the western boundary of Escambia County, Fla., as well as the Florida/Alabama State line. Coastal Alabama and northwest Florida are currently experiencing rapid growth and development similar to historical boom years in south Florida. It is likely that unless there are conscious efforts to apply growth management lessons learned from mistakes made in southern Florida, similar costly mistakes may be repeated. Current expansions associated with the Perdido River and Bay watersheds (fig. 1) provide an opportunity to apply "smart growth" principles to restore and protect their water bodies.

Although Perdido Bay covers approximately 130 km^2 (50 mi^2), its watershed encompasses over 3,238 km^2 (1,250 mi^2) of land, tributaries, lagoons, and bayous in Florida and Alabama. Perdido Bay is connected to the Gulf of Mexico through Perdido Pass, and the Gulf Intracoastal Waterway (GIWW) passes through the southern portion of the bay. Perdido Bay includes areas along the coastal and inland sections of Escambia County in Florida, and Baldwin and Escambia Counties in Alabama. Because of its small size, Perdido Bay features rapid changes in water quality in response to rainfall, wind, and tide (Perdido Ecosystem Restoration Group, 1998).

Because Perdido Bay includes oligohaline, mesohaline, and euryhaline systems, this estuary possesses high species diversity. Seagrasses in Perdido Bay provide spawning, nursery, and adult habitat for many commercially and recreationally important catches such as shrimp (*Penaeus* sp.), crabs (*Callinectes* sp.), scallops (*Argopecten* sp.), speckled trout (*Cynoscion* sp.), redfish (*Sciaenops* sp.), and mullet (*Mugil* sp.). Increased shoreline and watershed development, stormwater runoff, septic tanks, wastewater treatment plant effluent, industrial discharges, agriculture, silviculture, and natural occurrences (e.g., hurricanes) have all contributed to the degradation of water quality and the resulting loss of seagrasses in Perdido Bay. Hurricane Frederick, for example, which made landfall at Mobile Bay in 1979, did considerable damage to Perdido Bay seagrasses (Perdido Ecosystem Restoration Group, 1998). Seagrass monitoring efforts in Perdido Bay have shown that from 1940 to 1992, approximately 90% of the historical continuous seagrass coverage has been lost (table 1). Wolf Bay, the largest tributary to southwestern Perdido Bay, has undergone decreasing water quality because of nitrate and phosphate runoff (Livingston, 2001). Eutrophication of Wolf Bay has been attributed to an increase in agricultural land use, which is just one example of a nutrient source within the Perdido Bay watershed (Livingston, 2001).

Two-thirds of Escambia County, Fla., residents live within 16 km (10 mi) of a major waterway. Like most Florida surface waters, Perdido Bay is threatened by human activities. Most residential stormwater sources are located in the southern portions of the watershed, particularly the highly developed Ono Island and Perdido Key areas. Recent statistics reveal that in the years 1997–99, nearly 5,000 new single-family housing units and over 1,400 new multifamily housing units were permitted in Escambia County. Census results indicate that the population of Escambia County increased nearly 13% between 1990 and 2000 (University of Florida, 2000).

Upper Perdido Bay is impacted by runoff from agricultural and silvicultural lands. In 1998, it was determined that 75% of the basin in Baldwin County, Alabama; 70% of the basin in Escambia County, Ala.; and 85% of the basin in Escambia County, Fla., were used for timber production (Perdido Ecosystem Restoration Group, 1998).

Table 1. Perdido Bay seagrass values in hectares (acres) from 1940 to 2002.

Seagrasses	1940	1979	1987	1992	2002
Continuous	365 (880)	38 (94)	124 (307)	40 (99)	
Patchy	114 (282)	171 (423)	103 (255)	74 (183)	
Present					112 (277)

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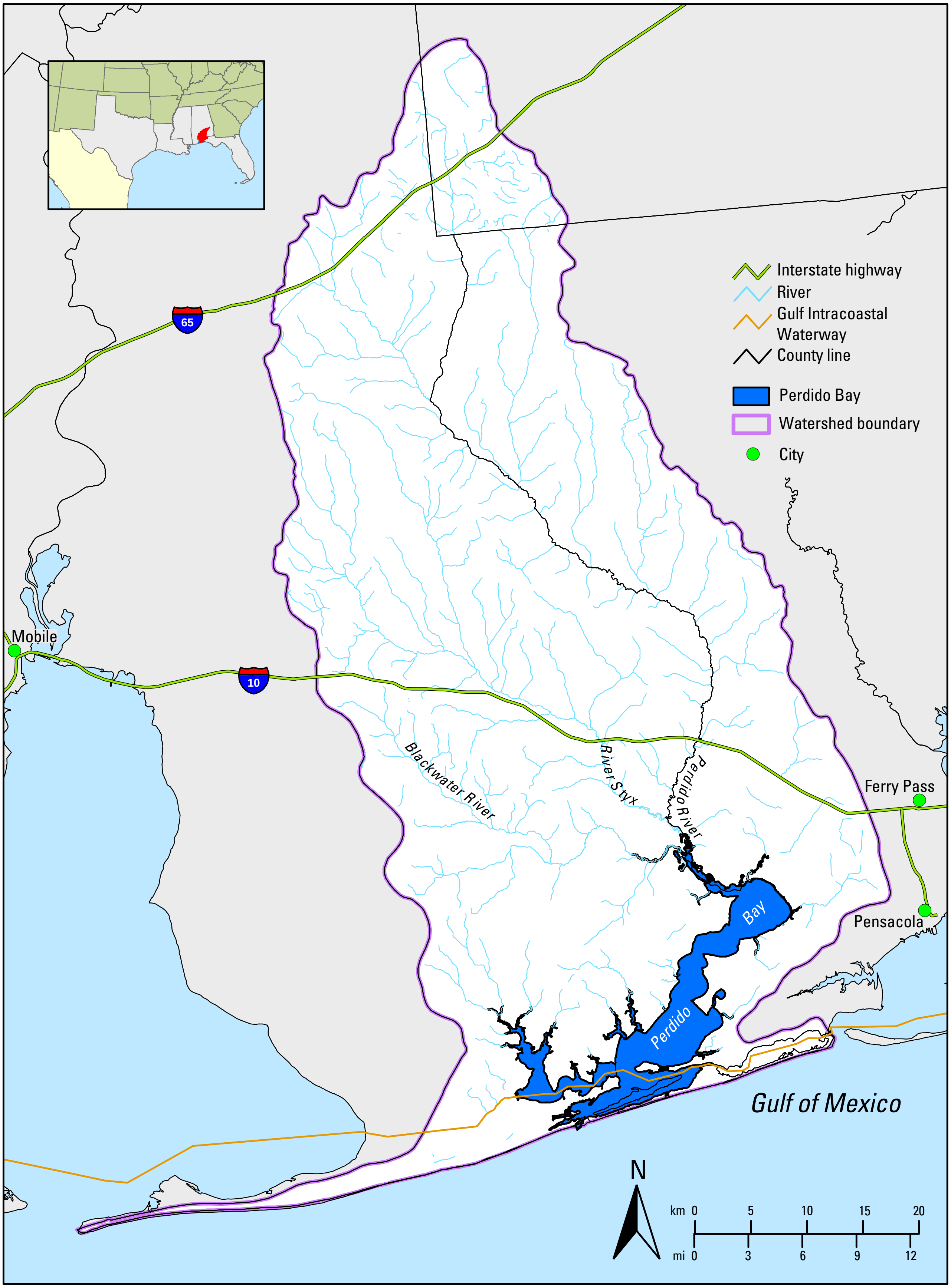

Figure 1. Watershed for Perdido Bay.

The most significant point source discharge into Perdido Bay is a paper mill. The mill discharges approximately 0.92 m^3/s (21 million gal/d) of kraft process effluent into Elevenmile Creek (Perdido Ecosystem Restoration Group, 1998). In addition, effluent from other industrial and domestic wastewater treatment plants increase nutrient levels in the bay. Much of Perdido Bay has been included on the 303(d) list of impaired waters because of low dissolved oxygen and high nutrient levels. As a result of citizen concerns about water quality in Perdido Bay, in July 1998 the Perdido Ecosystem Restoration Group, Florida Department of Environmental Protection (FDEP), National Oceanic and Atmospheric Administration (NOAA), and Florida Department of Community Affairs (DCA) published “Perdido Ecosystem Management Strategies.” This publication outlines management policies, action plans, and best management practices for the Perdido Bay watershed.

Scope of Area

Perdido Bay is oriented approximately perpendicular to the Gulf of Mexico, with a length of 53 km (33 mi) and an average width of 4 km (2 mi). Average water depth in the bay is approximately 2 m (7 ft) (Livingston, 2001). Perdido Bay can be divided into three segments: Upper, Middle, and Lower (fig. 2).

The Perdido River, designated by the State of Florida an “Outstanding Florida Water,” is a characteristic blackwater stream and the major freshwater tributary of Perdido Bay. The river is also the State boundary between Alabama and Florida. Much of the adjacent lands are used for silviculture. The majority of land along the river is owned by only two corporations. In contrast, the River Styx, which converges with the Perdido River from the west, has experienced substantial residential development.

Upper Perdido Bay

Upper Perdido Bay begins at the mouth of the Perdido River and extends south past the mouths of Elevenmile Creek and Bayou Marcus to Cummings Point. Although a substantial portion of the watershed remains forested, residential development is increasing.

Middle Perdido Bay

Extending southwest from Cummings Point to Innerarity Point, Middle Perdido Bay is the largest segment of the bay. Located on the eastern side of the bay, Tarkiln Bayou is one of the last remaining pristine bayous in northwest Florida. Tarkiln Bayou State Preserve and the Perdido Pitcher Plant Prairie (prioritized for preservation by Escambia County, Fla., and the State of Florida) surround Tarkiln Bayou.

Lower Perdido Bay

Lower Perdido Bay extends from Innerarity Point southward to the Gulf of Mexico and includes Wolf Bay, Cotton Bayou, Old River, and the GIWW eastward to the Theo Baars Bridge. The Lower Perdido Bay watershed includes Innerarity Point and Perdido Key in Florida, and Ono Island and Gulf Shores in Alabama. Although highly developed, with extensive residential and resort areas, this portion of the watershed also includes three State parks, as well as Gulf Islands National Seashore.

Methodology Employed To Document Current Status

The most current mapping study of seagrass coverage for the Perdido Bay area was conducted more than 10 yr ago by the U.S. Geological Survey (USGS) National Wetlands Research Center (NWRC) by using natural-color aerial photography taken in 1992 at a 1:24,000 scale as part of the northeastern Gulf of Mexico seagrass mapping project.

The mapping protocol consisted of stereoscopic photointerpretation, cartographic transfer, and digitization in acccordance with strict mapping standards and conventions. Other important aspects of the protocol included the development of a classification system, groundtruthing, quality control, and peer review.

The information derived from the photography was subsequently transferred using a zoom transfer scope onto a stable medium overlaying USGS quadrangle basemaps. In those cases where the data were inadequate or incomplete, contemporary supplemental data were acquired from other sources and used to complete the photographic coverage.

The seagrass classification system (see appendix for full discussion) used consists of two classes of open water—RIV (riverine, fresh water) and EST (estuarine or marine open water)—and five classes of seagrass habitats. One class is continuous seagrass, CSG, for which no density distinction was made, and the other four classes of patchy seagrass are based on percent ground cover of patches and in increments of 5%—PSG1 (0%–10% very sparse), PSG2 (15%–40% sparse), PSG3 (45%–70% moderate), and PSG4 (75%–95% dense).

The groundtruthing phase included the participation of field staff from Gulf Islands National Seashore, U.S. Fish and Wildlife Service (USFWS), Dauphin Island Sea Lab, Mississippi State University, Alabama Department of Conservation and Natural Resources, and Florida Department of Environmental Protection. Draft maps were sent to the aforementioned agencies and staff for review and comments. All comments received were incorporated into the final maps prepared and delivered.

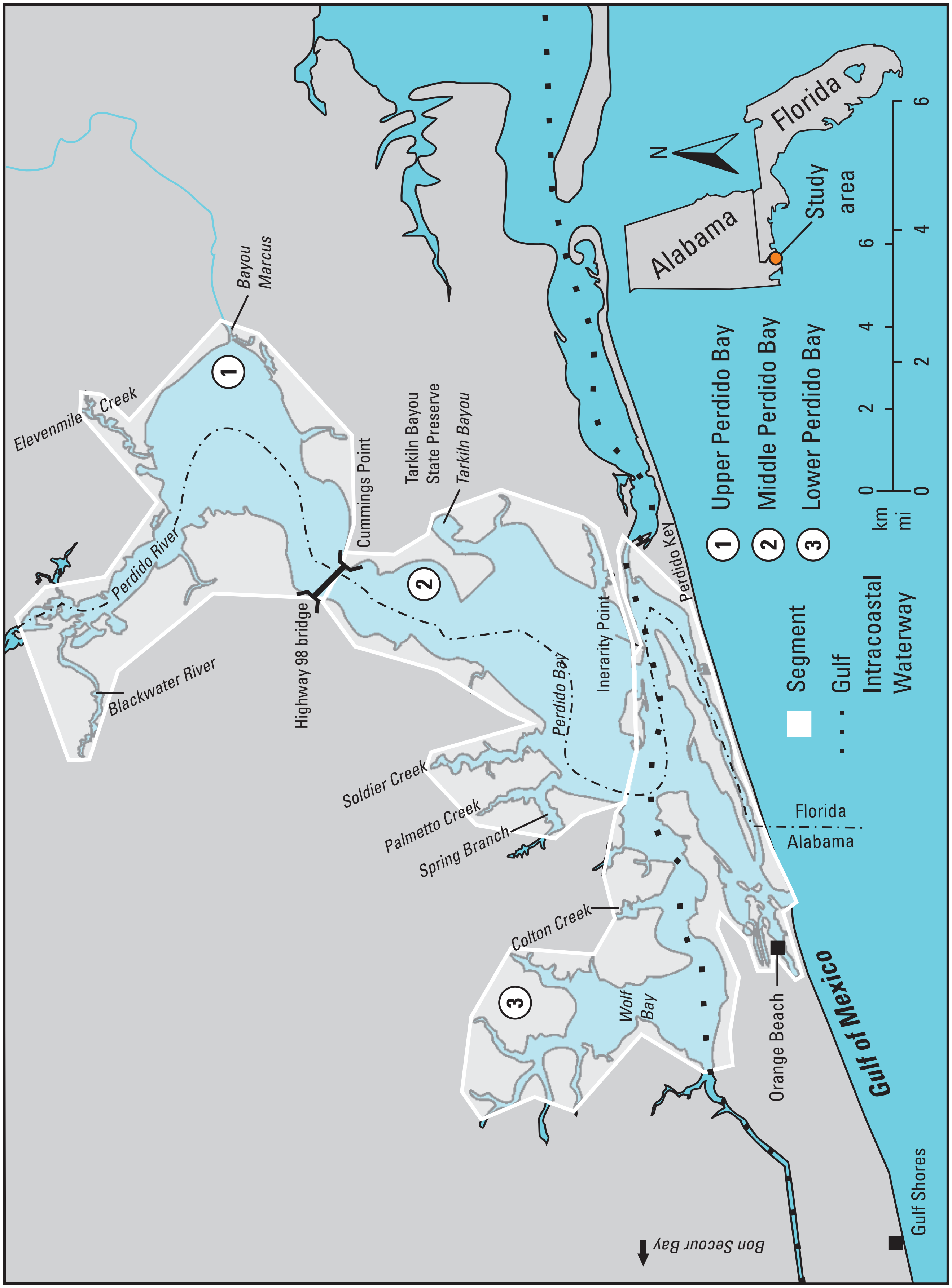

Figure 2. Scope of area for the Perdido Bay seagrass vignette.

Methodology Employed To Analyze Historical Trends

To produce a trend analysis for Perdido Bay, in 1988 and 1989, the NWRC produced a series of historical seagrass maps for the USFWS's Panama City Ecological Services Office and the U.S. Environmental Protection Agency's (EPA) Region IV Near Coastal Waters Program. Black and white photography from 1940–41 (1:20,000 scale), color infrared photography from 1979 (1:65,000 scale), and color infrared photography from 1987 (1:65,000 scale) was used to develop the trend analysis. Each date of aerial photography was analyzed with stereoscopic visual equipment, and seagrasses were delineated onto Mylar® overlays by using a zoom transfer scope with USGS 1:24,000 scale topographic quadrangles as the base maps. The overlays were digitized by using the Wetland Analytical Mapping System (WAMS) and converted into Arc format files. All dates of photography were of good to high quality for delineating seagrasses.

For consistency, classification of the seagrass for the trend analysis followed the same protocol as was used in the development of the Northeastern Gulf of Mexico seagrass mapping using the 1992 aerial photography. The seagrass classification system used consists of the same two classes of open water and five classes of seagrass habitats used to document current seagrass status.

Draft maps were sent out to the USFWS for review and comments. All comments received were incorporated into the final maps prepared and delivered. Although groundtruthing of the 1987 seagrass delineations was performed by NWRC, no groundtruthing took place for the 1940–41 or 1979 historical seagrass delineations.

The NWRC currently holds the aerial photography, the interpreted overlays, and the Mylar® overlays for the 1940–41, 1979, and 1987 maps.

Status and Trends

Seagrass monitoring efforts in 1940 (fig. 3), 1979 (fig. 4), 1987 (fig. 5), 1992 (fig. 6), and 2002 (fig. 7) have documented the decline of seagrass in Perdido Bay (table 1). In the 52-yr period from 1940 to 1992, Perdido Bay lost 355 ha (877 acres), or 74%, of its seagrass coverage. In the 10-yr period from 1992 to 2002, it appears that the rate of seagrass loss may have declined, but Perdido Bay still continued to lose an additional 3 ha (7 acres), or 2.6%, of its seagrass coverage during this 10-yr period. Total loss of seagrass coverage from 1940 to 2002 was 358 ha (885 acres), or 76%.

Upper Perdido Bay

The 1940 data (fig. 3) show only 0.4 ha (0.9 acres) of seagrass in Upper Perdido Bay. By 1979 (fig. 4), the seagrass coverage in this segment of the bay had apparently increased to 4.5 ha (11 acres), and the 1987 (fig. 5) data show 4.3 ha (11 acres) of seagrass. By 1992 (fig. 6), the seagrass coverage in Upper Perdido Bay had decreased to 1.3 ha (3 acres). By 2002, data showed no seagrass present in Upper Perdido Bay.

Middle Perdido Bay

Middle Perdido Bay showed seagrass coverage of 46.1 ha (114 acres) in 1940 (fig. 3). The majority of this coverage was located at the mouths of Soldier Creek and Palmetto Creek on the west side of the bay. By 1979, (fig. 4) seagrass coverage was greatly reduced to only 1.5 ha (4 acres), and by 1987, it had declined to 0.5 ha (1 acre). The 1992 data showed no seagrass present in Middle Perdido Bay, while the 2002 data shows 0.3 ha (0.7 acres).

Lower Perdido Bay

Since 1940, the majority of seagrass in Perdido Bay has been documented in the lower segment. Data show 434 ha (1,072 acres) of seagrass in Lower Perdido Bay in 1940 (fig. 3) and 209.7 ha (518 acres) in 1979 (fig. 4). By 1987 (fig. 5), seagrass coverage was 233.9 ha (578 acres). Photography from 1992 (fig. 6) shows 122.9 ha (304 acres) of seagrass in Lower Perdido Bay, while 2002 data indicate 121.1 ha (299 acres).

Entire Study Area

In 1940 (fig. 3), total Perdido Bay seagrass coverage was 480 ha (1,186 acres). As in subsequent years, most of this acreage (>90%) was located in Lower Perdido Bay. Seagrass maps from 1979 (fig. 4) showed total Perdido Bay seagrass coverage of 215.7 ha (533 acres). By 1987 (fig. 5), the total seagrass coverage in the bay was 239 ha (590 acres); by 1992 (fig. 6), it was 124.2 ha (307 acres); and by 2002, it was 121.3 ha (300 acres).

U.S. Geological Survey seagrass maps from 1987 (fig. 5) indicate that, in the 47-yr period between 1940 and 1987, approximately 50% of the total seagrass in Perdido Bay had been lost (figs. 3–5). Seagrass acreage decreased from 480 ha (1,186 acres) in 1940 (fig. 3) to 239 ha (590 acres) in 1987 (fig. 5), a loss of 241 ha (596 acres) baywide. Assuming a relatively steady rate of decline, the average rate of seagrass loss over this 47-yr period was approximately 5 ha (12 acres) per year, or just over 1% per year. Middle Perdido Bay lost about 45 ha (111 acres) of seagrass, or 98% of its 1940 total. Lower Perdido Bay lost 199 ha (492 acres) of seagrass, or 46% of its 1940 total. Upper Perdido Bay actually showed an increase of about 4 ha (10 acres) of seagrass during this time period. The decline of seagrass in Lower Perdido Bay was the greatest acreage loss, accounting for 199 ha (492 acres) of the total 241 ha (596 acres) lost baywide.

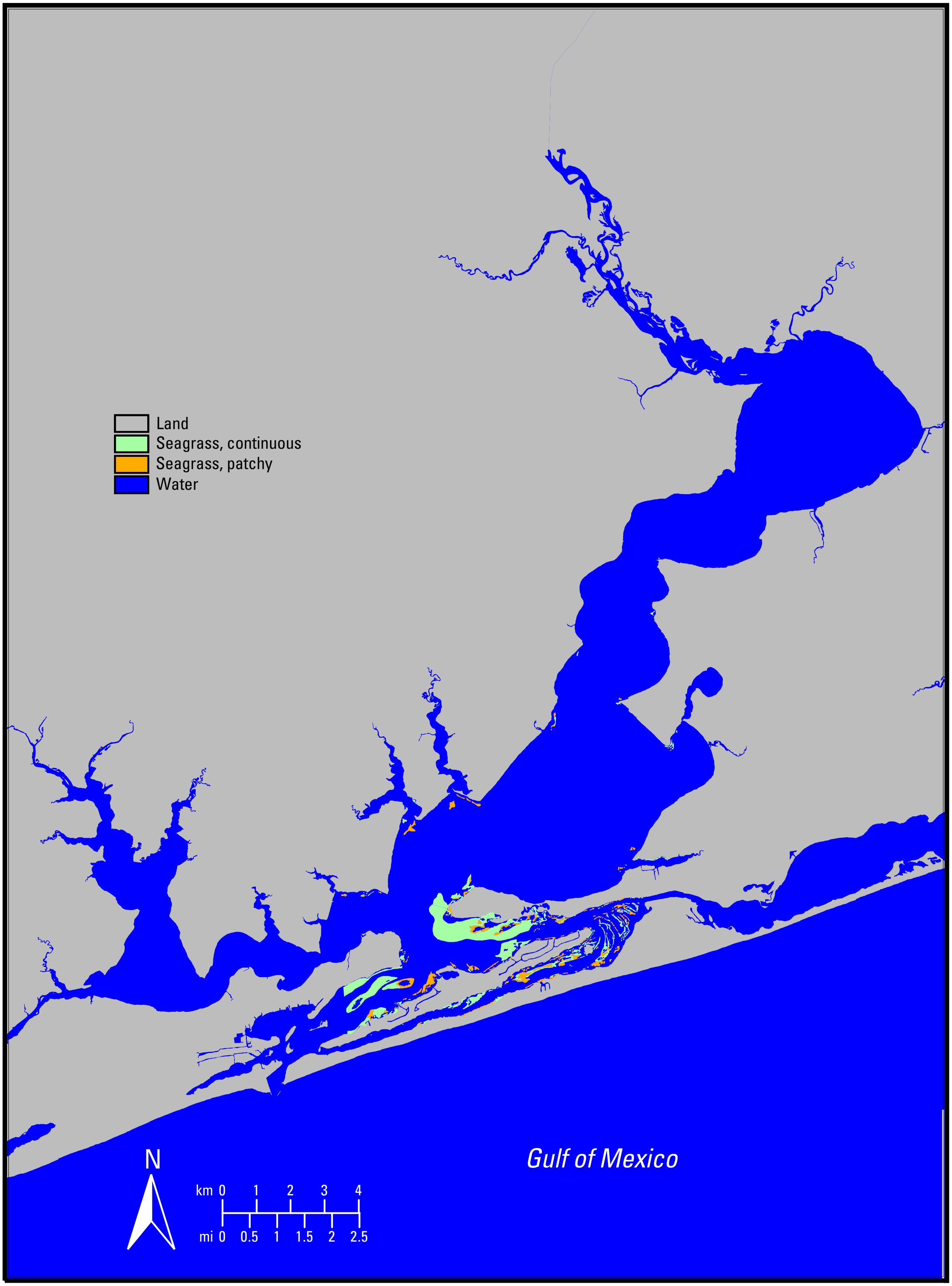

Figure 3. Distribution of seagrass in Perdido Bay, 1940.

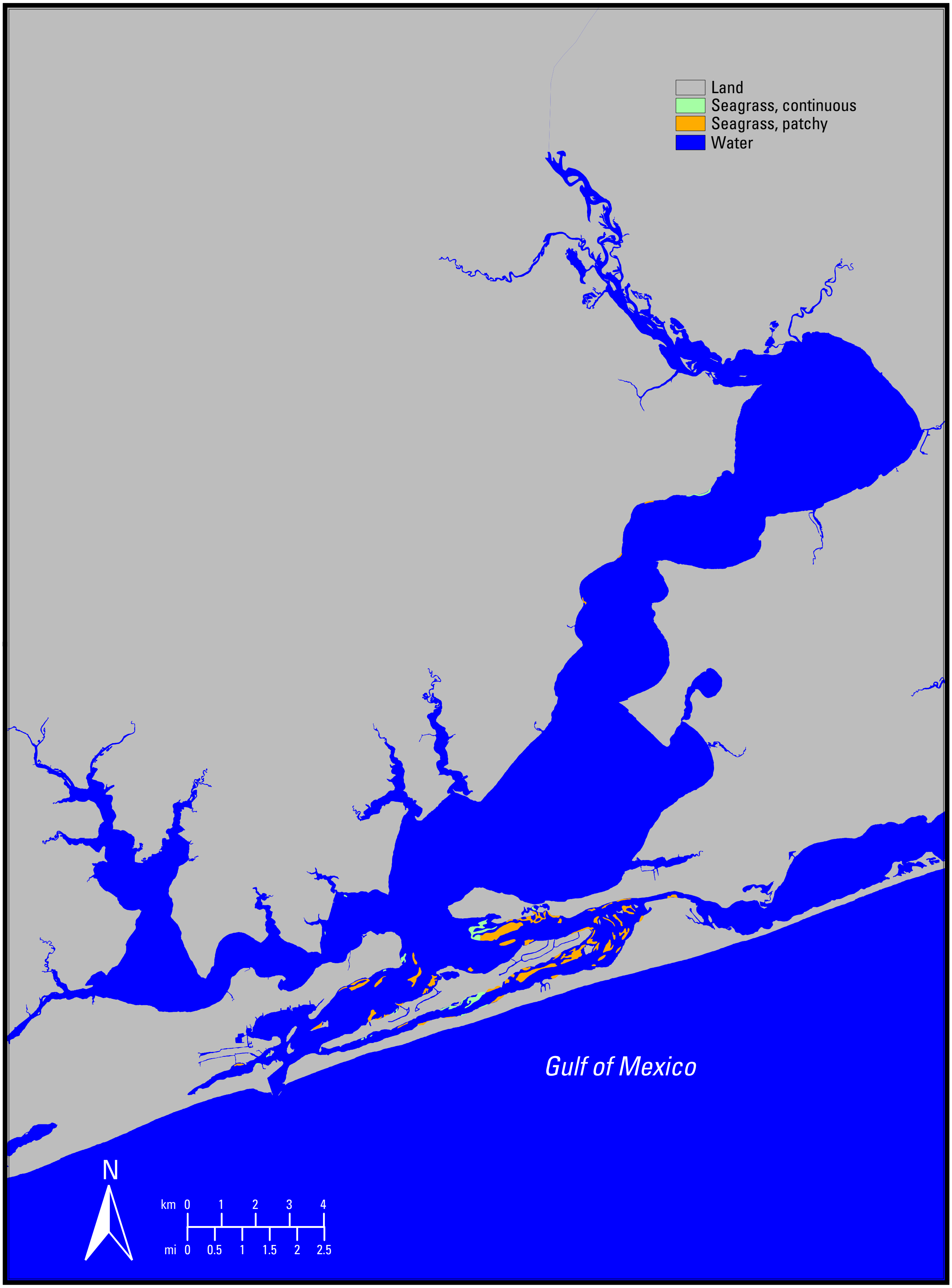

Figure 4. Distribution of seagrass in Perdido Bay, 1979.

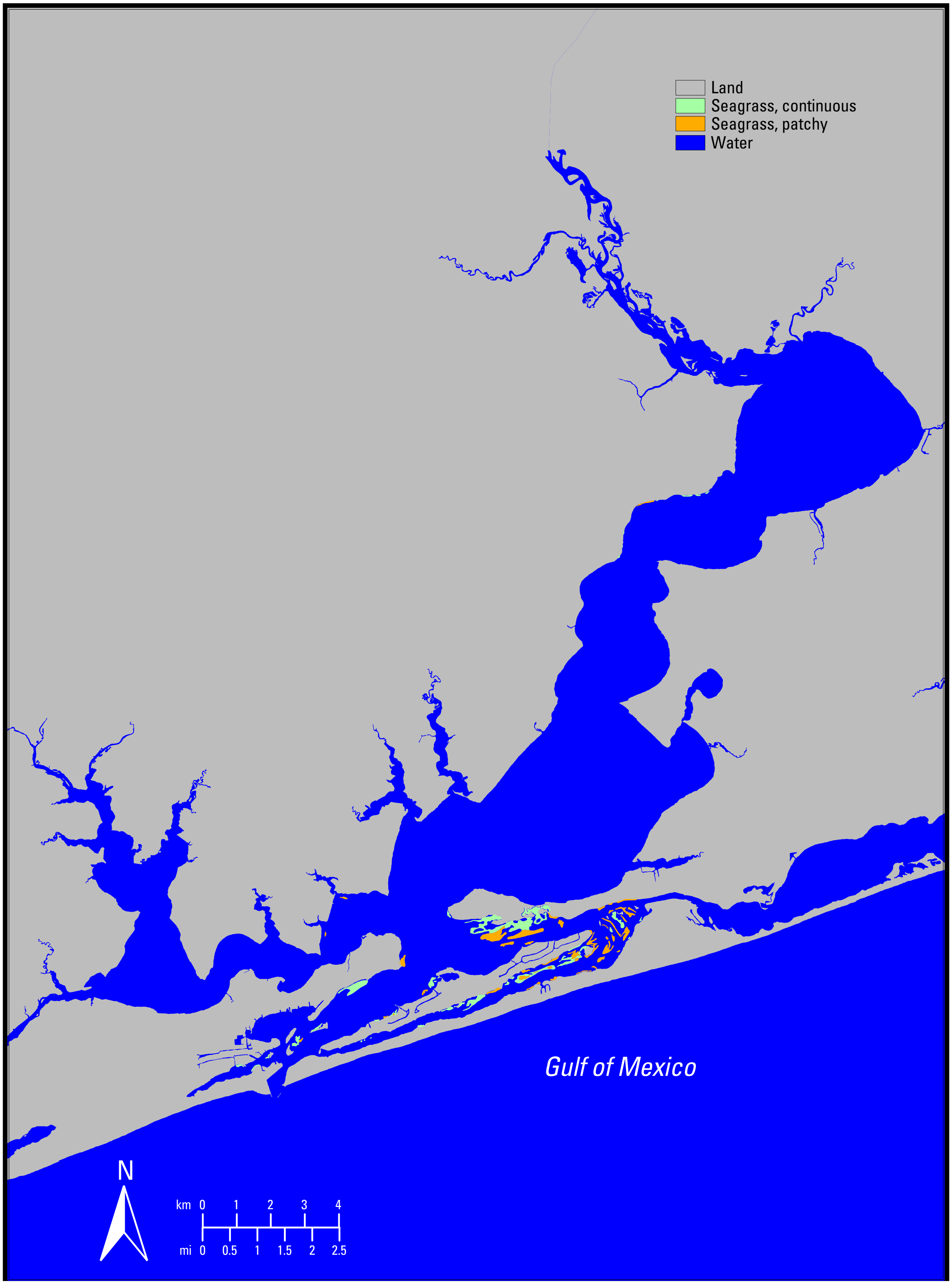

Figure 5. Distribution of seagrass in Perdido Bay, 1987.

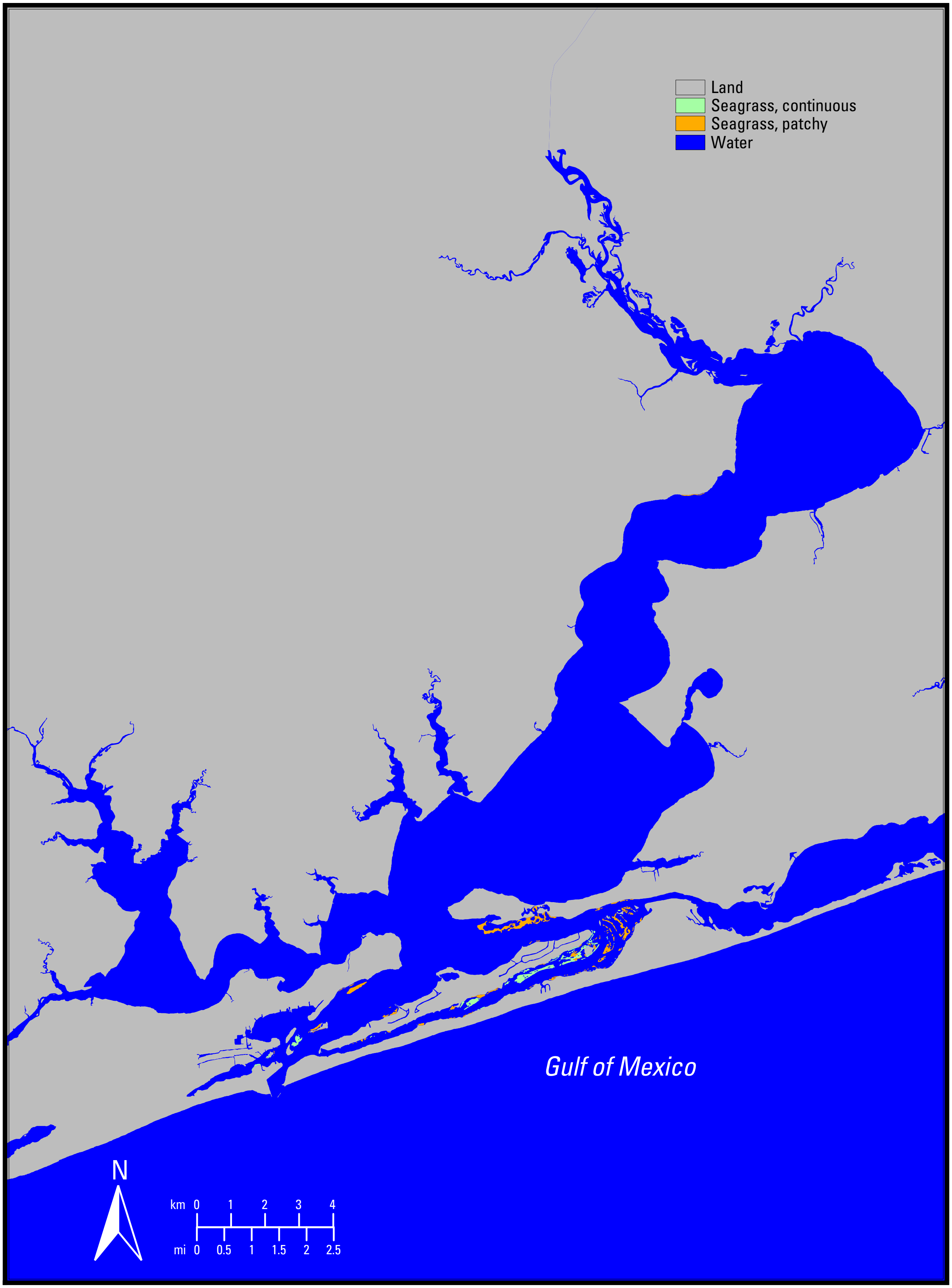

Figure 6. Distribution of seagrass in Perdido Bay, 1992.

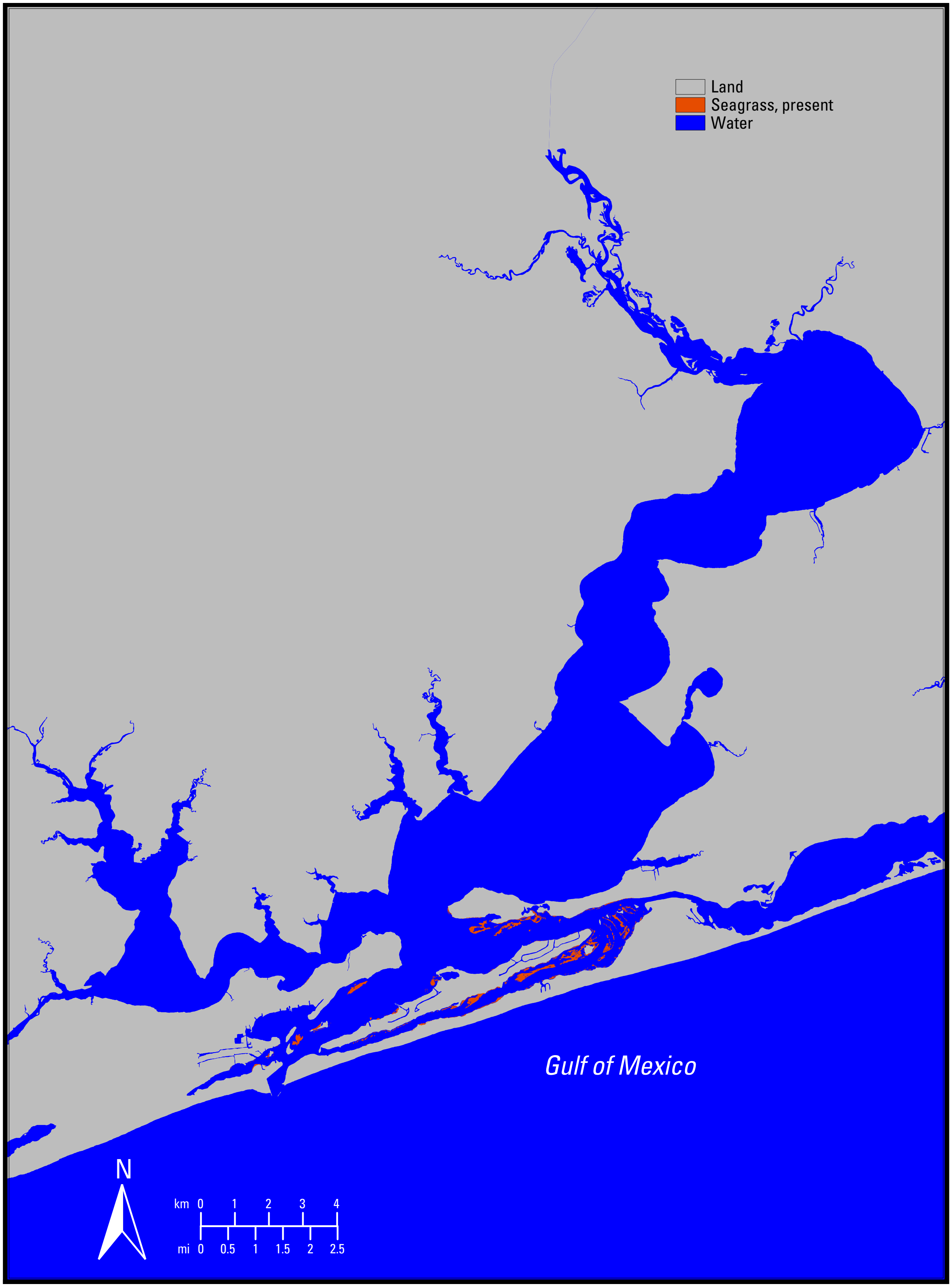

Figure 7. Distribution of seagrass in Perdido Bay, 2002.

U.S. Geological Survey seagrass maps from 1992 (fig. 6) show that total seagrass coverage in Perdido Bay was approximately 124 ha (306 acres). The percentage of seagrass lost in the 5-yr period from 1987 to 1992 was 48% (figs. 5 and 6). Total Perdido Bay seagrass acreage decreased from 239 ha (591 acres) to 124 ha (306 acres) during this time period. The average rate of seagrass loss during this 5-yr period was 141 ha (348 acres) per year, or almost 10% per year. Only about 1.2 ha (3 acres) of seagrass remained in Upper Perdido Bay, and only about 122 ha (301 acres) remained in Lower Perdido Bay. Unlike the 1940 (fig. 3) and 1987 (fig. 5) data, the 1992 (fig. 6) data indicated that no seagrass remained in Middle Perdido Bay. Seagrasses that were present at Palmetto and Soldiers Creeks had been lost by 1992.

In the 52-yr period from 1940 to 1992 (figs. 3–6), Perdido Bay lost 355 ha (877 acres), or 74%, of its seagrass coverage. The greatest loss of seagrasses occurred in Lower Perdido Bay where 310 ha (766 acres) of these 355 ha (877 acres) were lost. The lower section of Perdido Bay is the area that experienced the greatest developmental pressure during this 52-yr time period. Most of this development was related to vacation home construction and tourism.

It appears that during the 1992 to 2002 period, the rate of seagrass loss has declined. Data show a loss of 3 ha (7 acres), or 2.6%, during this recent 10-yr period, which equates to an estimated rate of decline of less than 0.3% per year.

Causes of Change

Upper/Middle Perdido Bay

Waters in Upper/Middle Perdido Bay are greatly influenced by the Perdido River, Elevenmile Creek, Bayou Marcus, Soldier Creek, Palmetto Creek, and numerous smaller tributaries. Wastewater and paper mill effluent are major point sources of pollution that can contain high levels of nutrients. Nonpoint pollution sources within the watershed include agricultural and silvicultural runoff that also can contribute high levels of nutrients to Perdido Bay. Other tributaries of Middle Perdido Bay (e.g., Palmetto Creek, Soldier Creek, and Bayou Garcon) have been residentially developed. Residential runoff (e.g., fertilizers, pesticides, and herbicides) also may affect water quality. Although primarily used for silviculture, lands surrounding Middle and Upper Perdido Bay and Perdido River are vulnerable to future residential development.

Lower Perdido Bay

Lower Perdido Bay has been greatly affected by accelerated residential, resort, and marine development. Numerous homes, condominiums, hotels, and marinas have crowded the coastline resulting in increased impervious surfaces and increased stormwater runoff, which usually cause increased nutrient enrichment, increased turbidity, decreased water clarity, decreased light penetration, and loss of seagrass. Ono Island and Perdido Key are approaching the maximum limits of development. Increased impervious surfaces generally contribute to greater amounts of urban stormwater running off into surrounding surface waters and transporting various pollutants such as fertilizers, pesticides, hydrocarbons, metals, and other toxins. Numerous marinas and residential docks, with associated vessels and activities, can provide additional stress to water quality and seagrasses.

Throughout Florida, the primary cause of water quality degradation is eutrophication Florida Department of Environmental Protection, 2000. Eutrophication has been blamed for seagrass losses in Florida and elsewhere (Cambridge and others, 1986; Tomasko and others, 1996). Loss of Perdido Bay seagrasses may be due to the result of decreased light, which is likely caused by increased turbidity and eutrophication. The degradation of water quality in the bay has occurred over time in proportion to population growth, increased land use, and development within the watershed. Over the past 10 yr, there has been a steady increase in population and development in the watershed and a concomitant loss of seagrasses.

Nutrient input is reported to be a major factor in Perdido Bay's water quality problems (Livingston, 2001). Residential and agricultural runoff are likely nutrient sources, as are domestic wastewater treatment facilities and paper mill discharges. Excess nutrients cause the reduction of light available to seagrass for photosynthesis in two general ways: (1) epiphytic growth, stimulated by nutrients, directly shade seagrass photosynthetic pigments; (2) nutrients cause phytoplankton blooms that absorb photosynthetically active radiation (PAR) in the water column.

Long-term trends are often viewed as gradual change over time; however, natural events such as hurricanes may cause large magnitude changes in a short time period. For example, the 1979 (fig. 4) seagrass maps were developed from data collected several months after a major hurricane (i.e., Frederick) heavily impacted Perdido Bay. Thus, when analyzing long-term trends, caution should be used when viewing short-term fluctuations in seagrass coverage that are to the result of isolated events.

Species Information

Three species of seagrasses have been found in Perdido Bay: wigeon grass (*Ruppia maritima*), shoal grass (*Halodule wrightii*), and turtle grass (*Thalassia testudinum*). Although not a true seagrass, the freshwater species water celery (*Vallisneria americana*) has been reported as the dominant submerged aquatic vegetation in the shallow, oligohaline waters of Upper Perdido Bay (Davis and others, 1999). Less frequently, wigeon grass occurs in shallow waters of Upper

Perdido Bay (Kirschenfeld and Turpin, personal observation). In contrast, euhaline species, shoal grass and turtle grass, are predominantly found in Lower Perdido Bay (Kirschenfeld and Turpin, personal observation).

Monitoring for Seagrass Health

Perhaps because of greater attention given to the larger Pensacola Bay System in Florida, or Weeks Bay and Mobile Bay in Alabama, management of Perdido Bay water quality and marine resources (e.g., seagrasses) has not been given adequate attention.

In 1991, the USFWS prepared a report on the changes in submerged vegetation coverage in Perdido Bay from 1940 to 1987 by analyzing and interpreting historical maps developed from aerial photography (U.S. Fish and Wildlife Service, 1991). In this report, a distinction was made between the true seagrasses and submerged vegetation, the latter of which additionally includes water celery. While no new field monitoring or water quality data were collected, the map analysis concluded that there was a loss of 235.5 ha (582 acres) of submerged vegetation, or a 48% net loss, from 1940 to 1987 (figs. 3–5).

Several transplanting efforts involving wigeon grass and water celery in upper Perdido Bay have indicated that water quality may be suitable for seagrass restoration efforts (Davis and others, 1999; Davis and Kirschenfeld, personal observation).

Weekly and monthly water quality monitoring data have been collected in Perdido Bay by biologists at the FDEP office in Pensacola. Measured parameters include nutrient levels, chlorophyll a, turbidity, Secchi depth, fecal bacteria, dissolved oxygen, temperature, and conductivity/salinity. In addition, Livingston (2001) published an extensive collection of many years of phytoplankton and nutrient data that was the result of studying the effects of pulp mill effluent on the Perdido Bay ecology.

Mapping and Monitoring Needs

As existing data show, the rate of seagrass loss in Perdido Bay appears to have increased from a loss rate of about 1% per year, from 1940 to 1987 (figs. 3–5), to a loss rate of about 10% per year, from 1987 to 1992 (figs. 5 and 6). More recently, however, from 1992 to 2002, it appears that the loss rate of seagrass in Perdido Bay has declined to less than 0.3% per year. Efforts should be undertaken to determine what activities and what best management practices have been implemented in Perdido Bay from 1992 to 2002 because data show an apparent successful effort to reduce the rate of decline of seagrass in the bay. Other estuarine systems could benefit from this information. If seagrass loss is still continuing, as historical trends indicate, additional seagrass preservation efforts and water quality improvements will be required in order to reverse this downward trend in seagrass coverage. The continued loss of seagrass, and the continued loss of its important intrinsic functions and values, will result in the further degradation of the ecology of Perdido Bay.

Restoration and Enhancement Opportunities

A seagrass monitoring effort and development of a seagrass management plan have recently been completed for nearby Big Lagoon and Santa Rosa Sound (Florida Department of Environmental Protection, 2001). Efforts such as this one should be expanded to include Perdido Bay.

In 1991, EPA biologists transplanted water celery in several different planting configurations in Upper Perdido Bay (Davis and others, 1999). In 1997, EPA and FDEP biologists successfully transplanted water celery and wigeon grass in Upper Perdido Bay (Davis and Kirschenfeld, personal observation). These results suggested that seagrass transplant efforts may be successful.

In recent years, there has been an elevated public awareness of environmental concerns in Escambia County, Florida, including water quality deterioration and loss of seagrasses. In 1998, the Pensacola FDEP office, in association with the Perdido Ecosystem Restoration Group (1998), led an effort to organize stakeholder meetings, develop action plans and management strategies, and publish *Perdido Ecosystem Management Strategies*. This document highlighted the public's concerns with poor water quality in Perdido Bay and the need for coordinated restoration efforts. In 1999, a special State of Florida Grand Jury issued a report on air and water quality concerns in Escambia County, which included 27 recommendations. These recommendations called for increased water quality monitoring efforts, more stringent water quality rules and regulations, reduced point and nonpoint sources of pollution, and the protection and preservation of environmentally sensitive lands. Perdido Bay has been classified as an Outstanding Florida Water, which means that there are more restrictive permit conditions required for developmental activities in this area.

Local efforts have been made to improve the water quality of Perdido Bay. Routine water quality monitoring by volunteers and publication of the data in the local newspaper (*Pensacola News Journal*) have heightened the public's environmental awareness. Citizens' efforts to reduce fertilization of lawns and reduce the use of pesticides and herbicides will help improve water quality. Local government efforts to reduce stormwater runoff and retrofit old storm drains will also help to improve water quality. The major industrial source of pollution to Perdido Bay is currently improving its process to improve the quality of its 0.92 m^3/s (21 million gal/d) of kraft mill effluent that is discharged into Elevenmile Creek and Perdido Bay. The cumulative positive effects of these efforts should be documented for positive reinforcement that these improvements and efforts are worthwhile.

Additional development will surely occur along the coast, but more stringent State and local regulations could be implemented to minimize negative environmental impacts. Improving the water quality of Perdido Bay, implementing coastal growth management plans, and continuing seagrass restoration efforts could reverse the trend of declining seagrass abundance.

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Pensacola Bay

By Lisa Schwenning,[1] Traci Bruce,[1] and Lawrence R. Handley[2]

Background

The Pensacola Bay system (fig. 1) has historically supported a rich and diverse ecology, productive fisheries, and considerable recreational opportunities. It has also provided an important resource for commercial shipping and military activities and has enhanced area aesthetics and property values. Unfortunately, for many years, point and nonpoint source pollution, direct habitat destruction, and the cumulative impacts of development and other activities throughout the watershed have combined to degrade the health and productivity of much of the Pensacola Bay system (Northwest Florida Water Management District, 1997).

During the 1800s and early 1900s, extensive logging of hardwood trees occurred on tributaries entering the Pensacola Bay system (Rucker, 1990). Historically, it is likely that the timber industry adversely impacted the upper bays of the system. Blackwater Bay and lower reaches of the Blackwater River and Pond Creek were covered with sawdust from mills in Bagdad, Fla. Sawn timber and sawdust are rich in turpines and other toxic substances. During logging operations, turbidity, nutrient concentrations, biological oxygen demand, and chemical oxygen demand were probably high, and dissolved oxygen levels were probably low. Benthic communities may have been scarce or nonexistent while the mills were operating at peak levels (Northwest Florida Water Management District, 1991).

Barge traffic and log rafting from tributaries to the Port of Pensacola unquestionably resuspended fine sediments in shoal areas of the system. Fairly extensive dredging of the Blackwater River was authorized and completed in 1916 by the U.S. Army Corps of Engineers (1990). Moskovits (1955) reported general silting problems in Pensacola Bay during 1949–50, as well as settlement of fouling organisms (barnacles, serpulid worms, hydroids, bryozoans, sponges, jingle shells, and *Mytilus* sp.) on artificial substrata. These organisms tend to settle in relatively clear water, so the presence of these animals suggests that suspended sediment loads were not sufficient to inhibit the settlement of zooplankton and that water quality may have been better during the early to mid-1950s (Northwest Florida Water Management District, 1991).

Predominant land uses within the riverine watershed include forestry, agriculture, military, and public conservation and recreation, as well as residential and other urban land uses around several communities. Much of the economic base of this area is provided by the extraction of natural resources, primarily timber and agriculture, but there are also indirect economic benefits provided by military activities and the service sector. Major public landholdings include portions of Eglin Air Force Base, Blackwater River State Forest (Northwest Florida Water Management District, 1997), and water management district lands along the Escambia and Blackwater Rivers.

Population growth and development apply the greatest pressure to the environment of Florida's coastal areas. As population increases, so do the demands on coastal resources. Over 75% of Florida's residents live in coastal counties, and a 50% increase is estimated by 2020 (Florida Department of Environmental Protection, 1998). In the three counties included in the Florida portion of the Pensacola Bay watershed, there has been a 220% increase in population since 1960. As a result of the population increase, there has been a dramatic change in land use of the Pensacola Bay area from agricultural to urban. Escambia County alone lost 13,247 ha (32,732 acres) of agricultural lands between 1992 and 1997; during the same time, the county gained approximately 6,071 ha (15,000 acres) of urban land use.

Scope of Area

The Pensacola Bay system consists of about 50,992 ha (126,000 acres), including its coastline of 885 km (550 mi) and open surface waters of about 373 km^2 (144 mi^2). The watershed covers nearly 18,130 km^2 (7,000 mi^2). The final outlet for the system is the Gulf of Mexico at the mouth of Pensacola Bay. The watershed of the Pensacola Bay system includes many different counties, cities, and incorporations. Approximately 65% of the entire watershed is located outside of Florida State lines; Alabama counties—including portions of Covington, Escambia, Conecuh, Butler, Crenshaw, Pike, Bullock, Montgomery, and Coffee—make up 11,632 km^2

[1] Florida Department of Environmental Protection, Northwest District.

[2] U.S. Geological Survey.

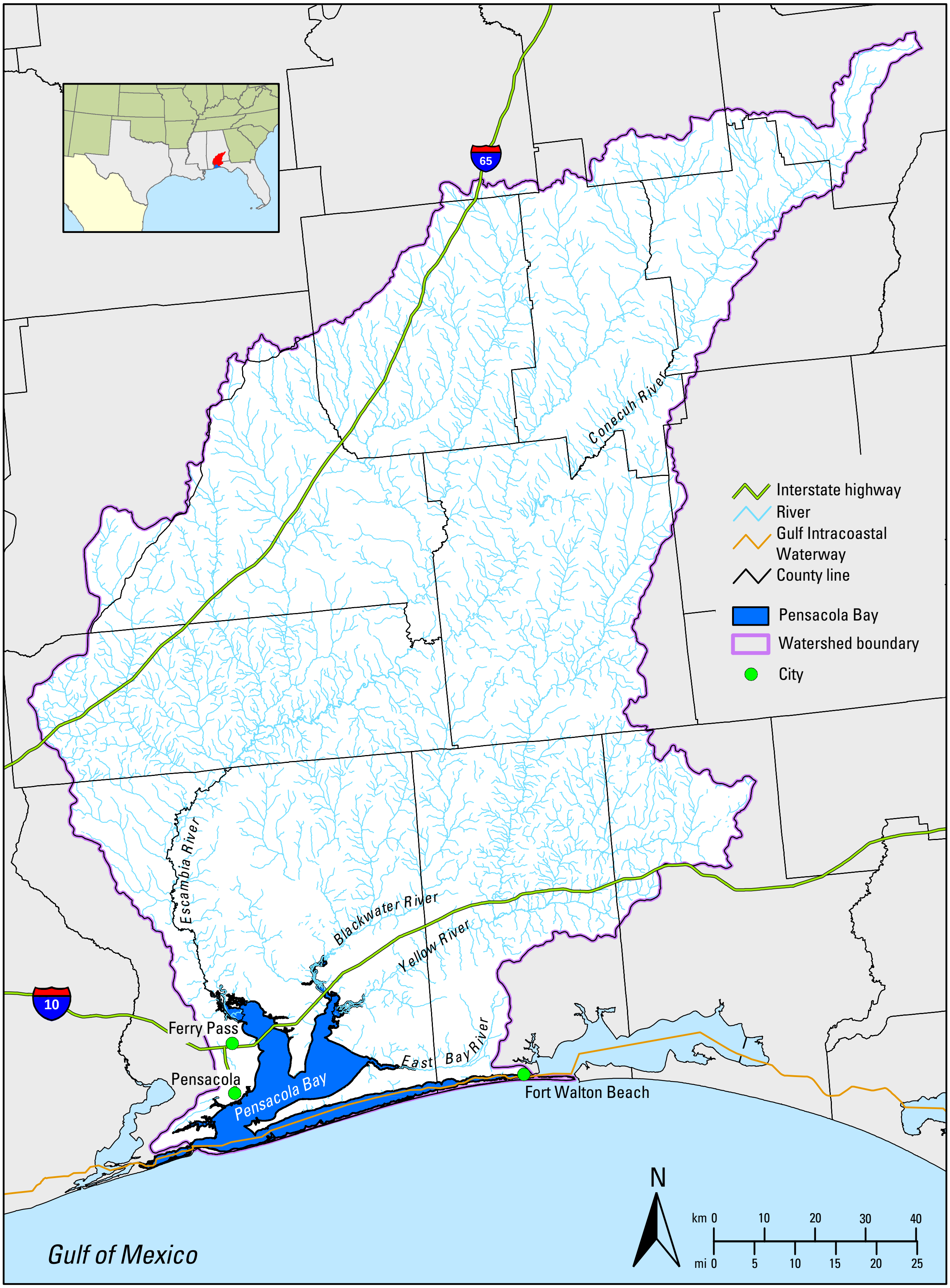

Figure 1. Watershed for the Pensacola Bay system.

(4,491 mi^2) of the Pensacola Bay system (Florida Department of Environmental Protection, 1998).

Approximately 857 km^2 (331 mi^2) (45%) of Escambia County form the western portion of the Pensacola Bay watershed. Nearly all of the 2,631 km^2 (1,016 mi^2) of Santa Rosa County, located to the east, are included in the watershed. Seventy-five percent of Okaloosa County (1,940 km^2, or 749 mi^2) lies in the watershed. Within Okaloosa County are the Yellow and Shoal Rivers, their tributaries, and the other half of Santa Rosa Sound. Only 648 km^2 (250 mi^2) (20%) of Walton County drain into the Pensacola Bay system and contain a portion of the Shoal River and tributaries. The western portion of the basin is predominantly urban, whereas the eastern portion is primarily low density, rural, and undeveloped.

The Pensacola Bay system has been divided into five segments: Escambia Bay, East Bay, Pensacola Bay proper, Big Lagoon, and Santa Rosa Sound (fig. 2).

Escambia Bay

Escambia Bay may be the most vulnerable of the Pensacola Bay segments. It is located north of the line between Garcon Point to the east and Magnolia Point to the west. Escambia Bay is commonly divided into upper and lower parts, with Interstate 10 delineating the boundary. This dividing line falls in an area that divides species between the dominant brackish water plant water celery (*Vallisneria americana*) in the upper bay and the dominant seagrass shoal grass (*Halodule wrightii*) in the lower part of the bay.

East Bay

East Bay is located east of the line between Garcon Point and Redfish Cove. East Bay, like Escambia Bay, is commonly divided into upper and lower segments, but with Robinson Point delineating the boundary. The area north of Robinson Point harbors all brackish water submerged aquatic vegetation. The area south of Robinson Point, or East Bay proper, contains shoal grass and turtle grass (*Thalassia testudinum*).

Pensacola Bay

Pensacola Bay is west of the Garcon Point Bridge. Escambia and East Bays drain into Pensacola Bay, along with Bayous Texar, Chico, and Grande. Pensacola Bay empties into the Gulf of Mexico at the Pensacola Pass.

Big Lagoon

Big Lagoon is oriented west to east and connects lower Perdido and Pensacola Bays. It is bordered on the south by Perdido Key and on the north by Pensacola Naval Air Station.

Santa Rosa Sound

Santa Rosa Sound is oriented west to east and connects Choctawhatchee and Pensacola Bays. It is bordered on the south by Santa Rosa Island and on the north by the Gulf Breeze Peninsula.

Methodology Employed To Determine and Document Current Status

The most current mapping study of seagrass coverage for the Pensacola Bay area was conducted more than 10 yr ago by the U.S. Geological Survey (USGS) National Wetlands Research Center (NWRC) by using natural-color aerial photography taken in 1992 at a 1:24,000 scale as part of the northeastern Gulf of Mexico seagrass mapping project.

The mapping protocol consisted of stereoscopic photointerpretation, cartographic transfer, and digitization in accordance with strict mapping standards and conventions.Other important aspects of the protocol included the development of a seagrass classification system, groundtruthing, quality control, and peer review.

The information derived from the photography was subsequently transferred by using a zoom transfer scope onto a stable medium overlaying USGS 1:24,000-scale quadrangle base maps. The primary data source was 1:24,000-scale natural color aerial photography flown by the National Aeronautics and Space Administration Stennis Space Center in fall 1992. In those cases where the data were inadequate or incomplete, contemporary supplemental data were acquired from other sources and used to complete the photographic coverage.

The seagrass classification system that was developed (see appendix for full discussion) consisted of two classes of open water—RIV (riverine, fresh water) and EST (estuarine or marine open water)—and five classes of seagrass habitats. One seagrass habitat class is continuous seagrass, CSG, for which no density distinction was made. The other four classes are patchy seagrass based on percent ground cover of patches in 5% increments: PSG1 (0%–10%, very sparse), PSG2 (15%–40%, sparse), PSG3 (45%–70%, moderate), and PSG4 (75%–95%, dense).

The groundtruthing phase included the participation of field staff from Gulf Islands National Seashore, U.S. Fish and Wildlife Service, Dauphin Island Sea Lab, Mississippi State University, Alabama Department of Conservation and Natural Resources, and Florida Department of Environmental Protection. Draft maps were sent out to these agencies for review and comments. All comments received were incorporated into the final maps.

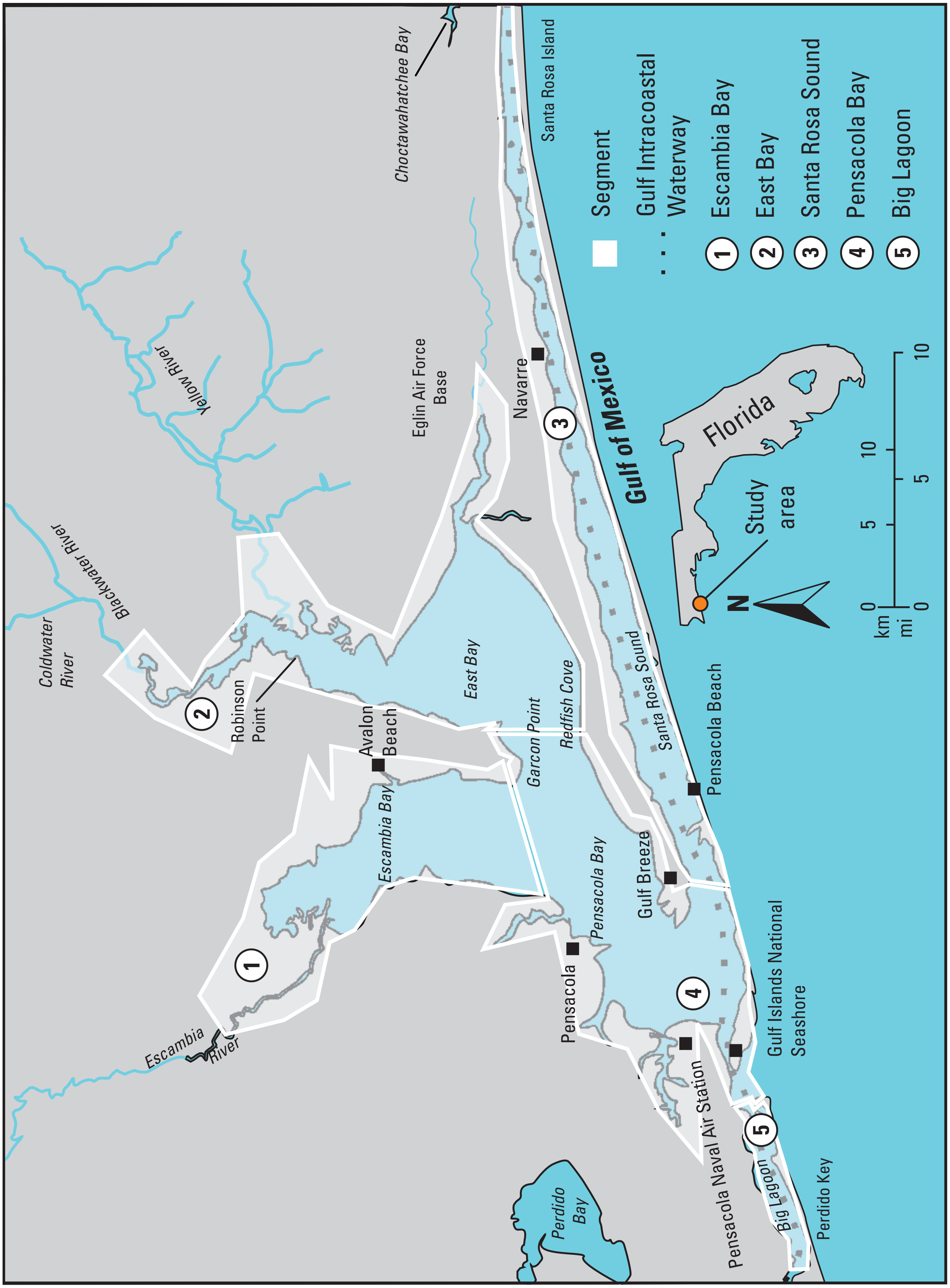

Figure 2. Scope of area for the Pensacola Bay vignette.

