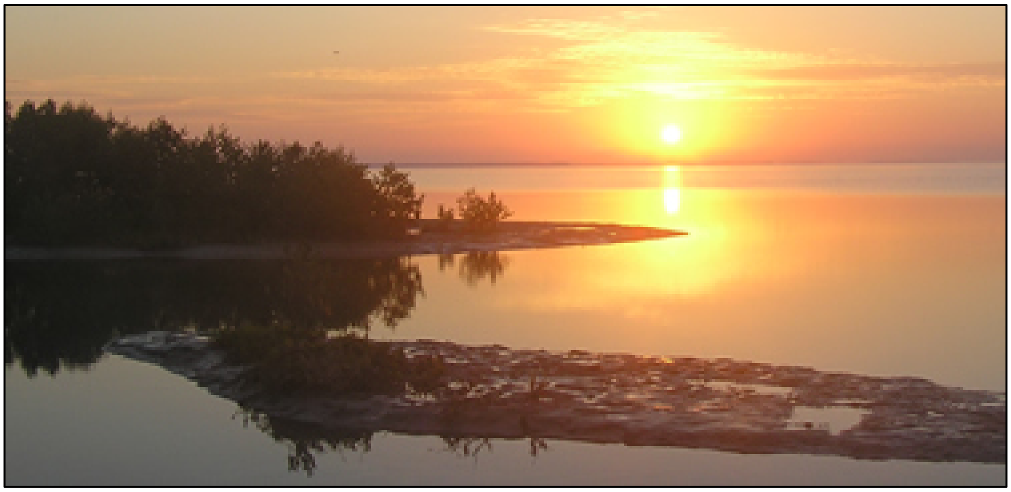
CONDITION OF THE NATURAL RESOURCES OF FLORIDA BAY, EVERGLADES NATIONAL PARK



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CONDITION OF THE NATURAL RESOURCES OF FLORIDA BAY, EVERGLADES NATIONAL PARK

A STATE OF THE PARKS TECHNICAL REPORT

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Acronyms Used in This Report

C&SFP	Central and Southern Florida Project for Flood Control and Other Purposes
CERP	Comprehensive Everglades Restoration Plan
CMECS	Coastal Marine Ecological Classification System
FHAP	Fish Habitat Assessment Program
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NPCA	National Parks Conservation Association
NPS	National Park Service
R.E.E.F	Reef Environmental Education Foundation
SEFSC	Southeast Fisheries Science Center
SFWMD	South Florida Water Management District
SFERPM	South Florida Ecosystem Restoration Prediction and Modeling Program
SEFCAR	Southeast Florida and Caribbean Recruitment Project
WCA	Water Conservation Area

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Executive Summary

Florida Bay, a shallow sub-tropical lagoon located within the southern end of Everglades National Park, is one of America's ecological treasures. The bay is home to the most significant breeding ground for wading birds in North America, provides habitats for a diverse collection of marine animals, including some high profile species that comprise a valuable recreational and commercial fishery, and is bordered by the southern extent of the largest preserved mangrove system in the Western Hemisphere. However, the bay's natural resources have been severely degraded amidst a series of landscape-scale water management projects that have altered the natural flows of freshwater into the bay. Currently, Florida Bay and the larger Everglades region are the focus of the largest restoration program ever initiated—a program that aims to reverse the decline of this world-class ecosystem while sustaining the economic viability of south Florida.

We assess the condition of Florida Bay's natural resources in this report using the methodology developed by the National Parks Conservation Association (NPCA) for its State of the Parks Program. This technical report will be used by NPCA as a resource to develop a special State of the Park Report for the Florida Bay portion of Everglades National Park. Our assessment consists primarily of a thorough review and summary of existing published scientific research, supplemented by expert input from natural resource professionals and scientists familiar with the issues affecting Florida Bay. For each natural resource indicator, we review its current condition within a historic context, describe its role and importance within the bay ecosystem, summarize the threats to the resource, and consider opportunities for restoration and management. For the purposes of this report, we focus on those portions of Florida Bay that lie within Everglades National Park, though many of the findings are relevant to the entire bay.

The ecological health of the bay's natural resources is intimately tied to the hydrology of south Florida. Under natural conditions, a delicate balance of fresh and marine waters flowed through a complex series of shallow basins in the bay, creating a mosaic of salinity and nutrient conditions that in turn supported a wide array of marine life. Although the details of this hydrologic regime are not well understood, it is clear that the hydrologic balance of the bay has been significantly altered, particularly the amount, timing and distribution of freshwater flows, by a series of extensive water drainage and diversion projects implemented in south Florida beginning in the late 1800s. In turn, these anthropogenic (or human-caused) changes have led to changes in ecosystem structure and function. In this report, we describe the extent to which the natural resources of Florida Bay are showing signs of ecological stress and the degree to which this stress is caused by natural versus anthropogenic forces. Below we highlight key findings related to water resources, air quality, marine and estuarine ecosystems, and representative species of wildlife.

Water Resources

- Over the 20th century, Florida Bay **salinity levels** rose slightly and the bay became more hydrologically isolated from both marine and freshwater influences. The primary driving force of salinity levels within the bay is annual rainfall. Anthropogenic disturbances play a secondary role, with more localized effects.

- The timing, distribution, and quantity of overland **freshwater inflows** into Florida Bay have been altered by water diversion and drainage practices in south Florida. Only 20% of the freshwater that flows through the Everglades is delivered to the bay. Freshwater inputs have been reduced by as much as 59%.

- Florida Bay, especially the central bay, has recently (1974-75, 1989-90, 2000-01, and 2004-05) experienced **hypersalinity** events, where hydrologic isolation coupled with drought conditions—and perhaps exacerbated by diminished overland freshwater flows—have raised salinity above normal oceanic levels.

- Water quality in the bay has improved since monitoring began in the late 1980s, though **phytoplankton blooms** and **decreased clarity** have been detected, especially in the bay's central and western portions.

Air Quality

- High local emission and deposition rates of mercury have led to **mercury contamination** and bioaccumulation in birds and fishes in Florida Bay and nearby areas. Local sources of mercury include incineration of municipal and medical waste, sugar industry activities, and fossil fuel power plants. Rates of mercury emissions and depositions peaked in the late 1980s and have subsequently declined due to strict regulations. However, mercury levels in many animals in Florida Bay and Everglades National Park have remained stable or increased, with some fish species still listed on consumption advisory lists.

Marine and Estuarine Ecosystems

- Seagrasses such as turtle grass are a keystone ecosystem within the bay, providing refuge, spawning areas and a food source for many important fish and invertebrate species. Florida Bay experienced a massive **seagrass die-off** event in 1987-1989, initially killing 4,000 hectares of turtle grass beds, and thinning the standing stock in an additional 23,000 hectares. A series of cascading ecological events that followed is likely related to this die-off, including phytoplankton blooms, increased turbidity, sponge die-off, and shifting nursery habitats for spiny lobster. While the cause of the seagrass die-off has not been definitively identified, a leading hypothesis invokes multiple stressors, including drought, reduced freshwater flows, and hypersalinity, leading to physiological stress and eventual sulfide poisoning of the turtle grasses.

- A combination of sea-level rise and diminished freshwater inflows has led to changes in the distribution and functioning of **mangroves**, an important foraging habitat for wading birds and a nursery area for many fish species.

Wildlife

- **Wading bird** populations within Florida Bay and surrounding areas have declined by 90 to 95% from early observations in the 1930s. The roseate spoonbill, a representative species chosen for this assessment, was nearly extirpated by plume hunting in the late 1800s and early 1900s, then rebounded after protection until the 1970s, and has since declined. Spatial patterns of nesting suggest that destruction and alteration of mangrove habitats, which are key foraging habitats for spoonbills, may be a primary cause of the decline.

- Florida Bay serves as an important nursery for many marine species, including **spiny lobster**, a valuable commercial fishery in Florida. Spiny lobster populations within Florida Bay declined in the early 1990s, but have since rebounded.

- Spotted seatrout and gray snapper, two representative and commercially important **marine fish** species in Florida Bay, are currently overfished. The population levels of a third representative species, goliath grouper, have been drastically reduced from historic numbers, indicating that the current fishing moratorium for this species should be continued.

Beginning in 1993 and continuing to date, scientists and local, state and federal agencies have conducted a substantial body of scientific research to determine the cause of these many ecosystem changes. This research is informing a massive effort to restore the Everglades, known as the Comprehensive Everglades Restoration Plan, as well as related restoration efforts aimed specifically at Florida Bay. These plans call for restoring the historic flows of freshwater into Florida Bay, a step that is essential to restoring and maintaining the ecological health and economic value of this unique natural treasure. Given the complex nature of coastal hydrology in the region and the limited understanding of its effects on natural processes and biological resources, restoration goals will only be met if scientific research continues to inform restoration and management efforts.

Introduction

Origins and Orientation

At 1.5 million acres, Everglades National Park is the largest designated subtropical wilderness area in North America, contains the largest mangrove system preserved in the western hemisphere, safeguards habitats that historically supported vast flocks of wading birds, and harbors significant marine and estuarine resources in Florida Bay (NPS 2003, NPS 1999). Everglades N.P. (herein referred to as "Everglades N.P." or "the park") is situated in southern Florida, comprising the southern extent of the Florida peninsula, and beyond into a marine lagoon known as Florida Bay (Figure 1). It is unique within the National Park System for containing significant terrestrial, freshwater and marine ecosystems and resources, and the only site in the United States that is designated as a World Heritage Site, International Biosphere Reserve, and a Wetland of International Importance (NPS 2003).

The park owes its uniqueness to the convergence of large geologic, bio-geographical and hydrologic processes that impart a blending of tropical and temperate floras, mixing of marine and fresh waters, and balancing of rising and falling sea levels (Lodge 2005). The story of Everglades N.P. is a dynamic story of water, how water flows through the land and shapes landscape elements, how water influences the functioning of ecosystems, how water flow drives the distributions of plants, animals and ecosystems, and importantly, how water flow has been altered by human activities to the detriment of resources within its boundaries.

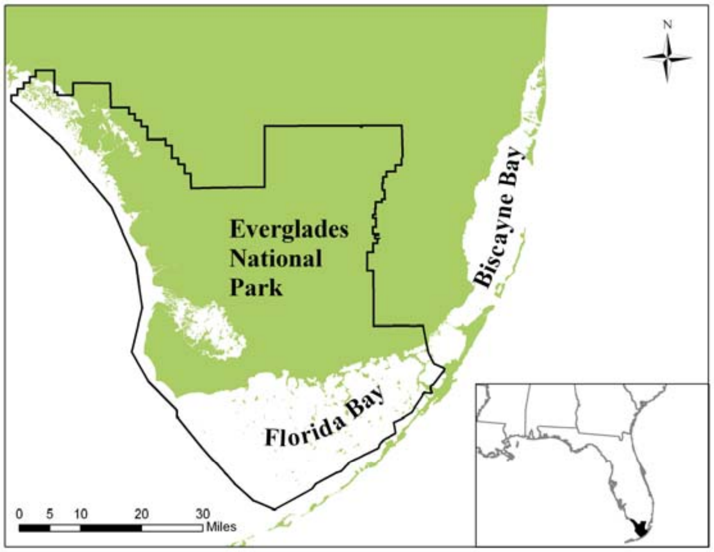


Figure 1. Map of Everglades National Park, located in southern Florida. Sources: NPS 2005a, National Atlas 2005.

Three regions comprise Everglades N.P., and these are intimately tied to one another by water flow: in the north, a large wetland complex dominated by sawgrass marsh ridges, but also characterized by river sloughs and tree islands, grading into mangrove estuaries that filter and process water and materials into a large marine lagoon, known as Florida Bay. Water flows from the north to the south along a slight elevation gradient, declining southward at three inches per mile (SCT 2003), in a process known as sheet flow. Unlike typical wetlands that receive water from seasonal or episodic flooding of river basins, water flow in the Everglades is much more subtle and almost imperceptible, flowing at an average rate of only one inch per second (SCT 2003).

Florida Bay, the focus of this report, is a shallow sub-tropical lagoon that forms the southern end of Everglades National Park. The lagoon harbors seagrass meadows and mangrove islands (Figure 2) dispersed throughout a mosaic of shallow basins and mud banks (Figure 3, Fourqurean and Robblee 1999). Many fish species range between the bay and adjacent coastal reefs within their life cycle. Over 250 species of fishes occur within the coastal waters of the park (Loftus 2000, NPS 2005b) and by (or before) reaching maturity, most leave the boundaries of the Everglades N.P. for adult habitat and spawning sites located elsewhere (Tabb 1966).

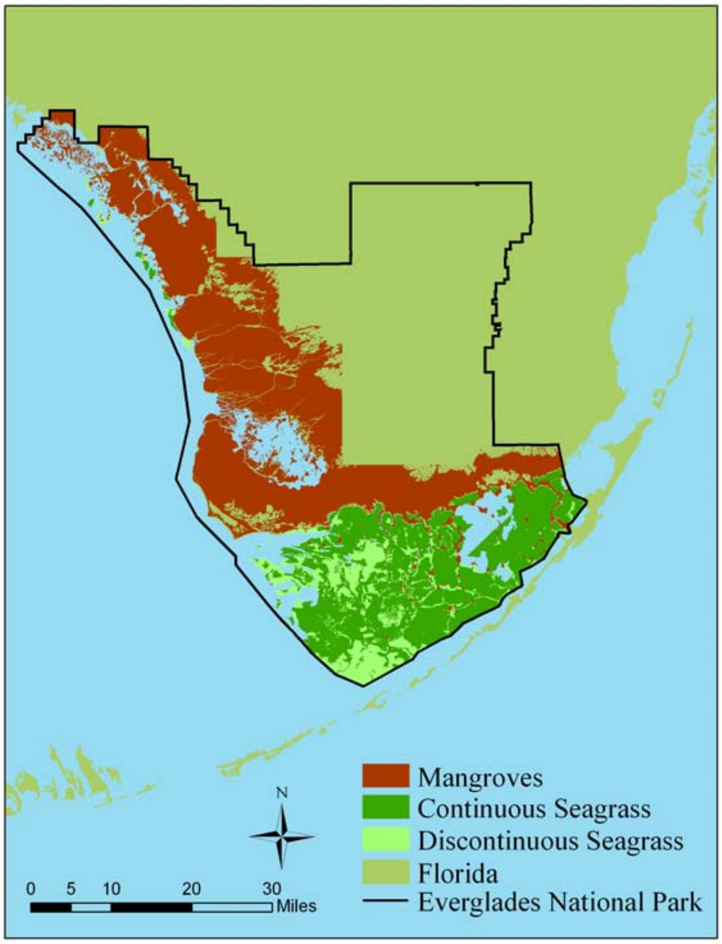


Figure 2. Vegetation communities of Florida Bay, Everglades National Park. Sources: NPS 2005a, National Atlas 2005, FMRI 2001, FMRI 1990.

The bay is a world class destination for recreational fisherman, bringing in an estimated $300 million dollars in annual revenue. Though commercial fishing is not permitted within Florida Bay, it supports habitats and nurseries for game fisheries (including spiny lobster and pink shrimp) that form the foundation of commercial fishing in waters bordering Florida Bay, bringing in an additional $100 million per year. Marine and mangrove estuary habitats within the bay also support the most significant breeding grounds for wading birds in North America (NPS 1999), a resource that has been diminished by up to 90-95 % (NPS 1997, Odgen 1994).

Authorized in 1934 and established in 1947 by President Harry S. Truman, Everglades N.P. was one of the first national parks to be established to protect significant biological resources (NPS 1999). Commercial fishermen were allowed to fish in Florida Bay until 1950, when the bay was included in Everglades N.P. Early conservationists recognized that the unique biological resources within Everglades N.P. were being jeopardized by exploitation and adjacent land uses that were incompatible with their persistence. Unfortunately, those resources are still in peril, even though the park has been under protection for almost sixty years. Everglades N.P. illustrates, perhaps more than any other park in the National Park System, that the condition of park resources is dependent not only on land management within the park, but more importantly, land and water management outside the park.

Because all natural resources within Everglades N.P. (including Florida Bay) are dependent upon a delicate ecological and hydrological balance, many ecosystems within the park are exhibiting signs of ecological stress. On the mainland, hydrology and the historic ridge and slough landscapes have been severely altered due to major water drainage efforts beginning in the late 1800s and continuing to present. Only one-half of the original “Rivers of Grass” wetland landscape exists today, only one-fifth is protected within Everglades N.P. (Davis et al. 1994). Vegetation change, invasion by alien plant species, changes in the fire regime and soil subsidence have all occurred in the post-drainage era (Sklar et al. 2002). In Florida Bay, human disturbances have been implicated in a series of abiotic and biotic changes: increase in hypersaline events, loss of water clarity, massive seagrass die-off, sponge mortality, phytoplankton blooms, reductions in wading bird populations, and diminishment of recreationally important fisheries (Nuttle et al. 2003).

Persistent human perturbations exacerbate variations in the natural environment and thus contribute to ecosystem stress (NMFS 1995). Effective restoration and management cannot be achieved without understanding the complex ecological controls and linkages inherit in such a system. Beginning in 1993 and continuing to date, a substantial body of scientific research by local, state and federal agencies has been conducted to determine the cause of these changes, and to inform a massive effort to restore the Everglades, called the Comprehensive Everglades Restoration Plan (CERP). An overarching hypothesis that has driven much of the research in Florida Bay and a central theme of this report is that water diversion has led to the changes in the quantity, quality, timing, and distribution of freshwater flow delivery into Florida Bay, and that this change has led to concomitant changes in ecosystem structure and function.

Biogeographic and Physical Setting

Park Location and Size

Florida Bay (ca. 25°05'N 81°45'W) is a wedge-shaped body of water that has an area of 2200 km^2, 1800 km^2 of which occur in Everglades N.P. (Figure 1, McIvor et al. 1994). For the purposes of this report, the condition of natural resources in Florida Bay that are within Everglades N.P. are documented, though many of the findings are relevant to the entire bay. The northern boundary of the bay is the southern tip of the Florida peninsula, while the eastern boundary is formed by the Florida Keys, an archipelago that sweeps in a southwestward arc from the Florida peninsula to Key Largo. The bay opens to the west into the Gulf of Mexico, which bathes the western bay in oceanic water.

Everglades N.P., in turn, covers 1.5 million acres (2,358 mi^2) in south Florida (Figure 1; NPS 2002), having been expanded several times since its establishment. Originally 460,000 acres in 1947, it was expanded in 1950 to 1.2 million acres to incorporate Florida Bay and other lands, in 1958 to 1.4 million acres, and finally in 1989, where the eastern margin of Everglades N.P. was expanded by the acquisition of an additional 109,000 acres (NPS 1999).

Climate Regime

Florida Bay has a tropical climate characterized by two seasons: a wet season from May to October and a dry season from November to April (NPS 2002). The average precipitation, which has remained relatively constant for the past 100 years, is 60 inches (152 cm) per year (Duever et al 1994). Seventy-five percent of this precipitation falls in the wet season (Duever et al 1994). Average annual temperature is 24.5°C, with a low monthly mean of 20°C in January, and a high of 28°C in August (NPS 2002).

Other than relatively high rainfall and mild temperatures, the region experiences extreme climate events, including freezes, drought and tropical cyclones (e.g. tropical storms, hurricanes). These events vary over time and space, but can have a large effect on the abiotic and biotic processes (Duever et al 1994). Hurricanes have received recent attention, given the occurrence of four hurricanes in Florida in 2004. From 1871-1986, at least one tropical storm or hurricane has impacted Everglades N.P. every year, with a high of 21 events in 1933 (Duever et al. 1994). These storms can generate storm surges in the coastal areas and heavy rains and winds in the non-coastal areas. Hurricane Andrew, for example, laid bare a 50 km swath of vegetation, where damage was greatest in areas of hardwood vegetation and less so in pine and cypress lands (Duever et al. 1994).

Geology and Landforms

Florida Bay is underlaid by a karst limestone bedrock, called Miami limestone, formed during the Pleistocene (Gleason and Stone 1994), and comprised of many sink holes (Fourqurean and Robblee 1999). The formation of the Everglades marshland and Florida Bay occurred recently, taking form roughly 5,000 years ago. The ecosystems emerged and formed during a warming trend after the last ice age which began 15,000 years ago and was associated with rising sea levels (Gleason and Stone 1994).

The bathymetry of Florida Bay is notable for a network of large shallow mud banks and emergent islands or keys (Figure 3). The mud banks create numerous distinct submerged basins that are more or less hydrologically isolated from each other (Wanless et al. 1994), and are biogenic in origin, consisting of the remains of green algae, seagrass epiphytes, mollusks and stony corals (Fourqurean and Robblee 1999). Sediment levels within the basins themselves increase along a northeast to southwest gradient, a gradient that is also notable for increasing benthic plant productivity (Zieman et al. 1989).

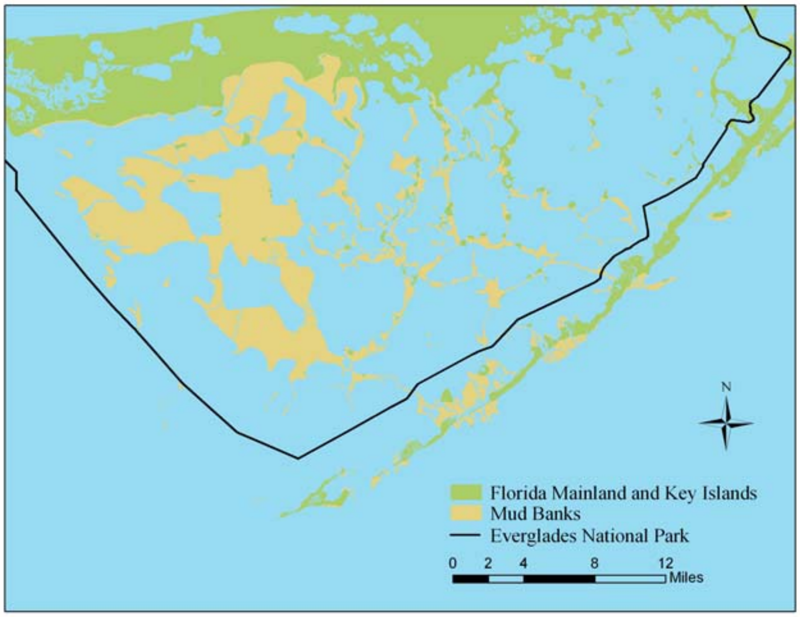


Figure 3. Key Islands and mud banks in Florida Bay, Everglades National Park. Sources: NPS 2005a, National Atlas 2005, FGDL 1999.

The network of banks and keys reduce penetration of fresh water and nutrient inputs from the Everglades, restrict tidal exchange with the Atlantic Ocean, and constrain circulation from the Gulf of Mexico (Halley and Roulier 1999). Because the mud banks dampen tidal energy, almost all of the hydrologic circulation in the bay is wind-driven, especially in the east where the islands of the Keys restrict the tidal signal from the Atlantic Ocean. In the western bay, where the Gulf of Mexico can exchange freely with bay waters, tidal circulation is driven by a gulf tide range with a diurnal amplitude of no greater than 0.5 m (Smith 1994). Banks and shoals attenuate the Gulf tidal signal so that the interior bay effectively has no astronomical tide. Restricted circulation within the estuary prevents both sea water and fresh water from circulating quickly through the bay and, frequently in the late dry season and summer months, this allows evaporation to raise salinity in the central bay to above oceanic levels, sometimes considerably so (Smith and Pitts 2002).

Hydrologic Regime

To a large extent, the distribution and flow of water, and how these changed over time and space under natural conditions, define the Everglades. Historically, Florida Bay was an estuarine-lagoon system (Zieman 1982), with considerable fresh water entering the bay via slow, diffuse discharge from the Everglades. Fresh water entered the bay through channels, overland sheetflow and groundwater (Wanless et al. 1994). Flow from Taylor Slough and the Everglades Panhandle seasonally maintained an increasing estuarine salinity gradient from the northeast to the southwest bay (Wanless et al. 1994, Light and Diveen 1994). Local rainfall has also been important to the hydrologic cycle of the bay (Anderson et al. 1999).

The drainage and compartmentalization within the Greater Everglades Ecosystem in the last 150 years has had a major impact on the natural hydrologic regime of the entire region and specifically within Everglades N.P. Specific to Florida Bay, Smith et al 1989 estimates that as much as 59% of the freshwater flow into Florida Bay from Shark River Slough has been lost to upstream water management. Recent measurements of freshwater discharge in the area indicate that only 20% of the freshwater input is discharged into Florida Bay, with 80% discharged to the Atlantic Ocean (Light and Dineen 1994). The major hydrologic changes to the terrestrial and marine (Florida Bay) components of Everglades N.P. are discussed below.

Hydrologic Regime: Terrestrial Everglades N.P.

Water dynamics drive the landscape functions, species distributions, and community dynamics in the terrestrial and freshwater portions of Everglades N.P. Specifically, water flow played an important role in the maintenance and functioning of the slough/ridge landscape in the Greater Everglades Ecosystem (SCT 2003, Nuttle et al. 2003). Hydroperiod, the number of days that a region is inundated with water, influenced the distribution and biological activities (e.g. feeding, breeding, nesting) of many species, especially wading birds (Odgen 1994).

Hydrologic Regime: Florida Bay–Zones of Influence

As noted earlier, water flow within Florida Bay is relatively restricted by the complex structure of basins formed by mud banks and the low water depths (~ 1 meter) within these basins. Restricted hydrology is confounded by a northeast to southwest inflow of freshwater from Everglades N.P. and a western to eastern influx of marine waters from the Gulf of Mexico. These spatially heterogeneous patterns of flow and geomorphology create "zones of influence" within the bay, or ecologically distinct areas that exhibit similar patterns of hydrologic flow, water quality, nutrient flux, and biological processes (Figure 4, Boyer et al. 1997). A multivariate analysis applied to a six year record of water quality parameters resulted in the aggregation of data into three zones of influence within Florida Bay proper (Figure 4, Boyer et al. 1997). The northeast region is most influenced by freshwater influxes from Everglades N.P., both from Taylor Slough creeks and the C-111 canal. It is isolated from tidal and oceanic processes. The central region is the most geomorphologically isolated region in Florida Bay. Direct freshwater influxes into the region occur only during extreme climatic events. Both freshwater and marine influxes into the region are restricted by the configuration of mud banks. The western region is affected by marine inflows from the Gulf of Mexico, and only indirectly by freshwater flows from shark river slough that drains through Oyster Bay and through the Ten Thousands Lakes region. We use a modification of the three-sector regionalization scheme that adds a fourth to include the mangrove transition zone at the Everglades N.P. and Florida Bay interface.

Hydrologic Regime: Florida Bay–Driving Forces

Water flow within Florida Bay is controlled by at least five important driving forces: precipitation, evaporation, surface freshwater inputs, groundwater inputs, and Gulf of Mexico currents. These driving forces, which exhibit high degrees of spatial and temporal heterogeneity, determine patterns of salinity across the bay (i.e. the relative balance of freshwater and marine water inputs, less the outputs of evaporation), which in turn, influences the distribution, abundance, and interactions of many species.

Rainfall varies on multiple time scales: seasonal, interannual, decadal, and even longer time periods, and can be punctuated by episodic climatic events, like droughts and hurricanes (Nuttle et al. 2003). In a synthesis of the major ecosystem history studies conducted in the bay, rainfall was found to be the most significant driver of historic salinity patterns.

Across the bay, evaporation is roughly equal to precipitation (Nuttle et al. 2000). Seasonal and annual fluctuations in these rates can led to local changes in salinity, especially in the estuary zone closest to the mainland and in the hydrologically isolated basins of the central bay. For example, the central basins of the bay receive minimal freshwater inputs (e.g. rainfall inputs only), and because of restricted water exchange and long resident times, evaporation rates can be high, especially in the summer months, leading to hypersaline events and to a concentration of nutrients (Robblee et al. 2001).

Northeast Florida Bay receives concentrated inputs of surface freshwater from Taylor Slough and the C-111 basin (Nuttle et al. 2003). Under natural conditions, these flows mimicked regional rainfall patterns, lower in drought years, higher during episodic climatic events (Light and Dineen 1994). However, to meet various water management objectives, flows into northeast Florida Bay are asynchronous from natural flows, leading to wide variations in salinity in this region. Florida Bay also receives indirect inputs of surface freshwater from Shark River Slough discharge, after it has been mixed with ocean currents with the Gulf of Mexico.

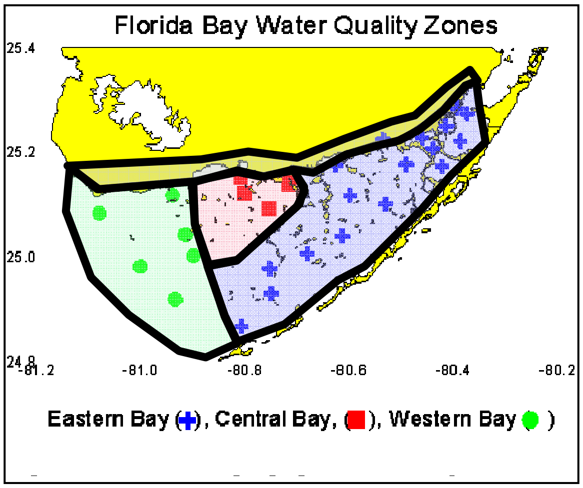


Figure 4. Water Quality Zones or Zones of Influence in Florida Bay, Everglades National Park. Concept adapted from Boyer et al. 1997.

The current rates of groundwater flows into Florida Bay are poorly known, but are thought to be reduced from natural levels due to the lowering of the water table caused by water diversion and subsequent saltwater intrusion (Nuttle et al. 2003). Some investigators have hypothesized that groundwater flows act as a significant external source of nutrients in the bay (Nuttle et al. 2003). In addition, Florida Bay's open boundary with the Gulf of Mexico and the prevalent currents lead to an important influx of marine waters and exogenous nutrients into the bay (Fourqurean et al. 1993).

Hydrologic Regime: Florida Bay–Sea Level Rise

From 1846 to 1992, the sea level at Key West, just southwest of Florida Bay, has risen by 30 centimeters, at a rate of 0.19 cm/yr (Maul and Martin 1993). Though it is unclear how much of this change is natural as opposed to human-induced climate change, sea level rise is an important factor that has influenced the structure and function of Florida Bay from the beginning. In fact, sea level rise has characterized the recent history of the region, where during the formation of the Everglades it rose by six feet per year, slowing to three-quarters feet per year from 5,500 to about 3,000 year ago, and slowing again to one foot per thousand years (Lodge 2005). Since 1932, sea level rise has increased sharply, where the present rate is almost ten-fold of this previous rate (Lodge 2005, Maul and Martin 1993). This assessment has found no empirical evidence that indicates what proportion of this recent acceleration in sea level rise, if any at all, may be accounted for anthropogenic climate change.

Hydrologic Regime: Florida Bay–Water Budget

From a water budget perspective, Nuttle et al. 2000 estimated that precipitation and evaporation are roughly equal on an annual basis, whereas water flows from freshwater sources are about 10% of annual rainfall totals. The majority of this freshwater inflow occurs in northeastern Florida Bay, where creek discharges amount to 67% of annual rainfall. Trout creek carries approximately 50% of this flow into this region of the bay (Hittle 2001). As noted earlier, the major source or marine waters is from the west, where there is an open boundary between Florida Bay and the Gulf of Mexico.

Hydrologic Regime: Florida Bay–Influences on Nutrient Fluxes and Biological Processes

These hydrologic flows influence patterns of nutrient exchange and productivity. Overall, Florida Bay is considered an oligotrophic, or nutrient-poor, lagoon. Though phosphorus (herein referred to as “P”) seems to be the primary limiting nutrient, there are actually gradients in P and nitrogen (herein referred to as “N”) availability across the bay. P is highest in the west and lowest in the east, following the flow of the major source of external P to the bay, the Gulf of Mexico (Fourqurean and Robblee 1999, Nuttle et al. 2003, Rudnick et al. 1999). N is highest in the northeast and lowest in the west given that freshwater inflows are the major source of external N (Fourqurean and Robblee 1999, Nuttle et al. 2003, Rudnick et al. 1999). These patterns in nutrient availability influence patterns of benthic community standing stock and production (Fourqurean et al. 1993, Zieman et al. 1989) and phytoplankton blooms within the bay.

Ecological and Habitat Classifications

As a framework for organizing ecological information associated with Florida Bay, the emerging NatureServe habitat classification, called the Coastal Marine Ecological Classification System (CMECS) (Madden and Grossman 2004) was employed. CMECS is an ecosystem-oriented, science-based framework for the identification, inventory, and description of coastal and marine habitats and biodiversity. The CMECS is designed to provide a framework for developing a consistent and universally recognized inventory of all habitats of the North American coasts and oceans. The flexibility of this classification supports a variety of local and regional applications.

Consisting of eight nested levels, the CMECS is hierarchical, extending from ecological regions at the largest spatial scale to habitat and associated biotopes at the smallest scale, within the following structure:

Level 1	**Ecological Region**: large regions of the coasts and oceans defined by similar physical and/or biological characteristics;
Level 2	**Regime**: areas defined by the presence or absence of fresh water;
Level 3	**System**: areas that form estuaries, estuarine-influenced areas, or marine waters of shallow (0-30 m), deeper (30-200 m) or very deep (>200 m) water columns;
Level 4	**Hydroform/Geoform**: large physical structures formed by either water or solid substrate within systems;
Level 5	**Zone**: the water column, littoral or sea bottom;
Level 6	**Macrohabitat**: large physical structures that contain multiple habitats;
Level 7	**Habitat**: a specific combination of physical and energy characteristics that creates a suitable place for colonization or use by biota;
Level 8	**Biotope**: the characteristic biology associated with a specific habitat.

Linkages between levels of the hierarchy are defined by ecosystem processes and by spatial relationships. The classification was designed to articulate with existing national fresh water and terrestrial classification standards.

Florida Bay can be classified at the upper levels of the CMECS as follows: (1) Floridian-Bahamian ecological region; (2) Fresh water influenced; and (3) Estuarine and Nearshore Marine. Within each of the two level 3 divisions, several ecological units are found in Florida Bay: (1) Island, Reef, Estuary and Marine Lagoon Geoforms; and (2) Littoral, Benthic and Water Column Zones. Each of the large geoforms in the bay can be further classified into the following macrohabitats: Mangrove Swamp, Submersed Bed, Softbottom, Hardbottom, and Coral Reef, with each having multiple habitats and biotopes.

The major vegetation communities within Florida Bay assessed in this report are macrohabitats within Level 6 of the hierarchy. They consist of the mangrove ecotone at the interface between the Everglades marsh and the Florida Bay estuary, the mangroves on key islands, and the prominent seagrass beds that carpet the bay bottom (Figure 2). Mangrove species include: *Rhizophora mangle, Avecinnia germinans, Laguncularia racemosa,* and *Conocarpus erectus*. The primary seagrass species include: *Thalassia testudinum, Halodule wrightii, Ruppia maratima*, and *Syringodium filiforme.*

Native and Introduced Species

Everglades N.P. contains roughly 1,000 vascular plants, 250 fish species, over 300 bird species, and smaller numbers of mammals, birds, reptiles and amphibians (NPS 2005b, Lodge 2005, Loftus 2000). The flora and fauna, especially for marine plants and animals, are characterized by a convergence of species of temperate and tropical origins (Lodge 2005). Many terrestrial plants, especially hardwood trees, originated in the tropics, while many of the marine vertebrate and invertebrate species are also from the tropical region, where planktonic larvae are dispersed by ocean currents, most notably the Florida Current (Lodge 2005).

Non-native invasive species are a major force in Everglades N.P., but not as much in Florida Bay, where non-native species have not yet been able to become established. Though outside the scope of this project, non-native terrestrial species, especially vascular plants, are a major stressor in the terrestrial ecosystems of Everglades N.P., where they comprise up to one-third of the plants now found in Everglades N.P. Non-native plants with large areas of infestation include: Brazilian pepper (*Schinus terebinthifolius*), Australian pine (*Casuarina equisetifolia*), Old World climbing fern (*Lygodium microphyllum*), and Melaleuca (*Melaleuca quinquenervia).*

The National Park Service Inventory and Monitoring Program conducts inventories of all vascular plant and vertebrate taxa, but to date, only the list of fish within Everglades N.P. has been completed and certified (see Appendix). Therefore, we are confident only of the fish tallies; the other numbers reported here should be used only as rough estimates of the diversity within the park.