Methodology for Historical Trend Mapping

In 1999, the NWRC produced historical seagrass maps from 1:20,000-scale black and white aerial photography from 1964 for the U.S. Fish and Wildlife Service's Panama City Ecological Services and Fisheries Resources Office to use in the development of a trend analysis for Pensacola and Escambia Bays, Santa Rosa Sound, and Big Lagoon. These seagrass maps for 1964 and the previously described seagrass maps from 1992 were used to develop the trend analysis. The aerial photography from 1964 was analyzed with stereoscopic visual equipment; seagrasses were delineated onto Mylar® overlays by using a zoom transfer scope and USGS 1:24,000-scale topographic quadrangles as the base maps. The overlays were digitized by using ArcInfo (Environmental Systems Research Institute, Inc., Redlands, Calif.) and converted into Arc format files. The black and white aerial photography was of good to high quality for delineating seagrasses.

Classification of the seagrasses for the trend analysis followed the same protocol as that followed in the development of the northeastern Gulf of Mexico seagrass mapping using the 1992 aerial photography (see appendix 1).

Draft maps were sent out to the U.S. Fish and Wildlife Service for review and comments. All comments received were incorporated into the final maps. No groundtruthing took place for the historical seagrass delineations for 1964.

The NWRC currently holds the aerial photography, the interpreted overlays, and the Mylar® overlays for the 1964 maps.

Status and Trends

Seagrass beds that were extensive in 1949 were documented in 1975 as having receded or disappeared in most areas of the Pensacola Bay system (Northwest Florida Water Management District, 1997). In the 1940s and early 1950s, seagrass was abundant in all segments of the system, but less than 5%–10% of these levels remain. Figures illustrating these documented losses in all segments are available in chapter 12 of *Environmental and Recovery Studies of Escambia Bay and the Pensacola Bay System, Florida* (U.S. Environmental Protection Agency, 1975).

Escambia Bay

The loss of seagrass in Escambia Bay may be the most significant in the system, affecting the biological quality of the entire Pensacola Bay system (table 1; fig. 3). The earliest documented assertions of seagrass bed loss were in a 1955 report to the Florida State Board of Conservation by Murdock (1955) after an industrial plant began operating and discharging under permit into the Escambia River.

Table 1. Seagrass area for the Pensacola Bay system.

[Values are given in hectares (acres)]

	Escambia Bay	East Bay	Pensacola Bay	Big Lagoon	Santa Rosa Sound
1960	105 (259)	476 (1,175)	372 (918)	271 (670)	2,634 (6,508)
1980	24 (60)	99 (245)	55 (137)	236 (582)	1,489 (3,680)
1992	178 (441)	165 (408)	114 (282)	218 (538)	1,140 (2,816)

The Murdock report clearly linked the loss of seagrasses in Escambia Bay with industrial discharges into the drainage basin (Hopkins, 1973).

During the 1940s, seagrass was sparse in lower Escambia Bay. Transient beds, usually small, appeared, disappeared, and reappeared several times during the 50 yr of survey. For example, a recession and disappearance of most of Escambia Bay's true seagrass occurred from 1950 to 1975. In 1960 (fig. 4; table 2), Escambia Bay had approximately 105 ha (259 acres) of seagrass. Attempts in the 1970s to reestablish shoal grass in lower Escambia Bay were not successful. Seagrass beds were believed to have disappeared completely by 1976. By 1980 (fig. 5; table 3), it was found that 24 ha (60 acres) of mostly patchy beds were present. A greater reestablishment of several small beds was observed in the late 1980s, and by 1992 (fig. 6; table 4), patchy beds were observed, mostly around the mouths of bayous, totaling approximately 178 ha (441 acres).

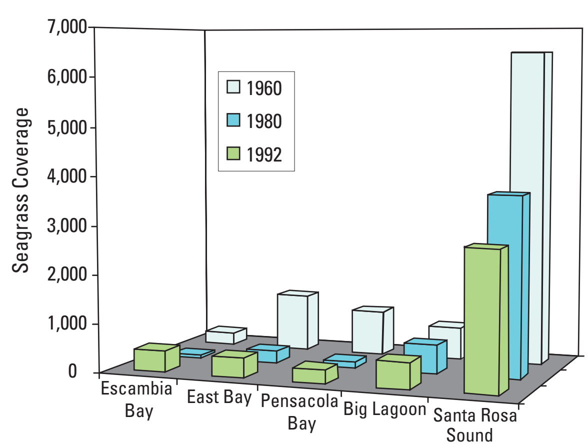

Figure 3. Summary of seagrass coverage in the Pensacola Bay system, 1960, 1980, and 1992.

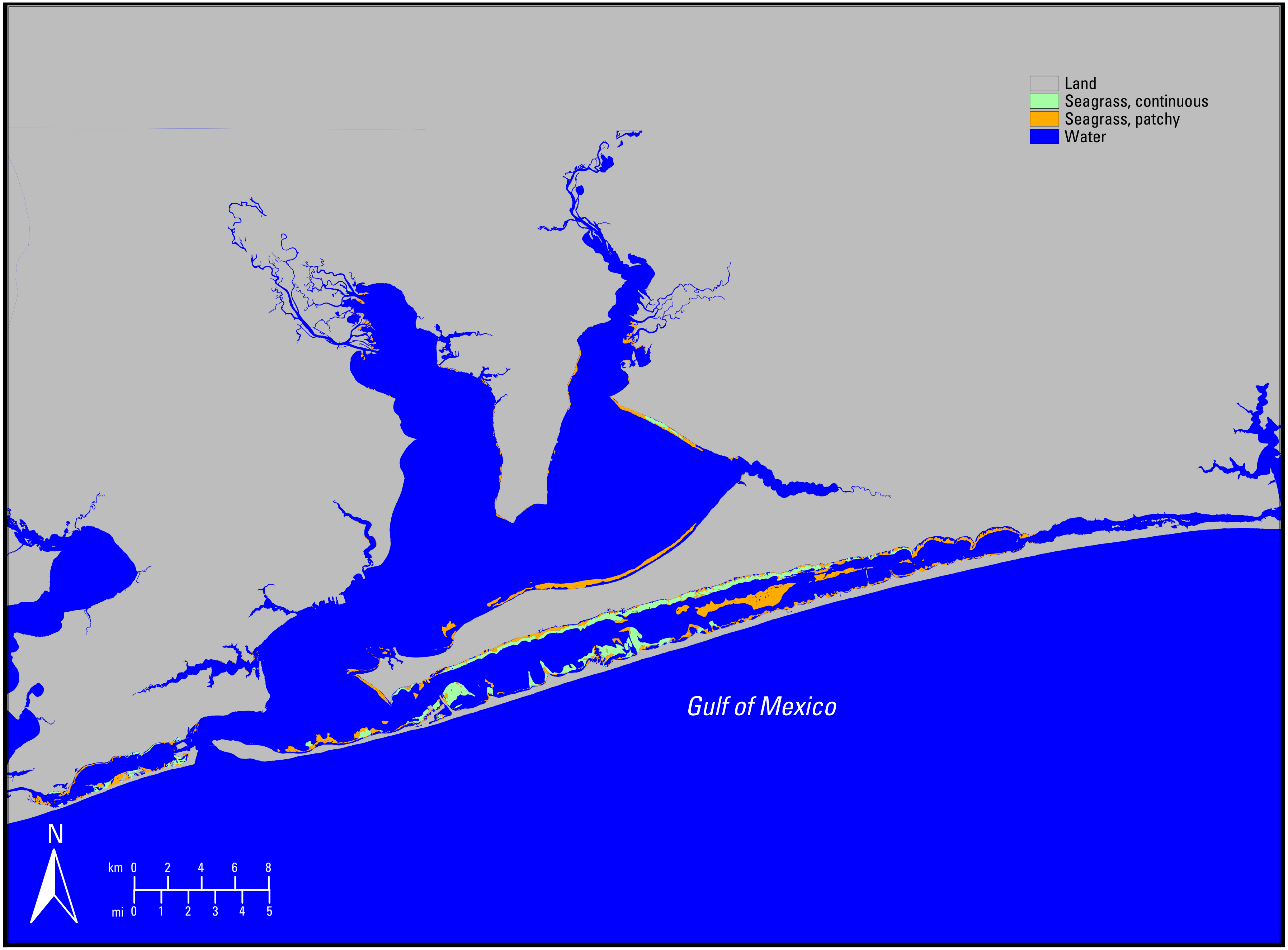

Figure 4. Distribution of seagrass in the Pensacola Bay system, 1960.

Table 2. Seagrass area in the Pensacola Bay system for 1960.

[Values are given in hectares (acres). CSG = continuous seagrass; PSG = patchy seagrass]

Segment	Class	Area
Escambia Bay	CSG	4 (10)
	PSG	101 (249)
East Bay	CSG	45 (110)
	PSG	431 (1,065)
Pensacola Bay	CSG	44 (108)
	PSG	328 (810)
Big Lagoon	CSG	107 (265)
	PSG	164 (405)
Santa Rosa Sound	CSG	1,247 (3,081)
	PSG	1,387 (3,427)

East Bay

From 1950 to 1975, East Bay experienced a decline similar to that in Escambia Bay (Northwest Florida Water Management District, 1991). In 1960 (fig. 4; table 2), 476 ha (1,175 acres) of seagrass were present. This area declined by 21% by 1980 (fig. 5; table 3) to 99 ha (245 acres); however, by 1992 (fig. 6; table 4), 165 ha (408 acres) were mapped in East Bay.

Pensacola Bay

According to the U.S. Environmental Protection Agency (1975), the disappearance of several small beds near the north end of the Pensacola Bay Bridge was documented in 1951 and was likely attributable to dredging. In 1960 (fig. 4; table 2), 372 ha (918 acres) of seagrass were mapped. In that same year, the Port of Pensacola was enlarged, which involved extensive dredge and fill activities. Additional dredging was done to the port in 1967. Most beds declined along the southern shore of Pensacola Bay and East Bay and disappeared by 1974. Based on historical data, seagrasses in Pensacola Bay declined from 372 ha (918 acres) in 1960 to 56 ha (137 acres) in 1980 (fig. 5; table 3). In 1992 (fig. 6; table 4), seagrass beds had increased to 114 ha (282 acres).

Big Lagoon and Santa Rosa Sound

Seagrass disappearance has not been quite as dramatic in Big Lagoon or Santa Rosa Sound; however, seagrasses have gradually decreased over the years in some areas of Big Lagoon and southwestern portions of Santa Rosa Sound along the Gulf Breeze Peninsula (Northwest Florida Water Management District, 1991).

In Big Lagoon, seagrass beds declined slightly from 271 ha (670 acres) in 1960 (fig. 4; table 2) to 236 ha (582 acres) in 1980 (fig. 5; table 3). Seagrass beds declined even further in 1992 (fig. 6; table 4) to 217 ha (538 acres). In contrast to the previously mentioned segments, Big Lagoon has not had dramatic losses but has had steady, continued decline in its seagrass meadows.

Santa Rosa Sound, like Big Lagoon, has experienced a steady decline from 1960 (fig. 4; table 2), when it had 2,634 ha (6,508 acres), to 1980 (fig. 5; table 3), when there were 1,489 ha (3,680 acres). There was a further decline by 1992 (fig. 6; table 4), when 1,139 ha (2,816 acres) were mapped.

Causes of Change

Major causes of seagrass loss were sewage and industrial waste discharges, dredge and fill activities, beachfront alteration, and changing watershed and land-use characteristics. In the Pensacola Bay system, many factors have affected the entire system, with certain factors having an increased local effect. Along the southern shore of East Bay, bulkheads and groins likely caused changes in nearshore water movements and energy; erosion of seagrass beds likely occurred as a result. Industrial discharge caused the loss of vegetation in the northeast section of Escambia Bay since these effluents remained near the shore in that area. For instance, the loss of seagrass around the sewage treatment plant there was caused first by the laying of a discharge pipe directly through the bed and later by sewage effluents (U.S. Environmental Protection Agency, 1975).

Although seagrass loss is thought to have occurred in the early 1900s, no data have been found to support the hypothesis. Two activities occurred in the area that could have caused loss. One possible impact was the dredging of the Gulf Intracoastal Waterway in Pensacola in 1910. A continuous 3 by 31 m (9 by 100 ft) channel was completed between Apalachicola River in Florida and New Orleans, La., in 1936. In 1942, the dimensions of the channel were increased to 4 m deep by 38 m wide (12 ft deep by 125 ft wide). The expansion of the waterway may account for increased traffic, providing indirect effects over a longer period of time.

Pensacola Bay waters have been impacted by numerous sources of pollution, resulting in a system that does not have the natural biodiversity and productivity expected of a system of this complexity. The massive quantities of point source waste previously discharged into the Pensacola Bay system between 1955 and 1964 have been significantly reduced. Based on surveys by the U.S. Environmental Protection Agency and its predecessor agencies, the combined quantity of waste discharged by the several major dischargers into the Escambia Bay drainage area was reduced by 40% for biological oxygen demand, 71% for total nitrogen, and 96% for total phosphorus by 1975 (U.S. Environmental Protection Agency, 1975).

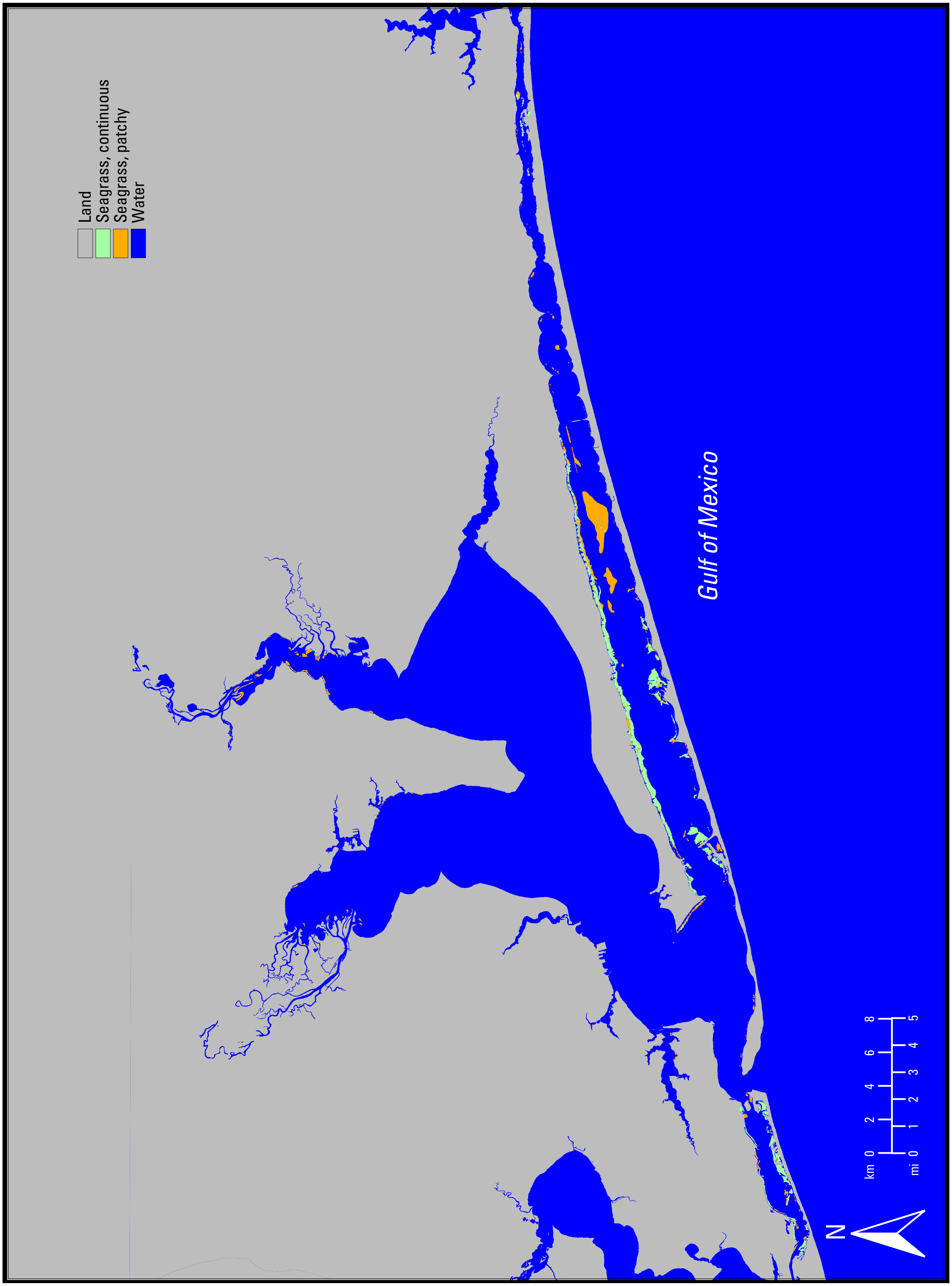

Figure 5. Distribution of seagrass in the Pensacola Bay system, 1980.

Table 3. Seagrass area in the Pensacola Bay system for 1980.

[Values are given in hectares (acres). CSG = continuous seagrass; PSG = patchy seagrass]

Segment	Class	Area
Escambia Bay	CSG	4 (10)
	PSG	20 (50)
East Bay	CSG	12 (30)
	PSG	87 (215)
Pensacola Bay	CSG	10 (24)
	PSG	46 (113)
Big Lagoon	CSG	193 (477)
	PSG	43 (105)
Santa Rosa Sound	CSG	860 (2,126)
	PSG	629 (1,554)

While it is true today that point sources have decreased significantly, nonpoint sources such as stormwater runoff are increasing, and pollution continues to degrade water and sediment quality throughout much of the system. Santa Rosa Sound and Big Lagoon are two of the few remaining bodies of water within the Pensacola Bay system that still harbor seagrass beds.

Livingston and others (1972) reported that 784,428 m^3 (1,025,933 yd^3) of sediments were dredged from Mulatto Bayou in 1965 during construction of the Interstate 10 bridge across upper Escambia Bay. Additional dredging occurred in 1970 when finger canals were created for housing developments and a narrow channel was dug to open the southern portion of the bayou to Escambia Bay. Livingston and others (1972) made a compelling case that the creation of deep borrow pits within the bayou and the new narrow entrance to the bay were nutrient traps that reduced circulation and flushing of the bayou and depressed oxygen availability. In addition, it has been reported that seagrass beds in Mulatto Bayou were replaced by sludge deposits containing hydrogen sulfide as a result of dredging (Northwest Florida Water Management District, 1991).

Species Information

The Pensacola Bay system contains four species of true seagrasses—turtle grass, manatee grass (*Syringodium filiforme*), star grass (*Halophila engelmannii*), and shoal grass—and two brackish water species, water celery and wigeon grass (*Ruppia maritima*).

The upper portion of Escambia Bay contains mostly water celery, with a few patches of wigeon grass; the lower portion has lost all seagrass. East Bay has lost all seagrass with the exception of a few areas in upper East Bay around the mouth of the Blackwater River. Big Lagoon and Santa Rosa Sound have the last two remaining stands of turtle grass and shoal grass in the Pensacola Bay system. Santa Rosa Sound holds, in addition, a few patches of manatee grass and star grass.

Though shoal grass is a pioneering species and is usually the first to colonize an area, no historical data have been found supporting transition from one species to another.

Monitoring for Seagrass Health

In 1974, the Escambia Bay Recovery Study Team obtained photographs from the Florida Department of Transportation (FDOT) and used them to develop maps of the seagrass beds. Their FDOT file of aerial photographs of Escambia and Santa Rosa Counties dated back to 1949. The study team project personnel also conducted aerial surveys in 1974. These overflights of the bays were made sporadically and occurred more frequently over areas of greater highway construction. (Refer to U.S. Environmental Protection Agency, 1975, to see these maps.)

In 1996, a study was conducted by Heck and others (1996) for the Gulf Islands National Seashore within Big Lagoon and Santa Rosa Sound, where two of the last stands of seagrass within the Pensacola Bay system remained. Annual groundtruthing, inventory of the seagrass community types, and water-quality parameters were monitored. The study suggested a significant decline in seagrass meadows in the area. The investigators of this study hypothesized a light-attenuation problem caused by an increase in turbidity in these water bodies (Kirschenfeld, 1999).

In 1999, the Florida Department of Environmental Protection was awarded a seagrass monitoring grant from the U.S. Environmental Protection Agency's Gulf of Mexico Program to determine the extent of the problem. Seagrass monitoring, combined with water-quality and light-attenuation monitoring, was conducted over 2 yr, resulting in the publication of the *Seagrass Management Plan for Big Lagoon and Santa Rosa Sound* (Florida Department of Environmental Protection, 2001).

The seagrass monitoring component of this study involved measurements of the depth of growth, percent cover, epiphytic coverage, Braun-Blanquet density (the second year), species composition, and canopy height (for more information on the Braun-Blanquet density, see *http://chla.library.cornell.edu/cgi/t/text/text-idx?c=chla;idno=2917578*). The monitoring parameters (Hach Environmental, Loveland, Colo.) measured were temperature, pH, conductivity, salinity, and dissolved oxygen. Water samples were also taken from each monitoring site once a month to measure turbidity, color, chlorophyll a, total suspended solids, total Kjeldahl nitrogen, ammonia, nitrate + nitrite, total phosphorus, orthophosphorus, and algal growth potential. The *Seagrass Management Plan for Big Lagoon and Santa Rosa Sound* (Florida Department of

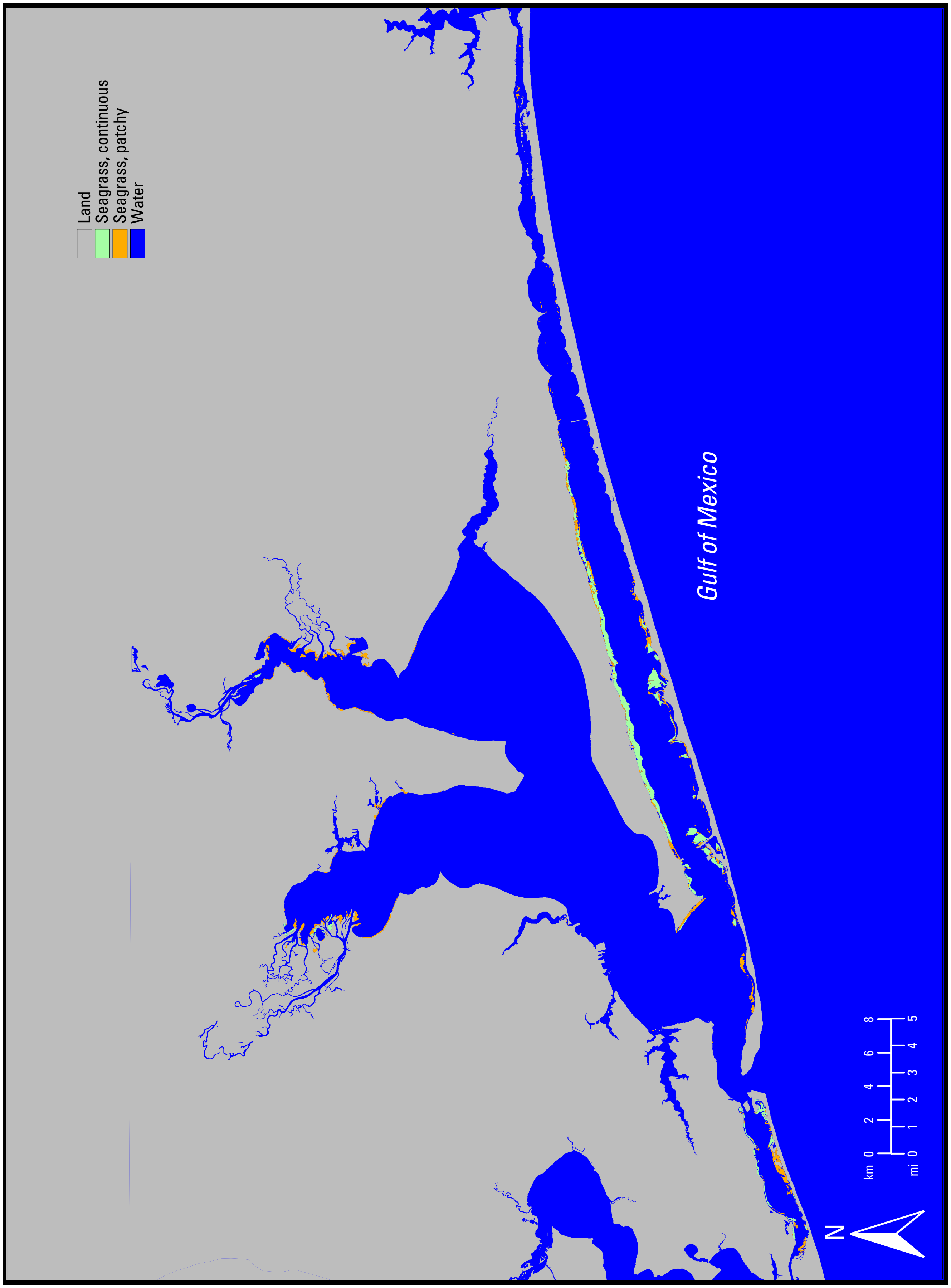

Figure 6. Distribution of seagrass in the Pensacola Bay system, 1992.

Table 4. Seagrass area in the Pensacola Bay system for 1992.

[Values are given in hectares (acres). CSG = continuous seagrass; PSG = patchy seagrass]

Segment	Class	Area
Escambia Bay	CSG	36 (88)
	PSG	143 (353)
East Bay	CSG	5 (13)
	PSG	160 (395)
Pensacola Bay	CSG	13 (33)
	PSG	101 (249)
Big Lagoon	CSG	99 (245)
	PSG	119 (293)
Santa Rosa Sound	CSG	796 (1,968)
	PSG	343 (848)

Environmental Protection, 2001) contains an overview of the results from the entire monitoring effort.

Epiphytic Coverage

In summary, 17 out of 20 sites monitored in spring 2001 indicated an increase of epiphytic coverage compared to the previous 2 yr, with the most dramatic increase in Big Lagoon. This change suggests that there may have been an increase in the amount of nutrients over the past 2 yr. If eutrophication has taken place, there should be more indications of this in other areas of monitoring, such as in water quality (Florida Department of Environmental Protection, 2001).

Water-quality Monitoring

Water-quality monitoring was conducted every month at each of the 20 monitoring sites. The data were categorized into two sets of parameters, Hydrolab and laboratory. The Hydrolab parameters were measured on site with a water-quality meter and included temperature, dissolved oxygen, pH, and salinity. Secchi depth was also included with the Hydrolab parameters. Laboratory parameters required chemical analysis and included turbidity, color, conductivity, nonfiltered residue, ammonia, Kjeldahl nitrogen, nitrate + nitrite, total phosphate, and chlorophyll a.

Algal growth potential was measured every other month. Algal blooms and excessive turbidity in the water column can reduce the amount of sunlight available to seagrass for photosynthesis. One indication of increased algae caused by excessive nutrients is an increase in both chlorophyll a and color. When turbidity increases but chlorophyll a and color remain low, however, stormwater runoff or some other stirring mechanism (i.e., boating) may be the cause of light attenuation. Results of water-quality monitoring for the parameters of chlorophyll a, color, and turbidity suggest eutrophication over the past 2 yr. The summer months of July and August normally show seasonal trends of increased chlorophyll a and color compared to other months of the same year. Not only is this trend now evident in the data, but those seasonal highs have also increased over the last 2 yr.

Light-attenuation Monitoring

Photosynthetically active radiation (PAR) is the part of the spectrum of sunlight, between 400 and 700 nanometers (nm), that plants use for energy. Seagrass requires about 25% of the surface PAR for photosynthetic production. For example, if the PAR is 9.5 Moles per square meter per second ($M/m^2/s$) just below the surface of the water, then the seagrass requires 2.4 $M/m^2/s$ of PAR for optimal survival. Underwater vegetation is less likely to receive sufficient PAR to sustain growth when light reaching the plants is reduced because of water turbidity and algal blooms.

In this study, light-attenuation monitoring suggested that, immediately after a heavy rainfall, stormwater input inhibits the amount of PAR available to the seagrass. Figure 7 illustrates light attenuation at the deep edge of a seagrass bed in 1.4 m (4.5 ft) of water. Although the monitored storm event lasted only 1 d, the inhibited PAR persisted for several days after the rainfall because of increased turbidity and nutrient enrichment from stormwater runoff.

Mapping and Monitoring Needs

Although historical data documenting extensive losses of seagrass in Pensacola Bay are available, recent (post-1992) data are scarce. Turtle grass and shoal grass are nearly absent from Pensacola and Escambia Bays; only water celery and wigeon grass are surviving in limited areas of upper Escambia Bay. These species are also doing well in portions of Blackwater Bay, and shoal grass, turtle grass, manatee grass, and star grass are surviving in portions of Santa Rosa Sound (Northwest Florida Water Management District, 1997) and Big Lagoon.

Furthermore, standardized mapping of seagrass beds at appropriate temporal and spatial scales should be made every 5–10 yr. Groundtruthing is recommended for verification of those mapping initiatives.

Continuation of a strategic long-term monitoring program that includes biotic and abiotic parameters of the community is needed. Such parameters include, but are not limited to, areal coverage, shoot length, epiphyte load, percent coverage, density, and water- and sediment-quality indicators. This monitoring should determine causes of changes in health, seagrass species composition, density, and distribution including total loss or gain. Shoot density and faunal communities should be measured to determine

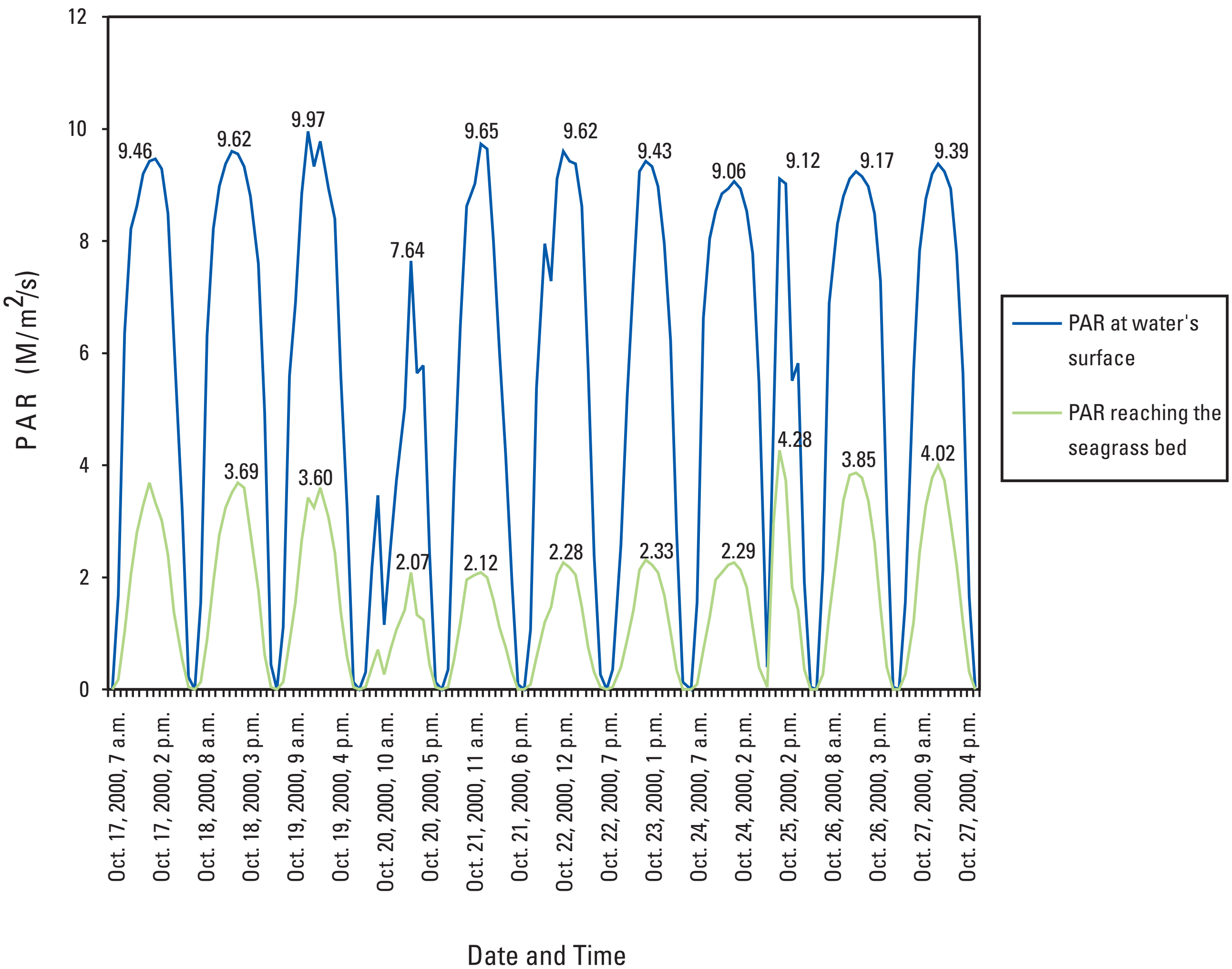

Figure 7. The amount of photosynthetically active radiation (PAR) was diminished for several days while turbidity settled after a storm event at Big Lagoon in Florida. October 18 indicates a normal day, with surface PAR around 9.6 Moles per square meter per second (M/m²/s) and underwater PAR (at grass blade depth) around 3.7 M/m²/s. Available PAR to the seagrass bed is about 38% of surface PAR. October 20 indicates a storm event with decreased surface and underwater PAR caused by clouds and rain. Following the storm event, surface PAR returned to normal, while underwater PAR remained inhibited for days because of increased turbidity from stormwater runoff. Available PAR decreased to insufficient levels of around 22%, which persisted for 4 d before returning to normal.

habitat function and productivity. Changes in light quality and quantity as they affect seagrass health may be related to nutrient loading and stimulation of phytoplankton blooms, epiphytes, and drift macroalgae and should be included in the monitoring program. Assessment of the biogeochemical environment occupied by belowground tissue, such as porewater composition, would also be useful, as would the monitoring of contaminants in surface water, sediment, and seagrass tissue. All of these monitoring activities, however, would not be able to be accomplished without a large source of funding. The prioritization of activities would be to study the parameters related to seagrass health and growth, such as areal coverage, density, shoot length, etc. If these attributes were found to be declining, then other parameters—such as contaminants in water, sediment, and tissue and light quality and quantity—would be useful to measure.

Studies should address the following:

- effects of boating impacts (trawling and boat traffic, including jet skis, motorboats, and sailboats),
- effects of nonpoint pollution sources on seagrass beds,
- socioeconomic values and impacts of management, on users
- effects of the increase in the human population (nonpoint nutrient loading, user impacts),
- factors involved in repairing propeller scar damage,
- dredging effects on light attenuation, and
- indirect effects of dredging.

Restoration and Enhancement Opportunities

Habitat functions and productivity of natural seagrass community types should be identified, as well as should linkages with other habitats to support habitat conservation, creation, enhancement, and restoration in the Pensacola Bay system.

The Florida Department of Environmental Protection is working with Federal, State, and local agencies and with local businesses, individuals, and civic organizations to promote and facilitate a large-scale restoration project. Project Greenshores is a habitat restoration project designed to restore an area in Pensacola Bay that was once lightly populated with seagrass. This project will attempt to restore between 4 and 8 ha (10 and 20 acres) of oyster reefs and emergent and submerged grasses, as well as to educate the public on the importance of seagrass communities in estuarine habitats. Seagrasses to be planted include turtle grass, shoal grass, and wigeon grass. This project will also be used as a template for further restoration projects in northwest Florida and throughout the Gulf of Mexico coastal area.

The Ecosystem Restoration Section of the Florida Department of Environmental Protection has initiated many local shoreline restoration projects involving emergent and submerged vegetation and propeller scar restoration. Restoration sites have included Wayside Park in Gulf Breeze, the mouth of Bayou Texar, and propeller scarred areas in Big Lagoon and elsewhere. The Florida Marine Research Institute states that the Pensacola Bay system has, in general, light to moderate scarring with a few areas labeled as severe. Propeller scar restoration is new to the Pensacola Bay system: few restoration attempts have been made within the past 5 yr, but restoration opportunities are becoming more attainable.

Future restoration opportunities should include experimental research on seagrass bed creation and restoration to determine how donor stocks, including drift grass (detached plants), should be selected to achieve maximum success. The research should also determine if there are methods to accelerate natural recruitment of seagrasses.

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Choctawhatchee Bay

By Barbara Ruth[1] and Lawrence R. Handley [2]

Background

The Choctawhatchee River and Bay system historically has supported a rich and diverse ecology that provides substantial economic and quality-of-life benefits to residents of northwest Florida (Northwest Florida Water Management District, 1996). The Choctawhatchee Bay area has become a highly desired area for relocation for many people: the perceived overdevelopment in southern Florida has led many to look for less highly developed areas. Although the area has no major industry, urban and suburban development, along with businesses that support Eglin Air Force Base (AFB) activities and an extensive retirement community, is creating impacts on the ecosystem through additional stormwater runoff, resource utilization, and similar pressures that are caused by development. In addition, as population along the bay's shoreline has increased, the number of seawalls and docks has also increased, reducing the intertidal areas that protect the bay's seagrass resources.

The Choctawhatchee Bay watershed encompasses nearly 13,856 km^2 (5,350 mi^2) and spans portions of northwest Florida and southern Alabama (fig. 1). Much of the watershed is composed of agricultural, silvicultural, and rural land, with a large portion of the Florida part of the watershed also making up the lands of Eglin AFB.

Historically, northwest Florida has been primarily an agricultural area, as is the case with the Choctawhatchee Bay watershed. Prior to the acquisition of lands for Eglin AFB in the early 1900s, the area was heavily timbered, causing major system alterations including extensive dirt road building, overharvest of longleaf pine (*Pinus palustris*), and conversion of lands to other uses. Northwest Florida still contains more than 50% of the remaining old-growth stands of longleaf pine, however, and is widely recognized as a nationally significant reservoir of rare biota (Feller and others, 1997).

The bay is supplied with fresh water from the Choctawhatchee River and several creeks and springs and with ground water from the Floridan aquifer system (Northwest Florida Water Management District, 1996). The only direct opening to the Gulf of Mexico is through the relatively shallow East Pass (immediately west of Destin), an artificial channel opened in 1929 to provide a permanent pass through a previously intermittent opening. The bay also opens to the Gulf Intracoastal Waterway (GIWW) in the east and to Santa Rosa Sound and the GIWW in the west. (The GIWW is regularly maintained by the U.S. Army Corps of Engineers.) Pensacola Pass is separated from the west side of the bay by an 84-km (52-mi) portion of the GIWW that was dredged initially in the 1940s (Northwest Florida Water Management District, 1996). Choctawhatchee Bay is a stratified system with low tidal energy, limited flushing (Blaylock, 1983; Livingston, 1986), and a halocline (noticeable changes in salt concentrations between the surface waters and lower waters) (Blaylock, 1983; Livingston, 1986). The bay's main freshwater influence is the Choctawhatchee River, which has a rate of 243 m^3/s (8,580 ft^3/s) (Northwest Florida Water Management District, 1996) annual average discharge. With the freshwater influence accompanied by the influx of salt water from the Gulf of Mexico, the bay's salinity level is constantly fluctuating.

The seagrass beds in Choctawhatchee Bay are considered extremely important because they support diverse populations of fish and invertebrates, including many recreational and commercial species (Northwest Florida Water Management District, 1996). Commercial and sport fisheries are important in the area, as is oyster harvesting. Significant portions of Choctawhatchee Bay are designated as Class II waters of the State (designated for shellfish propagation or harvesting). Commercially and recreationally important fish and shellfish include shrimp (*Penaeus* spp.), eastern oysters (*Crassostrea virginica*), spotted seatrout (*Cynoscion nebulosus*), gulf menhaden (*Brevoortia patronus*), red drum or redfish (*Sciaenops ocellatus*), blue crab (*Callinectes sapidus*), gulf flounder (*Paralichthys albigutta*), striped mullet (*Mugil cephalus*), and white mullet (*Mugil curema*). In addition, the bay system includes endangered species including the gulf sturgeon (*Acipenser oxyrinchus*), which migrates through the bay into the Choctawhatchee River to spawn. The easternmost part of the bay and the river delta, as well as the Rocky Bayou State Park Aquatic Preserve, are designated by the Environmental Regulation Commission of the Florida Department of Environmental Protection as Outstanding Florida Waters (chapter 62-302.700, Florida Administrative Code), worthy of special protection because of their natural attributes.

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[1] Florida Department of Environmental Protection, Northwest District.

[2] U.S. Geological Survey.

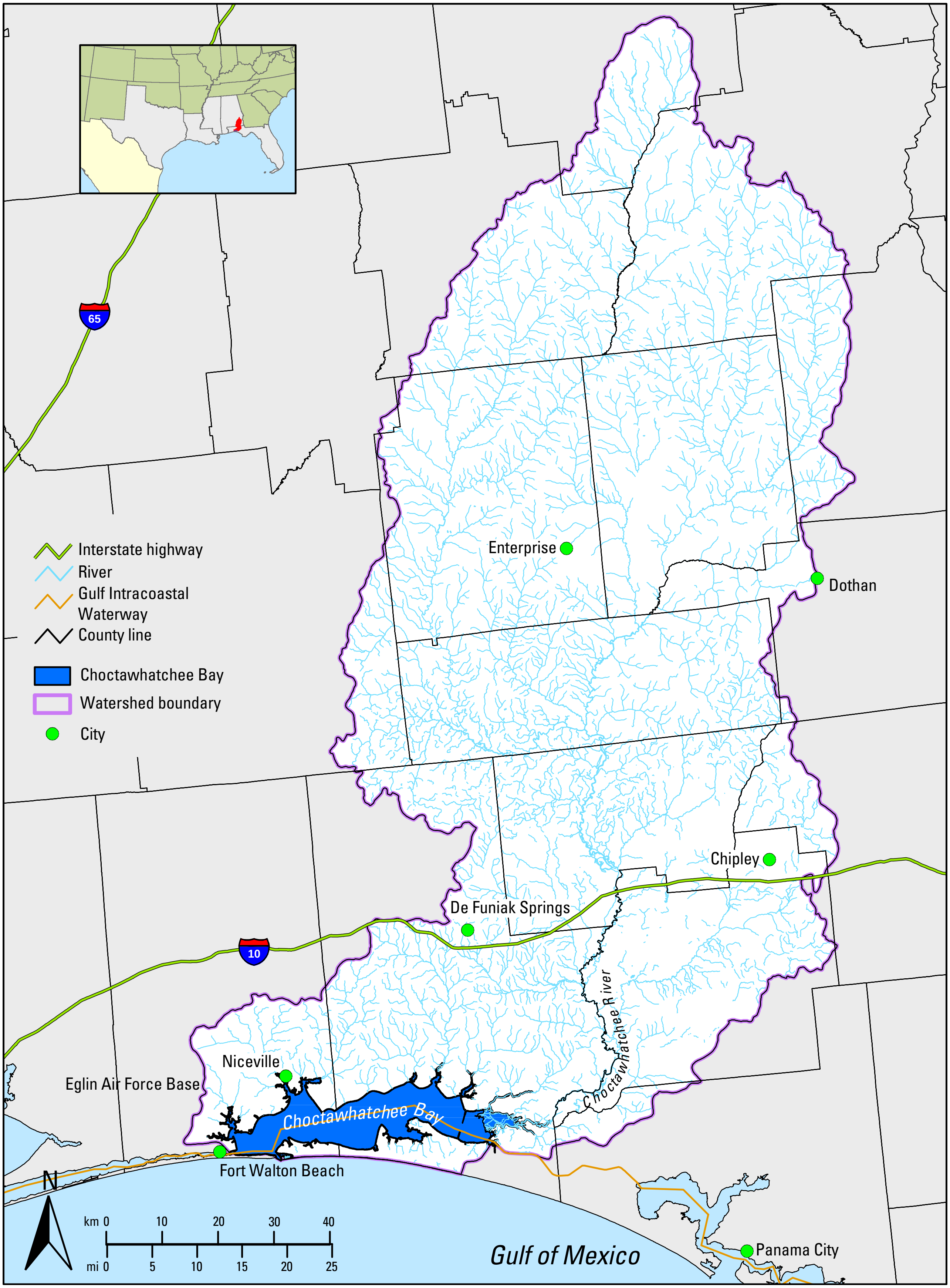

Figure 1. Watershed for Choctawhatchee Bay.

As in much of northwest Florida in the 1990s, the Choctawhatchee Bay watershed is experiencing rapid development, particularly along the eastern and southern shores. Numerous hotels and condominiums are being constructed along the perimeter of the bay. The human population of the watershed is growing rapidly, with that of the Florida counties increasing 41% from 1980 to 1995 (Northwest Florida Water Management District, 1996). Changes in population recorded between the 1990 and 2000 censuses show an increase of approximately 28% for the Choctawhatchee Bay area compared to 23% for Florida overall and 13% for the United States as a whole. Increased development threatens the water quality of the bay through increases in impervious surfaces that reduce water filtration as well as nutrient and other pollutant additions. Development in the northern portion of the bay is less intensive because Eglin AFB occupies much of the northern drainage area.

Baseline studies conducted in 1980 identified important resources and productive areas along with potential problem areas. Potential human impacts in problem areas included urban development, stormwater runoff, causeway construction, marina development, overfishing, riverborne agricultural pesticides and fertilizers, and military activities (Livingston, 1986). Natural events impacting the bay system included major storms, such as hurricanes and tropical storms, as well as drought periods, such as the 3-yr drought that northwest Florida experienced from 1999 through 2001.

There were 13 major floods in the Choctawhatchee River in the 20th century, including 2 in the 1990s (Northwest Florida Water Management District, 1996). The flood of 1929 reportedly created the inlet at East Pass. Local residents dug a trench where an intermittent stream connected with the Gulf of Mexico to help relieve the swelled bay. The U.S. Army Corps of Engineers has maintained the pass since Congress approved maintenance of it in 1930. Tropical Storm Alberto (1994) caused extreme flooding in the Choctawhatchee Bay watershed, providing inputs of large amounts of fresh water, sediments, and nutrients to Choctawhatchee Bay. Hurricane Opal (1995), a category 3 hurricane, came ashore just east of Fort Walton Beach on the western edge of the bay and caused significant damage to the area, including washing large amounts of sand into the southern portion of the bay. In 1999 the area began experiencing a drought that resulted in rainfall amounts 25–38 cm (10–15 inches) below annual averages. Drought conditions continued through summer 2002. The lack of rainfall reduced stormwater runoff, providing less nutrient and sediment loading to the system. Salinity levels increased because less fresh water entered the bay through the Choctawhatchee River and local creeks.

Scope of Area

Choctawhatchee Bay is more than 43 km (27 mi) long and follows an east-west orientation along the upper gulf coast of Florida in Okaloosa and Walton Counties. It varies from 2 to 10 km (1 to 6 mi) in width, with depths ranging from 3 to 13 m (10 to 43 ft). The bay has four basic natural habitats: shallow slope areas (vegetated, unvegetated, and oyster beds), deep central basin regions (unvegetated), bayous, and a river delta area, with East Pass connecting the bay to the Gulf of Mexico. Tidal exchange in the bay is minimal (a range of about 0.15 m, or 0.5 ft), and flushing is limited to the narrow, shallow opening to the Gulf of Mexico at East Pass.

The bay receives most of its freshwater inflow from Choctawhatchee River, which receives flow from several tributaries including Pea River, Wrights Creek, Sandy Creek, Pine Log Creek, Seven Runs, Holmes Creek, and Bruce Creek. Several creeks feed the various bayous surrounding the bay, including Turkey Creek, Rocky Creek, Swift Creek, and Alaqua Creek. Freshwater sources also include groundwater contributions from springs and the Floridan aquifer. The Choctawhatchee Bay watershed has the highest elevations in Florida. Soft, sandy soil, intense rainfall, and steep topographical relief make this area highly susceptible to erosion, particularly after removal of vegetative cover, during road construction, and with agricultural activities. Human impacts, in concert with natural forces, can affect the health and distribution of seagrass beds.

The area covered in this report extends from the delta area in the east where the Choctawhatchee River enters the bay to the Brooks Bridge on U.S. Highway 98 in Fort Walton Beach in the west and includes the several bayous around the bay. For purposes of describing seagrass status in this report, Choctawhatchee Bay was divided into three segments: western, middle, and eastern (fig. 2).

Western Choctawhatchee Bay

Western Choctawhatchee Bay includes all areas west of the Hwy 293 Mid-Bay Bridge to Brooks Bridge (where Santa Rosa Sound begins), which is the more saline portion of the bay and includes deeper sections (maximum depth is 13 m, or 43 ft), and historically has been the most developed area. The towns of Fort Walton Beach, Niceville, Valparaiso, and Destin as well as the main activities area of Eglin AFB (runways, support facilities, etc.) surround this area. The western segment includes Cinco Bayou, Garnier Creek, and Weekley Bayou in the Fort Walton Beach area. In the Niceville

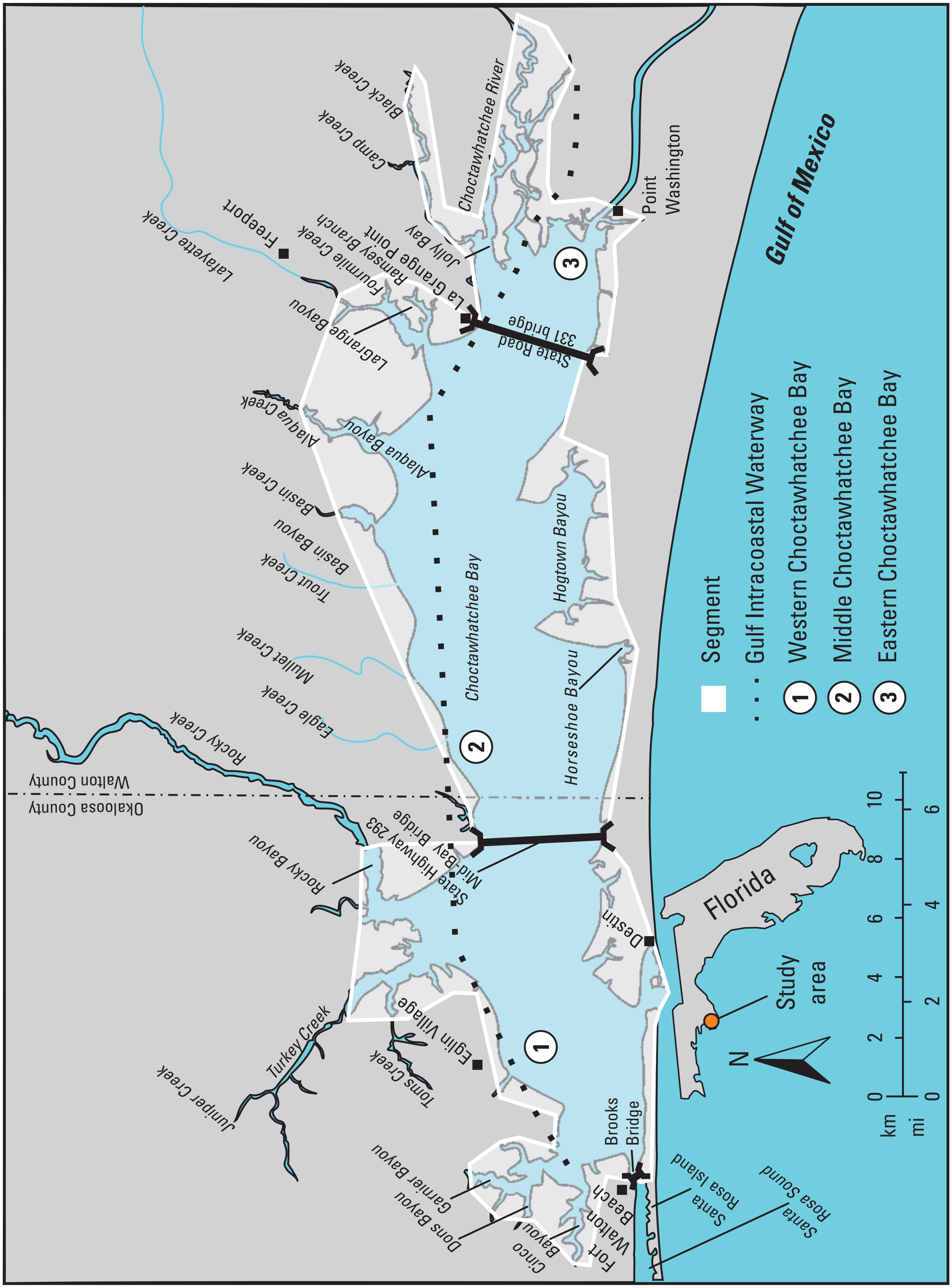

Figure 2. Scope of area for the Choctawhatchee Bay vignette.

and Valparaiso area, Turkey and Juniper Creeks enter Boggy Bayou, Toms Creek flows to Toms Bayou, and Rocky Creek enters Rocky Bayou, much of which is surrounded by a State park, the Rocky Bayou State Park Aquatic Preserve. These waterways all drain portions of Eglin AFB property through urbanized areas. The western section includes East Pass (with its limited tidal exchange), the connection to Santa Rosa Sound, and the western continuation of the GIWW.

Middle Choctawhatchee Bay

Middle Choctawhatchee Bay includes all areas of the bay between the State Road (SR) 331 bridge and the Mid-Bay Bridge. Much of the northern shore consists of Eglin AFB property. Mullet Creek, Trout Creek, the portion of Basin Creek to Basin Bayou, and the portion of Alaqua Creek to Alaqua Bayou, plus several smaller creeks, drain much of the Eglin Wildlife Management Area. East Branch Fourmile Creek and Lafayette Creek drain the Freeport area through LaGrange Bayou to the eastern portion of the middle bay. The southern shore includes Horseshoe and Hogtown Bayous and the resort area of Sandestin. Most of this section of the bay is classified as Class II, Shellfish Propagation or Harvesting, and contains the majority of the oyster resources in the bay.

Eastern Choctawhatchee Bay

Eastern Choctawhatchee Bay includes all areas of the bay east of the SR 331 bridge, which includes the Choctawhatchee River Delta area, the primary source of fresh water to the bay. The Choctawhatchee River is considered the fourth largest river in the State of Florida (Fernald and Purdum, 1998). River discharge dominates the circulation pattern of the bay (Northwest Florida Water Management District, 1996). The river provides approximately 90% of the total supply of fresh water to the bay system, and its inflows exert the greatest influence on average bay salinity (Eglin Air Force Base, 1996). In addition to the Florida ERC's designation of the Choctawhatchee River as an Outstanding Florida Water, the eastern bay waters are classified as Class II, Shellfish Propagation or Harvesting. Water depth averages 3 m (10 ft) in the eastern portion of the bay. The GIWW for this portion of the bay continues to the east out of Choctawhatchee Bay, just south of the delta near Point Washington.

Methodology Employed To Determine and Document Current Status

The most current mapping study of seagrass coverage for the Choctawhatchee Bay area was conducted more than 10 yr ago by the U.S. Geological Survey (USGS) National Wetlands Research Center (NWRC) by using natural-color aerial photography taken in 1992 at a 1:24,000 scale as part of the northeastern Gulf of Mexico seagrass mapping project.

The mapping protocol consisted of stereoscopic photointerpretation, cartographic transfer, and digitization in accordance with strict mapping standards and conventions. Other important aspects of the protocol include the development of a classification system, groundtruthing, quality control, and peer review.

The information derived from the photography was subsequently transferred by using a zoom transfer scope onto a stable medium overlaying USGS 1:24,000-scale quadrangle base maps. The primary data source was 1:24,000-scale natural color aerial photography flown by the National Aeronautics and Space Administration (NASA) Stennis Space Center in fall 1992. In those cases where the data were inadequate or incomplete, contemporary supplemental data were acquired from other sources and used to complete the photographic coverage.

The seagrass classification system that was developed consisted of two classes of open water—RIV (riverine, fresh water) and EST (estuarine or marine open water)—and five classes of seagrass habitats. One seagrass habitat class is continuous seagrass, CSG, for which no density distinction was made; the other four classes of patchy seagrass are based on percent ground cover of the patches in 5% increments: PSG1 (0%–10%, very sparse), PSG2 (15%–40%, sparse), PSG3 (45%–70%, moderate), and PSG4 (75%–95%, dense).

The groundtruthing phase included the participation of field staff from the Gulf Islands National Seashore, U.S. Fish and Wildlife Service, Dauphin Island Sea Lab, Mississippi State University, Alabama Department of Conservation and Natural Resources, and Florida Department of Environmental Protection. Draft maps were sent out to these agencies for review and comments. All comments received were incorporated into the final maps.

Methodology Employed To Analyze Historical Trends

Studies conducted in Choctawhatchee Bay to determine the status and trends of seagrass resources since 1949 have been limited and mostly were either part of a larger, general study (McNulty and others, 1972) or site-specific projects (U.S. Army Corps of Engineers, 1973, 1976) with limited documentation of methodologies employed.

Burch (1983) reviewed the earlier studies and different types of aerial photography furnished by the Florida Department of Transportation. Burch noted that the photography, besides being at different scales, did not cover the entire bay at any given time. Color aerial photography at a 1:24,000 scale from 1980 was used to delineate areas in the western portion of the bay, while photography taken from the NASA ER-2 aircraft at a 1:65,000 scale from 1979 was used for the remainder of the bay. Black and white aerial photography (1:20,000 scale from 1949 and 1955 and 1:40,000 scale from 1969) was used for historical comparisons. Field verification was conducted with transect sampling by using an Ekman dredge (Wildco®, Buffalo, N.Y.) and a "weed rake" similar to that used by the Florida Department of Natural Resources (now the Florida Department of Environmental Protection). North-south oriented transects 0.5 m (1.6 ft) apart were sampled from April to September 1982. Military restrictions limited field verifications in Rocky Bayou and from Boggy Bayou to White Point.

Status and Trends

Although the seagrass beds in Choctawhatchee Bay are considered extremely important, only limited studies have been conducted in Choctawhatchee Bay to determine the status and trends of those seagrass resources, and these studies have used different methods that make direct comparisons to current data difficult. Furthermore, these studies placed most of the seagrass beds in Choctawhatchee Bay in shallow nearshore areas, but seagrass has been found in water depths up to approximately 3 m (10 ft).

Burch (1983) summarized earlier studies, including those of McNulty and others (1972) and the U.S. Army Corps of Engineers (USACE) (1973, 1976). McNulty and others (1972) reported 1,237 ha (3,057 acres) of submerged aquatic vegetation in Choctawhatchee Bay as part of a larger study that covered the entire Gulf of Mexico coast (fig. 3). Mapping was done with limited field verification, and species were not identified. The USACE studies were limited to specific projects in the eastern section of the bay.

Interpretation of aerial photographs by Burch (1983) indicated significant loss of seagrass beds for most areas in the bay between 1949 and 1985. Burch determined that the submerged aquatic vegetation was limited to two species, shoal grass (*Halodule wrightii*) and wigeon grass (*Ruppia maritima*). The author concluded that the bay has supported varying amounts of submerged aquatic vegetation, with declines evident in several locations since 1949, and noted that less decline was evident between 1972 and 1982, with vegetation areas at the time of the study similar in location and configuration to those mapped in 1972.

The USGS conducted an analysis from aerial photographs taken in 1992, differentiating between continuous or patchy seagrass areas (fig. 4). This analysis provided an estimate of 1,726 ha (4,265 acres) of seagrass present in the bay (table 1). These data show that seagrass communities within Choctawhatchee Bay occurred primarily in higher salinity waters of the western and middle bay (fig. 5).

Western Choctawhatchee Bay

Burch (1983) reported six major areas with significant populations of shoal grass in 1982, five of which were in the western bay area: Moreno Point to Joes Bayou, East Pass, Santa Rosa Sound entrance, Black Point, and White Point. The coverage of shoal grass ranged from 20% to 80%, but most areas had more than 40% coverage. Seagrass beds with less than 40% coverage were reported northwest of Destin, Smack Point, and Eglin Village.

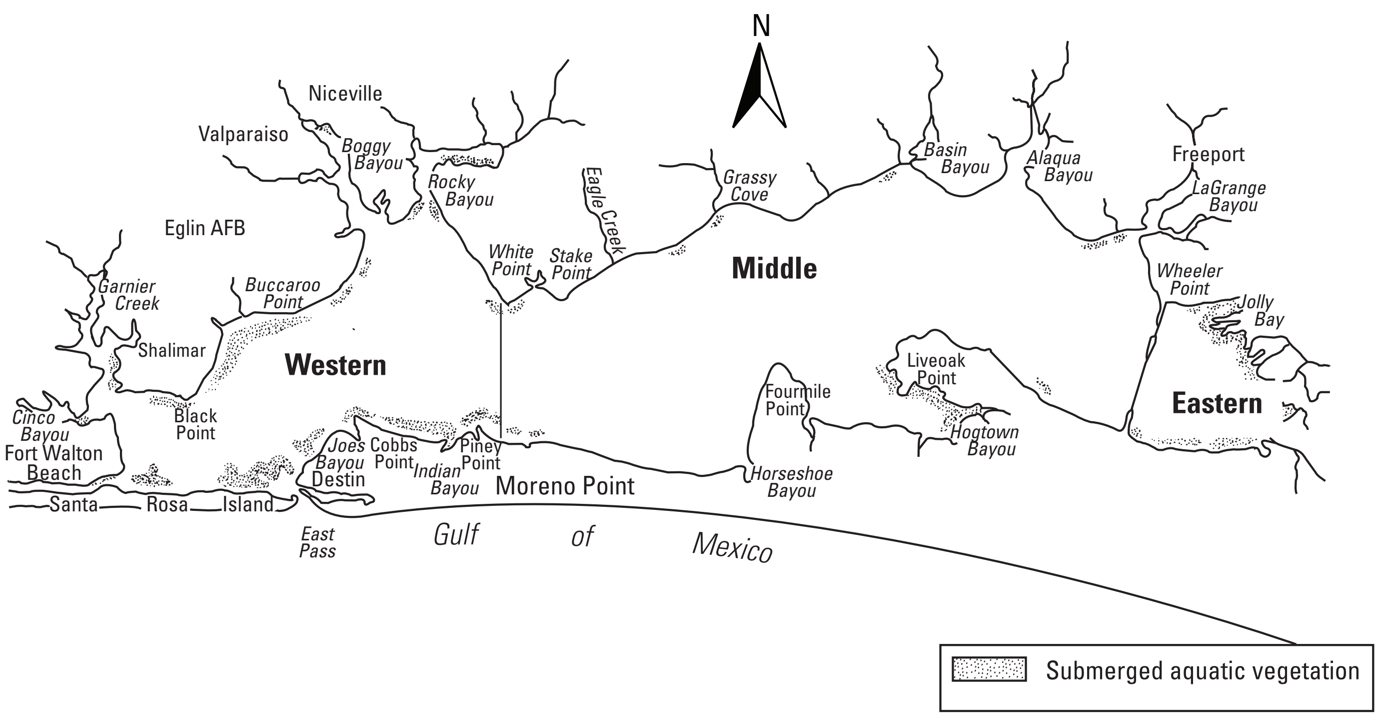

Figure 3. Distribution of submerged aquatic vegetation in Choctawhatchee Bay, 1972.

The USGS data show that most seagrass areas contained patchy beds (1,428 ha, or 3,529 acres) and were located primarily in the western bay, where 1,109 ha (2,740 acres) of patchy beds were delineated (table 1). Only 317 ha (783 acres) of continuous seagrass beds were present (table 1). The continuous seagrass beds were mainly located north of Santa Rosa Island, west of East Pass, in the area bordered on the south by the national seashore (fig. 4). The USGS analysis showed seagrass beds in areas similar to the 1972 data; however, the 1992 set showed patchy seagrass beds along the eastern shore of Fort Walton Beach, while the 1972 delineation does not indicate any seagrass in this area.

An extensive shoal grass bed that ranged in depth from less than 1 m (3 ft) to at least 3 m (10 ft) and extended offshore for more than 274 m (900 ft) was observed in the White Point area in 1991 (Donald Ray, Florida Department of Environmental Protection, written commun., 1991).

Livingston (1986) estimated 307 ha (759 acres) in the western portion of the bay and reported a loss of nearly 20% between 1955 and 1985.

Middle Choctawhatchee Bay

Hogtown Bayou in the south-central part of the bay was one of the major areas with significant populations of shoal grass in 1982 (Burch, 1983). Livingston (1986) reported wigeon grass mixed with shoal grass in Hogtown Bayou. Seagrass beds with less than 40% coverage were reported near Fourmile Point, east of the Okaloosa-Walton County line (Burch, 1983). Seagrass beds appeared to be more extensive in Hogtown Bayou in 1972 than in the later analysis in 1992 (figs. 3 and 4).

Table 1. Choctawhatchee Bay seagrass status, 1992.