Endangered Species

There are 21 species that occur within Everglades N.P. with designations as endangered or threatened under the U.S. Endangered Species Act (Table 1, NPS 2005c). An additional seven species are candidates for listing. Of the threatened and endangered species, seven are birds, seven are reptiles, two are mammals, one is an invertebrate, and four are vascular plants. On this list include several high profile species: the Florida panther, where perhaps only 30-50 individuals, 10 within Everglades N.P., may exist in

Scientific Name	Common Name	G-RANK	S-RANK
Endangered			
Ammodramus maritimus mirabilis	Cape Sable seaside sparrow	G4T1	S1
Amorpha crenulata	crenulate lead-plant	G4T1	S1
Cereus eriophorus var. fragrans	fragrant pricklyapples	G1	S1
Chamaesyce deltoidea ssp. deltoidea	deltoid spurge, Redland sandmat	G2T1	S1
Chelonia mydas	green sea turtle	G3	S2
Crocodylus acutus	American crocodile	G2	S1
Dermochelys coriacea	leatherback sea turtle	G2	S2
Eretmochelys imbricata	hawksbill sea turtle	G3	S1
Lepidochelys kempii	Kemp's ridley sea turtle	G1	S1
Mycteria americana	wood stork	G4	S2
Picoides borealis	red-cockaded woodpecker	G3	S2
Puma concolor coryi	Florida panther	G5T1	S1
Rostrhamus sociabilis plumbeus	Everglade snail kite	G4G5T2	S2
Sterna dougallii dougallii	roseate tern	G4T3	S1
Trichechus manatus	West Indian manatee	G2	S2
Threatened			
Caretta caretta	loggerhead sea turtle	G3	S3
Chamaesyce garberi	Garber's spurge	G1	S1
Charadrius melodus	piping plover	G3	S2
Drymarchon corais couperi	eastern indigo snake	G3	S3
Haliaeetus leucocephalus	bald eagle	G5	S3
Orthalicus reses	stock island tree snail	G2	S1
Candidate			
Argythamnia blodgettii	Blodgett's silverbrush	G2	S2
Chamaesyce deltoidea pinetorum	Pineland sandmat	G2T1	S1
Chromolaena frustrata	Cape Sable thoroughwort	G1	S1
Dalea carthagenensis floridana	Florida prairie-flower	G5T1	S1
Digitaria pauciflora	Florida pineland crabgrass	G2	S1
Hemiargus thomasi bethunebakeri	Miami blue butterfly	G3G4T1	S1
Sideroxylon reclinatum ssp. austrofloridense	Everglades bully	G4G5T1	S1

Table 1. List of species occurring in Everglades National Park and on the federal endangered species list. Source: NPS 2005c.

the wild; the Florida manatee, which is impacted by boat prop propellers, poaching and habitat destruction; and the wood stork, whose numbers within Everglades N.P. and the surrounding regions have declined precipitously in the 20th century as has the abundance of other wading birds (NPS 1997).

Given the paucity of marine species listed on the federal endangered species list, several other state and federal lists, including the IUCN and the American Fisheries Society list, are important in distinguishing marine species of concern. Included on these lists is the smalltooth sawfish, (*Pristas pectinata*), a fish that is restricted to Florida Bay and Ten Thousand Islands. An Everglades N.P. gamefish monitoring program serves as the major fisheries monitoring tool for this species (Schmidt 2005).

Unique Designations

Everglades N.P. is a protected area of international and global significance. Designated in 1976, 1979, and 1994 respectively, Everglades N.P. is one of only three sites to be recognized on three international lists: Biosphere Reserve, World Heritage, and Wetland of International Importance (Maltby and Dugan 1994).

Regional and Historical Context

Florida Bay and Everglades National Park are situated at the southern end of what once was a vast watershed known as the *Greater Everglades Ecosystem* (Figure 5). With headwaters in the Kissimmee River Valley and an outlet into the Florida Bay, this watershed once spanned 10,890 mi^2, and was 160 miles long and 62 miles wide (Nuttle et al. 2003). Water flowed in a 30-mile expanse of marshes, from north to south, almost imperceptibly, along a subtle downward gradient of three inches per mile. During the wet season, the southern banks of Lake Okeechobee would overflow and flood into the vast marshes. Ridges of sawgrass marsh covered the landscape, broken up by lower elevation sloughs that meandered in a northeast to southwest direction and delivered water below. Tree islands, higher elevation and drier sites, dotted the landscape (Figure 6a), appearing like tear-drop ellipses across the landscape. These landscape elements were orientated in a north-south axis that ran parallel to the direction of water flow.

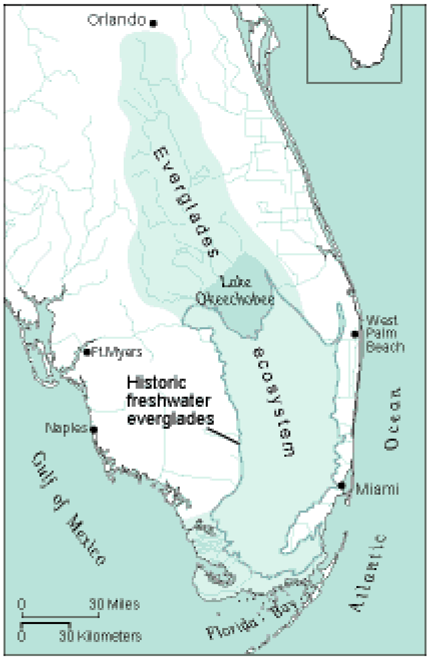
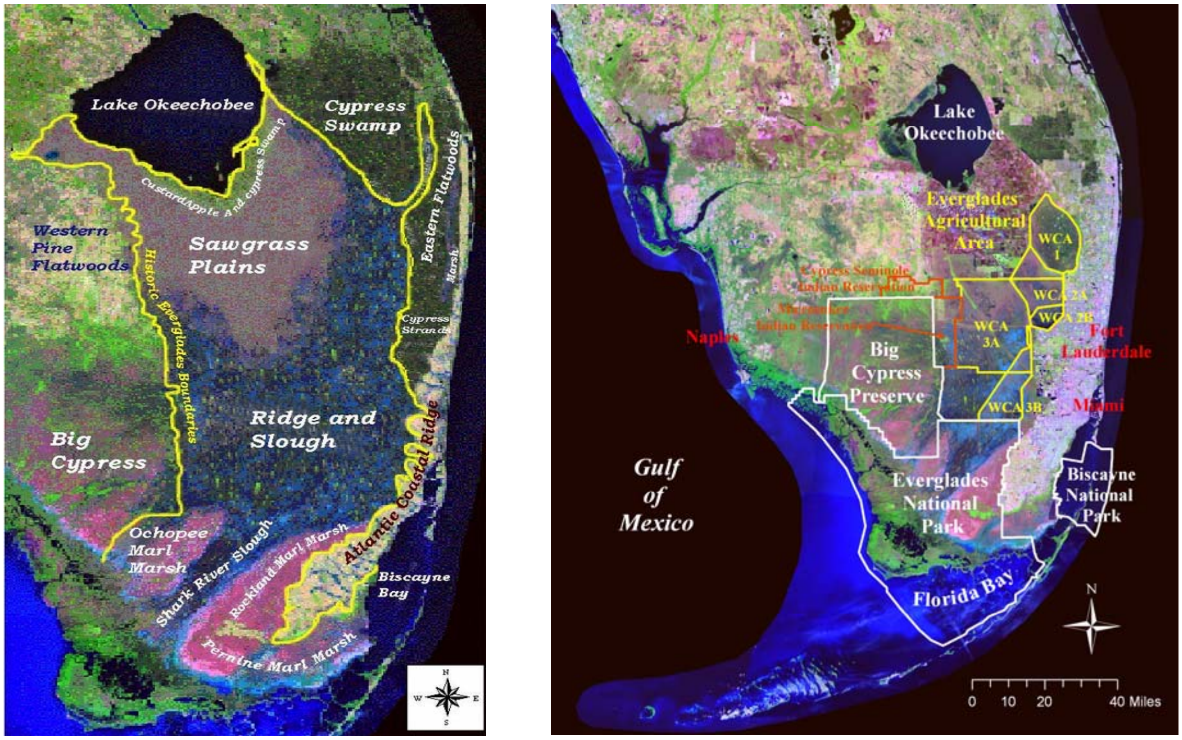


Figure 5. Map of the historic distribution of the Greater Everglades Ecosystem before human modification. Source: Ingebritsen et al. 1999.

These landscape units can also be depicted as vegetation units, sawgrass plains and wet prairie/slough, in a map the historic Everglades vegetation pieced together by a series of early pre-drainage surveys (Davis et al. 1994). In addition to these vegetation types, a custard apple swamp forest on the southern rim of Lake Okeechobee, wet prairie, cypress strands, and a marl marsh were also evident in the early landscape (Davis et al. 1994).



(a) **(b)**

Figure 6. Simulated satellite image of the Everglades landscape prior to human modification (a), and satellite image of the modern-day Everglades landscape (b), showing surrounding land uses and loss of original habitats and landscape units. Sources: (a) from SCT 2003 and (b) from NPS 2005a, SFWMD 2005a, FGDL 1999. Concept for (b) after Lodge 2005.

Land Use History

Well before the establishment of Everglades N.P., dating to the late 1800's, humans began to alter this historic landscape and vegetation in efforts to make the region more suitable for human uses and exploitation. Under a series of local, state and federal engineering projects to promote urban growth, control flooding, and secure water supplies for agriculture and groundwater recharge, the Everglades has been drained, dredged and compartmentalized.

Three waves of development occurred. In the first wave from 1880-1883, Hamilton Disston, a Philadelphia businessman who was granted four million acres by the state of Florida, drained 50,000 acres, connected headwater lakes, canalized the Kissimmee river for navigation, connected Caloosohatchee river to Lake Okeechobee, and completed a preliminary canal, eleven miles long, in the direction of Miami (Sklar et al. 2002, Light and Dineen 1994).

The second wave occurred from 1906 to the late 1920's. Empowered by the Everglades Drainage Act passed by the Florida legislature in 1907, four major canals, the West Palm Beach (42 miles), the Hillsboro canal (51 miles), the North New River canal (58 miles) and the Miami canal (85 miles) were built into the heart of the sawgrass plains (Figure 7). In addition, the Tamaimi trail, a road linking Naples to Miami, was completed in 1928, effectively dissecting the Everglades landscape into two pieces (SCT 2003).

In the third and final wave, from 1948-1973, guided by the passage of the Central and Southern Florida Project for Flood Control and Other Purposes (C&SFP) passed by the U.S. Congress in 1948, the Everglades landscape was transformed into compartmentalized and pooled regions, separated by canals and levees, and connected by water pumps (Figure 7, Sklar et al. 2002, Light and Dineen 1994). The C&SFP created six hydrologic and management units: Big Cypress, Lake Okechobee, Water Conservation Area 1, Water Conservation Area 2, Water Conservation Area 3, and Everglades National Park (Figure 6b).

As a result of these major changes in the landscape, half of the historic Everglades wetlands has been lost to agriculture and development; three major vegetation types have completely disappeared, and three-fourths of the sawgrass plains has been replaced by agricultural crops (Davis et al. 1994). Land use change has been implicated in the following large-scale changes in environmental, biological and ecological processes: (1) change in the timing, quantity and distribution of freshwater inflow into Florida Bay, (2) exposure and oxidation of soils leading to subsidence; (3) change in fire regime due to drying out of peat soils; (4) change in the structure and function of terrestrial, freshwater and marine ecosystems; (5) diminishment in abundance of wading birds due to loss of habitat; (6) increase in occurrence of invasive species (Sklar et al. 2002, Davis et al 1994).

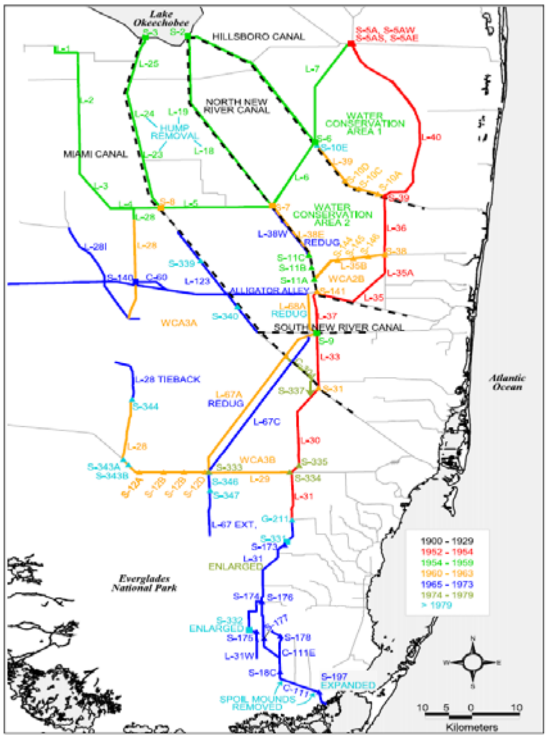


Figure 7. Sequence of construction of water control structure in the historic Everglades region from 1900 to present. From: Light and Duveen 1994. Concept from Sklar et al. 1999.

Adjacent Land Use

Adjacent Land Use: Water Conservation Areas

Three water conservation areas (WCA 1, WCA 2, and WCA 3) of more than 1,300 m^2 were created as a result of the C&SFP (Figure 6b). These areas are large impoundments or ponding areas of water that serve multiple purposes, including water storage, water recharge, flooding control, delivery of freshwater to Everglades N.P., and providing wildlife habitat. The C&SFP has led to several detrimental effects to the structure and function within these areas including: alteration of hydroperiod and flow, disruption of the ridge and slough landscape, vegetation change including loss of tree islands and wet prairie habitat, changes in the fire regime, increased non-native species, and water quality problems from efflux from the Everglades Agricultural Area just to the north (Sklar et al. 1999, Fennema et al. 1994).

Adjacent Land Use: Everglades Agricultural Area

Other than the WCAs, which are under some nominal conservation status, the Everglades Agricultural area is a 700,000 acre area of former Everglades wetlands that is devoted to agricultural production (Figure 6b). The major crop is sugar cane, but other crops include sod, beans, celery, lettuce, corn, radishes, and rice (Lodge 2005). Annual production is estimated to bring in revenues of $3 billion (Nova 2005), but the long-term status of the area is in jeopardy because agricultural practices have led to the subsidence of the peat soil level, at a rate of about one inch per year (Ingebrisen et al. 1999). At this rate, the native peat layers of between five and twelve feet have been severely compromised in this region, and not only does this put the future of agriculture in this region in doubt, this major change in topography complicates efforts to restore the natural flow regimes under the Comprehensive Everglades Restoration Plan (Ingebrisen et al. 1999).

Adjacent Land Use: Urban Areas

Everglades N.P. and Florida Bay are bordered to the east by major urbanized areas, including the Miami metro region (Figure 6b). Human populations in South Florida have exploded in the last century, where population estimates after the 1900 census were 26,000, growing to almost 6 million people after the 2000 census (Figure 8). Given that the intent of many of the water drainage and diversion practices in the region was to foster greater residential and agricultural development in the area, these projects have largely been successful. As demands on water supply and point and non-point water and air pollution continue to rise and as groundwater recharge continues to decline (due to impervious ground structures), human development around Everglades N.P. will continue to adversely impact natural resources within the park.

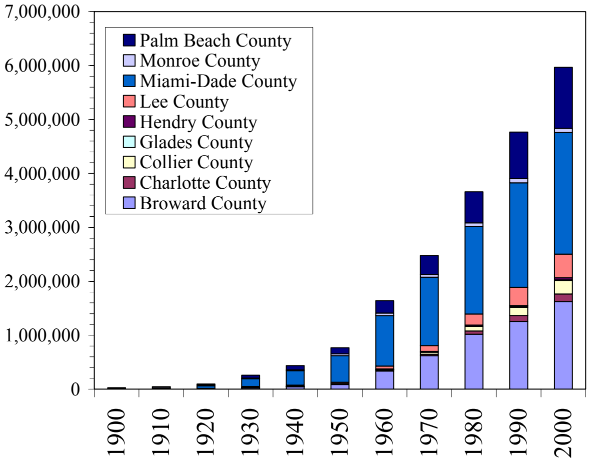


Figure 8. Population growth in south Florida from 1900-2000. Source: Updated from USCB 1995.

The eastern margin of Florida Bay is flanked by the chain of islands comprising the Florida Keys (Figure 3). As in other counties in south Florida, Monroe County, which encompasses all of the keys, is growing rapidly, from 18,000 residents in 1910 to almost 80,000 by 2000 (though most of this growth is in the Key West area south of Florida Bay). Other than the impact of a rising residential population and a burgeoning tourism industry, the natural environments of the keys are also threatened by water pollution, urban development that destroys of degrades wetland and mangrove habitats, introduced species, overfishing, damage to coral reefs caused by boat propellers, and global climate change as sea levels continue to increase (FMNH 2005).

In 1990, Congress passed the Florida Keys National Marine Sanctuary and Protection Act, creating a marine sanctuary of the same name that spans 2,800 square miles of coastal waters surrounding the Florida Keys, protecting mangrove, seagrass and coral habitats (FDEP 2004).

Park Science and Resource Management

Restoration Plans

The Everglades region, including Everglades N.P., is the site of an ambitious restoration program called the Comprehensive Everglades Restoration Program (CERP) (CERP 2005a). The overarching goal of CERP is to "get the water right" by restoring the natural hydrologic conditions of the region, while ensuring the continued economic and social development of South Florida. Ecological restoration goals include: increase the total spatial extent of natural areas; improve habitat and functional qualities; and improve native plant and animal species abundance and diversity (CERP 2005a). Economic and social goals include: increase availability of freshwater; reduce flood damages; provide recreational and navigational opportunities; and protect cultural and archeological resources and values (CERP 2005a). At an estimated cost of $7.8 billion dollars (50:50 cost share of the federal government and the state of Florida), with over 60 component projects, and a thirty year timeline to completion, CERP is largest restoration project yet conceived (Figure 9, CERP 2005a). CERP was authorized by the signing of the Water Resources Development Act of 2000.

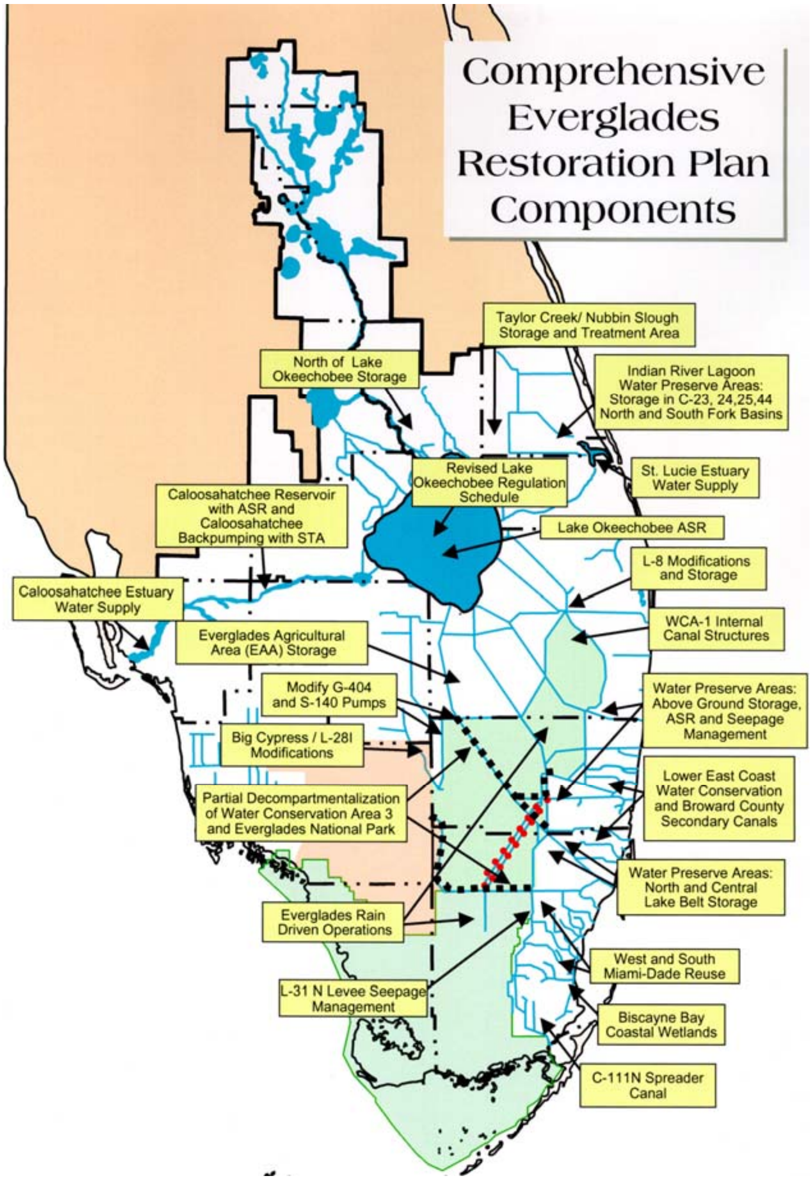


Figure 9. Map of Comprehensive Everglades Restoration Plan projects. Source: CERP 2005a.

CERP is a multi-agency effort, led at the federal level by the US Army Corps of Engineers and at the state level by the South Florida Water Management District, and supported by many scientific and policy teams. The idea of CERP is to capture water (approximately $1.8 million acre-feet by the completion of the project), the majority of which is currently diverted to the Atlantic Ocean, and redistribute it within the region to various land uses, including natural (Everglades N.P., Big Cypress National Park, Lake Okeechobee and Kissimmee River, Water Conservation Areas), agricultural and urban (RECOVER 2005). Water will be stored in more than 217,000 acres of new reservoirs and 300 aquifer storage and recovery wells. In order to restore more natural flow conditions, more than 240 miles of canals and levees will be destroyed, including most of the Miami canal in WCA 3 (Figure 7,9), and twenty miles of Tamiami trail will be raised (CERP 2005a). While the original goal stated that about 80% of these new sources of water be delivered for restoration of natural areas (CERP 2005a), the latest Interim CERP update indicates that far less (64%) may be delivered to these natural areas when the project is fully implemented (RECOVER 2005).

Specific projects within CERP that will directly impact resources within Florida Bay include the C-111 spreader canal project and the Florida Bay Feasibility Study. The purposes of the C-111 project are to: re-hydrate the Model Lands, establish sheet flow and hydropatterns that will sustain ecosystems in the Southern Glades and Model lands, provide more natural sheet flow to Florida Bay by eliminating point sources of freshwater discharges through C-111 to the estuarine systems of Manatee Bay and Barnes Sound, and maintain some level of flood protection for agricultural and urban areas in the project area (CERP 2005a). To implement the project, a 3,200-acre stormwater treatment area will be constructed, several pump station will be enlarged, the spreader canal will be extended two miles under U.S. Highway 1 and Card Sound Rd. to the Model Lands, and culverts will be installed under U.S. Highway 1 and Card Sound Road (CERP 2005a).

The goal of the Florida Bay Feasibility Study is to evaluate the connections of Florida Bay to the Everglades, the Gulf of Mexico and the Florida Keys in order to determine the modifications that are needed to successfully restore water quality and ecological conditions of the bay (CERP 2005a). Study objectives include the following: (1) determine the quantity, timing, quality and distribution of freshwater that should flow to Florida Bay and provide recommendations for any modifications of water deliveries that will result from the current Comprehensive Everglades Restoration Plan for Everglades wetlands; (2) determine the nutrient sources and loads to the study area, evaluate their impacts to reef and Bay ecosystems, and recommend restoration targets and implementation plans; (3) establish water quality and ecological performance measures; (4) evaluate the effects of restoring historical connectivity between Florida Bay and the Atlantic Ocean; and (5) evaluate management alternatives in a holistic manner employing where necessary, hydrodynamic, water quality and ecological models (CERP 2005a).

Preliminary work in the CERP has commenced. The South Florida Water Management District and the US Army Core of Engineers, the two implementation agencies, are currently acquiring lands, conducting pilot projects, and developing various other programs to manage and implement the project (CERP 2004). As of January 2004, 205,179 acres have been acquired which is an estimated 50% of the lands needed to successfully implement CERP (CERP 2004).

In addition to CERP, there are other restoration activities that will impact resources in Florida Bay. For example, the Modified Waters Delivery Project of the Combined Structural and Operational Plan will modify and add to the existing canal network system in order to enhance and restore water delivery to Everglades N.P. (USACE 2005). This project is aimed at restoring more natural flow conditions of water across the landscape, including enhanced conveyance of waters from WCA 3a to 3b, enhanced water conveyance from WCA 3b to northeast Shark River Slough in Everglades N.P., and the raising of U.S. Highway 41 (Tamiami Trail) at the L-31N canal.

Management Plans

Everglades N.P. is currently engaged in the process of developing a General Management Plan (GMP) which is scheduled for completion in summer 2006 (NPS 2003). The purpose of the GMP is to provide a vision for the next twenty years of Everglades N.P., specifically to ensure the continued persistence of the park's wilderness value and ecological integrity while providing public access to the park for the benefit and enjoyment of the people. The GMP must also ensure that NPS conforms to its responsibilities under existing legislation, including the protection of threatened and endangered species (NPS 2003).

After seeking public input, a draft GMP should be completed by fall of 2005. The six main subject materials that will be addressed in the GMP are: boating, access, facilities, partnering, ecosystem restoration, and education and interpretation (NPS 2003).

Research and Monitoring

In addition to and preceding the activities of the various scientific panels involved in the CERP, many scientific bodies have assembled to assess the condition of Everglades N.P. and Florida Bay and to guide future research, monitoring and restoration activities. Perhaps the most important of these groups is the Florida Bay and Adjacent Marine Systems Science Program (FBAMS). For the last ten years, this group of scientists from multiple federal and state agencies have conducted research, developed monitoring protocols, and developed modeling tools in support of restoration efforts in the region as well as to ensure the execution of individual agency mandates (PMC 2004).

Beginning in 1997, the program has focused Florida Bay research around five central questions:

1) How and at what rates do storms, changing freshwater flows, sea level rise, and local evaporation/precipitation influence circulation and salinity patterns within Florida Bay and the outflow from the bay to adjacent waters?
2) What is the relative importance of the influx of external nutrients and of the internal nutrient cycling in determining the nutrient budget of Florida Bay? What mechanisms control the sources and sinks of the bay's nutrients?
3) What regulates the onset, persistence and fate of planktonic algal blooms in Florida Bay?
4) What are the causes and mechanisms for the observed changes in the seagrass community of Florida Bay? What is the effect of changing salinity, light and nutrient regimes on these communities?
5) What is the relationship between environmental and habitat change and the recruitment, growth, and survivorship of animals in Florida Bay?

In 2003, FBAMS published a draft document that synthesized the latest scientific findings for these essential questions (Nuttle et al. 2003). In addition to supporting the restoration and planning activities of CERP, this document proved invaluable for this assessment report.

There are many additional scientific activities on Florida Bay and Everglades N.P., including an inventory and monitoring program and hydrological monitoring program of the National Park Service, a long-term water quality monitoring program (funded by SFWMD and conducted by the Florida International University), seagrass monitoring programs (NPS, SFWMD), and research conducted at the long term ecological research program in Everglades N.P. (funded by the National Science Foundation, conducted by researchers at Florida International University).

Marine and Estuarine Ecosystems

Water Resources

Salinity is a water parameter that integrates the influence of hydrologic processes on the structure and function of Florida Bay (Nuttle et al. 2000), including plant and animal productivity, distribution, abundance, and competitive interactions. Salinity, in turn, is governed by the natural and anthropogenic variability of hydrologic fluxes, including precipitation, evaporation, and freshwater inflows (Nuttle et al. 2003). Human perturbations, which can exacerbate high salinity, include high rates of ground water withdrawal and diversions of fresh and marine waters from natural flow paths through the Everglades to Florida Bay.

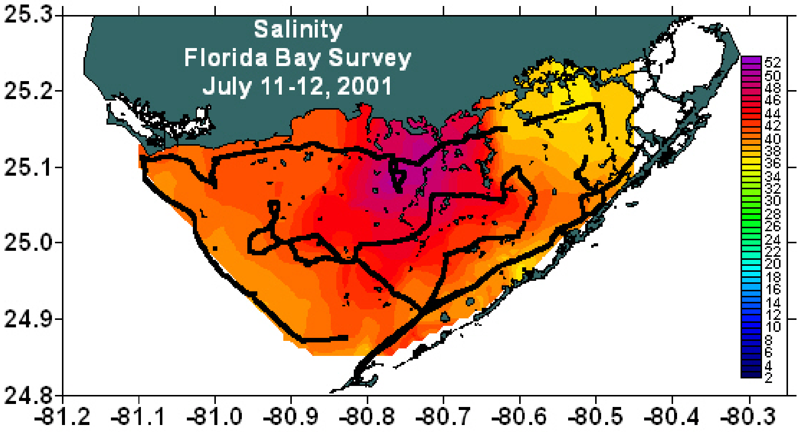


Figure 10 (a)

Salinity patterns within Florida bay vary significantly in time and space. Temporally, salinity varies at multiple scales: seasonal, interannual, decadal, and longer time frames (Nuttle et al. 2003, Nelson et al. 2002). Whereas interannual or decadal variations may be linked to climatic variations caused by the El Nino Southern Oscillation (Nuttle et al. 2003), seasonal variations are caused by relative changes in hydrologic fluxes. As the dry season ends in April, temperatures rise and evaporation rates increase, with concomitant increases in salinity (Figure 10a). If seasonal precipitation and/or water flow from the Everglades are below normal, as in 2001, salinity in the bay can begin to rise above normal oceanic levels, especially in the summer months, as water input fails to keep pace with evaporation (Figure 10b). These hypersaline events dissipate in the fall as temperatures decline, leading to low evaporation rates, and moderate salinities across the bay (Figure 10c).

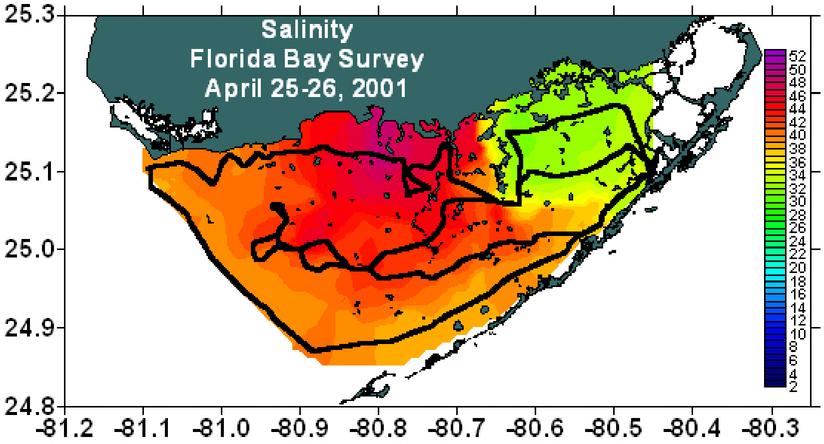


Figure 10(b)

A series of hypersaline events have been recorded in Florida Bay in the recent past. Salinity as high as 53 psu has been observed in the bay (2001, Figure 10b), while in the early 1990s, salinity of 70 psu was measured (Robblee et al. 2001). Excessive salinity can force mobile estuarine species to shift to different areas of the bay (Lorenz 1999), or to leave the bay entirely, but non-mobile species, such as rooted plants and some benthic fauna, exhibit reduced productivity or even mortality. A study of seed and seedling response has shown that both seed germination and seedling growth of *Thalassia* are negatively impacted by hypersalinity, with seedlings grown in salinity above 50 psu dying off completely within less than 10 days (Durako et al. 2002).

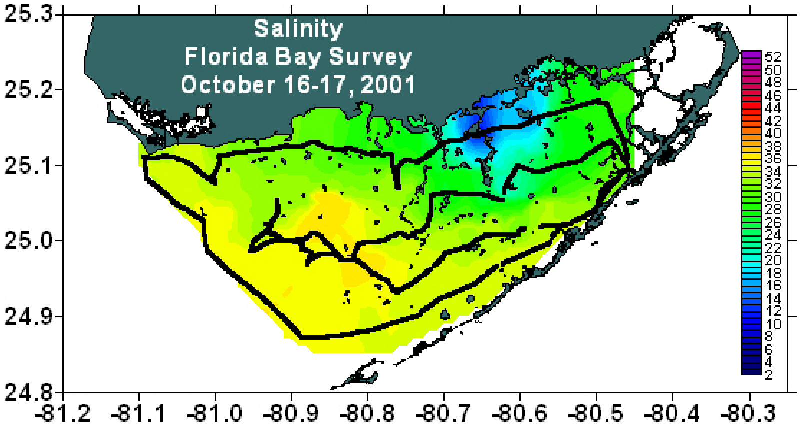


Figure 10(c)

Figure 10 (a) (b) (c). Salinity in Florida Bay in 2001, measured in (a) April, (b) July, and (c) October. Central Florida By experienced hypersalinity in spring and summer 2001. Source: NOAA 2005.

Salinity within Florida Bay also varies spatially due to non-uniform inputs of freshwater and the restricted hydrologic mixing within the bay (Boyer et al. 1997). Given that the major freshwater inputs into Florida Bay are in the northeast sector, salinity tends to increase along a northeast to southwest gradient within Florida Bay, though the central bay is prone to the highest salinities in the bay due to its hydrological isolation (Figure 10). Hypersaline events often occur in the central bay. The western bay is influenced more by currents from the Gulf of Mexico and so tends to exhibit salinities that are typical of marine waters.

Water Resources: Historic Changes in Hydrology

Even though focus of report is Florida Bay, terrestrial and marine ecosystems are linked by exchange of freshwater, and, therefore, changes is freshwater flow implicate change in Florida Bay processes. Though it is clear that reductions in freshwater water flow into Florida Bay have occurred, it is not clear by how much (Nuttle et al. 2000). Light and Duveen (1996) show that between 1980-1989, eighty percent (3.3 million acre-feet) of freshwater flushing through the Everglades was diverted to flow directly into the Atlantic Ocean, while only twenty percent is discharged into Everglades N.P. (Figure 11).

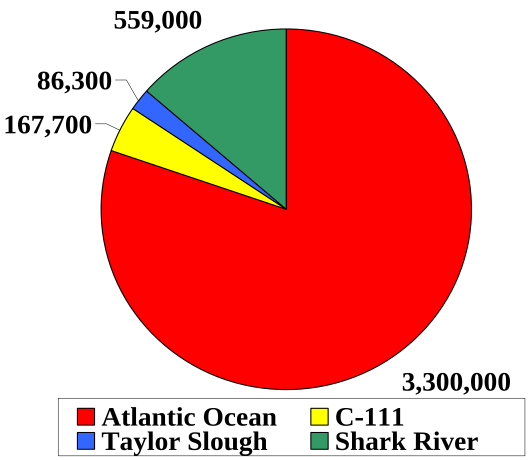


Figure 11. Annual freshwater discharge from Everglades from 1980-1989. Source: Light and Dineen 1994.

Hydrologic simulation models can be used to evaluate the effects on land use change, specifically water diversion, on hydrology. The South Florida Water Management District developed the Natural Systems Model (NSM) and the South Florida Water Management Model (SFWMM) to compare hydrologic dynamics under pre-management and post-management scenarios.

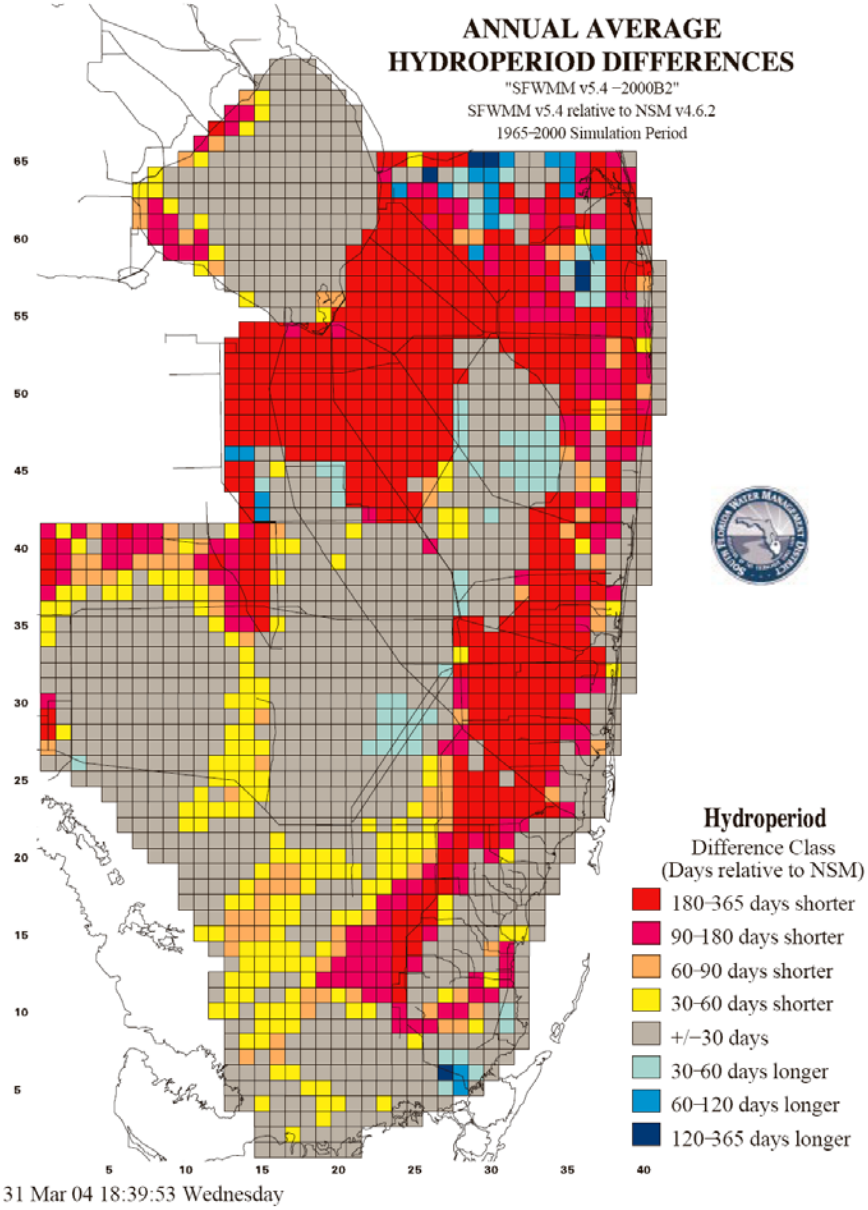
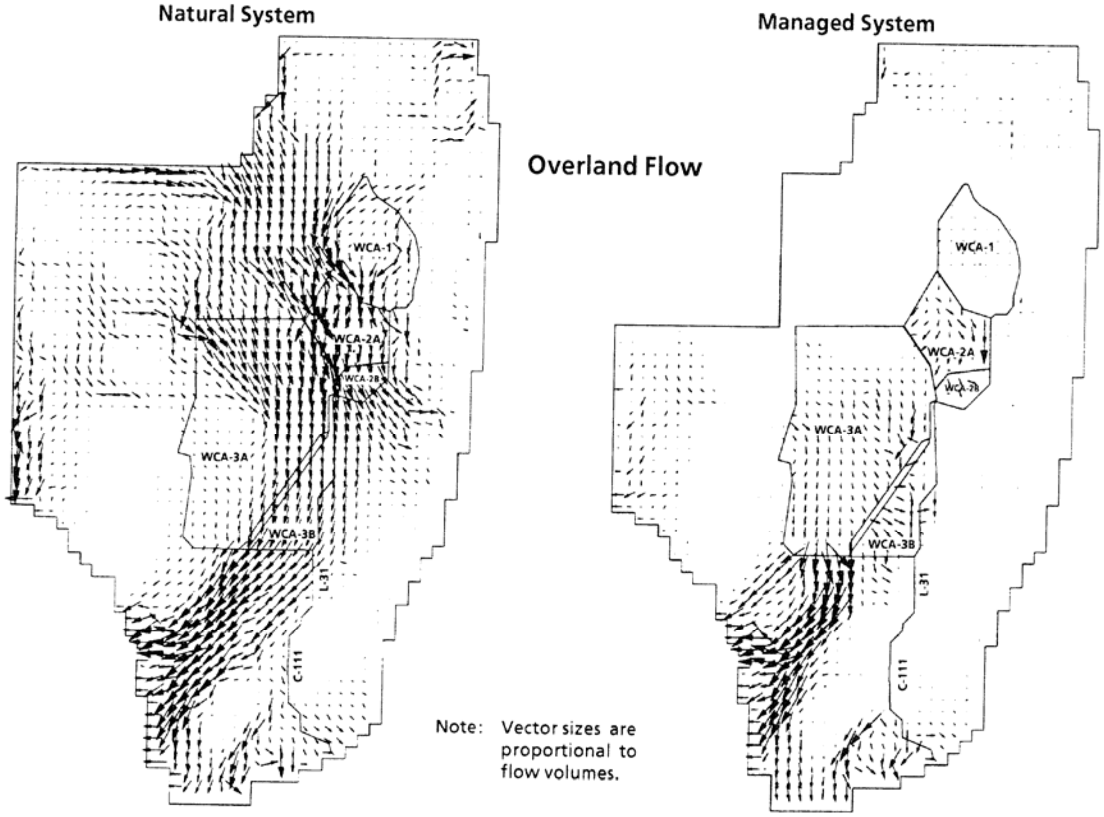


Figure 12. A simulation model comparison of hydroperiods under natural conditions (NSM v 4.6.2) and current management conditions (2000, SFWMM v.5.4 model), showing that the Everglades region is

Simulation results indicate that the spatial extent, distribution, and magnitude of water flow have been altered by the water diversion activities of the 20th century in the region (Figure 12, Fennema et al. 1994). The historic sheet flows, which once spanned a 30 mile expanse, are now largely confined to the water conservation areas and Everglades N.P. Groundwater flow patterns and recharge locations have been moved westward from the coastal ridge to the levees of the water conservation areas (Fennema et al. 1994). Simulation of hydroperiod indicates that the Everglades region was inundated with water across a greater spatial extent and with greater frequency in pre-management conditions (Figure 12, Fennema et al. 1994).

In addition, the patterns of water flow that once shaped the ridge and slough landscape have now been confined to a much smaller area than under natural conditions (Figure 13, SCT 2003, Fennema et al. 1994). Canal and levee structures have restricted natural flow patterns to the WCAs and Everglades N.P. Even within these areas, flows patterns have been altered.



Natural System

Managed System

Overland Flow

Note: Vector sizes are proportional to flow volumes.

Figure 13. A simulation model comparison of overland flow vectors under natural conditions (NSM, a) and managed conditions (SFWMM, b).

Water Resources: Historic Changes in Salinity

In the absence of long-term monitoring of salinity in Florida Bay, examinations of the paleoecological record, changes in isotope ratios within sediments and coral skeletons, and studies of banding patterns within large corals have been used to piece together the hydrologic and salinity history of Florida Bay (Nuttle et al. 2003). By examining current benthic microfaunal distribution and abundance with known environmental conditions, the microfaunal assemblages found in sediment cores of the bay can be used to infer past environmental conditions, most notably salinity. Similarly, ratios of certain minerals (Ca/Mg) within the shells of current and sediment micro-fauna record past environmental conditions, and can be used to infer salinity changes. The degree of coral inflorescence in the growth bands of large corals (e.g. similar to tree rings), a phenomena caused by humic and fluvic acids which are deposited into the bay by freshwater inputs, can be used to infer changes in freshwater inputs. All of these tools have been used to reconstruct the ecosystem history of the bay in an effort to determine natural patterns of salinity and any evidence of anthropogenic change to these patterns.

Collectively, these studies suggest that since the 1900s, salinity in the bay has increased slightly, fluctuations in salinity have risen, and the bay has become more hydrologically isolated from both freshwater and marine influences (Nuttle et al. 2003, Fourqurean and Robblee 1999). However, the causal link between declining freshwater inputs and rising salinities has not been definitive. Instead, the primary driver of historic salinity patterns in the bay is rainfall, while anthropogenic alterations to hydrology play a secondary role (Nuttle et al. 2003, Nelson et al. 2002).

Two studies have found that micro-fauna assemblages have shifted in Florida Bay, beginning in the 1900s, shifts that are indicative of a slightly increasing trend in bay salinity (Brewster-Wingard and Ishman 1999, Nelson et al. 2002). Though another study did not find this rising trend, it did indicate that the highest salinities ever to be recorded in Florida Bay may have occurred in the early 1990s (Swart et al. 1999). The Brewster-Wingard and Ishman (1999) study also found that salinity variability with the bay has increased, and this may be a significant trend because benthic microfauna may be more sensitive to changes in salinity rather than absolute salinity (Nelson et al. 2002).

Other studies suggest that human activities have altered freshwater and marine water flows into Florida Bay (Halley and Roulier 1999, Smith et al. 1989, Swart et al. 1999). In a study of coral inflorescence banding patterns, Smith et al. (1989) estimated that freshwater inputs into the bay may have been reduced by up to 59%. Though this estimate is associated with a large degree of variability, it is the only empirically based estimate of historic reductions in freshwater flows into Florida Bay.

Marine flows into Florida Bay may also have become restricted due to human activities, including the construction of the Overseas Railroad on the Florida Keys between 1906-1914 (Fourqurean and Robblee 1999). Chronologies of ^{18}O values in sediment cores from throughout the bay, which indicate historical evaporation rates, show a change beginning in about 1910 after a century of hydrologic stability (Halley and Roulier 1999). Given that these changes coincided with the railroad construction, this disturbance may have impeded circulation between Florida Bay and the Atlantic Ocean. Swart et al. (1996, 1999) found a similar signal of perturbation by examining chemical proxies in corals, noting slightly higher salinities between 1907 and 1912. Restricted water flows and hydrologic isolation could have led to hypersaline conditions in the bay.

It would be worthwhile to target more of the ecosystem history studies in different regions of Florida Bay because it may be expected that human-induced alterations to the bay would be regional or localized. None of the paleoecological microfauna studies, for example, focused on northeast Florida Bay, where recent studies have shown the largest impact in freshwater flow on salinities and fluctuations in salinity. Still, at least on a baywide basis, there have been only weak causal linkages between separate trends: slightly increasing salinity, reduced freshwater inputs, and restrictions in marine flows.

Water Resources: Recent Patterns and Trends in Salinity

Prompted by a massive seagrass die-off, comprehensive monitoring of water parameters within Florida Bay began in 1989. Salinity has continued to fluctuate temporally, declining slightly in the 1990s (Boyer et al. 1999), but exhibiting zones of hypersalinity in 2000-1 and again, in 2004-5. These hypersaline events were coincident with drought conditions. Declines in diversity and increasing dominance of salinity tolerant species in several benthic invertebrate groups have been noted beginning in the 1980s (Nuttle et al. 2003). Some studies indicate that salinity variation, rather than absolute salinity, plays a larger role in controlling benthic flora and fauna (Nelson et al. 2002).

Water Resources: Recent Patterns and Trends in Salinity–1989-1997

According to a recent study from 1989-1997, northeast Florida Bay has the lowest salinities of all three regions in the bay, but high variability in salinity (Boyer et al. 1999). Salinity averaged 28.1 psu, with a significant declining trend of –2.3/yr in the eight year period. Given the high temporal variability in salinity, decadal trends can be misleading as this time period includes high values from a drought in 1989-1990 followed by a wet years in 1994-1995 (Nuttle et al. 2003). Other studies have found that this region contains high variability in salinity (Robblee et al. 2001). Overall, declining salinities and high fluctuations in salinity can at least be partially accounted for by the mixing of freshwater from Taylor Slough and C-111 canal (Nuttle et al. 2003).

The mean salinity in central Florida Bay from 1989-1997 was 34.1 psu, showing a significant declining trend of –1.9/yr (Boyer et al. 1997). Like the northeast, salinity variability was high (Boyer et al. 1997, Robblee et al. 2001). This region is most prone to hypersalinity (Nuttle et al. 2003) most likely due the configuration of its isolated basins that act to restrict water flow, increase water residence times, and exacerbate high rates of evaporation.

Western Florida Bay exhibits relatively high salinity during this period, averaging 35.2 psu, a value that has been declining at –0.93/yr (Boyer et al. 1997). Salinity variability within this region is the lowest in Florida Bay, which is seen as evidence of the dominance of marine water flows in the western portion of Florida Bay (Nuttle et al. 2003).

Water Resources: Recent Patterns and Trends in Salinity–2000-2005

A recent drought in 2000-2001 afforded researchers with the opportunity to observe the development of hypersalinity within Florida Bay. In June, July and September, 2001, salinity in eastern, northern and central Florida Bay was mapped (Madden 2004) using a high-speed water sampling system (Madden and Day 1992). The entire eastern bay was found to be at approximately oceanic salinity in June, when the wet season normally has begun reducing salinity to lower levels. At this time, there was virtually no evidence of fresh water flowing from the Everglades, despite several heavy rains in the lower Everglades for several days prior to sampling. In lower Taylor Slough, salinity was 35 psu, at a period when fresh water is usually flowing. Waters in the central bay interior were hypersaline, ranging from 42-50 psu.

Finally, after significant rainfall in August and September, flow began to discharge from the Everglades to the northern bay, which quickly reduced salinity to about 10 psu in the northern part of the eastern bay (Figure 10c). The major input of fresh water was to the eastern bay from the Everglades Panhandle. Notably, these freshwater inputs have no effect on salinity in the central bay, where salinities remained in the 40 psu range.

From July 12-14, a second comprehensive mapping effort was undertaken. Despite very heavy rains in the area and throughout the watershed during the month of June, hypersaline conditions intensified in Florida Bay. The severity of the condition differed by region, but virtually all of the eastern bay exhibited high salinities, ranging from 36-38 psu (Figure 10b). The mangrove transition zone and creeks connecting the Everglades to the bay were also hypersaline, somewhat surprising, given the recent rains locally and throughout the Everglades watershed. In the central bay, salinity rose from 36 psu in Madeira Bay to 52 psu in Whipray Bay and Rankin Lake. The dramatic salinity difference between the eastern and central bay emphasizes the barrier to circulation between the two regions. Despite the rains beginning in June, central Florida Bay increased by 2-4 psu between June and July.

In contrast to conditions during 2001, there was little evidence of hypersalinity in 2002 and hypersalinity was light in 2003. The years 2004 and 2005 have exhibited a return to hypersaline conditions, however. Figure 14 shows mean annual salinity curves for four USGS stations in the bay averaged from data taken between 1996 and 2003, and reflecting the annual winter increase in salinity. Superimposed on the annual salinity curve is the most recent salinity data from Dec 2004-Jan 20, 2005, showing historically high levels of salinity in the bay, far above average for the period.

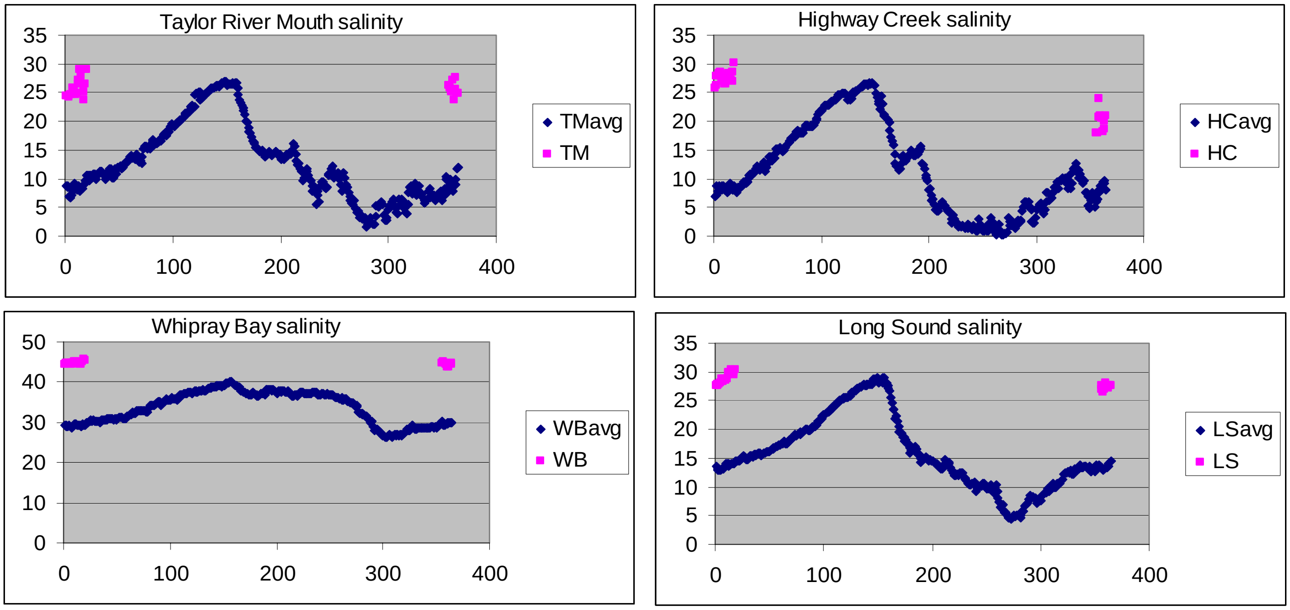


Figure 14. Salinities record averaged from 1996-2003 (blue dots), and from December 2004 – January 2005 (pink dots), for two discharge points (upper panels) and receiving water stations (lower panels) in Florida Bay. Extremely high salinity in 2004-5 is due to historically low fresh water flow into Florida Bay during the dry season Source: SFWMD 2005b.

Water Resources: Water Quality

Florida Bay is generally an oligotrophic, marine-estuarine lagoon with low nutrient levels and low phytoplankton productivity (Phlips and Badylak 1996). The trophic status is somewhat unusual for coastal water bodies receiving terrigenous inputs (Day et al. 1988), which usually are high in nutrients and are often turbid. The oligotrophic condition of Florida Bay can be largely attributed to the enormous filtering capacity of the Everglades wetland that occupies the watershed and to the carbonate chemistry of the substrate of south Florida. The wetland presents a large, biologically active area for nutrient uptake. The limestone basement rock and carbonate particulates that pervade the geology of south Florida and Florida Bay are highly attractive to P molecules and act to bind and sequester this nutrient (Koch et al. 2001).

Recent changes in the ecosystem since the early 1990s have caused concern about the possibility that nutrient enrichment may be occurring in Florida Bay (Lapointe 1989, Boyer et al. 1999). These changes mostly pertain to increased phytoplankton blooms and decreased water clarity. These patterns are discussed below in this section. Other changes in the system, such as a massive die-off of seagrasses that occurred in the late

1980s (Robblee et al. 1991) and increasing rates of coral death (Lapointe and Clark 1992) have also caused concern although these phenomena may have initiated too early (seagrass die-off) or to distantly (coral) to be directly related to increases in nutrients within Florida Bay. A more in-depth analysis of the potential relationship between seagrass die-off and autotrophic eutrophication is discussed later in this report, in the section *Seagrasses – Disturbance – Anthropogenic Eutrophication*.

The most proximate ecosystem response to terrigenous nutrient inputs and enrichment of the system is the occurrence of phytoplankton blooms (Phlips et al. 1999). As discussed below, bloom events increased in frequency and concentration in central Florida Bay after about 1991. A comprehensive monitoring program and numerous research projects (Nuttle et al. 2000) have been devoted to determining the origin of nutrient inputs to the bay and nature of the nutrient regime and of water quality in general within the bay.

Water Resources: Water Quality–Nutrient Budgets

Hydrologic flows influence the amount, timing and pattern of nutrient distribution in the bay. A nutrient budget for the bay (Rudnick et al. 1999) shows that loading from the major Everglades flow inputs to Florida Bay is extremely low in P. Annually, less than 2.6 metric tons of P as total phosphorus (TP) are exported from the eastern Everglades (C-111 Basin and Taylor Slough) to the bay and less than 12 tons are imported by the Shark Slough wetlands from the western bay and Gulf of Mexico. This compares to 38 tons input to the bay from the atmosphere and 42 tons from Florida Keys wastewater. By far the largest source of P to the bay, 500 tons, is advected from the ocean through the bay's boundary with the Gulf of Mexico. It is generally thought that P from land drainage along the west coast of Florida may be transported by long shore currents southward, where they are carried into the bay (Lapointe et al. 1994, Rudnick et al. 1999). Outputs of TP from the bay via advection and dispersion sum to 270 tons, about half of the estimated inputs, indicating significant retention and burial of P within the bay.

The budget for N shows the Everglades to be a more important source of total nitrogen (TN) than of TP to the bay. Input of TN from the Everglades is 250 metric tons annually from the eastern Everglades. Atmospheric N deposition is about three times the eastern Everglades input at over 700 tons, while the Keys' wastewater input to Florida Bay is estimated to be 170 tons. Advection of N from the Gulf of Mexico to the bay totals 13,000 metric tons. This total input of about 14,000 tons is balanced by total outputs of 17,000 tons, indicating that, unlike TP, TN is net exported from the bay to the adjacent marine systems (Rudnick et al. 1999). This export term represents the source of potential concern for the reef tract where TN may cause eutrophication (Lapointe et al. 2004). However, the small amount of TN supplied by Everglades inputs (250 tons) relative to both total input (14,000 tons) and net export (3,000 tons) terms suggests that an Everglades source that is one or two orders of magnitude is not likely be a significant factor in overall bay trophic status (Rudnick et al 1999).

Two terms in the nutrient budgets that are not well understood or quantified are groundwater inputs of N and P, and the net effect of several biological N transformations (Rudnick et al. 1999). Nitrogen fixation, denitrification and nitrification processes affect the speciation of N forms among nitrate, nitrite, ammonium and elemental di-nitrogen (N_2) (Owens and Cornwell 2001). This speciation also determines the relative balance of dissolved, solid phase or gaseous forms of N. Elemental N and ammonia are both gases and could represent a significant export from the estuarine system (Day et al. 1988). Continuing research on the magnitude of these biological processes and pools and the net of these reactions will enable the refinement of our estimates of N budgets for the bay. Because P does not have a gaseous form, the P that enters the bay is either cycled within the system, sequestered and buried, or exported from the system in dissolved or particulate form. Phosphorus is the limiting nutrient to autotrophic production across much of the bay (Fourqurean et al. 1993), and it is anticipated that increasing managed water flows to the bay for hydrological restoration will not result in harmful nutrient enrichment or eutrophication, as the P load from Everglades sources are small (Rudnick et al. 1999, Nuttle et al. 2003). The "controllable" inputs of both TN and TP via land drainage from the Everglades watershed are small compared to natural inputs, locally uncontrollable inputs from the atmosphere and inputs from the Florida Keys.

Water Resources: Water Quality–Spatial Trends

Gradients in P and N distribution and their availability across the bay create a shift in the potential limiting nutrient for autotrophic production (Rudnick et al. 1999). Phosphorus concentrations in the water column are highest in the western bay and lowest in the eastern bay, owing to the major source of external P from the Gulf of Mexico (Fourqurean and Robblee 1999, Nuttle et al. 2003, Rudnick et al. 1999). N concentrations are highest in the northeast and lowest in the west due to inputs of N from freshwater inflows, notably from the C111 Basin and Taylor River, and an evaporation driven concentration of nutrients in the hydrologically

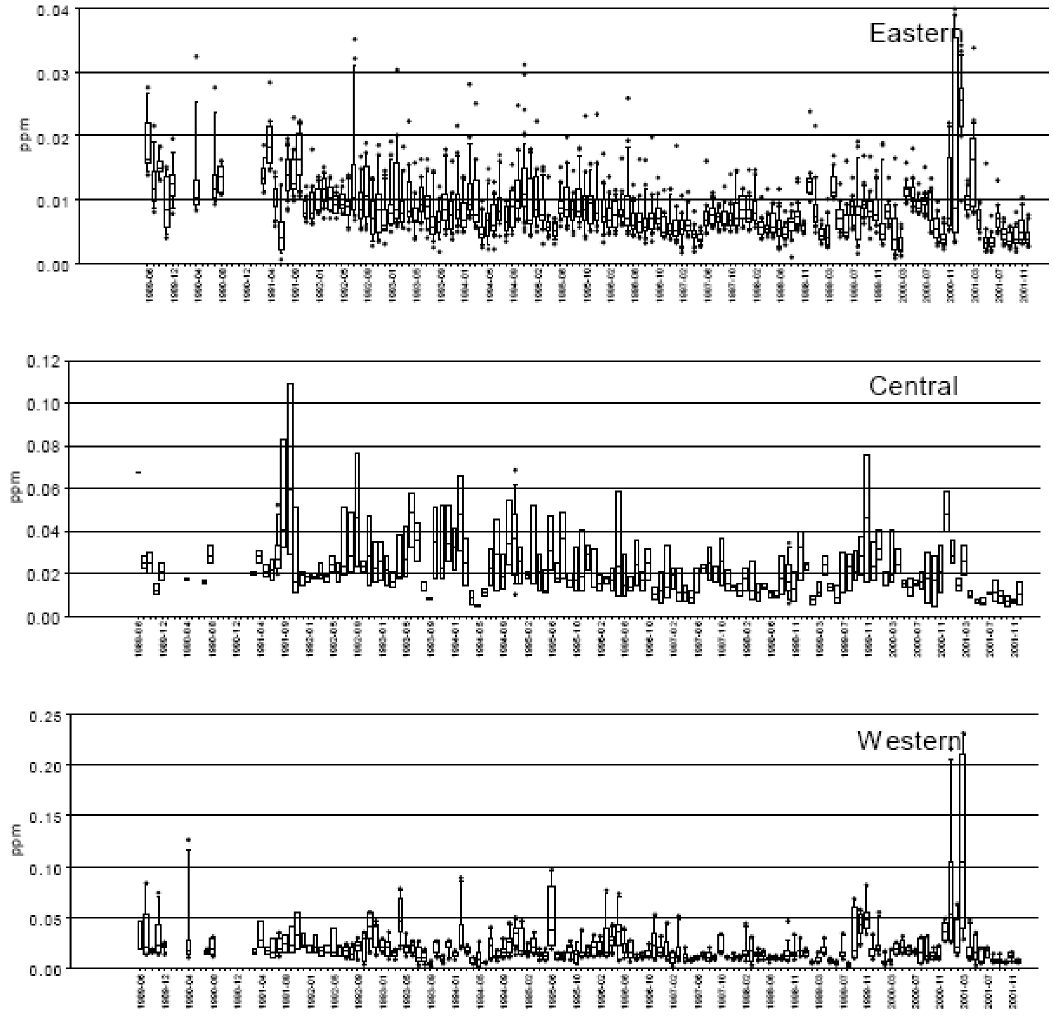


Figure 15. Total phosphorus (TP) in Florida Bay from 1989-2001 for the eastern, central and western regions of the bay in micro-grams. Source: Nuttle et al. 2003.

isolated central portions of the bay (Fourqurean and Robblee 1999, Nuttle et al. 2003, Rudnick et al. 1999, Fourqurean et al. 1993).

Water Resources: Water Quality–Temporal Trends

A recent analysis of water quality in Florida Bay beginning in the late 1980s indicates a decreasing trend in both TN and TP, following an initial increase in the early part of the decade (Figures 15, 16). There is some variability in phosphate in recent years possibly associated with storms and hurricanes, but Boyer et al. (1999) report that there have been significant (though small) declines in inorganic P for all three regions of

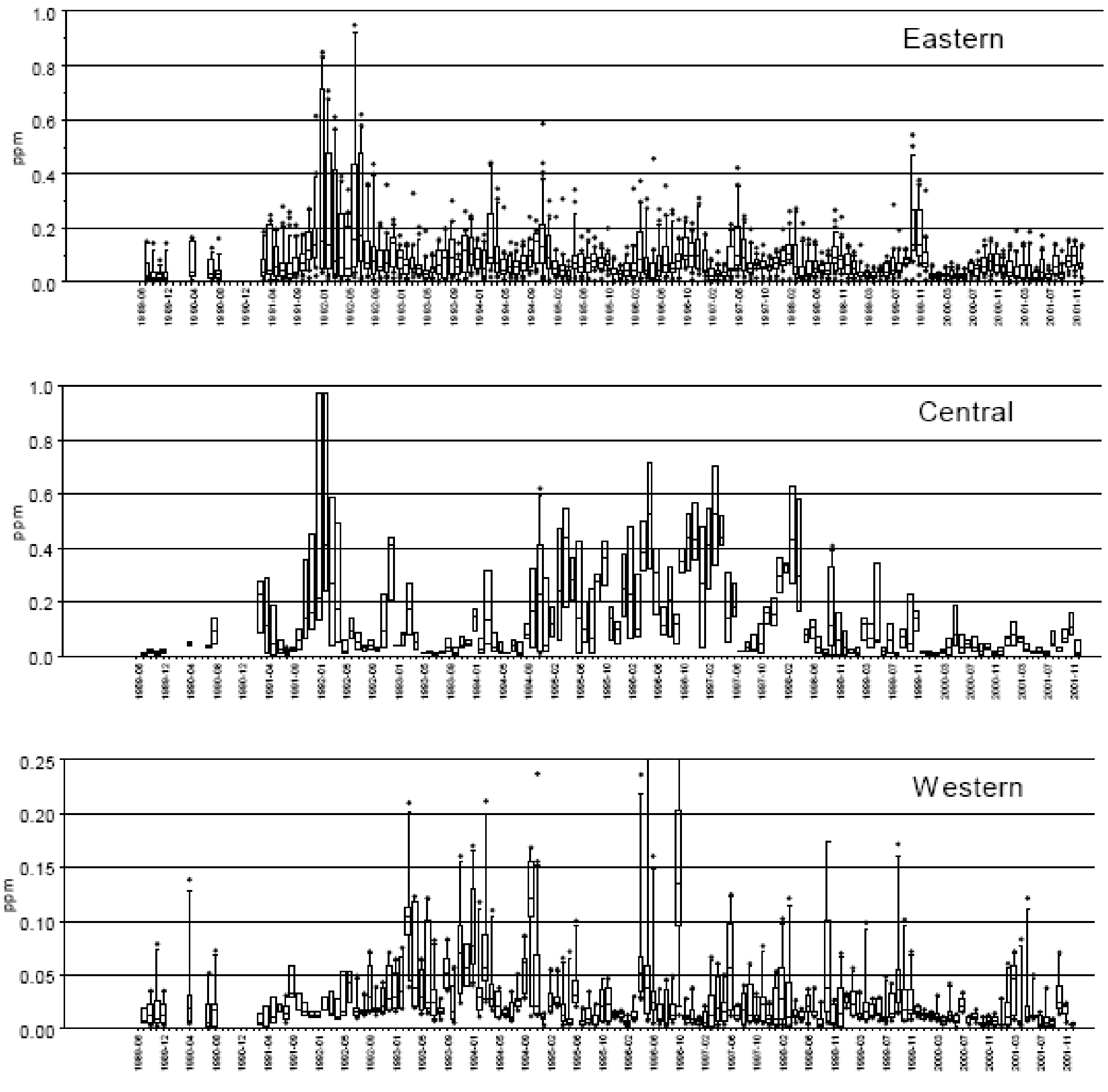


Figure 16. Dissolved inorganic nitrogen (DIN) in Florida Bay from 1989-2003 for the eastern, central and western regions of the bay in micro-grams. Source: Nuttle et al. 2001.

the bay. P levels in all three regions are so low as to be at or below the limit of kinetic uptake thresholds for many species (Boyer et al. 1999).

Dissolved inorganic nitrogen (DIN) also increased initially in the early 1990s, in all three sectors of the bay, being most pronounced in the central bay, where the moving average concentration increased from below 1 to 10 micro-moles. The eastern bay also showed a strong spike beginning in 1991-2, although at about a 50% lower level than in the central bay. The trend was muted and lagged by many months (beginning in 1993) in the western bay, where concentrations remained below 1 microgram throughout the episode. In the central bay, following the initial peak and decline, concentrations of DIN declined briefly, then again increased during the mid decade before finally declining in 1999 to levels observed prior to the 1990. Temporal and spatial patterns in nutrient availability can influence patterns of standing stock and production of benthic macrophytes and microalgae (Fourqurean et al. 1993, Zieman et al. 1989) as well as phytoplankton blooms within the bay. Fourqurean found that the gradient in Thalassia production and biomass, increasing from east to west, followed the P availability gradient, suggesting that seagrass production is limited by P (Fourqurean et al. 1992). Indeed, a nutrient content analysis of leaves in the eastern bay shows low P content, high C:P and N:P ratios, whereas leaves in western bay have higher P content, and lower C:P and N:P ratios (Fourqurean et al. 1992). Leaf C:N ratios showed little variation indicating that perhaps is N saturated, at least in terms of seagrass production.

Water Resources: Water Quality–Phytoplankton– Northeast Florida Bay

Chlorophyll a (herein referred to as Chl-*a*) is routinely monitored as an indicator of the concentration of living phytoplankton in the water column (Boyer et al. 1999). Despite increases in flows from the eastern Everglades subcatchment during the past 10 years of increased rainfall and increased managed flows, the northeast region of Florida Bay exhibits lower concentrations of suspended phytoplankton than the other two regions by far (Phlips et al. 1999, Boyer et al. 1999). Phlips et al. (1999) found that Chl-*a* concentrations in the northeast were consistently < 2 mg m-3 from 1993-1997, and Boyer et al. (1999) estimated a mean Chl-*a* concentration of 0.85 mg m-3, with a significantly declining trend of –0.74 mg m-3 yr-1. It is likely that phytoplankton blooms are relatively insignificant in this area of the bay because of a severe limitation in P availability. Freshwater inputs of P are estimated to be extremely small (Rudnick et al. 1999), leading to N:P ratios in the range of 100-120, significantly higher than the 16:1 Redfield ratio threshold for P limitation (Fourqurean et al. 1993). Bioassay studies show that growth is stimulated by the addition of P to incubated water samples from northeast bay further supporting P as a limiting nutrient for phytoplankton (Tomas et al. 1999).

Water Resources: Water Quality–Phytoplankton in Central Florida Bay

Throughout the 1990s, the central bay began to exhibit phytoplankton blooms and increased turbidity beginning in 1991 (Figure 17). Chl-*a* concentrations regularly exceeded 10 mg m-3, up to 20 mg m-3 (Phlips et al. 1999), and averaged 2.34 mg m-3 (Boyer et al. 1999). These blooms were seasonal, with peaks in Chl-*a* during the late summer and fall. The composition of the blooms was dominated by cyanobacteria, especially *Synechococcus elongates cf* (Nuttle et al. 2003, Phlips et al. 1999). The blooms were located in

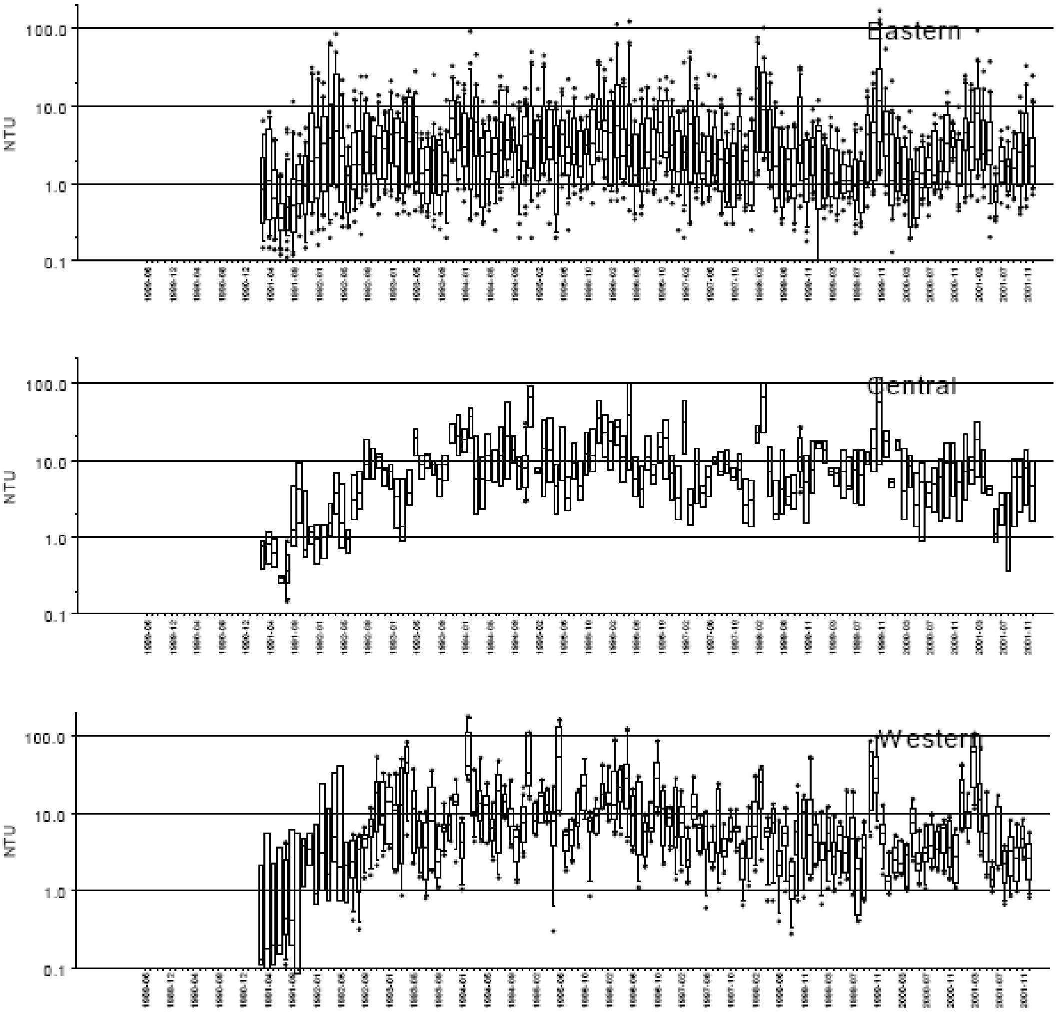


Figure 17. Turbidity in Florida Bay from 1991-2001 for the eastern, central and western regions of the bay in NTU. Source: Nuttle et al. 2003.

the north central portion of the bay, but were seasonally shifted southward, likely by circulation patterns driven by cold fronts in the fall and winter months (Phlips et al. 1999). N:P ratios are high, 140:1, and Chl-*a* concentrations are correlated with P concentrations, indicating P limitation of phytoplankton in this area (Fourqurean et al. 1993), but a bioassay study showed growth responses to addition of both P and N, indicating perhaps shifting P and N limitation of growth (Tomas et al. 1999). These blooms have declined since the late 1990s although remain higher than in either the eastern or western bays (Boyer et al. 1999).