[Data source: U.S. Geological Survey photointerpretation of 1:24,000-scale natural color aerial photography flown by Stennis Space Center (National Aeronautics and Space Administration) in fall 1992]

Segment	Classification	Hectares	Acres
Western	Continuous	317	783
	Patchy	1,109	2,739
Middle	Continuous	0.4	1
	Patchy	297	733
Eastern	Patchy	2	5
Total	Continuous	314	784
	Patchy	1,408	3,477

Eastern Choctawhatchee Bay

The USACE described wigeon grass occurring in the shallow, muddy areas of the bay at the LaGrange Bayou entrance in a 1973 environmental impacts study (U.S. Army Corps of Engineers, 1973). In a 1976 environmental impacts study, the USACE noted submerged grasses present near the mouth of the Choctawhatchee River in Bunker Cove and west of Point Washington but did not name which species were present (U.S. Army Corps of Engineers, 1976). Burch's (1983) analysis of the 1972 aerial photography indicated that seagrass beds appeared to be more extensive in the eastern section of the bay than the 1992 data show (figs. 3 and 4).

Causes of Change

Because the Choctawhatchee Bay system consists of three segments with diverse characteristics, only broad observations regarding change are possible. In addition, there is a lack of comparable historical information, as the available seagrass data were derived by different methods and for different areas within the bay. Because freshwater inflow and poor flushing heavily influence Choctawhatchee Bay, the bay exhibits extremes of fresh water, which are marine conditions that may contribute naturally to declines and expansions of seagrass beds. While it appears that the bay has supported varying amounts of seagrass in the last 50 yr, seagrass areas have declined in several locations since the 1950s.

The health of the bay and its seagrass beds is being threatened by increased development in the Choctawhatchee Bay watershed. Increasing numbers of people desire to be on or near the water, whether for living, recreation, or both. As environmentally attractive and sensitive lands are more heavily used, stress on the resources, including sensitive habitats such as seagrass beds, is increasing. This increased development also threatens the water quality of the bay through nutrient and other pollutant additions. The National Oceanic and Atmospheric Administration has described the bay as exhibiting strong symptoms of eutrophication, with seagrass loss being one of the secondary symptoms indicative of eutrophication (Bricker and others, 1999). Seagrass scarring has also been indicated as a problem in the bay (Sargent and others, 1995).

Western Choctawhatchee Bay

Older developed areas (Fort Walton Beach, Valparaiso, Niceville, and Eglin AFB main base and the earlier Destin development) border western Choctawhatchee Bay. During the 1970s, this area displayed eutrophication, extensive plankton "blooms," seagrass bed die-off, and fish kills (Young and Butts, 1984). A study by Young and Butts (1984) attributed much of the earlier pollution to sewage treatment plants in the peripheral bayous and eastern Santa Rosa Sound but

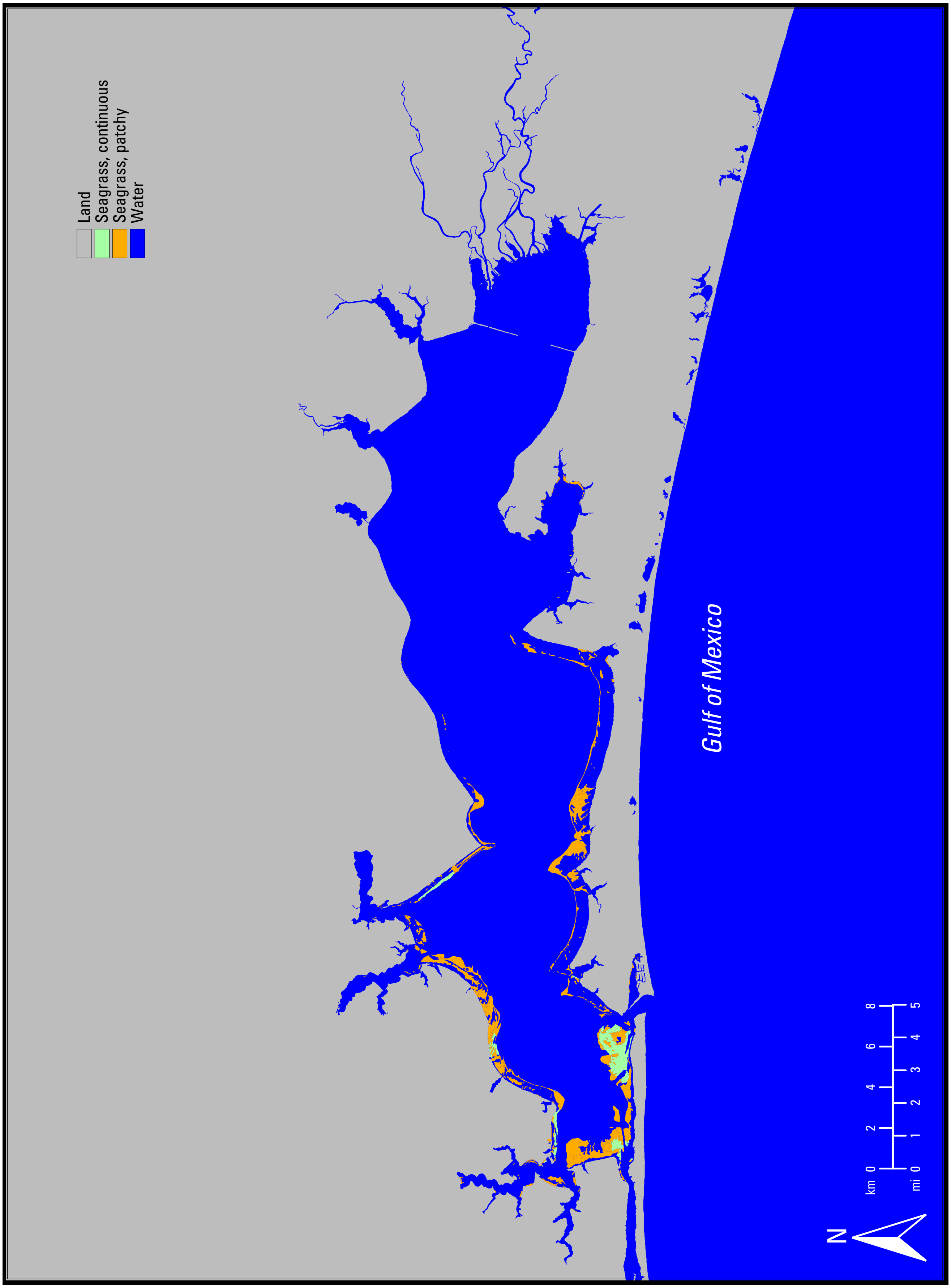

Figure 4. Distribution of seagrass in Choctawhatchee Bay, 1992.

noted that discharges had been converted to upland disposal, which led to improved water quality, expanded shoal grass beds, and increased macroinvertebrate species richness in the western bay at that time. Livingston (1986) suggested that spatial and temporal patterns indicated that losses were probably caused by changes in optical characteristics of the water column. Livingston further suggested that nutrient enrichment, phytoplankton, and coliform values indicated cultural eutrophication of western Choctawhatchee Bay, which was impacting seagrass resources through loss of light. He suggested that other cultural perturbations that could impact seagrasses included dredging and related sediment suspension and deposition, operation of recreational and commercial watercraft, and sedimentation from construction and other land-based activities. Florida Marine Research Institute data (Sargent and others, 1995) indicated that propeller scarring, while it did not impact a large portion of the bay (0.2% scarring), was evident along the northern shore of Santa Rosa Island and north of Brooks Bridge.

Middle Choctawhatchee Bay

Although development in the northern portion of middle Choctawhatchee Bay is minimal because Eglin AFB occupies most of the northern drainage area, the southern shores are experiencing rapid development. Numerous hotels, condominiums, and residential developments, as well as expanding commercial areas, are being constructed along the perimeter of the bay, changing the land use from rural to urban or suburban in character (Northwest Florida Water Management District, 2002). The Florida Department of Environmental Protection (FDEP) Northwest District Submerged Lands and Environmental Resources Program has seen a 58% increase over the last 5 yr in regulatory permitting actions in Walton County, which includes approximately two-thirds of Choctawhatchee Bay and much of the Florida portion of the watershed (fig. 2). These include permits for dredging and filling activities, as well as exemptions for private docks. While the population increased in Walton County by 46% over the last 10 yr, the higher rise in permitting actions over population increase indicates a greater pressure on wetland and water resources. As the population swells along the bay coastline, additional docks and seawalls are changing the hydrological patterns in the bay. These changes can lead to erosion in seagrass beds and to sedimentation, which causes increased turbidity, reducing light available to the plants and thus restricting their growth or smothering them. Increased population also leads to increased nutrients through additional use of fertilizer on lawns and a higher concentration of onsite sewage treatment systems (septic tanks), which can lead to algal blooms and epiphytic algae that also restrict light availability by attaching to plant blades. Hogtown Bayou is an area of concern; it is served by septic systems and has shown increasing nutrient trends, as has much of the Freeport area north of LaGrange Bayou (Ross Hamilton, Choctawhatchee Basin Alliance, oral commun., 2002).

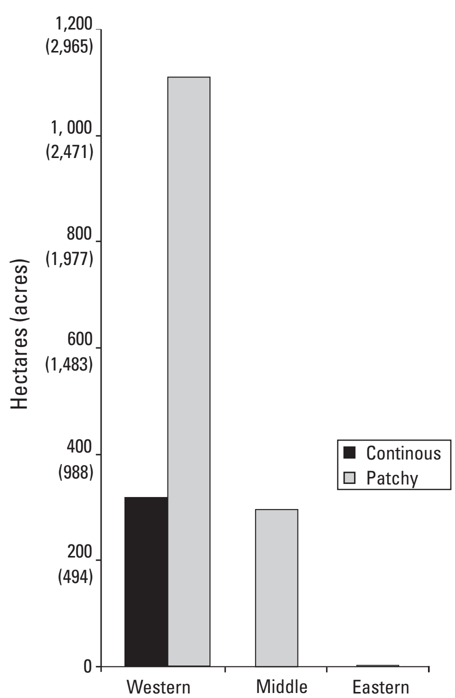

Figure 5. Choctawhatchee Bay seagrass coverage by area, 1992.

Eastern Choctawhatchee Bay

Early studies indicated that seagrasses were once present in the eastern portion of the bay in the Choctawhatchee River Delta area (Burch, 1983) (fig. 3). The current dataset (1992) does not show any seagrass populations in that area (fig. 4). There has been only limited development along the eastern end of the bay, although that appears to be changing as population is increasing on both the north and south ends of the State Road 331 bridge, which delineates the western edge of this section of the bay. In addition, the delta area is subject to sedimentation and nutrient loads from the Choctawhatchee River. Anecdotal evidence from local residents indicates that sedimentation has become a problem at the mouth of the Choctawhatchee River.

Overall, the information available on seagrass status and trends for the Choctawhatchee Bay is limited. While earlier studies have indicated reductions in seagrass populations and extent, no analysis of comparable datasets has been done.

Species Information

The dominant species of seagrass is shoal grass, with some wigeon grass found occasionally.

Western Choctawhatchee Bay

Studies by Livingston (1986) identified the presence of shoal grass beds in the higher salinity, western portion of the bay in the vicinity of Moreno Point, Joes Bayou, East Pass, the Santa Rosa Sound entrance, and Black and White Points.

Middle and Eastern Choctawhatchee Bay

The USACE observed wigeon grass at the entrance to LaGrange Bayou in eastern/middle Choctawhatchee Bay (U.S. Army Corps of Engineers, 1973). Livingston (1986) also described wigeon grass but only at Hogtown Bayou, east of the Okaloosa-Walton County line, in the middle bay segment.

Monitoring for Seagrass Health

There is no current, ongoing seagrass monitoring in the Choctawhatchee Bay system.

Mapping and Monitoring Needs

The need to quantitatively and qualitatively assess the status of seagrass beds has been identified in several plans (Burch, 1983; Young and Butts, 1984; Eglin Air Force Base, 1996; Northwest Florida Water Management District, 1996, 2002; Choctawhatchee Basin Alliance and Florida Department of Environmental Protection, 1998). Additional mapping and monitoring should be undertaken in Choctawhatchee Bay to provide more information on current status and changes compared to the previous mapping efforts in 1992. These efforts should include evaluating and digitizing the earlier aerial photographs used for comparison by Burch (1983). Standardized mapping of seagrass beds at appropriate temporal and spatial scales is needed every 5 to 10 yr. Monitoring the health of seagrass beds and the groundtruthing of digitized mapping should also be undertaken.

The Choctawhatchee River and Bay System Surface Water Improvement and Management (SWIM) Plan (Northwest Florida Water Management District, 2002) proposed a seagrass assessment as a priority activity for fiscal years 2002–03 and 2003–04. According to the plan, components of the project should include evaluating conditions of submerged aquatic vegetation, comparing data with earlier studies to identify trends that could be potential causes of adverse effects, and relating to nonpoint source projects to develop and pursue recommendations for protecting and restoring seagrass communities in the bay.

Restoration and Enhancement Opportunities

Eglin AFB has influenced the health of the Choctawhatchee Bay by preserving the environmentally important features of a federally owned land with nationally recognized research, preservation, and restoration projects (Eglin Air Force Base, 1996). Although the projects are not directly focused on seagrass communities, many of the projects address potential nonpoint source contributions to the bay that affect seagrass health. Projects include restoring shoreline vegetation and identifying and restoring stream segments impacted by erosion and sedimentation that ultimately enters the bay (Donald Ray, Florida Department of Environmental Protection, written commun., 1991).

The Choctawhatchee Basin Alliance, FDEP, and the Northwest Florida Water Management District (NWFWMD), along with local citizen volunteers, have undertaken several shoreline restoration activities. These activities include habitat restoration and enhancement on the Liveoak Point Peninsula and shoreline restoration in Valparaiso, on the southern shoreline of Choctawhatchee Bay just east of Fort Walton Beach along U.S. Highway 98 on Okaloosa Island, and near the 4-H Camp Timpoochee in Niceville (Choctawhatchee Basin Alliance, 2001; Northwest Florida Water Management District, 2002). The Valparaiso restoration included planting of wigeon grass at four transects in Toms Bayou, although survival was limited to only one transect because of foot traffic and boating activities (Florida Department of Environmental Protection, 2001). The report concluded that with proper protective measures, however, the potential to establish submerged aquatic vegetation in Toms Bayou does exist. As part of the restoration project, a shoreline protection and restoration brochure for waterfront owners was developed and distributed (Northwest Florida Water Management District, n.d.). These projects are restoring shoreline communities that were historically more prevalent in Choctawhatchee Bay, educating the local citizens about the importance of aquatic and emergent vegetation, and helping protect bay resources from pollutant runoff and erosion that can adversely affect seagrass resources.

The NWFWMD has recently acquired 130 ha (321 acres) of wetlands bordering Hogtown Bayou that will provide restoration opportunities and should help protect the remaining seagrass beds from further decline (Northwest Florida Water Management District, 2002). Additional land acquisition and conservation easements would help protect against nonpoint source impacts on seagrass communities. The NWFWMD has identified the need to evaluate the status of seagrass communities and relate nonpoint source pollution and ecological restoration projects in order to develop and pursue recommendations for protecting and restoring seagrass communities in the bay.

Many activities that have the potential to affect water quality and environmental resources, such as seagrass, require review and permitting through the FDEP. While review and permitting actions are designed to protect the State's resources, the policy of the FDEP is to grant the highest protection to Outstanding Florida Waters. In the Choctawhatchee Bay system these Outstanding Florida Waters include the easternmost part of the bay and the river delta, as well as the Rocky Bayou State Park Aquatic Preserve portions of the Choctawhatchee Bay system. This additional protection should help protect existing seagrass resources in these areas; however, impacts from unregulated or nonpoint sources will not necessarily be controlled in these areas any more than in other areas. Therefore, educating local citizens remains an important deterrent to activities that might adversely affect seagrass resources.

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St. Andrew Bay

By Michael S. Brim[1] and Lawrence R. Handley[2]

Background

St. Andrew Bay has a watershed of approximately 297,576 ha (735,300 acres, or 1,149 mi^2) (Beck and others, 2000) (fig. 1). The bay is almost entirely within Bay County, an area of over 2,590 km^2 (1,000 mi^2), which has a resident population of more than 148,000 (U.S. Census Bureau, 2002). Panama City is the largest of seven municipalities surrounding the bay. Much of the county, however, is unincorporated land traditionally supporting silviculture.

The primary industries in Bay County are tourism and the military, with Tyndall Air Force Base playing a dominant role in the community. The U.S. Navy's Coastal Systems Station and the U.S. Coast Guard also share the bay's shoreline. Most tourist activity occurs on Panama City Beach or upon bay waters. Other significant industries include the Smurfit-Stone Container Corporation paper mill, Arizona Chemical, Port Panama City, the Panama City-Bay County International Airport, and the Gulf Power Lansing Smith generation plant.

St. Andrew Bay has a rich history. The bay undoubtedly received its name from some of the early Spanish navigators who explored the northeast gulf coast in the 16th century, between 1516 and 1558 (West, 1922). Archaeological evidence supports that the first Native Americans in the area found food in the bay and shelter along its shores. A skirmish took place within the bay during the Civil War. In 1836, James Watson built a sawmill on what is now Watson Bayou (West, 1922). In 1931, International Paper constructed Florida's first paper mill on the site. The mill (currently owned by Smurfit-Stone Container Corporation) is still in operation. In 1938, the U.S. Army Corps of Engineers (USACE) constructed the main entrance channel by excavating through a barrier peninsula to create a rock-jettied inlet some 9.7 km (6 mi) west of the historical East Pass entrance that was once used by the Spanish explorers. In about 1950, USACE constructed the Gulf Intracoastal Waterway connecting western St. Andrew Bay with Choctawhatchee Bay and connecting eastern St. Andrew Bay with Lake Wimico and with St. Joseph Bay via the Gulf County Canal. In the 1960s, a dam was constructed across a portion of North Bay to create Deer Point Lake.

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[2] U.S. Geological Survey.

St. Andrew Bay is unique because of its wealth of biological diversity. Keppner (2002) documented the diversity associated with the bay and compared it with surveys of Indian River Lagoon, Fla., which has been touted as the most biologically diverse estuary in North America. His report documents 2,913 species of plants and animals associated with St. Andrew Bay, nearly 400 more species than found in the lagoon.

The bay is, however, a fragile ecosystem. Because of its high-salinity waters, the depths of the bay, the lack of significant freshwater inflow from land drainage, and the minimal tidal regime, the bay is highly susceptible to becoming polluted or contaminated. Chemicals and nutrients are not readily flushed from this bay, and the nature of the deep sediments (rich in fine clays, silt, and organic carbon) is such that they could easily become reservoirs for metals and organic-compound contaminants. While several point-source discharges occur in the bay, the greatest future threat to water and sediment quality (and thus to the diverse biota) is urbanization and its associated stormwater runoff.

Additional and more detailed descriptions of St. Andrew Bay and its resources can be found in the ecosystem management plan (Keppner and Keppner, 2001) and the State Surface Water Improvement and Management (SWIM) Plan for the bay (Northwest Florida Water Management District, 2000).

Scope of Area

St. Andrew Bay covers an area of about 27,714 ha (68,480 acres, or 107 mi^2). It is unique among gulf coast estuaries for several reasons. Waters are deep and clear because little fresh water flows into the bay (Saloman and others, 1982), the primary source being Econfina Creek, which has an average discharge of just 15.3 m^3/s (538 ft^3/s) (U.S. Geological Survey, 1990). The total discharge of all natural surface-water sources entering this estuary is probably less than 28.3 m^3/s (1,000 ft^3/s) (Brim, 1998). By comparison, the average flow of the Apalachicola River into Apalachicola Bay to the east is about 707 m^3/s (25,000 ft^3/s). Because of the absence of a large river emptying into St. Andrew Bay, there is little sedimentation and associated turbidity in this bay, a situation contrary to that of most "true" estuaries, which have

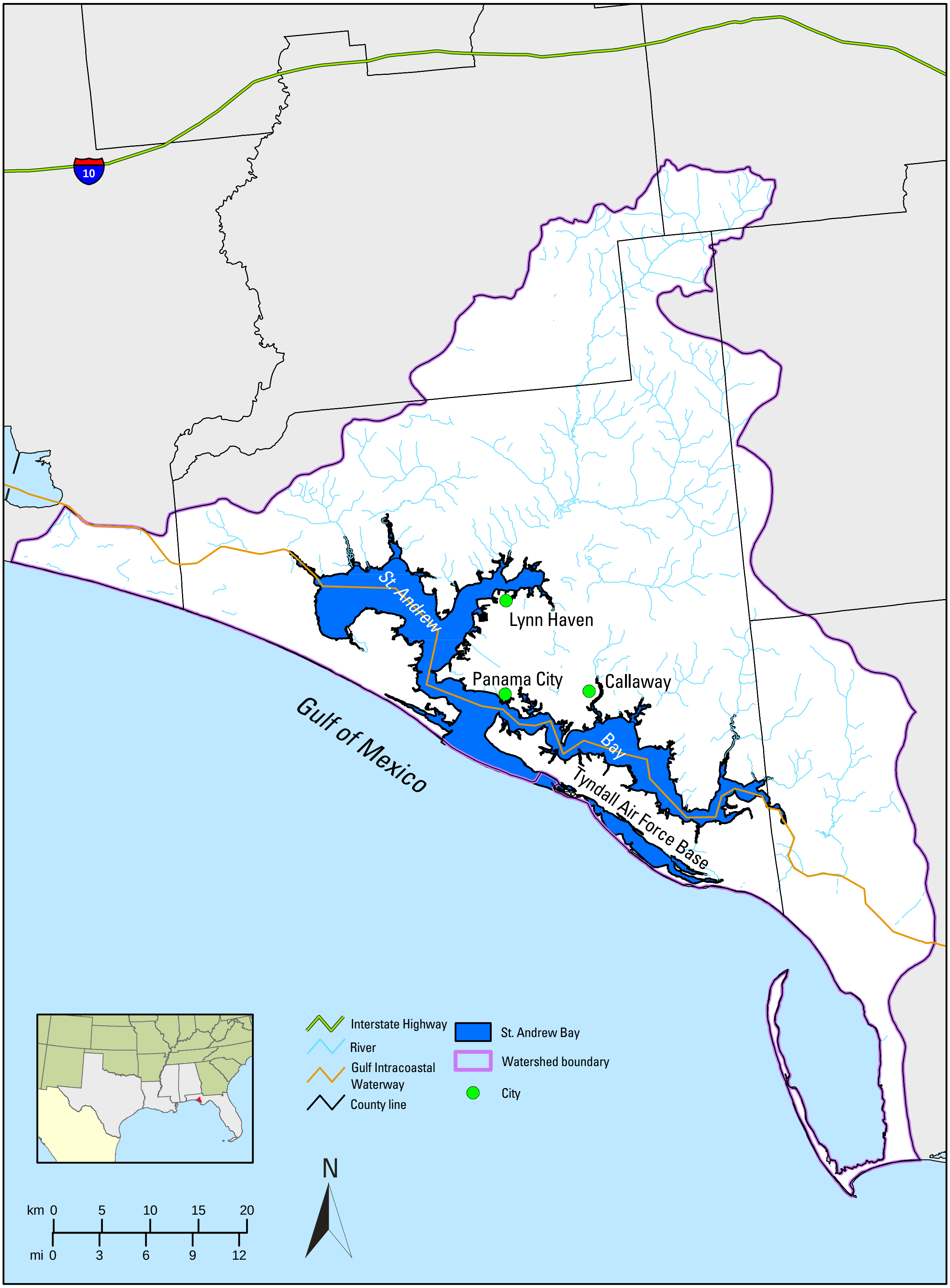

Figure 1. Watershed for St. Andrew Bay.

robust rivers draining into them. Bay depths of 12 m (40 ft) are not uncommon, and seagrasses flourish because of the clear, high-salinity waters. Furthermore, tidal flushing is minimal, with spring tides having a vertical amplitude of only about 0.67 m (2.2 ft) and neap tides often only 0.06 m (0.2 ft).

The St. Andrew Bay system lies in the Coastal Plain physiographic province, has a surface stratigraphy composed largely of post-Pleistocene sands, and is classified as coastal integrated drainage because of the set of small local streams draining its coastal regions (Young and others, 1987; Wolfe and others, 1988; Fernald and Purdum, 1992; Hydroqual, Inc., and Barry A. Vittor and Associates, Inc., 1993). The sediment composition within the bay varies, but several studies have revealed a positive correlation of increased silt and clay content as distance from the inlet increases. Deepwater sampling stations within the bay that were farthest from the jettied inlet, in the vicinity of Dyers Point and Bear Point, had sediments containing 67%–68% fine material (silts and clays) (U.S. Army Corps of Engineers, 1994).

St. Andrew Bay, which surrounds the Panama City metropolitan area on three sides, is about an equal distance (160 km, or 100 mi) from Pensacola to the west and Tallahassee to the east. The study area for this vignette extends from approximately 29^0 57’ to 30^0 18’ N. latitude and from 85^0 24’ to 85^0 51’ W. longitude. The gulf’s coastline at this location is a northwest/southeast axis.

The study area is divided into five logical segments (fig. 2): St. Andrew Bay (specifically the lower bay area); East Bay, North Bay, and West Bay (the three named arms); and St. Andrew Sound (a separate lagoonal embayment to the southeast). Information about the size and volume of these segments is taken from McNulty and others (1972).

St. Andrew Bay Segment

The first segment constitutes that southern portion of the bay between the two major bridges (Hathaway and DuPont) and extending south to the gulf inlets. The bay has two inlets: the recently reopened historical East Pass at the eastern end of Shell Island and the rock-jettied West Pass, which serves as the entrance for commercial vessels. Shell Island is an 11-km (7-mi) barrier island separating the lower bay from the gulf. The St. Andrew Bay segment covers 10,607 ha (26,209 acres), with a mean high water (mhw) volume of 5 x 10^8 m^3 (405,512 acre-feet).

West Bay Segment

West Bay is defined as the bay water northwest of an imaginary line connecting West Bay Point with Shell Point. It covers 7,113 ha (17,576 acres), with a mhw volume of 1.7 x 10^8 m^3 (136,135 acre-feet). Freshwater inflow comes primarily from two small creeks: Burnt Mill and Crooked Creeks. The Gulf Intracoastal Waterway enters West Bay on its west side. Botheration Bayou enters the bay at its southeast end. The municipally treated effluent discharge from the City of Panama City Beach enters West Bay on the southern shoreline.

North Bay Segment

North Bay is east of West Bay and north of the Hathaway Bridge. It covers 2,702 ha (6,676 acres) and has a mhw volume of approximately 68 million m^3 (55,189 acre-feet). It is primarily characterized by the presence of Deer Point Lake at its northeast end. The lake, a reservoir constructed in 1961, was once the most estuarine portion of St. Andrew Bay, the point at which Econfina Creek flowed into the system. Deer Point Dam impounded approximately 2,226 ha (5,500 acres) of estuarine area (Barkuloo, 1963); eventually the reservoir became a freshwater lake, which now serves as the potable water source for Bay County and its municipalities. North Bay’s east shoreline is highly urbanized and is also the location of the Panama City-Bay County International Airport. Sixteen bayous are found along the North Bay shoreline.

East Bay Segment

East Bay extends southeast of the DuPont Bridge. It covers 7,551 ha (18,659 acres) and has a mhw volume of 2.9 x 10^8 m^3 (231,705 acre-feet). East Bay is characterized by the presence of Tyndall Air Force Base, which includes almost all of the bay’s southern shoreline. Two creeks, Sandy and Wetappo, enter the bay’s eastern end, and 10 bayous are found along the shoreline. The Gulf Intracoastal Waterway enters East Bay along an alignment that cuts through the Wetappo Creek historical pathway. East Bay’s northern shoreline has extensive residential development to the west and moderate or sparse to the east.

St. Andrew Sound Segment

St. Andrew Sound is a lagoon not connected with St. Andrew Bay. The sound is approximately due south of East Bay. This lagoon extends along a northwest/southeast axis, is roughly 16 km (10 mi) long and 1.6 km (1 mi) wide, and covers an area of 1,905 ha (4,707 acres). The area features Tyndall Air Force Base, which covers approximately 95% of the shoreline, including the two barrier peninsulas which separate the sound from the gulf.

Methodology Employed To Determine and Document Current Status

The most current mapping study of seagrass coverage for the St. Andrew Bay area was conducted more than 10 yr ago by the U.S. Geological Survey (USGS) National Wetlands Research Center (NWRC) by using natural-color aerial

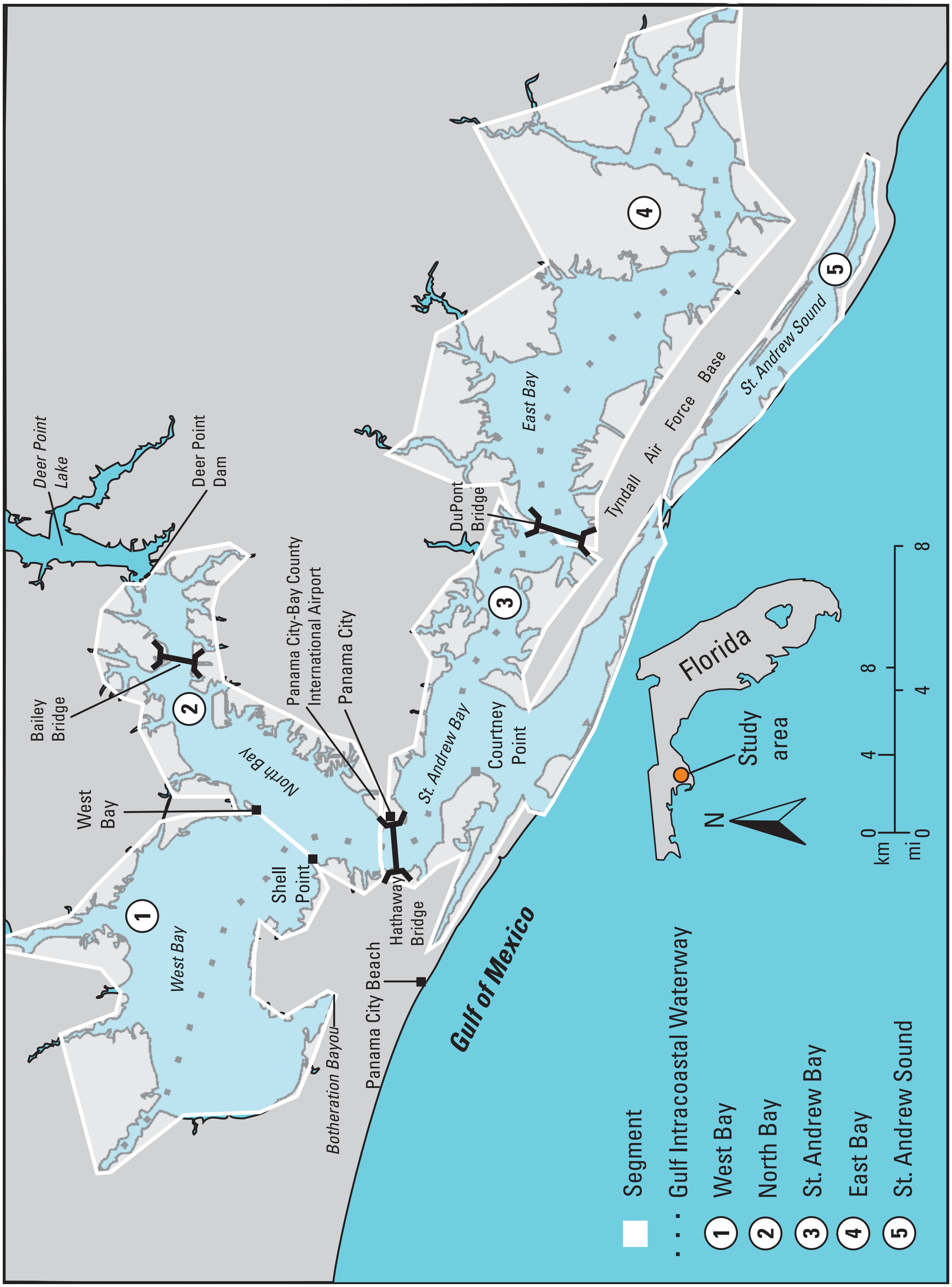

Figure 2. Scope of area for the St. Andrew Bay vignette.

photography taken in 1992 at a 1:24,000 scale as part of the northeastern Gulf of Mexico seagrass mapping project.

The mapping protocol for the project consisted of stereoscopic photointerpretation, cartographic transfer, and digitization in accordance with strict mapping standards and conventions. Other important aspects of the protocol included the development of a classification system, groundtruthing, quality control, and peer review.

The primary data source was 1:24,000-scale natural color aerial photography flown by the National Aeronautics and Space Administration (NASA) Stennis Space Center in fall 1992. In those cases in which the data were inadequate or incomplete, contemporary supplemental data were acquired from other sources and used to complete the photographic coverage. The information derived from the photography was subsequently transferred by using a zoom transfer scope onto a stable medium overlaying USGS 1:24,000-scale quadrangle base maps.

The seagrass classification system that was developed consisted of two classes of open water habitats—RIV (riverine, fresh water) and EST (estuarine or marine open water)—and five classes of seagrass habitats (see appendix 1 for full description). One seagrass habitat class is continuous seagrass, CSG, for which no density distinction was made. The other four classes are patchy seagrass based on percent ground cover of patches in 5% increments: PSG1 (0%–10%, very sparse), PSG2 (15%–40%, sparse), PSG3 (45%–70%, moderate), and PSG4 (75%–95%, dense).

The groundtruthing phase included the participation of field staff from the U.S. Fish and Wildlife Service and the Florida Department of Environmental Protection. Draft maps were sent out to these agencies for review and comments. All comments received were incorporated into the final maps.

Methodology Employed To Analyze Historical Trends

From 1995 through 1998, the NWRC produced a series of historical seagrass maps for the U.S. Fish and Wildlife Service's Panama City Ecological Services and Fisheries Resources Office to use in producing a trend analysis for St. Andrew Bay. Black and white photography from 1953 at a 1:20,000 scale, black and white photography from 1964 at a 1:20,000 scale, and natural color photography from 1980 at a 1:24,000 scale were used to develop the trend analysis. Each group of aerial photography was analyzed with stereoscopic visual equipment; seagrasses were delineated onto Mylar® (DuPont Teijin Films) overlays by using a zoom transfer scope and USGS 1:24,000-scale topographic quadrangles as the base maps. The overlays were digitized by using the Wetland Analytical Mapping System (WAMS) and converted into ArcInfo (Environmental Systems Research Institute, Inc., Redlands, Calif.) format files. All groups of photography were of good to high quality for delineating seagrasses.

Classification of the seagrass for the trend analysis followed the same protocol as the development of the northeastern Gulf of Mexico seagrass mapping using the 1992 aerial photography (see appendix 1). The seagrass classification system consisted of two classes of open water—RIV (riverine, fresh water) and EST (estuarine or marine open water)—and five classes of seagrass habitats. One seagrass habitat class is continuous seagrass, CSG, for which no density distinction was made. The other four classes are patchy seagrass based on percent ground cover of patches in 5% increments: PSG1 (0%–10%, very sparse), PSG2 (15%–40%, sparse), PSG3 (45%–70%, moderate), and PSG4 (75%–95%, dense).

Draft maps were sent out to the U.S. Fish and Wildlife Service for review and comments. All comments received were incorporated into the final maps. Although there was some limited groundtruthing in 1982–83 by NWRC and the U.S. Environmental Protection Agency (EPA) of the 1980 aerial photography, no groundtruthing took place for historical seagrass delineation.

The NWRC currently holds the aerial photography, the interpreted overlays, and the Mylar® overlays for the 1953, 1964, and 1980 maps.

Status and Trends

Status and trends for the study area and segments will be described by using just two density classifications, continuous and patchy, except where specific conditions warrant a more detailed description.

Entire Study Area

As of 1992, the study area contained 1,710 ha (4,225 acres) of continuous beds and 2,269 ha (5,607 acres) of patchy beds (table 1). Table 1 reveals a trend which ends with a total loss (of both types of coverage) amounting to 816 ha (2,011 acres) between 1953 and 1992, an 8% decrease. Furthermore, between 1980 and 1992, the system experienced a 592-ha (1,463-acre) loss of continuous coverage, while patchy coverage increased by 448 ha (1,107 acres). Both the areal extent and the robustness of the seagrasses in the study area have declined significantly.

St. Andrew Bay Segment

The St. Andrew Bay segment comprises 10,607 surface hectares (26,210 acres). Table 2 reveals the same general trend of seagrass areal loss and robustness in the St. Andrew Bay segment as that which is typical for the overall study area.

While total seagrass coverage is almost the same for 1953 and 1992, there was a loss of about 102 ha (250 acres) of total coverage between 1964 and 1992. Also, the decline

Table 1. Seagrass coverage for the entire St. Andrew Bay study area for 4 individual years of evaluation.

[Values are in hectares (acres)]

Coverage	1953	1964	1980	1992
Continuous	1,527 (3,773)	2,183 (5,394)	2,302 (5,688)	1,710 (4,225)
Patchy	3,268 (8,075)	2,523 (6,234)	1,821 (4,500)	2,269 (5,607)
Total	4,795 (11,848)	4,706 (11,629)	4,123 (10,190)	3,979 (9,832)

Table 2. Seagrass coverage for the St. Andrew Bay segment for 4 individual years of evaluation.

[Values are in hectares (acres)]

Coverage	1953	1964	1980	1992
Continuous	491 (1,213)	567 (1,401)	693 (1,712)	536 (1,324)
Patchy	543 (1,342)	580 (1,433)	415 (1,025)	509 (1,258)
Total	1,034 (2,555)	1,147 (2,834)	1,108 (2,738)	1,045 (2,582)

Table 3. Seagrass coverage for the East Bay segment for 4 individual years of evaluation.

[Values are in hectares (acres)]

Coverage	1953	1964	1980	1992
Continuous	263 (650)	643 (1,589)	510 (1,260)	660 (1,631)
Patchy	778 (1,922)	504 (1,245)	403 (996)	360 (890)
Total	1,041 (2,572)	1,147 (2,834)	913 (2,256)	1,020 (2,520)

in seagrass robustness between 1980 and 1992 amounted to a loss of nearly 157 ha (400 acres) of continuous beds. In that same period, a total loss of 63 ha (157 acres) of any type of coverage occurred.

Specifically, conditions appear to have improved in the beds behind Shell Island but have declined along the west shoreline south of the Hathaway Bridge, from the bridge to Courtney Point, and also along the shoreline from Redfish Point to Davis Point. Most of this change appears to have occurred between 1980 and 1992.

East Bay Segment

The East Bay segment comprises 7,551 surface hectares (18,659 acres). This segment appears to have experienced an increase in overall bed robustness from 263 ha (650 acres) of continuous-type coverage in 1953 to 660 ha (1,631 acres) in 1992 (table 3). In that same period, the total coverage of seagrass (both coverage types) has remained nearly unchanged.

North Bay Segment

The North Bay segment comprises 2,702 surface hectares (6,677 acres). The seagrasses in this segment have declined in total coverage from 1953 to 1992 by 55 ha (134 acres) (table 4). This decline appears to be due to a shift in robustness, with continuous beds declining from 455 ha (1,123 acres) in 1953 to 400 ha (988 acres) by 1992. Patchy coverage, at 355 ha (877 acres), is identical for those same two years. Most of the decline was lowest in 1980, with some recovery by 1992.

West Bay Segment

The West Bay segment comprises 7,113 surface hectares (17,576 acres). This segment has suffered the greatest seagrass loss in the St. Andrew Bay system (table 5). Continuous beds, which peaked in 1980 at 543 ha (1,343 acres), dropped to just 92 ha (227 acres) by 1992, an 83% decline in 12 yr. Patchy beds, which peaked in 1953 at 1,229 ha (3,037 acres), steadily declined to a 1992 total of 698 ha (1,725 acres), a 43% decline. Total loss of seagrass coverage (both categories) occurred in a steady downward trend, with a net loss of seagrass coverage between 1953 and 1992 of 750 ha (1,853 acres).

St. Andrew Sound Segment

The St. Andrew Sound segment comprises 1,905 surface hectares (4,707 acres). Seagrass coverage in this segment has not varied much over the 39-yr study span (table 6). Total coverage has ranged from a minimum of 336 ha (832 acres) to a maximum 370 ha (914 acres). The most notable shift appears to be a peak in seagrass robustness, with a tripling of the occurrence of continuous beds from 1964 to 1980. By

1992, however, continuous beds had dropped back to the 1964 quantity. The shifts in robustness appear to be minor changes from continuous to dense patchy (75%–95% coverage) and back again (figs. 3, 4, 5, and 6).

Causes of Change

Entire Study Area

Because of the varied nature of the St. Andrew Bay complex, being almost five independent bay areas, it is difficult to make anything but general observations on changes or losses for the entire study area. Each segment has experienced different types and degrees of development on the adjacent uplands. Thus seagrass impacts related to these land uses are best described on a segment basis. There are, however, stressors which are of general concern, including hurricanes and tropical storms, atmospheric deposition of nutrients (not as yet quantified), ubiquitous stormwater runoff (but of varied composition), propeller scarring (primarily from recreational boats), and algal blooms, light attenuation, and turbidity caused by multiple factors.

The overall decrease of total seagrass coverage within the bay amounts to an 8% loss over the 39-yr evaluation span. Between 1953 and 1964 (11 yr) the bay lost 89 ha (214 acres) of coverage; between 1964 and 1980 (16 yr) it lost 583 ha (1,439 acres); and between 1980 and 1992 (12 yr) it lost another 144 ha (358 acres). Even though the last five decades have seen increased regulatory programs, such losses would appear to indicate that somehow regulation, management, and education are still not entirely adequate to conserve St. Andrew Bay's seagrass resources.

St. Andrew Bay Segment

The lower bay segment appears to be stable, despite rapid urbanization and public use. This stability may be in part due to improvements of both industrial and municipal treatment processes and to the fact that St. Andrew Bay has not undergone any major port or harbor improvements, as other bays have. There is no easily perceived cause for the declines in seagrass robustness from 1980 to 1992 which appear to have occurred on the west shore between the Hathaway Bridge and Courtney Point and at the end of the "Tyndall peninsula" between Redfish and Davis Points.

East Bay Segment

East Bay also appears to be a fairly stable seagrass ecosystem, with only minor variations in total coverage and no trend of decline. One factor surely contributing to this stability is the undeveloped shoreline along Tyndall Air Force

Table 4. Seagrass coverage for the North Bay segment for 4 individual years of evaluation.

[Values are in hectares (acres)]

Coverage	1953	1964	1980	1992
Continuous	455 (1,124)	514 (1,270)	483 (1,193)	400 (988)
Patchy	355 (877)	262 (647)	238 (588)	355 (877)
Total	810 (2,002)	776 (1,917)	721 (1,782)	755 (1,866)

Table 5. Seagrass coverage for the West Bay segment for 4 individual years of evaluation.

[Values are in hectares (acres)]

Coverage	1953	1964	1980	1992
Continuous	311 (768)	437 (1,080)	543 (1,342)	92 (227)
Patchy	1,229 (3,037)	830 (2,051)	500 (1,236)	698 (1,725)
Total	1,540 (3,805)	1,267 (3,131)	1,043 (2,577)	790 (1,952)

Table 6. Seagrass coverage for the St. Andrew Sound segment for 4 individual years of evaluation.

[Values are in hectares (acres)]

Coverage	1953	1964	1980	1992
Continuous	7 (17)	22 (54)	72 (178)	22 (54)
Patchy	361 (892)	348 (860)	264 (653)	347 (857)
Total	368 (909)	370 (914)	336 (830)	369 (912)

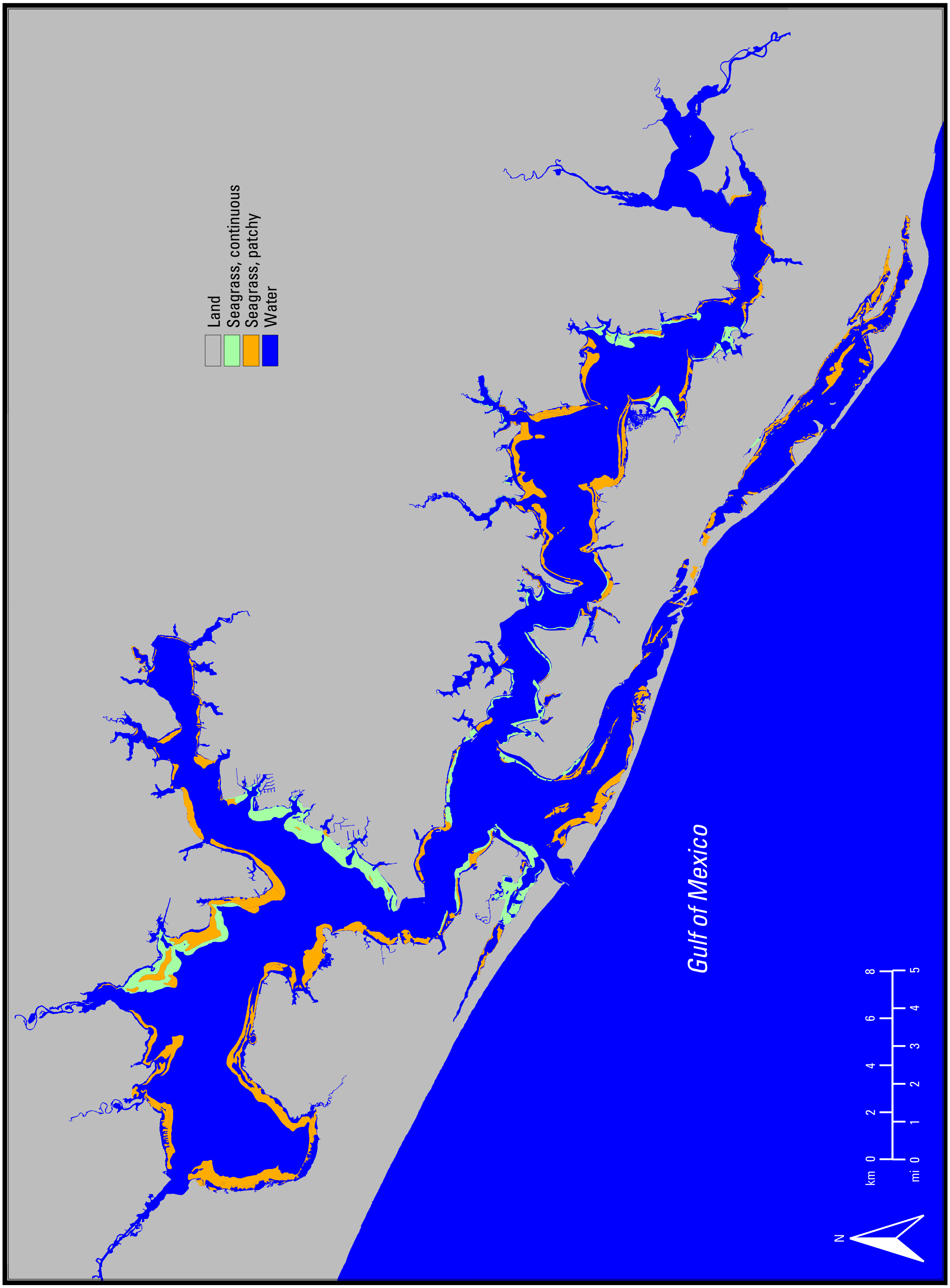

Figure 3. Distribution of seagrass in the St. Andrew Bay system, 1953.

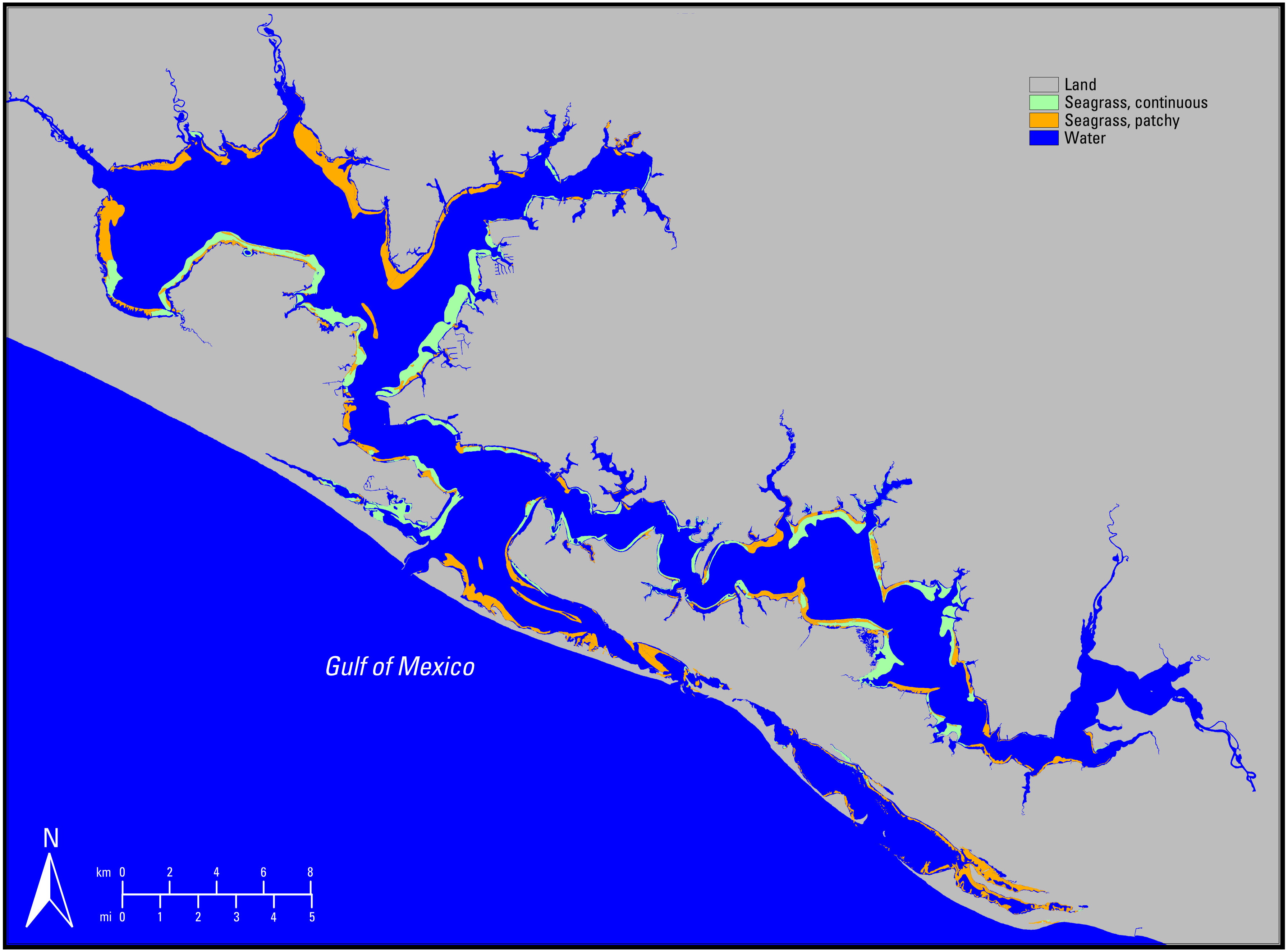

Figure 4. Distribution of seagrass in the St. Andrew Bay system, 1964.

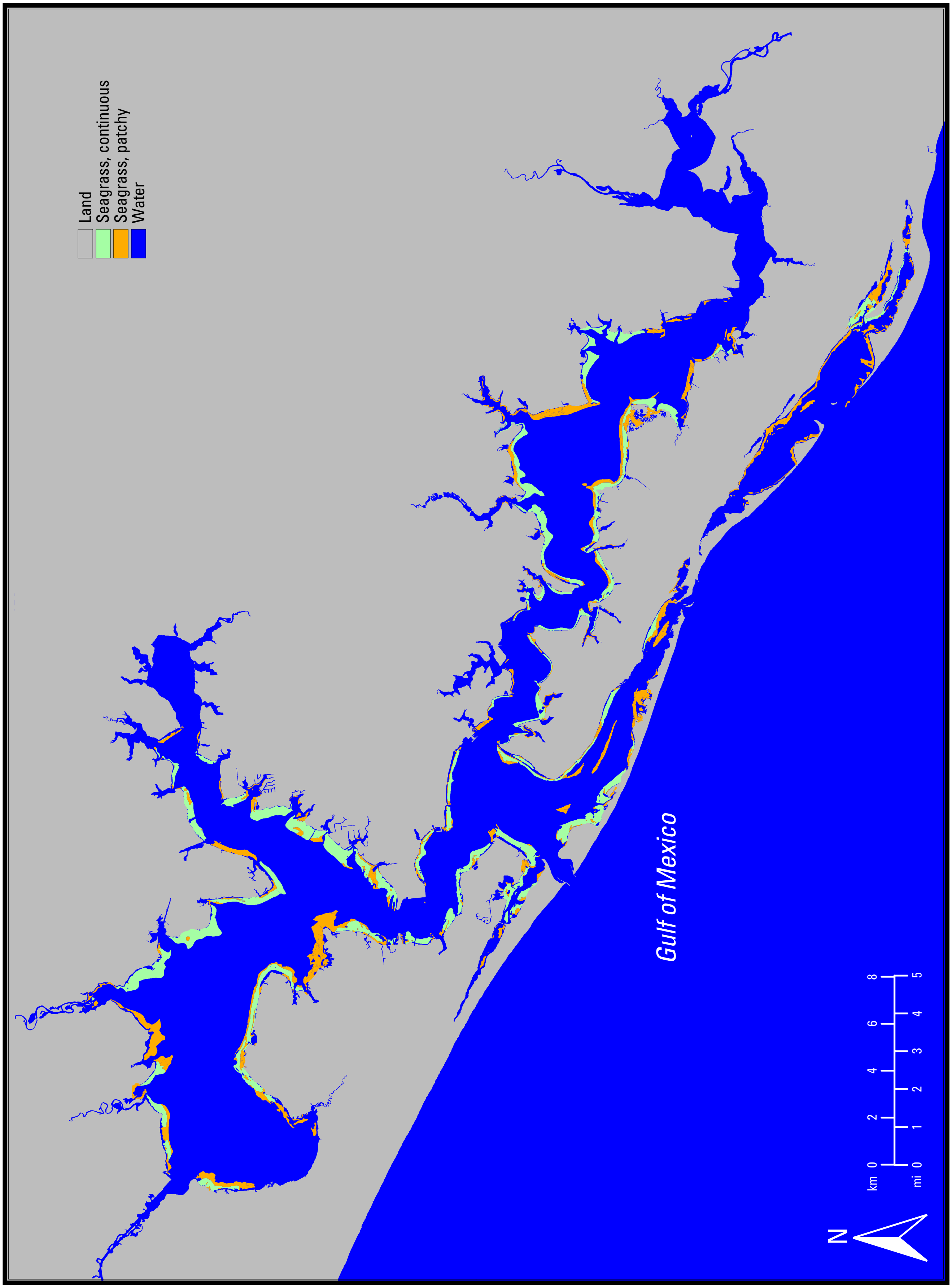

Figure 5. Distribution of seagrass in the St. Andrew Bay system, 1980.

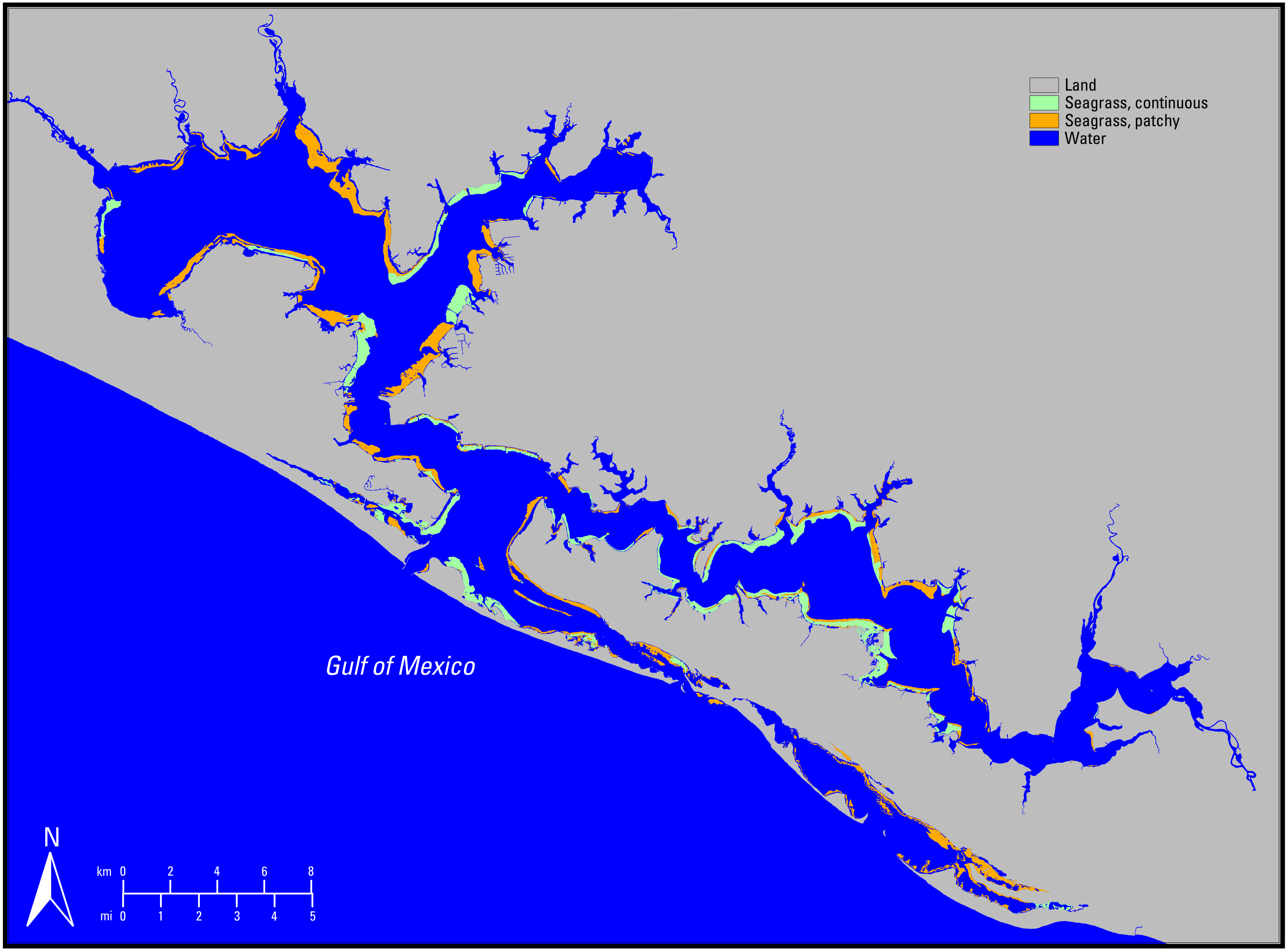

Figure 6. Distribution of seagrass in the St. Andrew Bay system, 1992.

Base covering almost all of the south shore of East Bay. Other factors include the relatively moderate urban development and a lack of any significant heavy industry on the north shore. Careful management of north shore development will be crucial to maintaining the seagrass resources of East Bay.

North Bay Segment

Most seagrass decline in this segment was along the south and southwest shorelines of North Bay where urban development has occurred.

From Little Oyster Bar Point southward to the Hathaway Bridge, seagrass abundance declined from continuous to patchy beds between 1964 and 1992 (see figs. 4–6). This is an area that was subject to the creation of numerous dead-end residential canals which can be seen in figure 6. In addition, a treated effluent discharge was once located south of Posten Bayou, but that point source has been removed and relocated to the lower bay, and effluent treatment levels improved. Only future seagrass monitoring will reveal whether or not any further decline in bed robustness has taken place since 1992.

Losses of beds and a decline in bed robustness can also be observed in North Bay along the south shore between the “Bailey” bridge and the dam at Deer Point Lake. The municipal sewage effluent once drained into Beatty Bayou and could have contributed undesirable nutrients to the bay. This outfall has since been relocated, treatment has been improved, and this possible source of seagrass stress has been eliminated. Other causes of seagrass decline along this shallow shore zone may include increased urban runoff associated with moderate to heavy urbanization in the Lynn Haven area.

West Bay Segment

In 1953, West Bay had a total coverage of 1,540 ha (3,805 acres); this segment lost 750 ha (1,312 acres) of seagrass by 1992, when total coverage had been reduced to just 790 ha (1,952 acres). The history of West Bay and the possible causes for these seagrass losses are complex and complicated. The losses are probably the result of cumulative stresses from several sources. These currently unquantified stresses can only be qualitatively described at this point; however, a restoration effort (described later in this vignette) may shed light on the most significant causes.

The first consideration within West Bay is the Gulf Intracoastal Waterway (GIWW). Over the last four decades, increased vessel traffic, larger vessels, and unstabilized banks may have adversely affected the West Bay system by transporting fine sediments and nutrients into the bay. West Bay is connected with Choctawhatchee Bay by a 29-km (18-mi) long GIWW land cut. At least one incident of complete blockage of the land cut occurred in September 1988 because of sediment runoff and accumulation. In addition, the hydrologic interaction between West Bay and Choctawhatchee Bay via the GIWW is not well defined. The sometimes sediment-laden Choctawhatchee River enters the eastern end of Choctawhatchee Bay not far from the GIWW land cut leading to West Bay. The long-term hydrologic interactions and sediment exchange of this canal have not been clearly defined and quantified. The GIWW could be a source of stresses which may adversely affect seagrass health, even though the land cut connecting the two bays is many kilometers long.

In 1970, two events took place that may have significantly affected the welfare of seagrass beds in southern West Bay. In that year the State of Florida, the USACE, and the EPA issued permits for two separate major projects.

The permits for the first project were issued to the City of Panama City Beach for the discharge of treated municipal effluent into the bay. The city’s effluent enters the bay via a 1.6-km-long (1-mi-long) drainage ditch. Discharge commenced with an initial volume of about 9.5 million liters/day (2.5 million gallons/day). Although the treatment level of the effluent has been significantly improved since 1970, currently the permitted discharge volume to the bay is 26.5 million liters/day (7 million gallons/day). Plans by the city and regulatory agencies are in place to remove the discharge from the bay upon or before expiration of the current EPA National Pollutant Discharge Elimination System permit. Nutrients from this discharge, particularly during the 1970s, could have adversely affected seagrass growth by encouraging the excessive growth of epiphytic algae on the seagrass leaves or by stimulating excessive growth of light-blocking phytoplankton on the water surface above seagrass beds.

Permits for the second project were issued in 1970 to a company named Marifarms for a commercial shrimp-farming operation. A permit and lease allowed the netting-off of 1,012 ha (2,500 acres) of southern West Bay with a net made of mesh with holes less than 1 cm^2 (0.4 $inch^2$), the size required to confine raised shrimp to the leased area. The confining net was approximately 3 m (10 ft) in depth and 5,500 m (18,000 ft) in length (U.S. Army Corps of Engineers, 1970). To ensure confinement, two nets (with a combined total of 11,000 m (36,000 ft) of netting) were kept in place. The permit also allowed the removal of marine fishes within the netted area through the application of rotenone. Fish removal was necessary to prevent unwanted predation upon the shrimp crop. A complaint detailing and challenging the legalities of the Marifarms project was filed in the Circuit Court of the Second Judicial District on August 3, 1970, by the Organized Fishermen of Florida (Circuit Court of the Second Judicial District of Florida, 1970).

The Marifarms company operated from 1970 through 1975, when Hurricane Eloise tore out the netting, and the company subsequently went bankrupt. It was not until 1999, however, that it was discovered that the netting had been treated with an antifouling compound to keep it free of the growth of various marine organisms (algae, tunicates, hydroids, barnacles, etc.). In that year, the shrimp off-loading area and net-treatment location were rediscovered, and paint chips from the treatment operation were chemically analyzed.

The 25-yr-old paint chips still contained nearly 3% copper and 0.2% organo-tin (Hemming and others, 2003).

The effects of continuous shrimp harvesting by trawl, chemical treatment of the nets, and removal of native fish species from the 1,012-ha (2,501-acre) shrimp grow-out area may all have had a significant adverse effect on seagrass survival.

Other potential causes of the West Bay seagrass losses could include less-than-optimal silviculture practices adjacent to the bay. Although residential development is still minimal along West Bay, future development plans are extensive and include the proposed relocation of the international airport to north of West Bay, between Crooked and Burnt Mill Creeks. For these reasons, extensive and careful management is being undertaken to protect and restore the seagrass beds of West Bay.

St. Andrew Sound Segment

The seagrass beds in St. Andrew Sound and their health have remained relatively stable over the 39-yr span of evaluation, probably because of the almost entirely encompassing presence of Tyndall Air Force Base. The shoreline around the sound has remained almost completely unaltered because of the Air Force presence and management.

The only noteworthy event within the sound during the span of study was Hurricane Eloise, which altered the area geologically and hydrologically. The storm and its surge effects have been documented by Burdin (1976). Prior to 1975, the entrance to the sound was an inlet at the northwest end, but in September 1975 the hurricane breached the 16-km-long (10-mi-long) barrier peninsula at its midpoint. Within 2 yr, the northwest inlet had closed. The present-day entrance is at the midpoint of the sound. This relocation of the inlet appears to have had some moderate effects on seagrass distribution.