Further study is required fully understand the dynamics of these blooms, but several hypotheses have been generated to explain the source of the bloom phenomenon. The phenomenon may be the result of a state shift in the bay due to the catastrophic die-off of seagrasses in the 1980s. By reducing the amount of seagrass biomass in the benthos, nutrient uptake capacity has been lost, perhaps allowing other autotrophs, such as phytoplankton increased access to nutrients. A simulation modeling study of Florida Bay (Madden et al. 2003) has shown that the degree of loss of seagrass observed in the bay could result in a commensurate increase in nutrients available to phytoplankton for growth causing bloom formation. Additional factors could include increased heterotrophic volume in the system due to a more turbid water column resulting in increased release of nutrients from sediment-derived decomposition products. The most prevalent species of inorganic N in the central bay (by an order of magnitude) is ammonium (Boyer et al. 1999), the concentration of which can exceed 20 micro-molar. This regenerated form of N is a decomposition product and could contribute additional internally recycled N as a result of increased organic decomposition. Boyer et al. (1999) reported a statistical independence of Chl-*a* concentrations and TN loading from the Everglades, supporting the concept that Everglades loading is not responsible for observed blooms.

More research needs to be conducted on the exchange rates of the central bay with neighboring basins, the residence time of water and organic materials, nutrient cycling rates, including the role of N-fixation in N dynamics, and the influence of N versus P in limiting phytoplankton production before a definitive cause can be identified (Nuttle et al. 2003). It is important to note that these hypotheses are not mutually exclusive and that a combination of factors may lead to phytoplankton blooms in the area.

Water Resources: Water Quality–Phytoplankton in Western Florida Bay

Phytoplankton concentrations in the western bay are moderate, where Chl-*a* concentrations ranged from 2-9 mg m-3 from 1993-1997 (Phlips et al. 1999) and averaged 1.93 mg m-3 from 1989-1997 (Boyer et al. 1997). Blooms there exhibited seasonal variation, with peaks in late summer to winter. Unlike in the central bay, the composition of blooms in the western bay was dominated by centric (*Phizosolenia* sp.) and pennate diatoms (*Cocconeis*, *Navicula*, and *Surirella* sp.) (Phlips and Badylak 1996). Steidlinger et al. (2001) also found the presence of cyanobacteria.

Phytoplankton in the western bay are stimulated by experimental additions of both N and Si indicating that these two nutrients limit production (Tomas et al. 1999). Blooms occurring in the western bay originate in the southwest Florida shelf, outside of Florida Bay, where they are supported by the discharge of freshwater from Shark River Slough containing high amounts of N and Si, or by the eastern limb of the Loop Current carrying Mississippi River water. These Gulf blooms are then often advected into western Florida Bay (Nuttle et al. 2003). Given this scenario, it is at least possible that increased freshwater inputs from Everglades restoration projects could lead to enhanced N loading into the shelf from Shark River Slough and potentially exacerbate diatom blooms in western Florida Bay.

Water Resources: Water Quality–Turbidity

In addition to higher phytoplankton concentrations (Boyer et al. 1999), areas of the bay became more turbid beginning in the early 1990s (Figure 17). Stumpf et al. (1999) reported data from 13 sites across the bay that showed the water column to be clear (mean 0.51 Kd) in 1987. By 1994-5, turbidity of 2.82 Kd were being reported at the same sites, conditions that can cause potential light limitation of benthic vegetation in even shallow waters. Boyer et al. (1999) also found increased levels of turbidity from 1991 to 1996, where turbidity increased by factors of 2, 4, and 20 in the eastern, western, and central bay regions, respectively. Some of the measured increase in turbidity is attributable to the phytoplankton blooms, although much is from increased bottom sediment resuspension due to the loss of belowground biomass. As phytoplankton blooms have declined in recent years, the turbidity decline has not been commensurate, indicating that a component of the turbidity is due to increased resuspension of sediments.

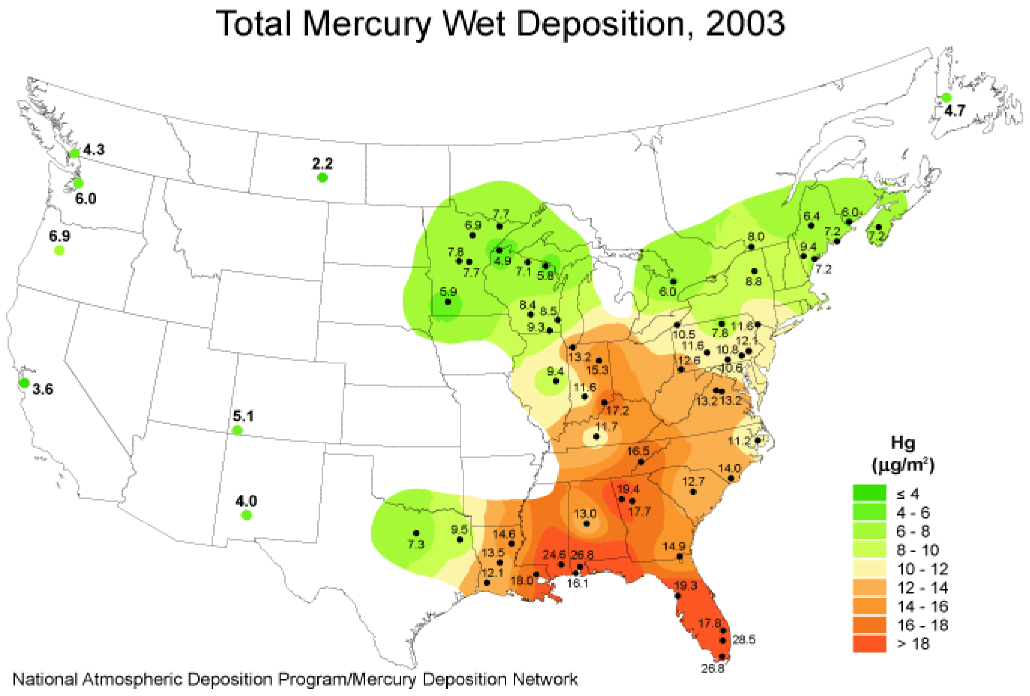
Water Resources: Water Quality–Conclusion

Florida Bay is historically an oligotrophic system, and nutrient enrichment would be expected to cause major shifts in ecosystem function and community structure, particularly in the coral reefs in and beyond Florida Bay. One of the major objectives of CERP is the restoration of fresh water flows and the removal of barriers to circulation in Florida Bay that will bring a more natural flow and salinity regime to Florida Bay (CERP 2005a), but an unintended consequences of these actions may be enhanced nutrient loading into the bay. At the present time, a nutrient budget analysis indicates that N and P inputs from freshwater sources are much smaller when compared to inputs from marine waters advected from the Gulf of Mexico (Rudnick et al. 1999). Other analyses indicate that P availability limits production in the bay (Fourqurean et al. 1993), and is not likely to increase with the increased freshwater flows. Further, simulation modeling indicates that the loading of N to the bay would increase by 1% or less if CERP restoration plans move forward (Perry 2005). All of this evidence suggests that the potential risk of enhanced nutrient loading under current restoration plans to increase freshwater flows to the bay is relatively low, though continued research in other sources of nutrients that are not well understood (e.g. groundwater) and nutrient exchange and transformations within the bay are needed.

Air Quality

Air Quality: Mercury

The death on an endangered Florida panther in 1989, later found to have elevated concentrations of mercury in the liver, raised national awareness that Everglades N.P. may experience significant levels of mercury contamination (Loftus and Bass 1992, Jordon 1990). Though the exact cause of death of this panther was not


Total Mercury Wet Deposition, 2003

National Atmospheric Deposition Program/Mercury Deposition Network

Figure 18. Patterns of mercury wet deposition in 2003 in the lower 48 U.S. States, showing south Florida as a mercury deposition hotspot. Source: (NADP/MDP 2003).

conclusive, many freshwater and marine fish and wading birds have also shown elevated concentrations of mercury, indicating that mercury contamination and bioaccumulation has occurred both within Everglades N.P. and the surrounding WCAs. In fact, the Florida Department of Health has issued many consumption advisories for fisheries in the regions because of elevated mercury concentrations of tissue, including snook, red drum, great barracuda, spotted sea trout, crevalle jack, gray snapper, and tripletail (FDH 2004)

As recently as 2003, south Florida experienced some of the highest wet deposition rates of mercury in the continental US (Figure 18, NADP/MDP 2003, Atkeson and Axelrad 2004). In fact, the major pathway of mercury into the central and southern Everglades is through air deposition, accounting for more than 90% of the inputs (Atkeson and Axelrad 2004). About one-third of this is in the form of dry deposition, the remainder wet (Atkeson and Axelrad 2004). Under the prevailing wind currents, south Florida is somewhat isolated from the rest of the continental US, and sources of mercury have been determined to be local. These anthropogenic sources include incineration of municipal and medical waste, activities of the sugar industry, and fossil fuel power plant emissions, especially coal produced electricity (Axelrad et al. 2005, Atkeson and Axelrad 2004, Loftus and Bass 1992).

Local mercury emission rates and deposition rates in the central and southern Everglades peaked in the 1980s (~ 3,000 kg/yr peak emission rate from local sources), after substantial increases in the 20th century from background levels, and subsequently have declined due to the implementation of strict regulations (Axelrad et al. 2005, Atkeson and Axelrad 2004). Mercury concentrations in freshwater fish and wading birds have followed this trend of decreasing levels in the 1990s to present (Axelrad et al. 2005), but notably, concentrations within marine fish species in Florida Bay have not declined (Evans et al. 2003, Strom and Graves 2001), suggesting that these resources are still in jeopardy from mercury contamination.

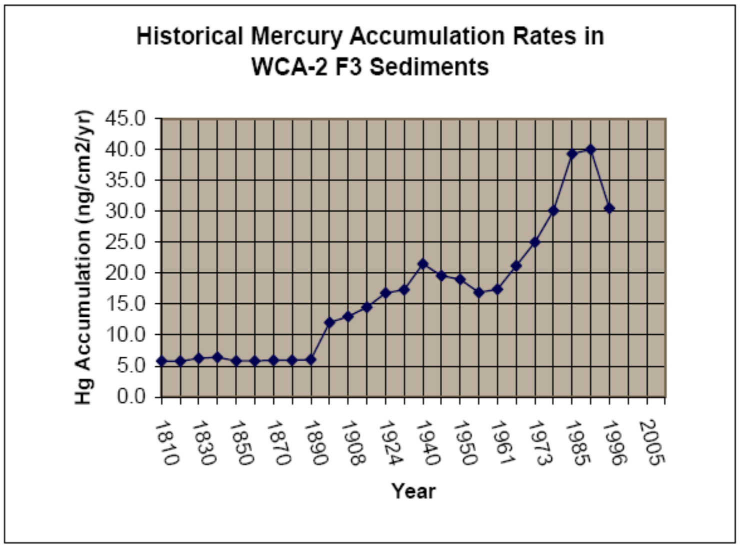


Figure 19. Long mercury accumulation rates records in Everglades soils in WCA-2, showing rise in mercury contamination in the last 100 years. After: Atkeson and Axelrad 2004. Source: Rood et al. 1995.

Air Quality: Mercury–Historic Trends

Monitoring of emission and deposition rates were not initiated until the late 1980s. However one study of sediments cores taken from WCAs and Everglades N.P. indicates that mercury concentrations within soils have increased eight-fold from the late 1800s to the mid 1990s (Figure 19, Rood et al. 1995). Though the fluctuations in emissions and deposition rates are not clear, it is unequivocal that increasing anthropogenic inputs of mercury to the region from the late 1880s to the 1980s have occurred (Husar and Husar 2002, Atkeson and Axelrad 2004).

Air Quality: Mercury–Historical Effect on Biota

A recent historical analysis shows that bioaccumulation in south Florida wading species has indeed occurred (Figure 20, Atkeson and Axelrad 2004). Though variance is high, mercury concentrations within the feathers of great egret and great blue heron birds are ten-fold higher in the 1990s compared to previous times.

Air Quality: Mercury–Current Trends

Under stricter regulations, local mercury emission rates have been reduced from high rates of 2-3,000 kg/yr in the late 1980s and early 1990s to much lower rates 250-600 kg/yr by 2000 (Figure 21, Axelrad et al. 2005). This close to 90% reduction has been due in large part to diminished emissions from municipal and medical waste incineration.

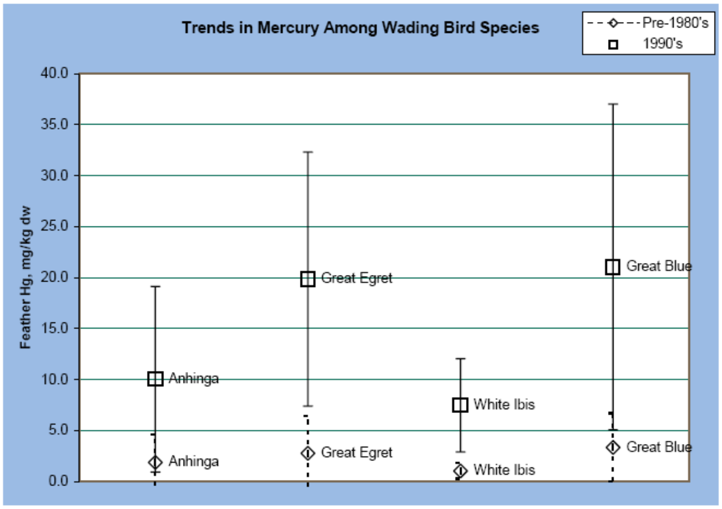


Figure 20. Historical comparison of mercury content in wading bird feathers, showing increased mercury content in recent years. Samples collected from North American museums for specimens tagged with a south Florida origin. Source: Atkeson and Axelrad 2004.

Air Quality: Mercury–Transformations in Aquatic Everglades

Mercury is a naturally occurring heavy metal found in the earth, air and water. Inputs of mercury into the central and southern Everglades are inorganic in form, but undergo several transformations in the aquatic environment. The form of mercury that is most toxic to animals is known as methymercury, which is a byproduct of the activities of sulfate reducing bacteria in aquatic sediments. Methymercury strongly bioaccumulates in the food chain once it is formed. Methymercury production is influenced by concentrations of mercury, sulfur (sulfate and sulfide), and dissolved organic carbon, and can form with a period of hours to days once deposited from the atmosphere (Axelrad et al. 2005).

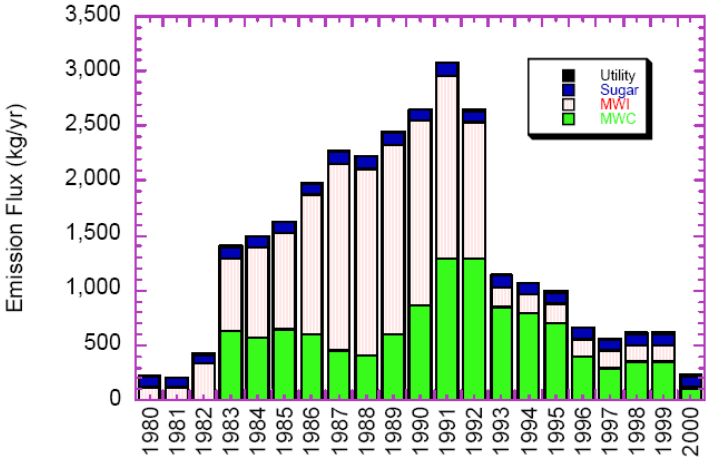


Figure 21. Annual atmospheric mercury emissions in south Florida from 1980-2000, estimated as a function of major sources: utility (power generation facilities), sugar (sugar refineries), MWI (municipal waste incinerators), and MWC (municipal waste combustors). After: Axelrad et al. 2005. Source: RMB Consulting & Research, Inc. 2002).

Though deposition rates of mercury are believed to be relatively uniform across the region, methymercury production is not. Complex biogeochemical interactions cause methymercury to vary both spatially and temporally. The central Everglades (WCA 3) have higher production rates than the northern Everglades (WCA 1, 2). A "hotspot" in WCA 3 had been noted for several years but has subsided substantially since 1993 (Strober et al. 2001, Axelrad et al. 2005). Current sulfate concentrations in this area of WCA 3 are well below the optimal levels needed for methymercury production, and it is possible that recent hydrologic changes related to the implementation of several Stormwater Treatment Areas may have diverted the sulfate inflow away from WCA 3 and to Everglades N.P. (Axelrad et al. 2005). In fact, now that mercury emissions have been reduced, current research is focused on the role of sulfate in determining methymercury production in the central and southern Everglades (Atkeson and Axelrad 2004).

Near Florida Bay, high rates of methymercury production in the mangrove transition zones occur in late summer months (Rumbold et al. 2003). Taylor slough production rates are higher than the neighboring C-111 basin, and in both basins there is evidence of *in situ* production of methymercury (Rumbold et al. 2003).

Air Quality: Mercury–Current Effect on Biota

Recent monitoring of bioaccumulation (1994-2003) indicates that mercury tissue concentrations are declining in freshwater fish and wading birds in the central Everglades (WCAs), but are stabilizing or increasing within Everglades N.P. and Florida Bay. Mercury in largemouth bass in the central Everglades (WCAs), for example, has declined by 60-80% in this period, though current levels often exceed the 0.3 mg/kg human-health fish tissue criterion proposed by the US EPA (Axelrad et al. 2005). Mercury concentrations within great egrets have followed a similar trend. Along with reduced mercury emission and deposition rates, sulfate concentrations within this region are now low enough to limit meythlmercury production.

Concern is high for patterns of mercury bioaccumulation in marshes and estuaries that lead into eastern Florida Bay, and eastern Florida Bay itself. Unlike the trend for central Everglades noted above, mercury concentrations in largemouth bass within Everglades N.P. have not declined recently, and in the case of bass in North Prong creek, have actually increased from 1999-2003 (Axelrad et al. 2005). Mercury concentrations in bass of this creek are about 1.75 mg/kg, greatly surpassing the recommended EPA health standard. This trend is also notable for other gamefish in this region, where many species continue to have mercury concentrations that exceed the Florida health advisory level (1.5 ug/g wet weight, Evans et al. 2003). From 2000-2003, 50% of the spotted seatrout, 28% of Jack crevelle, and 9% of the gray snapper that were sampled exceeded the Florida state health advisory level (Evans et al. 2003).

Air Quality: Mercury–Conclusion

Bioaccumulation and methymercury patterns suggest that the mercury hotspot occurring in the central Everglades during the 1990s most likely has shifted from WCA 3 to southern Everglades N.P. and Florida Bay (Axelrad et al. 2005). Though the causal mechanisms explaining these patterns are complex, it is believed that differential patterns of sulfate concentrations might explain this shift. Sulfate loading from the Everglades Agricultural Area canals directly into Everglades N.P. are 100 times greater than historical levels, and have recently shifted so that more EAA inflows are received directly by Everglades N.P. (Axelrad et al. 2005). While mercury concentrations in fish and birds north of Everglades N.P. have declined recently, they have remained steady or increased in many fish species within Everglades N.P. (Axelrad et al. 2005). Several marine species within Florida Bay demonstrate mercury concentrations that exceed state and federal health standards (Evans et al. 2003), and for which there are current consumption advisories that are specific for these fish in Florida Bay and other coastal waters (FDH 2004). Accordingly, more research needs to be conducted in order to determine the role of these agricultural water inflows into Everglades N.P. and the influence of sulfate concentrations on methymercury production. This information is needed given that restoration plans call for even greater inputs of freshwater into Florida Bay (Axelrad et al. 2005, Evans et al. 2003).

Representative Estuarine and Marine Ecosystems

Seagrasses

The seagrass community is keystone community within the Florida Bay ecosystem, playing a role in nearly every trophic and physico-chemical function of the bay's ecology (Stumpf et al. 1999, Matheson et al. 1999, Fourqurean et al. 2002, Ley and McIvor 2002, Lorenz et al. 2002). Dominated by turtle grass (*Thalassia testudinum)*, seagrasses stabilize sediment and sequester nutrients, processes that help reduce epiphyte and phytoplankton blooms (Zieman 1982). The sediment-binding capacity of the rhizomatous seagrasses serves to ameliorate turbid resuspension events, reduce scouring, promote a clear water column, and contribute to high rates of primary and secondary productivity (Zieman 1982).

Seagrasses provide refuge, spawning areas, and a food source for numerous important fish and invertebrate species (Zieman 1982, Sogard et al. 1989, McIvor et al. 1994, Thayer et al. 1999). Fish densities tend to be greater in the seagrass beds than outside the beds (Weinstein et al. 1977). In Rookery Bay to the west of Florida Bay, Yokel (1975) reported trawl catches in seagrass beds that were 3.5 times greater than those in other habitat types. Pink shrimp develop in the bay, favoring seagrass habitat (Sheriden 1992), before moving to the Dry Tortugas, where they support one of the major commercial shrimp fisheries in the Gulf of Mexico (Ehrhardt and Legault 1999). Spiny lobsters use the bay as juveniles before moving across the Keys to take residence in the reef from the Dry Tortugas to Pacific Reef near Miami (Davis and Dodrill 1989). The highest growth rates of juvenile spiny lobsters in the world were measured in Florida Bay, considered an optimum habitat for this species (Davis and Dodrill 1989).

The seagrass community is vulnerable to perturbation, undergoing a catastrophic die-off in 1987 (Carlson et al. 1990a, Robblee et al. 1991), and exhibiting signs of impairment on a wider time and space scale (Hall et al. 1999, Durako et al. 2002). This series of events has caused alarm about large scale restructuring of the community, degradation of habitat quality, declines in biodiversity and in the number of fish caught, and possible irreversible damage to the ecology of Florida Bay (Durako et al. 2002). We will summarize the key findings from the many investigations that ensued following the seagrass die-off event in the "Seagrasses – Disturbances" section of this report.

Seagrasses: Community Extent

The Florida Bay seagrass community is large in spatial extent, its range extending over virtually the entire bay (Zieman 1982). At any given time, 80% of the 1800 km^2 of the bay that is within Everglades National Park boundaries is vegetated by seagrasses (Iverson and Bittaker1983). Including areas outside the park, the seagrass community covers an estimated 5,500 km^2 of the greater Florida Keys area, and is "one of the largest seagrass resources on earth" (Zieman 1982).

Figure 22. Photo of seagrass in Florida Bay, Everglades National Park. Source: SFWMD 2005b.

Seagrasses: Species Distribution and Abundance

Much of Florida Bay is carpeted by submerged aquatic vegetation (SAV, Figure 22), rooted vascular plant beds that are dominated by turtle grass (*Thalassia testudinum*) throughout, often mixed with shoal grass (*Halodule wrightii*). Manatee grass (*Syringodium filiforme)* is found in generally deeper waters closer to the Gulf of Mexico and widgeon grass *(Ruppia maritime)* occurs in the fringes of the mangrove transition zone where water is fresher. In parts of the far western bay, the paddle grass (*Halophila decipiens)* is found and the related species *Halophila englemanii* occurs (Zieman 1982).

Spotted sea trout (*Cynoscion nebulosus*), gray snapper (*Lutjanus griseus*), red drum (*Sciaenops ocellatus*), snook (*Centropomus undecimalis*), mullet (*Mugil cephalus*), bay anchovy (*Anchoa mitchelli*), and a variety of forage fishes are permanently or transiently resident in Florida Bay (Sogard et al. 1989). Pink shrimp (*Pennaeus duorarum*) and the spiny lobster (*Panulirus argus*) use Florida Bay as a primary nursery ground (Butler et al. 1995).

Seagrasses: Ecosystem Processes/Function

Zieman et al. (1989) notes that a change in the frequency of hurricanes and large storms that left Florida Bay untouched by natural disturbance for over 20 years likely resulted in a succession away from mixed communities and pioneering species and toward a community dominated by the climax species *Thalassia testudinum*. Stable meteorological conditions, reinforced by a human-induced stabilization of the salinity regime away from one of estuarine variability and toward constant high salinity, acted to favor slower-growing and higher biomass species like *Thalassia* while suppressing fast-growing, fresh-tolerant pioneer species like *Halodule* and *Ruppia* in the open bay. This allowed a stable climax of *Thalassia* to form, out-competing other species and reducing the occurrence of mixed seagrass beds through many parts of the bay (Zieman 1982) and may have set up a situation that is less sustainable than a mixed bed. An immediate consequence of this monoculture was the reduction in some fish species like mullet and sea trout in the northeast and north central bay (Zieman et al.1989), as the areas silted in and formerly sparsely vegetation mud bottom habitat became crowded with vegetation. A long-term consequence was the set-up of the climax community for a catastrophic collapse.

Seagrasses: Ecosystem Processes/Function–Production

Because of the shallow depths (ranging from 1-3 m) and relatively clear waters, light penetrates to the bottom of Florida Bay, resulting in dominance of primary producers by benthic flora, especially turtle grass, *Thalassia testudinum* (Zieman et al. 1989, Stumpf et al. 1999). Very fine-grained carbonate sediments can be frequently and easily suspended by winds, reducing light penetration (D'Sa et al. 2002). These same carbonate sediments bind inorganic P very effectively such that the available P is often scavenged by particulate carbonates resulting in extremely low concentrations of dissolved inorganic P in the water column that limit plant production in the bay (Jones and Amador 1992, Rudnick et al. 1999, Koch et al. 2001). Fresh water entering from the Everglades is also very low in P (Amador et al. 1992, Rudnick et al. 1999, Brand 2002), and the bay is generally considered to be oligotrophic (Rudnick et al. 1999, Fourqurean and Robblee 1999).

Seagrass production increases from the Northeast to the western portions of the bay near the Gulf of Mexico (Zieman et al. 1989). A regression equation between standing crop and productivity is significant ($p < 0.001$) and explains 92% of the variation, indicating that standing crop or biomass can be used as a surrogate to estimate seagrass productivity. Standing crops were 21 $g\ m^{-2}$, 47 $g\ m^{-2}$ and 89 $g\ m^{-2}$, and productivity rates were 0.28 $g\ m^{-2}d^{-1}$, 0.85 $g\ m^{-2}d^{-1}$, and 2.29 $g\ m^{-2}d^{-1}$, in eastern, central and western regions, respectively (note: the eastern bay standing crop and productivity estimates presented here are cumulative of the northeast, Atlantic, and interior sectors portrayed in Zieman et al. 1989).

Nutrient and sediment depth are important factors determining Thalassia dynamics. Phosphorus is the apparent limiting nutrient to Thalassia production (Fourqurean et al. 1992) and photosynthesis is generally light saturated (Madden 2001). Higher P concentrations and a thicker sediment layer in the western bay are responsible for generally higher seagrass biomass and productivity (Zieman et al 1989). The sediment thickness to bedrock varies from a few cm in the eastern bay to nearly a meter in the western bay, on average (Wanless et al. 1994). It is hypothesized that greater sediment volume affords the extensive below ground biomass of *Thalassia* the opportunity to expand and better access the nutrients, particularly P, required for growth that exist in the sediment pore waters (Fourqurean et al. 1992).

Measurements of nutrients in the sediments and water column of the bay indicated that the increasing east to west gradient of P in porewaters and in the water column are reflected in nutrient ratios in the tissues of *Thalassia* (Fourqurean et al. 1992). A clear relationship between the depth of the sediment to bedrock and *Thalassia* standing crop and short shoot density implicates sediment geochemistry as a primary factor determining the biomass and production of the *Thalassia* community (Zieman et al. 1989, Fourqurean et al. 1992). Sediment depth alone accounted for about 30% of both density and biomass (Zieman et al. 1989). The volume of porewater P available is likely the primary determinant of *Thalassia* productivity and biomass. The long term spatial gradient in the ratio of TN:TP observed in the water column (Boyer et al. 1999) is also reflected in bay sediments. This pattern in turn is expressed in a gradient of tissue nutrient ratios in Thalassia as reported by Fourqurean et al. (1992) measuring 60-70 in the eastern bay, declining to 30-40 in the western bay. As detailed earlier, the gradient in seagrass production follows the P availability gradient, increasing from east to west, suggesting that production is limited by P in the sediment porewater pools (Fourqurean et al. 1992). Leaf C:N ratios showed little variation indicating that perhaps is N saturated, at least in terms of seagrass production.

The ecological service performed by the seagrass community as a pathway of nutrient removal from the water column and long-term sequestration can have a significant impact on the nutrient cycling and nutrient budget of the bay (Madden 1999). Loss of this uptake capacity is expected to enable the transfer of free nutrients into other nutrient pathways, such as to epiphytes and phytoplankton. Additionally, with die-off of *Thalassia* the decay products of the considerable biomass and remineralization of associated nutrients with seagrass would create an opportunity for the abrupt injection of mobilized, newly available nutrients into the water column.

Seagrasses: Vegetation Patterns–1950s to early 1980s

Seagrasses: Early Accounts–1950s-1970s

In order to understand community change, it is necessary to know the state of the benthic communities of the bay throughout recent history. Early knowledge of seagrass ecology in Florida Bay, however, is based on anecdotal evidence from local fishers and guides. In the first half of the 20th century, it was generally known only by word of mouth that seagrasses dominated the Florida Bay bottom (Zieman 1982, Zieman 1999). Knowledge of the fishes that used the seagrass beds as habitat was scarcely better, supported only by industrial catch statistics. Information on Florida Bay, its habitat and living resources was not reported or compiled in any scientific way. Tabb and Roessler (1989) reported that "Since biologists really knew nothing at all of the park fish fauna, virtually the entire period between 1957 and 1967 was devoted to listing of species, and recording basic environmental parameters such as climate, tides, salinity, temperature, bottom characteristics and community distribution."

The first modern recorded scientific descriptions of Florida Bay and its ecology were initiated by Tabb and Manning (1961), who produced a checklist of flora and fauna of the waters of Florida Bay in the late 1950s and a subsequent report on the ecology of northern Florida Bay (Tabb et al. 1962). That report qualitatively described the extensive seagrass community as mixed stands of *Thalassia* and *Halodule*, or dense monotypic stands of *Halodule*. The western bay seagrass community was described as being impacted by turbidity and salinity fluctuations that caused stunting of the sparse plants. In the interior bay, the authors even referred to small seagrass die-off events in the late 1950s that occurred when hypersalinity of 45-59 psu persisted for several months. Tabb and Manning's 1961 report on hydrographic data from the inshore bays and estuaries of Everglades National Park covered the period 1957-1959 and represented the first spatially distributed synoptic salinity measurements recorded in the interior of Florida Bay. Although the distribution of the water quality study sites was not comprehensive, mostly centered in the western bay, they are informative: the sites nearest the interior central bay showed salinity reaching to near 50 psu on several occasions during the two year study, the first quantitative evidence that the bay was becoming hypersaline.

Changes to bay hydrology initiated by landscape alterations for water management upstream that began in the early 20th century (Light and Dineen 1994) and culminated in the 1960s had led to shifts in species composition and community structure in Florida Bay. These may ultimately have resulted in an unstable equilibrium in the seagrass community in the late 1980s. Though there was scant documented evidence, Zieman (1982) and Zieman et al. (1999) pieced together information about Florida Bay seagrass community from interviews with local watermen. The picture that emerges of the bay during the 1970s is of an estuary that was starved for fresh water for over a decade and had become a clear quiescent lagoon that was prone to episodes of hypersalinity. The ecological response of the seagrass community to changes in the landscape and seascape was a change in community structure from that which Tabb and Manning had described in the 1950s as mixed beds of *Thalassia* and *Halodule* towards development of monospecific stands of *Thalassia*, with a general loss of *Halodule* and macroalgae species. *Thalassia* thrived in the high light, high salinity environment presented by the altered hydrological regime, at the expense of *Halodule*, *Ruppia* and macroalgal species. Dense, overgrown monocultures of *Thalassia* developed, particularly in the western bay and interior bay, fueled by P, the limiting nutrient of the seagrasses (Fourqurean et al. 1993), that was imported through the western bay boundary from the Gulf of Mexico (Boyer and Jones 2002). These dense stands persisted into the late mid 1980s (Zieman et al. 1989).

Seagrasses: Recent Accounts–Early 1980s

A 1984 vegetation survey showed that the four geographic sectors of the bay were characterized by distinctly different species compositions and vegetation densities (Zieman et al. 1989). The eastern bay was mainly sparse and patchy *Thalassia*, with a standing crop of 0-10 g dw m^{-2}, mixed with *Halodule*, which was more prominent in disturbed areas. It was noted that on the tops of banks, *Thalassia* standing crop of as much as 30 g dw m^{-2} was common and that often the leaves were heavily epiphytized. *Laurencia, Batophora, Acetabularia,* and *Penicillus* species of macroalgae were found in specialized habitats such as the lee sides of banks and bedrock outcroppings. In the central bay, dense monospecific stands of *Thalassia* were found with little evidence of *Halodule*. Standing crop varied little from 50-60 g dw m^{-2}. In the western bay, the densest stands of *Thalassia* were measured in extensive, lush beds of 75-125 g dw m^{-2} in the north and central western bay and up to 400 g dw m^{-2} on the bank tops in the southwestern bay. *Thalassia* was interspersed with *Halodule* and *Syringodium* in these western beds. *Syringodium* dominated the deeper parts of the western bay at 3 m depth.

Interestingly, in what Zieman terms the "mainland" community, corresponding to the lagoons and ponds in the mangrove transition sector, the submerged plant community exhibited the greatest heterogeneity. This area was most influenced by terrestrial drainage and still retained some of the variable salinity characteristics of the estuary that much of the bay had previously exhibited. The vegetation in the transition zone was described as dense stands of *Thalassia*, and equally dense and monotypic stands of *Halodule* of up to 90 g dw m^{-2}. *Syringodium* grew in the deeper waters of the western part of the sector, while *Ruppia* grew well in areas experiencing fresh water inflow, generally associated with the red mangrove stands in the eastern part of the sector. Much of the vegetation in this sector was epiphytized (Zieman et al. 1989).

An independent survey by Hall et al. (1999) also showed that in 1984, the densest *Thalassia* beds were in the south central and western bay with densities as high as 2200 short shoots (ss) m^{-2}. Of 107 stations sampled throughout the bay, the highest shoot densities, which were generally greater than 1000 ss m^{-2}, were at sites extending from the southern bay toward the west, corresponding exactly to the region where die-off took firm hold three years later. In terms of standing crop, these sites also reflected the highest biomass levels in the bay, all greater than 50 g dw m^{-2} and more than one third of the 44 sites in this area supporting greater than 125 g dw m^{-2}.

Halodule was distributed throughout the bay in a 1984 survey, with highest short shoot densities (> 1500 ss m^{-2}) reported in the western bay, and lowest (0-1 ss m^{-2}) in the south bay. The eastern bay had intermediate densities of *Halodule* in the range of 0-500 ss m^{-2}. (Hall et al. 1999). Standing stocks ranged as high as 12 g m^{-2} in the mangrove transition zone, and 5 g m^{-2} in the central and western bays. *Syringodium* was only significantly present in the western bay at deeper water depths (up to 2400 ss m^{-2}), although it was found in sparse numbers in all parts of the bay. Standing stock of this species was virtually zero throughout the bay and about 12 g m^{-2} in the western bay.

Seagrasses: Vegetation Change–Die-off

Seagrasses: Die-off–1987-1989

Die-off in the *Thalassia* community was first noted in the fall of 1987 in the northern interior bay in Rankin Lake and the southeastern bay at Cross Bank (Robblee et al. 1991). It quickly followed in Johnson Key Basin in the western bay and Rabbit Key Basin in the south central bay. The common factor across die-off sites was the rapid, near-total death of dense stands of *Thalassia*. Less dense stands were not affected, meaning that the northeastern bay was virtually untouched (Figure 23). The central and western bay were the sites of the most severe die-off, while in the eastern bay, die-off was practically non-existent. Die-off affected huge areas covering thousands of hectares and could also be patchy, with abrupt, sharp boundaries between living and non-living plants. Die-off continued through 1989 and severely impacted SAV community

Figure 23. Spatial patterns of seagrass die-off, showing areas where major die-off occurred. Source: modified by Robblee et al. 1991.

initially killing 4000 hectares of *Thalassia* beds outright, and thinning the population in 23,000 additional hectares during the primary event (Robblee et al. 1991, Hall et al. 1999). Within the major die-off areas, 95% of plants were killed and mortality eventually consumed 30% of the entire *Thalassia* community in Florida Bay (Hall et al. 1999, Durako et al 2002). *Halodule* and *Syringodium* were not apparently involved in the initial die-off event (Zieman et al. 1999, Hall et al. 1999).

Seagrasses: Post Die-off Status of Turtle Grass

The initial die-off event continued for two years through 1989, then subsided, replaced by a general, slower decline of the seagrass community involving *Halodule* and *Syringodium*. By 1995, the seagrass community stabilized and *Thalassia* ceased the widespread, rapid loss observed in 1987-89 although die-off still continued in some parts of the bay (Zieman et al. 1999). Although the rate of *Thalassia* die-off slowed considerably, and the spatial distribution of *Thalassia* changed little after 1994, density continued to decline more slowly through the 1990s. By 1995, the specific productivity rate of the remaining *Thalassia* was restored to its pre die-off level of (Zieman et al. 1999), although there was much less standing stock, and so total system productivity remained severely reduced. As of 2005, the total standing stock has not yet returned to historic levels and smaller, isolated die-off events have continued to occur in the southern and western bay areas, albeit not on the scale of the original die-off (FHAP unpublished data).

Hall et al. (1999) measured the status of the seagrass community in 1994, five years after the 1987-89 die-off event, and compared it to measurements of density and distribution at the same sites sampled in 1984, prior to the die-off. Mean short shoot density of all species had declined, in most cases significantly, between the two samplings. Many areas had completely lost all cover of *Thalassia* and others were much less dense than in 1984. Fully seven years after the die-off had begun, nearly as many sites (40 stations) had significantly lower *Thalassia* density (33-100% loss) as had not changed in density (42 sites), while only 18 sites had increased. Notably, short shoot density had declined by more than 50% (from 500 to 200 ss m^{-2}) in transition zone sites, and by almost 50% at western (800 to 550 ss m^{-2}) and interior (700 to 400 ss m^{-2}) sites. At stations with high levels of biomass in 1984, the decline in standing crop averaged 52%. Stations with a medium level of biomass declined in standing crop by 12% on average (Hall et al. 1999).

A longterm dataset (Zieman et al. 1999) shows that there were high density and low density beds throughout Florida Bay when primary die-off occurred. The high density beds were the ones that collapsed while the low density beds were unaffected. In ensuing years, the dense beds re-grew to a density somewhat lower than prior to the collapse, and the cycle repeated. A cyclic oscillation is observed where dense beds in optimum conditions (mostly the western and central bay) supported rapid regrowth of *Thalassia*, followed by an abrupt decline to a lower level. Over time, the trend for the high density beds has converged almost exactly on the biomass level of the low density beds, which may be the equilibrium density that is sustainable by the bay.

Seagrasses: Post Die-off Status of Shoal Grass

Although *Halodule* did not experience an abrupt die-off along with *Thalassia*, it declined markedly following the die-off, between 1989 and 1994 (Durako et al. 2002). These plants are hypothesized to have been adversely affected by the secondary effects of the initial die-off- notably by an increase in water column turbidity that began in 1991. *Halodule* was reported to have declined by 92% across the bay between 1984-94 (Hall et al. 1999). Although Syringodium also did not experience primary die-off, by 1994 its short shoot density had declined by 66-100% at almost every site in the western bay (Hall et al. 1999). This may be evidence of the keystone role that *Thalassia* plays in the survival of other seagrasses, as the light climate of the bay seems to have been altered by the loss of *Thalassia*'s sediment stabilization properties. The implication therefore, is that *Halodule* can survive in a mixed bed with *Thalassia*, while a monotypic stand of *Halodule* may not be able to sustain itself over a widespread area in large parts of Florida Bay, due to resuspension and light attenuation. Thus a mixed seagrass bed may be the most robust, and if a stress causes a reduction in either of the two components of the community, the entire community can be impacted negatively.

Seagrasses: Post-Die-off Patterns–1995-2003

The Fish Habitat Assessment Program (FHAP) was initiated to track changes in the vegetative habitat important to fish in Florida Bay since 1995. The vegetation maps produced by FHAP in 1995, 2001 and again in 2003 allow the comparison of distribution and bottom cover (measured by the Braun-Blanquet technique) between the base year (1995) and later years. Comparison of the two maps side by side for *Thalassia* and *Halodule* is instructive. The *Thalassia* distribution during the six years ending in 2001 shows extensive areas of red and scarlet, indicating continuing losses of seagrass density over significant areas of the western and central bays. Smaller areas of red, pink, and ochre in the central and even eastern bay indicate lower levels of seagrass loss. The eastern bay was largely unchanged and there are two notable areas of *Thalassia* gain in the central and western bay by 2001.

Following the trends through the 2003 sampling in the right panel, the western bay continues to show red and pink, indicating continued declines from 1995, although some of the areas indicate modest recovery of the beds (FHAP unpublished data). The central bay and eastern bay show more areas of green, indicating recovery in some of these areas by repopulation by *Thalassia*. In contrast to *Thalassia*, since 1995, *Halodule* has increased its presence significantly, especially in the western bay, where *Thalassia* was most affected by die-off. The indication is that after a period of early losses due to secondary die-off and difficulty in establishing itself in the early 1990s, as reported by Hall et al. (1999), *Halodule* has since begun rapidly colonizing western bay areas vacated by *Thalassia*.

Seagrasses: Disturbance

The 1987 die-off event precipitated a dramatic change in state of the Florida Bay community, including severe cascading effects that are still reverberating today as entire groups of living resources were impaired (Durako et al. 2002). Within a year of the die-off, a 100% mortality of sponges ensued and several genera of sponge (*Speciospongia, Ircinia, Hippiospongia, Spinosella, and Vergangia*) disappeared from the bay (Fourqurean and Robblee 1999). Butler et al. (1995) hypothesized that the loss of sponges resulted in a decreased nursery function of the estuary for early life stages of the spiny lobster, *Panulirus argus*. Landings of pink shrimp at Tortugas Banks plunged in 1988 to their lowest levels in decades (Robblee et al. 1991). Game fish landings declined as seagrass community composition shifted.

Following the die-off event, a number of investigations to understand the scope and the origin of the problem and its ramifications through the ecosystem were initiated. The bay water quality network was initiated in 1989. Numerous surveys of seagrass distribution and ecology were conducted and experiments were initiated. The first published observations on the die-off phenomenon appeared in short reports (Carlson et al. 1990a, Carlson et al. 1990b) describing the primary areas of die-off and some experiments to determine causality. These hypotheses, the evidence gathered to support or refute them, and the emerging scientific consensus on the cause of the seagrass die-off are discussed below.

Seagrasses: Disturbance–Wasting Disease

One of the first hypotheses for die-off initiation that was examined was wasting disease associated with the parasitic slime-mold, *Labryrinthula*. It was well-known that an infection by a *Labyrinthula* species had ravaged the temperate seagrass *Zostera marina* throughout the world (den Hartog 1987, Short et al. 1988, Rasmussen 1977). A related *Labyrinthula* parasite that infects *Thalassia* was detected at the Florida Bay die-off sites and the symptoms of infection were noted in lesions on the leaves (Conroy et al. 1999, Blakesley et al. 1998). *Labyrinthula* infection is likely transmitted by leaf to leaf contact, which would argue for increased mortality in denser beds, so the spatial pattern of die-off fit the hypothesis of a density-dependent model. Further, the parasite is only active and infectious at higher salinities (above 20 psu), which also conforms to the observation of die-off in high salinity areas. Although much of the spatial pattern of the die-off fits the paradigm of parasitic disease transmission, it is also consistent with other hypotheses, such as an overbuilt canopy limited by self-shading. Further analysis has led to the belief that the die-off probably was not primarily precipitated by a parasitic disease infection (Fourqurean et al. 1999). Many areas of die-off did not exhibit infection, while other areas that were infected did not exhibit die-off (Hall et al. 1999). Laboratory experiments (Carlson et al. 1990b) showed that the disease develops and transfers too slowly to account for the rapid spread of die-off. It is widely thought now that the parasite is not responsible for the initiation of primary die-off. The scientific consensus suggests, however, that the disease may exploit plants that have already been compromised by other stressors and primary die-off itself and intensify the effects of die-off (Durako et al. 2002).

Seagrasses: Disturbance–Anthropogenic Eutrophication

The hypothesis that nutrient enriched water from land drainage caused the die-off is another conceptual model that is also applicable to many impaired seagrass systems around the world, such as Chesapeake Bay (Kemp et al. 1983), Cockburn Sound in Western Australia (Cambridge et al. 1986, Cambridge and McComb 1984) and Waquoit Bay, MA (Valiela et al. 1991). Increasing anthropogenic loads of nutrients can cause phytoplankton growth, replacement by macroalgae, and epiphyte overgrowth of seagrass plants. This model implicates eutrophication of the water column and attenuation of light over seagrass beds as the primary agent causing reduction of photosynthesis and productivity. In Florida Bay, nutrient enrichment has been promoted as a possible model to explain the 1987 die-off event and subsequent seagrass losses (Brand 2002, Lapointe et al. 2002, Lapointe et al. 2004). While potentially contributory to seagrass community deterioration, nutrient enrichment is not a likely a primary cause of die-off (Fourqurean and Robblee 1999, Hall et al. 1999, Nuttle et al. 2003).

Because wide scale water quality monitoring was not in place prior to and during the seagrass die-off in 1987, the nutrient status of the bay during the time leading up to the die-off is speculative but certain limited inferences can be drawn from data that do exist. Inspection of the water quality record shows that with few exceptions, the long-term trend in Florida Bay nutrients is of low concentrations and decreasing levels for all P, and particularly N species (Boyer et al. 1999). In addition, these nutrient forms have decreased in all regions of the bay and are at their lowest levels in a decade (Boyer et al. 1999, Boyer and Jones 2002, Rudnick et al. 1999).

Available evidence indicates no severe nutrient enrichment problem and no apparent phytoplankton blooms or epiphyte overgrowth at the time of the die-off (Fourqurean and Robblee 1999). Although Hurricanes Andrew in 1992 and Irene in 1999 and tropical storms in 2002 caused transient increases in the water column nutrient profile, the overall pattern has been a downward trend in terrestrial nutrient loads and an improvement in water quality throughout the 1990s (Rudnick et al. 1999, Childers et al. 2002). The

Everglades delivers water to Florida Bay that is extremely low in P and only moderately loaded with N (Boyer et al. 1999). Before and during the period of the initial die-off, there were no reports or observations of phytoplankton blooms in the bay (Boyer et al. 1999), nor were there observations of excessively epiphytized seagrass blades in die-off areas (Zieman et al. 1999) . Given the hypersaline conditions in 1986-1987 it is likely that there was little to no appreciable fresh water flowing into the Florida Bay system in that period and that virtually no terrestrial enrichment of Florida Bay could have occurred, as there would have been no hydrologic mechanism to deliver a nutrient load to the central, western and southern bay.

The areas most severely affected by die-off in the central, western and southern parts of the bay, are well-removed from the terrigenous flows from the mainland via the northeastern channels that deliver water from Taylor Slough and the Everglades Panhandle (Robblee et al. 1991). These terrestrial flows discharge into the eastern and northeastern bay, and waters there remain largely oligotrophic (Rudnick et al. 1997). Seagrass in these areas have experienced little to no die-off (Hall et al. 1999). The central bay, although close to the mainland, is hydrologically isolated from terrestrial inputs (Nuttle et al. 2000); fresh water nutrient loads can be assumed to have not impacted the central bay area when die-off occurred there. In fact in the mid and late 1980s, the central bay had become the most hypersaline part of the entire bay at the time it was a location of primary die-off. Due to plant decomposition, it is expected that increased nutrient concentrations would be more likely be a *consequence* of die-off, rather than a cause, and this may explain the increase in nutrient concentrations noted in 1991-1992 (Durako et al. 2002). Sediments and porewater nutrients would also be more apt to register as increases in the water column due to reduced biomass of seagrass available to stabilize or sequester the substrate. Therefore, it is not surprising that nutrient levels measured after 1989 are higher than prior to die-off.

Seagrasses: Disturbance–Multiple Stressors

A third conceptual model for the die-off invokes multiple stressors simultaneously impacting the community (Robblee et al. 1991, Carlson et al. 1994, Durako et al. 2002). The elements of the conceptual model included a suite of stressors that converged due to both natural (drought) and human (hydrologic alteration) factors contributing to reduced fresh water flow. These stressors include high salinity, elevated temperature, toxic accumulation of hydrogen sulfide in sediments and wasting disease (Durako et al. 2002). This conceptual model predicts that these stressors can become acute such that plants are unable to photosynthesize sufficiently to maintain a positive balance of photosynthesis to respiration (P/R) (Borum et al. 2005). High salinity is insufficient by itself to create the conditions necessary for die-off. Marine plants can tolerate elevated salinities for significant periods of time (Koch and Durako 2004). High salinity, however, sets up a stress in the plant which must generate osmolytes to maintain a sharp osmotic gradient for survival, which then leaves the plant less able to deal with additional stressors.

The overgrown, dense beds of *Thalassia* that developed during a long period of favorable conditions and quiescence encouraged the development of below ground biomass. Cumulative stresses would reduce photosynthetic efficiency and decrease the chance for the photosynthate and oxygen produced in the leaves to support a large biomass and its respiratory demand. Large amounts of accumulated detrital material in the sediments also create a high biological oxygen demand (Carlson et al. 1994). Peak microbial oxygen demand during summer temperatures, low water circulation, and water renewal rates would contribute to the formation of anaerobic conditions in the sediments, simultaneously activating a microbial sulfate reduction process that generates hydrogen sulfide (H_2S) (Pedersen et al. 2004, Borum et al. 2005). Normally, *Thalassia* combats this sediment anaerobic zone by forcing oxygen in the leaves down to the rhizome via air spaces or lacunae and “leaking” this photosynthically produced oxygen from its roots, oxidizing the surrounding rhizosphere and the sulfide it contains (Carlson et al. 1994). The conceptual model considers a combination of stress factors that reduces photosynthetic oxygenation of the sediments and permits intrusion of hydrogen sulfide into the meristem of the plant.

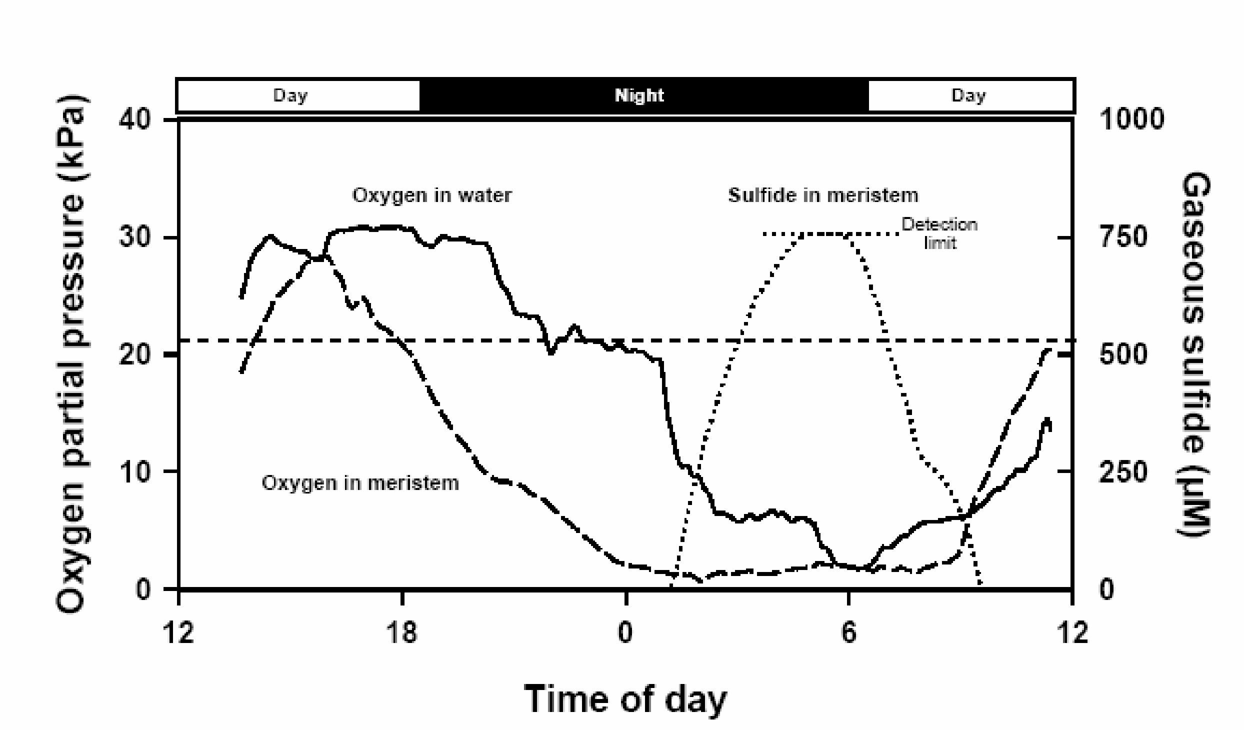


Figure 24. Micro-electrode measures of oxygen and hydrogen sulfide in a *Thalassia* meristem and in the surrounding water column in situ over a 24 h period in an area of active die-off. Source: Borum et al. 2005.

There is empirical evidence from microelectrode measurements in the field and laboratory to suggest that the result of conditions that reduce the P/R ratio within the plant is to poise the seagrass near the point where toxic sulfides in the sediments can enter the plant lacunae, especially during the night when respiration stores of oxygen are at their lowest point (Borum et al. 2005; Figure 24). Sulfide poisoning of the meristem results in death of the short shoot. Divers *in situ* during the die-off reported that bubbles were exuding from green leaf tissue of plants in active die-off zones indicating that the photosynthetic apparatus was functional, but that upon touching the plants, live leaves literally floated off the dead meristems (Zieman 2002).

Seagrasses: Disturbance–Conclusion

After more than a decade of research on the conditions which led to seagrass die-off, there is a growing consensus on a general conceptual model for seagrass die-off. Seagrasses tolerate high salinities, although to do so they must increasing osmolytes in their tissues to counter a strong osmotic gradient, an energy-expending process. Seagrasses also have adaptations to tolerate sulfitic and anaerobic sediments, but also with an energy expenditure and cost to the plant. When several stresses converge, the plants can be overwhelmed such that the community cannot sustain the levels of production required to maintain an oxidized rhizosphere below ground, and anoxia can kill the meristem.

A prolonged drought of historic proportions had gripped that area during the years leading up to the die-off, leading to hypersaline events that were perhaps exacerbated by long-term anthropogenic reductions in freshwater flow into Florida Bay. By 1987, during this period of drought and high salinity and temperature, it is considered quite likely that the plants could not produce enough oxygen to maintain a positive oxygen balance in late summer and early fall when day length and sunlight was declining. P/R balance in the plant shifted as insulation and photosynthesis became reduced, while temperatures remained high, which maintained high sediment respiration.

The long-term trend of nutrient concentrations in the bay has been declining from already low levels in the late 1980s. The nutrient levels in period leading to the die-off are not known, but there is little reason to believe that nutrients were much higher than the relatively low concentrations measured in 1989. Because conditions just prior to the die-off were reported to be of unusually low rainfall, high water temperature and clear, quiescent water, the condition of the bay was that of an oligotrophic, hypersaline marine lagoon. A major hurricane had not impacted the area since Donna, Betsy and Inez of the early and mid 1960s, which is hypothesized to have allowed detritus and mats of organic material to accumulate in the sediments. From 1984-86, the three years prior to initiation of the die-off, the bay experienced rainfall deficits of historic proportions. The years 1984, 1985, and 1986 were the ninth, fifth and third driest years in the 30-year precipitation record, with an average annual deficit of 20% over three years. By 2 inches, 1986 missed being the driest year in the previous 30 years. Salinity was rising in the bay during these years, with levels routinely reaching 50 and 60 psu. The shallowness of the bay and clarity of the water column preclude light limitation as being a chronic problem or primary cause of seagrass die-off. The most likely paradigm that fits the data is one that also fit the facts peculiar to the unique environment of Florida Bay—a lagoon estuary that has been choked off from its historical levels of fresh water input—leading to repeated hypersaline events, overgrowth of *Thalassia* in quiescent conditions, and collapse of an unsustainable population.

A consensus is growing that the restoration of hydrologic flow to Florida Bay is essential to recovery and long-term maintenance of the seagrass community and the ecosystem. The variability associated with seasonal alterations in salinity, the flow effect on removing detrital buildup, and the niche-creation due to a variable salinity regime should work to strengthen the ecosystem by providing a more energetically variable and spatially heterogeneous community. Further studies in controlled laboratory conditions, continued field monitoring, a calibrated hydrodynamic model of the bay and ecological simulation modeling efforts together are required to fully address the questions of how die-off is controlled and how to manage the system to avoid similar imbalances in the future.

Water being discharged to the bay must be of good quality in nutrients. The Everglades is historically an oligotrophic system, and nutrient enrichment would be expected to cause major shifts in ecosystem function and community structure, particularly as may impact the coral reefs in and beyond Florida Bay. One of the major objectives of the Comprehensive Everglades Restoration Plan (CERP) is the restoration of fresh water flows and the removal of barriers to circulation in Florida Bay that will bring a more natural flow and salinity regime to Florida Bay. An increase in water exchange with the adjacent marine systems and a reduction in turnover time of water in the bay, should re-introduce more natural spatial gradients and schedules of salinity variation that will lead to a more stable and robust seagrass community.

Mangroves

Another major vegetation community of major importance to the ecology of Florida Bay is the sub-tropical wetland mangrove forest community. Although not as extensive as the seagrass community, it is the second largest rooted vegetation community in Florida Bay (Zieman 1982, Odum et al. 1982) and serves a habitat to a number of important faunal groups, including the endangered American crocodile (Mazotti 1999), nesting and wading birds, such as the endangered roseate spoonbill and wood stork (Lorenz 1999), and numerous fish and invertebrate that reside in the habitat or use it during critical parts of their life cycle (Ley and McIvor 2002). The mangroves of Florida Bay on the mainland form an ecotone along the hydrologic/salinity gradient of the lower Everglades from the sawgrass *Cladium* dominated freshwater wetland to the Florida Bay estuary (Snedaker 1989). Mangroves are also the primary vegetation community populating over 200 keys and islands in the bay. The mangroves are connected to the bay hydrologically, geochemically and via faunal populations that migrate between the bay and the ecotone (Zieman 1982, McIvor et al. 1994).

There are important natural perturbations to the mangrove systems, most notably fire, sea level rise, and hurricanes (Odum et al. 1982), and there are human perturbations associated with the significant alteration of the hydrology of the southern Everglades (Lorenz 1999). As described below, the result of these perturbations is to exacerbate the already stressful environment in which the mangroves live, threatening their viability and degrading the habitat. In contrast to other mangrove systems, the mangroves in Florida Bay are not threatened by land use conversion, dredging, filling or development because they are protected as part of the Everglades National Park. However, the hydrologic perturbations to the mangrove system originate outside the park boundaries and impact the community nonetheless.

Mangroves: Community Extent and Habitat Characteristics

The mangrove vegetation type populates most of the emergent land in the bay, including the southern Everglades wetland and the islands and keys that are found throughout the bay and surrounding areas, comprising a significant part of the 1295 km^2 of mangroves in south Florida (Lodge 1994). The spatial extent of the mangrove community is dwarfed by the seagrass community. Less than 7% of area of the Florida Bay estuary, including the mangrove fringe, is comprised of mangrove vegetation, as compared to the 45% comprised by seagrasses (Zieman 1982). Nonetheless, the role played by the mangrove system in the ecology of the bay is essential to bay function in terms of biological productivity, habitat, and geochemical cycling (Odum and Heald 1972, 1975, Odum et al. 1982, Snedaker 1989).

Of the six mangrove community types identified by Lugo and Snedaker (1974), the Florida Bay community has five: the *overwash forest*, particularly on the keys subjected to wave action, the *fringe forest* along the banks of waterways and Florida Bay, *the basin forests* in interior depressions, *hammock forests* on elevated outcrops, and the *dwarf or scrub forest* in the marl soils of the flat coastal plains. Although an analogue of the riverine forest type is present along the banks of tidal creeks, technically no true rivers, with associated hydrological cycles and heavy nutrient and sediment loads, are present in the south Florida mangrove community and the "riverine-like" forests are more similar to fringe mangrove forests. The mangroves in Florida Bay are quite small which is attributable to their location at the northern limit of their temperature range (Odum et al. 1982), and to the generally nutrient poor conditions in which they grow (Koch 1996). The majority of the community is considered to be "scrub" or dwarf mangrove (Lorenz 1999), reaching only 1-3 m in height (Odum et al. 1982, Ross et al. 2000).

The mangrove community on the mainland forms what is known as the *mangrove transition zone* or *mangrove fringe* in the southern area of the Everglades wetland. This landscape of this ecotone habitat is characterized by a network of creeks and shallow ponds interspersing broad, regularly flooded marl-peat wetlands (Snedaker 1989). These landscape components are important to parts of the life cycles of crocodiles (Mazzotti 1989, Mazzotti 1999), fish (Ley and McIvor 1999, McIvor et al. 1999) and birds (Lorenz 1999). The mangrove areas on the mainland are less evaporative than the keys, but have seasonally high salinities in overlying waters during the dry season, as marine salinities can encroach several km upstream (Rudnick et al. 1999). The percolation of these waters into the soil and evaporative process can drive soil salinities to 70 psu in the mangrove transition zone (Snedaker 1989).

On the keys themselves, a characteristic topography and vegetation pattern is common: the outer perimeters of the islands are populated by red mangroves and a high berm (30 cm) provides a well-drained, oxygenated substrate where the vegetation tends to be largest and most productive (Enos 1989). This topographic high (important nesting areas for crocodiles and many birds) grades to a central basin in the interior of the island often known as a salina, which tends to be frequently flooded, acts as an evaporative basin and collects and concentrates salt. Salinities can approach 100 psu and water temperatures 40 C in these central basins. Often the interior basins are devoid of vegetation due to the harsh conditions, although in less extreme cases, scrub black mangroves are found, owing to their high tolerance of hypersalinity and high temperatures.

Mangroves: Species Composition, Distribution, and Abundance

The mangrove community in South Florida is dominated by the red mangrove (*Rhizophora mangle)* along the waters edge and close to the land margins that are regularly flooded, and the black mangrove (*Avicennia germinans)* in interior areas and basins of longer hydroperiod (Odum et al 1982). The less common white mangrove (*Laguncularia racemosa*) is found on slightly higher ground, (Ley and McIvor 2002) and the buttonwood (*Conocarpus erectus)* (not considered a true mangrove) is found along naturally formed ridges that are least flooded on streamside banks of tidal channels interspersed with red mangroves. At the margin between the mangrove wetland and the Florida Bay estuary is a topographically high ridge that has been built up by wave deposition of sediments, shell hash, and organic material to a height of about 0.5 m above the surrounding wetland (Wanless et al. 1994, Ley and McIvor 2002). This feature, known as the Buttonwood Ridge, runs the length of the wetland-estuary interface and is notable for the vegetation which tends to grow more productively, with higher standing crops and trees of greater heights (Odum et al. 1982). The ridge plays an important role in the hydrology of the area as the formation tends to trap water and create ponded areas (Madden 2002).

The mangrove community represents important shelter and refuge habitat for Florida Bay's fish populations, particularly within the prop root systems of red mangroves, which provide habitat to adult fishes, including mangrove snapper and snook (Ley and McIvor 2002). The primary fish inhabitants of the mangroves in Florida Bay are: tarpon (*Megalops atlanticus*), ladyfish (*Elops saurus*), cravalle jack (*Caranx hippos*), goliath grouper (*Epinephelus itajara*), gaftopsail catfish (*Bagre marinus*), and snook (*Centropomus undecimalis*). Mangroves also provide habitat to juvenile fishes, pink shrimp, and lobster (Odum et al. 1982).

The close proximity of the mangrove and seagrass communities creates a synergy where marine species can use both communities to fulfill different functions, migrating between them on a daily basis to feed or using each habitat during different life stages (Zieman 1982). Gray snapper (*Lutjanus griseus*), sheapshead (*Archosargus probatocephalus*), red drum (*Sciaenops ocellota*), and spotted seatrout (*Cynoscion nebulosus*) all are recruited into the Florida Bay seagrass community as larvae and juveniles but move into the mangrove habitat for several years to feed and grow to maturity (Odum et al. 1982, Heald and Odum 1970).

Two thirds of the American crocodiles (*Crocodylus acutus*) in Florida inhabit Florida Bay (Mazzotti 1989). Mazzotti 1999 notes that the combination of a well-drained, elevated streamside berm, and a moist soil adjacent to relatively deep, low salinity creeks are ideal nesting sites for crocodiles. The abundance of such conditions in the mangrove ecotone is responsible for the nesting success of crocodiles in Florida Bay (Mazzoti 1989).

Topography and landscape of the mangrove zone contribute to an important feeding ground for fish and birds. The seasonal drawdown of water levels during the dry season is a natural concentrating mechanism for fish which must move from the desiccating flats into increasingly small ponds and creeks. This concentration is of importance to wading birds and other consumers that depend on these forage fish at critical times in the year (Lorenz 1999).

Mangroves: Ecosystem Processes–Production, Decomposition and Nutrient Cycling

Although direct grazing on mangrove leaves is estimated to be minor, or the order of 10% (Odum and Heald 1975), a substantial portion of the primary productivity of the mangrove zone is exported as particulate detritus (POC) and as dissolved organic carbon (DOC) to the estuarine waters of Florida Bay (Odum and Heald 1972, 1975). The tracking and identification of this material in estuarine food webs is complicated and there are many unknowns in the process, but the first evidence that mangroves do export significant amounts of material to adjacent systems was developed for Florida mangrove systems by Heald (1969) and Odum (1970). Subsequent work by Lugo and Snedaker (1974, 1975) and Lugo et al. (1980) provided supporting data showing that exported mangrove detritus was important to higher consumers. This carbon provides the base of an important food web that supports numerous trophic groups outside of the mangrove zone itself, such as shrimp, crabs, and other invertebrates. Materials originating in the fringe mangroves also contribute to the energy flow of Florida Bay via a well-developed microbial community. Microbes utilize mangrove-derived DOC and POC and these microbes subsequently form part of the food complex of the Florida Bay deposit feeding community including grass shrimp (*Palaemonetes*) and mullet (*Mugil*) (Odum et al. 1982). Snedaker asserts that the importance of DOC produced by the mangrove community is at least as great as that of POC exported to adjacent waters in terms of ecological services including alternate food sources, formation of the basis for a rich heterotrophic food web, environmental cues that control the distribution of estuarine species and as regulators of estuarine biochemical processes (Snedaker 1989).

One of the most obvious impacts of mangrove community on the estuarine community is the export of deep red tannic waters during the wet season. These waters are highly loaded with complex organic compounds, including humic acids and fulvic acids, and have unknown impact on the nutrient cycling of the estuarine system (Jaffe 1999, Qualls and Richardson 2003). There is evidence that these compounds may be important in phytoplankton or seagrass productivity (Prakash and Rashid 1985, Day et al. 1988, Rivera et al. 1998). The mangrove system also plays an important role in the cycling of N in the bay. Measurements of N transformations in the swamp floor, and in pond and creek bottoms of the mangrove system in ecotone indicate that a portion of the N load in waters flowing from the Everglades to the bay may be removed as a result of denitrification and a significant portion via sequestration and burial in the mangrove zone (Cornwell et al. 2000). This has implications for the planned restoration of the Everglades (CERP) because increases in flow to the bay may be accompanied by increased N loads (but not P). It remains to be seen how effectively the mangroves may ameliorate any enhanced nutrient inputs.

Mangroves: Disturbance and Vegetation Change

Collectively, changes to the hydrologic regime, both natural and anthropogenic, are the largest stressor to mangrove communities. The basin mangrove community on the keys is vulnerable to extreme conditions of hypersalinity and high temperatures. The natural topography of the keys with a perimeter berm and central depression creates a "one way valve" such that during storms and moderate winds, waves can overwash the berm, introducing salt water to the evaporative central basin (Armentano 1995). The water effectively has no way to leave except through evaporation, concentrating the hot brine. Salinities of 100 psu are not uncommon and can kill all mangrove vegetation in the basin (Armentano 1995). Carlson (1997) hypothesized that during the seagrass die-off of the late 1980s, the large rafts of dead seagrass leaves contributed to the mangrove die-off by washing up on and accumulating on the keys, creating an enhanced berm that trapped additional water and raised salinity levels even further.

On several occasions, the fringe mangroves of Florida Bay have been notably stressed by high salinity conditions, with leaves of red mangroves turning yellow and falling off, most recently in 2001 in the areas of Little Madeira Bay and Terrapin Bay (Coronado-Molina 2000) and ongoing in 2004-5 in the area of Long Sound (Madden unpublished data). The most obvious cause of these events is the natural variation and temporary decrease in local rainfall, but aquifer drawdown and reduced freshwater flow are also important in determining the magnitude and the length of a hypersalinity episode. The role of altered hydrology and reduced fresh water inputs in is as yet not quantified, although hydrologic models of the coastal fringe are being developed (Swain 2000, USGS 2001) that will provide insight to the magnitude of the problem and the degree to which it can be ameliorated by water management.

Sea level rise and the reduction of fresh water flow due to the alteration of hydrology in south Florida are the two factors that most influence structural and functional changes within the fringe mangroves of Florida Bay in the last 100 years (Wanless et al. 1994). Both of these factors are manifest as increases in salinity that create a stress to the plant, reduce productivity, and can be lethal. However, the most common natural perturbations are hurricanes and strong storms. The primary ways in which hurricanes can cause impacts to the mangrove zone are by storm surge and inundation of the community by salt water, deposition of large layers of sediment on pneumatophores and other structures of the trees (Davis et al. 2004), and physical breakage and uprooting (Odum et al. 1982). The mangroves of the Everglades were strongly impacted by Hurricane Donna in 1960, which impacted 40,000 ha, destroying 25-100% of the trees, depending on the area (Craighead and Gilbert 1962).

Hurricane Andrew in 1992 impacted the upland forest several km north rather than the coastal mangroves of Florida Bay, although the mangroves of the western Florida peninsula were heavily damaged by the hurricane (Smith 1992, Armentano et al. 1995). Recovery there has been slow (Wanless et al. 1994). In the Florida Bay mangroves, no extensive storm damage has occurred since the early 1960s and recovery from previous hurricane events has been complete. A cycle of hurricane impacts the Florida Bay mangroves of every 25-30 years has been detected in the historical data by Lugo et al. (1976), who hypothesize that the mangroves there are adapted to reach climax stage and maximum biomass and maturity on that time scale (Odum et al. 1982).

Mangroves: Disturbance and Vegetation Change – Die-off

Concomitant with the historic drought and seagrass die-off event in the late 1980s in Florida Bay, a serious mangrove die-back event occurred in 1989. Reports of mangroves dying back on the islands of the keys, and on areas of the shore, particularly in the north central part of the bay, were made by several researchers

(Armentano 1995, Carlson et al. 1995). This die-back was different than the seagrass die-off in that only parts of the mangrove trees (foliage and branches) were dying, as opposed to the entire tree or, as in the seagrass community, elimination of large swaths of all vegetation in a specific area. Still, the event was serious and given the general sense of alarm over the still-ongoing seagrass event, there was concern that Florida Bay was in the middle of an ecological collapse (Dewar 1992).

Although the die-back persisted for two more years, the problem was put in perspective when, beginning in the early 1990s, it was observed that the trees in the die-back areas were "greening up" with the return of more normal precipitation patterns and salinity levels (Armentano 1995). It was hypothesized that the mangrove community, which can easily tolerate a normal salinity regime, became physiologically stressed by the prolonged hypersaline conditions of the late 1980s. The habitat for the vegetation community itself began to break down and the salinity and temperature ranges for several years were farther and farther outside the range of tolerance for the species.

Mangroves: Disturbance and Vegetation Change–The "White Zone"

A major impact to the Florida mangrove community in the southern Everglades is related to hydrology and is manifest as what is known as the "white zone," which was investigated in the 1930s-40s by Egler (1952) and more recently by Ross et al. (2000). The white zone acquired its name because of its highly reflective white color in remote sensing images, and the greater exposure of the ground substrate due to reduced vegetation cover (Ross et al. 2000). The zone is commonly hypothesized to be the result of salinity increases which have killed much of the vegetation, reduced productivity, and caused changes in community structure in the southern Everglades. In general, a former sawgrass (*Cladium*) and *Eleocharis* marsh has been killed off, giving way to salt-tolerant dwarf red mangrove and creating a salt pan soil substrate that is readily identifiable as a white reflective surface in satellite images. By comparing Egler's data to recent transect and remotely sensed data, Ross et al. (2000) have shown that over the past sixty years the white zone has increased in area and moved northward, particularly in the impounded area cut-off by US Highway 1. The expanding white area is a region of little habitat value, providing no shelter or food resources, and which has lost much of its ecological function of nutrient uptake and transformation (Ross et al. 2000).

Ross et al. 2000 demonstrate that the alteration of hydrology (by reducing fresh water availability and by cutting off natural flows by canal and road building in the southern Everglades) is responsible for the expansion and differential distribution of the white zone. It is believed that these unnatural hydrological changes have exacerbated sea level rise of almost 10 cm since Egler's (1952) study. Areas in the Everglades panhandle just west of US Highway 1, still receive a limited amount of fresh water, and though restricted, show the least expansion of the white zone during the sixty years between measurements, with the boundary moving northward an average of about 1 km.

In areas where flow was most reduced or completely cut off in the "Triangle Lands" to the east of US Hwy 1 the fastest and most severe encroachment of the white zone upstream has occurred, moving inland an average of 2.24 km since Egler's (1952) study. Moreover, Ross et al. (2000) make note of the fact that beyond the dramatic movement of the white zone, changes in species composition and community structure have occurred since Egler (1952) that reach much further inland than the white zone boundary. They note that the shift in community structure of the marsh-mangrove ecotone extends for three times the distance of the white zone encroachment. The disruption of the natural flooding-drying cycle has impeded the formation of periphyton mats (Sklar et al. 2002), which are essential to building the soil substrate, maintaining marsh levels against sea level rise, and provide important food source and habitat (Browder et al. 1994). With the loss of these processes, associated changes in higher trophic level ecosystem function are expected. Efforts to re-hydrate these wetlands are directed at stopping and reversing the perturbation caused by restricted sheetflow and overland circulation (CERP 2000). It is unknown how difficult this will be due to the hysteresis in habitat suitability for the community given that soil salinity has been driven to extremely high levels and large amounts of peat substrate have been oxidized and lost due to exposure (Sklar et al. 2002, Gleason and Stone 1994).