Species Information

Several species of seagrasses occur within St. Andrew Bay and St. Andrew Sound, and their presence has been reported regularly (Saloman and others, 1982; Florida Department of Natural Resources, 1991; Fonseca, 1994; Koenig and others, 1998).

The dominant seagrass species, undoubtedly providing the most valuable seagrass habitat, is turtle grass (*Thalassia testudinum*). This subtidal species grows to depths of 1.8–2.4 m (6–8 ft) mean low water in the southern bay areas near the gulf (St. Andrew Bay segment) and in St. Andrew Sound. Light penetration appears to limit maximum growth depths to 1.2–1.8 m (4–6 ft) in the interior bay segments.

The intertidal and subtidal species shoal grass (*Halodule wrightii*) dominates the shallow and intertidal bay areas. These beds are often exposed to the air in winter when north winds create “wind tides” and push large amounts of water from the bay and in spring when the north winds combine with low tide conditions of the solunar spring tide. Manatee grass (*Syringodium filiforme*) can be found scattered within turtle grass beds, or sometimes in pure stands thriving near the influence of clear, highly saline water entering the bay from the gulf. While it is not a dominant species, it provides valuable diversity and cover where it is present. Wigeon grass (*Ruppia maritima*) occurs in some of the fresher parts of the bay, including areas of some bayous. Finally, the presence of star grass and paddle grass (*Halophila* spp.) has been noted, albeit rarely, amongst turtle grass beds of the southern bay.

Monitoring for Seagrass Health

The St. Andrew Bay Resource Management Association, in partnership with Gulf Coast Community College and the Florida Department of Environmental Protection, initiated in 2000 a seagrass monitoring program at three locations in the St. Andrew Bay estuarine system. Transects have been established at sites near Shell Island, in Grand Lagoon (both within the St. Andrew Bay segment), and in West Bay. Monitoring is accomplished in fall each year. Data collected include species composition, shoot density, percent cover, and canopy height. In addition, water-quality data, including turbidity and photosynthetically active radiation, are collected monthly at each site. The work is supported by a grant from the National Oceanic and Atmospheric Administration to the Florida Department of Community Affairs. Because the program is only 3 yr old, no trends regarding seagrass health, gains, or losses have yet been documented.

Mapping and Monitoring Needs

Seagrasses within the St. Andrew Bay system and St. Andrew Sound have not been mapped since 1992. The ecosystem management plan developed by the St. Andrew Bay Environmental Study Team (Keppner and Keppner, 2001), however, includes an action plan (SG1) encouraging monitoring of seagrass beds by aerial photography and interpretation every 5 yr in partnership with the U.S. Fish and Wildlife Service. The action plan also encourages continuing the St. Andrew Bay Resource Management Association-Gulf Coast Community College seagrass monitoring program in the bay. New aerial photography for the Florida Panhandle was taken in October 2003, and the resulting maps and geographic information system data layers will be made available to agencies, participating conservation groups, and county and municipal government planning departments in 2005.

The other action plans (Keppner and Keppner, 2001) regarding seagrasses are discussed in the following section.

Restoration and Enhancement Opportunities

The Bay Environmental Study Team (BEST) for St. Andrew Bay is composed of all users of the bay including State, Federal, and municipal governments, industries, academic institutions, conservation organizations, commercial fishing interests, and the general public. BEST is structured very similar to the EPA National Estuary Program and serves a similar purpose for St. Andrew Bay.

The BEST ecosystem management plan for St. Andrew Bay includes other action plans, besides the previously explained SG1 (above), to protect and preserve seagrasses:

- SG2, Protection of Seagrass Beds, calls for pursuit of additional State, Federal, and local legislation that would provide additional protection for the seagrass beds in the St. Andrew Bay ecosystem. The action plan will also include actions on the bay such as the placement of marker buoys and channel markers to reduce propeller scarring in seagrass beds.

- SG3, Restoration of Lost or Damaged Seagrass Beds (anywhere within the bay), includes work for inventorying all of the bay, identifying areas of loss, and designing and imple menting restoration at any sites where such action is appropriate.

- SG4, Restoration of Seagrass Loss in West Bay, is aimed at the restoration of the lost West Bay seagrass beds. Currently BEST is the project sponsor for this USACE Aquatic Ecosystem Restoration Project (under the authority provided by section 206 of the Water Resources Development Act of 1996). The project was initially evaluated by the Mobile District Corps of Engineers in 2001. In spring 2002, the Preliminary Restoration Plan was completed (U.S. Army Corps of Engineers, 2002). To date, the scope of work is being developed for the Ecosystem Restoration Report, a study which should provide critical information regarding seagrass stressors and their elimination, as well as an optimal strategy for reestablishing seagrass beds.

- SG5, Education About the Significance of Seagrass Meadows in the Ecosystem, involves (1) the production of an underwater video depicting the importance of seagrass meadows, (2) the production of a video about seagrass monitoring, (3) distribution of “A Boater’s Guide to St. Andrew Bay,” (4) placement of educational signs at boat ramps, and (5) the development of an educational display about seagrasses for use at conservation meetings and other public gatherings.

- SG6, Innovative Pier and Dock Construction, is aimed at developing and encouraging the use of pier and dock materials and designs which encourage seagrass growth beneath them, are minimally disruptive to seagrasses, and are created of materials that are chemically compatible (nontoxic) with seagrasses or estuarine organisms.

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Florida Big Bend

By Robert A. Mattson,[1] Thomas K. Frazer,[2] Jason Hale,[2] Seth Blitch,[3] and Lisa Ahijevych[3]

Background

Those who have visited Florida's Big Bend coast are often struck by the distinctly "wilderness" feel of the area. It is possible, even today, to venture out onto the water and not encounter another boat for several days. The area has been described by Livingston (1990, p. 554) as "one of the least polluted coastal regions of the continental United States." Exceptional water quality and clarity in the shallow waters of the region provide a favorable growing environment for seagrasses.

Along the Big Bend coast of Florida (from Anclote Key north to Apalachee Bay), seagrass coverage is extensive (3,000 km^2 or 1,158 mi^2) (see Zieman and Zieman, 1989; Mattson, 2000). In fact, seagrass beds are often the dominant structural feature in the shallow, subtidal estuaries and nearshore, coastal waters in the region. As such, seagrasses provide essential refuge and forage habitats for a myriad of ecologically and economically important fauna. Approximately 85% of the recreational and commercial fishery species in Florida spend some portion of their life in estuaries (Comp and Seaman, 1985), and many of these species are considered obligate seagrass inhabitants. Blue crabs (*Callinectes sapidus*) and bay scallops (*Argopecten irradians*), for example, are largely dependent on seagrass resources (Orth and van Montfrans, 1987, 1990). The Big Bend region accounts for between 25% and 33% of the total commercial blue crab fishery landings in Florida and supports the largest recreational scallop fishery in the State. Seagrass beds are considered essential to the ecological integrity and health of Florida's estuarine and nearshore coastal ecosystems.

Land cover in the region includes natural wetland and upland types (Berndt and others, 1996). Land use in the region includes commercial forestry, agriculture (row crops, poultry, dairy cattle, beef cattle, horses, and hay, as well as citrus in the southern part of the region), and urbanized land (residential and commercial), with the latter being more extensive in the southern part of the region. Industrial development is not extensive; the major facilities are the Buckeye Florida, L.P., pulp mill in Taylor County and the Florida Power Corporation (FPC) generating complexes in Citrus and Pasco Counties. Small commercial port facilities are at St. Marks and the FPC Crystal River generating complex.

The remoteness of the Big Bend and its relatively pristine character have been both a blessing and a curse. Overall, this area is one of the least populated in Florida and is also one of the poorest in terms of per capita income and ad valorem tax base. These factors, in large part, are the very reasons that this area is so undeveloped and undisturbed. Yet, the lack of economic and political power in the area contributes to a related lack of investment of State and Federal resources to conduct the studies needed to effectively manage the area's seagrass resources. This situation is in contrast to that of the more highly urbanized areas of Florida's gulf coast, where the economics at stake (recreational and commercial fishery and waterfront property values) and heavy public use have generated the political pressure and resources needed to assess, manage, and restore seagrass habitats.

Over half of the entire Big Bend region is part of the Big Bend Seagrasses Aquatic Preserve, managed by the Florida Department of Environmental Protection (FDEP). Aquatic preserves are areas of State-owned, submerged lands permanently set aside and protected for the benefit of future generations. This preserve was designated by the Florida Legislature in 1985 in recognition of the area's "exceptional biological, aesthetic, and scientific value" (Chapters 18–20 FAC/chapter 253, 395 FS). The Big Bend region also includes five U.S. Fish and Wildlife Service (USFWS) national wildlife refuges (St. Marks, Lower Suwannee, Cedar Keys, Crystal River, and Chassahowitzka) and several other State conservation areas (Econfina River State Park, Cedar Key Scrub State Reserve, Waccasassa Bay State Preserve, St. Martins Marsh Aquatic Preserve, and Homosassa Springs Wildlife State Park). Extensive areas of the fringing intertidal marshes bordering the Big Bend coastal waters have been acquired by the State of Florida and are State wildlife management areas (e.g., Spring Creek, Tide Swamp, and Jena). A substantial investment of public dollars has been made to acquire and manage these lands. The immense seagrass ecosystem which forms a part of these conservation areas is one of the key components of their natural value.

Some of the earliest basic research on the ecology of Florida seagrasses was conducted in the Big Bend. For example, Reid (1954) first documented the existence of a distinct seagrass ichthyofauna in the Cedar Key area. Phillips

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Figure 1. Watershed for the Florida Big Bend region.

(1960a, b) studied seagrass community characteristics in the areas around Crystal River and to the south, and Strawn (1961) investigated patterns of seagrass zonation in the Cedar Key area. Ballantine and Humm (1975) sampled seagrass epiphytes in the area around Anclote Key. Researchers at Florida State University studied seagrass-faunal interactions in Apalachee Bay beginning in the mid-1970s and continued this work into the 1990s (see Livingston, 1984a, b; Livingston and others, 1998). Iverson and Bittaker (1986) conducted the first regionwide survey of seagrasses in the Big Bend during the mid- and late 1970s. Despite these past and current investigations, however, we cannot answer many important questions about the Big Bend seagrass ecosystem:

1. What is the current status of seagrasses in the Big Bend in terms of areal coverage, species composition, standing crop, productivity, or other characteristics?
2. Is seagrass coverage in the region declining, increasing, or not changing? Are there similar trends in standing crop, shoot density, or productivity, or is species composition changing?
3. What water-quality conditions need to be met to maintain seagrasses in the region?

Scope of Area

The Big Bend area discussed in this report extends from Lighthouse Point, in the southwest corner of Apalachee Bay, east and south along the Florida gulf coast to Anclote Key off the mouth of the Anclote River near Tarpon Springs (fig. 1). This is the area that has typically been described as "Florida's Big Bend Coast." Geologically, the entire area is similar, consisting of drowned karst, with limestone at or near the surface of the land or submerged bottom (Terrell, 1979; Davis, 1997). Geomorphic and meteorologic characteristics of the area result in a low-energy coastline in terms of wave and wind activity (Davis, 1997). It is also a "sediment-starved" coastline, as the rivers draining to the region all carry low sediment loads (Hine and Belknap, 1986; Davis, 1997). The area's climate is a combination of southern temperate and subtropical. Summers are hot and wet; generally half or more of the total annual precipitation falls between late June and September (Henry, 1998). Winters are cool and somewhat wet to the north and drier and warmer at the southern end of the region. Dry seasons typically occur in the spring (late April to early June) and fall (October to December).

Two segments of this coastal region are currently recognized (Wolfe, 1990; Estevez and others, 1991): the northern half, or "Big Bend Proper," which extends from Lighthouse Point, east and south to Waccasassa Bay, and the southern half, or "Springs Coast," which extends from Withlacoochee Bay south to the Anclote area (fig. 2). This geographic division and the segment names will be maintained in this report.

Eight stream systems drain into the Big Bend Proper: the St. Marks, Aucilla, Econfina, Fenholloway, Steinhatchee, Suwannee, and Waccasassa Rivers and Spring Warrior Creek. These streams are fed partly by springs discharging ground water from the Floridan aquifer system and partly by surface-water runoff from their drainage basins. Surface runoff dominates the hydrology of these rivers during high or flood flows, while groundwater inflow from springs predominates during low or base flows. The Springs Coast is fed by seven river systems: the Withlacoochee, Crystal, Homosassa, Chassahowitzka, Weeki Wachee, Pithlachascotee, and Anclote Rivers. The Withlacoochee River is similar to the rivers that drain the Big Bend Proper in that it is fed by a combination of spring flow and surface runoff. The other six rivers that feed/drain the Springs Coast are spring-run streams (Nordlie, 1990; Estevez and others, 1991), fed almost entirely by ground water discharging from first-magnitude springs or spring groups. The two southernmost rivers, Pithlachascotee and Anclote, are mostly fed by surface runoff, with a lesser amount of groundwater inflow.

River flood seasons usually occur in winter in the Big Bend Proper segment (January–March) and in late summer in the Springs Coast segment (July–September). Regionwide, the combination of geologic and hydrographic conditions creates an immense area of shallow, clear water, which allows for the growth of seagrasses. Because most of the seagrass species present in this region have tropical affinities (see the section on species information in this vignette), they are growing near their northern limits of distribution in North America here in the northern Gulf of Mexico.

Methodology Employed To Determine and Document Current Status

There is presently no comprehensive program to monitor and assess seagrass coverage or the health of seagrasses throughout the Big Bend region. Mattson (2000) and Zieman and Zieman (1989) summarized existing data from the region that were derived from the published and grey literature.

The most current mapping study of seagrass coverage for the Florida Big Bend area was conducted more than 10 yr ago by the U.S. Geological Survey (USGS) National Wetlands Research Center (NWRC) by using natural-color aerial photography taken in 1992 at a 1:24,000 scale as part of the northeastern Gulf of Mexico seagrass mapping project.

The mapping protocol consisted of stereoscopic photointerpretation, cartographic transfer, and digitization in acccordance with strict mapping standards and conventions. Other important aspects of the protocol included the development of a seagrass classification system, groundtruthing, quality control, and peer review. The

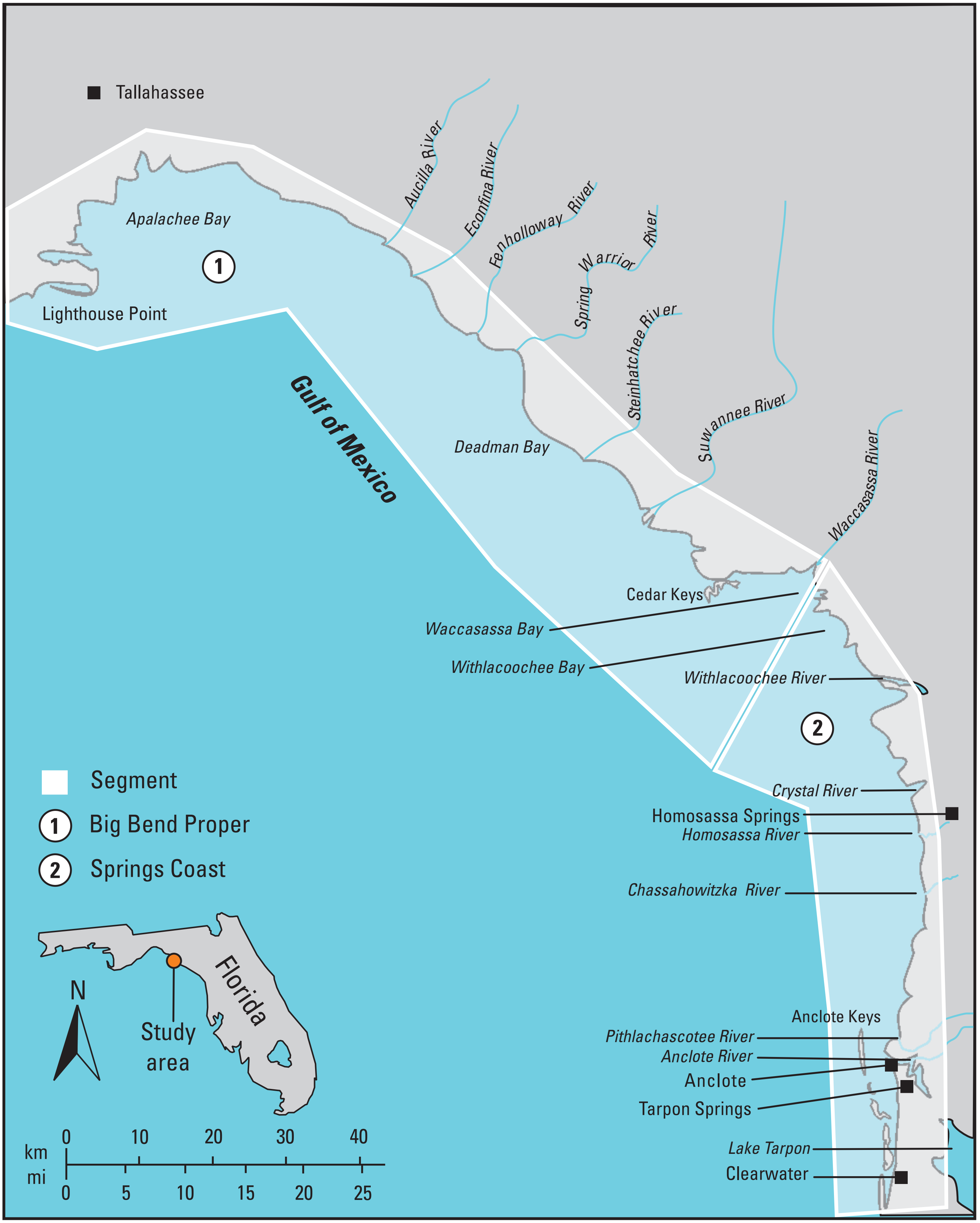


Figure 2. Scope of area for the Florida Big Bend vignette.

information derived from the photography was subsequently transferred by using a zoom transfer scope onto a stable medium overlaying USGS 1:24,000-scale quadrangle base maps. In those cases in which the data were inadequate or incomplete, contemporary supplemental data were acquired from other sources and used to complete the photographic coverage.

The seagrass classification system that was developed consisted of two classes of open water—RIV (riverine, fresh water) and EST (estuarine or marine open water)—and five classes of seagrass habitats. One seagrass habitat class designated continuous seagrass, CSG, for which no density distinction was made, and the other four classes designated patchy seagrass based on percent ground cover of patches in 5% increments: PSG1 (0%–10%, very sparse), PSG2 (15%–40%, sparse), PSG3 (45%–70%, moderate), and PSG4 (75%–95%, dense). For purposes of this vignette, we report the data only as continuous or patchy.

The groundtruthing phase included the participation of field staff from Gulf Islands National Seashore, U.S. Fish and Wildlife Service, Dauphin Island Sea Lab, Mississippi State University, Alabama Department of Conservation and Natural Resources, and FDEP. Water conditions during this time were not optimal (F. Sargent, Florida Marine Research Institute, oral commun.); therefore, the usefulness of the photography was limited. In addition, because of personnel and resource limitations, groundtruthing of this photography was also limited. Draft maps were sent out to the aforementioned agencies for review and comments. All comments received were incorporated into the final maps.

A more recent effort to map seagrass cover in the Springs Coast segment was completed in 1999 by the Department of Fisheries and Aquatic Sciences at the University of Florida under contract to the Southwest Florida Water Management District (SWFWMD) (Frazer and Hale, 2001). Natural color aerial photography at a scale of 1:24,000 was flown in December 1999 and was groundtruthed and scanned into digital format for analysis by using a combination of image processing and traditional photointerpretation techniques. After adapting the original maps to the results of an accuracy assessment protocol that was conducted by personnel from SWFWMD and FDEP, the maps were estimated to be approximately 80% accurate in describing seagrass abundance and distribution along the Springs Coast.

Methodology Employed To Analyze Historical Trends

To date, there has been no comprehensive study of trends in seagrass cover over time in the Big Bend region. Current and previous seagrass mapping efforts suffer from a lack of consistency in approach (summarized in Mattson, 2000) that makes determination of trends difficult. In addition to the 1992 mapping effort (U.S. Geological Survey, 1992) described above, the only other regionwide mapping survey was conducted by a private contractor for the Minerals Management Service (MMS) in 1984. Natural color aerial photography at a 1:40,000 scale was flown in October and November 1984 and interpreted stereoscopically (Continental Shelf Associates, Inc., and Martel Laboratories, Inc., 1985). Groundtruthing efforts (both by divers and by using remote video cameras) concentrated mainly on deepwater areas because of the focus of the study and the lack of data on deepwater seagrass beds (Continental Shelf Associates, Inc., and Martel Laboratories, Inc., 1985; D. Deis, oral commun., October 1998). The results of the 1984 survey for MMS and the later 1992 survey by USGS are not considered comparable, and no trend analysis using these two datasets is conducted in this report.

In an effort to create comparable datasets of seagrass abundance and distribution for the Springs Coast, researchers at the University of Florida reclassified the aerial photographs acquired in 1992 and used the same combination of traditional photointerpretation and image processing developed for the aerial photography taken in 1999 (Frazer and Hale, 2001). The product was the first estimate of changes in seagrass abundance and distribution occurring in the 1990s, but aerial photographic coverage was restricted to the nearshore waters along the Springs Coast.

An additional effort to assess historical changes was recently conducted by the University of Florida, SWFWMD and the Florida Marine Research Institute (FMRI) in 2000. This project was an attempt to locate and revisit sites assessed by Iverson and Bittaker in the late 1970s in their Florida seagrass study (Iverson and Bittaker, 1986). This effort included much of the northern Big Bend Proper, as well as the Springs Coast. The effort was hampered by the lack of accurate geographic reference coordinates on the sites visited by the earlier investigators. The sites were located as best as possible, and latitude/longitude was obtained by using a handheld Global Positioning System unit. Upon location of a site, the revisit entailed qualitative inspections of the bottom to assess seagrass species present for comparison with the past effort. Acquisition of coordinates on these sites now permits revisits to be made reliably in future years. The main goal of this revisit was to see if broad-scale changes in seagrass species composition or distribution had occurred over the past 25 yr by comparing the species of seagrass seen historically at a site with those observed at the time of the study. The main limitation of this effort was that it could not be assured that the exact sites visited by Iverson and Bittaker in the 1970s were being revisited in 1999.

Status and Trends

Three regionwide mapping studies of seagrass coverage were conducted in the last two decades of the 20th century (Continental Shelf Associates and Martel Laboratories, 1985; U.S. Geological Survey, 1992; Sargent and others, 1995).

Table 1. Estimates of seagrass cover in the Big Bend region.

[Area is given in hectares (acres)]

	Continuous Cover	Patchy Cover	Total
Minerals Management Service (1984)			
Big Bend Proper	80,081 (197,880)	250,768 (619,648)	330,849 (817,528)
Springs Coast	135,871 (335,737)	53,572 (132,376)	189,443 (468,114)
Total	215,952 (533,617)	304,340 (752,024)	520,292 (1,285,642)
U.S. Geological Survey (1992)			
Big Bend Proper	27,159 (67,110)	81,153 (200,529)	108,312 (267,639)
Springs Coast	1,202 (2,970)	140,972 (348,342)	142,174 (351,312)
Total	28,362 (70,083)	222,125 (548,871)	250,487 (618,953)
Florida Marine Research Institute (1995)			
Big Bend Proper[1]			181,000 (447,251)
Springs Coast[1]			154,000 (380,534)
Total			335,000 (827,785)
University of Florida (2001)			
Springs Coast 1992		39,714 (98,133)[2]	81,742 (201,985)[3]
Springs Coast 1999		46,545 (115,013)[2]	92,028 (227,401)[3]

[1] For this study, Big Bend Proper was defined as the five counties from Apalachicola River to the Withlacoochee River. Springs Coast was the three counties from Withlacoochee River to the Anclote River.

[2] Interpreted as >25% cover/m^2.

[3] Represents total area surveyed.

The results of these studies as well as a subregional mapping study conducted by the University of Florida in 2001 are shown in table 1. In spite of different project goals, methods, and scales of source data, estimates of seagrass abundance for the region have generally arrived at the same basic area of seagrass coverage. The MMS mapping project (Continental Shelf Associates and Martel Laboratories, 1985) used aerial photographs and underwater video transects to sample depths to 20 m (66 ft). The USGS (1992) and Sargent and others (1995) collected and interpreted aerial photographs of the region. All of these projects estimated seagrass coverage along the Big Bend as roughly 300,000 ha (741,300 acres). A few subregional mapping studies of smaller extent have been conducted; however, their study areas were confined to nearshore areas, e.g., within 3 km (2 mi) of shore (McNulty and others, 1972), or did not cover the entire region (e.g., Livingston, 1993, covered only Apalachee Bay).

Regionwide Mapping Studies

Minerals Management Service

The MMS effort in 1984 (Continental Shelf Associates and Martel Laboratories, 1985) mapped 520,292 ha (1,285,642 acres) of seagrass habitat (fig. 3; table 1) in the Big Bend region. Differences in the scale and quality of the photography used and the focus of this earlier effort on offshore seagrass resources mean that the 1984 data cannot be directly compared to subsequent mapping data; therefore caution should be used in comparing figures 3 and 4 and the data in table 1. It should be noted that the MMS effort in 1984 is the only attempt to date to map the extent of the offshore, deepwater seagrass beds of the Big Bend.

USGS

The 1992 seagrass mapping study by the USGS yielded an estimate of 250,487 ha (618,953 acres) of total seagrass cover in the entire Big Bend region (fig. 4; table 1). This estimate somewhat corresponds with Iverson and Bittaker's (1986) estimate of 300,000 ha (741,300 acres) and less so with the FMRI 1995 estimate (Sargent and others, 1995) of 335,000 ha (827,785 acres). Of the total acreage mapped in 1992, 23% was mapped as continuous seagrass cover, and 77% was patchy cover. Of this total Big Bend seagrass cover reported in 1992, more than half (57%) occurred in the southern portion of the region, the Springs Coast; the remainder (43%) was in Big Bend Proper.

Florida Marine Research Institute

This effort by Sargent and others (1995) mapped 335,000 ha (827,785 acres) of seagrass habitat (fig. 4; table 1). Sources of photography varied depending upon which area of the State the study was in. It appears that the photography they used from the Big Bend region was the 1992 USGS photography, which they remapped. This study observed an opposite trend in seagrass cover between the two major segments than did the USGS study: the majority of the seagrass mapped (54%) was in the Big Bend Proper, with less (46%) in the Springs Coast.

Other Subregional Mapping Studies

Springs Coast

The mapping study conducted by the University of Florida in 1999 and 2000 (Frazer and Hale, 2001) identified 92,028 ha (227,401 acres) in 1999. Comparing their estimate to the 81,742 ha (201,985 acres) found in 1992 (U.S. Geological Survey, 1992), their study suggested an increase in seagrass coverage in this subarea. The analysis of change (Frazer and Hale, 2001) in seagrass abundance and distribution along the Springs Coast produced several interesting results. The area in which seagrass density increased was twice the area in which seagrass density decreased. Seagrasses in the St. Martins Keys and the Homosassa River estuarine areas appear to have increased in coverage; however, some declines in coverage appear to have occurred near the mouth of Crystal River and in the Anclote Key and Tarpon Springs region.

Apalachee Bay

A subregional study not listed in table 1 was conducted in the region of the Fenholloway River estuary. Livingston (1993) compared seagrass cover off the Fenholloway River, which was affected by highly colored wastewater from a pulp mill discharge upstream, with cover in adjacent, unimpacted drainages (Econfina River, Aucilla River, Spring Warrior Creek). From this comparison, he estimated that about 2,330 ha (5,757 acres) of historical seagrass coverage had been lost since 1954 (when the mill began operation) as a consequence of light reductions from the mill's wastewater (Livingston, 1993; Livingston and others, 1998).

The resurvey of the sites sampled by Iverson and Bittaker (1986) in the 1970s suggests that changes in species composition may have occurred over the past 25 yr (Hale and others, 2004). Although some seagrasses were observed at more sites in the 1999 survey than in the earlier survey (table 2), the distribution of particular species seems to have changed. Turtle grass (*Thalassia testudinum*) decreased in occurrence from deeper areas of the region, while other species seem to have disappeared from areas near the mouths of several coastal rivers. In addition, some species which were recorded during the Iverson and Bittaker (1986) survey were not observed at all during the recent survey (Hale and others, 2004).

Table 2. Records of seagrass species occurrence made by Iverson and Bittaker (1986) and later by researchers at the University of Florida, Southwest Florida Water Management District, and Florida Marine Research Institue in 2000 (unpublished data). The number of sites at which a species was observed in each geographic subregion for each sampling period is listed.

	1974–80	2000
Big Bend Proper		
Turtle grass (*Thalassia testudinum*)	17	15
Manatee grass (*Syringodium filiforme*)	18	18
Shoal grass (*Halodule wrightii*)	16	23
Star grass (*Halophila engelmannii*)	14	4
Springs Coast		
Turtle grass (*Thalassia testudinum*)	46	39
Manatee grass (*Syringodium filiforme*)	25	33
Shoal grass (*Halodule wrightii*)	27	30
Star grass (*Halophila engelmannii*)	9	7

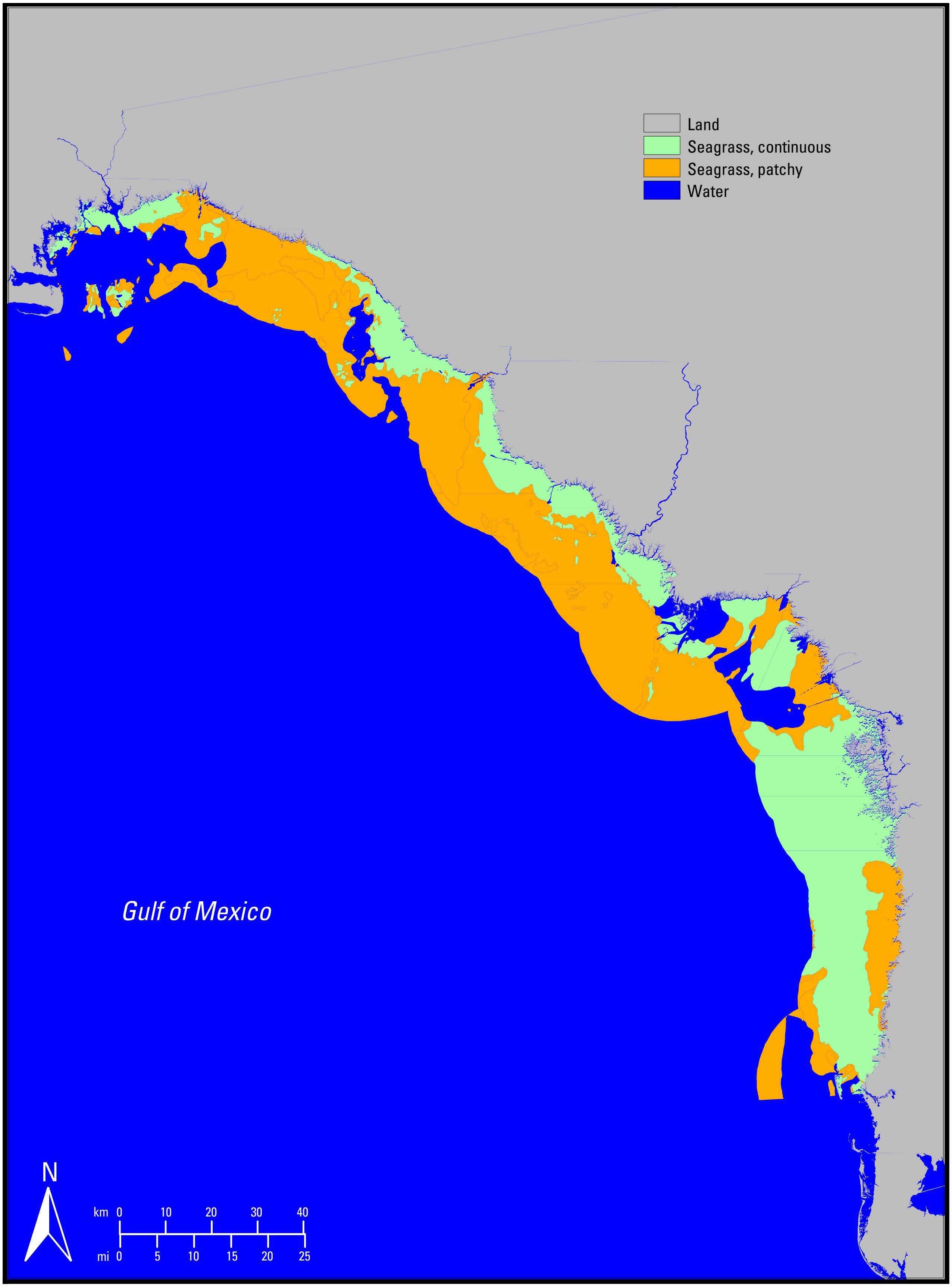


Figure 3. Distribution of seagrasses in the Florida Big Bend region, 1984.

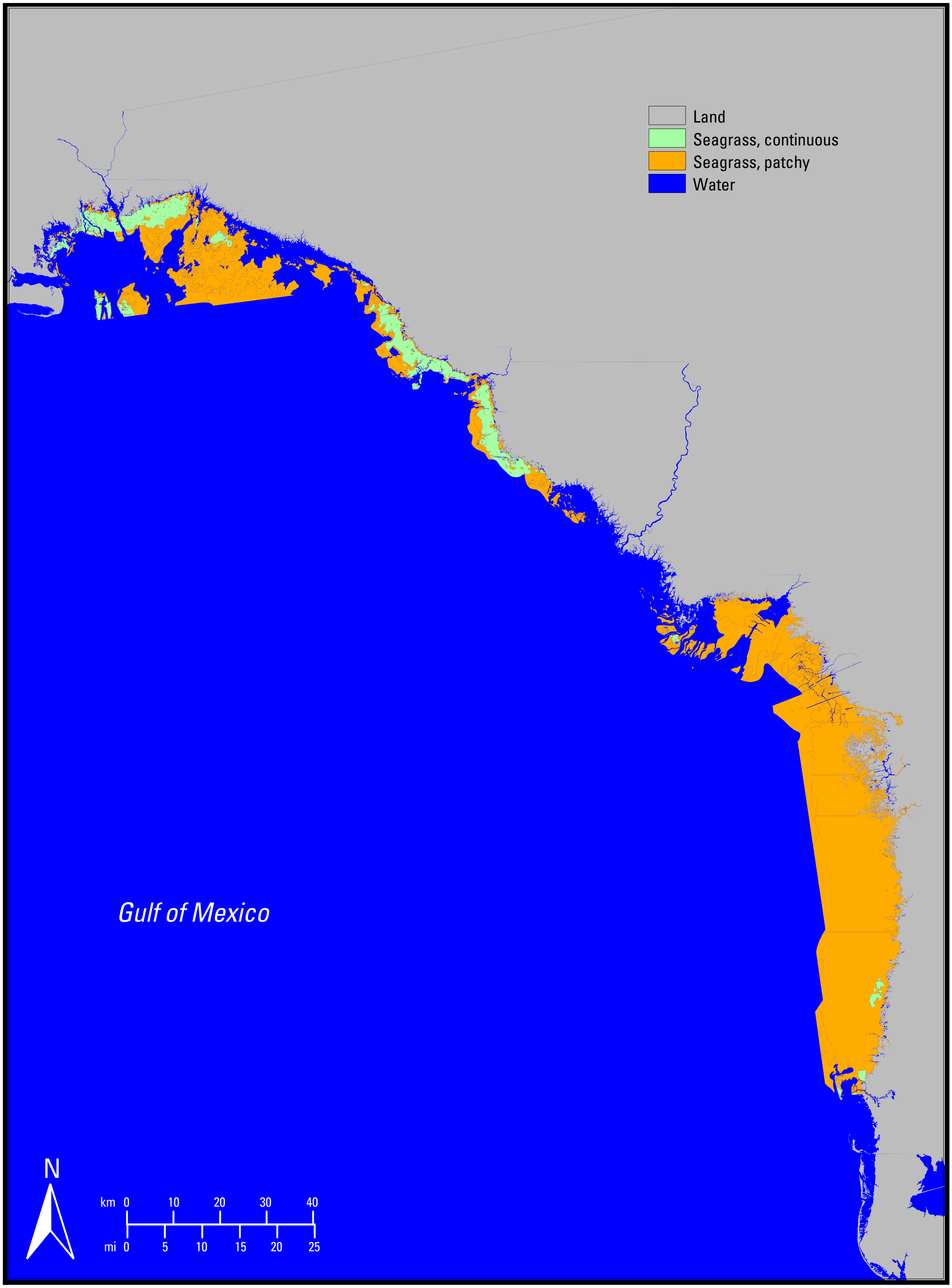


Figure 4. Distribution of seagrasses in the Florida Big Bend region, 1992.

Although several seagrass mapping projects have been conducted in the Big Bend region over the past 20–25 yr, differences in the actual area mapped, methodology, and quality of base data quality make trend analyses and comparisons impossible. There are anecdotal accounts of changes in seagrass cover in portions of the region; this anecdotal evidence suggests that there should be future efforts to conduct trend analyses. Moore (1963) and Grinnel (1971) both provided credible observations which suggest that inshore areas off the Suwannee River that are now unvegetated were historically vegetated with seagrasses. In a study in Waccasassa Bay in the early 1960s, Putnam (1967) indicated that seagrass cover may have been more extensive historically than is currently seen in this estuary.

Causes of Change

As noted previously, documented trends in seagrass cover along the Big Bend are lacking. Consequently, the causes of change cannot be discussed in great detail at this time. Two major issues of concern, however, can be identified as possibly having an effect on seagrass resources in the Big Bend region.

Hydrologic Alteration of Watersheds

Investigations in the Steinhatchee River subbasin have suggested that historical drainage activities associated with commercial forestry may have affected river flow regimes (KBN Engineering and Applied Sciences, 1990). The main effect of this alteration was shown to be an increase in river peak flows during wet seasons (KBN Engineering and Applied Sciences, 1990). Drainage activities were mainly conducted in headwater wetland areas, such as San Pedro Bay (Taylor County) and Mallory Swamp (Lafayette and Dixie Counties), and involved increased flow of highly colored surface-water runoff into the estuary of the river, with possible effects on seagrass communities. Therefore, it is possible that the changes in flow regimes in the Steinhatchee River subbasin have resulted in increased organic load to the river and nearshore coastal area with a concomitant increase in color and a resultant decrease in light available to seagrasses in the area.

In a nearshore area adjacent to the Fenholloway River, Livingston and others (1998) found that higher color as a result of industrial discharges was associated with changes in the absolute quantity and quality of light penetrating the water column and that these changes had measurable negative effects on seagrass biology and community characteristics. These findings suggested that there are similar effects from hydrologic alteration in these coastal watersheds (i.e., increased discharge of highly colored runoff, with resulting negative effects on seagrasses because of changes in water clarity). They also suggested that forestry practices have the potential to cause changes in estuaries and nearshore areas along the Big Bend and merit the attention of seagrass ecologists and water resource managers. Unfortunately, there are few historical data that can be used to determine whether or not changes in water clarity or seagrass cover as a result of forestry-related activities have occurred in the Steinhatchee estuary (or other watersheds where forestry activities in the headwater wetland areas may have resulted in similar alterations of flow regimes and chemical characteristics, e.g., the Suwannee and Waccasassa Rivers). Currently, many of the headwater wetland areas are being purchased by government agencies outright, or conservation easements are being acquired, and efforts are underway to restore more natural, historical drainage patterns and river flow regimes.

Nutrient Enrichment of Estuaries and Nearshore Coastal Waters

Because of the karst geology of the land which drains to the Big Bend coastal waters, many regions are moderately to highly prone to groundwater contamination (Katz and others, 1997; Hornsby and Ceryak, 1999). Land-use activities can rapidly and profoundly affect water quality in the underlying Floridan aquifer system (Katz and others, 1997). Elevated levels of nitrate-nitrogen have been found in portions of the Floridan aquifer throughout the Big Bend (Jones and others 1997, 1998; Hornsby and Ceryak, 1999) and have manifested as increasing levels of nitrate-nitrogen in some river systems (Ham and Hatzell, 1996) and in the individual springs which feed the rivers (Jones and others, 1997, 1998; Hornsby and others, 2000). Nitrogen sources include agricultural land uses, fertilizer from golf courses, and urban development (residential and commercial).

Increased nutrient concentrations in the coastal river systems and subsequent nutrient delivery to the Gulf of Mexico have led to concerns about algal blooms affecting light penetration to seagrasses. Phytoplankton blooms in the water column, increased epiphyte loads on seagrass blades, or macroalgal blooms covering seagrass beds all may produce undesirable effects on seagrass cover and community characteristics. Increased water column nitrogen has been shown to affect phytoplankton production and standing crop in the Suwannee estuary (Bledsoe and Phlips, 2000; Phlips and Bledsoe, 2002), and increased nitrate-nitrite concentration appears to be associated with higher periphyton standing crops in the Suwannee River and tributaries (Hornsby and others, 2000). In contrast, algal production along much of the Springs Coast may be phosphorus limited (Frazer and others, 2001), and increases in this nutrient may elicit similar responses. Trends in surface-water phosphorus concentrations have received much less attention but merit further investigation. In order to properly protect and manage the seagrass resources in the region, research is needed to determine the appropriate water-quality criteria (nutrients, color, and turbidity) necessary to maintain adequate water clarity for light penetration to seagrasses.

Species Information

Five species of seagrasses and a common freshwater associate are found in the Big Bend (Zieman and Zieman, 1989; Mattson, 2000). The seagrasses are shoal grass (*Halodule wrightii*), turtle grass (*Thalassia testudinum*), star grass (*Halophila engelmannii*), paddle grass (*Halophila decipiens*), and manatee grass (*Syringodium filiforme*). The associate plant is wigeon grass (*Ruppia maritima*), a freshwater plant tolerant of salinity and usually included as one of the "seagrasses." Paddle grass is less common and is mainly confined to the offshore, deepwater areas greater than 10 m (33 ft) in depth (Continental Shelf Associates and Martel Laboratories, 1985; Iverson and Bittaker, 1986).

Strawn (1961), Zimmerman and Livingston (1976, 1979), and Mattson (1995) presented species-specific data on seagrass community characteristics in the region (standing crop, cover, and blade characteristics). Mattson (2000) summarized the results of these and other existing studies on seagrasses in the Big Bend. Inshore areas more influenced by freshwater inflow are more commonly vegetated with wigeon grass and shoal grass (fig. 5), with few or no benthic algal associate species (e.g., rhizophytic algae in the Bryopsidales order). Offshore regions are more commonly dominated by turtle grass and manatee grass (fig. 5), with numerous algal associates such as *Caulerpa* spp., *Udotea* spp., and *Halimeda* spp. and with shoal grass and star grass as associate seagrasses. Paddle grass and star grass are distributed across large areas in waters as deep as 20 m (66 ft) (Continental Shelf Associates and Martel Laboratories, 1985). Wigeon grass is generally most abundant adjacent to river mouths, where freshwater influence is high (Zimmerman and Livingston, 1979; Mattson, 1995). Although star grass can be abundant in deeper, offshore areas, it is also common in nearshore areas around river mouths, often in association with shoal grass and wigeon grass. Although star grass appears to prefer higher salinity areas (Mattson, 1995), this species is tolerant of the low light conditions that are present adjacent to some river mouths, enabling it to grow in these inshore areas. Overall, manatee grass and turtle grass, account for most of the total seagrass standing crop in the region (fig. 6) (Zimmerman and Livingston, 1976; Iverson and Bittaker, 1986), although at some locations, benthic algal standing crop may exceed that of seagrasses (Zimmerman and Livingston, 1979; Mattson, 1995).

The staff of the FDEP Big Bend Seagrasses Aquatic Preserve has been monitoring seagrasses at two locations in the region: St. Martins Keys (near Crystal River) and Deadman Bay (off the Steinhatchee River). The St. Martins Keys area is monitored twice annually and is characterized by dense mixed beds of seagrasses and benthic macroaglae; however because of the karstic nature of the region, the distribution of seagrasses is somewhat patchy within the sampling area. shoalgrass occurs more commonly among the nearshore sites, while turtle grass and manatee grass occur in mixed beds among the westerly sites. The Deadman Bay monitoring site is monitored annually and is characterized by dense contiguous seagrasses that do not extend as far offshore as those off St. Martins Keys. The distribution of species is similar to that of Citrus County, but species usually occur in monotypic stands and there is much less benthic macroalgal cover.

Highest seagrass standing crops and percent cover are generally observed in the summer (Zimmerman and Livingston, 1976; 1979; Mattson, 1995), although temporal studies of this type are few. A secondary peak in standing crop and cover is observed in the winter/early spring in some areas (Zimmerman and Livingston, 1976; Mattson, 1995, 2000). The same pattern generally holds true for the individual species of seagrasses, although some species occasionally exhibit high aboveground standing crop in the fall (Mattson, 1995). Turtle grass persisted as the dominant species, in terms of Braun-Blanquet abundance (a semiquantitative technique for assessing plant cover and abundance; for more information, see *http://chla.library.cornell.edu/cgi/t/text/text-idx?c=chla;idno=2917578*), over the past 5 yr of FDEP sampling in the St. Martins Keys area (fig. 7). Shoal grass and manatee grass have similar values for Braun-Blanquet abundance (fig. 7). Star grass occurs less frequently in this region, but this species where present is generally more abundant than either shoal grass or manatee grass. The data for the spring samples of turtle grass indicate that there has been a slight trend towards decreasing abundance from year to year (fig. 7). Other than these data, there are no evident trends or patterns either interannually or seasonally for these four seagrass species. For the sites in Deadman Bay, turtle grass and manatee grass codominate Braun-Blanquet abundance (fig. 8). Shoal grass and *Halophila* sp. are less dominant in the seagrass landscape there.

Based on field observations throughout the Big Bend region, turtle grass and manatee grass are known to flower (Zimmerman and Livingston, 1976; Mattson and Frazer, personal observations); less is known about seed production in the region. Phillips (1960b) wrote that the Anclote region was the northernmost limit of flowering and seed production for turtle grass on the Florida gulf coast, but seeds of this seagrass were observed during a period of sustained drought in the shallow waters adjacent to Keaton Beach (Taylor County) in summer 2000 (Mattson, pers. obs.), and flowers were observed in the estuarine waters between Steinhatchee and Homosassa during summer 2000 and 2001 (Frazer, pers. obs.). Year-old seedlings of turtle grass were found off the Econfina River in May 2002 by biologists with the Florida Marine Research Institute, and fruiting star grass was also collected there. During droughts, it is possible that high salinities promote seed production in turtle grass, even as far north as Taylor County. Manatee grass flowering was observed throughout the region from spring through fall in all years from 2000 to 2002, and star grass and wigeon grass flowers have been observed in the Crystal River-Homosassa River area (Frazer, pers. obs.).

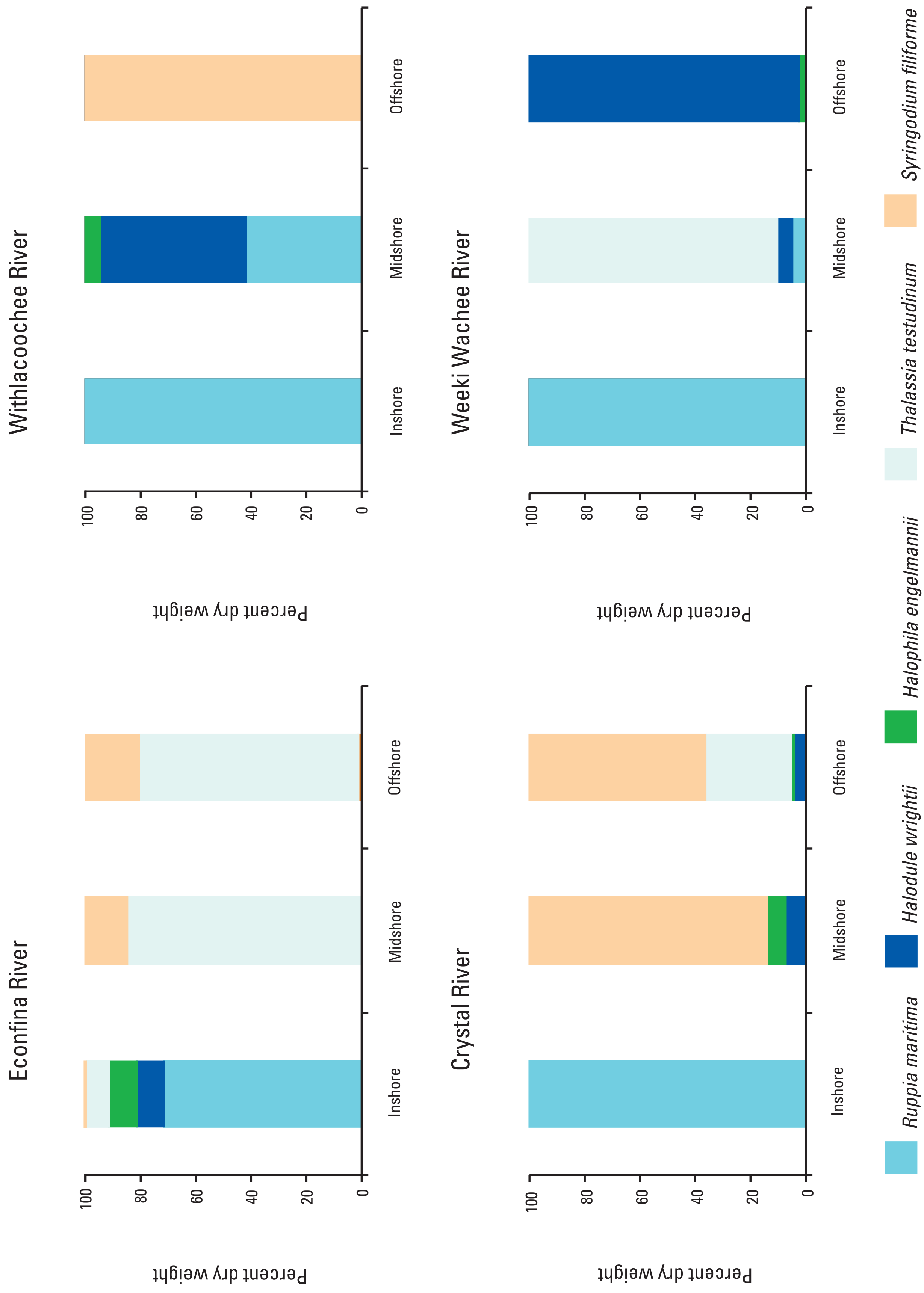


Figure 5. Seagrass species composition (as percent of the total dry weight standing crop) at four Big Bend river estuaries. Econfina River data are from Zimmerman and Livingston (1979); the remaining data are from Mattson (1995).

Monitoring for Seagrass Health

Recent efforts are beginning to address the need to better assess seagrass resources in the Big Bend. Field surveys and mapping are being conducted by the University of Florida and SWFWMD to assess seagrass resources in the Springs Coast portion of the Big Bend. The Suwannee River Water Management District (SRWMD)—in partnership with the U.S. Environmental Protection Agency (EPA) Gulf of Mexico Program, the Florida Marine Research Institute, and the USFWS Lower Suwannee and Cedar Keys National Wildlife Refuges—is beginning to map and assess seagrasses in the Big Bend Proper. These parallel programs need to coordinate activities as best as possible in order to generate comparable data.

Field-based methods to assess seagrass health have yet to be developed. The use of biological assessment methods to generate indices of ecological integrity or health is well established in river systems (e.g., Davis and Simon, 1995). Similar tools need to be developed for estuarine ecosystems, including seagrass indices or "metrics" of seagrass health or condition. One example of such a tool is the plastichron interval (Duarte and others, 1994). Carlson and others (2003) evaluated various seagrass metrics (morphological and chemical) to evaluate the effects of water quality changes related to El Niño and the Southern Oscillation. These metrics may have utility for general assessment of seagrass health.

Available data for the Big Bend indicate that various seagrass community characteristics are comparable to those of other areas of Florida and the Gulf of Mexico (Zieman and Zieman, 1989; Mattson, 2000). Standing crop values for turtle grass, shoal grass, and manatee grass are similar to those reported for Texas and are generally in the middle to lower portion of the range of values from Florida and the Caribbean. These values might be expected, since the Big Bend (and Texas) seagrasses are near the northern limit of their geographic range. Iverson and Bittaker (1986) reported higher shoot densities in Florida Bay than in the Big Bend, and Mattson (2000) indicated a trend of roughly increasing shoot densities from north to south in the Big Bend. Epiphyte loads on seagrasses in the region are only now beginning to be quantitatively assessed. Overall, metrics such as standing crop and shoot density of Big Bend seagrasses are comparable to those reported for similar, relatively undisturbed seagrass areas in the Gulf of Mexico.

Mapping and Monitoring Needs

One of the most important needs for proper management of seagrass resources in the Big Bend is to implement a sustained program of mapping and monitoring to assess status and trends in seagrass abundance and distribution. Seagrasses are often touted as good overall indicators of coastal water quality (Dennison and others, 1993; Kemp, 2000) because a variety of water-quality variables affect water clarity, which in turn influences seagrasses. Recent monitoring efforts begun by the SWFWMD, SRWMD, University of Florida, FMRI, and USGS are beginning to address the need for data; however, these efforts need future funding in order to

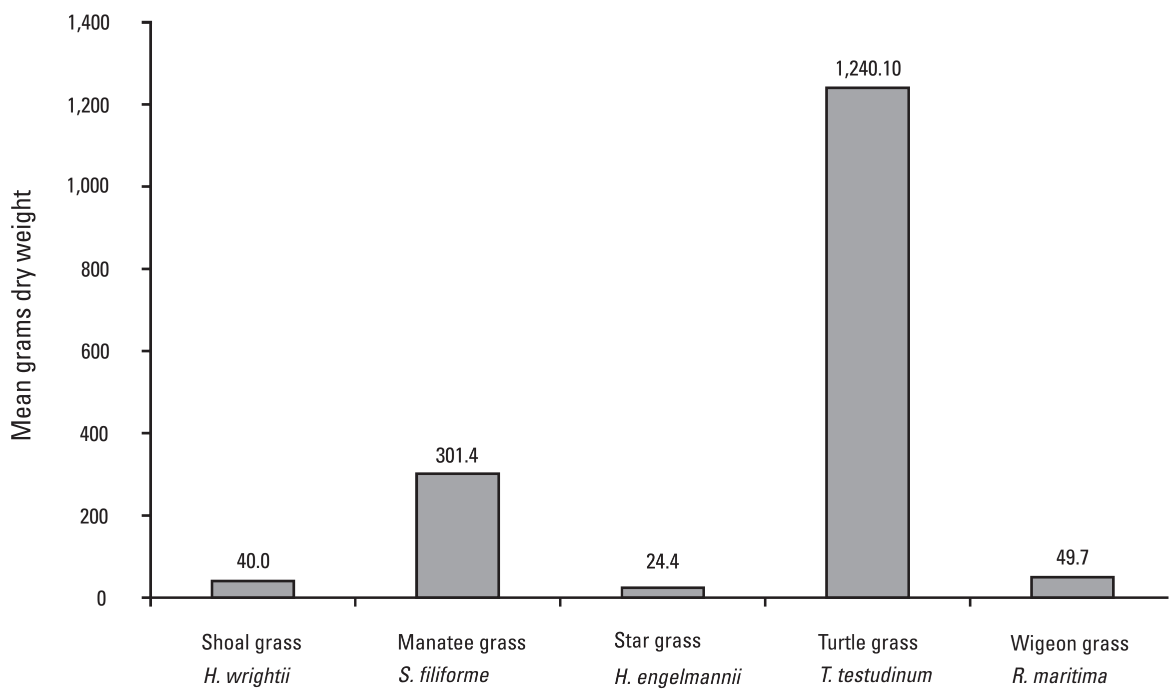


Figure 6. Total seagrass standing crop (aboveground and belowground) by species in Apalachee Bay. Data are from Zimmerman and Livingston (1976).

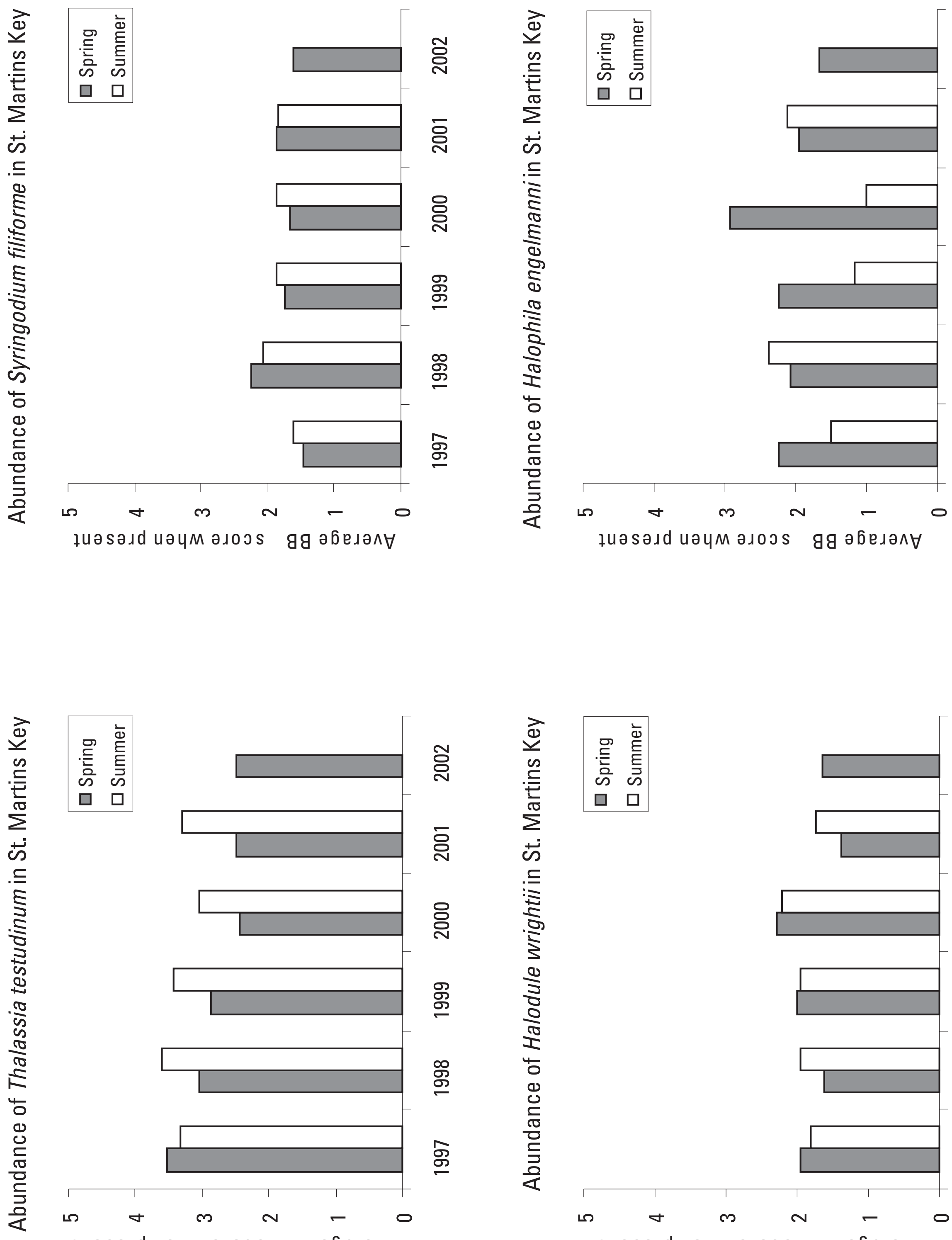
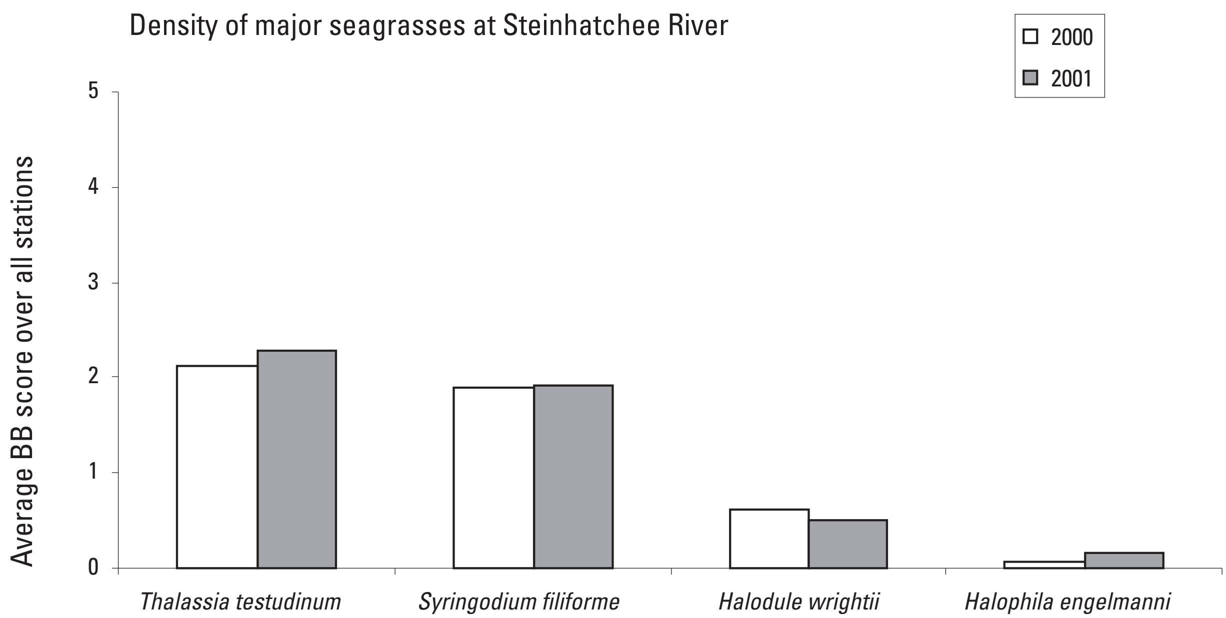


Figure 7. Braun-Blanquet abundance (BB) scores by seagrass species from a Florida Department of Environmental Protection monitoring site in the St. Martins Key area.

A



B

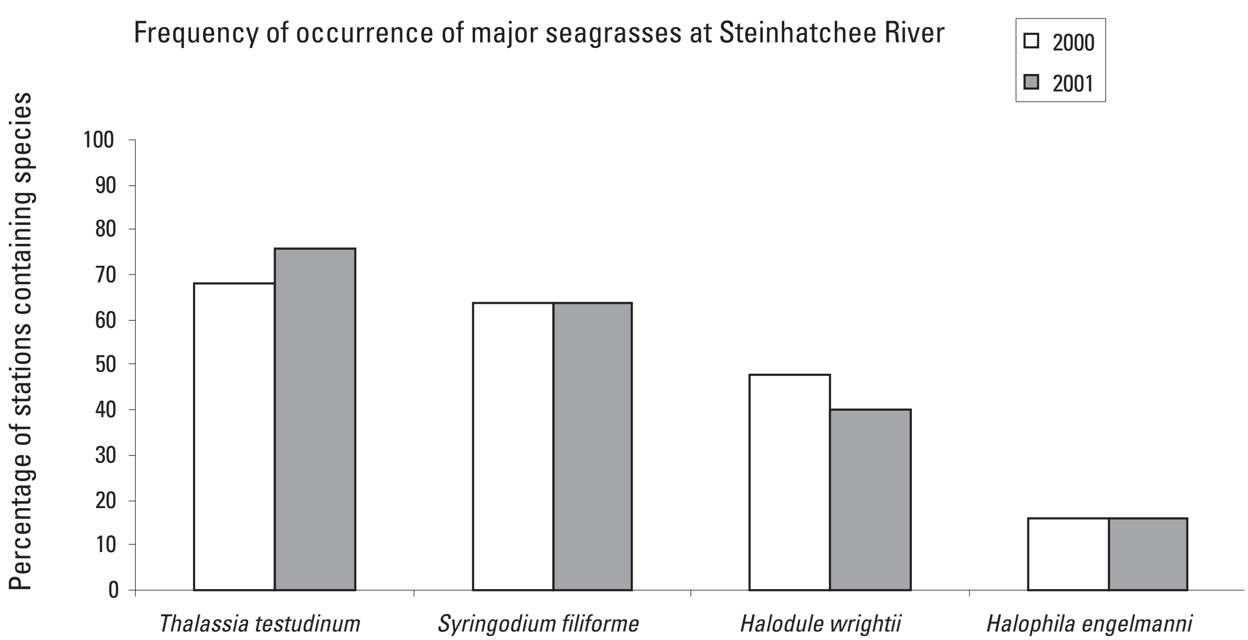


Figure 8. Braun-Blanquet (BB) abundance (*A*) and frequency of occurrence (*B*) by species of seagrass in Deadmans Bay off the Steinhatchee River. Data are from Florida Department of Environmental Protection annual surveys.

continue. The sheer size and extensiveness of the Big Bend seagrass ecosystem make mapping and monitoring by current conventional methods timeconsuming and costly. The use of alternative technologies, such as acoustic doppler surveys and multispectral remote sensing imagery, warrants investigation. The effectiveness of existing regulatory programs and ongoing basin management efforts in the region cannot be determined without an ongoing monitoring program.