Mangroves: Disturbance–Degraded habitat

Finally, it is worth noting that another effect of altered hydrology in south Florida may be of importance to the mangrove fish community as suggested by the work of Ley and McIvor (2002). An expanded interpretation of the match-mismatch theory of habitat value (Cushing 1969, 1983, Crowder et al. 1990) requires that, in order for a habitat to be favorable to a particular fish species for successful recruitment from the larval stage, all habitat components (e.g. food, temperature, salinity) must be present at appropriate levels and at the appropriate times. If the mangrove habitat is largely functional and structurally viable, but the salinity regime has increased measurably, many estuarine fish species that require a period of lower salinity to be able to effectively use the mangrove community may be unable to exploit the habitat. The value of the mangrove as habitat is compromised by the shifted salinity regime. Reduced mangrove habitat value for larval and juvenile fish initiates a trophic cascade that may impact higher consumers, particularly wading birds and possibly crocodiles that depend on the viability of this habitat (Lorenz et al. 2002). It is not yet known how hydrological changes may have affected the usefulness of the mangrove habitat in Florida Bay.

Marine and Estuarine Representative Species

Wading Birds

One of the defining characteristics of the pristine Everglades environment was the great numbers of wading birds that occupied the marshes, mangroves and bays of the region. Though actual numbers of wading birds under pre-management conditions are not known (Powell et al. 1989a), anecdotal accounts and observations by Audubon and others indicated that wading bird presence was indeed large. Early estimates of millions of birds or hundreds of thousands of birds may be apocryphal (Odgen 1994), but the area certainly was notable for the huge flocks of wading birds. Currently, there are fourteen wading bird species that breed in south Florida (Lodge 2005).

From the last half of the 1800s until the early 1900s, plume hunters decimated wading bird populations in the region, driving many populations to the point of extirpation from the region (Powell et al. 1989a). Though plume hunting regulations and prohibitions were implemented in the early 1900s, many populations did not begin to recover until the 1940s (Powell et al. 1989a). In the early 1930s, the National Audubon Society began to record scientific census data on wading bird populations and nesting locations. This scientific record has been continued, with a few interruptions, to the present day, affording researchers with a long-term history of wading bird population dynamics in the Everglades region, albeit one that began after drainage activities had already commenced.

An analysis of this long-term data set for five species (Ciconiiformes family) in the central and southern Everglades (present day Everglades N.P. and WCAs 1, 2, and 3) indicate that the number of wading birds has declined by approximately 80%, from an early drainage period (1930-1946) high of 180,000-245,000 birds to the high in a late drainage period (1974-1989) of only 50,000 birds. Nesting sites have shifted from headwaters on present day Everglades N.P. to ponds and impoundments on WCAs (Odgen 1994). More recent estimates of wading birds indicate a 90-95% decline in south Florida populations, though these declines have not been as precipitous in Florida Bay as in other areas (NPS 1997).

Figure 25. Photo of roseate spoonbill. © StockTrek.

It is presumed that these changes in distribution and abundance were precipitated by water management activities that altered the quantity and timing of delivery of freshwater into Florida Bay, and that disrupted the natural sheet flow of water through the Everglades marshes. Specifically, drainage has led to the decline in foraging habitat, increased the frequency of extreme drydown events, and relocated areas with long hydroperiods from Everglades N.P. to the WCAs (Odgen 1994). In this sense, wading birds are good indicator species for the region because their populations are sensitive to anthropogenic disturbance, in this case the disruption of the natural hydrologic cycle.

Roseate Spoonbill

One wading bird species, the roseate spoonbill (*Ajaia ajaja*), has been selected as a representative species for this assessment. Roseate spoonbill (Figure 25) is a good indicator species for the aforementioned reasons, and also because it has a unique feeding behavior called tactolocation that makes it dependent upon certain hydrologic conditions. Tactolocation is a method of feeding that involves submerging a partially open bill into the water to search for food, and then, once encountering prey, quickly snapping the bill shut (Allen 1942). This feeding behavior requires that the spoonbill have access to prey in shallow waters. In addition, spoonbill breeding is dependent on the seasonality of hydrology, where the wet season allows the prey base to flourish and the dry season concentrates this prey base into drying wetlands (Lorenz et al. 2002). Given that water management has altered water depth, hydroperiods, and the wet/dry seasonality, there is a natural link between human alteration and spoonbill biology.

Roseate Spoonbill: Land Use History

Like other wading bird species, roseate spoonbills were severely impacted by plume hunting in the late 19th and early 20th century in the Everglades region. By the 1930s, only 200 nesting pairs occurred in Florida. Spoonbills were almost extirpated in 1935, where it was estimated the Florida population consisted of only 15 nesting pairs (Grimes and Sprunt 1936). All these nesting pairs were located in the only functioning breeding colony in Florida, on Bottle Key in eastern Florida Bay.

Roseate Spoonbill: Population Dynamics

From the brink of extirpation in the 1930s, spoonbill populations exhibited two phases: a dramatic recovery from the late 1930s until late 1970s, followed by a decline from 1979-1999 (Figure 26, Lorenz et al. 2002, Powell et al. 1989a). From 1955-1978, the population roughly doubled every decade, so that by the 1978-1979 season, the highest number of nesting pairs, over 1200, were recorded in Florida Bay (Powell et al. 1989a). Subsequently, from 1979-1999, the population had been reduced to half of this peak population, down to approximately 600 nesting pairs (Lorenz et al. 2002).

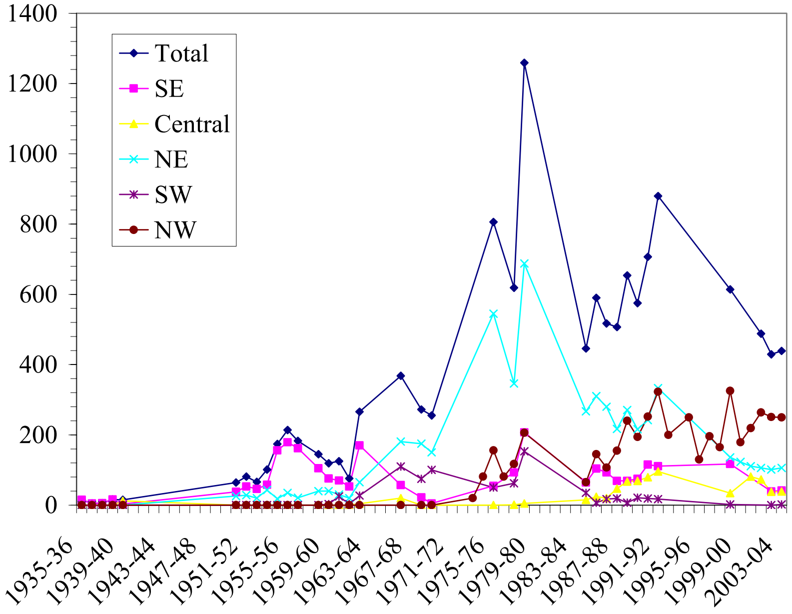


Figure 26. Number of roseate spoonbills nests in Florida Bay from 1935-2002, including number of nests per colony. Source: Updated from Lorenz et al. 2002.

A detailed spatial analysis of spoonbill colonies indicates that roseate spoonbill nest locations have periodically shifted to different regions of Florida Bay in the last half of this century (Lorenz et al. 2002). The majority of nesting colonies have shifted in three periods from predominance in the southeast (mid-1930s to early 1960s) to the northeast (early 1960s to 1985) to the northwest (1985-1999) (Figure 27, Lorenz et al. 2002). Allen (1942) was first to note that spoonbill colonies follow a pattern of slow establishment, peaks and declines with Florida Bay and offered four factors that may help explain this pattern. These factors are: (1) disease/parasites, (2) flushing from human visitors, (3) raccoon predation, and (4) food source availability.

Roseate Spoonbill: Disease

Diseases caused by parasites have been noted within spoonbill populations in Florida Bay (Allen 1942, Bjork and Powell 1994), but it is thought that these diseases affect only individual nesting sites rather than entire colonies, and perhaps play a secondary role in nest abandonment (Bjork and Powell 1994, Lorenz et al. 2002).

Roseate Spoonbill: Predation

Raccoons are considered to be the primary predators of spoonbills (Allen 1942). They can reach nesting locations along the mangrove fringe environments on the northern border of Florida Bay and some of the mangrove keys within the bay, provided they are accessible by exposed mud banks. There is little to no overlap been the distribution of raccoons and spoonbill nesting sites, indicating that spoonbills may in fact avoid areas where raccoons are present (Lorenz et al. 2002). Raccoon predation does not explain the spatial and temporal patterns of spoonbills directly, though they may limit the areas that are available for spoonbill nests.

Roseate Spoonbill: Visitor Impacts

There is data available that might suggest that human visitors directly impact spoonbill nesting activities, though this has been a concern for some time (Allen 1942). Recreational fishers in the bay can flush birds from nesting sites in mangrove keys (King 2005), but it is unclear what impact these visits may have on nest abandonment. Researchers have found no pattern or impact of their visits to nesting site locations (Lorenz et al. 2002), and there is no reason to believe that visiting impacts would correlate spatially with the spoonbill nesting patterns noted over time (e.g. visits shifting from the southeast to northeast over the time periods where nesting sites shifted).

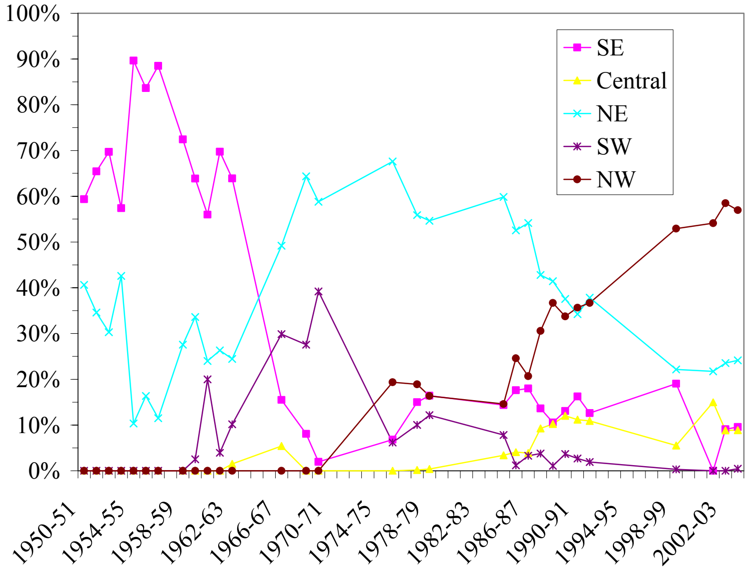


Figure 27. Percent of total spoonbill nests found in each colony from 1950-2004. Source: Updated from Lorenz et al. 2002.

Roseate Spoonbill: Foraging Habitat

The alteration and destruction of foraging habitat is the most likely cause for recent declines in abundance and changes in distribution of spoonbills in Florida Bay (Lorenz et al. 2002). Several lines of evidence suggest that these alterations occurred at the same time that spoonbill nesting patterns shifted in the bay, first from the southeast to northeast, and next from the northeast to the northwest. Both events were spurned by anthropogenic changes that impacted mangrove fringe habitats. These habitat impacts, in turn, are thought to have impacted prey densities, eventually forcing spoonbills to relocate into areas with high densities of prey. The restoration of freshwater flows that are closer to historic flows are expected to reverse this trend.

Between 1957-1959, mangroves were destroyed or degraded along the northern Florida Keys. Monitoring of flight patterns suggest that these areas were destinations for foraging of spoonbills that nested in southeastern Florida Bay. Starting in 1958, spoonbill flight patterns began to shift northward, until sometime in the period 1963-1967, spoonbills had abandoned their colonies in the southeast and moved to the northwest.

A similar abandonment of nests occurred in the northeast in the period from the early 1960s to 1985, also inferred to be caused by habitat destruction of mangroves, but in this case, a result of changes to the hydrologic regime. In the estuaries that border northeastern Florida Bay, upstream water management has led to greatly reduced water flows, alteration of hydroperiods, and the dampening of the water flow seasonality (Johnson and Fennema 1989). Others have found that this region experienced increased salinities during this period (McIvor et al. 1994). Though no direct measurements have been made that specifically link reduced water flow and higher salinities with lower densities of prey, other studies suggest that prey densities would be diminished under these circumstances. Lorenz (1999) showed that reduced hydroperiods and increases in salinity decrease prey density in the same mangroves in northeastern Florida Bay. Though not measured, it is inferred that out-of-season influxes of freshwater in the dry season, when prey are normally concentrated, also serves to diminish successful feeding by spoonbills. Johnson and Fennema (1989) demonstrated that the seasonality of freshwater inputs in Taylor Slough and C-111 basin has been dampened, especially after 1982, with drier wet seasons, and wetter dry seasons. Finally, due to water diversion and eventual drydown, the mangrove fringe has actually moved inland, 3.3 km since the mid 1940s (Ross et al. 2000).

It is clear that anthropogenic changes to water flow in Florida Bay have altered the structure and function of the coastal mangroves that served as foraging habitat for spoonbill. Impacts include either direct destruction of mangroves or a significant degradation of the hydrological functions that maintain them. Additionally, there is strong circumstantial evidence that this habitat alteration may have led to an overall reduction in the number of spoonbills nesting in Florida Bay since the late 1970s and shifts in nesting areas. If this hypothesis is valid, then a resumption of freshwater inflows into northeast Florida Bay that more closely resemble natural conditions are predicted to restore mangrove foraging habitats, and stabilize roseate spoonbill abundance and distribution.

Marine Invertebrates

Spiny Lobster

For economic and ecological reasons, the Caribbean spiny lobster (*Panulirus argus,* Figure 28) merits consideration as a species of special concern in south Florida, and the presence of large lobster nursery areas in southern Florida Bay within the Everglades National Park (Everglades N.P.) confirms its importance for this assessment (Field and Butler 1994, Herrnkind et al. 1997). Since 1986, spiny lobster has consistently ranked as the first or second most valuable commercial fishery in Florida, alternating with pink shrimp. The dockside value of the commercial lobster fishery now varies between $20 and $30 million/yr (Harper 1995), which does not include ancillary economic benefits (i.e.,“multiplier effects”) such as the value of the large recreational sport diving fishery for lobster. Each year,

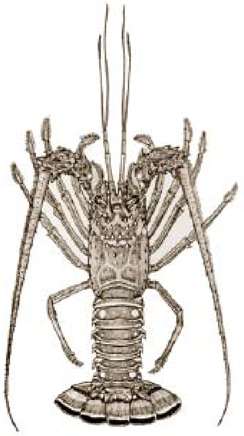


Figure 28. Spiny Lobster. Source: FMRI 2003a.

over 150,000 people purchase permits to recreationally fish for lobster in south Florida, accounting for ~20% of the total recorded catch.

Ecologically, *P. argus* is an important component of the Florida Bay marine ecosystem for several reasons. It is a locally abundant, benthic predator of a variety of gastropods, bivalves, crustaceans, echinoderms, and small fishes (Andree 1981, Herrnkind et al. 1988). Its juveniles also fall prey to an enormous variety of even larger predators, comprising a major portion of the diet in many species (e.g., nurse sharks, bonnethead sharks, rays), some of which are important to sport fisheries (e.g., permit, bonefish) and commercial fisheries (e.g., gray snapper, grouper) (Smith and Herrnkind 1992). Lobsters are thus an integral component of the trophic structure in southern Florida Bay, both as predators and prey.

The emphasis on “southern Florida Bay” highlights another aspect of the species ecological importance as an indicator of ecosystem transition. Lobsters are true marine, not estuarine species. They persist in southwestern Florida Bay at its juncture with the adjacent Florida Keys and Southeast Gulf Shelf marine ecosystems, being physiologically limited by the prevailing salinity. So like many other marine species (e.g., hard and soft corals, large sponges, bonefish, groupers, sea turtles, etc.), lobsters are indicators of ecological change in Florida Bay because they are sensitive to alterations in freshwater input to the system, as well as to changes in water quality that affect benthic habitat structure (Butler et al. 1995).

Everglades N.P. does not participate in or fund the study or monitoring of spiny lobster or hard-bottom habitat where lobsters occur. Most research on spiny lobster occurs in the adjacent Florida Keys marine ecosystem. However, several independent studies of lobster population dynamics have taken place within the Everglades N.P. boundaries and a recently implemented (2002) program for monitoring hard-bottom habitat and spiny lobster in south Florida includes sites within the Everglades N.P.

Spiny Lobster: Life History

The Caribbean spiny lobster has a complex life history, involving several distinctly different life stages each requiring a separate marine habitat for completion of its life cycle (Butler and Herrnkind 2001). Adult *P. argus*, which may live for > 25 yrs, dwell primarily in coral reef environments, taking refuge by day on the reefs and foraging at night in rubble and seagrass habitats up several km from the reef (Cox et al. 1997). Reproductive activities and spawning take place exclusively on the reef, primarily from March – July in Florida. The resultant planktonic larvae are long-lived (estimated 6 – 12 month duration; ~ 11 larval stages), planktotrophic predators of the open ocean realm where they are presumably transported great distances. This is no doubt the reason for their widespread geographic distribution (Brazil to Bermuda) and what is believed to be a pan-Caribbean population with little genetic substructure (Silberman and Walsh 1994, Silberman et al. 1994).

Following larval dispersal, the subsequent postlarval stage travels inshore each month on new moon flood tides (Acosta et al. 1997). Those that encounter structurally complex vegetation settle, preferably in clumps of red macroalgae (e.g., *Laurencia* spp.) within hard-bottom habitat, but sometimes in seagrass meadows or among heavily fouled mangrove roots (Marx and Herrnkind 1985, Herrnkind and Butler 1986, Acosta and Butler 1997). The postlarvae then metamorphose into the early benthic juvenile (EBJ) stage and remain hidden and solitary within their settlement habitat for ~ 3 - 4 months (Butler and Herrnkind 1991, Herrnkind and Butler 1994). Later, the juveniles emerge from the macroalgae, become social, and take up daytime refuge in crevice-bearing shelters such as sponges, corals, and solution holes that occur on hard-bottom habitat (Eggleston et al. 1990, Forcucci et al. 1994, Butler et al. 1995, Herrnkind et al. 1997). Postalgal juveniles initially occupy relatively small home ranges, but at 45 mm CL (carapace length – the standard scientific measure of size in lobster) they become more nomadic (see Butler and Herrnkind 2001). After 1–2 years in the shallow, inshore nurseries, the lobsters then migrate tens of kilometers offshore to the coral reefs fringing the Florida Keys. The lobsters that inhabit Florida Bay are nearly all juveniles; no reproductive activity has been reported within this region. Therefore, the ecological processes that naturally limit lobster populations in Florida Bay are those that impact juveniles within nursery habitat.

Spiny Lobster: Hard-bottom Habitat

Hard-bottom habitat with plentiful macroalgae and crevice shelters is the primary nursery for spiny lobster in Florida and within the Everglades N.P. in southwestern Florida Bay. Hard-bottom habitat (also called: live-bottom, hardgrounds, bedrock outcrops, alcyonarian-sponge communities, nearshore rock-bottom, and algal-dominated hard-bottom; Chiappone 1996) are typically shallow (< 3m) and are characterized by low relief (< 0.5 m), limestone bedrock overlain by a thin veneer of sediment. Hard-bottom covers ~ 40 % of the seafloor in the Florida Keys (Zieman et al. 1989) and is interspersed with sand, calcareous mud-bottom, and seagrass habitats.

As the term “algal-dominated hard-bottom” implies, numerous species of red, green, and brown macroalgae occur there. However, bushy, red macroalgae (primarily *Laurencia spp.*) dominant the algal biomass and serves as the primary settlement habitat for lobster. Sponges, octocorals, and hard corals are the most conspicuous sessile fauna found in hard-bottom habitats and, like lobsters, are sensitive to changes in salinity. Massive sponges (e.g., loggerhead sponge, *Speciospongia vesparium*; vase sponge *Ircinia campana*; stinker sponge, *Ircinia felix*, etc.) and a few species of tall, branching octocorals (sea plume, *Pseudopterogorgia spp.*; angular sea whip, *Pterogorgia anceps*) are common, and they provide three-dimensional structure in these otherwise flat seascapes. A few species of sponge (sheepswool sponge, *Hippiospongia lachne*; yellow sponge, *Spongia barbara*; glove sponge, *Spongia graminea*) of commercial importance also occur on hard-bottom in Florida Bay, although commercial fishing of sponges is prohibited in the Everglades N.P.

A large but untold number of motile macrofauna also dwell in hard-bottom areas. Many motile species use hard-bottom habitat opportunistically (e.g., stone crabs, bonefish, tarpon, various sharks, sea turtles, etc.). Some are obligate dwellers of hard-bottom and are rarely found elsewhere (e.g., spider crabs, octopus, cleaner shrimps, etc.). For others, hard-bottom is a nursery (e.g., Red grouper, Nassau grouper, Hogfish, various species of snapper, and parrotfish), as it is for spiny lobster.

Spiny Lobster: Population Dynamics–Factors limiting abundance

Postlarval Supply: The local availability of new recruits (i.e., planktonic postlarvae) transported from outside of Florida Bay ultimately determines the maximum lobster population possible. Large-scale oceanographic processes operating offshore of the Florida Keys, coupled with larval and postlarval planktonic mortality, are primarily responsible for changes in the temporal pattern of supply of lobster and other postlarvae to Florida Bay (Acosta et al. 1997, Acosta and Butler 1999, Yeung et al. 2001; Figure 29). In contrast, the local abundance of postlarvae available for settlement in coastal hard-bottom nurseries is probably driven by the complex interactions between the tidal and wind-driven flow of ocean water from offshore and Florida Bay’s intricate bathymetry (Smith 1998, Lee et al. 2001, Smith and Pitts 2002). Florida Bay is crisscrossed by shallow (<1 m), carbonate mud banks that separate the bay into a series of shallow (< 3m) basins interconnected by deeper (up to 4m), narrow channels (Zieman et al., 1989). The mudbanks restrict water circulation in the bay (Nuttle et al. 2000) and, presumably, the transport of planktonic larvae.

Settlement of lobster postlarvae in hard-bottom areas north (bayside) of the mud banks is generally an order of magnitude lower than in similar areas to the south (oceanside)(Field and Butler 1994). The abundance of *Laurencia* on a site also plays a role in determining local patterns of settlement, because its chemical scent attracts the strong swimming lobster postlarvae (Butler and Herrnkind 1991, Herrnkind and Butler 1994). Following settlement, post-settlement mortality (i.e., predation, disease, starvation) takes it toll.

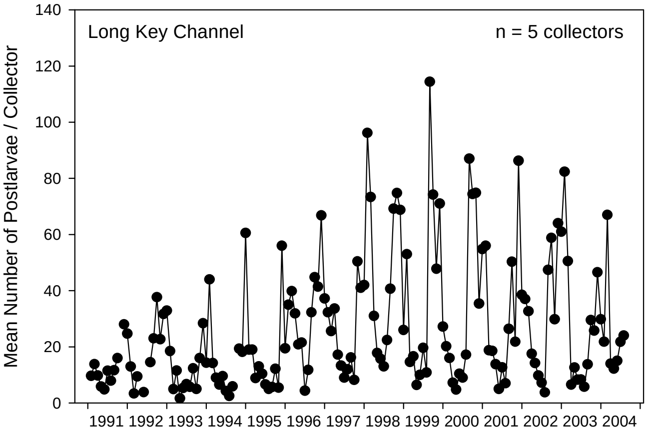


Figure 29. Monthly temporal pattern in the supply of postlarval P. argus to Florida Bay from 1991 – 2004, as estimated from five Witham-type postlarval lobster collectors deployed on the oceanside of Long Key Channel.

Predation and Habitat Availability: Experimental studies indicate that the local abundance of juvenile lobsters is limited in a density-dependent manner by shelter-imposed demographic bottlenecks that set an upper threshold, below which the juvenile population fluctuates in response to local changes in larval supply (Butler and Herrnkind 1997, Herrnkind et al. 1997). Predation by various species of fish (Mintz et al. 1994) and by octopus (Berger and Butler 2001) is normally the most significant source of mortality for juvenile lobsters in Florida Bay. It is estimated that < 5% of the postlarvae that settle in Florida Bay survive their first year (Butler et al. 1997, Sharp et al. 2000). For the smallest lobsters (i.e., recently settled postlarvae and EBJ), the availability of red macroalgae is crucial to survival, so they rarely leave it until they are too large to effectively shelter there (Herrnkind and Butler 1986, Butler and Herrnkind 1991, Acosta and Butler 1997). Not only are EBJs sheltered from predators within the bushy algae, their cryptic behavior, asocial nature, and camouflage coloration makes them less visible as well. For larger, crevice-dwelling juvenile lobsters, the risk of predation diminishes significantly when shelters of suitable dimensions are available, and for lobsters dwelling in social aggregations and thus capable of group defense (Eggleston et al. 1990, Eggleston and Lipcius 1992, Mintz et al. 1994, Herrnkind et al. 2001).

Food Availability: Postlarval lobsters are the first stage to enter the nearshore waters of Florida Bay, but they are incapable of feeding, living instead off internal energy stores accumulated during the previous larval stages. Food for EBJs does not normally appear to be limiting, because they have access to abundant prey (e.g., copepods, mollusks, shrimp, etc.) that also inhabit their macroalgal settlement habitat (Herrnkind et al. 1988). However, siltation can reduce prey abundance in macroalgae, inducing emigration of EBJs from the food depleted habitat (Marx and Herrnkind 1985, Herrnkind et al. 1988). There is no evidence that food limits the growth or local abundance of crevice-dwelling juvenile P. argus in Florida Bay (Behringer 2003). In fact, field estimates of growth in Florida Bay are among the highest reported for this species, averaging 0.95 mm CL/wk (range: 0.35 - 1.25 mm CL/wk for individuals 20-25 mm CL and 40-45 mm CL, respectively) (Forcucci et al. 1994). These results indicate that lobsters in some areas of Florida Bay can reach Florida's legal harvestable size (76 mm CL) ~ 1.5 years after settlement.

Disease: There are four diseases known to infect *P. argus*, but there are undoubtedly more that have not been discovered (Evans et al. 2000, Porter et al. 2001, Shields and Behringer 2004). Three of the four known diseases were reported from *P. argus* populations in Florida: shell disease (Porter et al. 2001), microsporidiosis (Bach and Beardsley 1976), and PaV1 virus (Shields and Behringer 2004). The first two diseases are rarely fatal, but the PaV1 virus is pathogenic and lethal. Most infected lobsters die within 90 days. Methodical sampling for disease in lobsters in south Florida has only been conducted for PaV1 viral infections, and only since its discovery in 1999. The disease is widespread in the Florida Keys, appearing at about 25% of the sites where lobsters were surveyed, including sites in Florida Bay. The prevalence of PaV1 infections in juvenile lobsters has remained around 5% per site (max = 30%/site). The disease is most frequent (mean = 16%) among the smallest crevice-dwelling juveniles (<20mm CL) and its incidence declines with size; few adults (<1%) are affected. Its prevalence in EBJs in nature is unknown, but laboratory evidence suggests that it is likely to be even higher than in larger juveniles. Laboratory experiments indicate that the virus is transmitted most effectively by direct contact with infected individuals, but can also be contracted via ingestion of infected tissue (Behringer 2003).

Water Quality: The water quality conditions typically found in Florida Bay that are most relevant to spiny lobsters are temperature, salinity, and dissolved oxygen. Temperature and salinity interact to impact the survival of *P. argus*; postlarvae and EBJ are particularly intolerant. Postlarval and EBJ lobsters die if exposed to salinities that differ more than a few units from 35 psu and mortality rates increase exponentially at low (<20°C) or high (>30°C) temperatures (Field and Butler 1994). Larger juveniles can tolerate more extreme salinities, but they increase their rates of movement at salinities other than 35psu, presumably in an effort to emigrate from the area (Butler 2002). Low dissolved oxygen concentrations, such as those induced by high temperatures and eutrophy, are lethal to lobsters of all sizes. Juvenile lobsters are also indirectly impacted by blooms of planktonic cyanobacteria and abnormal salinity, which kill sponges and octocorals that provide habitat for lobsters (Butler et al. 1995, Herrnkind et al. 1997, Phlips et al. 1999, Butler 2002).

Spiny Lobster: Population Distribution and Abundance

Spiny lobsters occur primarily in the southwestern section of Florida Bay within structurally complex hard-bottom habitats south of the first bulwark of carbonate mud-banks (e.g., Nine Mile Bank, Twin Key Bank). An exception being populations of juvenile lobsters found north of those banks in hard-bottom habitat within Twin Key Basin. However, large juveniles and subadults are highly mobile and nomadic, and can often be found in locally large numbers sheltering next to sponges, in holes, or under seagrass blow-outs within the channels that dissect the banks.

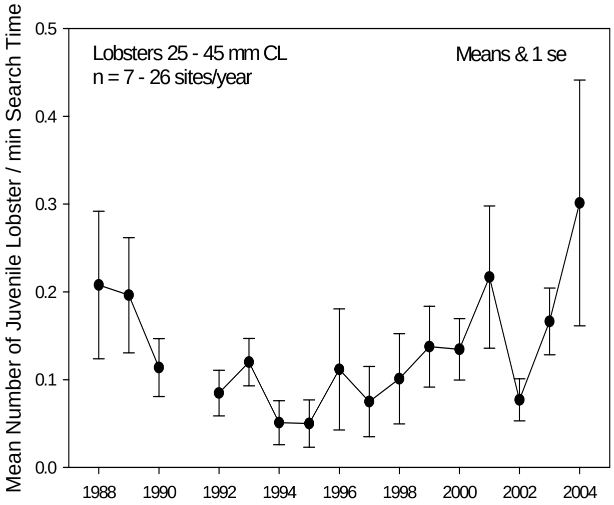


Figure 30. Diver catch-per-unit-time surveys of the relative abundance of crevice-dwelling juveniles (25 – 45 mm CL) on hard-bottom in the middle Florida Keys (including Florida Bay) from 1988 – 2004.

Densities of crevice-dwelling juveniles on hard-bottom habitat in Florida vary widely, but in prime nursery habitat their numbers can exceed 400 individuals/ha (Forcucci et al. 1994). The density of EBJ can not be easily determined, but based on estimates of mortality (Butler et al. 1997, Sharp et al. 2000) must be at least an order of magnitude greater than that of the crevice-dwelling juveniles. Surveys of crevice-dwelling juvenile abundance from 1988 – 2004 reveal a general decline in abundance in the early 1990's followed by a gradual recovery later in the decade and another decline and recovery 2002 – 2004 (Figure 30). Declines in juvenile abundance in Florida Bay in the early 1990's are attributed to the mass mortality of sponges over an area > 200 km^2 (Butler et al. 1995). The widespread mortality of sponges followed repeated and persistent (3 - 6 month duration) blooms of cyanobacteria that blanketed south-central Florida Bay in 1991 and 1992. The loss of the sponge community was nearly 100% at many sites in Florida Bay and at those sites, juvenile lobster abundance declined by an average of 49% (Herrnkind et al. 1997). The sponge community has recovered slowly, but has not assumed its previous diversity or size structure (Stevely and Sweat 1999).

Spiny Lobster: Threats and Management Issues

Potential threats to lobster populations in Florida Bay include: poaching, hurricanes, climate change, and diminished water quality. It is not known how severe a problem poaching is within the Everglades N.P., but strict enforcement within the park serves to minimize the problem. Poaching is probably most problematic along the southern edge of the Everglades N.P. boundary where lobsters and people are most abundant, and where errors in navigation may mistakenly place fishers inside the poorly demarcated park boundary. The number of fisherman that district rangers in Everglades N.P. have warned or cited for lobster poaching within the park has increased ten-fold from 30/yr to 296/year in the years 1990-2004 (King 2005).

Hurricanes may temporarily impact lobster populations via turbulent transport, local habitat destruction, and freshwater run-off, but their effects are not likely to be long lasting. The effects of global climate change, in particular, global warming on lobster populations in Florida Bay are unknown and probably more complex than one might assume. Geographically heterogeneous and more extreme climatic effects, along with indirect effects on lobster habitat, prey, and predators render simple hypotheses based on thermal stress alone meaningless. In contrast, anthropogenic impacts on water quality and marine habitats in Florida Bay are well documented and are likely to continue with perhaps unpredictable results in response to the Comprehensive Everglades Restoration Plan (CERP) (see Porter & Porter 2002 for a review). Diminished water quality (i.e., eutrophication) and altered salinity are the gravest threats to the persistence of lobsters in Florida Bay.

Hard-bottom communities and the lobster populations they sustain have proven to be sensitive indicators of ecosystem change (Butler et al. 1995, Herrnkind et al. 1997). Although the direct impact of inappropriate salinity on lobster are likely to be severe, field studies and simulation modeling suggest that significant effects of eutrophication and salinity change on lobster will also be manifested indirectly through changes in nursery habitat structure and disease epidemiology (Butler 2003, Butler et al. *in press*). Currently, much of the lobster nursery habitat in Florida Bay is subject to highly variable salinities in response to changes in rainfall and the release of freshwater from the Everglades (Boyer et al. 1999). CERP is expected to decrease salinity and perhaps increase its variability in this same region of Florida Bay (Nuttle et al. 2000). Salinities appreciably different from 35psu kill *P. argus* postlarvae and EBJ (Field and Butler 1994) and are a stressor to larger lobsters, which alter their patterns of movement when salinities change (Butler 2003). It is also feasible that inappropriate salinity may promote PaV1 disease in spiny lobster as it does in other crustaceans (LeMoullac and Haffner 2000). Sponge, hard coral, and octocoral abundances and community structure will diminish in Florida Bay if salinity changes more than a few psu. This loss of shelter for larger crevice-dwelling juvenile lobsters is likely to have two consequences: higher mortality due to predation (assuming predator abundance and their foraging preferences and efficiency are not diminished), and an increased probability of intraspecific encounters and perhaps transmission of the PaV1 virus among lobsters.

Marine Fishes

Many species of fishes range between the bay and adjacent reef habitats over their life cycles. While more than 250 species of fishes are found within the coastal waters of the park, more than half of these are known to occur within Florida Bay proper (Schmidt 1979, Loftus 2000). Most of the fishes that reside within the boundaries of the park do so only during their juvenile phases and leave the boundaries of Everglades N.P. for adult habitat and spawning sites located elsewhere upon reaching sexual maturity (Tabb 1966).

Tidal flows, upland freshwater input, mixing of water masses, and the migrations of marine organisms connect Florida Bay with the inland Everglades islands southwest Florida shelf and the Florida Keys reef tract, including the corals of the Dry Tortugas. Because of this connectivity, processes affecting change in the ecosystem of the bay extend well beyond the boundaries of Everglades N.P.

This complex system of linked marine ecosystems harbors diverse living resources and sustains lucrative commercial and recreational fisheries (Bohnsack et al. 1994), which require that a delicate ecological balance is maintained. Continually growing human activities have put these ecosystems under severe stress (NMFS 1995). Effective restoration and management cannot be achieved without improved understanding of the complex ecological controls and linkages inherent in these systems.

Marine Fish: Habitats

Everglades N.P. encompasses most of the Everglades, its freshwater inlets, and most of Florida Bay. It is bordered to the northwest by the Gulf of Mexico and to the south and east by the upper Florida Keys. Florida Bay is a lagoonal estuary with several narrow connections to the open ocean and 20 small streams that intermittently discharge fresh water from the Everglades. These outlets primarily originate in two catchments: Taylor Slough and Canal-111. The Bay is divided into 30 shallow sub-basins by shallow mudbanks (Ley et al. 1999). The Bay floor is covered in seagrass beds, mud, sand and limestone rubble. The shores of Florida Bay are predominantly bordered by mangroves and there are many small mangrove covered islands especially in the Ten Thousand Islands area which lies partly within the park and runs northwest immediately adjacent to its northwestern border.

Florida Bay and the Ten Thousand Island regions are thus connected to the freshwaters of the Everglades, coastal and insular mangroves areas, the Florida Keys, the Florida reef tract, the southwest Florida shelf, and the Dry Tortugas. This connectivity occurs via the flow of water, the transport of larvae, and the ontogenetic migration of fishes. Therefore, an assessment of the state of Everglades N.P. should extend beyond the boundaries of the park since processes occurring within the greater ecosystem affect the reproduction, larval transport, and replenishment of populations occurring within the park.

Marine Fish: Habitats–Mangroves & Seagrass

In Florida Bay, as a result of the many freshwater inlets, their variable input over time and variation in intrusion of oceanic water, there exist a number of basins within the bay that differ in depth and circulation patterns with resultant variation in bottom type, salinity and temperature. The great variation in these basins provides a wide variety of habitat types and results in a great diversity of fish assemblages (Sogard et al. 1989, Ley et al. 1999). The productive capacity of the park for each species is determined by the amount of overlap between areas which provide favorable salinity ranges and areas which provide the appropriate type habitat (e.g., mangrove, seagrass, shoreline, depth, etc.). However, the quality of seemingly appropriate habitats must also be considered as excess nutrients and organic detritus can degrade these via reduced water clarity, low dissolved oxygen, etc. (Browder et al. 2003).