A second important need is to conduct applied research to determine the water-quality thresholds which must be met to ensure that adequate light penetrates the water column to support seagrass growth. The deeper depths to which seagrasses grow in the Big Bend suggest that even subtle water-quality degradation may result in landward migration of the offshore edge of the bed or alteration in species composition of the deep beds.

Restoration and Enhancement Opportunities

With perhaps one exception (the Fenholloway River estuary), the major seagrass management issue in the Big Bend is protection, not restoration. Nowhere else in Florida, or perhaps even the entire Gulf of Mexico, is the opportunity greater to protect, not restore, an area. Despite the lack of recent data on the status of seagrasses in the region, the consensus among resource managers is that the system retains a high degree of ecological integrity. The investment of public and private funds to monitor and map seagrasses in the Big Bend and conduct applied research is imperative as part of an overall program of seagrass preservation in the region. The historical loss of seagrasses in the Fenholloway River estuary represents the one known restoration opportunity in the region. Federal and State regulatory agencies are working with the Buckeye Florida, L.P., pulp mill in Taylor County to improve the quality of their effluent so as to allow seagrass recovery in the estuary. Similar to efforts elsewhere in Florida (i.e., Tampa Bay and Indian River Lagoon), it is hoped that natural recovery of seagrasses will occur following improvement in water quality. Some seagrass recovery was observed after an initial pollution abatement effort in the 1970s (Livingston, 1984a). If natural recovery efforts prove to be inadequate, then planting efforts will have to be undertaken to restore seagrass cover.

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Tampa Bay and Saint Joseph Sound

By David A. Tomasko[1] and Holly S. Greening[2]

Background

Nearly 2 million people live in the Tampa Bay region, and the bay is the largest estuary in the State of Florida. The bay is home to the third largest port in the United States (in terms of domestic tonnage), and nearly 100,000 recreational boats are registered to owners in the counties surrounding the bay. Tampa Bay is also home to approximately 40,000 breeding pairs of shorebirds and over 100 bottlenose dolphins (*Tursiops truncatus*) (Tampa Bay National Estuary Program, 1996).

In the years following World War II, the human population in the Tampa Bay region increased at a rapid rate. Accompanying this population growth was a significant loss of native upland, intertidal, and subtidal plant communities. Perhaps half of the bay's seagrass meadows were lost between 1950 and 1982 because of combined impacts from dredge-and-fill activities and degraded water quality (Tampa Bay Estuary Program, 1996). The loss of seagrass acreage is thought to be at least partially responsible for declines in commercial and recreational fisheries for various species of finfish and shellfish.

In southwest Florida, seagrass meadows have been the focus of a significant amount of research on the relationships between pollutant loads, water quality, and seagrass health. In Tampa Bay, historical losses of seagrass coverage have been linked to both direct and indirect impacts (Lewis and others, 1991; Haddad, 1989; Lewis, 1989). In contrast, recent increases in seagrass coverage have been linked to improved water quality, which has in turn been linked to reductions in nitrogen loads caused by humans (Johansson, 1991; Avery, 1997; Johansson and Ries, 1997; Johansson and Greening, 2000). Recent improvements in the treatment and disposal of wastewater discharges by the city of Tampa, the city of St. Petersburg, and the city of Clearwater have been identified as major causes of improved water quality in the bay (Tampa Bay Estuary Program, 1996).

To the north and west of Tampa Bay are the areas of Clearwater Harbor and Saint Joseph Sound (fig. 1). While adjacent to Tampa Bay, Clearwater Harbor and Saint Joseph Sound have not yet benefited from a detailed examination of pollutant loads, water quality, and seagrass coverage as has Tampa Bay. Figure 1 shows the watershed for both Tampa Bay and the Saint Joseph Sound and Clearwater Harbor region.

Scope of Area

Tampa Bay is divided into seven bay segments: (1) Hillsborough Bay, (2) Old Tampa Bay, (3) Middle Tampa Bay, (4) Lower Tampa Bay, (5) Boca Ciega Bay, (6) Terra Ceia Bay, and (7) the Manatee River (fig. 2). These subareas have all been consistently included in seagrass mapping efforts dating back to the 1950s.

For Clearwater Harbor and Saint Joseph Sound, 1996 photography covered an area between The Narrows (27° 52.7' N. latitude) and Three Rooker Bar (28° 07.4' N. latitude). For 1999 and 2002 (the time period considered here), the area photographed was expanded northward up to Anclote Key (28° 10.7' N. latitude).

Methodology Employed To Determine and Document Current Status

Seagrass mapping efforts have played an important role in measuring the success, or lack thereof, made toward maintaining and expanding upon improvements to water quality in Tampa Bay and other estuaries. These mapping efforts are conducted on a typically biennial basis by the Southwest Florida Water Management District (SWFWMD) to fulfill its obligations under the Comprehensive Conservation and Management Plan created by the Tampa Bay National Estuary Program (1996). Seagrass maps are available for Tampa Bay for the years 1988 (fig. 3), 1990 (fig. 4), 1992 (fig. 5), 1994 (fig. 6), 1996 (fig. 7), 1999 (fig. 8), and 2002 (fig. 9). In addition, data from previous efforts are available for 1950 (Tampa Bay Regional Planning Council, 1986) and 1982 (Haddad, 1989) (see table 1). For Clearwater Harbor and Saint Joseph Sound, mapping efforts have been completed for the years 1996 and 1999 (figs. 7 and 8).

Seagrass maps were produced through multiple steps. First, aerial photography was obtained, usually in late fall to

[1] PBS&J.

[2] Tampa Bay Estuary Program.

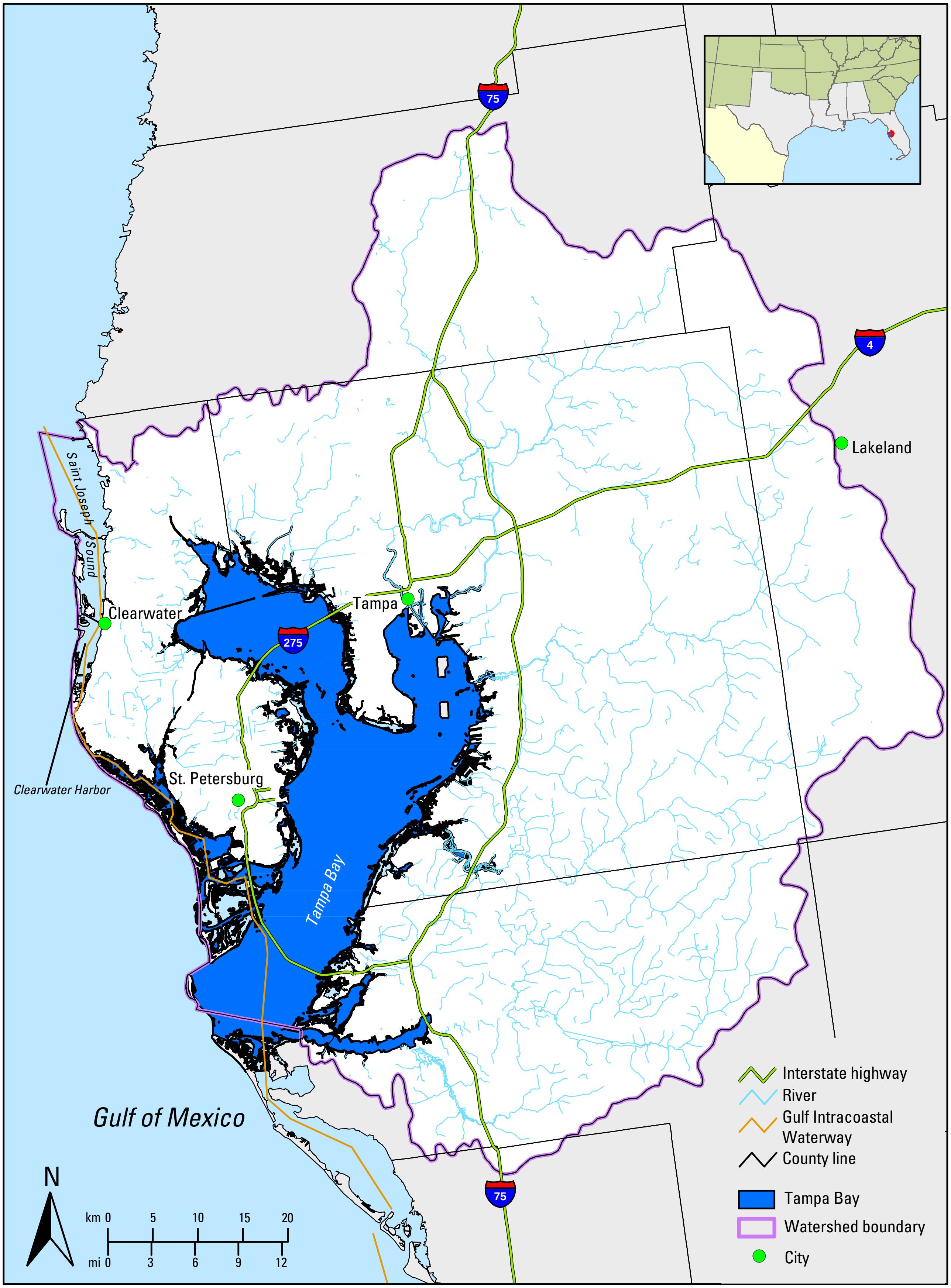

Figure 1. Watershed for Tampa Bay and Saint Joseph Sound.

early winter. This time of year is associated with both good water clarity and relatively high seagrass biomass.

Second, photointerpretation efforts were conducted in the field to allow for the successful evaluation of distinct photographic signatures. Seagrass signatures are divided into two classes (continuous and patchy), with a minimum mapping unit of 0.2 ha (0.5 acres).

Third, polygons were integrated into an ArcInfo program. For some past efforts (i.e., 1988, 1990, 1992, 1994, and 1996; figs. 3–7), individual polygons were delineated onto Mylar® overlays, cartographically transferred by using a zoom transfer scope to U.S. Geological Survey quadrangles, and then digitally transferred to an ArcInfo database for further characterization. These techniques allowed for the seagrass maps to meet U.S. National Map Accuracy Standards for 1:24,000-scale maps. For 1999 and 2002 seagrass maps (figs. 8 and 9), the 1:12,000-scale U.S. National Map Accuracy Standards were met. While photography remained at a scale of 1:24,000, the higher positional accuracy standard required the use of tighter ground control and more sophisticated mapping techniques. Analytical stereo plotters were used for photointerpretation rather than traditional stereoscopes. This technique allowed for the production of a georeferenced digital file of the photointerpreted images without the need for additional photograph-to-map transfer. Rather than redrawing seagrass coverage polygons for each mapping year's efforts, the previous efforts' digital coverages were used as the baselines, and only the changes in seagrass coverage were mapped. Areas with no change between efforts were coincident with the earlier effort's coverage.

Fourth, hard copies of plots were made of photointerpreted seagrass coverage, and 60 randomly chosen points were identified for a classification accuracy assessment of the finished map. A hand-held Global Positioning System unit was used, along with the map and the latitude and longitude of the randomly located stations, to develop an unbiased determination of the map's classification accuracy. A 90% classification accuracy standard is required for these efforts, and 96% accuracy was achieved for 1999 efforts (i.e., 53 of 55 stations that could be visited had been accurately described on the maps).

Methodology Employed To Analyze Historical Trends

To determine seagrass coverage for historical conditions in Tampa Bay (table 1), the Tampa Bay Regional Planning Council (1986) used 1:24,000-scale, natural color aerial photographs. The resulting seagrass maps were digitized by the Florida Marine Research Institute, converted from raster to vector format, and horizontally georeferenced with available ground controls. This mapping effort met U.S. National Map Accuracy Standards for 1:24,000-scale maps. No groundtruthing was conducted concurrent with the photography for this time period. Coverage was simply classified as polygons with or without seagrass coverage; the classification system did not distinguish between patchy and continuous coverage.

Seagrass coverage estimates for 1982 were also available for Tampa Bay (table 1). These estimates are from a joint project between the U.S. Fish and Wildlife Service (USFWS) and the Florida Department of Environmental Protection. This effort digitized existing 1:24,000-scale natural color aerial photography that had been delineated according to the USFWS National Wetlands Inventory standard classification system. This mapping effort met U.S. National Map Accuracy Standards for 1:24,000-scale maps. As with the 1950 photography, a statistically relevant groundtruthing effort was not conducted concurrent with the acquisition of photography. The classification system did not distinguish between patchy and continuous coverage.

All historical mapping efforts are available in digital format through either the SWFWMD or the Tampa Bay National Estuary Program.

Status and Trends

Tampa Bay

Across the entire Tampa Bay study area, the majority of coverage is in the higher salinity portions of the lower bay, with the most extensive seagrass meadows being found in the vicinity of Fort Desoto County Park, just west of the northern end of the Sunshine Skyway. Extensive seagrass meadows are also found on the southeastern shore of Tampa Bay, from Anna Maria Island up to the Little Manatee River. Significant coverage is found in Old Tampa Bay, particularly in the eastern portion adjacent to Interbay Peninsula. Coverage in Hillsborough Bay is much less extensive than in other portions of the bay.

Tables 1 and 2 show the overall trend in seagrass coverage for Tampa Bay from 1950 to 2002. In 1950, seagrass meadows covered 16,350 ha (40,401 acres) of bay bottom. By 1982, that number had dropped to 8,763 ha (21,653 acres). From 1982 to 1996, acreage increased by 2,130 ha (5,263 acres) to 10,893 ha (26,917 acres) total. The average rate of increase between 1982 and 1996 was 152 ha (376 acres) per year. From 1988 to 1996, the average rate of increase was 184 ha (455 acres) per year. From 1996 to 1999, there was a decrease in coverage of 840 ha (2,076 acres) to 10,053 ha (24,841 acres) total. From 1999 to 2002, there was an increase in coverage of 501 ha (1,238 acres) to 10,554 ha (26,079 acres) total. Seagrass coverage in Tampa Bay in 2002 was 65% of the 1950 values.

For individual segments of the bay, table 1 can be used, in addition to the text below, to assess changes in coverage on a segment-by-segment basis over time.

Figure 2. Scope of area for the Tampa Bay and Saint Joseph Sound vignette.

Hillsborough Bay

In Hillsborough Bay, seagrass coverage dropped from 931 ha (2,301 acres) in 1950 to a complete absence in 1982. Coverage in 2002 was 194 ha (479 acres), a rate of increase of about 10 ha (25 acres) per year between 1982 and 2002. Seagrass coverage in Hillsborough Bay in 2002 was 21% of the 1950 values (table 1).

Old Tampa Bay

In Old Tampa Bay, seagrass coverage declined from 4,330 ha (10,700 acres) in 1950 to 2,026 ha (5,006 acres) in 1988. From 1988 to 1994, coverage increased at a rate of about 61 ha (151 acres) per year to 2,392 ha (5,911 acres) total. From 1994 to 1996, coverage decreased to 2,332 ha (5,762 acres), a 30 ha (74 acre) per year decline. From 1996 to 1999, coverage again decreased, this time to 1,779 ha (4,396 acres), a rate of decline of about 185 ha (457 acres) per year. Coverage in 2002 increased by 355 ha (877 acres) to 2,134 ha (5,273 acres) total. Seagrass coverage in Old Tampa Bay in 2002 was 49% of the 1950 values (table 1).

Middle Tampa Bay

In the middle Tampa Bay subarea, seagrass coverage declined from 3,885 ha (9,600 acres) in 1950 to 1,636 ha (4,043 acres) in 1982. From 1982 to 2002, seagrass coverage increased by 680 ha (1,680 acres), a 42% increase. Seagrass coverage in middle Tampa Bay in 2002 was 60% of the 1950 values (table 1).

Lower Tampa Bay

Relative to other areas of Tampa Bay, portions of the lower bay appear to exhibit a more optimistic picture of seagrass recovery. In lower Tampa Bay, seagrass coverage in 1950 was 2,469 ha (6,101 acres). Seagrass coverage in lower Tampa Bay in 2002 was 2,271 ha (5,612 acres), or 92% of the 1950 values (table 1).

Boca Ciega Bay

In Boca Ciega Bay, seagrass coverage declined from 4,371 ha (10,801 acres) in 1950 to 2,335 ha (5,770 acres) in 1982. From 1982 to 2002, seagrass coverage increased at a rate of 39 ha (96 acres) per year. Seagrass coverage in Boca Ciega Bay in 2002 was 3,105 ha (7,673 acres), or 71% of the 1950 values (table 1).

Terra Ceia Bay

In Terra Ceia Bay, seagrass coverage remained similar between 1950 and 1982 (283 ha (699 acres) and 304 ha (751 acres), respectively); however, 1988 coverage was 383 ha (946 acres), a 26% increase from 1982. In 2002, seagrass coverage in Terra Ceia Bay was 380 ha (939 acres), or 34% higher than in 1950 (table 1).

Manatee River

In the Manatee River, seagrass coverage declined between 1950 and 1982 (81 ha (200 acres) and 53 ha (131 acres), respectively); however, 1988 seagrass coverage was 140 ha (346 acres), a 165% increase from 1982. In 2002, seagrass coverage in the Manatee River was 154 ha (381 acres), or 91% higher than in 1950 (table 1).

Clearwater Harbor and Saint Joseph Sound

For Clearwater Harbor and Saint Joseph Sound, figure 9 shows the location of seagrass coverage in 2002. To the south, in Clearwater Harbor, seagrass coverage diminished compared to the wider areas of Saint Joseph Sound to the north. The deep edges of seagrass meadows extend farther offshore to the north, which seems to correspond to the perceived improvement in water quality in the northern portion of the mapped area. Unfortunately, water-quality monitoring programs are not nearly as well developed in Clearwater Harbor and Saint Joseph Sound as they are in Tampa Bay.

Between 1999 and 2002, seagrass coverage in Clearwater Harbor and Saint Joseph Sound decreased from 5,958 ha (14,722 acres) to 5,713 ha (14,117 acres), a decrease of 245 ha (605 acres). Seagrass coverage in Clearwater Harbor and Saint Joseph Sound in 2002 was 96% of the 1999 coverage.

Causes of Change

Historical losses of seagrass coverage in Tampa Bay (i.e., between 1950 and 1982; table 1) are thought to be due to direct and indirect impacts associated with rapid human population growth in the watershed in the post-World War II years (Tampa Bay Estuary Program, 1996). Direct-impact losses occurred because of dredge-and-fill activities associated with waterfront development for residential and commercial land uses. Indirect-impact losses are thought to be associated with increased point and nonpoint source nutrient loads that accompanied the population growth and urbanization of the watershed.

In contrast, the overall trend of increasing seagrass coverage in Tampa Bay from 1982 to 2002 is related to increases in water clarity during that same time period (Johansson, 1991; Johansson and Ries, 1997; Johansson and Greening, 2000). Increased water clarity, in turn, appears to be related to decreased phytoplankton populations (Johansson, 1991; Janicki and Wade, 1996). Finally, the reduction in phytoplankton levels is thought to be related to the approximately 61% decline in nitrogen loads coming into Tampa Bay between 1976 and 1992–94 (Tampa Bay Estuary Program, 1996).

Seagrass planting efforts appear to have played a minor role in bringing about the sustained increases in acreage in Tampa Bay, as most of the areas where seagrass increases have occurred are in parts of the bay where no transplanting efforts have been undertaken (Tomasko, personal observation). Also, seagrass transplanting efforts have usually been on the level of 1 ha of effort (usually less), whereas increases in seagrass coverage averaged 184 ha (455 acres) per year between 1988 and 1996 (figs. 3–7; table 1).

In Clearwater Harbor and Saint Joseph Sound, systemwide water-quality monitoring programs are just beginning, and information on trends in water quality is not yet available. Furthermore, watershed-level pollutant loading models for the watershed scale have yet to be derived for this region.

Monitoring for Seagrass Health

In addition to aerial photography for estimating seagrass acreage, a series of fixed transects has been established throughout Tampa Bay. A similar monitoring effort is underway in Clearwater Harbor and the southern portion of Saint Joseph Sound. Approximately 60 transects are placed throughout Tampa Bay, and they are revisited every October. At each transect, seagrass cover is estimated for each species by using the Braun-Blanquet method (for more information, see *http://chla.library.cornell.edu/cgi/t/text/text-idx?c=chla;idno=2917578*). Also, the maximum distance offshore where seagrasses are found is noted, as is the relative water depth at each station. Year-to-year comparisons are thus possible for examining trends in seagrass meadow composition for individual transects. Periodic reports on these findings are produced by the city of Tampa (e.g., Avery, 2000) and provide useful groundtruthed information on seagrass health throughout the bay.

The transect-based monitoring effort has provided information useful for interpreting and verifying results from seagrass mapping efforts. In Old Tampa Bay, where mapping efforts concluded that seagrass coverage declined by 24% between 1996 and 1999 (figs. 7 and 8; table 1), four of eight transects showed evidence of a shoreward migration of the offshore edge of the seagrass meadow between 1998 and 1999 (Avery, 2000). In Boca Ciega Bay, where seagrass coverage declined only 3% between 1996 and 1999 (table 1), only 1 of 10 transects showed a shoreward retreat of the offshore edge of the meadow between 1998 and 1999 (Avery, 2000).

Table 1. Seagrass coverage in hectares (acres) by year for segments of Tampa Bay.

[HB = Hillsborough Bay, OTB = Old Tampa Bay, MTB = Middle Tampa Bay, LTB = Lower Tampa Bay, BCB = Boca Ci ega Bay, TCB = Terra Ceia Bay, MR = Manatee River]

	1950	1982	1988	1990	1992	1994	1996	1999	2002
HB	931 (2,301)		3 (7)	19 (47)	19 (47)	59 (146)	78 (193)	78 (193)	194 (479)
OTB	4,330 (10,699)	2,405 (5,943)	2,026 (5,006)	2,251 (5,562)	2,378 (5,876)	2,392 (5,911)	2,332 (5,762)	1,779 (4,396)	2,134 (5,273)
MTB	3,885 (9,600)	1,636 (4,043)	2,106 (5,204)	2,148 (5,308)	2,133 (5,271)	2,337 (5,775)	2,242 (5,540)	2,282 (5,639)	2,316 (5,723)
LTB	2,469 (6,101)	2,030 (5,016)	2,232 (5,515)	2,486 (6,143)	2,526 (6,242)	2,511 (6,205)	2,582 (6,380)	2,366 (5,846)	2,271 (5,612)
BCB	4,371 (10,801)	2,335 (5,770)	2,533 (6,259)	2,754 (6,805)	2,813 (6,951)	2,880 (7,116)	3,116 (7,700)	3,021 (7,465)	3,105 (7,673)
TCB	283 (699)	304 (751)	383 (946)	405 (1,001)	406 (1,003)	404 (998)	394 (974)	376 (929)	380 (939)
MR	81 (200)	53 (131)	140 (346)	147 (363)	147 (363)	148 (366)	148 (366)	152 (376)	154 (381)
Total	16,350 (40,401)	8,763 (21,653)	9,423 (23,284)	10,209 (25,226)	10,422 (25,753)	10,732 (26,519)	10,893 (26,917)	10,053 (24,841)	10,554 (26,079)

Species Information

Besides estimating acreage, the transect-based monitoring effort also allows for determination of the species composition of seagrass meadows throughout Tampa Bay. It was found that shoal grass (*Halodule wrightii*) was distributed throughout the bay, whereas wigeon grass (*Ruppia maritima*) was only encountered at transects located in Hillsborough Bay and Old Tampa Bay. Manatee grass (*Syringodium filiforme*) and turtle grass (*Thalassia testudinum*) were found in all portions of Tampa Bay except Hillsborough Bay (although Avery (2000) encountered turtle grass along a transect at Wolf Branch, on the border between Hillsborough Bay and Middle Tampa Bay). Star grass (*Halophila engelmannii*) was found along transects in lower Tampa Bay and Old Tampa Bay. In an earlier assessment, Lewis and Phillips (1980) found turtle grass and shoal grass to be equally abundant throughout the bay, with manatee grass and wigeon grass in lesser amounts. The least commonly encountered seagrass species was star grass.

In Clearwater Harbor, only shoal grass and turtle grass were encountered along the monitored transects (Avery, 2000). In the southern portion of Saint Joseph Sound, shoal grass, turtle grass, and manatee grass were all encountered.

Restoration and Enhancement Opportunities

Restoration and enhancement efforts for seagrasses in Tampa Bay are focused on implementation of a nitrogen-management strategy developed by the partners in the Tampa Bay Estuary Program. Efforts to reduce seagrass scarring by boat propellers have also been important elements in limited, but heavily impacted, areas of the bay, as summarized here.

The Tampa Bay Nitrogen Management Strategy

In 1990, Tampa Bay was accepted into the U.S. Environmental Protection Agency's National Estuary Program (TBNEP). The TBNEP, a partnership that includes three regulatory agencies and six local governments, has built on the resource-based approach initiated by earlier bay management efforts. Further, TBNEP has developed water-quality models (Zarbock and others, 1994; Janicki and Wade, 1996; Martin and others, 1996; Zarbock and others, 1996; Morrison and others, 1997; Wang and others, 1999) to quantify linkages between nitrogen loadings and bay water quality, as well as models that link loadings and water quality to seagrass goals (Squires and others, 1996; Janicki and Wade, 1996; Johansson and Greening, 2000).

The establishment of clearly defined and measurable goals is crucial for a successful resource management effort. In 1996, the TBNEP adopted a baywide minimum seagrass goal of 15,379 ha (38,002 acres). This goal represented 95% of the estimated 1950 seagrass cover (minus the nonrestorable areas) and included the protection of the existing 10,053 ha (24,841 acres) plus the restoration of an additional 5,325 ha (13,158 acres) (Tampa Bay National Estuary Program, 1996).

Recent research indicates that the deep edges of turtle grass meadows, the primary seagrass species for which nitrogen-loading targets are being set, correspond to the depth at which 20.5% of subsurface irradiance (the light that penetrates the water surface) reaches the bay bottom on an average annual basis (Dixon and Leverone, 1995). Water clarity and light penetration in Tampa Bay are affected by a number of factors such as phytoplankton biomass, nonphytoplankton turbidity, and water color. Janicki and Wade (1996) used regression analyses, based on long-term data provided by the Environmental Protection Commission of Hillsborough County, to develop an empirical model describing water-clarity variations in the four largest bay segments. Results of the modeling effort indicated that, on a baywide basis, variation in chlorophyll a concentration is the major factor affecting variation in average annual water clarity. The empirical regression model was used to estimate chlorophyll a concentrations necessary to maintain water clarity needed for seagrass growth for each major bay subarea. The adopted targets of average annual chlorophyll a (which are specific to each segment: 8.5 µg/L for Old Tampa Bay, 13.2 µg/L for Hillsborough Bay, 7.4 µg/L for middle Tampa Bay, and 4.6 µg/L for Lower Tampa Bay) are easily measured and tracked through time and are used as intermediate measures for assessing success in maintaining water-quality requirements necessary to meet the long-term seagrass goal (Johansson and Greening, 2000; Tomasko and others, 2005).

Water-quality conditions in 1992–94 appeared to allow an annual average of more than 20.5% of subsurface irradiance to reach target depths (i.e., the depths to which seagrasses grew in 1950). Thus, a management strategy based on maintaining the 1992–94 nitrogen-loading rates should be adequate to achieve the seagrass restoration goals. This maintenance approach, combined with careful monitoring of water quality

Table 2. Tampa Bay seagrass coverage from 1950 to 2002.

Year	Hectares	Acres
1950	16,350	40,401
1982	8,763	21,653
1988	9,423	23,284
1990	10,209	25,226
1992	10,422	25,753
1994	10,732	26,519
1996	10,893	26,917
1999	10,053	24,841
2002	10,554	26,079

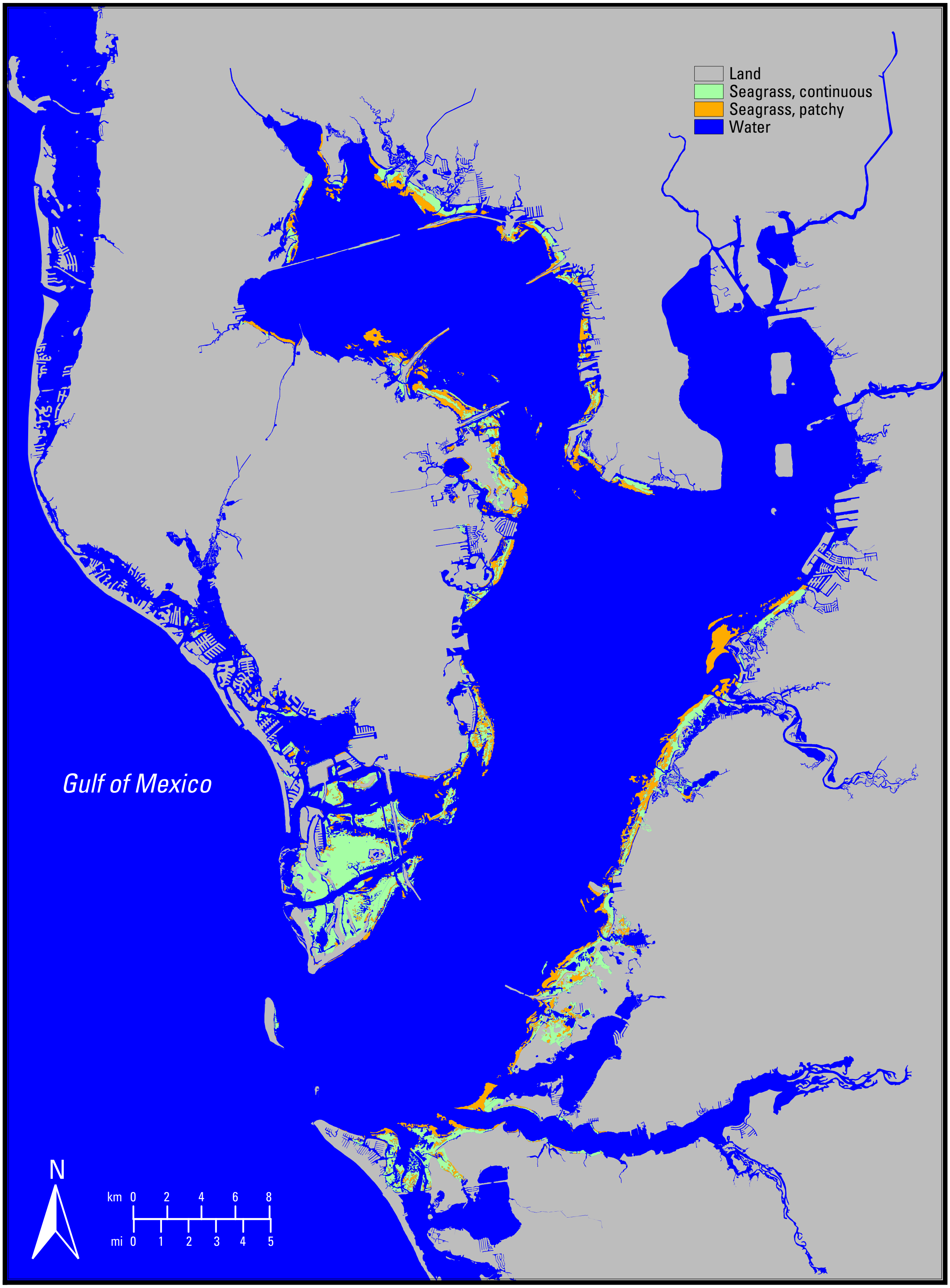

Figure 3. Distribution of seagrass in Tampa Bay, 1988.

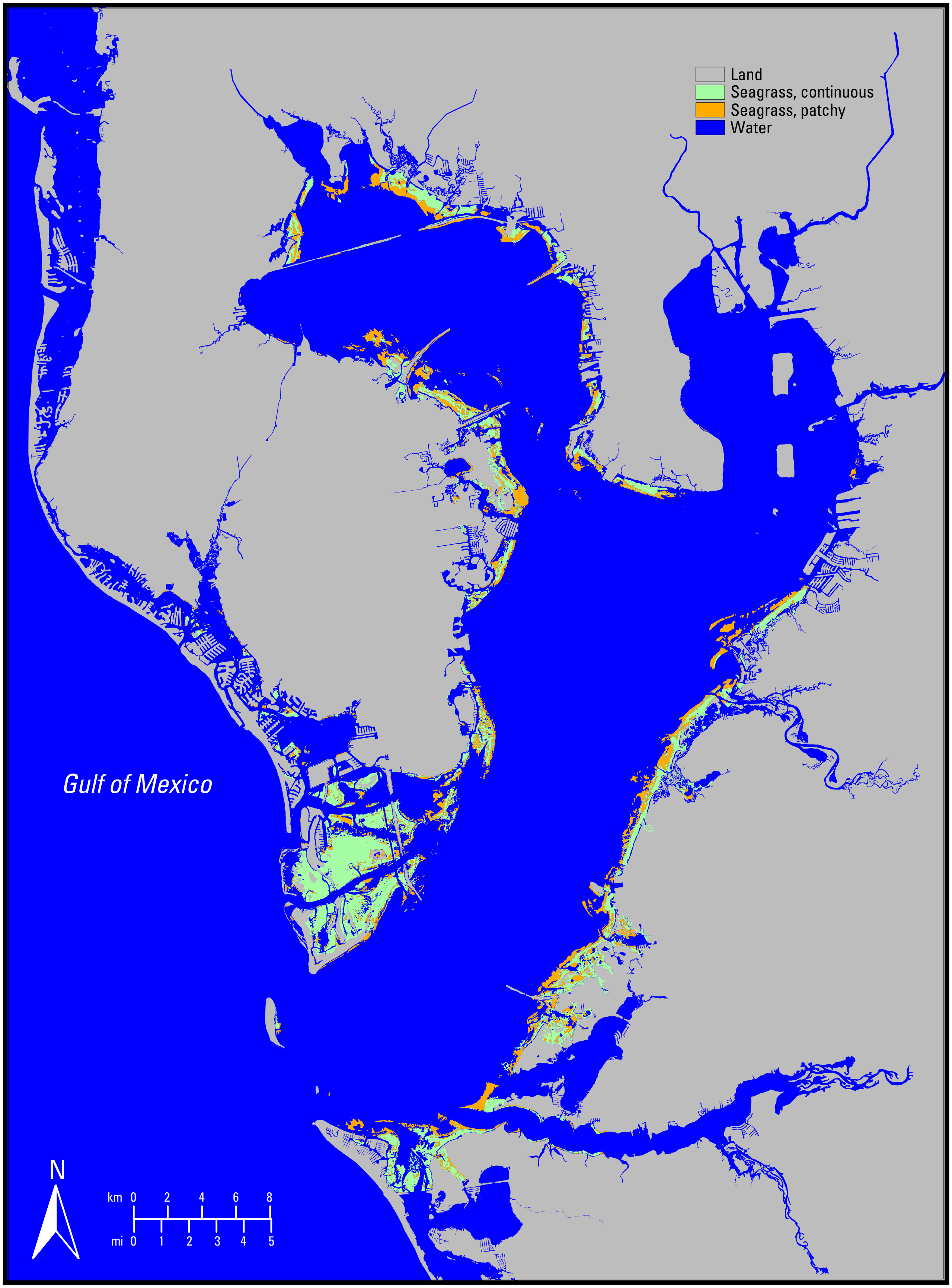

Figure 4. Distribution of seagrass in Tampa Bay, 1990.

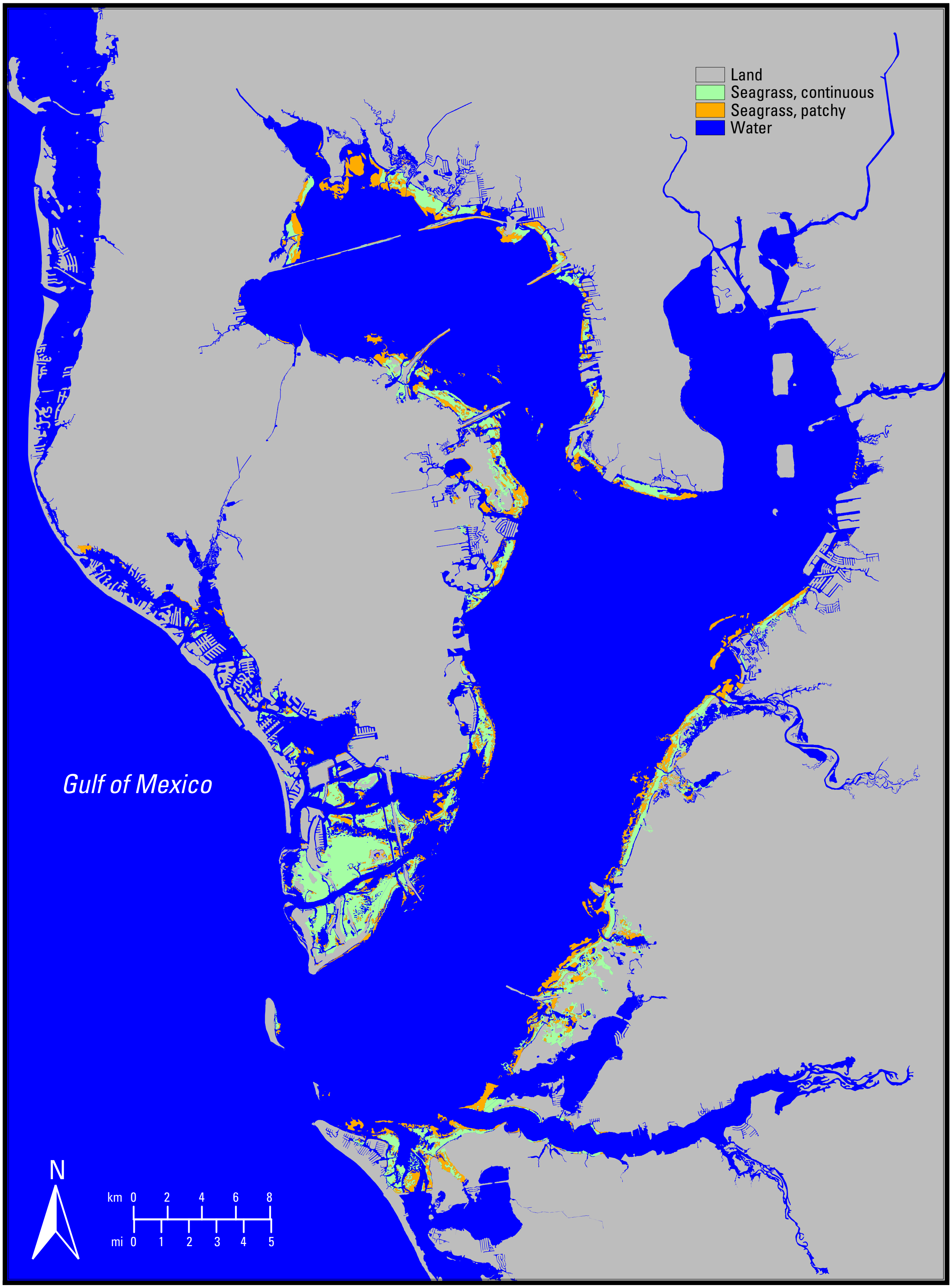

Figure 5. Distribution of seagrass in Tampa Bay, 1992.

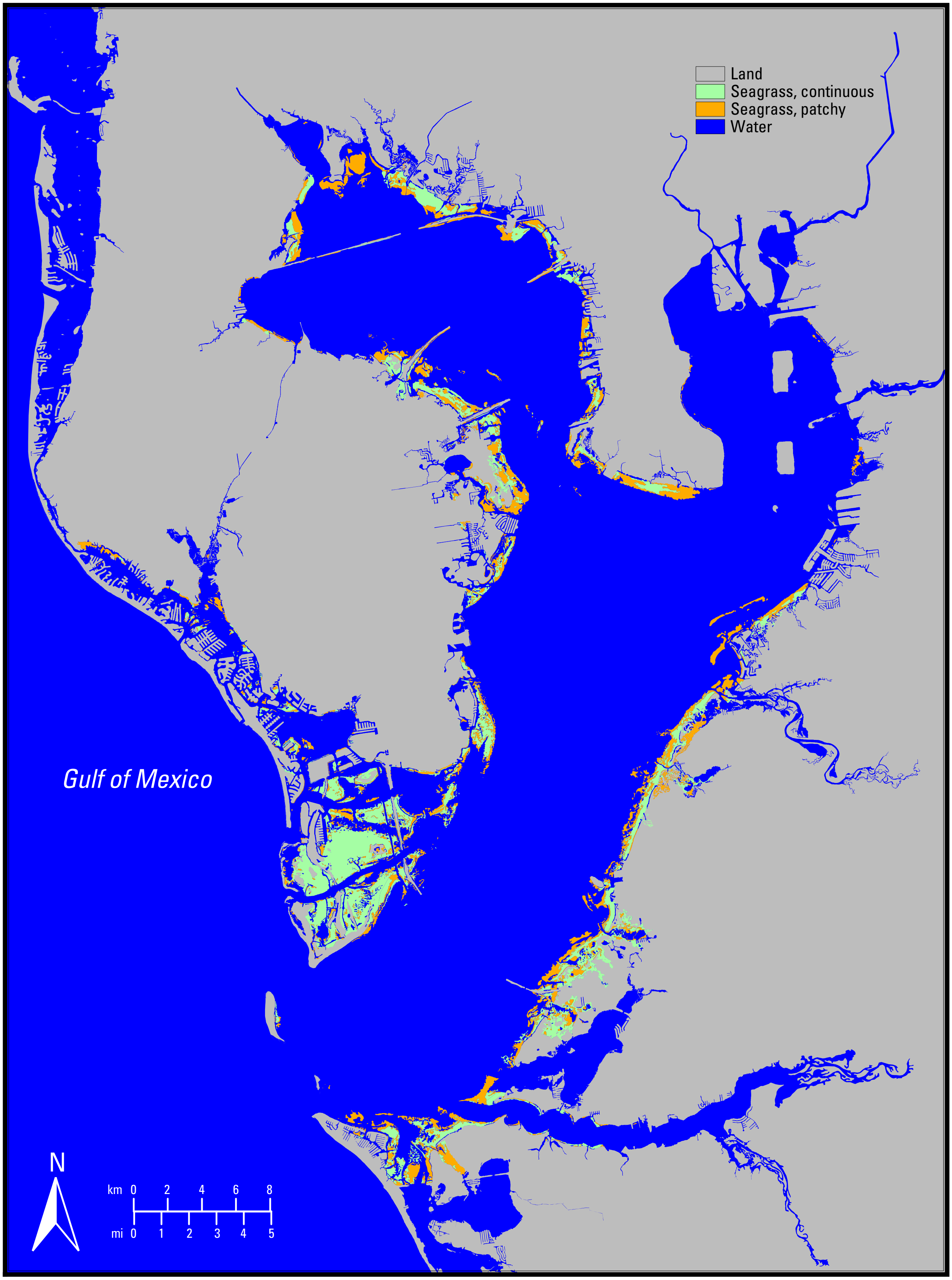

Figure 6. Distribution of seagrass in Tampa Bay, 1994.

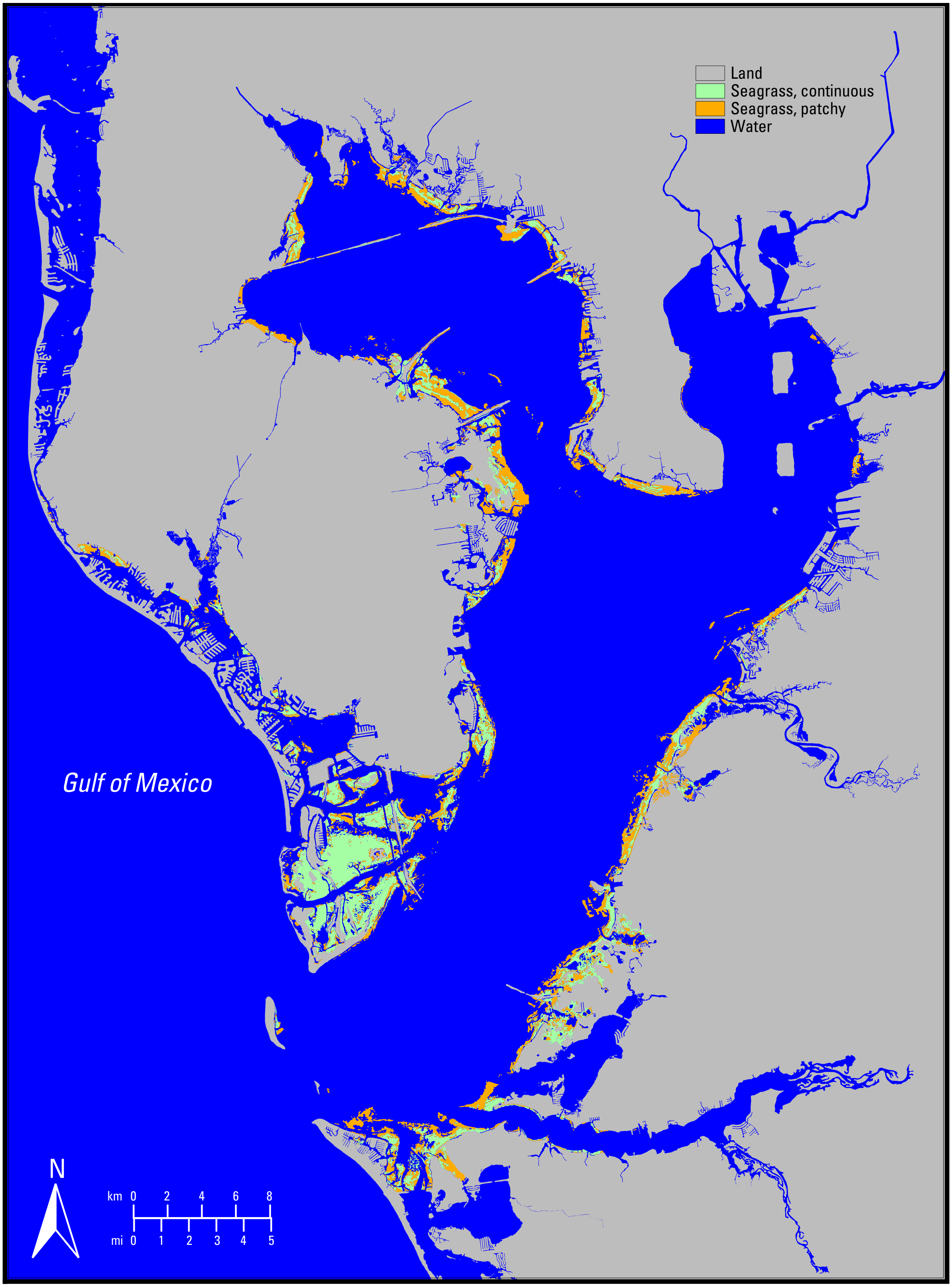

Figure 7. Distribution of seagrass in Tampa Bay, 1996.

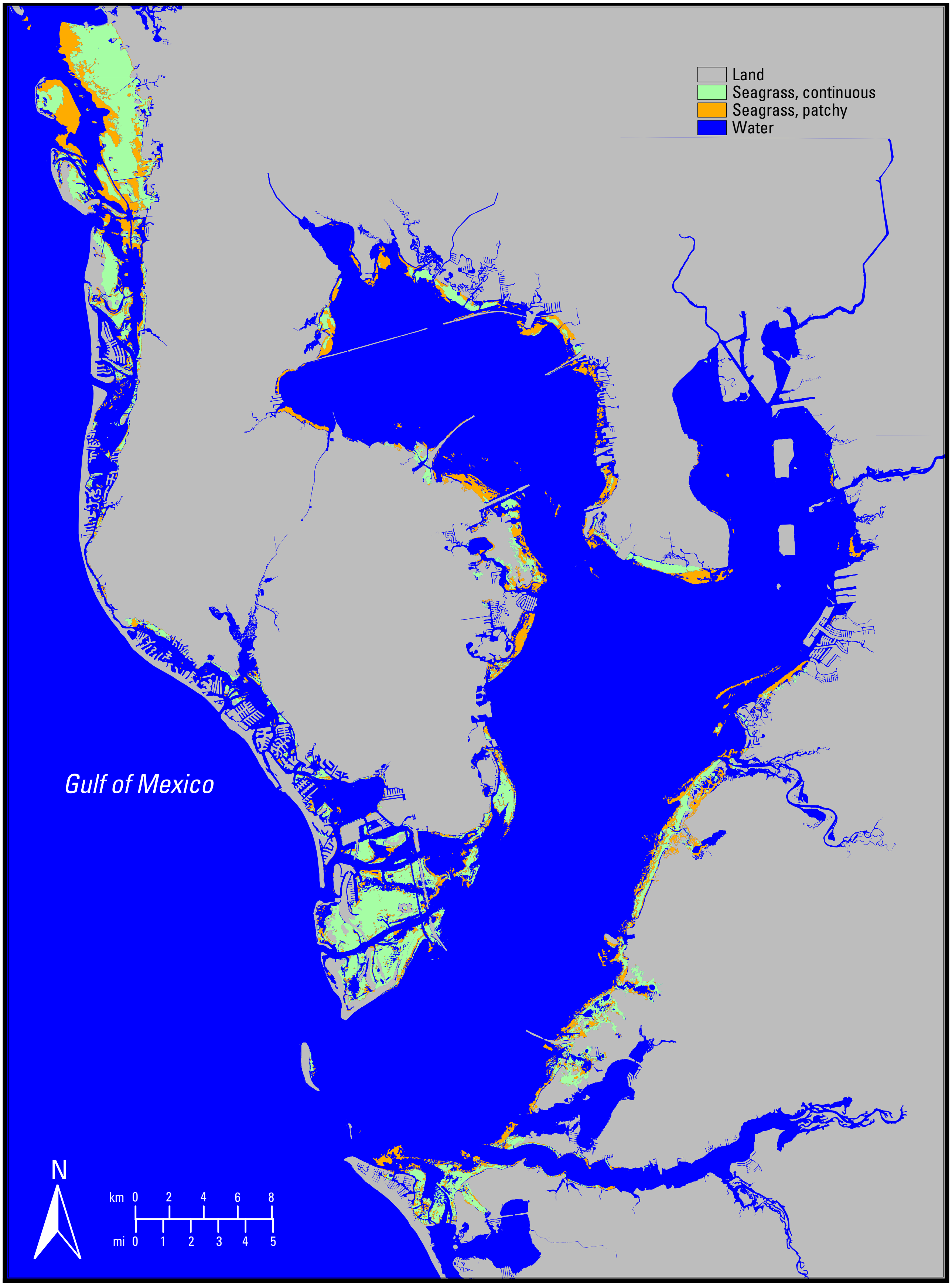

Figure 8. Distribution of seagrass in Tampa Bay, 1999.

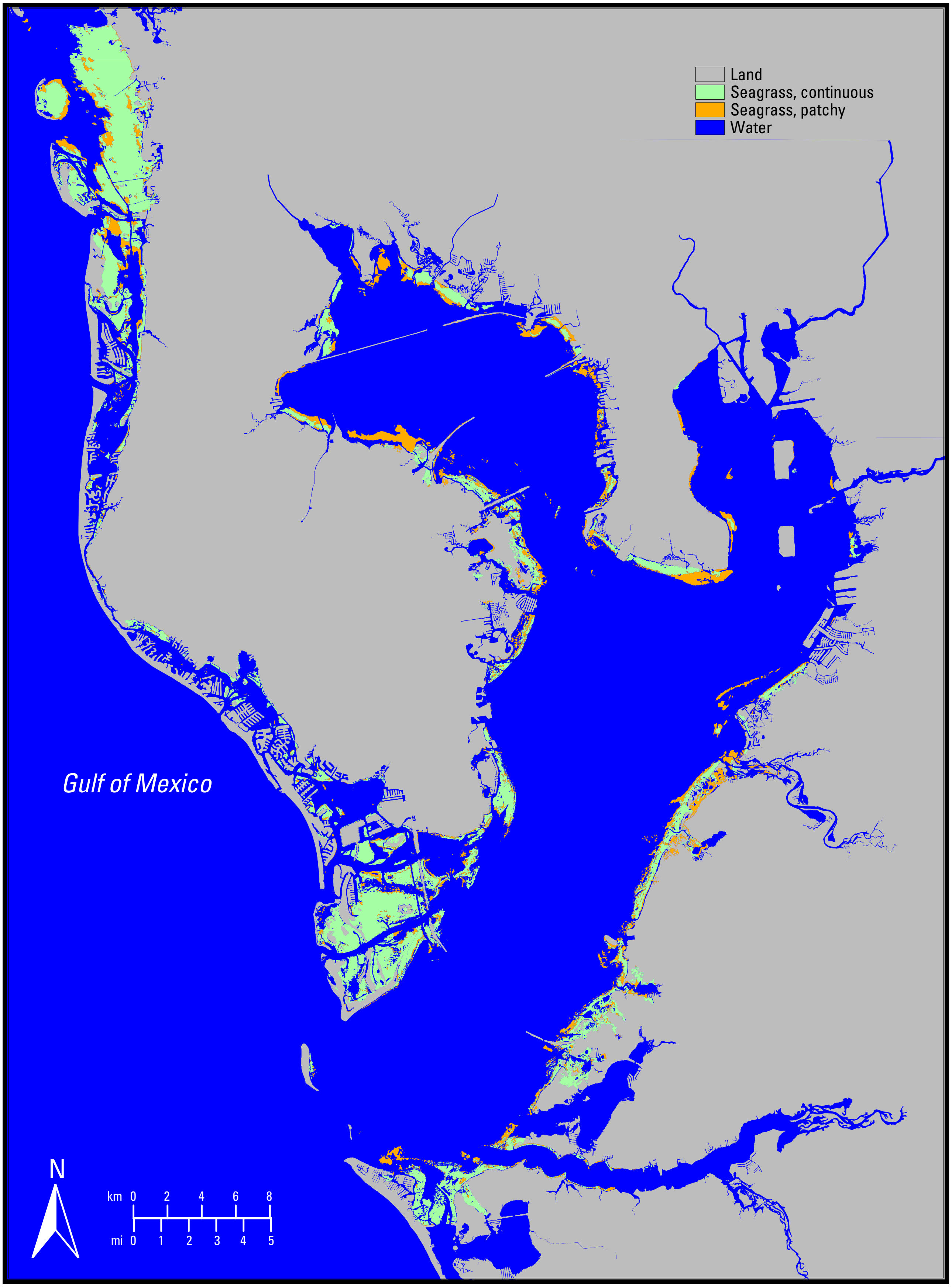

Figure 9. Distribution of seagrass in Tampa Bay, 2002.

and seagrass extent, was adopted by the TBNEP partnership in 1996 as its initial nitrogen-load management strategy.

A successful adherence to the maintenance nitrogen-loading strategy, however, may be hindered by the projected population growth in the watershed. A 20% increase in population and a subsequent 7% increase in annual nitrogen load are anticipated by the year 2010 (Zarbock and others, 1996). Therefore, if the projected loading increase (a total of 17 tons (~15 Mg) per year) is not prevented or precluded by watershed management actions, the maintenance nitrogen-load management strategy will not be achieved.

In 1996 local government, agency, and private industry partners formed a nitrogen management consortium to develop a plan to accomplish the nitrogen-reduction goal (17 tons (~15 Mg) per year) needed to meet long-term water-quality conditions necessary for seagrass restoration to historical levels. A nitrogen management action plan developed by the public and private partners in the consortium combined for each bay subarea all local government, agency, and private industry projects that will contribute to meeting the 5-yr nitrogen-management goal (Tampa Bay Estuary Program, 1998b). A total of 134 tons (~122 Mg) per year reduction in nitrogen loading to Tampa Bay was expected by the end of 2000, which exceeds the 1995–99 reduction goal by 60%.

Reduction of Seagrass Scarring

Fort Desoto Seagrass Protection Efforts

Assessments by the Florida Department of Environmental Protection and the Florida Marine Research Institute indicate that there is moderate to severe scarring in nearly 30% of the total seagrass coverage in Tampa Bay, some of the worst rates in the State. In response to these findings, and following the work of a coalition of regulatory and citizen advisory groups, Pinellas County adopted a seagrass protection ordinance in 1992 for the seagrass beds around Fort Desoto County Park, near the mouth of Tampa Bay (Stowers and others, 2000). Results of a 5-yr assessment indicated that, following placement of signage in the management area, the rate of increase of new scars was significantly reduced in both the caution zone and the exclusion zone as compared with the control (no signage) area (Stowers and others, 2000). The placement of signs, coupled with full-time, on-water presence of law enforcement officers in shallow draft boats, has reduced the rate of scarring (Jake Stowers, pers. commun.).

Cockroach Bay Seagrass Management

In addition to the governmental and private industry initiatives to manage nitrogen loadings to Tampa Bay, several efforts by citizens have contributed to the restoration and protection of seagrass. For example, when Hillsborough County proposed closing the area to boats and other restrictions in the Cockroach Bay Aquatic Preserve because of the extensive propeller scarring of seagrass in that area, local citizens and fishers organized in 1995 as "C-BUG" (Cockroach Bay Users Group) and proposed alternative, nonregulatory ways for protecting the seagrass from further propeller scarring. Hillsborough County agreed to allow C-BUG a 3-yr trial period for their strong protectionary approaches through education and voluntary slow speeds. C-BUG projects have included development and distribution of educational material to encourage the boating public to take responsibility for its actions. It has posted educational signs at public boat ramps in Cockroach Bay and placed 25 seagrass-area marker buoys at the deep edges of the seagrass meadows. Further, C-BUG has developed and placed a unique "stoplight" tidal gage to alert boaters of the tidal water level in relation to the seagrass. Recent monitoring of the Cockroach Bay area has indicated that several seagrass areas that were previously heavily scarred are recovering (Ehringer, 1998, 2000).

To date, results from these two seagrass management approaches support a similar conclusion: signage, education, and on-water presence appear to be just as effective in reducing rates of new scarring as is closure. This conclusion, if it continues to hold in these and other areas, is an important finding for seagrass management.

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Sarasota Bay

By David A. Tomasko[1] and Gary E. Raulerson[2]

Background

Sarasota Bay first took shape approximately 5,000 yr ago (Estevez, 1992) as a result of the formation and development of offshore barrier islands during a period of continuing sea-level rise. The Sarasota Bay watershed (fig. 1) contains a number of archeological sites, including shell middens, sand mounds, and cemeteries of various prehistoric cultures, some of which date back to 10,000 B.C. (Deming and others, 1990).

During the 1870s, a resort hotel in Osprey was one of the first business ventures to take advantage of Sarasota Bay's wealth of natural beauty, although commercial fishing had occurred since the late 1700s (Whelan, 1992). During the period of 1895–1903, the first large-scale channel alteration activities took place in Sarasota Bay as the dredging project "Suwanee" enlarged or created channels at the pass out of Palma Sola Bay, at Longboat Pass, and in the area between Little Sarasota Bay and Venice (Whelan, 1992).

Population growth in the watershed has been dramatic, especially in the post-World War II years. For example, in 1989 the population of Manatee and Sarasota Counties together (an area which includes areas outside of the Sarasota Bay watershed) was estimated at 425,400 (Sarasota Bay National Estuary Program, 1995); by the year 1995, that number was at approximately 513,900. As population in these two counties was less than 150,000 in 1940, this change represents an increase of nearly four times in population in just over 50 yr.

During this period of rapid growth, much environmental damage occurred as a result of large-scale dredge and fill projects, such as the conversion of Bird Key into a finger fill canal community and the dredging (in the 1960s) of the Gulf Intracoastal Waterway. The dredging of the Gulf Intracoastal Waterway may have increased the hydrologic instability of Midnight Pass—the movement of which precipitated its permitted closure and subsequent failed reopening in winter 1983—which has altered circulation in Little Sarasota Bay (Sheng and Peene, 1992).

Water quality in Sarasota Bay is influenced by the area of watershed that drains into different parts of the bay. In the northern portion of the bay (hereafter Upper Sarasota Bay) (see fig. 2), 153 km^2 (59 mi^2) of watershed drain into 117 km^2 (45 mi^2) of open water. In the southern portion of the bay, 236 km^2 (91 mi^2) of watershed drain into 18 km^2 (7 mi^2) of open water. Thus, the ratio of watershed to open water in the northern part of Sarasota Bay is 1:3, while in the southern part of the bay this ratio climbs to 13:4, a roughly tenfold increase.

According to data from 1990, residential land use accounted for approximately 42% of the total watershed, while 36% of the watershed was a combination of forested upland, rangeland, and open/recreational uses (Heyl, 1992). Commercial and industrial land uses accounted for 10% of the watershed, and agriculture (cropland and citrus) accounted for 9% of the land. The remainder of the watershed (about 4%) consisted of wetlands and open water bodies (lakes and streams).

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Scope of Area

In keeping with standard methodology for seagrass mapping efforts in Sarasota Bay, the study area is divided into five segments: (1) Sarasota County portion of Upper Sarasota Bay, (2) Manatee County portion of Upper Sarasota Bay, (3) Roberts Bay, (4) Little Sarasota Bay, and (5) Blackburn Bay (fig. 2). These delineations are coincident with mapping efforts and analyses dating back to 1988.

Methodology Employed To Determine and Document Current Status

Seagrass mapping efforts have played an important role in measuring the success and failure of maintaining and expanding on improvements to water quality in Sarasota Bay and other estuaries. These mapping efforts are conducted on a roughly biennial basis by the Southwest Florida Water Management District (SWFWMD), which conducts seagrass mapping to fulfill obligations under the comprehensive conservation and management plan (Sarasota Bay National Estuary Program, 1995) of the U.S. Environmental Protection Agency's National Estuary Program.

Seagrass maps are produced through a multiple-step process. First, aerial photography is obtained, usually in the

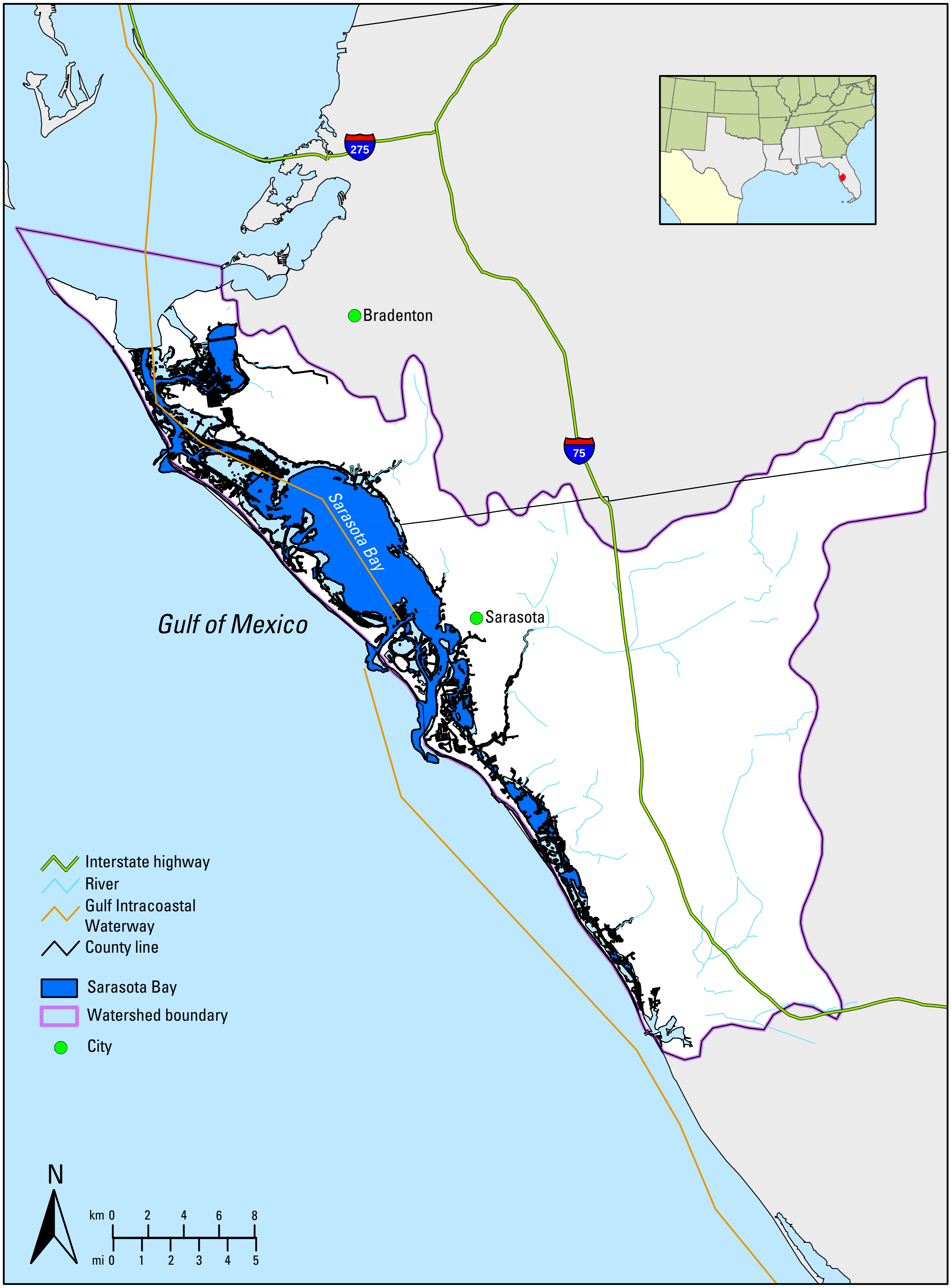

Figure 1. Watershed for Sarasota Bay.

late fall to early winter. This time of year is associated with good water clarity and relatively high seagrass biomass.

Second, photointerpretation efforts are conducted in the field to verify seagrass coverage suggested in the photography. The seagrass classification system is divided into two classes—continuous (<25% unvegetated bottom visible within a polygon) and patchy (>25% unvegetated bottom visible within a polygon)—with a minimum mapping unit of 0.2 ha (0.5 acres).

Third, polygons are integrated into an ArcInfo (Environmental Systems Research Institute, Inc., Redlands, Calif.) program. For some past efforts (i.e., 1988, 1990, 1992, 1994, and 1996), individual polygons were delineated onto Mylar® (DuPont Teijin Films) overlays, cartographically transferred by using a zoom transfer scope to U.S. Geological Survey (USGS) quadrangles, and then digitally transferred to an ArcInfo database for further characterization. These techniques allowed for the seagrass maps to meet U.S. National Map Accuracy Standards for 1:24,000-scale maps. For 1999 and 2002 seagrass maps, the 1:12,000-scale U.S. National Map Accuracy Standards were met. While photography remained at a scale of 1:24,000, the higher positional accuracy standard required the use of tighter ground control and more sophisticated mapping techniques. Analytical stereo plotters were used for photointerpretation rather than stereoscopes. This technique allowed for the production of a georeferenced digital file of the photointerpreted images without the need for additional photograph-to-map transfer. Rather than redrawing seagrass coverage polygons for each mapping year's efforts, the previous efforts' digital coverages were used as the baselines, and only the changes in seagrass coverage were mapped. Areas with no change between efforts are coincident with the earlier effort's coverage.

Fourth, hardcopies of plots are made of the photointerpreted seagrass coverage, and 60 randomly chosen points are identified for a classification accuracy assessment of the produced map. A hand-held Global Positioning System unit is used, along with the map and the latitude and longitude of the randomly located stations, to develop an unbiased determination of the map's classification accuracy. A 90% classification accuracy standard is required for these efforts; for example, 94% accuracy was achieved for 2002 efforts (i.e., 64 of 68 stations that could be visited had been accurately described in the map).

Status and Trends

For Sarasota Bay in 2002, the majority of coverage was in the northern portions of the bay, in the segments of Upper Sarasota Bay in Manatee County (62% of 2002 coverage) and Upper Sarasota Bay in Sarasota County (24% of 2002 coverage). Roberts, Little Sarasota, and Blackburn Bays contained 3%, 8%, and 3%, respectively, of the 2002 seagrass coverage.

In this study, historical seagrass changes between 1880 and 2002 were examined. The estimates for 1880 were based on the assumption that seagrasses grew down to approximately 2 m (6 ft) in water depth, which may be an underestimate of depth distribution in some areas and an overestimate of depth distribution in other areas. Estimates from 1950 were based on photography that was not groundtruthed or subject to intensive analysis based on geographic information systems. Therefore, estimates from 1988 to 2002 are more likely to be accurate than those from 1880 and 1950. Baywide, seagrass coverage declined from 1880 to 1950, with a further decline from 1950 to 1988. This decline was followed by a period of increased coverage which peaked in 1996, when values were 676 ha (1,671 acres) higher than in 1988, a 19% increase. Since 1996, coverage is down 462 ha (1,141 acres), an 11% decrease. Still, seagrass coverage in 2002 was 6% higher than in 1988. Despite evidence of a "leveling off" of seagrass coverage, the percentage of seagrass cover in the continuous coverage category has continued to increase since 1988.

In table 1 and the following text, changes in coverage on a more detailed basis for the years 1988 to 2002 are examined.

Upper Sarasota Bay in Manatee County

In the Upper Sarasota Bay segment in Manatee County, seagrass coverage increased from 2,213 ha (5,469 acres) in 1988 (see fig. 3) to 2,356 ha (5,821 acres) in 1994 (see fig. 4) and then to 2,541 ha (6,278 acres) in 1996 (see fig. 5), a 15% increase. Coverage declined from 1996 to 1999 (see figs. 5 and 6) to 2,312 ha (5,714 acres), a 9% decrease, followed by an increase from 1999 to 2002 (see figs. 6 and 7) of 13 ha (30 acres). Seagrass coverage in 2002 (see fig. 7) was 5% higher than in 1988.

Upper Sarasota Bay in Sarasota County

In the Upper Sarasota Bay segment in Sarasota County, seagrass coverage increased from 773 ha (1,909 acres) in 1988 (see fig. 3) to 850 ha (2,101 acres) in 1994 (see fig. 4) and then to 1,043 ha (2,578 acres) in 1996 (see fig. 5), a 35% increase. Coverage declined 209 ha (516 acres) from 1996 to 1999 (see figs. 5 and 6), a loss of 20%. From 1999 to 2002 (see figs. 6 and 7), seagrass coverage increased by 27 ha (106 acres). Seagrass coverage in 2002 (see fig. 7) was 13% higher than in the 1988 estimates.

Roberts Bay

In Roberts Bay, seagrass coverage increased from 134 ha (331 acres) in 1988 (see fig. 3) to 140 ha (345 acres) in 1994 (see fig. 4) and then to 145 ha (358 acres) in 1996 (see fig. 5), an 8% increase. Coverage declined between 1996 and 1999 (see figs. 5 and 6) to 136 ha (330 acres), followed by an additional decline of 26 ha (58 acres) between 1999 and 2002

Figure 2. Scope of area for the Sarasota Bay vignette.

Table 1. Seagrass coverage in hectares (acres) by year for segments of Sarasota Bay.

	1988	1994	1996	1999	2002
Upper Sarasota Bay, Manatee County	2,213 (5,469)	2,356 (5,821)	2,541 (6,278)	2,312 (5,714)	2,325 (5,744)
Upper Sarasota Bay, Sarasota County	773 (1,909)	850 (2,101)	1,043 (2,578)	834 (2,060)	877 (2,166)
Roberts Bay	134 (331)	140 (345)	145 (358)	136 (330)	110 (272)
Little Sarasota Bay	215 (532)	239 (591)	290 (717)	312 (770)	282 (698)
Blackburn Bay	166 (410)	166 (410)	162 (401)	151 (373)	122 (301)
Total	3,501 (8,651)	3,896 (9,628)	4,177 (10,322)	3,742 (9,247)	3,715 (9,181)

(see figs. 6 and 7). Seagrass coverage in 2002 (see fig. 7) was 18% lower than in 1988.

Little Sarasota Bay

In Little Sarasota Bay, seagrass coverage increased from 215 ha (532 acres) in 1988 (see fig. 3) to 239 ha (591 acres) in 1994 (see fig. 4) and then to 290 ha (717 acres) in 1996 (see fig. 5), a 35% increase. In contrast to all other bay segments, coverage increased from 1996 to 1999 (see figs. 5 and 6) by an additional 22 ha (53 acres), a 7% increase. From 1999 to 2002 (see figs. 6 and 7), seagrass coverage decreased by 30 ha (72 acres), a 10% decrease. Seagrass coverage in 2002 (see fig. 7) was 31% higher than in the 1988 estimates.

Blackburn Bay

In Blackburn Bay, seagrass coverage has shown a general pattern of decline over the period of record. Seagrass coverage was unchanged between 1988 and 1994 (see figs. 3 and 4) at 166 ha (410 acres). Between 1994 and 2002 (see figs. 4–7), coverage had declined by 44 ha (109 acres). Seagrass coverage in 2002 (see fig. 7) was 27% lower than in 1988 (see fig. 3).

Causes of Change

Historical losses of seagrass coverage in Sarasota Bay (i.e., post-World War II to 1988) are probably due to a combination of direct and indirect impacts associated with rapid urbanization of the watershed and of the edges of Sarasota Bay. It has been estimated that seagrass coverage declined by approximately 30% from the 1950s to 1988 (Sarasota Bay National Estuary Program, 2000). It must be noted that the seagrass coverage estimate for 1950 is an educated guess at best: the southern portion of the bay is characterized by reduced salinities that are associated with low water clarity, and it is likely that submerged vegetation could not be easily identified.

Increases in coverage between 1988 and 1996 (figs. 3–5) are thought to be mostly due to improvements in wastewater treatment practices in the watershed. Nitrogen loads from wastewater treatment plants were estimated to have declined from 516,188 kg/yr (1,138,195 lb/yr) in 1988 to 99,790 kg/yr (220,037 lb/yr) in 1999, an 81% decrease (Sarasota Bay National Estuary Program, 2000). Overall, nitrogen loads from all sources (i.e., point sources, nonpoint sources, atmospheric deposition, baseflow, and septic tank systems) are estimated to have declined by approximately 47% between 1990 and 1999 (Sarasota Bay National Estuary Program, 2000).

As a result of reductions in nitrogen loads, water quality has improved in nearly every portion of Sarasota Bay. Between 1968 and 1998, nitrogen and phosphorus concentrations declined in most parts of the bay, with no segments showing evidence of degradation (Dixon and Heyl, 1999).

A potential cause of the 1996–99 decreases in seagrass coverage in Sarasota Bay is the 1997–98 El Niño event, which caused annual rainfall amounts to be 20% to 48% higher than the 1988–99 average (as measured at Tampa International Airport). The excessive loads of nutrients and suspended

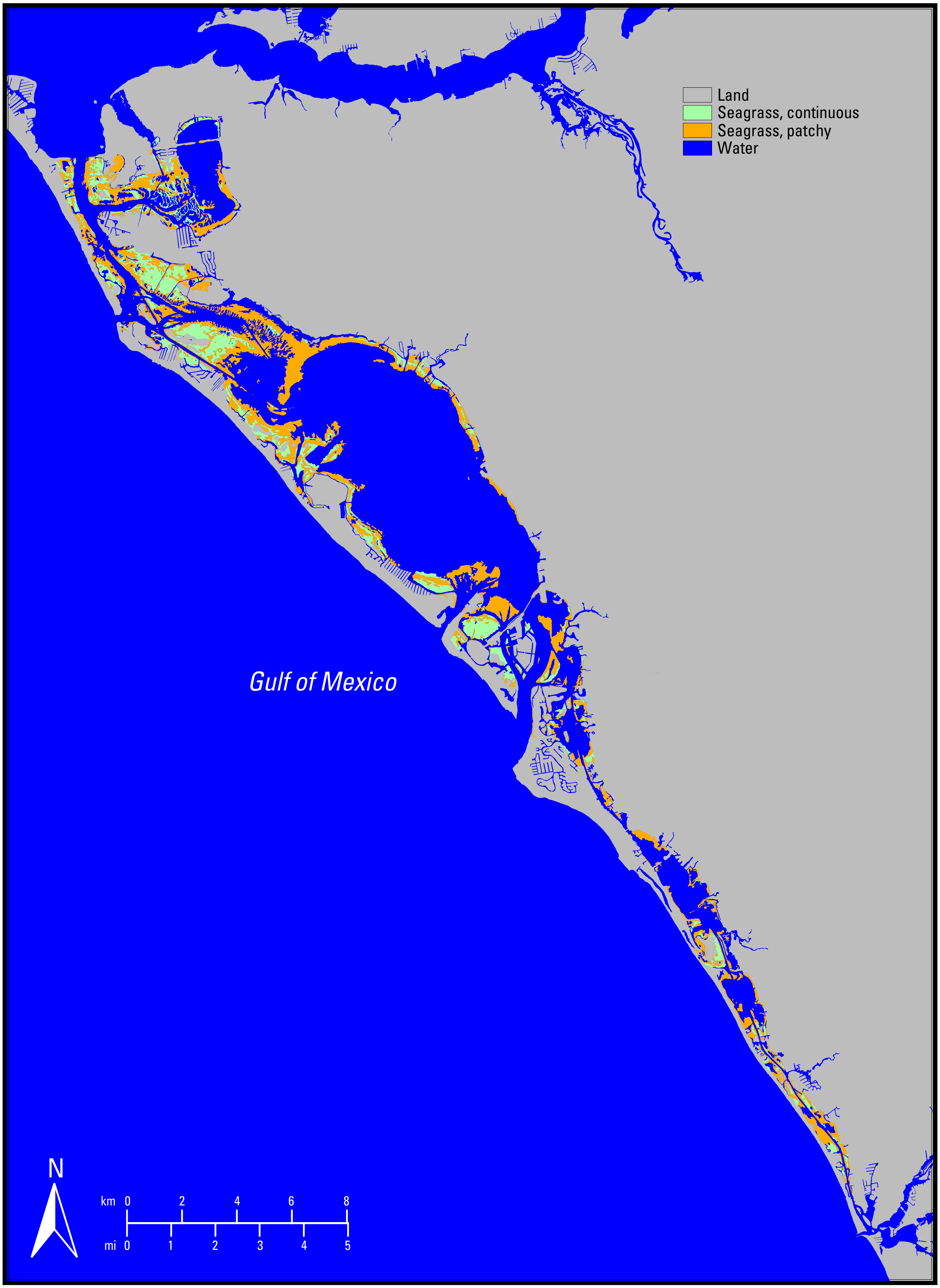

Figure 3. Distribution of seagrass in the Sarasota Bay system, 1988.

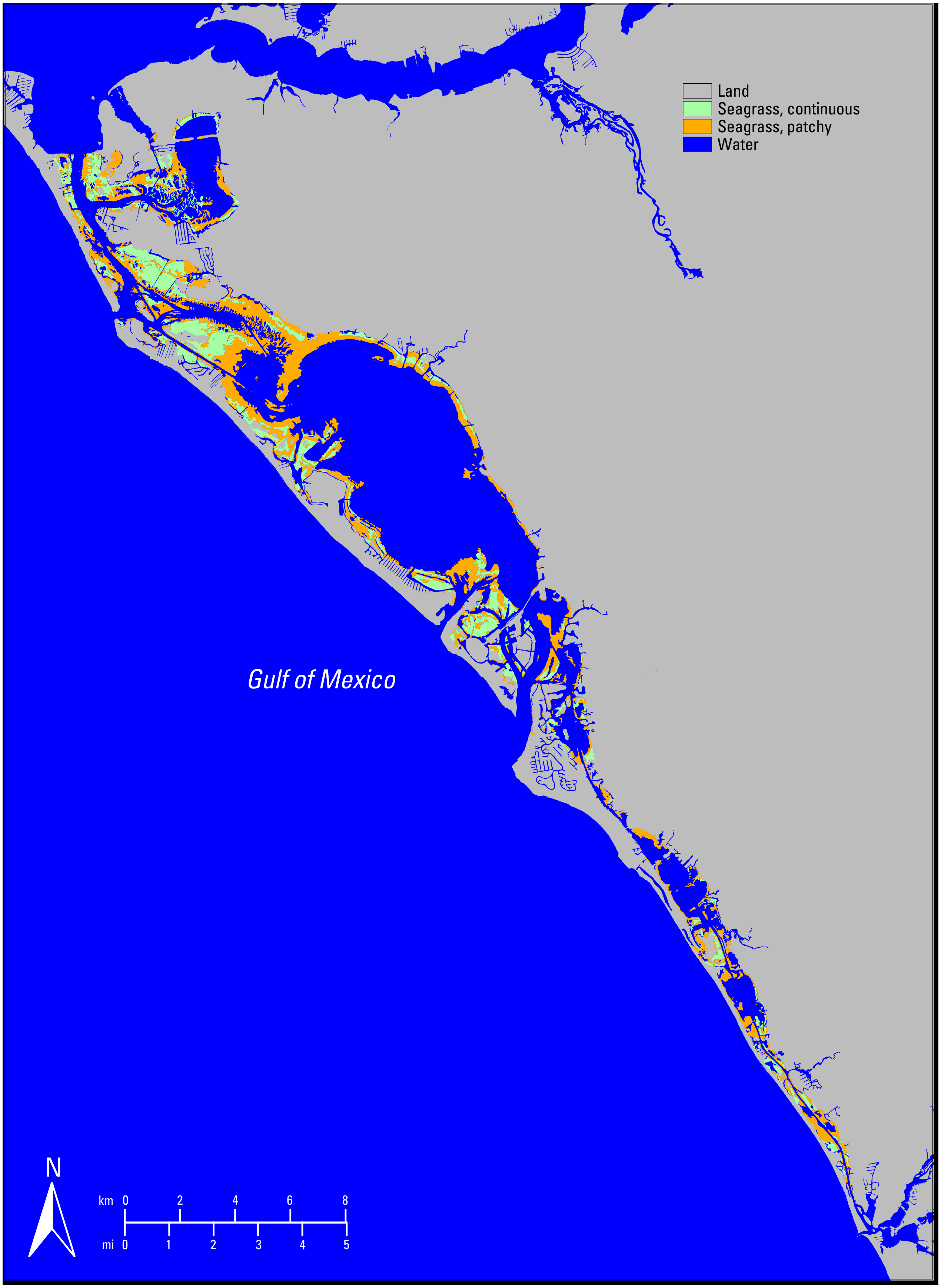

Figure 4. Distribution of seagrass in the Sarasota Bay system, 1994.

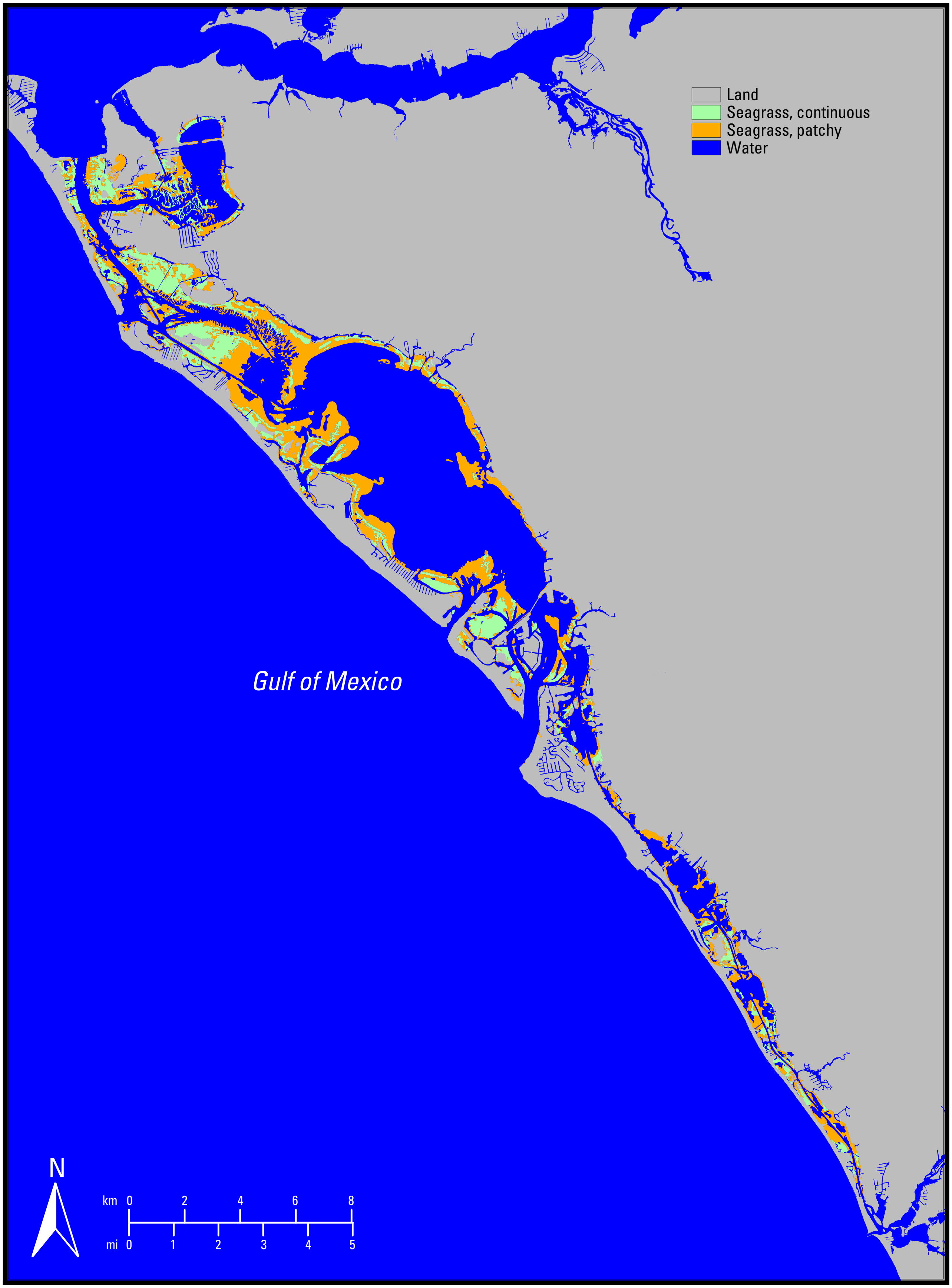

Figure 5. Distribution of seagrass in the Sarasota Bay system, 1996.

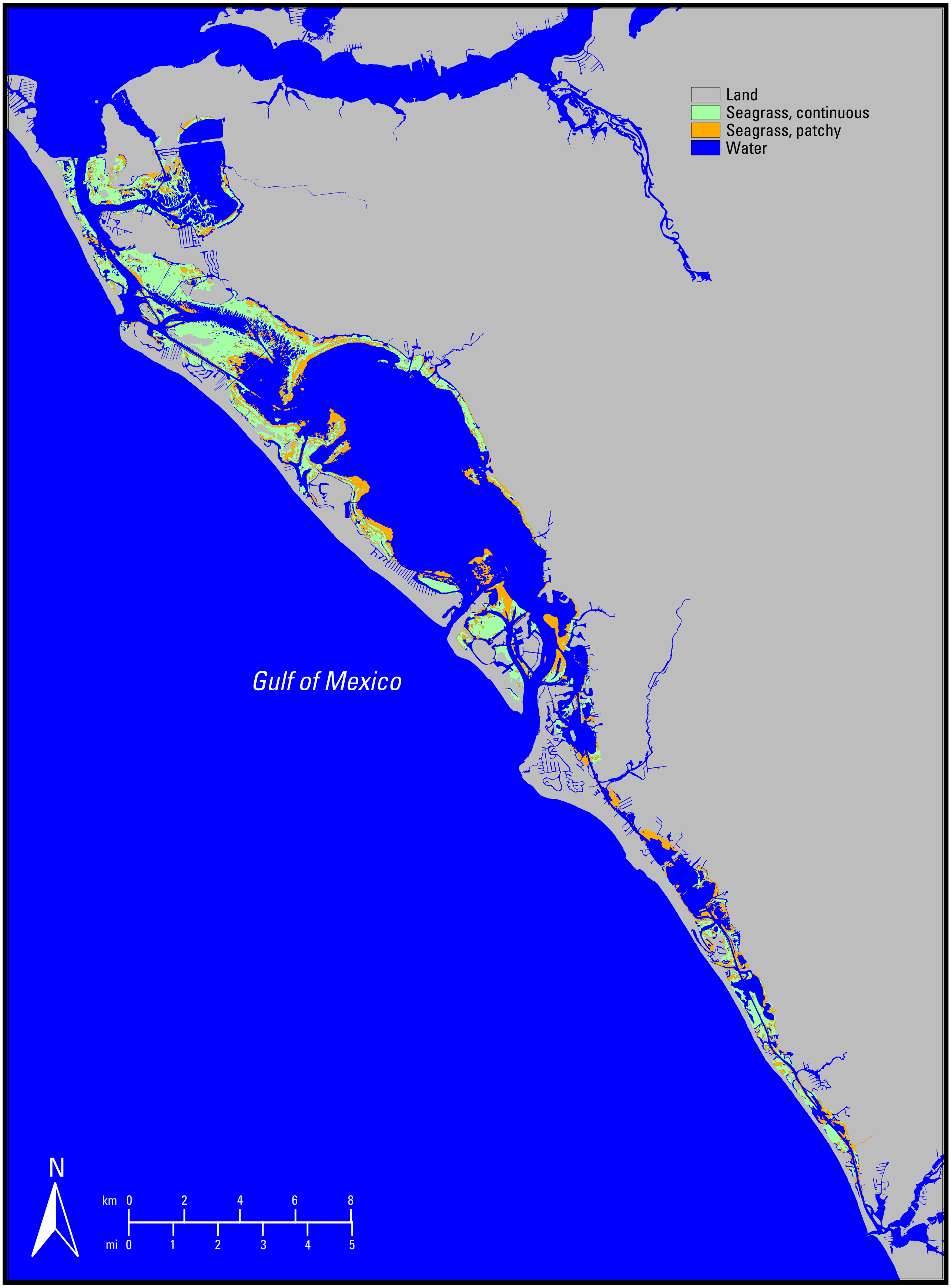

Figure 6. Distribution of seagrass in the Sarasota Bay system, 1999.

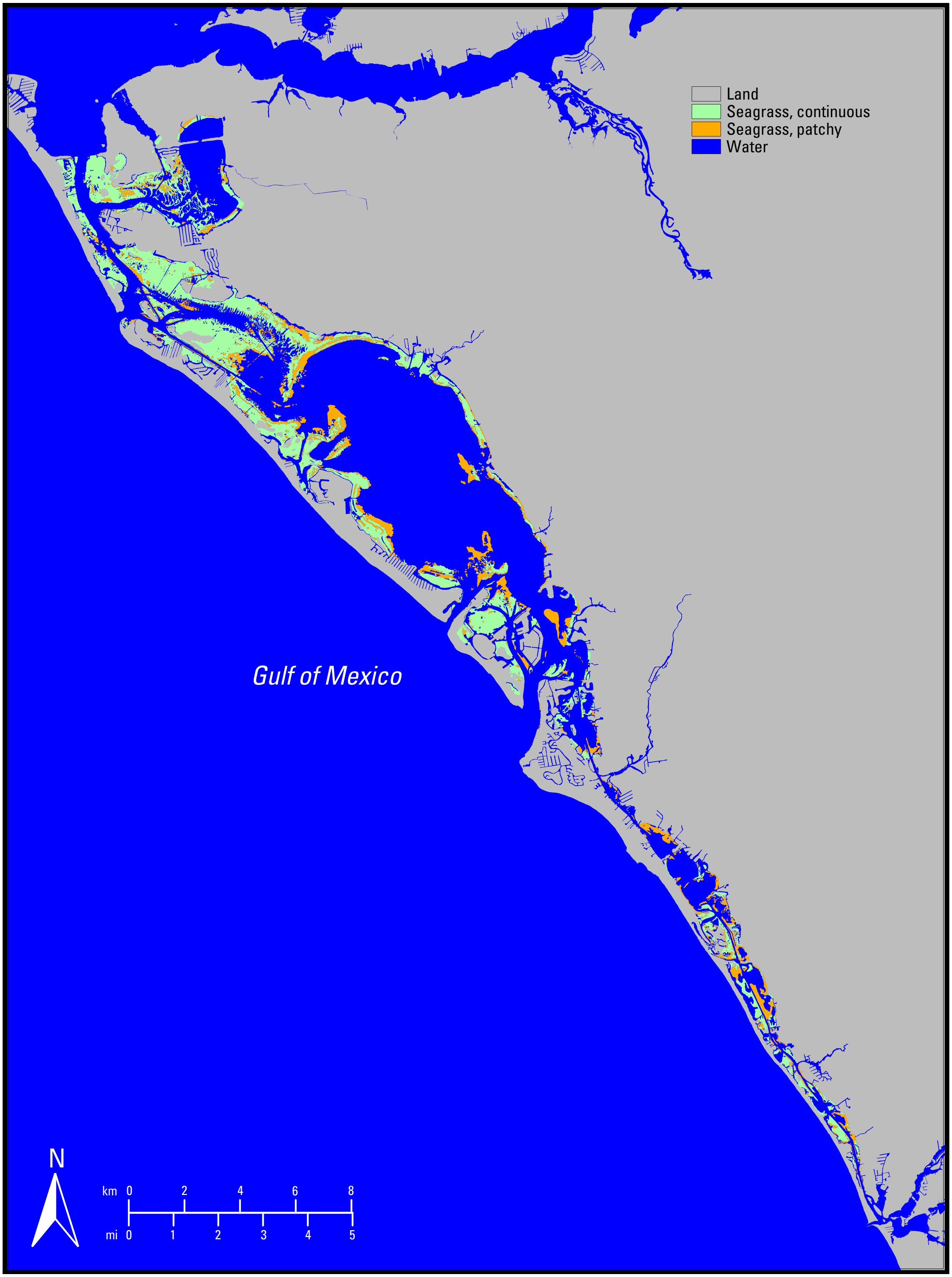

Figure 7. Distribution of seagrass in the Sarasota Bay system, 2002.

solids that accompanied this El Niño event would have most likely caused a considerable decrease in water clarity, which could account for the loss of seagrass coverage in most parts of Sarasota Bay. Little Sarasota Bay was the only bay segment to have an increase in coverage between 1996 and 1999 (table 1).

Between 1999 and 2002 (figs. 6 and 7), seagrass coverage increased in the northern portion of the bay, where most (86%) of the bay's seagrass is located. In contrast, coverage in Roberts and Blackburn Bays decreased by a combined total of 75 ha (185 acres) between 1996 and 2002 (figs. 5–7), while coverage in Little Sarasota Bay declined by 30 ha (72 acres) between 1999 and 2002 (figs. 6 and 7). Determining the cause of these declines is a pressing need for better understanding recent trends of water quality and seagrass health in the southern portions of Sarasota Bay.

Monitoring for Seagrass Health

Currently, aerial photography and seagrass mapping are anticipated to be continuing efforts that will be conducted every 2 yr or so. Furthermore, a series of 10 fixed transects have been established throughout Sarasota Bay (an effort that is based on the design of a similar effort in Tampa Bay). At each transect, seagrass coverage is estimated for each species by using the Braun-Blanquet method (for more information, see *http://chla.library.cornell.edu/cgi/t/text/text-idx?c=chla;idno=2917578*). Also, the maximum distance offshore where seagrasses are found is noted, as well as the relative water depth at each station. Year-to-year comparisons are thus possible for examining trends in seagrass meadow composition for individual transects. At present, there is no formal report produced based on these efforts; however, useful information can still be derived from this monitoring program. For example, results of these efforts thus far suggest a general trend toward expansion of the deep edges of meadows to waters farther offshore during the period of 1998–2000; however, changes in transect location have reduced, somewhat, the strength of any conclusions drawn from this dataset.

Species Information

Shoal grass (*Halodule wrightii*) is distributed throughout Sarasota Bay. Wigeon grass (*Ruppia maritima*) is mostly restricted to Little Sarasota Bay, particularly the area behind the now-closed Midnight Pass. Manatee grass (*Syringodium filiforme*) is mostly restricted to higher salinity (and higher water clarity) portions of the bay, particularly to the northern portions close to Longboat Pass and New Pass. Manatee grass is common in the northern and western portions of Anna Maria Sound. Turtle grass (*Thalassia testudinum*) is more widespread than manatee grass and is abundant in all portions of the bay except the Little Sarasota Bay segment. Finally, star grass (*Halophila engelmannii*) is found as an understory plant in several locations, particularly at the deeper edges of seagrass meadows in Roberts Bay.

Restoration and Enhancement Opportunities

Since its inception in 1989, the Sarasota Bay National Estuary Program (SBNEP) has been the primary agency focused on restoration and protection of the natural resources of Sarasota Bay. In 1992, after 3 yr of intensive technical assessment, the bay's health was documented by SBNEP in a report (Sarasota Bay National Estuary Program, 1992) in which principal investigators for projects on pollutant-loading models, shellfish contamination, fisheries, etc., proposed various projects that would be appropriate for protecting or restoring the bay's natural resources. These projects were extensively reviewed by various technical committees, and a number of formal public presentations were made.

The SBNEP also produced a comprehensive conservation and management plan (Sarasota Bay National Estuary Program, 1995) based partly on input from the public. The comprehensive plan included a list of action plans that were to be fulfilled by various local, regional, State, and Federal agencies. These action plans focused on major issues, such as declines in water and sediment quality, loss of wetlands and other coastal habitats, loss of seagrasses, and declines in finfish and shellfish populations.

In recent years, a number of projects have been undertaken that have benefited the bay's natural resources. Wastewater nitrogen loads to the bay are believed to have declined from 515 Mg (569 tons) per year in 1988 to 100 Mg (110 tons) per year in 1999, an 80% decrease (Sarasota Bay National Estuary Program, 2000). Baywide nitrogen loads are thought to have declined by approximately 47% (Sarasota Bay National Estuary Program, 2000). As a result, water quality has improved throughout most of the bay, and seagrass cover has expanded as well.

Continued efforts are required to reduce stormwater loads of nitrogen, especially because of continued urbanization of the bay's watershed. In addition, further projects are necessary to continue to reduce point sources of pollution by increasing the quantity of wastewater treatment plant effluent that is reused and by replacing malfunctioning septic tank systems with appropriate wastewater treatment practices.

At this time, there is little enthusiasm from the Technical Advisory Committee of SBNEP for projects focusing on transplanting efforts for seagrass recovery in Sarasota Bay.

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Charlotte Harbor

By Catherine A. Corbett[1] and Kevin A. Madley[2]

Background

Charlotte Harbor, located on the west coast of Florida south of Tampa and Sarasota Bays, is the second largest open water estuary in the State. It is also generally considered one of Florida's most pristine and productive estuaries. Most of the harbor is surrounded by an extensive conservation buffer system of well over 21,610 ha (53,398 acres) that the State of Florida began purchasing in the 1970s. Much of the shoreline in this buffer system is unaltered mangrove and salt marsh habitats, thereby providing abundant food and shelter for juveniles of many of the harbor's estuarine species. In addition to the State's ongoing land acquisition program, the U.S. Fish and Wildlife Service operates a series of national wildlife refuges along the Sanibel and Pine barrier islands that continues up into the tidal Caloosahatchee River. The largest of these refuges, J.N. "Ding" Darling, has almost 900,000 visitors every year and is residence to the federally endangered American crocodile (*Crocodylus acutus*). The harbor itself is home to more than 40 endangered or threatened species and boasts a world-class recreational fishing industry, including tarpon (*Megalops atlanticus*), snook (*Centropomus undecimalis*), reddrum (also called redfish; *Sciaenops ocellatus*) and spotted seatrout (*Cynoscion nebulosus*). Combined with the harbor's commercial fishing industry, fishing has an estimated impact of over one billion dollars annually (Charlotte Harbor Natural Estuary Program, 2000).

Agriculture encompasses the major land use in the greater Charlotte Harbor watershed and is second only to tourism in economic impact. In 1995, a total of 114,520 ha (282,978 acres) within the greater Charlotte Harbor watershed was dedicated to citrus crops—one-third of all Florida citrus acreage (Charlotte Harbor Natural Estuary Program, 2000)—while in 1990 over 404,680 ha or nearly one million acres was devoted to rangeland or pasture for cattle (Charlotte Harbor Natural Estuary Program, 1999). Simultaneously, Florida leads the nation in conversion of farmland to urban lands, and along the coast especially, residential and urban development is rapidly expanding. In 2020, the region is projected to have a population of almost 2 million residents, a 424% increase from the 1960s population of 363,200 (cited in Charlotte Harbor Natural Estuary Program, 2000). Finally, there is an extensive phosphate mining industry within the middle and upper reaches of the northern watershed. The "Bone Valley" phosphate deposit of more than 202,342 ha (499,987 acres) lies primarily within the Peace River subbasin. This phosphate deposit provides almost 75% of the Nation's phosphate supply and 25% of the world's (Charlotte Harbor Natural Estuary Program, 2000). Future mining is expected to move southward towards the harbor and last an additional 30 yr.

In the southwest Florida region, much research has focused on seagrass meadows to ascertain the implications of human impacts on estuarine resources, and to this end Charlotte Harbor is generally considered to be fairly healthy. For much of the greater Charlotte Harbor region, there has been little conclusive evidence of a substantial change in seagrass coverage, and mapping efforts in the harbor since 1982 have not demonstrated significant coverage trends. In addition, pollutant loads have not been documented as a threat to seagrass extent to date in the harbor. The subbasins that make up the greater Charlotte Harbor region face disparate issues, however, and in several subbasins, there may be cause for concern. For instance, in the southern Charlotte Harbor region there was documentation of an approximate 57% loss of seagrasses between the 1940s and 1980s (Harris and others, 1983), believed to be a result of physical alterations in harbor circulation patterns from the dredging of the Gulf Intracoastal Waterway and Sanibel Island Causeway construction. In Lemon Bay, seagrasses may be demonstrating a slight decline in recent years (Kurz and others, 2000), which could in turn be linked to increasing pollutant loads, nitrogen in particular (Tomasko and others, 2001). In Estero Bay, there is evidence of losses but, because of a lack of monitoring efforts and research into possible causes, little inference can be drawn at this time.

Scope of Area

The greater Charlotte Harbor watershed (fig. 1*A* and fig. 1*B*) extends approximately 210 km (130 mi) from the northernmost headwaters of the Peace River to southern Estero Bay and for this effort is divided into six major hydrologic subbasins: Charlotte Harbor; Lemon and Estero Bays; and

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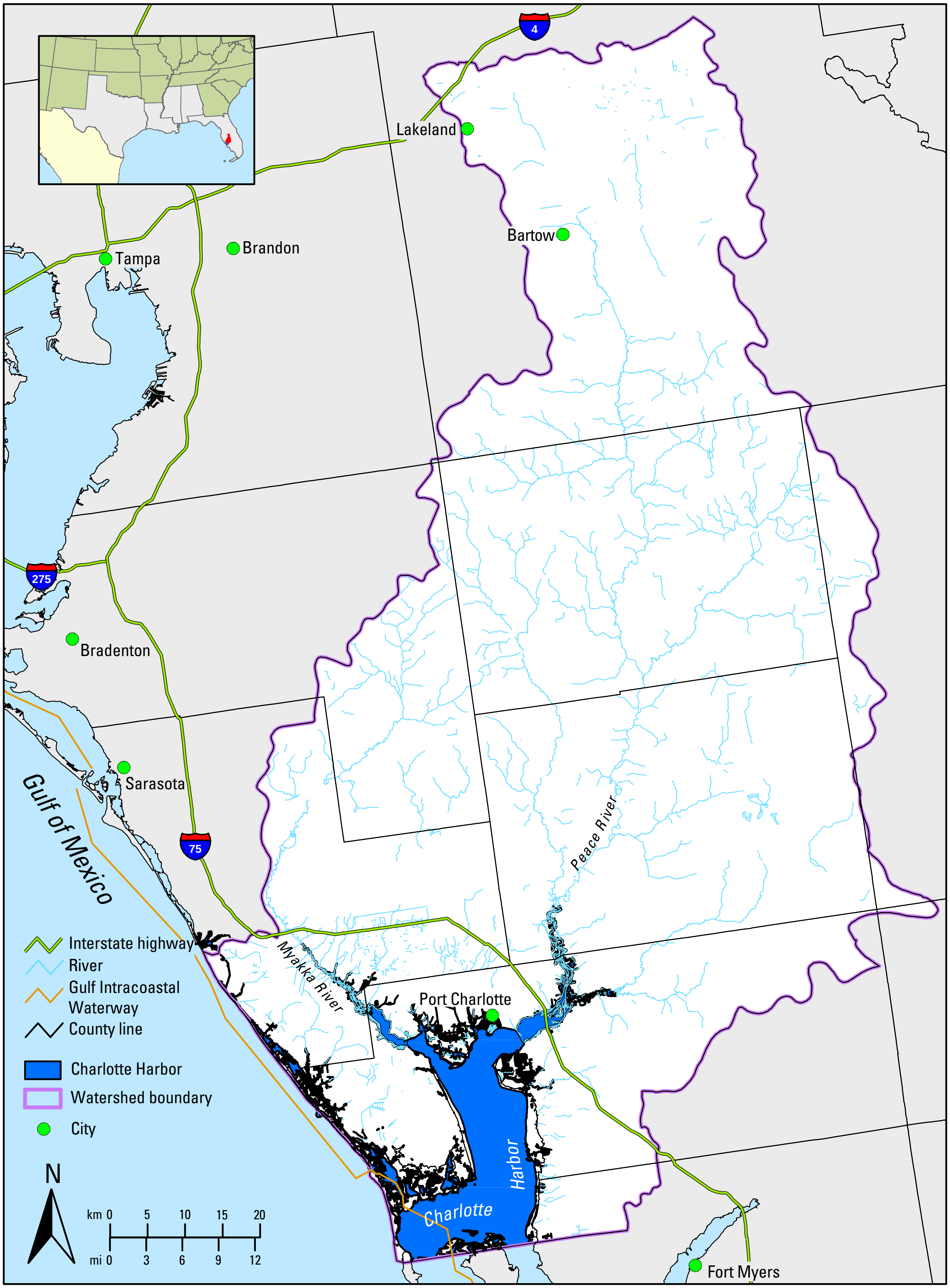

Figure 1*A*. Watershed for northern Charlotte Harbor region.

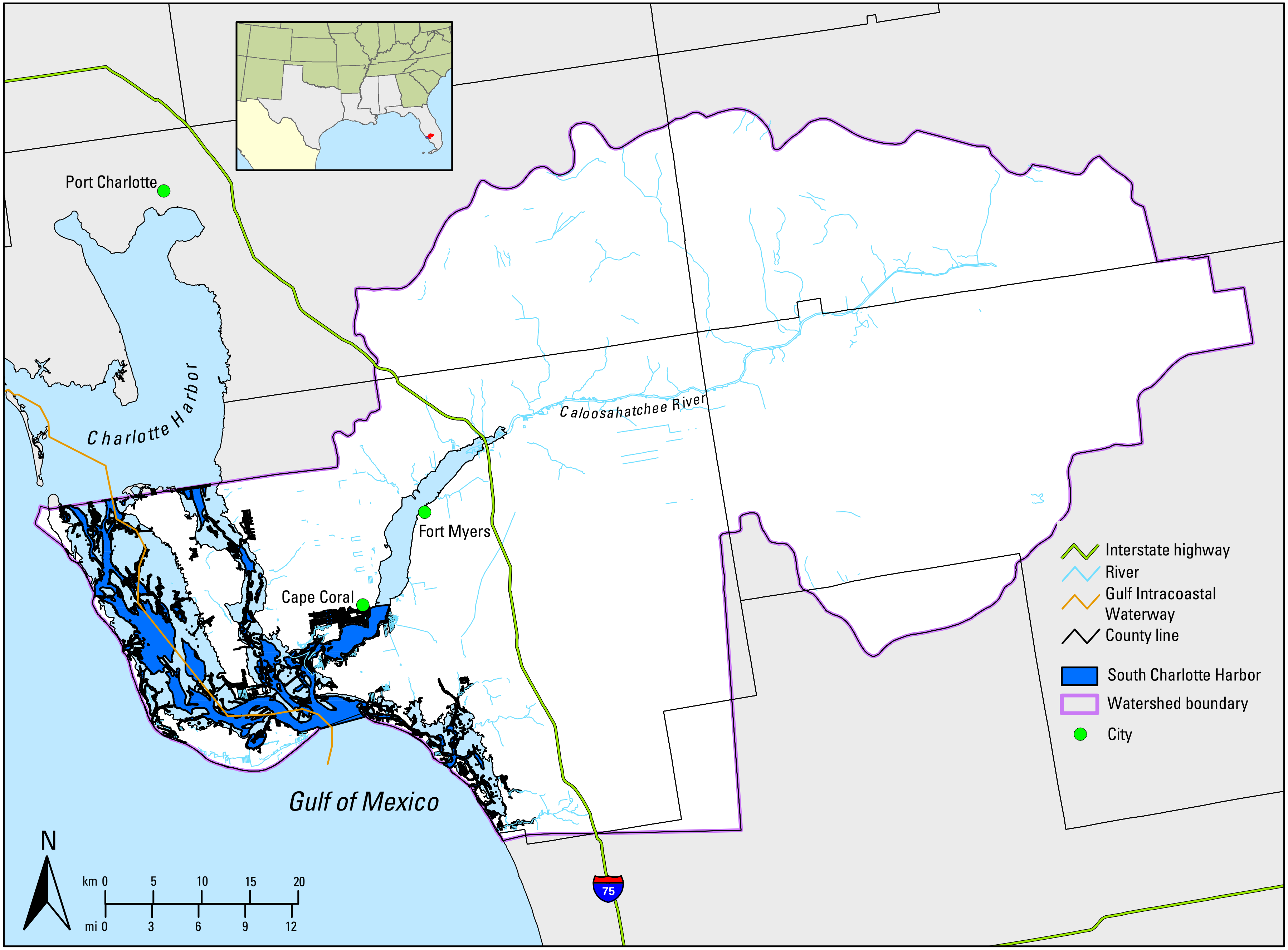

Figure 1*B*. Watershed for southern Charlotte Harbor region.

the Caloosahatchee, Peace and Myakka Rivers. The latter two subbasins, the Peace (a 6,090 km^2 or 2,350 mi^2 basin) and the Myakka (a 1,560 km^2 or 602 mi^2 basin) Rivers, serve as two major sources of fresh water to the Charlotte Harbor estuary (cited in Hammett, 1990). The average discharge from the Peace River is estimated at 60 m^3/s (2,010 ft^3/s) to 75 m^3/s (2,640 ft^3/s), and the Myakka River is estimated at 18 m^3/s (630 ft^3/s), although the discharge is much higher during early July through late September in the summer rainy season (Hammett, 1990).

The third major source of fresh water to the harbor is the Caloosahatchee River (3,570 km^2 or 1,378 mi^2 basin extending to Moore Haven), which contributes an annual average inflow to the lower harbor of approximately 57 m^3/s (2,000 ft^3/s) (Hammett, 1990). The river was channelized and connected to Lake Okeechobee in the late 1800s (although there is some evidence that the Calusa Indians may have created a link between the two waterbodies much earlier (South Florida Water Management District, 1998)) and repeatedly dredged over the next century to provide flood protection and serve as both a source of agricultural and urban water supply and a navigational channel. A series of three locks and dams was constructed along the river, one of which, the W.P. Franklin locks and dam, artificially truncates the river's estuarine system by blocking the natural gradient of fresh to salt water that historically extended upstream during the dry season. It is common to have disparate salinity regimes on the two sides of this lock system during the dry season when the locks are closed. Water exiting the Caloosahatchee River flows both north up lower Matlacha Pass and southern Pine Island Sound and south through San Carlos Bay and Estero Bay to the Gulf of Mexico.

South of the Caloosahatchee River is Estero Bay, a shallow 4,580 ha (11,317 acres) bay with a 780 km^2 (301 mi^2) basin (Charlotte Harbor Natural Estuary Program, 2000). Estero is the receiving water body for the Imperial and Estero Rivers and Hendry, Spring, and Mullock Creeks. This bay's watershed, situated between the cities of Naples and Fort Myers, is an area of very rapid population growth, including a high density of golf courses. The tributaries to Estero Bay all demonstrate a recent increase in phytoplankton blooms, and three are on the State's Impaired Waters List for nutrient impairments.

Finally, Lemon Bay, another shallow bay, is located to the northwest of the Charlotte Harbor subbasin and actually connects the harbor to Sarasota Bay via the Gulf Intracoastal Waterway. This bay is 21 km (13 mi) in length and 0.2 to 1.9 km (0.12–1 mi) wide with a surface area of only 31 km^2 (12 mi^2) (Tomasko and others, 2001). Lemon Bay receives fresh water from several small tributaries: Buck, Coral, Alligator, Forked, Gottfried, Rock, and Oyster Creeks. The waters leaving Lemon Bay in turn drain into the Gulf of Mexico through Stump Pass or by entering Gasparilla Sound and coalescing with the waters exiting Charlotte Harbor to the south.

These 6 hydrologic subbasins are further segmented into 14 segments for analyses of seagrass in this effort (fig. 2). Eight of these segments, from Lemon Bay to the South Harbor area of northern Charlotte Harbor, fall within the Southwest Florida Water Management District's (SWFWMD) jurisdiction (table 1). The other six segments fall under the purview of the South Florida Water Management District (SFWMD).

Table 1. Fourteen subsegments created for analyses of seagrass coverage in the greater Charlotte Harbor region.

Southwest Florida Water Management District Region	
Charlotte Harbor (Northern):	Charlotte Harbor (Southern):
Lemon Bay	Pine Island Sound
Peace River	Matlacha Pass
Myakka River	San Carlos Bay
Middle Harbor	Lower Caloosahatchee River
West Wall	Upper Caloosahatchee River
East Wall	Estero Bay
Placida Region	
South Harbor	

Methodology Employed To Determine and Document Current Status

The SWFWMD and SFWMD conduct seagrass mapping efforts within the Charlotte Harbor region as fulfillment of the districts' obligations under the Charlotte Harbor National Estuary Program's (CHNEP) Comprehensive Conservation and Management Plan (CCMP). The seagrass mapping effort is also included within the SWFWMD's Surface Water Improvement and Management (SWIM) Plan for the northern harbor and Lemon Bay regions. The SWFWMD has conducted the seagrass mapping efforts on a roughly biennial basis since 1988, and the SFWMD initiated the undertaking of biennial seagrass mapping efforts within their area in 1999. In 1999, which was the most recent comprehensive mapping effort for the harbor, the two districts used somewhat dissimilar methodologies in their mapping efforts; however, the SFWMD is using most of the same methodologies as the SWFWMD in future efforts to ensure better data comparability.

Figure 2. Scope of area for the Charlotte Harbor seagrass vignette.

Southwest Florida Water Management District

Seagrass maps are produced through a multistep process on a district-wide basis; therefore, this same methodology is used in Tampa and Sarasota Bays as well as in Charlotte Harbor. First, aerial photographs are obtained during times of good water clarity and moderately high seagrass biomass—usually November or December—after the summer rains have ceased. True color photographs at a scale of 1:24,000 are used, and the film is New Kodak Aerocolor negative film 2445, or a district-approved equivalent. The requisite end and side laps for the photographs are 60% and 30%, respectively. The SWFWMD requires that secchi disk depths be over 2 m, wave height less than 0.61 m (2 ft) and the wind speed less than 16 km/hr (10 mi/hr) on the day the photographs are obtained. The sun angle must be at least 35 degrees and tidal stage must be at no greater than mean tide level. These requirements necessitate that district staff, personnel from supporting agencies, and/or volunteers be out on the estuary checking for the correct conditions on several occasions before an aerial survey is actually flown.

Next, investigators examine bottom cover at various locations in the field to allow identification of distinct photographic signatures and investigate unusual signatures. In the office, the field classifications are matched to signatures on the photographs and used to train the photointerpretation staff. Seagrass signatures are divided into two classes: continuous coverage (<25% unvegetated bottom visible within a polygon) and patchy (>25% unvegetated bottom visible within a polygon. The Florida Land Use, Cover and Forms Classification System (FLUCCS; *http://www.dot.state.fl.us/surveyingandmapping/fluccmanual.pdf*) has been used to divide seagrass into two categories: patchy seagrass as 9113 (>25% unvegetated bottom visible within a polygon) and continuous seagrass as 9116 (<25% unvegetated bottom visible within a polygon). The minimum mapping unit is 0.02 ha (0.5 acres).

For the earlier mapping efforts (1988, fig. 3; 1992, fig. 4; 1994; fig. 5; and 1996, fig. 6) the individual polygons were delineated on Mylar® sheets placed over top of the aerial photographs, and then a zoom transfer scope was used to transfer the delineated polygons to USGS quadrangles. Next, the polygons were digitally transferred to an ArcInfo database. The resulting seagrass maps meet USGS National Map Accuracy Standards for 1:24,000 scale maps. For the 1999 and 2002 seagrass maps, tighter ground control and more sophisticated mapping techniques were used to meet 1:12,000 National Map Accuracy Standards while still using 1:24,000 scale photographs. Analytical stereo plotters were used for photointerpretation in lieu of the stereoscopes. This method allowed for the production of a georeferenced digital file of the photointerpreted images without the need for additional photo to map transfer. Instead of drawing complete polygons each year, effort and errors have been reduced by using the previous effort's digital coverage as the baseline and delineating any changes to seagrass extent for the current effort. This method has provided a change analysis as well as a current seagrass coverage.

Hard copy plots were produced and checked for errors. Finally, between 20 and 40 randomly chosen points were identified for the northern Charlotte Harbor and Lemon Bay regions and plotted for a classification accuracy assessment. The points were randomly selected by using ArcInfo processes and by first defining the coordinates of the study area. The point selection then involved the random generation of numbers based on the minimum and maximum values of the X and Y coordinates of the study area. The numbers that were generated were stored as variables, and a selection was made from the ArcInfo coverage to see if they fit the criteria specified (i.e., seagrass codes = 9113 or 9116). A variable was also set up to be used as a counter, and set to a value of zero (0). If the area did not fit the selection criteria, the "counter" variable was not calculated and the loop ran itself again. If the area fit the selection criteria, a point was placed at the position and the coordinates were stored in the variable; then the "counter" variable was calculated with the next value. This process was repeated until approximately 10–20 points per estuary region were selected. Field staff used the coordinates for the randomly chosen sites, a site map and a Global Positioning System (GPS) unit to visit the locations in the field and classify the bottom cover. These field checks were compared to the map classifications to develop an unbiased determination about classification accuracy of the map. The SWFWMD requires a 90% classification accuracy standard for the seagrass mapping efforts, and accuracies over 95% were achieved in 1999 (figs. 7*A* and 7*B*) and 2002 (fig. 8).

South Florida Water Management District

The 1999 seagrass mapping effort for southern Charlotte Harbor, including Pine Island Sound, Matlacha Pass, San Carlos Bay, the upper Caloosahatchee River, lower Caloosahatchee River and Estero Bay areas, used a different methodology than the SWFWMD's described above. New Kodak Aerocolor negative film 2445 was used for the acquisition of 1:24,000-scale, natural color aerial photographs in December 1999 (AGRA Baymont, 2001). End lap and side lap were required at 60% and 30%, respectively.

A Differential Global Positioning System unit was used to collect 20 ground control points. Also, photoidentifiable "pass points" (between frames following the flight lines) and "tie points" (between flight line strips) were selected and included to the previously surveyed ground control point network. Additionally, ground control points from the SWFWMD aerial photography photointerpretation project in 1999 were included to guarantee an accurate tie between the two areas and resulting maps. This suite of control points was used to accomplish analytical aerotriangulation of the aerial photographs well within the USGS National Map Accuracy Standards for 1:24,000-scale maps.

Prior to photointerpretation, site visits were made for the purposes of advance photosignature identification and investigation of unusual signatures. Photographs and GPS coordinates were taken during these site visits. A photointerpretation key was developed with definitions of each classification category, photographs from site visits, and images of aerial views to be used as a guide throughout the interpretation process.

Covertype boundaries were photogrammetrically digitized by using CADMAP®/dgn software. The data were initially captured digitally into MicroStation NT then later exported to ArcInfo. Classifications of vegetation covertypes were made based on FLUCCS as defined below (The minimum mapping unit was 0.10 ha or 0.25 acres.):

Continuous, dense seagrass beds. The dominant feature of these seagrass beds was continuity (>85% cover), but there may have been variable density within the bed. These beds contained interspersed areas of unvegetated bottom (<10% cover); however, unvegetated bottom or sand patches greater than 0.1 ha (0.25 acres) should have been distinguished within a continuous seagrass bed. The percent cover for continuous seagrass was approximately 85%.

Patchy seagrass beds. Patchy or discontinuous seagrass beds (<10% and >85% cover) greater than 0.1 ha (0.25 acres) that may also have been of variable density.

Algal beds. Beds of attached algae that were distinguishable from seagrass. Where feasible, they were mapped and field verified if >10% cover.

Groundtruthing of completed photointerpretation work involved visiting 40 sites and verifying actual cover type. Sites were selected to include complex areas (i.e., seagrass density variations and algae presence) and the variety of classification categories. These field checks were used as quality assurance checks for verification of accuracy.

Methodology Employed To Analyze Historical Trends

In 1983, the Florida Department of Natural Resources (FDNR) and the Florida Department of Transportation (FDOT) produced a document and associated maps that examined the land use of Charlotte Harbor and Lake Worth Lagoon, Fla. (Harris and others, 1983). Black and white photographs from 1946 and 1951 (referred to in the original document as the 1945 set) were acquired, and 1:24,000-scale, positive, false-color infrared transparencies were produced from flights in April 1982. The 1982 photographs were analyzed with stereoscopic visual equipment; the 1945 photographs did not have the required overlap and endlap to perform stereoscopic analyses. For the 1945 and 1982 photographs, seagrass was delineated onto Mylar® overlays, and then the data were digitized into the FDOT proprietary point-vector database. Maps were produced at 1:24,000 scale.

Classification of the seagrass from the 1945 aerials was performed to the level of only one category (submerged aquatic vegetation) because the quality of the photographs did not allow multiple category classification. The 1982 seagrass was classified to three categories:

- 901 Sparse underwater vegetation: this class was characterized by approximately 70% or more exposed sand in the actual meadow regardless of the patchiness observed within the meadow;

- 903 Moderate-to-dense underwater vegetation: this class encompassed all contiguous meadows with approximately 30% or less uniformly exposed sand; and

- 904 Patchy underwater vegetation: this category was characterized by large unvegetated patches within areas of 1 m^2 or more moderate to dense grass.

For this report, coverages for all datasets, including subsequent mapping efforts, are reported as combinations of all seagrass classification categories employed (i.e., sparse, dense, patchy) and are summarized in table 2.

The Florida Marine Research Institute (FMRI) currently holds the original Mylar® overlays for the 1945 and 1982 photographs. In 1990, FMRI staff registered the original 1982 Mylar® overlays to 7.5 minute quadrangles and redigitized the seagrass polygons to create a digital ArcInfo file. Also, to fill a void in the 1982 mapping effort that did not include southern Estero Bay, FMRI interpreted 1982 Estero Bay extent from 1990 photographs. Because the 1982 data exist in digital format, calculations of seagrass extent based on the 14 seagrass segments were possible for this report and are used as historical data for comparison with the more recent mapping results (fig. 9).

Lack of digital data for the 1945 effort, however, prohibit examination of the 14 seagrass segments for this effort. The total coverage for the 1945 analysis is included briefly in the discussion below, but it is not used for seagrass trend analysis. Seagrass coverage for the 1945 maps were evaluated by USGS quadrangle areas rather than the 14 segments that have been used to define coverage for this report. Also, the geographic boundaries for the 1945 study area do not match the boundaries of the 1982 and subsequent year analyses. (The original 1982 study area boundaries were identical to the boundaries of the 1945 study area boundaries; however, SWFWMD and FMRI recalculated the acreages based on the 14 segments used for recent Charlotte Harbor seagrass investigations as explained above.) Therefore, a comparison of total extent from the 1945 to recent efforts is not possible. Finally, the black and white photographs used in the 1945 effort were of too low a quality for delineating seagrass, and the absence of ground-verification during the year the photographs were produced is reason for caution when examining these data and resulting maps.

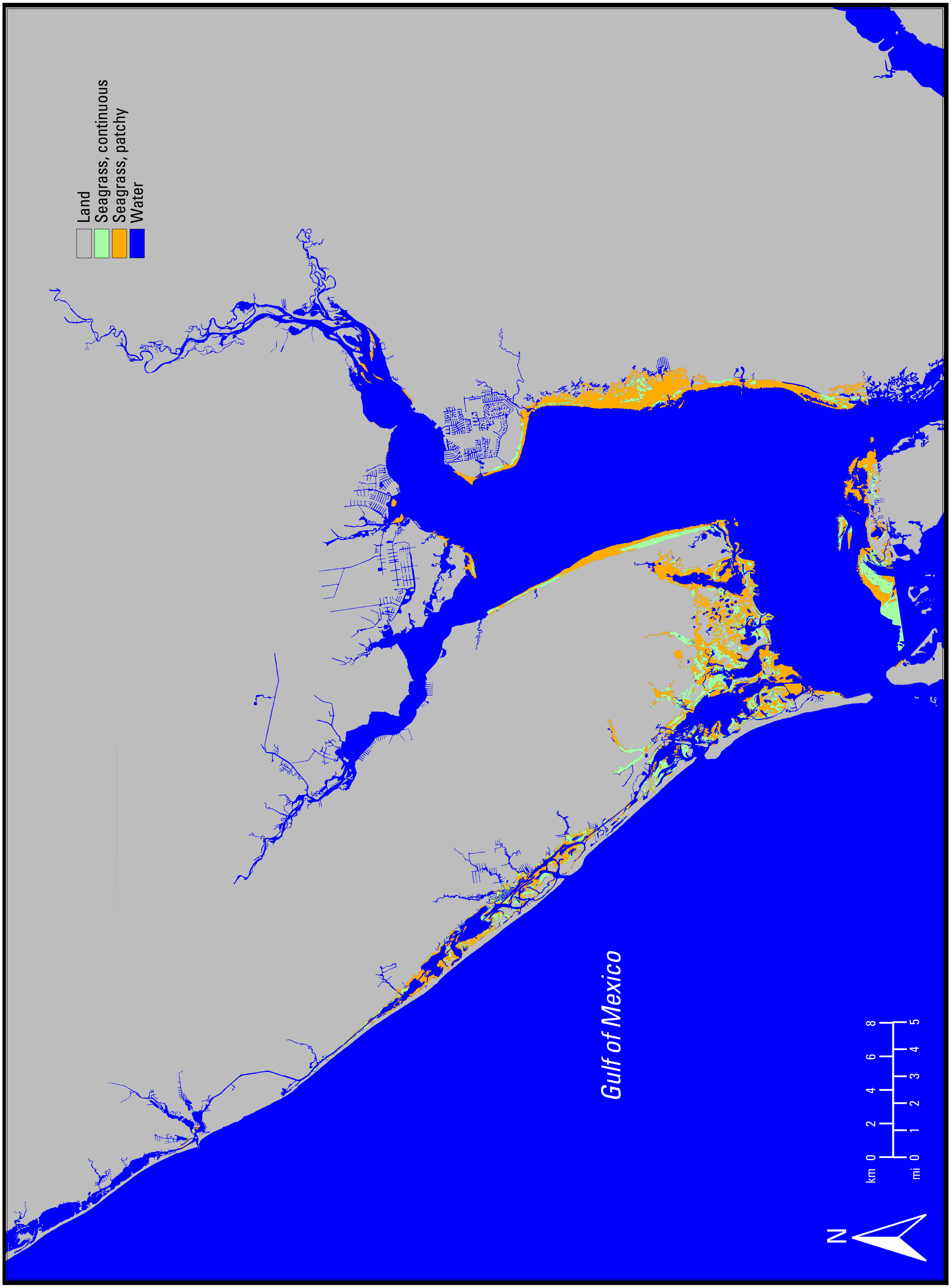

Figure 3. Distribution of seagrass in the northern Charlotte Harbor system, 1988.

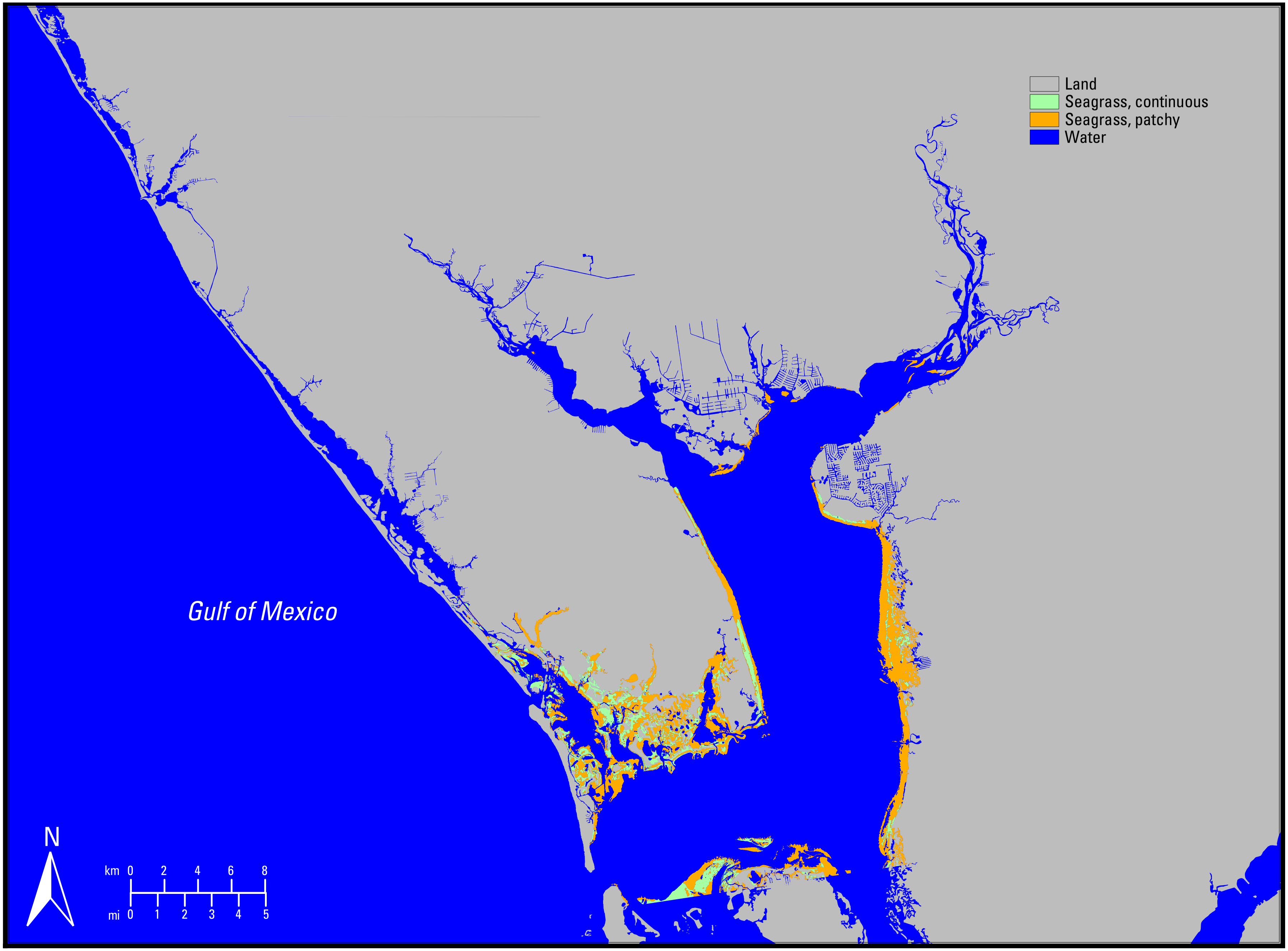

Figure 4. Distribution of seagrass in the northern Charlotte Harbor system, 1992.