Within Florida Bay many species of fishes are found to occur in greater densities within seagrass meadows vs. exposed, unvegetated bottom types (Sogard et al. 1989, Thayer and Chester 1989, Thayer et al. 1999). Similarly, fish density and biomass for some species (e.g., groupers) have been reported to be greater in mangrove prop root habitat than in adjacent fringing seagrass areas and serves as the primary habitat for juveniles of some species e.g. goliath grouper (Koenig et al. *in review*). Thayer et al. (1987) concluded that mangrove prop root habitat in Florida Bay and nearby areas were occupied a distinct faunal assemblage. More recent work indicates that several snapper and grunt species make nightly migrations from their mangrove "daytime resting" sites to feed in seagrass beds and sand flats on crustacean prey (Rooker and Dennis 1991). Stable isotope and gut content analyses on Florida Bay gray snapper collected from mangrove habitats (Harrigan et al. 1989) tends to support this scenario. This suggests the mangrove-seagrass habitat *combination* is more important than either habitat alone, especially during the juvenile and subadult stages.

Marine Fishes: Florida Bay Regional Characteristics

The northeast region of Florida Bay exhibits greater variation in salinity, lower density of seagrass, fewer mud banks, and is more restricted to larval input compared to other regions of Florida Bay (Browder et al. 2003). Schmidt (1977) reported lower forage fish diversity, biomass, and numbers in north central Florida Bay in salinities > 45 ppt. The western and southwestern sections of Florida Bay that lie within the park contain extensive estuarine, mangrove swamp and bordering mangrove areas which are ideal habitat for many fish species. Abiotic characteristics that potentially have a large influence on the distribution of abundance of fishes inhabiting the park on a region scale include bottom type, freshwater inflow, and the degree of mixing with Atlantic and Gulf waters. Temporal variation of these characteristics would further emphasize regional differences in habitats (e.g., seasonally variability in salinity is greatest in the northeast region of Florida Bay) (Browder et al. 2003).

The Ten Thousand Islands area appears to be a very important area of fish habitat within the Everglades N.P. It is located near Chokoloskee and Everglades City, Florida and most of this area is located within the boundaries of the Everglades N.P. (Cass-Calay and Schmidt *in review*). The Ten thousand Islands area is believed to be the current center of abundance and primary nursery grounds for goliath grouper (Sadovy and Ecklund 1999, Koenig et al. *in review*).

Marine Fishes: Florida Bay as a Nursery

Everglades National Park includes aquatic habitats that range from fresh water rivers and brackish inlets to shallow estuarine habitats and functionally marine areas. Some animals spend their entire life cycles within Florida Bay but some occur as adults outside of the bay on the offshore reefs off the Florida Keys (Ley et al. 1999, Ley and McIvor 2002). A large number of species of fish (including snappers, groupers and grunts) that occur as adults on these reefs spend their juvenile phase within Florida Bay. The habitats found in the park especially seagrass beds and mangroves seem to be of particular importance to juvenile fishes as well as many invertebrates. They provide cover as well as a source of detritus on which the invertebrate prey of these fishes and larger invertebrates (lobsters, shrimp) feed.

Florida Bay serves as a major nursery area for gray snapper, lane snapper, and schoolmaster. Juveniles are found primarily in mangrove and seagrass habitats of Florida Bay and Ten Thousand Islands regions of Everglades N.P. while adults of these species are mainly associated with reef habitats occurring with the

Florida's National Marine Sanctuary. For many of the species that inhabit Everglades N.P. for at least a portion of their lifecycle, spawning occurs outside the park along or offshore of the reef tract, the larval phase occurs in offshore, oceanic waters, and subsequent settlement by juvenile into their nursery habitat occurs within seagrass meadows and mangrove prop root habitats within Florida Bay and the Ten Thousand region.

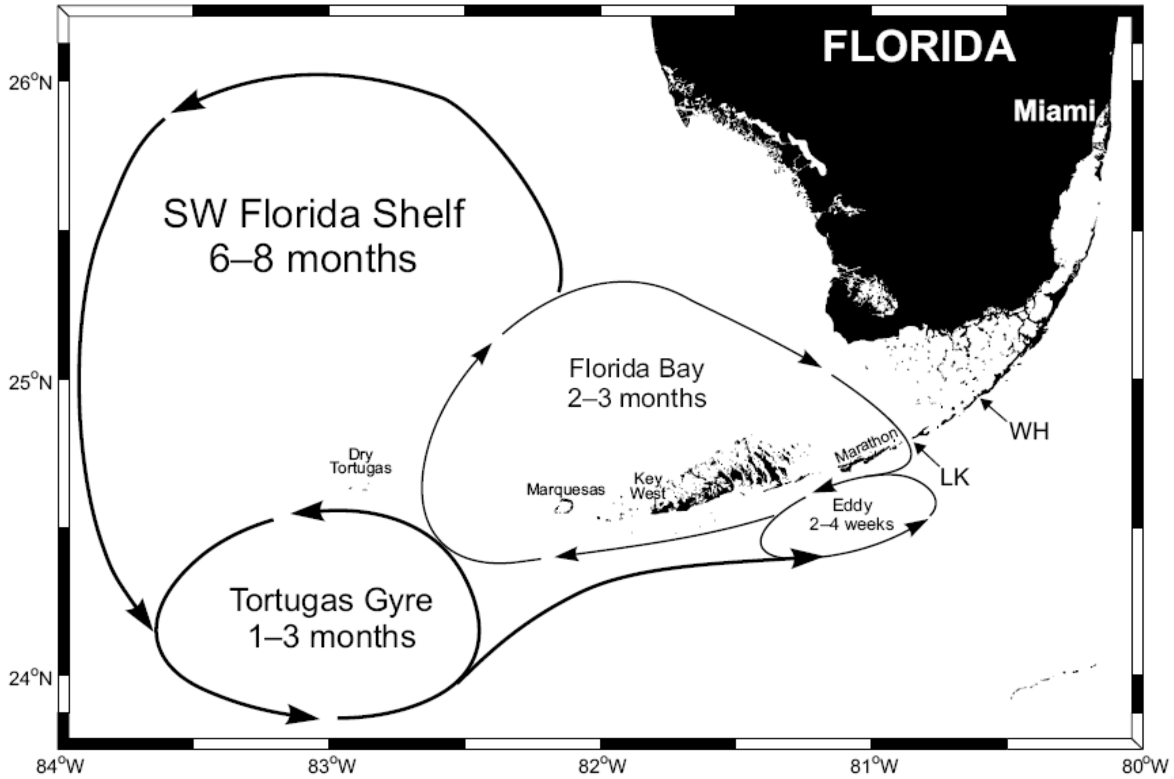


Figure 31. Schematic of possible recruitment pathways for pelagic marine larvae spawned locally in the Dry Tortugas and the Florida Keys (adapted from Fig. 7 of Lee and Williams 1999).

Marine Fishes: Oceanography & Larval Transport

Lee et al. (1992, 1994) have proposed a regional scale recirculation feature that links the Florida Keys inshore counter-current, the Tortugas Gyre, and SW Florida Shelf circulation patterns that could potentially retain larvae within South Florida for periods of several months. Such oceanographic linkage between the various marine ecosystems of South Florida may have been a necessary requirement enabling development and persistence of the marine biota now seen inhabiting the bay and reef tract today.

The Southeast Florida and Caribbean Recruitment (SEFCAR, NMFS--Univ. of Miami) project (1989-1995) made significant progress in understanding larval recruitment processes in South Florida through shipboard work describing and correlating circulation patterns and larval distributions in offshore waters of the Straits of Florida (Yeung and McGowan, 1991, Lee et al. 1992, Lee et al.1994, Cha et al. 1994, Diaz 1995, Limouzy-Paris et al. 1994, Criales and Lee 1995, Yeung 1996, Limouzy-Paris et al. 1997, Lee and Williams

1999, Yeung et al. 2000, Yeung et al. 2001). Results indicate that winds, Florida Current flow, and eddy-induced recirculation combine to influence the onshore transport and retention of larvae in the Keys coastal zone (Lee et al. 2002) and subsequent delivery of larvae to the park.

Oceanographic process occurring outside the park that directly influence larval supply to the park include variability in Loop Current/Florida Current system, the Tortugas Gyre, coastal eddies, and wind-driven onshore transport (Figure 31, Limouzy-Paris et al. 1997, Jones et al. 2001, Yeung et al. 2001, Criales et al. 2003, Criales et al. 2005). The effect of these processes on larval transport and the subsequent abundance and distribution of juvenile fish within the park must be addressed in order to determine the future effect of Everglades restoration efforts i.e. changes in freshwater inflow.

Low densities of juvenile predatory fishes (e.g., snappers, groupers, and drums) are found to occur in the northern and interior portions of Florida Bay in areas that appear to offer suitable nursery habitats for these fishes. It is suggested that one factor may be that these areas are too isolated from the offshore sources of larvae and that there is a limit of larval transport mechanisms *within* the bay for these areas to function as suitable nursery habitat for species having offshore pelagic larval stages (Dennis & Sulak 2001, Powell et al. 2002, Koenig et al. *in review*). Other areas within the park however are more open to the sea and to larval influx and additionally juveniles may migrate into these secluded areas after settling in these more accessible portions of the bay.

Marine Fish: Stressors

The major anthropogenic influences affecting changes within Everglades N.P. include: water management, pollution and nutrient input, construction, and fishing (Browder et al. 2003). Everglades National Park encompasses a region of south Florida that is immediately adjacent to the densely populated urban areas of Miami and the Florida Keys. Mainland areas adjacent to Florida Bay have been altered over the last two centuries by urbanization and the effects include loss of natural habitat, loss of natural water flow patterns and decline in water quality due to input of urban and agricultural waste products (Tilmant 1989, Schmidt et al. 2002). In addition this area has long been a popular fishing region and the impacts of overfishing of many species are now evident in the decreased numbers of large adult fishes of many species.

Marine Fish–Stressors–Fishing

Major recreational fisheries operate in the coastal waters of South Florida, but the economic value of the catch is not readily estimated, as it generates indirect, value-added revenue through support industries, e.g., contributing strongly to the economic base of the area as direct production ”value added”, and purchases generated in support industries (tourism, restaurants, fishing supply stores, and dive shops, etc. (Browder et al. 2003). Commercial fishing has been banned in Florida Bay in Everglades N.P. since 1985 after fishing data indicated the occurrence of overfishing and environmental perturbations. Commercial fishing was considered inconsistent with the purposes for which the National Park system was created.

Recent studies (Bohnsack et al. 2001, Ault et al. 2002a, Ault et al. 2002b) suggest that fishing affects reef fish populations, some of which spend part of their life cycle in Florida Bay. Decreases in the size and density of snapper and grouper populations and alterations in the trophic structure of reef fish populations inhabiting Florida National Marine Sanctuary has been attributed to increased fishing pressure (Ault et al. 1998, Bohnsack et al. 1999, Schmidt et al., 1999). Schmidt et al. (2001, 2002) analysis of the park's creel fishery data suggested that catch rates were positively correlated with fishing rates, at least for gray snapper, seatrout, and red drum and they concluded that there was no effect of fishing on overall abundance within

Everglades N.P. Assessments of fishing impacts usually consider fishing pressure and the resulting population effects within only a given area. None have specifically examined the effects of fishing in one location on populations of fish in other adjacent regions or considered areas/habitats separately (e.g. Florida Bay vs. offshore reefs). Because of ontogenetic migrations (e.g. bay to reef) and transitions in habitat use these effects are an important yet overlooked issue. This may explain discrepancies between these different findings.

Marine Fish*: *Stressors–Perturbations of Water Quality

Fish kills occur periodically within Florida Bay due to low levels of dissolved oxygen in the water (Schmidt 1993, Schmidt and Robblee 1994). These mass mortalities usually occur in the Snake Bight area (the northern interior Bay near Flamingo) where extremely shallow water, high summer temperatures, poor tidal and wind mixing, and high respiration rates result in low levels of dissolved oxygen. The C-111 canal is the major canal carrying fresh water into Florida Bay. Pesticide contaminants in this canal exceed current water quality standards and originate from the extensive agricultural drainage that supplies this canal (Browder et al. 2003). Restoration-associated alteration of C-111 canal flow to allow increased freshwater flow into the adjacent Everglades may result in an increase in the discharge of pesticides into Everglades N.P. and Florida Bay.

Elevated levels of mercury in fishes have been observed in the eastern region of Florida Bay where health advisories are posted warning against consumption of game fishes. Thirty percent of spotted seatrout from this region exceed the state's no consumption advisory level (Evans & Crumley *in review*). Other species of fishes have levels many times higher than those found in other parts of the bay or even the state. Consumption of these fishes put humans and wildlife at risk. Changes in the water management practices for Florida Bay through the Everglades restoration project by increasing freshwater input could alter the existing high levels of mercury in some species of fishes, changing the productivity and structure of the food web and influencing the bioaccumulation of mercury (Browder et al. 2003).

Marine Fish*: *Representative Fish Species

We chose three fish species to represent Everglades National Park for this assessment: goliath grouper (*Epinephalus itajara*), spotted seatrout (*Cynoscion nebulosus*), and gray snapper (*Lutjanus griseus*). This is an important trio of game fishes in Florida Bay currently (Rutherford et al. 1983), or has been historically (e.g., goliath grouper). Game fishes are of major ecological and economic value in South Florida and rely heavily on nursery grounds located within the park which supports several highly sought-after game fish, including red drum, spotted seatrout, and gray snapper (Schmidt et al. 2001, Schmidt et al. 2002). The gray snapper and spotted seatrout rank first and second, respectively, in the numbers of fish harvested in Everglades National Park (Rutherford et al. 1989). Despite its protected status, goliath grouper are also targeted by recreational anglers—indeed, 10 of the last 16 issues of *Florida Sportsman* magazine (Jan 2004-April 2005) have showcased successful catches of this species. Much of the use of Florida's national parks by visitors consists of recreational fishing for these species or diving and snorkeling to see them.

Goliath grouper, spotted seatrout, and gray snapper are dependent on habitats found within Everglades N.P. The spotted seatrout is unique in that it spends its entire life history within Florida Bay (Rutherford et al. 1989). Although goliath grouper and gray snapper adults are found outside of the estuary, their early stages depend upon the mangrove and seagrass environments within the Ten Thousand Islands and Florida Bay sections of Everglades N.P. as a nursery. In fact, Florida and Biscayne Bays appear to be the primary nursery habitat for gray snapper and the backwater areas of Florida Bay Ten Thousand Islands for goliath grouper. Everglades N.P. encompasses some of the most extensive areas of mangrove development in the South Florida region. These mangrove areas appear to be important for the survival of juvenile goliath grouper and the holes and caves formed by undercuts at the edges of mangrove islands provide important shelter for all life stages (Sadovy and Eklund 1999).

These three species represent a range of life history strategies and thus are susceptible to different pressures (fishing, environmental, habitat degradation) which make them good indicators of the effects of fishing pressure, habitat loss, environmental variability and the effects of other stressors within the park.

Goliath Grouper

Goliath Grouper: General Description

Goliath grouper is a the serranid formerly known as the Jewfish and is the largest reef fish with sizes recorded up to 216 cm and 318 kg and maximum recorded age of 37 years (Figure 32). It is probable that these fish reached larger sizes and longer ages but that the largest and oldest fishes have been removed from the population by fishing. They are long lived and reach maturity at over 5 years and 1100–1200 mm total length.

Adult goliath grouper display strong site fidelity and can be found in the same location year after year (Smith 1976, Koenig et al. 1997, Sadovy and Eklund 1999). These characteristics combined with their ambivalence towards divers have made them very vulnerable to fishing pressure and has led to their drastically reduced numbers (GMFMC 2005). Goliath grouper reached a point of being rare, but appear to be making a comeback following additional protection (Tilmant 2005). They are considered to be commercially extinct throughout significant parts of their geographic range (Sadovy and Eklund 1999).

Figure 32. Goliath Grouper. Source: NOAA 2004. Photo Credit: Don De Maria.

Goliath Grouper: Habitat

This species was once common in Florida and in Parts of the Gulf of Mexico (Randall 1996). Adults inhabit shallow inshore and offshore waters (usually < 40 m) (Sadovy and Eklund 1999) though they can occur in deeper water (Carpenter and Nelson 1971, Zinkowski 1971). They have not been observed at depths > 50 m around Florida (Sadovy and Eklund 1999). Adults are usually found in caves, holes, and undercuts (Nagelkerken 1981) or associated with high relief structures such as wrecks and pilings from bridges and docks (Smith 1971, Bullock and Smith 1991).

Juveniles have been collected from inshore, shallow habitats such as mangrove swamps along shallow mangrove shorelines, underneath mangrove prop roots (USDOC 2005), around bulkheads and bridges, in seagrass beds, and in poorly oxygenated canals (Springer and Woodburn 1960, Tabb and Manning 1961, Lindall et al. 1975, Thompson and Monro 1978, Bullock and Smith 1991, Sadovy and Eklund 1999) and below undercut ledges in swift tidal creeks draining mangrove swamps (Bullock et al. 1992). Areas with extensive mangrove development seem to be particularly important nursery areas for young goliath grouper and may limit their distribution. It is believed that the major factor in limiting abundance of juvenile goliath grouper is lack of suitable mangrove habitat. Abundance of juveniles is much higher in the Ten Thousand Islands mangrove island habitat (estimated at 54,000) than in Florida Bay (1,115) and is attributed to the more extensive mangrove development in the Ten Thousand area (Koenig et al. *in review*).

Goliath Grouper: Life History

Goliath grouper form spawning aggregations and return to these locations year after year. Individuals reach sexual maturity at a size of 1150-1350 mmTL (Sadovy and Eklund 1999) but it is larger fish of 68-136 mmTL that have been observed in these aggregations (GMFMC 2005). In southwest Florida these aggregations form in August and September on shipwrecks in 23–40 m depths of water (Sadovy and Eklund 1999). Of the aggregations that have historically been reported, most no longer are seen or consist of only 2–3 individuals (Sadovy and Eklund 1999). Of the few documented goliath grouper aggregations, one quarter no longer form, and the large historical aggregations have not been reported in the Atlantic for many years.

Larval goliath grouper settle out of the plankton but little is known about their abundance or the route they take into their juvenile habitat. Seventeen recently settled juveniles were collected from the Wood River in the Ten Thousand Islands area between October 1 and October 23, 2004 by the researchers at NOAA--SEFSC. These juveniles were 15–51.5 mm SL. The smallest individuals had very little pigment indicating size at settlement is probably around 15–16 mm SL.

Adult goliath grouper feed primarily on crustaceans, including pink shrimp (*Penaeus duorarus*), and xanthid crabs, spiny lobster (*Panulirus argus*), slipper lobster (*Scyllarides aequinoctialis*) and crabs such as *Calappa flammea*, *Menippe mercenaria*, *Ovalipes floridanus*, *Callinectes* sp., *Hepatus* sp. (Longley and Hildebrand 1941, Erdman 1957, Odum 1971, Randall 1996, Bullock and Smith 1991) though fish such as *Chaetodipterus faber*, *Lactophrys quadicornis*, *Etrumeus teres*, *Chilomycterus schoepfi*, *Dasyatis americana* and *Diodon* sp. and octopus, gastropod (*Fasciolaria tulipa*) and turtle parts have also been identified from stomach contents (Randall 1996, Bullock and Smith 1991). Recent studies of stomach contents of fish from the Ten Thousand Islands contain mostly crabs and fish and few lobsters.

Large adults of the species likely have few natural predators but smaller individuals and juveniles may be preyed upon by sharks such as the sandbar shark (*Carcharhinus plumbeus*) and the great hammerhead shark (*Spyrna mokarran*) and large fish such as barracuda, king mackerel, moray eels, large snappers and other groupers (UF 2005). While predators are known, there is no information on the natural mortality rates in goliath grouper.

Goliath Grouper: Environmental Stressors

Salinity, Temperature, Dissolved Oxygen: The goliath grouper is one of the few groupers that can live in brackish water, and has been captured in areas of low dissolved oxygen such as upland canals (Lindall et al. 1975, Sadovy and Eklund 1999). Goliath grouper does appear to be susceptible to exposure to low temperatures, however, at least to those below 13C (Gilmore et al. 1978). A negative correlation was found between abundance of juveniles and a combination of low salinity and low dissolved oxygen in the rivers and mangrove areas in the Ten Thousand Islands region (Koenig et al. *in review*).

Pollution: Little is known about the effects of pollution on the goliath grouper, but high levels of toxic substances can accumulate in the tissues of these top predators. Average levels of mercury concentrations exceeding the U.S. Food and Drug Administrations action level of 1 ppm methyl mercury were reported from several fish caught in southwest Florida and the Florida Keys between 1989 and 1991 (Sadovy and Eklund 1999). It has been suggested that relatively low density of goliath grouper juveniles in Florida Bay (vs. Ten Thousand Islands) may be due to lower water quality in Florida Bay as a result of anthropogenic influence (Koenig et al. *in review*).

Other: Goliath grouper, as many other large fishes can be host to parasites including trematodes, nematodes, isopods and copepods. It is not known at what level these infestations can affect the condition of the fish. Goliath grouper appear to be susceptible to the toxins released from red tide organisms. Smith (1976) recorded a fish-kill of goliath grouper associated with a red tide in 1971 that resulted in many dead individuals > 45 kg.

Goliath Grouper: Fishing

Pressure: Historically, the majority of the U.S. commercial catch had been landed along the Florida Gulf coast, where from 1979–1988, commercial catches increased from 15,454 kg to 61, 818 kg and then declined drastically. Although 5,000–17,000 fish were once taken recreationally each year, by the early 1990s, < 5,000 were caught and released annually (FMRI unpublished data) (Bullock et al. 1992). Before closure of the fishery in the U.S. in 1990, the south Gulf counties of Florida (i.e., Monroe, Collier, Charlotte, and Lee) accounted for 78% of the total Gulf landings, and all of Florida accounted for 99% of the Gulf landings for this species (GMFMC). Considering the short time between the most intensive fishing effort and the drastic decline in the stock, it is evident that goliath grouper populations can be very easily decimated within a short time (DeMaria 1996). A negative correlation was found between fishing pressure and juvenile abundance within Everglades N.P. (Cass-Calay and Schmidt *in review*). Spawning stock biomass per recruit has been estimated at close to 1%, far below the recommended minimum of 30% (Sadovy and Eklund 1999).

NMFS–SEFSC Reef Team visual point counts or predator searches, from 1979–1994, from Biscayne National Park to the Dry Tortugas, Florida, off the Florida Keys failed to observe a single jewfish (SEFSC). Jewfish were seen on only two of the 204 Reef Environmental Education Foundation (R.E.E.F.) dives logged in the Dry Tortugas, Florida, and none at the other R.E.E.F. census locations (Sadovy and Eklund, 1999). However, a recent study using tagging and recapture of juveniles (<1m SL) indicates that they are becoming more abundant especially in the Ten Thousand Islands portion of Everglades N.P. (Koenig et al. *in review*).

Protection: Goliath grouper spawning aggregations have historically been targeted by fishermen as they were a predictable occurrence of large numbers of fish. Targeting these aggregations has caused the reduction in numbers of fish of many species including the goliath grouper. This aggregating behavior for reproduction, together with the characteristics of slow growth, long time to maturity and site specificity has made the species exceedingly vulnerable to fishing pressure (Bannerot et al. 1987, Polovina and Ralston 1987, Sadovy and Eklund 1999). In addition, their dependence on geographically limited areas, such as mangrove covered shorelines for nursery habitat, makes the species vulnerable to key habitat degradation. Furthermore, their characteristic limited spawning periods and possibly narrow recruitment window have made the species particularly vulnerable to recruitment variability and susceptible to recruitment failure in poor years due to low numbers of spawning stock, making time to species stock recovery even longer (Sadovy and Eklund 1999).

Goliath Grouper: Current Status

Jewfish have been on the candidate list for possible listing as threatened or endangered under the Endangered Species Act since 1991 (Sadovy and Eklund 1999). In 1996, they were included on the "Red List" of the

IUCN–World Convseration Union. There is a moratorium now in effect in all waters of the U.S. and Bermuda (e.g. fishers are not allowed to keep any goliath grouper).

Recoveries in numbers of goliath grouper have already occurred from protection, as some researchers have noted that the most frequently encountered age in goliath grouper from surveys was around 12 years in 2002. This age exactly coincides with the closure of the fishery. It is estimated that it could be another 20–23 years before full population recovery occurs.

Sightings of goliath grouper are being used as criteria for success in the restoration of the Southern Florida ecosystem (Southern Florida Management and Coordination Working Group) based on the fact that, historically, the species was an abundant predator along the Everglades Ten Thousand Island region (Sadovy and Eklund 1999). Aggregations on four spawning sites off southwest Florida have been tracked since the 1980s. Since the closure of the fishery in 1990, these sites have shown some increase in numbers, but the number of individuals at each of the sites remains at half or less than half of the numbers observed in 1982. However, at the present time, there is no hard evidence that the protection measures in place for goliath groupers have brought about a change in the numbers of adult goliath grouper within Everglades N.P. (R.E.E.F. surveys). There is anecdotal evidence that their numbers are increasing and especially the number of juveniles observed within Florida Bay and surrounding areas (Koenig et al. *in review*).

The main center of research on goliath grouper in Florida is through the NOAA Fisheries Service Reef Research Team at the Southeast Fisheries Science Center (USDOC 2005). Recent research on goliath grouper in Florida has included studies of the distribution of goliath grouper, site fidelity, diet, and juvenile jewfish biology including their distribution and habitat use. This research is currently concentrated off southwest Florida, primarily in the Ten Thousand Islands area but also includes the Florida Keys and the Dry Tortugas. Jewfish nursery habitat has been identified around mangrove islands, and juvenile jewfish tagging studies are being used to estimate abundance and to eventually develop a recruitment index (GMFMC 2005). Within the Ten Thousand Islands area, NOAA Fisheries Service has divided its research into four discrete areas. Within each of these areas, 18 to 26 jewfish have been observed. Based on the hook and line catch rates of juvenile jewfish caught for tagging purposes, NOAA estimates that, if these areas were open to harvest, it would take just 1300 hook hours in each area to harvest all of the jewfish. If there is a recovery occurring, it is only just beginning and lifting the moratorium on goliath grouper any time soon is not recommended.

Spotted Seatrout

Spotted Seatrout: General Description

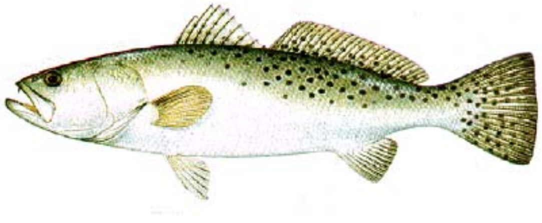


Figure 33. Spotted Seatrout. Source: FMRI 2003b.

The spotted seatrout (Figure 33) is one of the two most popular gamefish in Everglades National Park and ranks second in numbers harvested in the Everglades N.P. sport fishery (Rutherford et al. 1989). Landings of spotted seatrout totaled 1,144,304 kg during 2001 in Florida. Since 1996 the fishery consists almost entirely of recreational fishers (>95% by weight) (FFWCC 2003). The Bay serves as both a nursery for the juveniles and a sportfishing ground for the adult as this species spends its entire life history within Florida Bay (Rutherford et al. 1989). Males of this species reach a maximum age of 9 years and females an age of 8 years (FFWCC, 2003) and begin spawning at 0–2 years.

Spotted Seatrout: Habitat

Spotted seatrout appear to spend their entire lives within a single bay or estuary, utilizing the estuarine environment for feeding, spawning, and nursery habitat (Tabb 1966, Baker and Matlock 1993). Adults are found in waters that vary from brackish to hypersaline (0.2–70 ppt), but spawning occurs in waters from 20–37 ppt and eggs and larvae have been found in estuarine waters from 15–50 ppt. (Lassuy 1983, Holt and Holt 2003). Salinity does not appear to be a significant factor in the distribution of small juveniles (20–200 mm) in Florida Bay (Chester and Thayer 1990).

Spotted seatrout are distributed over a wide geographic area, but it is believed that individuals remain primarily in their natal waters throughout their life. Evidence from tagging studies shows that 95% of recaptured seatrout in Florida had moved less than 48 km from their release point, suggesting that they do not migrate even between adjacent estuaries (Iverson and Tabb 1962). Furthermore, evidence based on a study of spotted seatrout DNA (FMRI unpublished data), shows that there are three genetically distinct groups in Florida: an Atlantic coast group south through Biscayne Bay, a gulf coast group from Biscayne Bay north to Apalachee Bay (includes all of Everglades N.P. and Florida Bay) , and another gulf coast group from Apalichocola westward. These findings suggest that each region has its own distinct stock and that the fish from these different areas do not intermix to any great degree and have not for a long period of time.

Juveniles primarily inhabit seagrass beds (Moody 1950, Reid 1954, Tabb and Manning 1961). In a study in Florida, more young-of-the-year were caught over beds of *Thalassia* and *Halodule* marine grasses than over bare bottom. In western Florida Bay, they were most common in *Syringodium* beds (Rutherford et al. 1989). Proximity to mangroves also appears to be an important factor in the distribution of juvenile seatrout and may serve as a source of detrital material which forms the basis of the food web for seatrout and other fishes (Nelson and Leffler 2001). In bays with a large range of salinity, juvenile distribution appears to be negatively correlated with salinity (Nelson and Leffler 2001). This may also be true for adults (Helser et al. 1993).

Larvae are collected over a wide variety of bottom types at water temperatures between 20–35 °C and salinities 12–41 ppt. In a 1989–1999 study, high densities of larvae were consistently found in Whipray Basin (located in the central portion of Florida Bay). This appears to be a valuable juvenile nursery area despite low seagrass above-ground standing crop and occasional hypersaline conditions (Powell et al. 2001, Powell et al. 2002, Powell et al. 2004). Early juveniles (9–30 mm) occur in shallow and deep water, but most occur in shallow areas probably due to the location of the seagrass beds which provide shelter as well as food (McMichael and Peters 1989).

Spotted Seatrout: Life History

Spotted seatrout reach maturity at a young age, and first spawn between 0 and 2 yrs of age at a size of 11.8–15.7 in TL5. Central Florida Bay is believed to be a major spawning ground for spotted seatrout (Powell et al. 2004). Spawning has been documented in western Florida Bay primarily during the summer with peaks in May and June, though Jannke (1971) suggested that spawning occurs throughout the year in Everglades N.P. Adults are believed to spawn during the night in deep channels and depressions near grass flats when temperatures are >21°C (Tabb 1966, Helser et al. 1993).

Larvae are distributed over a variety of bottom types but appear to recruit primarily to areas with dense seagrass beds (McMichael and Peters 1989, Powell et al. 2001, Powell et al. 2002). Growth appears to be faster in juveniles in Florida Bay than in other areas (e.g., Tampa Bay) and juveniles of spotted seatrout are, in fact, most common in western Florida Bay (Chester and Thayer 1990) where seagrass beds are significantly denser (Iverson and Bittaker 1986). These beds are perhaps providing a higher abundance of invertebrate prey on which the juveniles depend as a food source (Holmquist et al. 1989, Hettler 1989, McMichael and Peters 1989, Matheson et al. 1999). In addition, warmer water temperatures have been observed in Florida Bay (Boyer et al. 1999) compared to Tampa Bay (McMichael and Peters 1989). These warmer temperatures are believed to work in conjunction with food availability to enhance growth in Florida Bay (Warren 1971).

The smallest juveniles (8–15mm) feed primarily on copepods, switching to caridean shrimp, amphipods and mysids, penaeid shrimp and finally fish in juveniles > 15mm. Abundant stocks of penaeid shrimp appear to be necessary to support large populations of juvenile spotted seatrout and made up large percentages by number and volume in all size classes > 15mm (Hettler 1989, McMichael and Peters 1989).

Spotted Seatrout: Environmental Stressors

Salinity, Temperature, Dissolved Oxygen: Powell et al. (2004) examined spatial and temporal variation in growth and survival of spotted seatrout. A strong parabolic relationship was found between temperature and growth in juveniles (22–60 d), but no significant relationship was found for larval growth. They could not look at variation of growth with salinity because of inappropriate distribution of samples. Lab experiments were conducted by Wuenschel (2002) examining the effect of temperature and salinity on routine metabolic rate of spotted seatrout. A significant interaction of temperature and salinity on metabolic rate was found.

Though field studies of the effect of salinity changes on spotted seatrout are limited, there is little evidence that salinity plays a major role in regulating growth or general condition of larvae or juveniles within areas where the fish are normally found. The site specific behavior exhibited by seatrout may allow the population to adapt to local salinity conditions. In addition, there is evidence for a genetic basis to the reproductive response (including egg diameter, tolerance of eggs and larvae to differing levels of salinity and differential buoyancy) and of spotted seatrout populations living in extreme-salinity environments to be successful. The salinity of the water where spawning occurs appears to direct these adaptations and allows the spotted seatrout to produce eggs and larvae that thrive at a variety of salinity levels (Holt and Holt 2003). It is not surprising that an estuarine species should have evolved to cope with fluctuations in salinity levels and the ability of this species to produce viable larvae over a 20 ppt salinity range demonstrates the adaptability of this species to a wide range of salinity environments.

The spotted seatrout is not without limits to salinity tolerance. Serafy et al. (1997) found spotted seatrout to be one of the least tolerant (among 10 species of estuarine fishes) of rapid salinity changes. The amount of fresh-water runoff is inversely related to annual average spotted seatrout catch rates in Everglades N.P. (Schmidt et al. 2001). Regarding salinity effects of young spotted seatrout, Banks et al. (1991) demonstrated that tolerance to salinity changes was at its minimum in 3-day-old larvae and time to 90% yolk depletion was significantly reduced for larvae spawned and raised in lower salinity regardless of origin (Kucera et al. 2002). Furthermore, at salinities > 45 ppt, survival and growth of larval and juvenile spotted seatrout could be diminished even in otherwise highly favorable areas (Wuenschel 2002). During low rainfall periods, salinities in the north central portion of the bay can exceed 45 ppt (Orlando et al. 1997, Boyer et al. 1999). The frequency and duration of hypersaline events in the Central Bay might, therefore, affect survival and growth of young seatrout and, consequently, the abundance of this species (Browder et al. 2003). Powell (2002) noted that densities of the postlarvae of many species collected with an epibenthic sled were higher in the Central Bay, so many species may be disproportionately exposed to hypersaline conditions in an otherwise favorable nursery area.

Larval concentrations suggest the central part of the bay is a major spawning ground (Powell et al. 2004). Juvenile seatrout appear to extend their range into the central portion of the bay when hypersaline conditions were absent (Thayer et al. 1999). Powell (2002), citing other published works, noted that seatrout eggs do not float at salinities < 0–15ppt; therefore egg survival might be virtually zero percent at these salinities. Lab experiments suggests that seatrout do not spawn at salinities > 30 ppt. (Wakeman and Wolschlag 1977, Taniguchi 1980).

Pollution: There have been few studies of the effects of run-off of pollutants into the bay on spotted seatrout. Johnson et al. (1977) found that considerable loss of larval seatrout can be expected in areas of chlorinated effluent disposal where the toxic products of sodium hypochlorite and seawater are above 0.17 ppm of sodium hypochlorite. The lethal levels for two additional toxic products of chlorine degradation were measured: chloramines, and 5-chlorouracil. Though the exact action of 5-chlorouracil is not known, it was found to cause mortality and anatomical malformations in larvae and at concentrations greater than 0.5-1.0 mg/liter reduced hatching and survival of spotted seatrout larvae in laboratory experiments.

Petroleum hydrocarbons may also enter into Florida Bay through spills from boats or from street run-off. Exposure to these chemicals has been found to cause decreased condition in fish including disorientation, weight loss, histopathological alterations and increase in intensity of trichodinid infestations (Blaylock and Overstreet 2003).

Input of a variety of chemical substances can result from run-off from the extensive agricultural areas adjacent to Florida Bay. These products can result in oxygen depletion by using available oxygen for degradation. In addition they can stimulate the growth of algae which in turn require oxygen further depleting the waters and leading to anoxic conditions (Blaylock and Overstreet 2003) that can lead to massive die-offs of fish. Laboratory studies have shown that there are a variety of organochlorine pesticides that have been found to accumulate in the tissues of seatrout and can cause liver, gill, spleen, intestine, kidney, skin, gonad and brain abnormalities (Meyers and Hendricks 1982). In a study on a related seatrout, *Cynoscion regalis*, pesticide residues such as chlordane, polychlorinated biphenyls (PCBs), and DDT continued to accumulate in the flesh of fish long their use was restricted (Kennish and Ruppel 1996, Kennish and Ruppel 1998) and it was found that fish with DDT residues as high as 8 ppm in the gonads failed to breed for as long as 2 years (Butler 1969, Butler et al. 1970).

Spotted Seatrout: Fishing

Pressure: In 2001 total landings of spotted seatrout in the state of Florida **reached** 1,144,304 **kg. Ninety** eight percent by weight of the total statewide landings was through the recreational fishery. Since 1996, the fishery has changed from a mixed-sector fishery, with 20% commercial landings, to an almost exclusive recreational fishery (>95% by weight) (FFWCC 2003). Recreational landings made during 2001 were high in all regions except for the Northeast, panhandle and Southeast regions of Florida (FFWCC 2003).