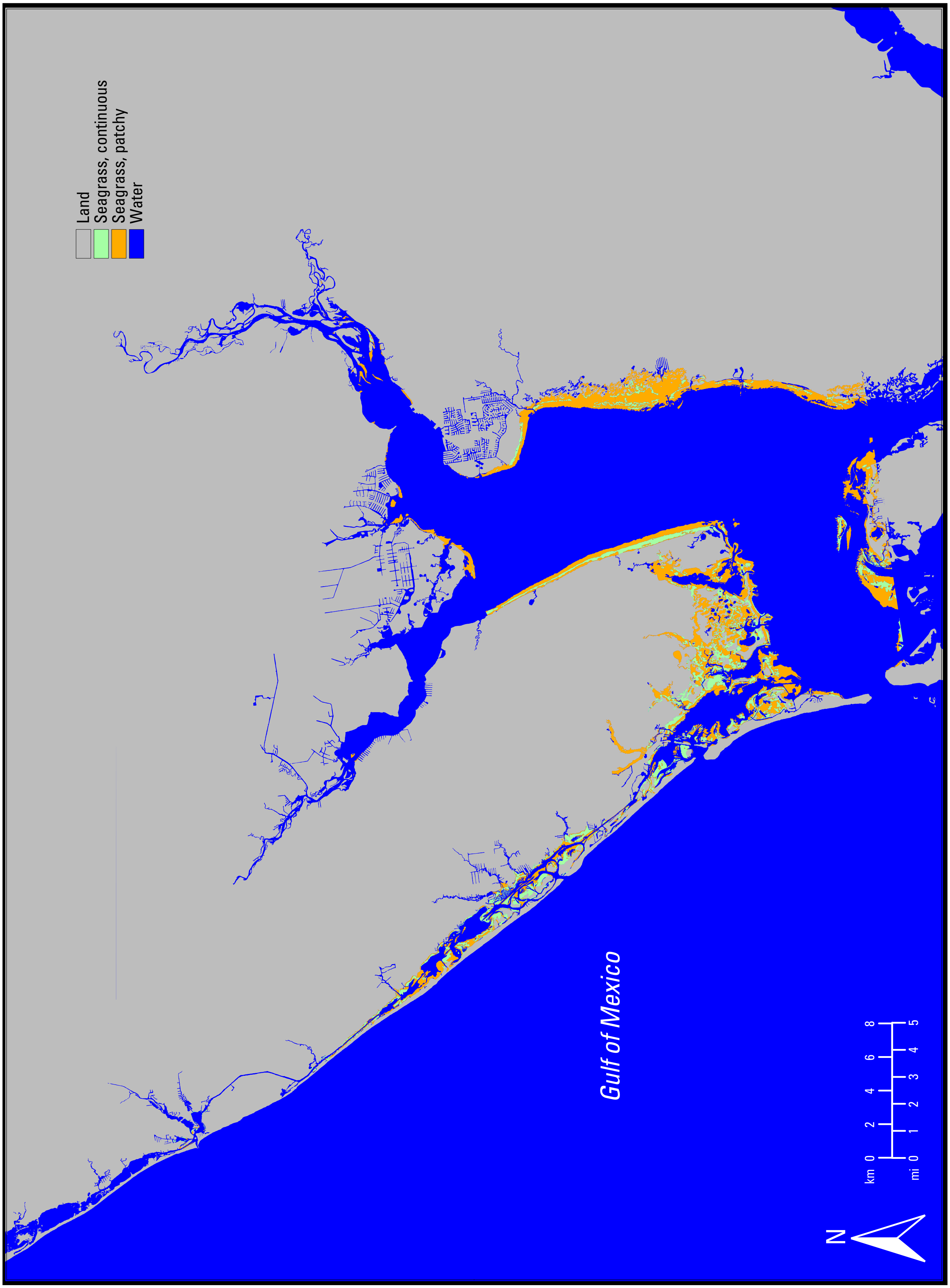

Figure 5. Distribution of seagrass in the northern Charlotte Harbor system, 1994.

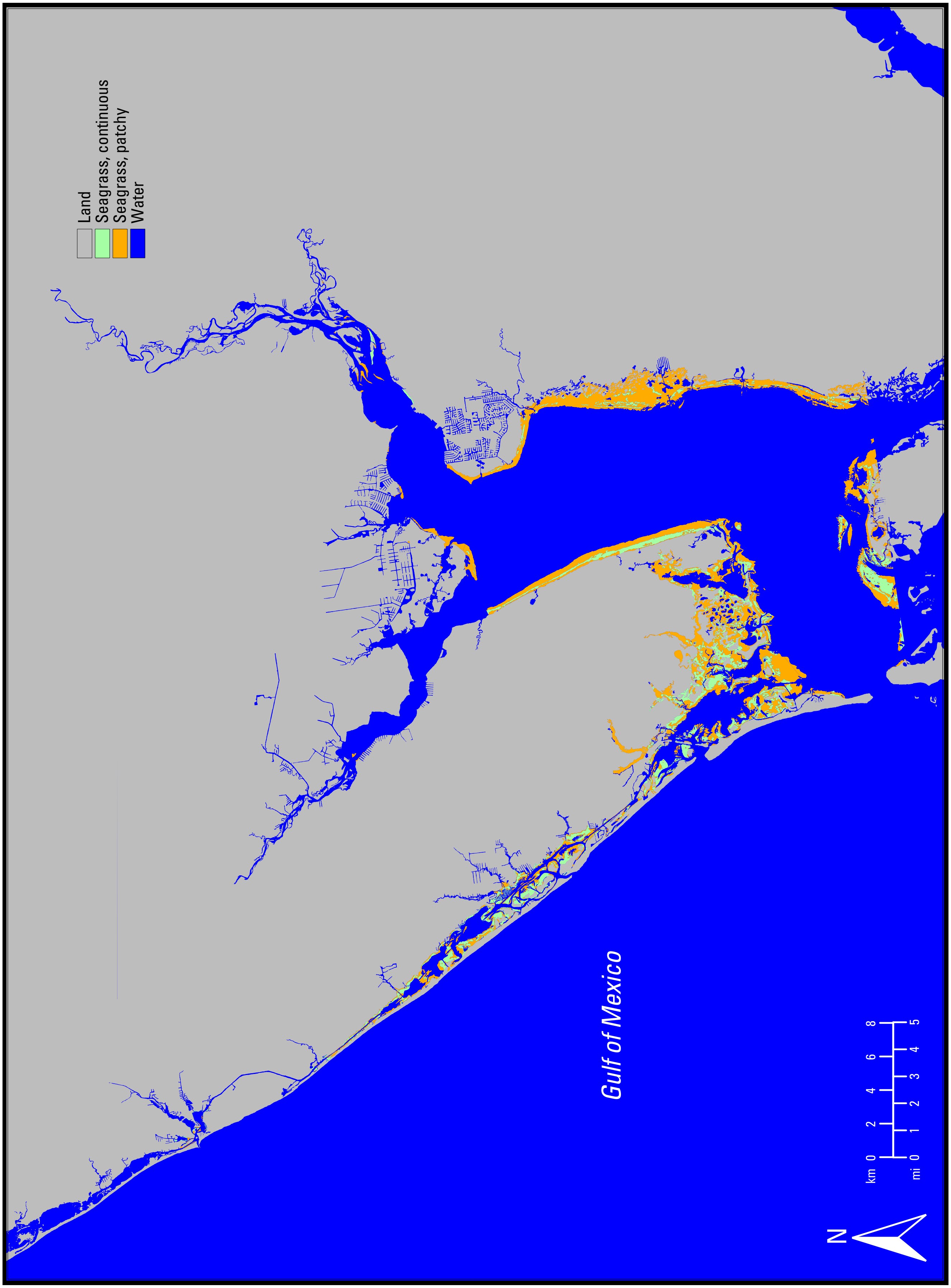

Figure 6. Distribution of seagrass in the northern Charlotte Harbor system, 1996.

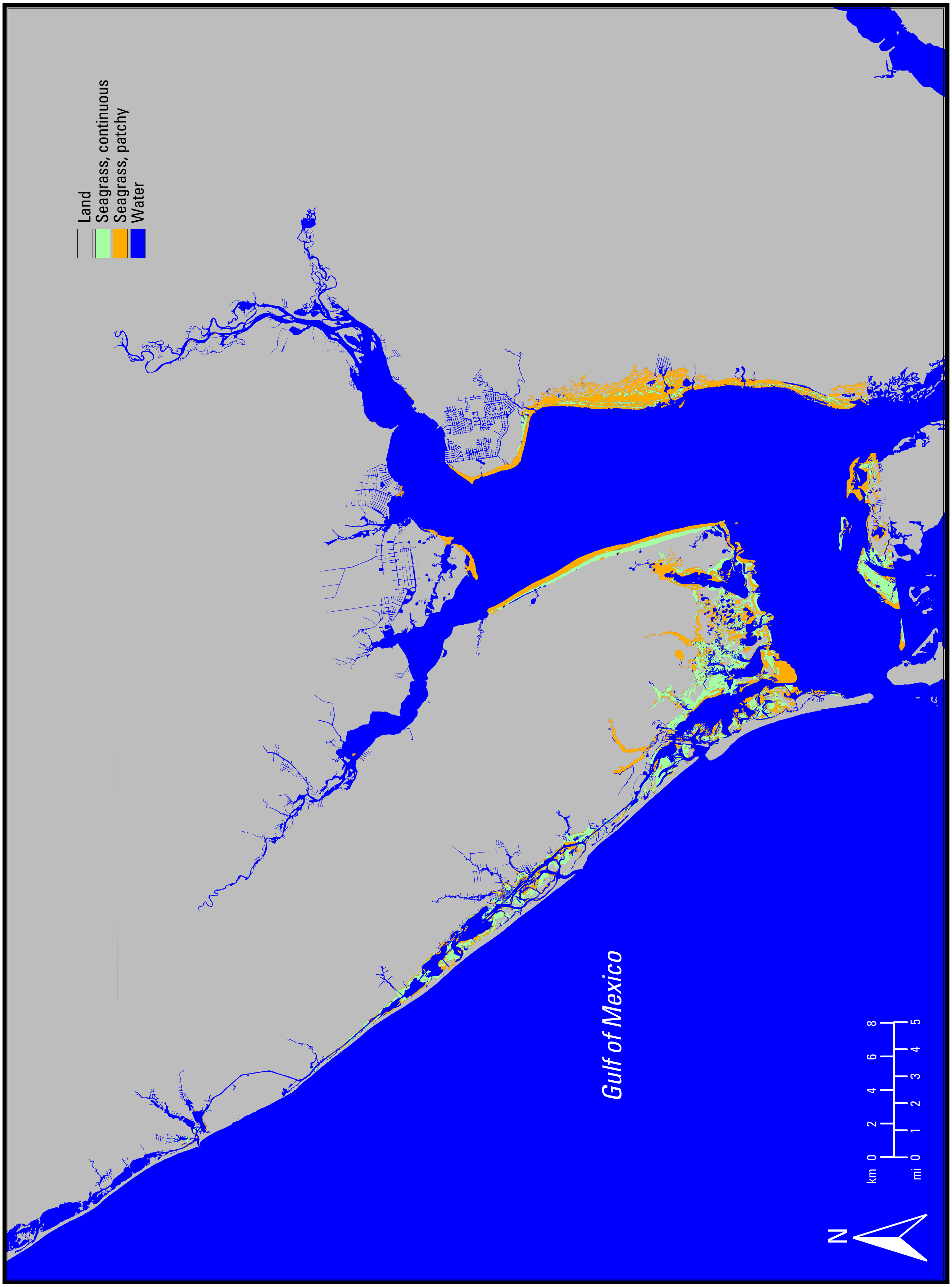

Figure 7A. Distribution of seagrass in the northern Charlotte Harbor system, 1999.

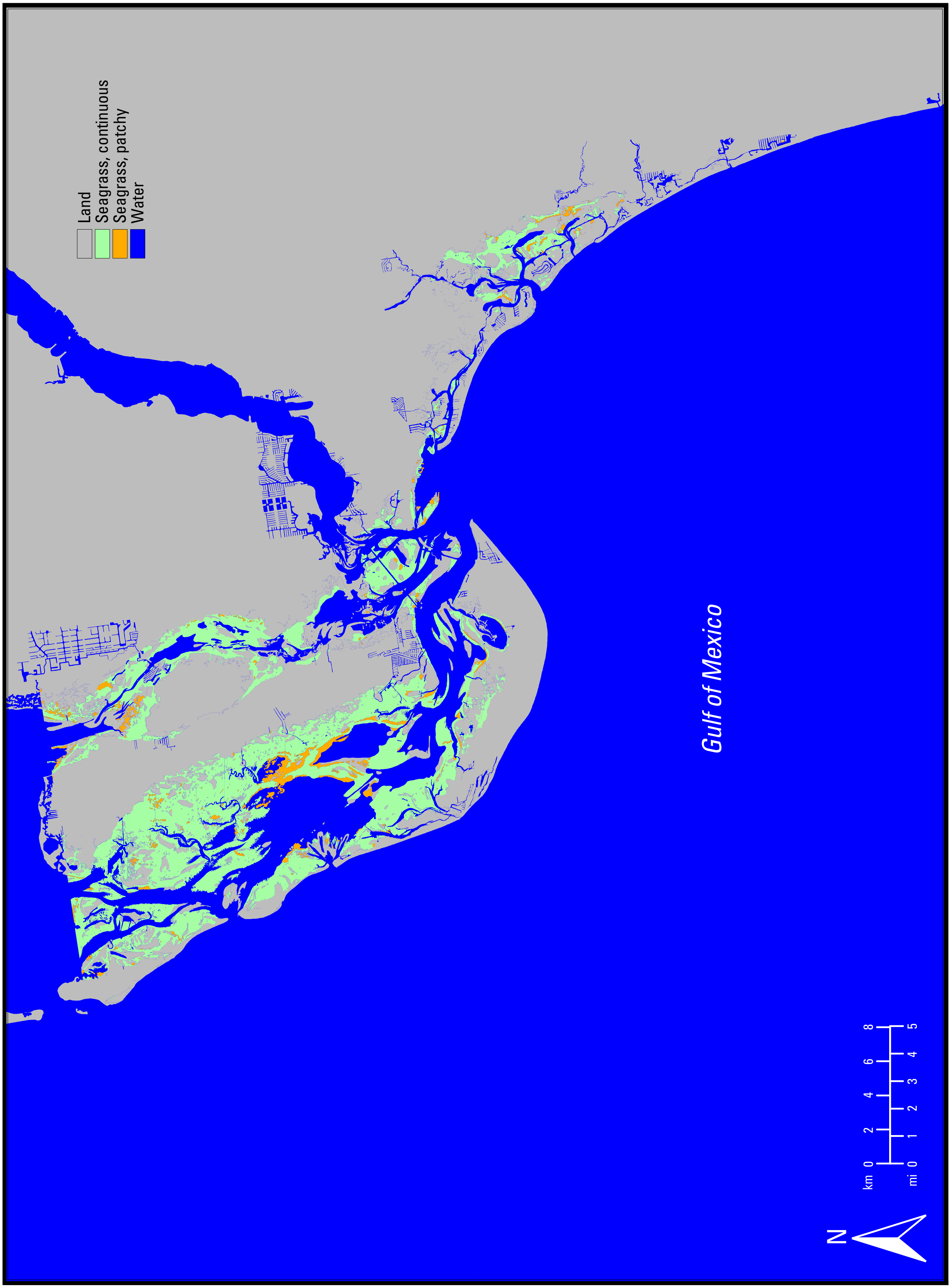

Figure 7B. Distribution of seagrass in the southern Charlotte Harbor system, 1999.

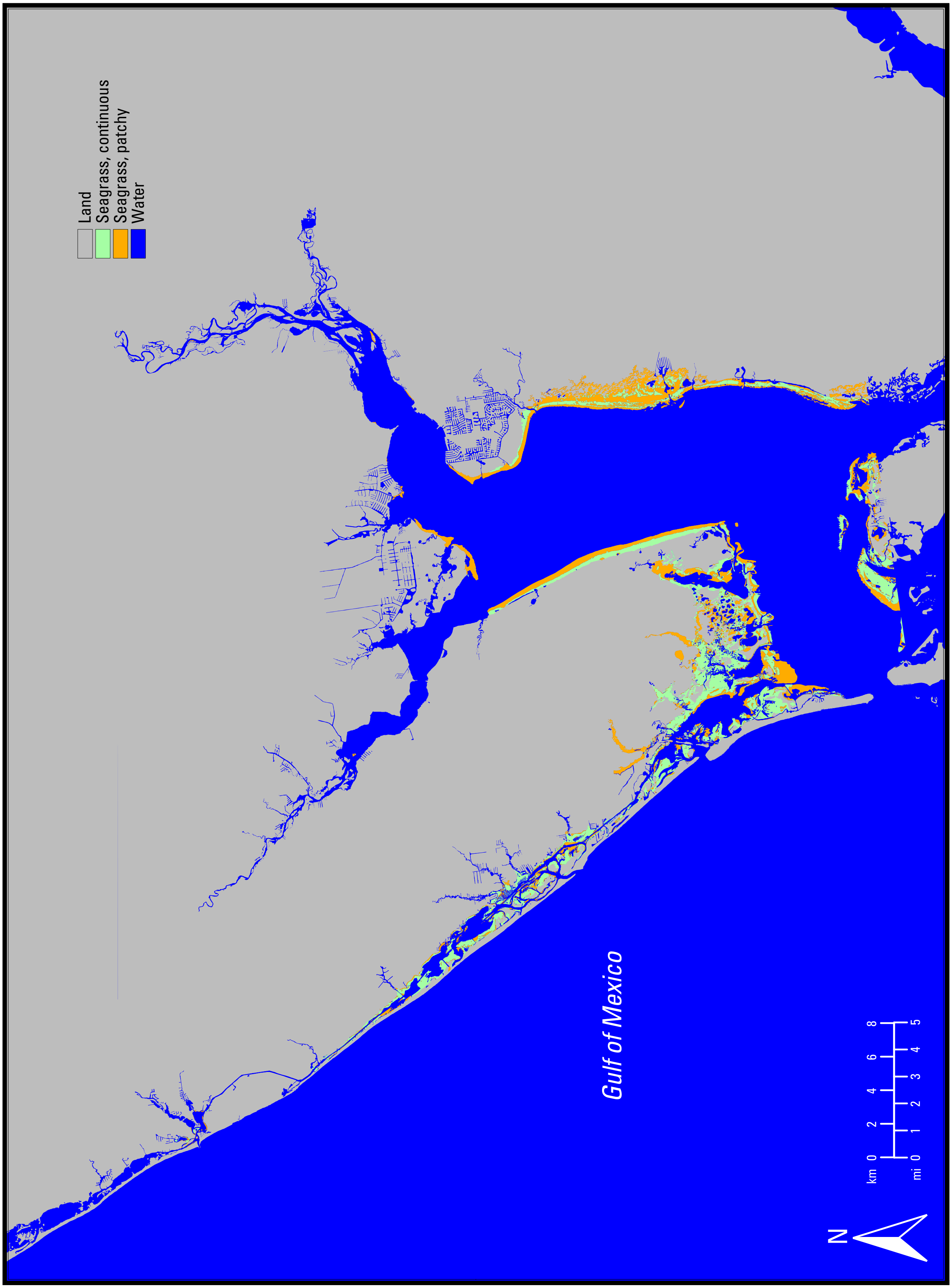

Figure 8. Distribution of seagrass in the northern Charlotte Harbor system, 2002.

Table 2. Seagrass coverage in hectares (acres) by year in the 14 subsegments of the greater Charlotte Harbor region.

[Note: 1982 coverage calculations differ from Harris and others (1983) report because of subsegment boundary delineations. The 1983 report calculated coverage within USGS quadrangles; 1988–96 extent taken from Kurz and others (2000)]

Subsegment	Year						
	1982*	1988	1992	1994	1996	1999	2002
Lemon Bay		1,055 (2,607)		1,073 (2,651)	1,054 (2,604)	1,044 (2,580)	1,043 (2,577)
Myakka River	238 (588)	202 (499)	130 (321)	189 (467)	209 (516)	191 (472)	185 (457)
Peace River	378 (934)	158 (390)	166 (410)	196 (484)	225 (556)	109 (269)	137 (339)
Charlotte Harbor (northern)							
East Wall	1,548 (3,825)	1,372 (3,390)	1,361 (3,363)	1,416 (3,499)	1,371 (3,388)	1,452 (3,588)	1,454 (3,593)
West Wall	672 (1,660)	585 (1,445)	495 (1,223)	675 (1,668)	794 (1,962)	699 (1,727)	699 (1,727)
Middle Harbor	70 (173)	50 (124)	50 (124)	60 (148)	76 (188)	63 (156)	64 (158)
Placida Region	948 (2,343)	1,408 (3,479)	1,376 (3,400)	1,337 (3,304)	1,450 (3,583)	1,503 (3,714)	1,531 (3,783)
South Harbor	3,513 (8,681)	3,684 (9,103)	3,636 (8,985)	3,633 (8,977)	3,626 (8,960)	3,340 (8,253)	3,313 (8,186)
Subtotal (excluding Lemon Bay)	7,367 (18,204)	7,458 (18,429)	7,214 (17,826)	7,505 (18,545)	7,751 (19,153)	7,357 (18,179)	7,383 (18,243)
Charlotte Harbor (southern)							
Pine Island Sound	9,853 (24,347)					10,484 (25,906)	
Matlacha Pass	3,245 (8,018)					2,456 (6,069)	
San Carlos Bay	2,420 (5,980)					1,504 (3,716)	
Upper Caloosahatchee River	0					0	
Lower Caloosahatchee River	242 (598)					1 (2)	
Estero Bay	2,504 (6,187)*					1,008 (2,491)	
Subtotal (excluding Estero Bay)	15,760 (38,943)					14,445 (35,694)	
Grand total	25,631 (63,334)	8,513 (21,036)	7,214 (17,826)	8,578 (21,196)	8,805 (21,757)	23,854 (58,943)	8,426 (20,821)

*The Florida Marine Research Institute interpreted Estero Bay from 1990 photographs to fill a void in the 1982 photographs.

Status and Trends

Since the first mapping of seagrass in Lemon Bay in 1988, coverage calculations have remained relatively consistent through the 2002 time period. However, seagrass coverage in Estero Bay may have dramatically changed because a 60% (1,496 ha or 3,697 acres) decrease was recorded from 1982 to 1999 (note: coverage estimates for the southern portion of Estero Bay reported in these 1982 data were actually obtained from 1990 photographs). It should be noted that mapping methodologies between these two time periods were not identical, and groundtruthing was scarce in Estero Bay for the 1982 study (Frank Sargent, oral communication). Field verification work for the 1999 study (as well as a preliminary study done in 2003) found large areas of accumulated drift algae over sand and seagrass areas; therefore, it is possible that the Estero Bay coverage in the 1982 study incorporates drift algae wracks. The decrease in coverage in Estero Bay from 1982 to 1999 must be viewed with these caveats in mind.

For the following comparisons of seagrass area in the greater Charlotte Harbor estuarine complex, area calculations for Lemon Bay and Estero Bay are not included. The first reliable seagrass interpretation work for Charlotte Harbor was created with the 1982 photographs. Interpretation of the photographs resulted in 23,127 ha (57,147 acres) total seagrass. In 1999 the most recent comprehensive mapping project for the greater Charlotte Harbor estuarine complex produced seagrass estimates of 21,802 ha (53,873 acres). Thus, from 1982 to 1999 the mapping results indicate an overall 6% (1,325 ha or 3,274 acres) decrease of seagrass (see table 3).

Combined estimates for the seven segments in Charlotte Harbor under the SWFWMD jurisdiction (Myakka River, Peace River, East Wall, West Wall, Middle Harbor, Placida Region, and South Harbor) have fluctuated up and down within a variance of less than 1,236 ha (3,054 acres) since the 1982 study. The 1999 extent is only 10 ha (25 acres) less than the 1982 value, while the 2002 coverage is 16 ha (39 acres) greater (fig. 10).

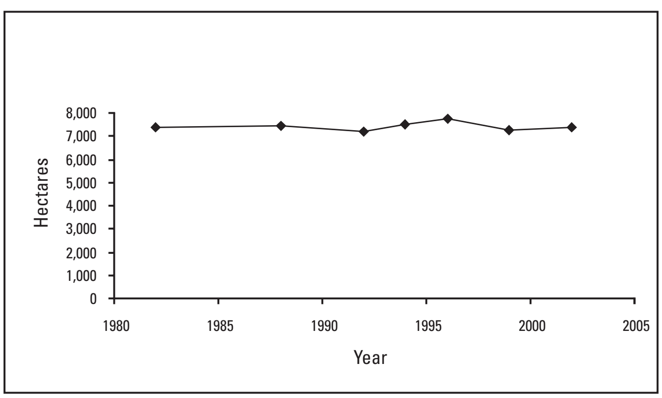

Figure 9. Seagrass extent in the northern Charlotte Harbor region (except Lemon Bay) since 1982.

Table 3. Comparison of 1982 to 1999 seagrass coverage by subbasin (in hectares; conversions to acres are provided in table 2).

Segment	Year		Change	% Change
	1982	1999		
Myakka River	238	191	-47	-20
Peace River	378	109	-269	-71
Charlotte Harbor (northern)				
East Wall	1,548	1,452	-96	-6
West Wall	672	699	27	4
Middle Harbor	70	63	-7	-10
Placida Region	948	1,503	555	59
South Harbor	3,513	3,340	-173	-5
Subtotal	7,367	7,357	-10	0
Charlotte Harbor (southern)				
Pine Island Sound	9,853	10,484	631	6
Matlacha Pass	3,245	2,456	-789	-24
San Carlos Bay	2,420	1,504	-916	-38
Upper Caloosahatchee River	0	0	0	0
Lower Caloosahatchee River	242	1	-241	-100
Subtotal	15,760	14,445	-1315	-8
Total	23,127	21,802	-1325	-6

The five Charlotte Harbor segments within SFWMD jurisdiction (Pine Island Sound, Matlacha Pass, San Carlos Bay, lower Caloosahatchee River, and upper Caloosahatchee River) constitute the majority of the seagrass coverage in the Charlotte Harbor region—almost double that of the northern region in 1999, for example. These five segments have experienced an 8% (1,315 ha or 3,249 acres) decrease in seagrass from 1982 to 1999. This southern area accounts for approximately 77% of the total 6% seagrass coverage decline in the greater Charlotte Harbor region from 1982 to 1999.

Causes of Change

The subbasins included in the Charlotte Harbor Estuarine Complex face disparate issues that can result in changes of seagrass extent from historical conditions. Northern Charlotte Harbor appears to lack a significant trend in seagrass extent since the 1982 mapping effort, and it appears that seagrass extent in this subbasin is largely a factor of inflows from the Peace and Myakka Rivers. Nonetheless, in other subbasins within the Charlotte Harbor region, there is growing concern

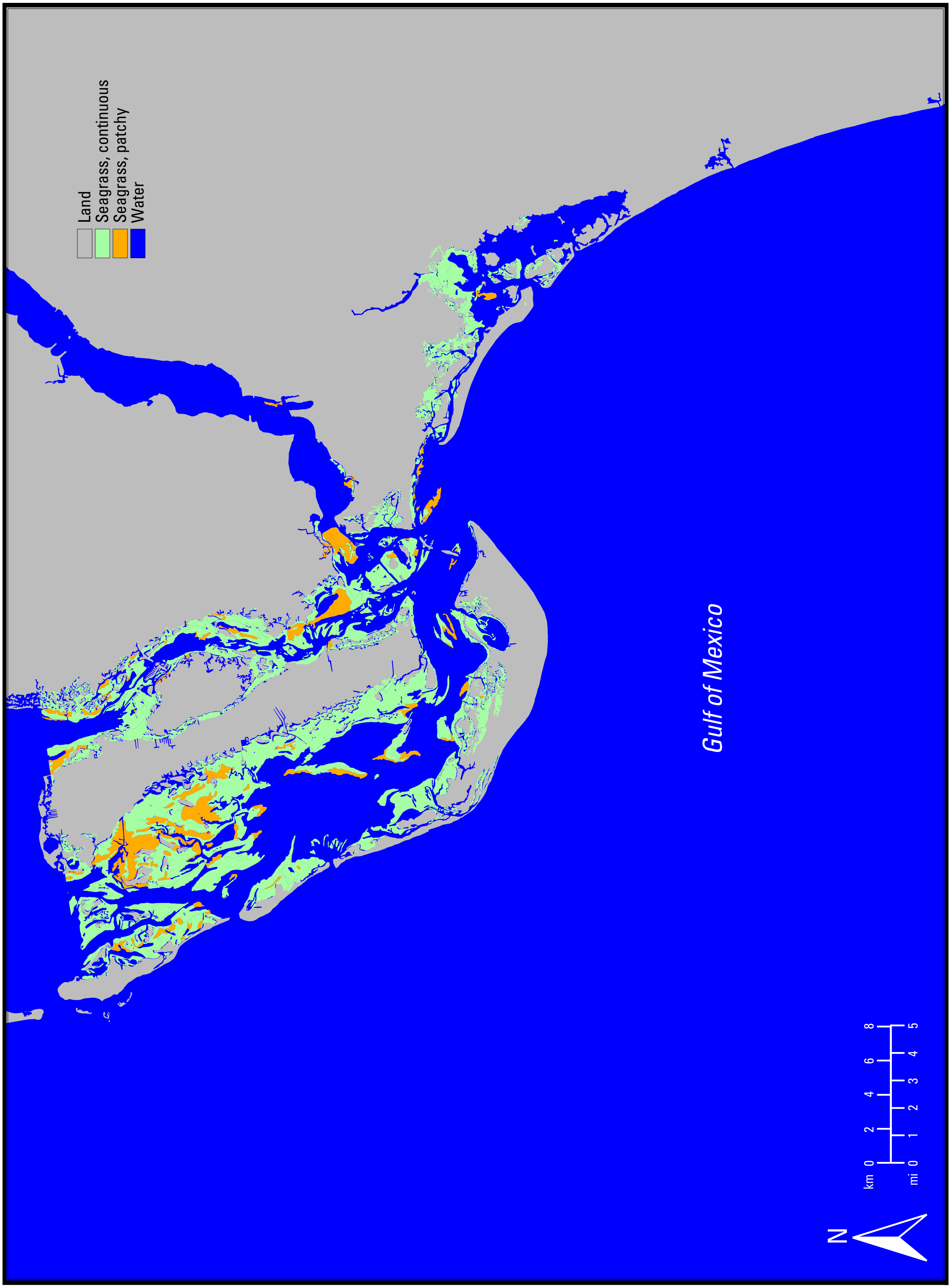

Figure 10. Distribution of seagrass in the southern Charlotte Harbor system, 1982.

that seagrasses are being harmed by human impacts. For instance, seagrass coverage in Lemon Bay showed a 1.8% decrease from 1994 to 1996 (Kurz and others, 2000); this coverage then remained constant in 1999 and 2002. The potential small decrease may be linked to decreases in water clarity associated with increases in nutrient loads (Tomasko and others, 2001) and resuspension of estuary bed sediments (Judy Ott, oral communication). There is also documentation of a large loss of seagrass coverage in southern Charlotte Harbor from the 1940s because of physical alterations and resulting changes in flow patterns within this region. The seagrass beds in all three of the rivers demonstrate marked changes in extent with each mapping effort, perhaps in part because of photointerpretation issues but also because of concomitant changes of river flows in the Peace and Myakka Rivers and water management in the Caloosahatchee River. Finally, the Charlotte Harbor region is considered one of the most severely impacted regions of Florida by boat propeller scarring (Sargent and others, 1995).

The most comprehensive review of historical seagrass extent for the Charlotte Harbor region is the 1983 FDNR report described above (Harris and others, 1983). This report documented a 29% decrease in seagrass, from 33,572 ha (82,959 acres) to 23,672 ha (58,495 acres) between 1945 and 1982 for the harbor, excluding Lemon and southern Estero Bays. As explained above, neither the study area boundary nor the segment delineations for the 1945 data within the 1983 report correlate with this current effort and are therefore problematic for use in direct coverage comparisons with subsequent year efforts; however, the report did theorize about the causes of change in the region that is very useful for this discussion. The authors noted a decrease in seagrasses in the deep edges of seagrass beds and the deeper portions of the harbor, and they theorized that the loss was a result of decreasing water clarity with increasing pollutants and changing drainage patterns. The study also determined that 40% of the total region-wide loss was located solely within the lower Pine Island Sound area. When combined with the loss of southern Matlacha Pass and San Carlos Bay areas, this loss equaled 57% of the total Charlotte Harbor region-wide loss of 29%. The authors contributed this substantial loss to the dredging of the Intracoastal Waterway and construction of a bridge and multiple causeway islands to Sanibel Island in the 1940s and 1960s. Using nautical maps, the authors noted that a shallow bar less than 1.5 m (5 ft) in depth with deeper channels (2.4–4.6 m or 8–15 ft) on either side, had served as a tidal node that extended entirely across Pine Island Sound. During an ebb tide, flow occurred to the north and south of this bar. This shallow bar was apparently the location of the one of the first channel dredging operations in the 1940s. In the 1960s, the Intracoastal Waterway was dredged through Pine Island Sound and up the Caloosahatchee River, and the Sanibel Causeway, including its spoil islands, was constructed across San Carlos Bay. The authors reasoned that the diversion of water from the Caloosahatchee River into Pine Island Sound and the changes of circulation patterns as a result of these two projects have lowered the salinities in the lower Caloosahatchee River estuary, San Carlos Bay, Matlacha Pass, and Pine Island Sound areas. Also, during the dredging of the Intracaostal Waterway and construction of the Sanibel Causeway, direct loss of seagrasses was due to the excavation and sidecast fill over the nearby seagrass beds (cited in Harris and others, 1983; James Beever, written communication). Following both projects, loss of seagrasses ensued because of turbidity and spoil spread (James Beever, written communication).

Indirect impacts to seagrass extent in Charlotte Harbor from pollutant loads have not been documented. Relative to Tampa, Sarasota, and Lemon Bays to the northwest, Charlotte Harbor is highly influenced by the freshwater inflows of its large watershed. The surface area of the harbor is approximately 700 km^2 (270 mi^2), whereas the watershed is almost 11,300 km^2 (4,362 mi^2), a ratio of watershed to open water of over 12:1 (Southwest Florida Water Management District, 2000). The result of this large watershed is that the waters of the harbor are often a dark, brownish, tea color caused by the influx of tannins and suspended mater from the watershed. For instance, a 1987 study found that nonchlorophyll suspended matter (including detritus, cellular material, and minerals) accounts for an average of 72% of light attenuation in the water column, color (dissolved matter) accounts for 21%, phytoplankton chlorophyll for 4% and water itself the remaining 3% (McPherson and Miller, 1987), and another evaluation of light attenuation in 1999 found that color, turbidity and chlorophyll accounted for 66%, 31% and 4% of light attenuation (Dixon and Kirkpatrick, 1999). Water clarity in the harbor increases with increasing salinity (McPherson and Miller, 1987; McPherson and Miller, 1994; Dixon and Kirkpatrick, 1999; Tomasko and Hall, 1999). Thus, the light reaching the tops of seagrass beds is largely a factor of basin runoff and flows from the three major tributaries—the Peace, Myakka, and Caloosahatchee Rivers (McPherson and Miller, 1987; Tomasko and Hall, 1999; Doering and Chamberlain, 1999). In turn, seagrass coverage changes in the harbor and the tidal reaches of the rivers are thought to be a function of changes in these freshwater inflows. For example, the East Wall segment of Charlotte Harbor has in general consistently more seagrass coverage and more stable meadows than the West Wall segment (the latter demonstrated by the greater average species abundance of turtle grass; Staugler and Ott, 2001), probably because of the freshwater inflows from the Peace and Myakka Rivers that concentrate more towards the Western Wall. Also, in general the maximum depths of seagrass beds increase with increasing distance from the mouths of the Peace and Myakka Rivers and increasing salinities (Dixon and Kirkpatrick, 1999).

In comparison, relative to Charlotte Harbor, the Lemon Bay subbasin has a much smaller watershed, with a ratio of watershed to open water of 5:1, and water clarity is much more strongly tied to phytoplankton levels (Tomasko and others, 2001). Phytoplankton biomass was calculated to contribute 12% to 39% of light attenuation within the water column with a mean percent of 29%, and depth distribution of seagrasses in Lemon Bay is largely a factor of chlorophyll a concentrations (Tomasko and others, 2001). Kurz and others (2000) noted a small decrease in seagrass coverage in Lemon Bay from 1994 to 1996, and this coverage then remained stable from 1996 to 2002 (table 2). The decrease represents less than a 2% loss and falls well within the margin of sampling error and so should be viewed accordingly. Nevertheless, with increasing urbanization pressure and accompanying increases in nonpoint source nutrient loads, there is increasing concern over the preservation of the seagrass beds in this subbasin. Annual nitrogen loads to the bay were estimated to have increased 59% since predevelopment (1850) to 1995 conditions, and they are predicted to increase 45% by 2010 from 1995 levels (Tomasko and others, 2001). Septic tank systems are thought to play a significant role in this increase, especially during the 9-mo dry season. Septic tanks are estimated to contribute 28% of the nitrogen load October through June and 14% July though September (Tomasko and others, 2001). Finally, Lemon Bay is also relatively shallow, and boating activities help stir up bottom sediments, further aggravating water-clarity problems.

In the southern Charlotte Harbor region, there has been less documentation of seagrass extent than in the northern areas, and less is understood about possible coverage changes and causes of such over time. In the northern Pine Island Sound and Matlacha Pass segments, suspended matter and dissolved matter are still the dominant factors in light attenuation, and turbidity increases in impact as one moves south in either region (McPherson and Miller, 1987; Dixon and Kirkpatrick, 1999). McPherson and Miller (1987) found that Pine Island Sound was dominated by noncolored waters from the Gulf of Mexico, while parts of Matlacha Pass and San Carlos Bay were affected by local runoff and discharges from the Caloosahatchee River. Suspended matter, however, the source of which appeared to be the bed of the estuary (McPherson and Miller, 1987), was the major contributor of light attenuation in the southern regions of these segments (McPherson and Miller, 1987; Dixon and Kirkpatrick, 1999). In the tidal head of the Caloosahatchee River and San Carlos Bay, light is attenuated mostly by dissolved materials or color, while in the more upper reaches of the river, chlorophyll a may be the major contributor (Doering and Chamberlain, 1999). As discussed earlier, however, a significant loss of seagrasses from historical conditions in the southern Charlotte Harbor region stemmed from the physical alterations caused by the Gulf Intracoastal Waterway dredging and Sanibel Causeway construction. Changes in seagrass extent in these areas largely result from inflows from the Caloosahatchee River that now flow into lower Pine Island Sound and Matlacha Pass and the resulting reduced salinities (Harris and others, 1983) and possible resuspension of sediments (McPherson and Miller, 1987).

Little is known about possible causes of coverage changes where coverage existed in Estero Bay, but anecdotal evidence points to resuspended sediments as the major problem with light reaching seagrass beds. At the same time, there is also strong evidence of increasing nutrient enrichment of the tributaries into this subbasin. A new water quality-monitoring program by the local county government that collects light attenuation and supplemental data and the expansion of both seagrass mapping and transect monitoring by the SFWMD and FDEP into the bay will help fill the knowledge gaps in this area.

Finally, the scarring and subsequent loss of seagrass beds by boat propellers has been a significant issue in the entire Charlotte Harbor region. Most of Charlotte Harbor is relatively shallow, averaging only 2.1 m (7 ft) in depth, with a deep depression in Boca Grande Pass (22 m or 72 ft) (Stoker, 1986), leaving it vulnerable to the propeller dredging of inexperienced or imprudent boaters. A 1995 effort by the State determined that the Charlotte Harbor region is one of the most heavily scarred areas in Florida (Sargent and others, 1995). Simultaneously, the area faces the pressures of a hearty tourism industry and a rapidly growing population, which includes an increase in boating activities and in the size of those boats. Area resource managers face a growing number of requests to allow dredging for greater access to the more shallow areas and the development of private docks for riparian landowners or marinas. A study of 27 docks averaging 113.94 m^2 (1,226 ft^2) in total area constructed over grass beds in Pine Island Sound and San Carlos Bay found boat propeller dredging associated with roughly one- third of the docks and an average area of 6.89 m^2 (74 ft^2)of dredged area per dock (Loflin, 1995). Additionally, the study found an average 128.84 m^2 (1,386 ft^2) seagrass “shadow” or area of seagrass loss associated with the total size of each dock. The study did not address the effects of changes in dock dimensions (height and width) or the possible cumulative impacts of docks to seagrasses on a region-wide basis, but it is apparent that boating activities are having a deleterious impact on the seagrasses of Charlotte Harbor.

Monitoring for Seagrass Health

As a supplement to the mapping efforts through aerial photography to estimate seagrass extent, the FDEP—Charlotte Harbor Aquatic Preserves Office has established a series of 50 transects distributed over most of the six subbasins (excluding only Estero Bay and tidal Caloosahatchee River). Beginning in 1999 these 50 fixed transects are visited annually during September through early November to determine declines or improvements in seagrass conditions by detecting changes in seagrass depth distributions, epiphyte coverage, short shoot densities, and species composition. Program researchers use the Braun-Blanquet Cover Abundance Scale (for more information, see *http://chla.library.cornell.edu/cgi/t/text/text-idx?c=chla;idno=2917578*) to estimate seagrass species abundance for individual species and collect blade length, sediment type, and epiphyte coverage and type at 50-m (164-

ft) intervals along each transect (or 10-m or 33-ft intervals for transects shorter than 50 m). Also, water quality data, such as photosynthetically active radiation and salinity, and short shoot density are gathered at mid bed. In 2000 the quarterly collection of seagrasses for disease analysis of turtle grass was added to the program. In 2002 this monitoring program was expanded to include the Caloosahatchee River and five transects within Estero Bay.

Species Information

The FDEP transect monitoring program allows for the determination of species composition within the various major seagrass beds in the Charlotte Harbor Estuarine Complex. Five seagrass species have been identified in these subbasins: turtle grass, shoal grass (*Halodule wrightii*), manatee grass (*Syringodium filiforme*), wigeon grass (*Ruppia maritima*), and star grass (*Halophila engelmannii*) (table 4). In 1999 and 2000, shoal grass was determined to occur most frequently throughout the region, with 55.6% and 62.9% occurrence and within each harbor segment, with the exceptions of Pine Island Sound and San Carlos Bay (Staugler and Ott, 2001). In Pine Island Sound, turtle grass occurred most frequently with 59.5% in 1999 and 65.9% in 2000. In San Carlos Bay, manatee grass, with 60% (1999) and 66.7% (2000) abundance, and turtle grass, with 46.7% (1999) and 77.8% (2000) coverage, were almost equally ubiquitous.

Table 4. Seagrass species present at the Florida Department of Environmental Protection transect monitoring sites (from Staugler and Ott, 2001).

[No transects were located within the Middle Charlotte Harbor, Caloosahatchee River, and Estero Bay segments in 2001]

Region	Number of Sites	Species Present
Lemon Bay	6	Shoal grass (*Halodule wrightii*), turtle grass (*Thalassia testudinum*), manatee grass (*Syringodium filiforme*)
Gasparilla Sound/ Placida Region	7	Shoal grass, turtle grass, manatee grass
Myakka River	6	Shoal grass, turtle grass (1999), wigeon grass (*Ruppia maritima*)
Peace River	6	Shoal grass
Western Charlotte Harbor	3	Shoal grass, turtle grass, wigeon grass (1999 in northernmost site)
Eastern Charlotte Harbor	4	Shoal grass, turtle grass
Pine Island Sound	9	Shoal grass, turtle grass, manatee grass
Matlacha Pass	6	Shoal grass, turtle grass, manatee grass (found at 1 transect in northern M.P.), star grass (*Halophila engelmannii*)
San Carlos Bay	3	Shoal grass, turtle grass, manatee grass

Anecdotal evidence suggests that wigeon grass can be found in the Peace and Myakka Rivers and may alternate in prevalence with water celery (*Vallisneria americana*), depending on river flows and salinity. The FDEP monitoring program located wigeon grass in the tidal Myakka River in both 1999 and 2000 and within the West Harbor Wall segment at its northernmost site in 1999. Star grass is found only in Estero Bay and Matlacha Pass. The recent expansion of the transect monitoring into Estero Bay in 2002 found the presence of shoal grass, turtle grass, and star grass in the bay. Finally, paddle grass (*Halophila decipiens*) has been documented in Estero Bay (Mackenzie, written communication) and the East Wall of Charlotte Harbor (Tomasko, written communication).

Mapping and Monitoring Needs

Southwest Florida is rapidly urbanizing, and there is a need to continually evaluate the impacts of this urbanization on invaluable estuarine resources. Currently, biennial seagrass mapping efforts exist in the entire Charlotte Harbor region that are important in allowing resources managers to ascertain the spatial and temporal changes of seagrass coverage in coastal waters. In the Charlotte Harbor region, the SFWMD and SWFWMD effectively share the responsibility of accomplishing these biennial seagrass maps. And while there has been much closer coordination and communication in the past several years between the districts for these efforts, there are several issues with the consistency of timing, accuracy of photointerpretation, data collection methodologies, and reporting that could be enhanced. For instance, intradistrict aerial surveys for the maps may be flown as late as April during one year's mapping effort and in November or December in another. Also, the interdistrict aerial surveys may be off by several months or more. In the most recent mapping effort, the SWFMD flew to their boundary—the "South Harbor" segment of northern Charlotte Harbor—in January 2002. The SFWMD, however, flew to their boundary in January 2003. These timing inconsistencies, caused in large part by uncontrollable water clarity and flight conditions, cause data comparability issues between these individual efforts. It is problematic to compare seagrass coverage data collected in April, during the winter dry season, to data collected in November, nearer the end of the summer wet season, because seagrass extent may vary substantially between the two seasons. Also, SFWMD may not include a postmap classification accuracy assessment requirement in their mapping efforts, while the SWFWMD requires a minimum of 90% accuracy. Although these relatively minor issues may continue to plague the mapping work in the future, the CHNEP and its many integral partners are fostering closer coordination between the two districts and spurring the use of similar mapping and postmap production methodologies. The mapping efforts by both districts are vital to the understanding and protection of coastal resources and need to continue.

In addition, there is now a transect monitoring program that encompasses the entire Charlotte Harbor Estuarine Complex. The fixed transects monitoring for seagrass health, species abundance, and bed length was expanded in 2002 into the Caloosahatchee River and Estero Bay and is expected to greatly broaden the region's knowledge of the seagrass beds in those subbasins. The FDEP Charlotte Harbor and Estero Bay Aquatic Preserves and South District offices conduct these monitoring programs, which can use up a great deal of their resources. For instance, because of the large size of their study area, the Charlotte Harbor Aquatic Preserves office alone monitors 50 transects with two dedicated staff members and a group of volunteers. This annual monitoring event usually occupies 2–4 d per week for approximately a 2-mo period. This program builds upon similar programs in the region, such as one in Tampa Bay, and is considered essential to the understanding of the health and quality of the essential fish habitat in the southwest Florida region.

Restoration and Enhancement Opportunities

Charlotte Harbor has not demonstrated a significant trend in overall seagrass extent since 1982, and therefore there has not been a strong impetus for seagrass restoration projects in the area, with the possible exception of the restoration of boat propeller scars. As explained above, the Charlotte Harbor region was determined to be one of the most severely impacted regions in Florida for boat propeller scarring in the 1995 statewide evaluation of the issue (see Sargent and others, 1995). Indeed, the local entities that make up the CHNEP incorporated the reduction of such scars as a major component of the CHNEP's management plan. In order to better understand this issue and determine future steps to alleviate the problem in Charlotte Harbor, the CHNEP and FMRI are currently undertaking a change analysis of the severity and location of propeller scars within the region since the 1995 effort. Also, there are several proposed efforts to attempt the restoration of heavily scarred areas by planting seagrass plugs or injecting nutrients in the scarred areas. FMRI is proposing a multiyear project to evaluate the various methods that seek to speed the recovery of boat propeller scars in seagrass beds and may use the Charlotte Harbor region in their efforts.

With support of the CHNEP, the SWFWMD Surface Water Improvement and Management (SWIM) Program has developed a pollutant load reduction goal for the northern Charlotte Harbor region that should reduce or maintain current nitrogen inputs into Charlotte Harbor. The goal was developed to alleviate human impacts on hypoxic events in the northern Charlotte Harbor region (Southwest Florida Water Management District, 1993). Hypoxia, or episodes of dissolved oxygen below 2 mg/L, occurs almost annually when the waters of northern Charlotte Harbor become stratified (Camp, Dresser, and McKee, Inc., 1998). When inflows from the Peace and Myakka Rivers reach over 3,785,400 m^3/day (1 billion gallons per day) during the summer wet season, the water column in the harbor stratifies, with the less dense fresh water flowing on top of the heavier, more saline waters. This flow creates a cap that reduces the movement of oxygen into the deeper waters. Nutrients and bacteria in the water column and sediments can combine to create a demand for oxygen that lowers the available oxygen in the water column. Increases in nutrient loading may be increasing the frequency, duration, and severity of the harbor's hypoxia events, and the district has proposed projects to reduce the nitrogen loads from the headwaters of the Peace River with a series of wetlands, media filtration, or settling ponds. The SWFWMD and the CHNEP hope these projects will at a minimum offset expected increases in nitrogen caused by development over the next decade. The reduction in nutrients may also have coincident benefits for the seagrass beds within northern Charlotte Harbor, because while nutrient inputs and resulting chlorophyll a concentrations may not contribute largely to light attenuation within the water column of Charlotte Harbor, they may cause heavier epiphyte loads on seagrass blades.

In Lemon Bay, there is an ongoing project by the SWIM Program with support from local governments and the CHNEP to determine the annual nitrogen loads from each of the six major tributaries into the bay. The district hopes that with this information, they, the CHNEP, and the local governments can then prioritize those basins that most contribute nitrogen to the bay and in turn develop projects to at minimum mitigate future nitrogen inputs. It is hoped that such projects will protect the seagrass beds in Lemon Bay from declines with future development, and likewise, similar research efforts are being proposed for Estero Bay to the south of Charlotte Harbor and the Caloosahatchee River as well.

Within the southern Charlotte Harbor region, including the Caloosahatchee River, Pine Island Sound, Matlacha Pass, and San Carlos Bay, the SFWMD is evaluating various methods to better protect estuary resources from harmful hydrologic conditions stemming from the artificial connection of the Caloosahatchee River to Lake Okeechobee. They have successfully promulgated minimum flows and levels rules that establish a minimum flow the SFWMD must meet to protect estuary resources from low flow conditions. The SFWMD is, in turn, currently designing projects, with support from State, Federal and local agencies, to increase storage within the Caloosahatchee River watershed to allow this rule to be met more frequently in the future through the Comprehensive Everglades Restoration Plan (*http://www.evergladesplan.org/index.cfm*). It is hoped also that these projects will allow the district to manage the timing and quantity of releases from the

Franklin Locks system to the estuary to better mimic a natural flow regime.

The SFWMD in 2003 also designated the southern Charlotte Harbor region and Estero Bay as SWIM plan waterbodies, and now development of pollutant load reduction goals is required for these areas. In close coordination with this effort, a local nonprofit entity, the Estero Bay Nutrient Management Partnership, which includes representatives from the SFWMD, local governments, the CHNEP, citizens and industry, is developing voluntary nutrient reduction goals for the Estero Bay subbasin.

Finally, the SFWMD is developing circulation models of the southern Charlotte Harbor region, including Pine Island Sound, Matlacha Pass, San Carlos Bay, and Caloosahatchee River to help better understand flow patterns in that area. The local government agencies are contemplating new designs for the Sanibel Causeway and are using this model to evaluate the causeway's implications on estuarine resources.

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Florida Bay

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Background

Florida Bay is a shallow, triangular lagoon with an average depth of less than 1.5 m (5 ft), located at the southern tip of the Florida peninsula (fig. 1). Florida Bay ranks among the world's largest estuarine systems with an overall area of 220,000 ha (543,620 acres), and seagrass communities, principally turtle grass (*Thalassia testudinum*), cover 95% of the bottom (Fourqurean and Robblee, 1999). Florida Bay seagrasses are critically important to the economy and ecology of the southeastern Gulf of Mexico, providing food and shelter to numerous fish and invertebrate species, including the economically important pink shrimp (*Farfantepenaeus duorarum*), stone crab (*Menippe mercenaria*), and spiny lobster (*Panuliris argus*) (Davis and Dodrill, 1989; Holmquist and others, 1989a; Thayer and Chester, 1989; Tilmant, 1989; Robblee and others, 1991). Various wading birds, as well as endangered species such as bald eagles (*Haliaeetus leucocephalus*), manatees (*Trichechus manatus*), American crocodiles (*Crocodylus acutus*), and sea turtles (*Chelonia mydas*) also depend, in part, on seagrass communities (Holmquist and others, 1989b; Mazzotti, 1989; Boesch and others, 1993).

Florida Bay is divided into numerous, discrete basins (locally called "lakes") by a series of interconnected, carbonate mudbanks. These mudbanks function as barriers to water circulation, leading to large spatial differences in water temperature, salinity, and turbidity within the region. Tidal energy is rapidly attenuated by the mudbanks so that there is essentially no lunar tide over most of central and northeastern Florida Bay. This extensive network of mudbanks covers more than 25% of the area and occurs atop an almost planar surface of limestone bedrock that slopes from the Florida mainland downward to the southwest (Perkins, 1977; Wanless and Tagett, 1989). Physical and chemical characteristics of both banks and basins vary along this gradient. For example, narrow banks separate relatively large basins in the northeast, while banks are wider and basins smaller in the southwestern portion of bay. Sediment accumulation, water depth, and phosphorus availability, which is the limiting plant nutrient in much of Florida Bay, also increase from northeast to southwest (Zieman and others, 1989; Fourqurean and Robblee, 1999).

Florida Bay is typically a negative estuary, with evaporation exceeding freshwater input during most of the year. Hypersaline conditions (i.e., salinities greater than 38 ppt) are thus common, especially in central Florida Bay where water circulation is most limited. Factors that influence salinity levels include proximity to the Gulf of Mexico or Atlantic Ocean, groundwater seepage, and local precipitation and runoff; however, the single most important factor affecting both seasonal and annual variation in salinity patterns throughout the system is the amount of rainfall. Salinities are usually highest during the late spring, which is the end of the dry season, and lowest during the peak rainy season in late summer. During extended drought periods, salinities greater than 70 ppt are not uncommon in central Florida Bay.

Florida Bay historically received large quantities of fresh water via The Everglades watershed. Unfortunately, substantial alteration of The Everglades during the past 100 yr to accommodate the rapidly growing south Florida population has resulted in greatly reduced freshwater inflow to the region when compared to historical levels. Most of the reduction in freshwater inflow is the result of an extensive series of drainage channels constructed to regulate flow through The Everglades for agriculture and flood control (i.e., the Central and Southern Florida Project). Fresh water that once might have flowed into Florida Bay has been diverted east into Biscayne Bay and the Atlantic Ocean via the canal system. In addition, construction of the Flagler Railway (1905–12) connecting Key West to the Florida mainland greatly changed circulation patterns within the region, essentially eliminating waterflow between Florida Bay, the Atlantic Ocean, and the Gulf of Mexico.

The majority of fresh water flowing directly into Florida Bay enters from the northeast through the Taylor River (fig. 2). Fresh water also enters the northeastern portion of the bay as overflow from the C-111 Canal, part of the South Florida Water Management District's canal system, and as sheet flow from The Everglades. Shark River Slough, the principal freshwater flowway of The Everglades, empties

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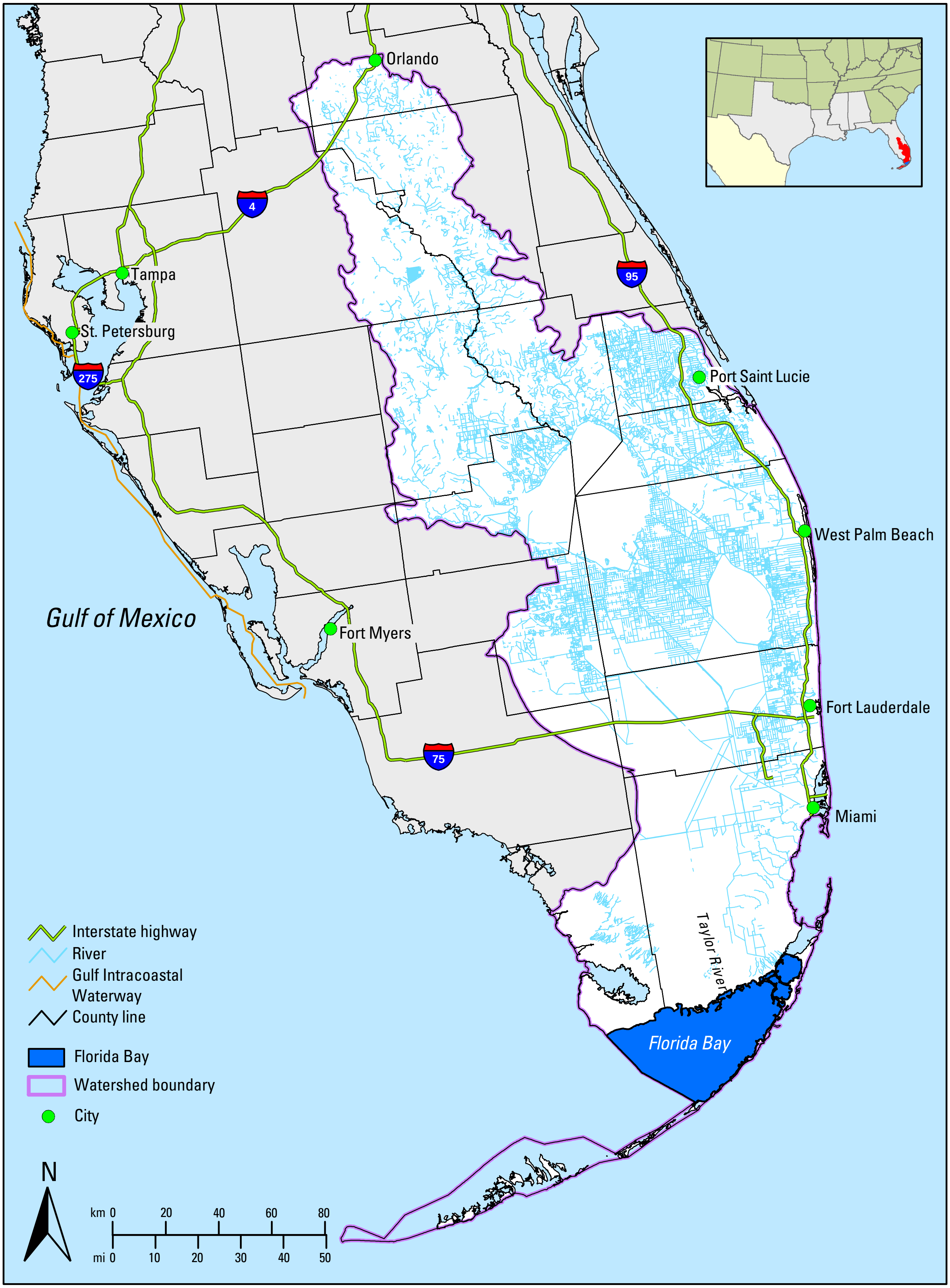

Figure 1. Watershed for Florida Bay.

into Whitewater Bay and the mangrove estuaries of the southwestern Florida coast. Fresh water from Shark River Slough enters Florida Bay from the west after mixing with near shore waters of the Gulf of Mexico. Inflows from both the Taylor River and Shark River Sloughs are significantly lower than historical volumes because of current water management practices.

Recent paleoecological investigations have provided evidence that salinities in Florida Bay have increased over the last century, and the temporal patterns of change coincide with human activities in the south (Swart and others, 1996; Brewster-Wingard and Ishman, 1999; Halley and Roulier, 1999). For example, increases in salinity in Florida Bay between 1910 and 1920 have been attributed to construction of the Flagler Railroad causeway. Salinity elevations observed after 1940 are most likely related to the canal system that diverts fresh water away from The Everglades before it can reach Florida Bay. Although the effects of increased salinity on the Florida Bay ecosystem are difficult to quantify given the lack of historical data, it is widely recognized that salinity greatly influences the distribution and abundance of estuarine organisms.

Scope of Area

Florida Bay is bordered to the north by the Florida mainland, to the southeast by the Florida Keys, and to the west by the Gulf of Mexico (fig. 1). The exact geographical boundaries of Florida Bay are difficult to establish given the open connection with the Gulf of Mexico. Florida Bay is generally considered to be the shallow, less than 3 m (10 ft), mudbank-dominated region located east of The Everglades National Park boundary (Fourqurean and Robblee, 1999).

Methodology Employed To Analyze Historical Trends

Aerial Photography and Interpretation

Three reports have provided acreage estimates for Florida Bay seagrasses that were based primarily on the interpretation of aerial photography. The first acreage estimates in 1972 were included in a natural resource inventory of estuaries along the Florida Gulf of Mexico coast found in McNulty and others (1972) (table 1). The authors collected photography from county offices, described as mostly 1:400-scale imagery, that was acquired within 5 yr before the study. The document does not specifically state the scale or source of the Florida Bay photography. Submerged vegetation was mapped by freehand on navigation charts while consulting the aerial photography, and field checks were performed to determine accuracy of the photointerpretation. Seagrasses were classified in a single category called “submerged vegetation.”

The second mapping report that included Florida Bay seagrasses was prepared by Continental Shelf Associates, Inc. (1989) under contract with the U.S. Department of the Interior’s Minerals Management Service (table 1; fig. 3). For this project, 1:40,000-scale, natural-color aerial photography was collected for a large area of southwestern Florida, including Florida Bay. The classification system consisted of 26 categories describing a variety of benthic habitat types (e.g., seagrasses, algae, live bottom, coral, sand), of which 14 contained a seagrass component. Extensive groundtruthing data were gathered to facilitate photointerpretation. In addition to seagrass acreage, the report included information on the characteristics of the seagrass community that included seagrass biomass and short-shoot density.

The third Florida Bay seagrass mapping effort was a joint venture between the Florida Marine Research Institute and the National Oceanic and Atmospheric Administration’s Coastal Services Center (table 1; fig. 4). At two different times between 1991 and 1995, 1:48,000-scale aerial photography was collected for the Florida Bay region. The imagery was interpreted for benthic habitats, including seagrasses. Of the 30 classification categories used in the interpretation process, 9 contained a seagrass component.

Differences in the quality of the photography, delineation techniques, classification systems and geographic extents of the study areas preclude using these datasets for determining trends in Florida Bay seagrass distribution and abundance. For this document, the Continental Shelf Associates, Inc. (1989) and the Florida Marine Research Institute and National Oceanic and Atmospheric Administration (1995) data were analyzed within the same extent boundary by using a geographic information system (GIS) software clip command. The data reported in McNulty and others (1972) were not available in digital form, so a similar GIS clip function was not possible.

Table 1. Historical coverage in hectares (acres) of submerged vegetation in Florida Bay.

Year of Photography	Seagrass Coverage	Source
1970	103,848 (256,608)	McNulty and others, 1972
1987	166,030 (410,270)	Continental Shelf Associates, Inc., 1989
1991–95	147,612 (364,602)	Florida Marine Research Institute/ National Oceanic and Atmospheric Administration, 1995

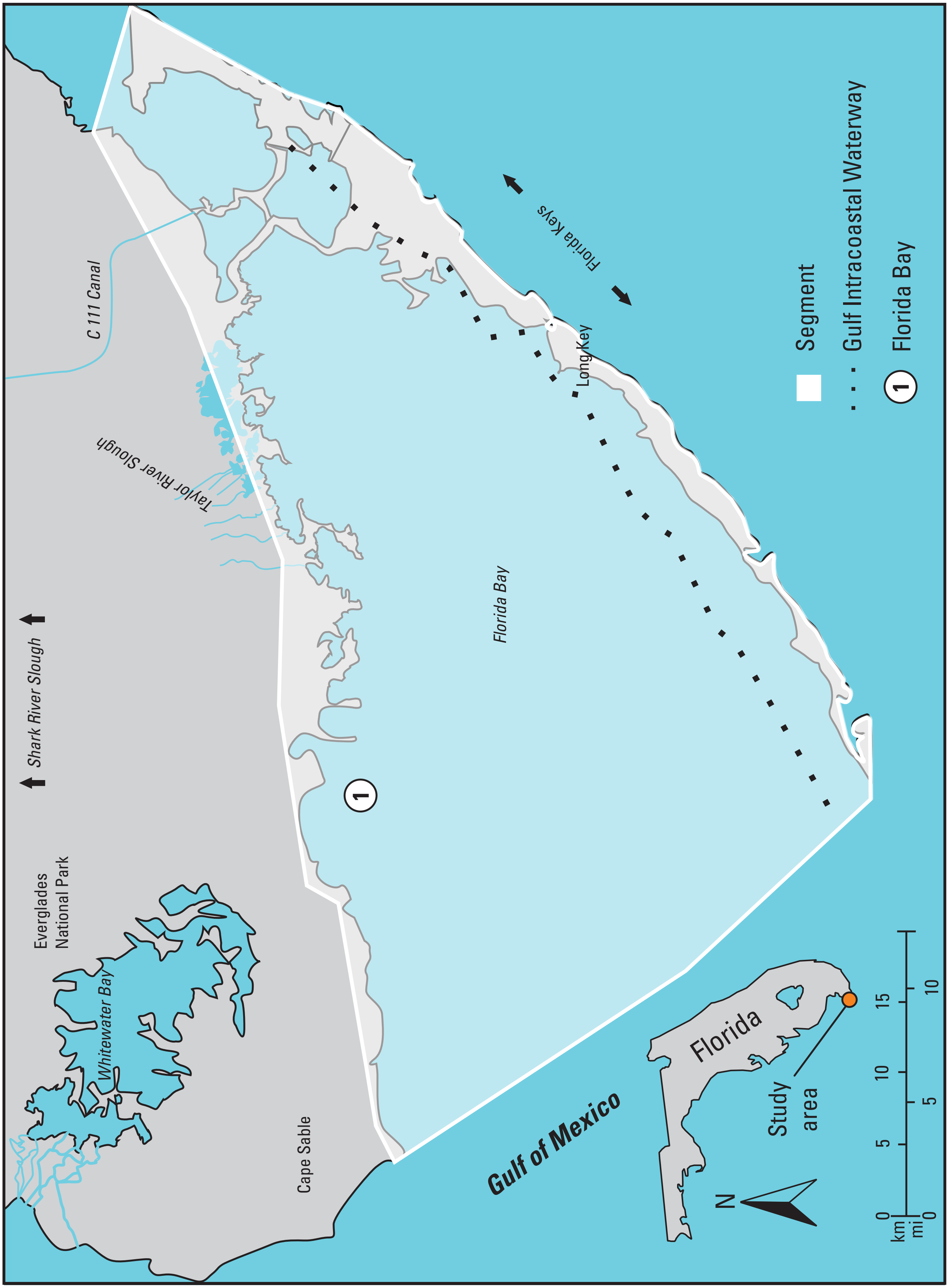

Figure 2. Scope of area for the Florida Bay seagrass vignette.

Field Investigations

The first quantitative seagrass survey in Florida Bay was conducted during the summer of 1984. Zieman and others (1989) were contracted by the National Park Service to characterize the status of seagrass habitats in the Everglades National Park. Seagrass species composition and abundance were determined at more than 100 stations distributed throughout the bay. Seagrass beds dominated by turtle grass covered more than 80% of the subtidal mudbanks and basins in Florida Bay.

Since the late 1980s, many components of the Florida Bay ecosystem have changed substantially, and changes in the seagrass community have been particularly conspicuous. Extensive areas of turtle grass began dying in summer 1987, particularly in central and western Florida Bay (Robblee and others, 1991). This die-off expanded rapidly, and by summer of 1988, 30% of the dense seagrass beds in western Florida Bay were affected. By 1990, more than 4,000 ha (9,884 acres) of seagrass had been completely lost, and an additional 23,000 ha (56,833 acres) were damaged. Although the rate has slowed, turtle grass die-off continues to occur in the bay.

The patchy mortality characteristic of die-off is very different from the gradual thinning and loss of seagrasses typically resulting from decreased water clarity. In turtle grass meadows affected by die-off, there is often a very sharp transition between die-off patches and visually healthy seagrasses. Factors that may contribute to turtle grass die-off are physiological stressors such as elevated water temperature and prolonged hypersalinity, excessive seagrass biomass leading to increased respiratory demands, hypoxia and sulfide toxicity, and disease; however, the causative mechanism behind die-off remains poorly understood (Robblee and others, 1991; Carlson and others, 1994; Durako, 1994; Durako and Kuss, 1994; Fourqurean and Robblee, 1999).

As the die-off subsided in 1991, a widespread decline in water clarity began that persisted in some parts of the bay until 1997 (Boyer and others, 1999; Stumpf and others, 1999). Before this event, most of the water in Florida Bay was exceptionally clear and often described as “gin” clear. Usually, the only turbid areas were in northeastern Florida Bay, where carbonate sediments are easily resuspended by wind and waves. This increased light attenuation was principally the result of microalgal blooms and resuspended sediments and was most severe in western and central Florida Bay (Phlips and others, 1995; Phlips and Badylak, 1996). Environmental changes that lead to reduced light availability have been implicated in seagrass declines worldwide (Peres and Picard, 1975; Cambridge and McComb, 1984; Orth and Moore, 1984; Giesen and others, 1990; Dennison and others, 1993; Onuf, 1994), raising concerns that Florida Bay seagrasses would suffer even further damage.

In response to the substantial ecosystem changes and probable negative impacts on seagrasses, Hall and others (1999) conducted a study to determine the current status of Florida Bay seagrass communities. During the summer of 1994, seagrass species composition, distribution, and abundance were measured at the same stations visited 10 yr earlier by Zieman and others (1989) to assess changes that had taken place since 1984, a time which preceded the seagrass die-off and microalgal blooms.

Despite seagrass die-off and persistent declines in light availability, turtle grass remained the dominant seagrass species in Florida Bay, and its distribution changed little between 1984 and 1994. On a bay-wide basis, however, turtle grass abundance declined significantly between surveys; mean short-shoot density dropped by 22% over the 10-yr interval. Turtle grass decline was not homogeneous throughout Florida Bay; largest reductions in shoot density occurred principally in the central and western regions. Both the distribution and abundance of two other seagrasses, shoal grass (*Halodule wrightii*) and manatee grass (*Syringodium filiforme*), declined substantially between 1984 and 1994. Bay-wide, shoal grass and manatee grass shoot densities declined by 92% and 93% between surveys, respectively. The spatial patterns of seagrass loss in Florida Bay between 1984 and 1994 suggested that chronic light reduction and die-off were the most likely causes for decline.

Current Seagrass Change—Florida Bay Fisheries Habitat Assessment Program

The Florida Bay Fisheries Habitat Assessment Program (FHAP) was initiated during spring 1995 in response to continuing concerns over environmental changes and seagrass loss within the region. The goal of FHAP is to provide spatially explicit information on the distribution, abundance, species composition, and population dynamics of Florida Bay seagrasses. The Miami-Dade Department of Environmental Resource Management (DERM) employs FHAP protocols in Biscayne Bay and northeastern Florida Bay, and the Florida Keys National Marine Sanctuary (FKNMS) also uses FHAP protocols in its seagrass monitoring program. The establishment of compatible sampling protocols among these three large-scale programs has provided a mechanism for the establishment of a regional management-oriented database of unprecedented geographic scope (Fourqurean and others, 2002).

Methods

Sampling for FHAP is conducted twice a year, during spring (May–June) and fall (October–November). Each of 10 basins representing a range of conditions and gradients in Florida Bay are partitioned into approximately 30 to 35 tessellated, hexagonal grid cells. Sampling-station locations are randomly chosen from within each cell, for a total of about 330 stations per sample period. Sampling grids and station locations were generated by using algorithms developed by the

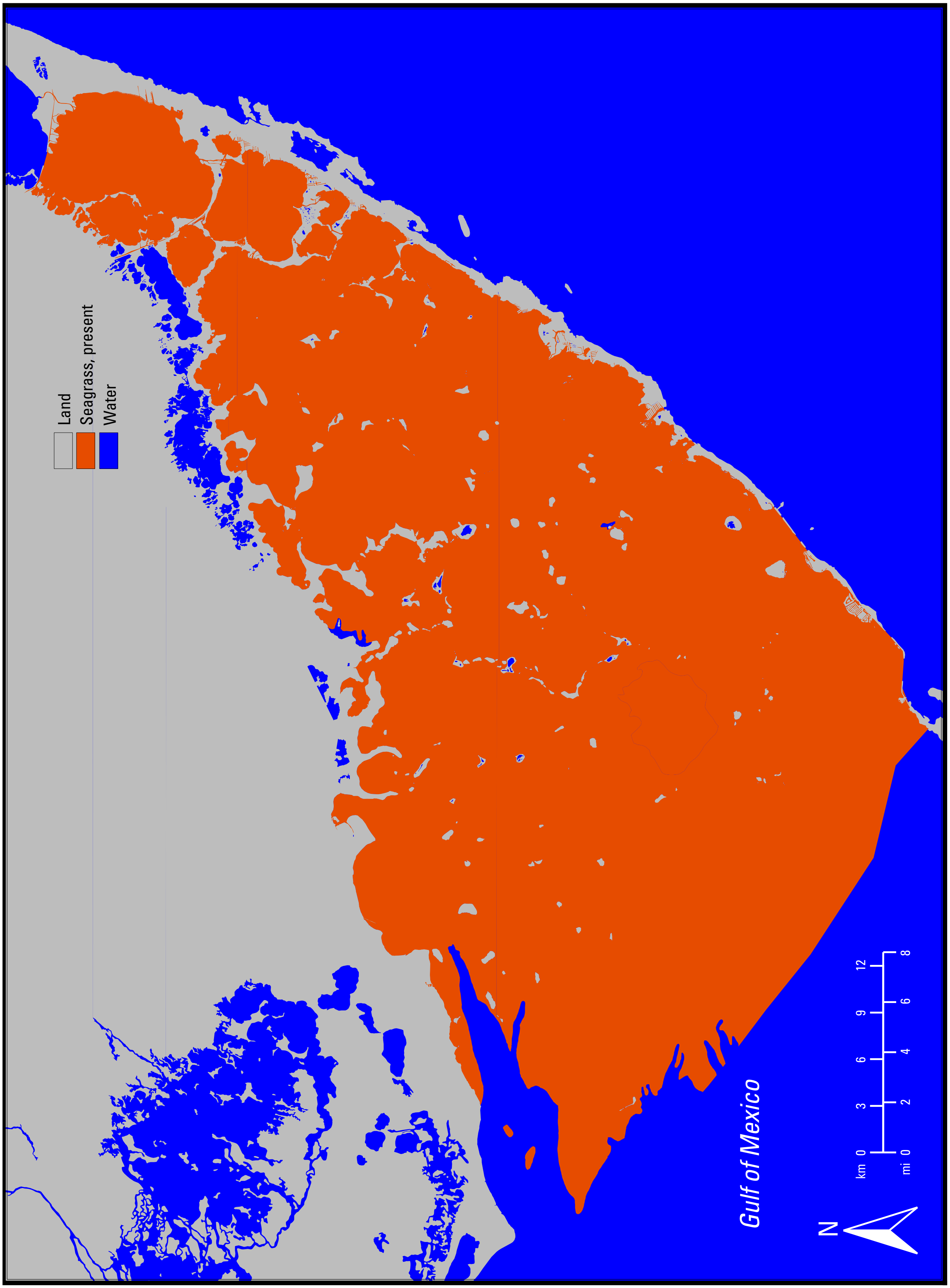

Figure 3. Seagrass distribution in Florida Bay, 1987.

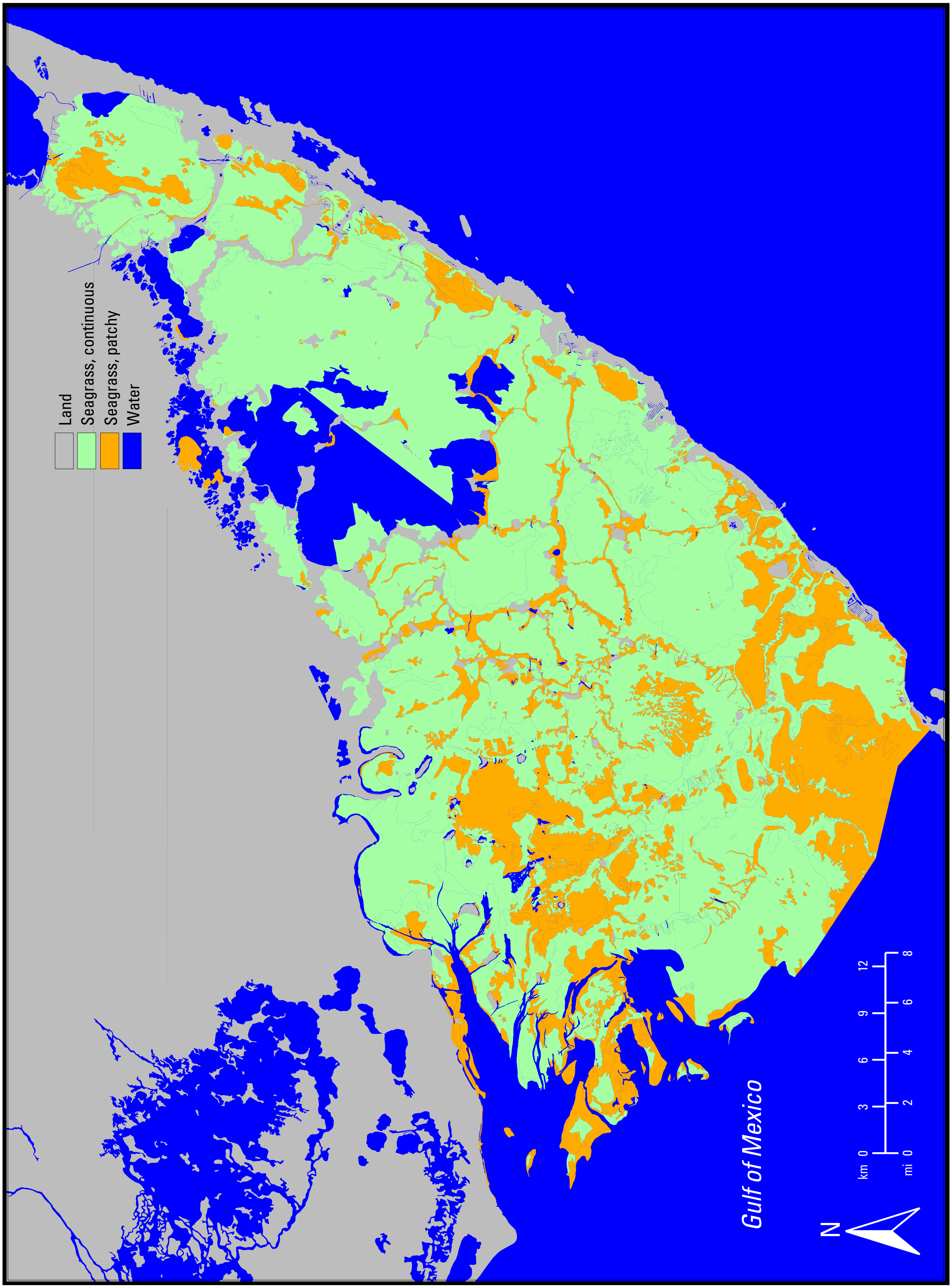

Figure 4. Seagrass distribution in Florida Bay, 1994.

U.S. Environmental Protection Agency's (EPA) Environmental Monitoring and Assessment Program (EMAP) and were provided by Dr. Kevin Summers of the EPA in Gulf Breeze, Fla. This type of sampling design results in systematic random sampling, scales the sampling effort to the size of the basin, and is well-suited for interpolation (i.e., kriging) and mapping of the data. Stations are located by using a handheld Global Positioning System (GPS) unit.

At each station, seagrass cover is visually quantified within each of four randomly located 0.25 m^2 (3 ft^2) quadrats by using a modified Braun-Blanquet frequency abundance scale (Mueller-Dombois and Ellenberg, 1974). This nondestructive, semiquantitative method requires relatively little time per sample (5 to 10 min for four quadrats), can be used for most plant communities, and approximates quantitative characteristics of shoot density and standing crop for seagrasses. Because of the efficiency of the Braun-Blanquet approach, information about seagrass abundance can be collected on a bay-wide scale; this type of geographical scope would be cost prohibitive if time-consuming direct shoot counts or biomass measurements from destructive sampling were used.

For a particular sample quadrat, the observer first lists all seagrass species that are observed. A cover-abundance rating is then assigned by using the following scale: 5 is assigned to any number with cover of more than 75% of the quadrat; 4 to any number with 50% to 75% cover; 3 to any number with 25% to 50% cover; 2 to any number with 5% to 25% cover; 1 to numerous, but less then 5% cover, or scattered with up to 5% cover; 0.5 to few, with small cover; and 0.1 to solitary, with small cover. The upper four scale values of 5, 4, 3, and 2 refer only to cover. The lower three scale values, 1, 0.5, and 0.1, are primarily estimates of abundance (i.e., number of individuals per species). Sampling of replicate (four) quadrats at each sample point allows assessment of within-station versus among-station variability.

Maps of seagrass distribution and abundance and changes in abundance are produced by using a contouring and 3-D mapping program (ArcView Spatial Analyst). The geostatistical gridding method of kriging is used to express the trends in the Braun-Blanquet data.

Status and Trends

Contour plots illustrating turtle grass distribution and abundance in spring 1995 and spring 2003 (fig. 5*A* and *B*) indicate that it continues to be the dominant and most widely distributed seagrass species in Florida Bay. In contrast, turtle grass abundance declined significantly in basins along the western margin of Florida Bay from spring 1995 to spring 2003 (fig. 5*C*). Although quantitative data on light availability in Florida Bay are limited, it appears that the greatest losses of turtle grass occurred from 1995 to 1998 in chronically turbid areas. Whether disease (i.e., the parasitic slime mold *Labyrinthula* sp.) also influenced decline of turtle grass is difficult to determine; however, spatial coincidence among the distributional patterns of *Labyrinthula* abundance in the fall and turtle grass loss in the following spring suggests that the slime mold affected turtle grass decline. Turtle grass abundance increased as water clarity improved during the period from 1997 to 2000 and has remained generally stable since that time.

The distribution and abundance of shoal grass increased substantially from spring 1995 through spring 2003 (fig. 6*A*, *B*, and *C*), most likely the result of improvements in water clarity. The very large gains in shoal grass relative to those of turtle grass in western Florida Bay may reflect the lower light requirement of shoal grass relative to that of turtle grass, as the gains may also reflect the ability of shoal grass to rapidly colonize areas where the turtle grass canopy has been removed.

Restoration Opportunities

Restoration of Florida Bay and The Everglades watershed is a long-term process that has only recently begun. Complex environmental, engineering, and management issues must be resolved to achieve restoration goals, requiring the cooperation of multiple Federal, State, local, and tribal organizations. To facilitate the restoration process, Congress established the South Florida Ecosystem Restoration Task Force through the Water Resources Development Act of 1996. The purpose of the task force is to coordinate policies, programs, and information exchange among all organizations responsible for restoration, preservation, and protection of the south Florida ecosystem.

Many significant restoration projects affecting Florida Bay will be part of the Comprehensive Everglades Restoration Plan (CERP). The Water Resources Development Acts of 1992 and 1996 gave the U.S. Army Corps of Engineers authority to review the Central and Southern Florida (C&SF) Project, and to develop a comprehensive plan to restore historical water flow in the south Florida region while meeting other water resource needs (e.g., flood control, agriculture, and urban water supplies). The CERP includes over 60 water improvement projects and will take more than 20 yr to complete at an estimated cost of 7.8 billion dollars.

Projects addressing Florida Bay restoration under CERP are being developed through the Florida Bay and Florida Keys Feasibility Study, a joint effort led by the U.S. Army Corps of Engineers and the South Florida Water Management District. The purpose of the feasibility study is to identify the modifications needed to restore the water quality and ecological conditions of Florida Bay while maintaining or improving conditions in the Florida Keys.

A crucial restoration project affecting Florida Bay but not included in CERP is the Modified Deliveries to the Everglades National Park (ENP) and C-111 Project. This project will modify water flow to the ENP to restore more natural

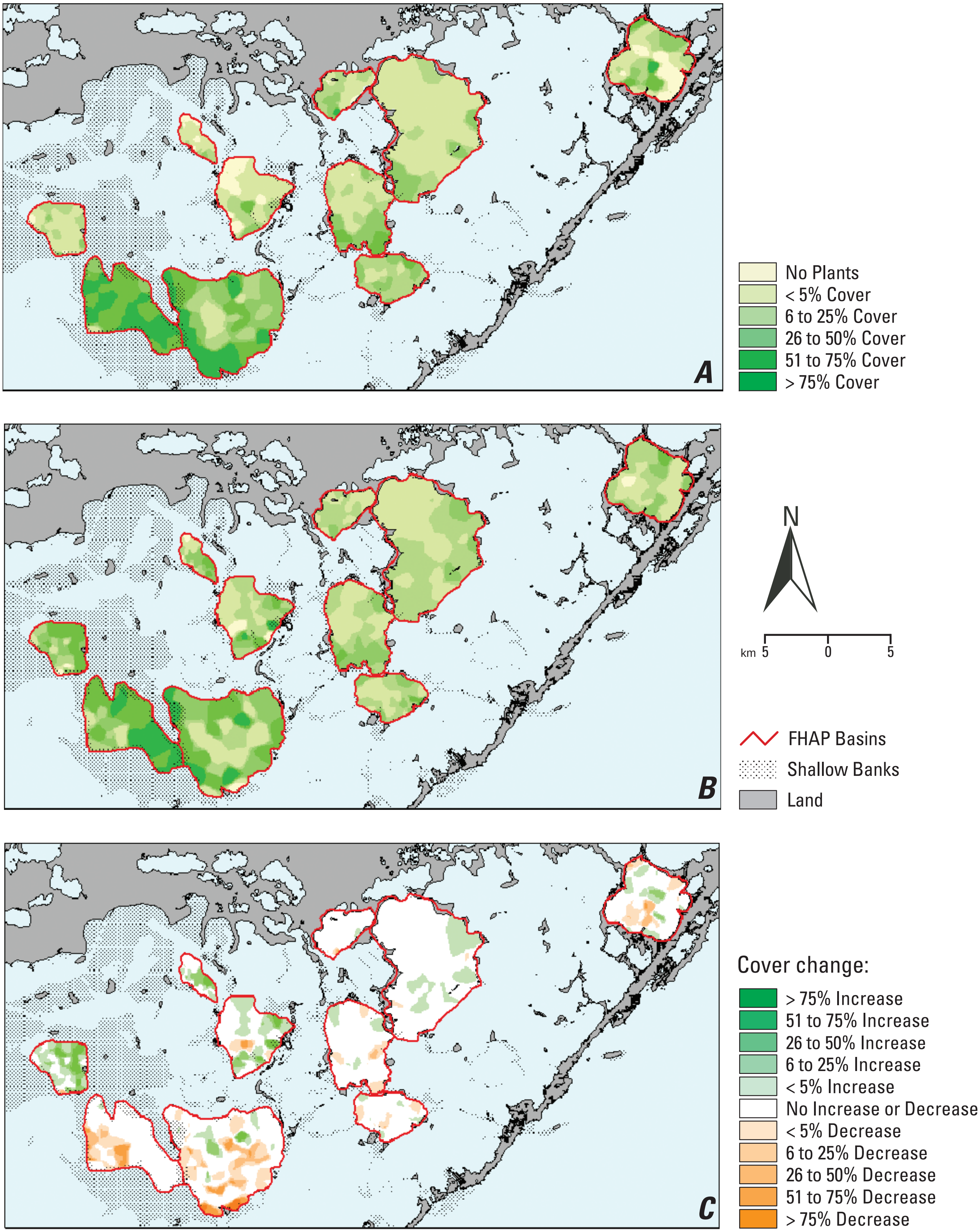

Figure 5. Distribution, abundance, and cover change of turtle grass (*Thalassia testudinum*) in Florida Bay in springs 1995 and 2003 (FHAP = the Florida Fish and Wildlife Conservation Commission Fisheries Habitat Assessment Program).

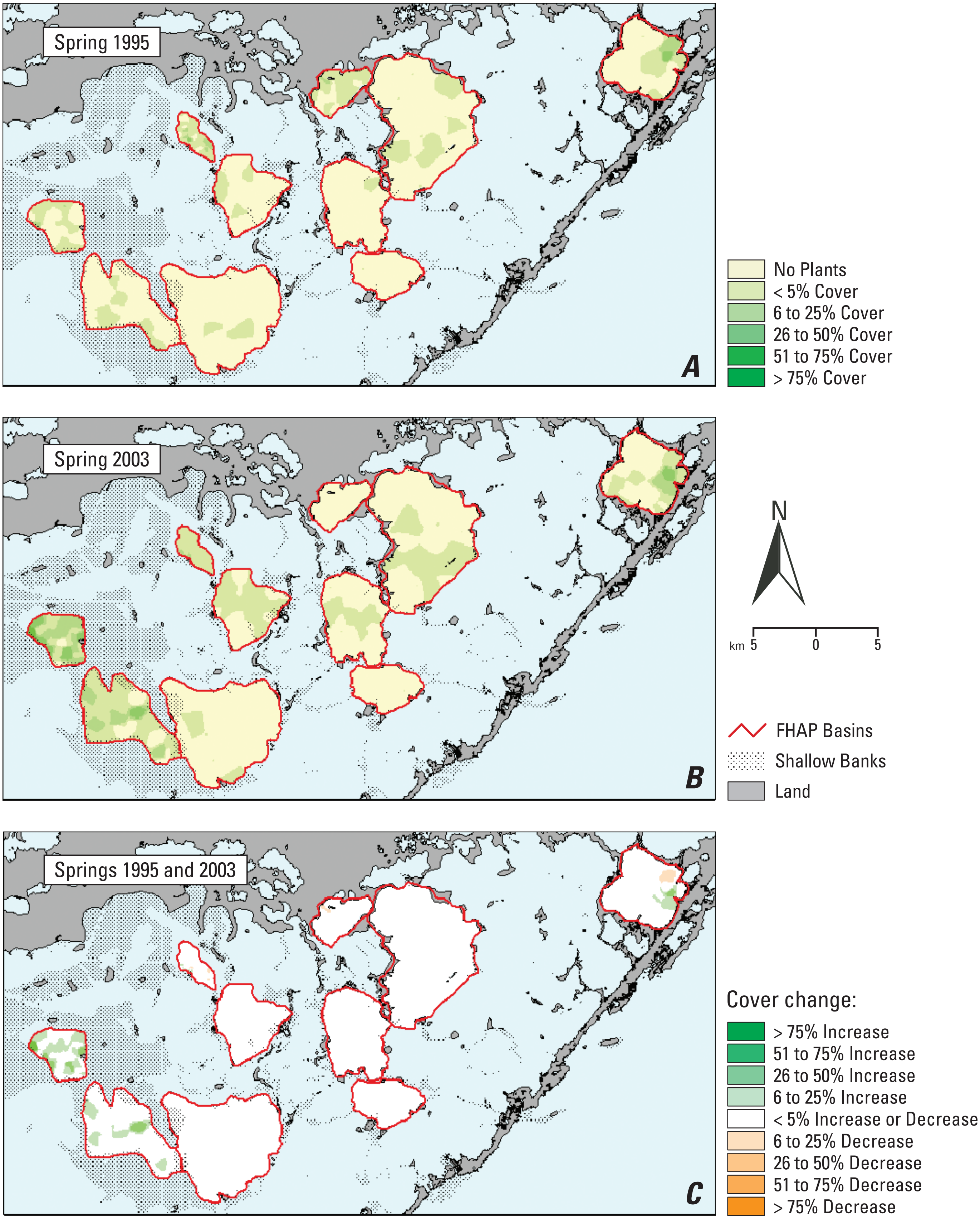

Figure 6. Distribution, abundance, and cover change of shoal grass (*Halodule wrightii*) in Florida Bay in springs 1995 and 2003 (FHAP = the Florida Fish and Wildlife Conservation Commission Fisheries Habitat Assessment Program).

hydrologic conditions to the southern Everglades and Florida Bay.

More fresh water alone will not return Florida Bay to its pre-1980s, clear water condition. Increasing freshwater delivery to the bay could actually result in decreased water clarity and further seagrass loss if nutrient loads and algal blooms also increase. It will be critical to the restoration process to determine proper timing, location, and quality of fresh water released to Florida Bay in addition to increasing input volume.

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Conclusions

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Importance of Gulf of Mexico Seagrasses

The vignettes in this report quantify and highlight the regional significance of seagrass ecosystems in the northern Gulf of Mexico. Seagrasses make many important ecological and economic contributions to communities around the Gulf of Mexico (fig. 1). These seagrass ecosystems are also of national and even global significance. Seagrass meadows of the northern Gulf of Mexico contain more than 50% of the total U.S. distribution of seagrasses and more than 5% of the known global occurrences of seagrasses (Green and Short, 2003). The areal extent of seagrass meadows in the northern Gulf of Mexico is greater than the known distribution of seagrasses in any other countries except Australia and Indonesia (Green and Short, 2003). Seagrasses of the Florida Keys and Florida Bay in Florida and the adjacent Continental Shelf make up the largest documented semicontinuous seagrass meadow on Earth (Fourqurean and others, 2002). The Gulf of Mexico harbors a significant component of the world's marine biodiversity for seagrass ecosystems.

Seagrass Losses

Over the past 50 yr, seagrass losses of 20% to 100% have been estimated in northern Gulf of Mexico estuaries (Duke and Kruczynski, 1992). Causes of these losses are many and include climate and water-level variations, physical removal, smothering with sediments, light extinction resulting from turbidity or phytoplankton, and inputs of excess nutrients. Natural perturbations, such as storm events, can cause erosion of sediments and high water turbidity that may stress seagrasses.

Human-induced impacts on seagrasses can have more long-lasting, and perhaps synergistic, effects. Seagrasses are found at the downstream position in watersheds and are susceptible to perturbations from many upstream sources. Land-based erosion is a major stress to seagrass systems because it leads to physical smothering and to increased turbidity of the water column and, thus, to decreased light for photosynthesis. Nutrient additions from point and nonpoint sources can result in blooms of phytoplankton that can shade seagrasses and cause stress or death. Stresses can eliminate seagrasses completely or produce patchy, discontinuous beds. Fragmented seagrass beds do not provide the same ecological goods and services as continuous seagrass meadows (Bell and others, 2001; Uhrin and Holmquist, 2003). Once seagrasses are lost from an area, physical binding of sediments by the

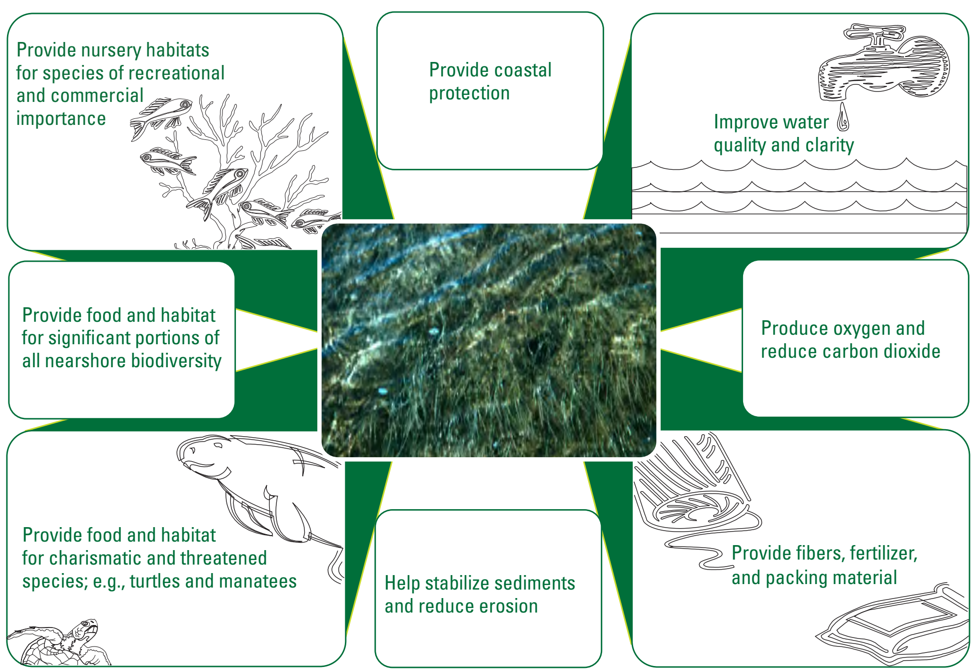


Figure 1. Examples of seagrass functions and values. Photograph by Tommy Michot, U.S. Geological Survey

[1] The Nature Conservancy.

[2] U.S. Environmental Protection Agency.

[3] National Oceanic and Atmospheric Administration.

root systems disappears and may result in increased turbidity and more seagrass losses. Physical damage to seagrass systems resulting from cumulative impacts from boat groundings and propeller scarring is also a significant contributor to losses of seagrass communities (Sargent and others, 1995; Uhrin and Holmquist, 2003).

Assessment of seagrass areal coverage and cause of losses for the 14 estuarine systems presented in this status and trends report by the various authors along the Gulf of Mexico coast are summarized in table 1 and show several recurrent themes:

- Historical losses for the most part far exceed any recent gains in acreage.
- The area of continuous coverage of seagrasses has markedly declined, and the area of fragmented beds has increased.
- Development pressure and nutrient additions are the main contributors to historical and continuing losses.
- Variability in climate can cause losses and can interact with human stresses.

Although some estuarine systems in this report document recent increases in seagrass coverage, the losses far outweigh the gains. For example, Laguna Madre, Tex., has lost approximately 10% to 20% of seagrass coverage since 1965, much of which has been due to a recent phytoplankton bloom. 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. Tampa Bay, Fla., has experienced some recent increases in seagrass coverage because of improved water quality, but coverage today is still reduced by 6,000 ha (14,826 acres) from the 1950s.

The principal cause of seagrass loss varies depending upon the location, but there are some commonalities. Physical removal of seagrass and burial with sediments are consequences of initial and maintenance dredging of the Gulf Intracoastal Waterway (GIWW). In Texas, dredging in the lower Laguna Madre resulted in increased turbidity and a significant loss of seagrasses because of physical removal, smothering, and reduced light penetration. Dredging of Redfish Bay not only resulted in a loss of about 536 ha (1,324 acres) of seagrasses but also resulted in a substantial loss (48%) of continuous beds and an increase (88%) in patchy, fragmented beds. Turbidity associated with dredging Redfish Bay, Harbor Island, and the back side of Mustang Island for oil and gas exploration resulted in blanketing seagrass habitats with sediments and subsequent disappearance of seagrasses. In Florida, dredging operations in the 1950s and 1960s were a major cause of seagrass losses.

It is well documented that human-induced sediment and nutrient inputs into estuaries lead to major declines in seagrass coverage and that cessation of those inputs leads to recovery. In Florida, approximately one-half of Tampa Bay's seagrass meadows (16,350 to 21,653 ha (40,401 to 53,505 acres)) were lost between 1950 and 1982 as a result of the combined effects of dredge and fill activities and degraded water quality. Recent improvements in treatment and disposal of wastewater by Tampa, St. Petersburg, and Clearwater have resulted in improved water quality and an increase in seagrass acreage (26,078 ha (64,439 acres)). Seagrass coverage in Tampa Bay during 2002, however, was only 65% of the 1950 values. Wastewater nitrogen loads to Sarasota Bay have decreased from 516 Mg/yr (569 tons/yr) in 1988 to 100 Mg/yr (110 tons/yr) in 1999 (an 80% decrease). This improved water quality has led to increased seagrass coverage in portions of Sarasota Bay (e.g., Little Sarasota Bay).

Natural nutrient releases may also initiate seagrass loss. The bloom of "brown tide" in the upper Laguna Madre in Texas is thought to have begun with inhibition of grazers by hypersalinity followed by a freeze-induced fish kill and a subsequent massive nutrient release. The phytoplankton bloom that occurred because of increased nutrient availability resulted in continuously reduced light penetration from 1990 until 1997, and the brown tide alga has bloomed sporadically since 1997. There has been little recovery of seagrasses because bottom sediments are now more prone to resuspension that results in decreased light availability and further nutrient releases.

Declining water quality because of significant changes in historical land uses is also thought to be a major contributor to seagrass losses in Mississippi Sound, Miss. Agriculture and forestry have given way to urban, casino, industrial, and port developments. Coastal Mississippi has experienced a 21.8% increase in human population over the past decade, which is twice that observed for the entire State. This growth has resulted in increased erosion and point and nonpoint sources of pollution that have detrimentally affected seagrass survival.

Although it has not been proven, the die-off of seagrasses in Florida Bay, Fla., was most probably due to severely reduced freshwater inflow following canal construction during the 1950s to 1970s to drain wetlands and prevent flooding of urban areas in south Florida. The canals routed fresh water to the sea before it reached The Everglades wetlands that form the northern border of Florida Bay. Decreased water flow by design, and a significant drought in the late 1980s, resulted in hypersaline conditions and changed sediment chemistry that stressed seagrasses. Once stressed, seagrass were then susceptible to disease, which may have been the ultimate cause of over 27,000 ha (66,717 acres) of seagrass damaged or lost (Robblee and others, 1991). Resuspension of sediments and phytoplankton blooms have restricted recovery of seagrasses in many areas of Florida Bay.

Propeller scar impacts to seagrass beds are an increasing problem for shallow beds throughout the Gulf of Mexico. The number of boaters continues to increase, which results in an increased threat to seagrass resources. Florida, for example, has documented that approximately 70,000 ha (172,970 acres) of seagrass have some degree of propeller scarring. Moderate to heavy scarring was found to be most prevalent in the Florida Keys, Charlotte Harbor, and Indian River Lagoon. Scarring

Table 1. Summary of areal coverage estimates from Gulf of Mexico nearshore areas and major threats to the resource.

Estuary System	Seagrass Coverage (most recent data)	Major Threats
1. Laguna Madre	69,517 ha (171,777 acres) 1998	1. Dredging 2. Texas brown tide
2. Texas Coastal Bend	11,385 ha (28,132 acres) 1994	1. Nutrient loading 2. Dredging
3. Galveston Bay	210 ha (519 acres) 1998	1. Habitat alteration 2. Nutrient loading
4. Chandeleur Islands	4,511 ha (11,147 acres) 1995	1. Tropical storms 2. Natural loss of sediment
5. Mississippi Sound and the Gulf Islands	298 ha (736 acres) 2002	1. Recreational uses 2. Decline in water quality
6. Perdido Bay	120 ha (297 acres) 2002	1. Wastewater effluent 2. Nutrient loading
7. Pensacola Bay	1,814 ha (4,482 acres) 1992	1. Sewage/industrial waste 2. Dredging
8. Choctawhatchee Bay	1,722 ha (4,255 acres) 1992	1. Boat propeller scarring 2. Nutrient loading
9. St Andrew Bay	3,979 ha (9,832 acres) 1992	1. Stormwater runoff 2. Boat propeller scarring
10. Florida Big Bend	250,000 ha (617,750 acres) 1992	1. Hydrological alterations 2. Nutrient loading
11. Tampa Bay	10,554 ha (26,079 acres) 2002	1. Dredging 2. Nonpoint pollution
12. Sarasota Bay	3,715 ha (9,180 acres) 2002	1. Rapid urbanization 2. Wastewater treatment
13. Charlotte Harbor	21,802 ha (53,873 acres) 1999	1. Increases in turbidity 2. Freshwater inflow
14. Florida Bay	124,787 ha (308,349 acres) 1994	1. Increased turbidity 2. Chronic light reduction

has also been documented in Redfish and Aransas Bays, Tex. Propeller scars may take many years to recover under ideal conditions, and many fail to recover because they are scoured by currents and wave action. Boater education, unambiguous signs, speed restrictions, and motor exclusion zones are being implemented in some areas in an attempt to limit future losses to scarring.

Significant changes in seagrass coverage can result from natural causes, but it is sometimes difficult to separate the impacts caused by nature from those caused by humans since they are often intertwined. For example, Hurricane Beulah (1967) caused the loss of 100% of seagrasses in Nueces Bay, Tex. This loss may have been exacerbated by poor erosion controls in adjacent farmlands which experienced record-level rainfall from the storm. Galveston Bay, Tex., was subjected to major destruction from Hurricane Carla (1961), and in the subsequent two decades lost 100% of its seagrasses. During that time, however, seagrasses experienced increasing stresses because of subsidence, erosion, and shoreline development in and around the bay. A 33% reduction in seagrass bed acreage in the Mississippi Sound, Miss., was attributed to a combination of erosion and sedimentation during Hurricane Camille (1969) and subsequent reductions in salinity because of flooding. Recent decreases of seagrass coverage in Sarasota Bay, Fla., have been linked to the 1997–98 El Niño event that resulted in rainfalls 20%–48% higher than average. Excessive nutrients and sediment loads most likely caused a decrease in water clarity and subsequent seagrass losses.

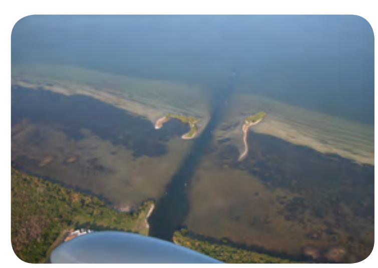
Other climatic variations can initiate changes in seagrass coverage. The Chandeleur Islands of Louisiana are a dynamic barrier island chain where the land area decreased 65% from 1855 to 1999 (including a 40% decrease with the passage of Hurricane Georges in 1998). Seagrass area surrounding those islands seems to decrease during and immediately after tropical cyclone passage and to recover between storm events. Winter storms and low water associated with cold front passages contribute to natural losses of seagrasses in Louisiana and other areas.

Sea-level rise as a result of the current global warming trend has resulted in gains of seagrass coverage at Harbor Island and Mustang Island, Tex. Elsewhere, seagrass increases with sea-level rise can be expected to occur in areas where depth, water quality, and sediment stability are suitable for seagrass colonization and growth.

Seagrass Decline Leads to Economic and Aesthetic Losses

Seagrass meadows are one of the building blocks that modulate productivity of coastal and estuarine ecosystems in the northern Gulf of Mexico. They are highly productive plant communities that provide habitat and forage for fish and wildlife, stabilize coastal sediments, and decrease wave energy. The biodiversity and productivity of seagrass meadows are directly linked to coastal economies (Duarte, 2000, 2002). Healthy seagrass meadows also filter nutrients and contaminants from the water, thus improving water quality. The resources that State and Federal agencies have expended to provide this status and trends report attest to the recognition of the goods and services that seagrasses provide.

Species that are dependent upon seagrass beds include large, charismatic herbivores, such as redhead ducks (*Aythya americana*), manatees (*Trichechus manatus*), and green sea turtles (*Chelonia mydas*), as well as resident and transient fishes and invertebrates, such as pinfish (*Lagodon rhomboides*), grass shrimp (*Palaemonetes* spp.), and seahorses (*Hippocampus erectus*), and their foods. Many species use seagrass beds as feeding and nursery areas, such as red drum (*Scianops ocellatus*) and bonefish (*Albula vulpes*). It has been estimated that 70% to 90% of economically important fishery species spend part of their life histories in seagrass beds, including penaeid shrimps, spiny lobster (*Panularus argus*), blue crab (*Callinectes sapidus*), and a wide variety of fishes. Although estimates of their monetary value vary widely, all Gulf of Mexico States agree that maintenance of healthy seagrass communities is essential for maintaining coastal economies. For example, seagrasses were determined to be worth $9,000 to $28,000 per acre for commercial, recreational, and storm protection functions in Texas. The loss of seagrasses can lead to changes in the types, abundance, productivity, and diversity of aquatic species and communities in an estuary.



These changes will eventually translate into altered economic and aesthetic values.

Resource Management and Monitoring

The States that border the northern Gulf of Mexico have significant natural beauty, and their coastal economies are dependent upon the maintenance and restoration of their natural resources. The States have recognized the importance of seagrasses to the coastal economies and must be congratulated for their conservation efforts and the effort and expense to conduct the status and trends monitoring reported in this document. It is hoped that these efforts can be made more consistent between States so that a gulfwide area figure can be regularly and accurately estimated.

The states have been pursuing a variety of regulatory programs in order to reduce seagrass losses and to promote recovery of stressed seagrass beds. Mississippi has imposed a ban on trawling within 1.8 km (1 mi) of the shoreline of Gulf Islands National Seashore in an effort to reduce turbidity adjacent to seagrass beds. Texas and Florida have enacted “no combustion engine zones,” improved signage, and education programs to reduce propeller scarring of seagrass beds. States are also considering restricting the size (draft) of vessels using certain waterways because of continuing losses to propeller wash and chronic sediment disturbance.

There is a need to reduce nutrient loading to surface waters from wastewater and stormwater. Coastal areas are generally not suitable for septic tank treatment of wastewater because of high water tables and paucity of nutrient binding and rapid percolation in sandy soils, and underlying karst in Florida. A major goal of the U.S. Environmental Protection Agency (EPA) Gulf of Mexico Program and of the seven gulf National Estuarine Programs is to provide an awareness the water quality issues through education, and to work with the gulf coast States and counties to (1) improve inadequate sewage treatment systems and septic systems, (2) replace wastewater treatment systems with state-of-the-art systems that provide improved treatment and disposal of wastewater, and (3) assist in the adoption and enforcement of stormwater treatment ordinances that will reduce runoff of untreated stormwater to surface waters. The example of Tampa Bay in reducing nutrient loads since the early 1990s illustrates that Federal agencies, State agencies, and local governments can work together to provide ecological payoffs that include the increase in seagrass habitat in the bay.

Planting of appropriate nonvegetated areas with seagrass shoots or seeds may increase the rate of recovery of an area towards a vegetated bottom. Methods to maximize success of creation or restoration of seagrass beds have been reviewed and analyzed by Fonseca (1994) and Fonseca and others (1998). There are, however, many examples of failed seagrass restoration efforts. To avoid failure, first and foremost, the area must have suitable environmental conditions for seagrass survival and growth. Restoration of areas devoid of seagrasses can only be successful if the physical and chemical conditions are amenable, or can be made amenable to seagrass survival. More research is needed on the definition of conditions that are conducive to seagrass growth and survival. Timing of plantings, genetic makeup of plant material, storage of plant material, anchoring methods, fertilization rates, currents, local stresses such as boat traffic, and other variables must be considered in developing seagrass restoration plans.

Proactive management of seagrass ecosystems requires the identification of indicators of seagrass health. Regular monitoring of those indicators will be required to detect areas that are stressed so that the proper responses can be applied to quickly and efficiently remove the stresses. More research is required on this important topic (Coles and Fortes, 2001).

Regular mapping of seagrass beds is required to define and quantify changes in coverage and may assist in monitoring progress toward restoration goals. Data from monitoring and mapping activities are particularly important in justifying the large expenditures required to produce changes in water quality conducive to seagrass survival and growth; however, accuracy of periodic gulfwide estimates of seagrass coverage will depend on standardization of the procedures used to make the estimates in different geographic areas. At present, there are different methods and definitions used between and within States. The EPA’s Gulf of Mexico Program provides an excellent forum to work toward standardization of techniques and definitions as well as for coordination of gulfwide mapping efforts.

Field of Dreams or Empty Meadows: If You Build It, Will They Come?

Most restoration goals are based on area of habitat restored (e.g., the Estuary Restoration Act of 2000, S. 835), but habitats are defined by the associated plants and animals, that is, the communities in the seagrass meadows, not just the areas of seagrasses. The ultimate goal of restoration should be the rejuvenation of the former seagrass habitats, their biodiversity, and the ecological services that they provide (fig. 1).

Most of our conservation and restoration focus for seagrasses in the Gulf of Mexico has been on the seagrasses themselves; we need to expand our vision for success. The underlying and often untested assumption of restoration efforts (e.g., by replanting or by improving water clarity) is that “If you build it, they will come.” That is, once the gardens of seagrasses are reestablished, the typical plants and animals of seagrass communities will come swimming and drifting back. We know from extensive research that this is a bad assumption for many salt marsh restoration projects (Minello and Webb, 1997; Zedler, 2000; Zedler and others, 2001). Some research in seagrass restoration indicated that plants and animals can return quickly (Fonseca and others, 1996), and overall our success in restoring seagrasses and their communities is likely to be higher than for salt marshes (French McCay and Rowe, 2003); nonetheless, we need to better monitor not only the acreage of seagrasses in the Gulf of Mexico, but also the communities that inhabit them. A common fallacy is that this community-level monitoring must be time- and cost-intensive. We can use presence and absence surveys of plants and animals and analyze those data with nonparametric, multivariate statistics that can robustly identify changes in seagrass communities (Warwick and Clarke, 1991; Clarke 1993; Anderson and Underwood, 1994). Similar community-based monitoring protocols are widely used in Europe and Australia, and we encourage their use in the Gulf of Mexico (Short and Coles, 2001).

Conservation Is a Necessary and Proactive Management Tool

Much of the focus of seagrass management has been on restoration of areal coverage, as discussed in the vignettes of this report. Some great strides have been made in restoration, for example in Tampa Bay, Fla., but we are still learning how to correct previous mistakes in management. It must be recognized that this form of retroactive management (i.e., restoration of lost resources) is expensive. The current restoration efforts in Florida in The Everglades and Florida Bay, and to a lesser extent in Tampa Bay, illustrate how expensive it can be for tax payers to redress problems with historical management practices.

There are many good opportunities for proactive management of seagrasses in the northern Gulf of Mexico. To be able to move forward on any cost effective conservation opportunities, however, we need to first remove ourselves from the endless cycle of crisis management. When funds are allocated and priorities are set for restoration activities, they should go hand in hand with conservation allocations and priorities.

There are places in the northern Gulf of Mexico where some relatively cost-effective restoration and conservation efforts can yield substantial benefits in seagrass meadows of regional and global significance. For example, the Big Bend of Florida contains the largest and least impacted seagrass meadows in the United States, and possibly the world. Mattson and others (this report) concluded that a conservation focus is needed to protect those meadows. The importance of conservation of Big Bend seagrass meadows is also highlighted in The Nature Conservancy’s regional plan for the northern Gulf of Mexico (Beck and Odaya, 2001). Nutrient levels in the coastal rivers and streams of the Big Bend are rising, and coastal development is impinging rapidly from the northwest along the Florida Panhandle and from greater Tampa/St. Petersburg area in the south. The current quality and health of these seagrass ecosystems in the Big Bend could decline rapidly. Although monitoring records are admittedly incomplete and inconclusive, it appears that there have been recent habitat losses in this area. The conditions that led to the declines of seagrasses throughout much of the rest of the Gulf of Mexico are set to be replayed in the Big Bend. Have we learned enough from the past to not allow history to repeat itself?

Another high priority area for conservation action is the Laguna Madre of Texas, which contains the largest seagrass area in the western Gulf of Mexico subregion. Previous studies have shown that maintenance dredging of the Gulf Intracoastal Waterway, which represents one of the largest impacts to seagrasses in this bay, is not only harmful to seagrasses, but is also cost inefficient. Very few ships use the waterway in this region, and there are several more practicable alternatives for transporting supplies along this route while maintaining habitat quality (Diaz and Kelly, 1994; Onuf, 1994; Sheridan, 2004).

Towards Restoration and Conservation Success

The completion of this gulfwide status and trends report is particularly timely. The first global survey of changes in seagrass acreage recently revealed that approximately 15% of this marine resource has been lost in the last 10 yr (Green and Short, 2003). In 2000, the World Summit on Sustainable Development (Johannesburg, South Africa) adopted a commitment to reverse the trend of seagrass losses by 2010. Hard facts, such as those presented in the United Nations

Environment Programme document, will help persuade countries of the need for their improved stewardship of this important natural resource. Similarly, it is hoped that this status and trends report will go a long way in providing the justification for strong actions by Gulf of Mexico States to their remaining seagrasses and restore their eliminated or degraded seagrass beds.

For successful protection and restoration of seagrasses of the Gulf of Mexico, this status and trends report points to the need for a program to inventory and monitor seagrass change around the Gulf of Mexico, develop a restoration plan, identify the Federal, State, and local governments' partnerships, and develop priorities for the implementation of the plan and program. The program and plan would devote more time and effort for research and monitoring of these ecologically important, nearshore nursery areas and the essential fish habitat that they provide. In particular, the National Oceanic and Atmospheric Administration has a mandate to better identify, manage, and conserve these habitats, and the EPA is required to assess the impacts of coastal water quality on nearshore environments.

Opportunities should also be created for more private investment in coastal restoration and conservation. State and local governments often rely on public-private partnerships to achieve restoration of coastal habitats, and overall these have been quite successful; however, much more effort is needed. For example, Gulf of Mexico coastal States have the opportunity to lease submerged lands specifically targeted for the restoration and conservation of nearshore habitats. Submerged lands are widely available for lease and ownership in the United States. These lands have been sold almost exclusively to business interests for the exclusive use, management, and harvest of natural resources (McCay, 1998). For example, Florida leases submerged lands to more than 22,000 marinas. The leasing of submerged land for fisheries and aquaculture is a large and growing business (Goldburg and Triplett, 1997; DeVoe, 1999). More than 160,000 ha (395,360 acres) of the coast of Louisiana have been leased for oyster harvest (Louisiana Department of Wildlife and Fisheries, 2003). The leasing and ownership of submerged lands by individuals and groups, however, have rarely been used as tools for marine restoration or conservation.

A few groups are testing leasing and ownership of submerged lands for restoration and conservation, and the public-private partnerships they engender may be a welcome addition to coastal management (Beck and others, 2004). Existing statutes in Texas make it possible to lease submerged lands for restoration and conservation purposes, and these opportunities are being used to help restore and conserve salt marsh habitats in Galveston Bay (Beck and others, 2004). More such experiments in public-private conservation need to be attempted in the Gulf of Mexico, particularly for seagrass habitats.

For the past decade, the Gulf of Mexico coastal States of Texas and Florida have increased planning and conservation activities toward seagrass habitats, and the State of Mississippi has been gradually moving forward with conservation activities for the past couple of years; however, in all cases funding for seagrass inventory, monitoring, protection, and restoration has not been available for full implementation of the State plans. In addition, within the Federal Government seagrass habitat has not been a high priority for program development and action toward seagrass restoration and conservation. Small steps have been made, some independent and some coordinated, calling to action State, local, and Federal Government agencies for seagrass inventory, monitoring, protection, and restoration, for example this report on Seagrass Status and Trends in the Northern Gulf of Mexico: 1940–2002, and the EPA Gulf of Mexico Program and The Nature Conservancy which has identified priority estuaries in the northern Gulf of Mexico (Beck and others, 2000; Beck and Odaya 2001), that provide reasonable starting points for discussions on development and implementation of goals and objectives surrounding seagrass restoration and protection.

Potential priority areas like the Big Bend of Florida and the Laguna Madre of Texas represent only a portion of restoration and conservation priorities for seagrasses and their communities in the northern Gulf of Mexico. The identification of priority areas is not meant to dissuade agencies, groups, and citizens from acting to better manage seagrass meadows in all of their local embayments; however, there are areas of regional and even global importance for seagrass conservation within the northern Gulf of Mexico, and there are key opportunities to act with forethought to conserve and to restore those areas.

It is our duty to provide future generations with the opportunity to enjoy the benefits of a healthy coastal ecosystem. Expanses of healthy seagrass meadows are an integral part of that vision.

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Appendix 1.

Appendix 1. Seagrass Classification System

The seagrass classification system that has been used for a large extent of the seagrass mapping of the northern Gulf of Mexico was developed beginning in 1990 at the "Seagrass Mapping Meeting" hosted by the Environmental Mapping and Assessment Program (EMAP) of the U.S. Environmental Protection Agency. The development of the classification scheme was a collaboration among the U.S. Fish and Wildlife Service's (now in the U.S. Geological Survey) National Wetlands Research Center (NWRC) in Lafayette, La.; the National Oceanic and Atmospheric Administration's (NOAA) Center for Coastal Fisheries and Habitat Research in Beaufort, N.C.; and the Florida Department of Environmental Protection's Florida Marine Research Institute (FMRI) in St. Petersburg, Fla. The classification scheme was presented in 1992 at the "Seagrass Monitoring and Research in the Gulf of Mexico Workshop" at the Mote Marine Laboratory in Sarasota, Fla.

The seagrass classification scheme developed was designed for use with natural color aerial photography at an optimal scale of 1:24,000 or larger (Handley, 1996). At 1:24,000 scale for both the aerial photography and the base maps, the minimum mapping unit is nominally 0.4 ha (1 acre); however, at this scale seagrass units have been delineated to 0.2 ha (0.5 acre) consistently. The seagrass classification scheme was developed as a hierarchical system (fig. 1). General consensus of the seagrass mapping community was that the simpler the system, the greater the chance of consistency and improved accuracy in the mapping results, the easier the use with multiple scales of aerial photography and emulsions, and the greater the opportunity for replication of the mapping over several time periods. The intent of the scheme is to describe the morphology of the seagrasses present (percent of ground cover with respect to spatial extent), not the biomass density (shoots per square meter) of the seagrasses within the beds. While interpretation of seagrass densities has been performed with inconsistent results, morphological classification of spatial structure (i.e., continuous versus patchy) of seagrass habitats has been possible to map accurately and with consistency (Handley, 1994).

The scheme begins with determining the presence or absence of submerged aquatic vegetation—either seagrass or algae on aerial photographs. The imagery signatures of seagrass and algae are generally identical; as a result algae is indicated as present only if groundtruthed or known by expert opinion, and otherwise the signature is considered to be seagrass. The seagrasses are further subdivided into two classes—continuous beds and patchy beds. The definition of a continuous bed is an area of seagrass over 0.4 ha (1 acre) in size with less than 15% bare sediment observed within the delineated area. Patchy beds are defined as scattered units of seagrasses that are less than 0.4 ha (1 acre) in size each and have 15% or more bare sediment observed between and among the patches. Thus, numerous patches are aggregated into a single area. The patchy class is further subdivided into subclasses based on the percent ground cover of patches within the area delineated and in 5% increments: very sparse patchy seagrass (PSG1) (0%–10%, very sparse); sparse patchy seagrass (PSG2) (15%–40%, sparse); moderate patchy seagrass (PSG3) (45%–70%, moderate); and dense patchy seagrass (PSG4) (75%–85%, dense). The U.S. Department of Agriculture (USDA)

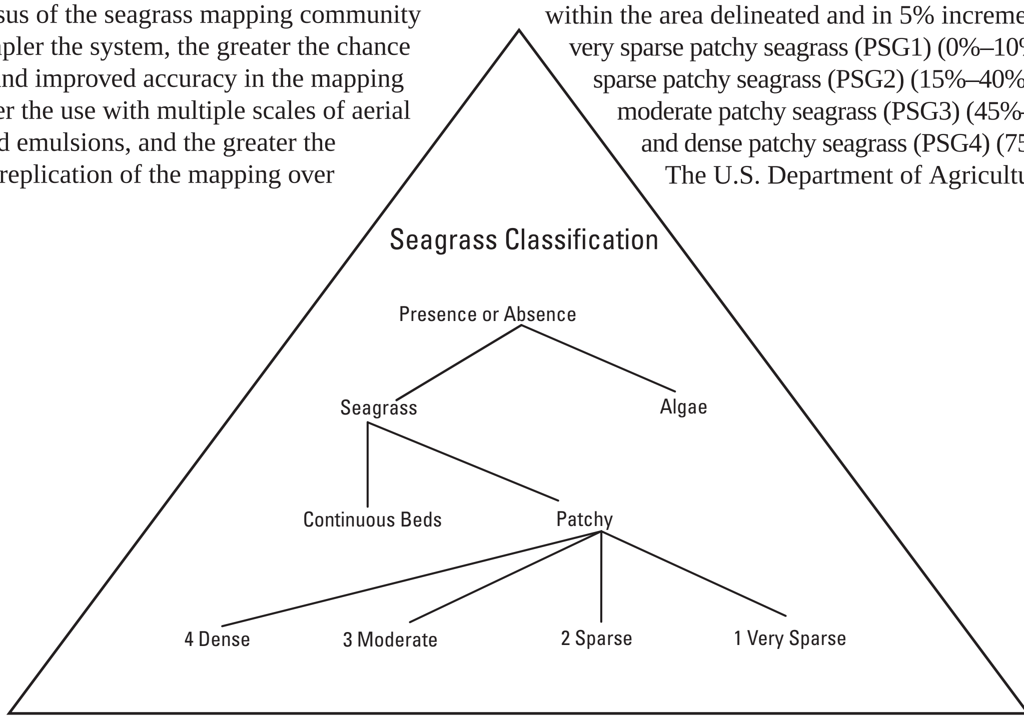

Figure 1–1. Seagrass classification scheme used in U.S. Fish and Wildlife Service and U.S. Geological Survey mapping projects.

Forest Service's percent crown cover density scale (Paine, 1981) has been modified to be used as the optical measure to determine percent cover at multiple scales of aerial photography used in the seagrass mapping projects (fig. 2).

If this seagrass classification scheme is used to map seagrass distributions, it will result in a hierarchical scheme by using a consistent methodology and proven source of imagery. This scheme does not provide a reference to the actual seagrass biomass of the continuous or patchy beds. For example, there can be sparse (less than 50 shoots per square meter) continuous beds; sparse (less than 50 shoots per square meter) continuous beds with areas of dense (greater than 50 shoots per square meter) seagrasses within them; and of course, sparse (less than 50 shoots per square meter) and dense (greater than 50 shoots per square meter) patchy beds.

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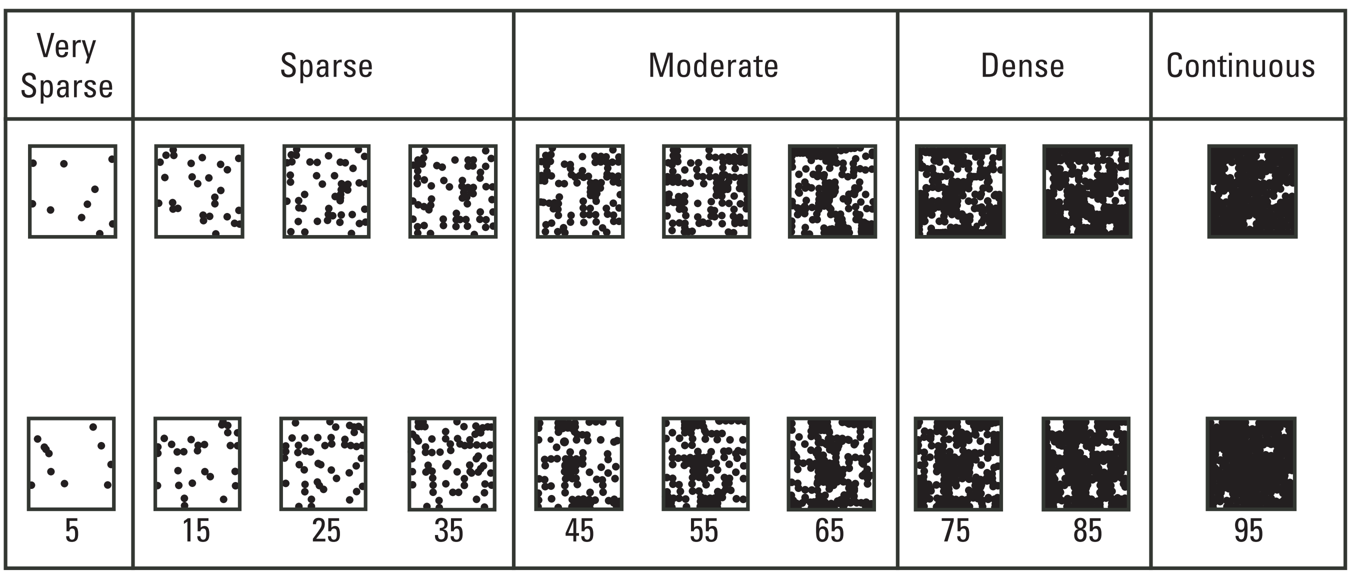

Figure 1–2. Percent cover (5% increments) of patch density and continuous seagrass for photointerpretation.

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Pages 2, 36, 37, 53, 75, 258, 259: T. Michot, U.S. Geological Survey.
Page 52: K. Madley, Florida Fish and Wildlife Conservation Commission.

Prepared by the USGS Lafayette Publishing Service Center for the
USGS National Wetlands Research Center and the Environmental
Protection Agency