Current estimates show that in most regions spotted seatrout are being overfished (Murphy 2003). Only in the northeast management region of Florida were 2001 fishing rates low enough to achieve and maintain the commission's 35% Spawning Potential Ratio (SPR) target. In other regions, static SPRs under 2001 fishing mortality rates range from 16–29% (Murphy 2003). Results from the 2003 assessment indicate that recruitment of age-0 spotted seatrout appears to be declining in the southwest and northeast regions, possibly increasing in the southeast and fluctuating without trend in the northwest region. However, though estimates of average fishing mortality have declined since 1990, they have not declined in the southeast region since the mid 1990's.

Gray Snapper

Gray Snapper: General Biology

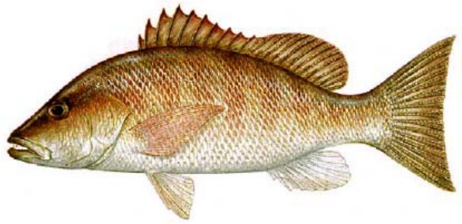


Figure 34. Gray Snapper. Source: FMRI 2003c.

Populations of adult gray snapper (Figure 34) are known from southeastern Brazil, the Caribbean, and the Bahamas to the northern Gulf of Mexico, Bermuda, and North Carolina (Grimes et al. 1982, Robins and Ray 1986, Smith 1997). The subtropical marine ecosystems of South Florida form a prime habitat for an extensive population of this species where adults occur primarily along the reef tract and juveniles utilize nursery grounds within coastal bays, including Florida Bay. Adults are believed to recruit to the reef from sea grass and mangrove nursery habitats of Florida Bay where they may spend their juvenile phase before migrating to the coral reefs as young adults. The source of larval recruits and extent to which population replenishment relies on upstream populations (e.g., Cuba and the Yucatan) is unknown.

Gray Snapper: Habitat

For gray snapper, Florida Bay serves primarily as a juvenile nursery habitat. While mature fish are known to occur within the bay (e.g. Faunce 2002), spawning takes place elsewhere as ripe and running adults are rarely found within the bay (Croker 1962, Rutherford et al. 1983). In South Florida gray snapper migrate to offshore areas and spawn during the new moon periods from June to September (Starck 1970, Rutherford et al. 1989, Domeier and Colin 1997). The waters off Alligator Reef (Starck 1970) and those near reefs off Marathon (Rutherford et al. 1989) and in the Middle Keys are important known pawning grounds for gray snapper. The resulting offspring have an offshore pelagic larval phase which has been described for laboratory-reared (Richards and Saksena 1980) and wild-caught specimens (Richards et al. 1994).

Gray Snapper: Early Life History

Very little is known of the biology and ecology of larvae and early juveniles of this species. This is a critical life history stage that is highly susceptible to predation and must undergo drastic developmental, trophic, and habitat transition. The relative survival success of these stages undoubtedly affects subsequent year class strengths and ultimately spawning success.

The pelagic larval phase of snappers recruiting to Florida Bay ranges from ca. 36–42 days (Jones et al. 2001). Evidence of the wide dispersal potential of this stage is supported by the presence of larvae within bay inlets of North Carolina (Hettler and Barker 1993) and the occurrence of juveniles as far north as Massachusetts (Sumner et al. 1911). Gray snapper in such northern extents have presumably been expatriated by the Gulf Stream and do not survive the winter.

Previous attempts to collect snapper larvae within the bay have resulted in little success. No gray snapper larvae were collected in ichthyoplankton sampling in Florida Bay conducted by Rutherford et al. (1989). Only a few specimens were collected by Powell et al. (1989b) or Collins and Finucane (1984), the youngest of which were actually external to the bay. Previous SFERPM work indicates only the latest larval stages were found to enter Florida Bay from offshore Atlantic waters through inter-island channels in the Florida Keys (Jones et al. 2001). Late stage larvae may also enter the bay from the Gulf across the southwestern Florida Shelf, but this transport scenario has yet to be sufficiently investigated.

At 10–12 mm SL (Starck 1970, Richards and Saksena 1980) gray snapper settle from the plankton into seagrass meadows in Florida Bay (Starck 1970, Chester and Thayer 1990). Early juveniles are solitary, diurnal feeders that exhibit high site fidelity at this stage. The seagrass beds offer shelter from predation and provide a rich source of small invertebrate prey. After reaching ca. 80 mm SL, fish initiate ontogenetic migrations and aggregate in mangrove prop root habitats and channel edges which offer larger structure for shelter (Starck 1970).

The lack of effective transport processes linking offshore Atlantic and Gulf waters with potential settlement sites in interior regions of Florida Bay might isolate otherwise suitable nursery grounds (Dennis and Sulak 2001, Powell et al. 2002) thus limiting the distribution of juveniles.

Gray Snapper: Juvenile Nursery

Settle (2003) examined juveniles of a closely related species (lane snapper, *Lutjanus synagris*) in Florida Bay. Cohort-specific growth rates (30–242 d) showed significant intra- and inter-annual variation. This species primarily inhabits the Gulf Transition and Western sub-regions. Growth rates ranged from 0.59–0.93 mm/day. Snappers spawned during the spring and summer grew faster than those spawned during fall and winter. Fish spawned during 1998 grew faster than those spawned in 1997.

Gray Snapper–Age & Growth

A number of studies have examined age and growth of snappers in the region of South Florida. Initially, scale-based techniques were employed in estimating age and determining growth rates of gray snapper from the Florida Keys and Florida Bay (Croker 1962, Starck 1970). Based on scale-derived ageing, Rutherford et al. (1983) concluded that gray snapper in South Florida displayed growth rates of 9 mm SL · month−1 and that ontogenetic migrations to offshore adult habitats occur at 3–4 years of age. Unfortunately, many life history parameters cannot be reliably estimated from scales as these structures are often replaced after loss, are frequently disfigured, and rarely validated for reliability.

More recently, otolith-based methods have been employed as a more reliable alternative to scalebased work. Unfortunately, age (Manooch and Matheson 1981, Johnson et al. 1994) and growth (Johnson et al. 1994) were estimated from otolith annuli prior to validation of annular deposition growth increments. Early works relying on otolith annuli were probably ripe with inaccuracies, most likely the result of misinterpretations of subannual growth rings as annual (Claro and Garcia-Arteaga 1994). Marginal increment analysis has only very recently been used to actually validate the annular deposition of growth increments of otoliths in gray snapper from Florida (Burton 2001). In addition, otolith marking methods have been used to validate daily increment deposition in otoliths of gray (Ahrenholz 2000, Allman and Grimes 2002) and lane snapper (Ahrenholz 2000). Daily increment deposition of yellowtail snapper have been validated through laboratory rearing (Lindeman 1997).

Gray Snapper: Otolith Microchemistry

Lara et al. (*in review*) traced the ontogenetic movements of juvenile snappers from the nursery grounds in Florida Bay to adult habitats along the offshore reef tract of the Straits of Florida using otolith microchemistry as natural tags. This study is based on the presence of trace metals and rare earth elements in the otoliths. Preliminary results define a unique suite of elements that distinguish fish residing in Florida Bay as juveniles from fish from other regions (i.e., Biscayne Bay, Dry Tortugas, Lower Florida Keys, and Ten Thousand Islands).

Gray Snapper: Fishing

Gray snapper comprise a large proportion of the commercial and recreational finfish harvest in the state of Florida. Landings in Florida from commercial harvesting from 1998–2001 averaged 158,757 kg · yr−1 with a value of $635,000 dollars in 2001(USDOC 2003). Recreational fishing was responsible for landings of 518,909 - 753,417 kg · yr−1 between 1998 and 2002 (USDOC 2003). The economic value of the recreational component of the catch is not readily estimated, as it generates indirect, value-added revenue through support industries, e.g., tourism, restaurants, fishing supply and boating stores, dive shops, etc. (Browder et al. 2003).

In Florida Bay, gray snapper are one of the most popular game fish comprising the highest harvest in the recreational fishery. Due primarily to its extreme fighting ability, all sizes of these fishes are highly valued by anglers (Manooch 1984). The recreational fishing effort in South Florida is continually expanding as the greatest number of reported fishing trips to Florida Bay alone reached an all time high in 2001 (Schmidt et al. 2002). Recreational fishing within Florida Bay along with recreational and commercial fishing along Florida's reef tract have already affected the size structure of snapper populations (Ault et al. 1998, Faunce et al. 2002). This intensive fishing combined with high rates of other sources of mortality have resulted in more than half of the species of snappers in south Florida being overfished (Ault et al. 1998).

Gray Snapper: Coastal Processes & Florida Bay Recruitment

Jones et al. (2001) back-calculated birth dates of snapper larvae entering Florida Bay through inter-island channels using daily otolith increment counts. They found that larval duration during the first year varied across species and ranged from 35.50 to 41.45 days. The peak in influx and age of the larvae together indicate spawning occurred around the time of the full moon and during the period when spawning aggregations are known to occur off the Dry Tortugas. Peak abundance of snapper larvae in the channels occurred in the summer of 1997. This was coincident with a well developed Loop Current and the presence of a stationary eddy off the Dry Tortugas, where snapper spawning aggregations occur. In contrast, during the second year the abundances of snapper larvae were lower, the Loop Current was poorly developed, and no eddy occurred during the spawning period. It is unclear whether Loop Current development during the 1997–1998 period was affected by the strong 1997–1998 ENSO event. A longer data time series is necessary to capture such inter-annual variability.

Marine Fishes: Restoration

The South Florida Ecosystem Restoration Effort and the Comprehensive Everglades Restoration Project (CERP) call for restoring historic freshwater input to the Florida Everglades. These efforts will undoubtedly alter the physical and biological characteristics of Florida Bay in an as yet undetermined and unpredictable manner. In order to accurately assess and predict how these changes will affect important species dependent on the bay, researchers need to examine 1) the existing levels of natural variability in life-history parameters and 2) how this variability is influenced by environmental factors and biological interactions. Only then will scientists be able to assess and predict what effects restoration efforts will have on habitat characteristics and subsequent biological responses to these changes.

Baseline information is needed to allow for prediction of the effects of environmental and habitat alterations in Florida Bay which will result from efforts by the South Florida Water Management District (SFWMD) and the U.S. Army Core of Engineers to restore historic surface water flow to the Everglades. As directed by the South Florida Ecosystem Restoration Prediction and Modeling Program (SFERPM), research should be conducted to supply scientific facts essential to constructing realistic models and increasing their predictive abilities to ultimately guide management of Florida Bay and direct Everglades restoration efforts.

The restoration of historic surface water flow patterns to the Everglades may bring about several changes to the ecosystem including: 1) alteration of average salinities and salinity ranges where water flow patterns will be changed from the present levels, 2) possible increase in outflow of contaminants from urban and agricultural areas and, 3) habitat alteration due to changes in average freshwater flow.

The most important of the three changes mentioned above, for all of the species discussed here, is habitat alteration, as these species and many others are dependent on extensive seagrass beds and mangrove shorelines as nursery areas for the juveniles of their species. It needs to be determined what changes will occur and how extensive these changes may be as a result of changes in water flow patterns before predictions can be made about the effect on these and other species. It is not known if areas such as seagrass beds and mangroves will increase, decrease or shift position as a result of restoration efforts. This is a key problem that needs to be addressed.

Marine Fishes: Recommendations

All three of these species appear to have a high tolerance to variations in salinity and occur in waters with a wide variety of salinities. This is not unexpected as these are estuarine species and must be adapted to natural changes in salinity in the areas in which they exist. However, there are upper and lower limits of salinity that effect the growth and performance of these three species, as well as stages in their life cycles that are more vulnerable to wide fluctuations in salinity. The stress of high and low salinities may be compounded by extreme temperatures and other stresses (e.g. pollutants, competition for limited habitat) to affect fish detrimentally. In addition, the prey items of these fishes may be affected by changes in salinity and may result in changes in assemblages of prey items, changes in abundance or migration of these organisms to other areas. There are to date a few studies on the assemblages of species that inhabit the habitats used as nurseries in the bay (e.g. Ley et al. 1999, Ley and McIvor 2002, Sogard et al. 1989), but little discussion about how these assemblages may change with changes in salinity.

Most of the waters of Everglades N.P. are considered to be altered to some degree by pollutants of anthropogenic origin. Waters in adjacent areas near lower human populations (e.g. Ten Thousand Islands) are considered to be more pristine, and indeed appear to be more productive, as they have more extensive, healthy, natural marine habitats and more fish. It is possible that the increase in water flow from agricultural areas of South Florida will increase the input of pollutants into Everglades N.P. and Florida Bay. Even if this is not the case it appears that the levels of contaminants of anthropogenic origin in Florida Bay are presently high and it is believed that the effects of this can currently be seen in the lower water quality and decreased numbers of fish and invertebrates in the waters near the urbanized areas of Miami and the Florida Keys. Ultimately the relationship between natural and anthropogenically induced environmental variables and population success parameters needs to be addressed for important keystone fish species.

Because of the myriad and unknown effects that would result by restoring historic water flow to the Everglades, we caution that restoration efforts proceed slowly, initially in a limited area with directed research to determine the resulting effects before proceeding on a larger scale. In order to do this, baseline data on fish assemblages and abundances before restoration are needed followed by analysis after controlled changes in water flow to limited areas occurs.

Marine Fishes: Fishing Recommendations

Currently, all three species are considered to be overfished in Florida Waters (Ault et al. 1998, Sadovy and Eklund 1999, Murphy 2003). The Goliath grouper population is drastically reduced from historical numbers and should continue to be protected and we support its addition to the list of threatened or endangered species. Though it is currently prohibited to take goliath grouper from U.S. waters, adults may move into unprotected waters to spawn. The most effective protective measures, therefore, would be those that would ban the harvesting of adults at spawning aggregations, and would conserve juvenile and adult habitat. Moratoriums on fishing and the establishment of marine reserves which encompass the proper habitats can protect both juvenile and adult biomass (Sadovy and Eklund 1999).

Though juveniles of gray snapper appear to be abundant in Everglades N.P., the numbers of large snappers of any species on Florida's reefs are greatly reduced (Ault et al. 1998). We believe that increased protections may be required in order to increase numbers of adult snappers to long-term sustainable levels. Spotted seatrout is presently considered overfished in most of the state and is almost exclusively taken by recreational fishers (Murphy 2003, FFWCC 2003). Protective measures have changed often and are considerably recent and positive results in the form of increased numbers of fish have yet to be seen.

Even with the current protections, one of the major problems in Everglades N.P. and other parks and reserves in Florida are fishing violations by recreational fishers. The vast majority of fishing violations in Florida Bay involve the taking of undersized gray snapper (Rays 2005). We feel the only way to curtail this is through further support for raising public awareness and support of enforcement efforts. Finally, ontogenetic migrations and transition in habitat use by different life stages is rarely considered in studies looking at the effects of fishing pressure. Removal of fish from one location may have more impact on populations in other areas than what is revealed immediately at that location. We suggest that this be taken into consideration when effects of fishing are assessed.

References

Acosta, C.A. and M.J. Butler IV. 1999. Adaptive strategies that reduce predation on spiny lobster postlarvae during onshore transport. Limnology and Oceanography 44:494-501.

Acosta, C.A., T.R. Matthews, and M. J. Butler IV. 1997. Temporal patterns and transport processes in recruitment of spiny lobster, *Panulirus argus*, postlarvae to south Florida. Marine Biology 129: 79-85.

Acosta, C.A. and M. J. Butler IV. 1997. The role of mangrove habitat as nursery for juvenile spiny lobster, *Panulirus argus*, in Belize. Marine and Freshwater Research 48:721-728.

Ahrenholz, D. W. 2000. Periodicity of growth increment formation in otoliths of juvenile gray snapper (*Lutjanus griseus*) and lane snapper (*Lutjanus synagris*). J. Elisa Mitchell Sci. Soc. 116: 251–259.

Allen, R.P. 1942. *The Roseate Spoonbill*. Dover Publications, New York.

Allman, R. J. and C. B. Grimes. 2002. Temporal and spatial dynamics of spawning, settlement, and growth of gray snapper (*Lutjanus griseus*) from the West Florida shelf as determined from otolith microstructures. Fish. Bull. U.S. 100: 391–403.

Amador, J. A., G. H. Richany, and R. D. Jones. 1992. Factors affecting phosphate uptake by peat soils of the Florida Everglades. Soil Science 153:463-470.

Anderson, G., T., J. Smith and W. K. Nuttle. 1999. Restoration of sheetflow to northeast Florida Bay: an assessment of structural modifications to the C111 canal. Abstracts Florida Bay Science Conference 1999, Miami FL.

Andree, S.W. 1981. Locomotory activity patterns and food items of benthic postlarval spiny lobsters, *Panulirus argus*. MS Thesis, Florida State University.

Armentano, T.V., R.F. Doren, W.J. Platt, and T. Mullins. 1995. Effects of Hurricane Andrew on coastal and interior forests of southern Florida: overview and synthesis.J. Coast. Res. 21: 11-144.

Armentano, T.V. 1995. Analysis of pattern and possible causes of die-back of Florida Bay keys mangroves. Report to the U. S. Army Corps of Engineers. National Park Service, Everglades National Park.20 pp.

Atkeson, T. and D. Axelrad. 2004. Mercury monitoring, research and environmental assessment. Pages 2B-1/2B-28 In: *2004 Everglades Consolidated Report*. South Florida Water Management District, West Palm Beach, Florida.

Audubon. 2005. Spoonbill Banding Project. Audubon of Florida. .Ault, J. S., J. A. Bohnsack, and G. A. Meester. 1998. A retrospective (1979-1996) multispecies assessment of coral reef fish stocks in the Florida Keys. Fish. Bull. U.S. 96: 395–414.

Ault, J. S., S. G. Smith, G. A. Meester, J. Luo, E. C. Franklin, J. A. Bohnsack, D. E. Harper, D. B. McClellan, S. L. Miller, M. Chiappone, and D. W. Swanson. 2002a. Synoptic habitat and reef fish surveys support establishment of marine reserves in Dry Tortugas, Florida USA. Reef Encounter 31: 22–23.

Ault, J. S., S. G. Smith, J. Luo, G. A. Meester, J. A. Bohnsack, and S. L. Miller. 2002b. Baseline multispecies coral reef fish stock assessments for the Dry Tortugas. NOAA Tech. Mem. NMFSSEFSC-487.

Axelrad, D.M., T.D. Atkeson, C.D. Pollman, T. Lange, D.G. Rumbold and K. Weaver. 2005. Mercury monitoring, research and environmental assessment in south Florida. Pages 2B-1/2B-31 In: *2005 South Florida Environmental Report*. South Florida Water Management District, West Palm Beach, Florida.

Bach, S.D. and G. L. Beardsley (1976) A disease of the Florida spiny lobsters. Sea Frontiers 22: 52-53.

Baker, W. B. and G. C. Matlock. 1993. Movement of spotted seatrout tagged in Trinity Bay,Texas. Northeast Gulf Science 13: 29–34.

Banks, M. A., G. A. Holt, and J. M. Wakeman. 1991. Age-linked changes in salinity tolerance of larval spotted seatrout (*Cynoscion nebulosus*, Cuvier). J. Fish. Biol. 39: 505–514.

Bannerot, S. P., W. W. Fox Jr., and J. E. PowersPolovina, J. J. and S. Ralston Eds. 1987. *Reproductive strategies and the management of snappers and groupers in the Gulf of Mexico and Caribbean*.

Behringer, D.C. Jr. 2003. The ecological ramifications of disease and density on juvenile spiny lobster, *Panulirus argus*. Ph.D. Dissertation, Old Dominion University.

Berger, Dianne K. and Mark J. Butler IV. 2001. Do octopuses influence den selection by juvenile Caribbean spiny lobster? Marine and Freshwater Research 52: 1049- 1054.

Bittaker, H. F. and R. L. Iverson. 1976. Thalassia testudinum productivity: a field comparison of measurement methods. Mar. Biol. 37:39-46.

Bjork, R.D. and G.V.N. Powell. 1994. Relationships between hydrologic conditions and quality and quantity of foraging habitat for Roseate Spoonbills and other wading birds in the C-111 canal basin, final report to the south Florida research center, Everglades National Park, Homestead, Florida.

Blakesley, B. A., J. H. Landsberg, B. B. Ackerman, R. O. Reese, J. R. Styer, C. O. Obordo, and S. E. Lucas-Black. 1998. Slime mold, salinity, and statistics: Implications from laboratory experimentation with turtlegrass to facilitate interpretation of field results from Florida Bay. Florida Bay Science Conference, Abstract. Miami FL.

Blaylock, R. B. and R. M. Overstreet. 2003. Diseases and parasites of the spotted seatrout, Biology of the spotted seatrout (S. A. Bortone, ed.), CRC Press, Boca Raton, FL, pp. 197–225.

Bohnsack, J. A., J. S. Ault, and S. G. Smith. 2001. Advances in reef fish monitoring and assessment in the Florida Keys. 2001 Florida Bay Science Conference.

Bohnsack, J. A. and 16 co-authors. 1999. Baseline data for evaluating reef fish populations in the Florida Keys, 1979–1998. NOAA Tech. Mem. NMFS-SEFSC-427.

Bohnsack, J. A., D. E. Harper, and D. B. McClellan. 1994. Fisheries trends from Monroe County, Florida. Bull. Mar. Sci. 54: 982–1018.

Borum, J., O. Pedersen, T. M. Greve, T. A. Frankovich, J. C. Zieman, J. W. Fourqurean and C. J. Madden. 2005. The potential role of plant oxygen and sulfide dynamics in die-off events of the tropical seagrass, *Thalassia testudinum*. J. Ecology 93: 148-158.

Boyer, J. N. and R. D. Jones. 2002. A view from the bridge: External and internal forces affecting the ambient water quality of the Florida Keys National Marine Sanctuary. Pp 609-628 In: Porter and Porter [eds] The Everglades, Florida Bay and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook. CRC Press Boca Raton. 1000 pp.

Boyer, J.N., J.W. Fourqurean, and R.D. Jones. 1999. Seasonal and long-term trends in the water quality of Florida Bay (1989-1997). Estuaries 22(2B): 417-430.

Boyer, J.N., J.W. Fourqurean and R.D. Jones. 1997. Spatial characterization of water quality in Florida Bay and Whitewater Bay by multivariate analyses: Zones of similar influence. Estuaries 20(4): 743-758.

Brand, L. 2002. The transport of terrestrial nutrients to south Florida coastal waters. Pages 361-414 In: Porter and Porter Eds. *The Everglades, Florida Bay and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook.* CRC Press, Boca Raton, Florida. 1000 pp.

Brewster-Wingard, L., T. Cronin, B. Wardlaw, J. Stone, S. Schwede, S. Ishman, C. Holmes, R. Halley, M. Marot, G. Dwyer, and J. Huvane. 1999. Long-term Florida Bay salinity history: a synthesis of multi-proxy evidence from sediment cores. Pages 182-183 in Program and abstracts, 1999 Florida Bay and Adjacent Marine Systems Science Conference.

Brewster-Wingard, G., ans S.E. Ishman. 1999. Historical trends in salinity and substrate in central Florida Bay: A paleoecological reconstruction using modern analogue data. Estuaries 22(2B): 369-383.

Browder, J. A., & 24 others. 2003. Synthesis of Florida Bay higher trophic levels research, Florida Bay Science Conference, April 2003 (W Nuttle, ed.), Tampa Bay, FL, 68 pp.

Browder, J. A., P. J. Gleason, and D. R. Swift. 1994. Periphyton in the Everglades: Spatial variation, environmental correlates and ecological implications. Pp. 379-418 In: J. C. Ogden and S. Davis [eds] Everglades: The ecosystem and its restoration.

Bullock, L. H., M. D. Murphy, M. F. Godcharles, and M. E. Mitchell. 1992. Age, growth, and reproduction of jewfish *Epinephelus itajara* in the eastern Gulf of Mexico: 23.

Bullock, L. H. and G. B. Smith. 1991. Seabasses (Pisces: Serranidae). Memoirs of the Hourglass Cruises. Florida Mar. Res. Inst. Dep. Nat. Resour. St. Petersburg FL. Vol.8(2). 243 pp.

Burton, M. L. 2001. Age, growth, and mortality of gray snapper, *Lutjanus griseus*, from the east coast of Florida. Fish. Bull. U.S. 99: 254–265.

Butler, M.J. IV, T. Dolan, J. H. Hunt, W. F. Herrnkind, and K. Rose. *In press.* Recruitment in degraded marine habitats: a spatially-explicit, individual-based model for spiny lobster. Ecological Applications.

Butler, M.J. IV. 2003. Incorporating ecological process and environmental change into spiny lobster population models using a spatially-explicit, individual-based approach. Fisheries Research 65: 63-79.

Butler, M.J. IV. 2002. Changing salinity effects on spiny lobster and hard-bottom habitat. Contract NA16OP256. Report to NOAA Coastal Ocean Program, Silver Springs, MD.

Butler, M.J. IV and W.F. Herrnkind. 2001. Puerulus and juvenile ecology, pp. 276 - 301 in B.F. Phillips and J. Kittaka Eds. *Spiny Lobsters: Fisheries and Culture*, 2nd ed. Blackwell Scientific Press, Oxford.

Butler, M.J. IV, W.F Herrnkind, and J. H. Hunt. 1997. Factors affecting the recruitment of juvenile Caribbean spiny lobsters dwelling in macroalgae. Bulletin of Marine Science 61: 3-19.

Butler, M.J. IV and W.F. Herrnkind. 1997. A test of recruitment limitation and the potential or artificial enhancement of spiny lobster populations in Florida. Canadian Journal of Fisheries and Aquatic Sciences 54: 452-463.

Butler, M. J., J. H. Hunt, W. F. Herrnkind, M. J. Childress, R. Bertelsen, . Sharp, T. Matthews, J. M. Field, and H. G. Marshall. 1995. Cascading disturbances in Florida Bay, USA: cyanobacterial blooms, sponge mortality, and implications for juvenile spiny lobsters, *Panulirus argus*, Mar. Ecol. Prog. Ser. 129:119-125.

Butler, M.J., IV and W.F. Herrnkind. 1991. Effect of benthic microhabitat cues on the metamorphosis of postlarvae of the spiny lobster *Panulirus argus*. Journal of Crustacean Biology 11:23-28.

Butler, P. A., R. Childress, and A. J. Wilson. 1970. The association of DDT residues with losses in marine productivity, FAO Technical Conference on Marine Pollution and its Effects on Living Resources and Fishing, FAO Technical Conference on Marine Pollution and its Effects on Living Resources and Fishing.

Butler, P. A. 1969. The significance of DDT residues in estuarine fauna, Chemical Fallout (M. W. Miller and G. B. Berg, eds.), Charles C. Thomas Publishers, Springfield, IL, pp. 205–220.

Cambridge, M. L., A. W. Chiffings, C. Brittan, L. Moore, and A. J. McComb. 1986. The loss of seagrass in Cockburn Sound, Western Australia, II. Possible causes of seagrass decline. Aquatic Bot. 24:269-285.

Cambridge, M. L. and A. J. McComb. 1984. The loss of seagrasses in Cockburn Sound, Western Australia. I. The time course and magnitude of seagrass decline in relation to industrial development. Aquat Bot. 20: 229-242.

Carlson, P.R., Jr. 1997. Sediment porewater chemistry of mangrove die-off sites in Florida Bay. Florida Department of Environmental Protection Report to Everglades National Park.

Carlson, P.R., Jr., S. Brinton, T. Armentano, D. Smith, and J. Absten. 1995. Mangrove mortality in Florida Bay.Page 227 in Florida Bay Science Conference: a report by principal investigators - abstracts and program.

Carlson, P. R., L. A. Yarbro, and T. R. Butler. 1994. Relationship of sediment sulfide to mortality of Thalassia testudinum in Florida Bay. Bull. Mar. Sci. 54: 733-746.

Carlson, P., Jr., M. J. Durako, T. R. Barber, L. A. Yarbro, Y. deLama, and B. Hedin. 1990a. Catastrophic mortality of the seagrass Thalassia testudinum in Florida Bay. Report to Florida Dept of Natural Resources.

Carlson, P., Jr., M. J. Durako, T. R. Barber, L. A. Yarbro, Y. deLama, and B. Hedin. 1990b. Investigations on the causes, extent, and characteristics of a seagrass die-back in Florida Bay. Report to Florida Dept of Natural Resources.

Carpenter, J. S. and W. R. Nelson. 1971. Fishery potential for snapper and grouper in the Caribbean area and the Guianas. Symposium on investigations and resources of the Caribbean Sea and adjacent regions. FAO 71.

Cass-Calay, S. and T. W. Schmidt. *In review*. Stardardized catch rates of juvenile goliath grouper, *Epinephelus itajara*, from the Everglades National Park creel survey, 1973-1999. Gulf Mex. Sci.

Cha, S. S., M. F. McGowan, and W. J. Richards. 1994. Vertical distribution of fish larvae off the Florida Keys, 26 May–5 June 1989. Bull. Mar. Sci. 54: 828–842.

Chester, A. J. and G. W. Thayer. 1990. Distribution of spotted seatrout (*Cynoscion nebulosus*) and gray snapper (*Lutjanus griseus*) juveniles in seagrass habitats of western Florida Bay. Bull. Mar. Sci. 46: 345–357.

Chiappone, M. 1996. Marine benthic communities of the Florida Keys, *Vol. 4 of Site Characterization for the Florida Keys National Marine Sanctuary and Environs,* Farley Court Pub., Zenda, Wisconsin.

Childers, D. L, R. D. Jones, J. C. Trexler, C. Buzzelli, S. Dailey, A. L. Edwards, E. E. Gaiser, K. Jayachandaran, A. Kenne, D. lee, J. F. Meeder, J. H. K. Pechmann, A. Renshaw, J. Richards, M. Rugge, L. J. Scinto, P. Sterling, and W. Van Gelder. 2002. Quantifying effects of low-level phosphorus additions on unenriched Everglades wetlands with in situ flumes and phosphorus dosing. Pp 127-152 In: Porter and Porter [eds] The Everglades, Florida Bay and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook. CRC Press Boca Raton. 1000 pp.

Claro, R. and J. P. Garcia-Arteaga. 1994. Crecimiento, Ecologia de los peces marinos de Cuba (R. Claro, ed.), Centro de Investigaciones de Quintana Roo, Mexico, pp. 321–402.

Collins, L. A. and J. H. Finucane. 1984. Ichthyoplankton survey of the estuarine and inshore waters of the Florida Everglades, May 1971 to February 1972. NOAA Tech. Rep. NMFS 6.

Comprehensive Everglades Restoration Plan (CERP). 2005a. The Comprehensive Everglades Restoration Plan. 17 March 2005. http://www.evergladesplan.org/about/rest_plan.cfm.

Comprehensive Everglades Restoration Plan (CERP). 2005b. The Comprehensive Everglades Restoration Plan: RECOVER – Restoration, Coordination and Verification. 17 March 2005. http://www.evergladesplan.org/pm/recover/recover..cfm.

Comprehensive Everglades Restoration Plan (CERP). 2005c. The Comprehensive Everglades Restoration Plan: System-Wide Modeling. Available: http://modeling.cerpzone.org/cerp_recover/pmviewer/pmviewer.jsp (accessed 14 February 2005).

Comprehensive Everglades Restoration Plan (CERP). 2004. The Comprehensive Everglades Restoration Plan: Annual Report – January 2004. 17 March 2005. http://www.evergladesplan.org/pm/cerp_annual_report.cfm.

Comprehensive Everglades Restoration Plan (CERP). 2000. Central and southern Florida project Comprehensive Everglades Restoration Plan Section 9 and 10 CERP. Yellow Book. Training.l June 19, 2000. US Army Corps of Engineers and South Florida Water Management District.

Conroy, R.M., Hyniova, J., Carlson, P.R.Jr., Arnold, H., Blakesley, B.A., Luca, S.E. & Yarbro, L.A. 1999. Turtlegrass (*Thalassia testudinum*) mortality on Florida Bay mudbanks. 1999 Florida Bay and Adjacent Marine Systems Science Conference. (abstr.): 26.

Cornwell, J., M. Owens, and C. J. Madden. 2000. Nitrogen cycling in Florida Bay mangrove environments: Sediment-water exchange and denitrification. Report to South Florida Water Management District. 33 pp.

Coronado-Molina, C. 2000. Personal communication. South Florida Water Management District.

Cox, C., J.H. Hunt, W.G. Lyons, and G.E. Davis. 1997. Nocturnal foraging of the Caribbean spiny lobster (*Panulirus argus*) on offshore reefs of Florida, USA. Marine Freshwater Research 48: 671-680.

Craighead, F. C. and V. C. Gilbert. 1962. The effect of Hurricane Donna on the vegetation of southern Florida. Q. J. Fla. Acad. Sci. 25: 1-18.

Criales, M. M., J. Wang, J. A. Browder, and M. B. Robblee. 2005. Tidal and seasonal effects on transport of pink shrimp postlarvae. Mar. Ecol. Prog. Ser. 286: 231-238.

Criales, M. M., C. Yeung, D. L. Jones, T. L. Jackson, and W. J. Richards. 2003. Variation of oceanographic processes affecting size of pink shrimp (Farfantepenaeus duorarum) postlarvae and their supply to Florida Bay. Estuarine, Coastal, and Shelf Science. 57: 457-468.

Criales, M. M. and T. N. Lee. 1995. Larval distribution and transport of panaeoid shrimps during the presence of the Tortugas Gyre in May–June 1991. Fish. Bull. U.S. 93: 471–482.

Croker, R. A. 1962. Growth and food of the gray snapper, *Lutjanus griseus*, in Everglades National Park. Trans. Am. Fish. Soc. 91: 379–383.

Croker, R. A. 1960. A contribution to the life history of the gray (mangrove) snapper, *Lutjanus griseus* (Linnaeus), University of Miami, University of Miami, Miami, FL, 93 pp.

Crowder, L. B., J. A. Rice, T. J. Miller and E. A. Marschall. 1990. Empirical and theoretical approaches to size-based interactions and recruitment variability in fishes. pg. 237-255 In: DeAngelis, D. L. and Gross, L. J. Eds. *Individual-based models and approaches in ecology*. New York, NY Chapman Hall.

Cushing D. H. 1983. Are fish larvae too dilute to affect the density of their food organisms? J. Plank. Res. 5:847-854.

Cushing, D. H. 1969. The regularity of the spawning season of some fishes. J. Conseil International pour l'Exploration de la Mer, 33, 81-92.

D'Sa, E. J., J. B. Zaitzeff, C. S. Yentsch, J. L. Miller, and R. Ives. Rapid remote assessments of salinity and ocean color in Florida Bay. 2002. Pages 451-460 In: Porter and Porter Eds. *The Everglades, Florida Bay and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook.* CRC Press Boca Raton. 1000 pp.

Davis, G. E., and Dodrill, J. W. 1989. Recreational fishery population dynamics of spiny lobsters, *Panulirus argus*, in Florida Bay, Everglades National Park, 1977-1980. Bull. Mar. Sci. 44(1): 78-88.

Davis, S. E., J. E. Cable, D. L. Childers, C. Coronado-Molina, J. W. Day, Jr., C. D. Hittle, C. J. Madden, E. Reyes, D. Rudnick, and F. Sklar. 2004. Importance of storm events in controlling ecosystem structure and function in a Florida Gulf coast estuary. J. Coastal Res. 20(4): 1198-1208.

Davis, S.M., L.H. Gunderson, W.A. Park, J.R. Richardson, and J.E. Mattson. 1994. Landscape dimension, composition and function in a changing Everglades ecosystem. Pages 419-444 In Davis, S.M. and J.C. Ogden Eds. *Everglades: The Ecosystem and its Restoration.* St. Lucie Press, Delray Beach, Florida. 826 pages.

Day, J. W., Jr., C. J. Madden, F. Ley-Lou, R. Wetzel, and A. M. Machado-Navarro. 1988. Aquatic primary productivity in Terminos Lagoon. Pages 221-235 In A. Yanez-Arancibia and J. W. Day, Jr. Eds. *Ecology of Coastal Ecosystems in the Southern Gulf of Mexico*. Editorial Universitaria, Mexico.

DeMaria, K. K. 1996. Changes in the Florida Keys marine ecosystem based upon interviews with experienced residents. The Nature Conservancy and Center for Marine Conservation Special Report, The Nature Conservancy, The Nature Conservancy, Key West, FL.

DenHartog, C. 1987. "Wasting disease" and other dynamic phenomena in *Zostera* beds. Aquatic Bot. 27:3-14.

Dennis, G. D. and K. J. Sulak. 2001. Mangrove prop-root habitat as essential fish habitat in northeastern Florida Bay. Proc. Florida Bay Science Conference: 180.

Dewar, H. 1992. Dying mangrove trees killing off bay islands. Page 1, The Miami Herald, April 19, 1992.

Diaz, G. 1995. Distribution and recruitment strategies of stomatopod larvae off the Dry Tortugas and Lower Florida Keys, M.S. Thesis, University of Miami, M.S. Thesis, University of Miami, Coral Gables, FL, 191 pp.

Domeier, M. L. and P. L. Colin. 1997. Tropical reef fish spawning aggregations: defined and reviewed. Bull. Mar. Sci. 60: 698–726.

Duever, M.J., J.F. Meeder, L.C. Meeder, and J.M. McCollom. 1994. The climate of south Florida and its role in shaping the Everglades ecosystem. Pages 225-248 In Davis, S.M. and J.C. Ogden Eds. *Everglades: The Ecosystem and its Restoration.* St. Lucie Press, Delray Beach, Florida. 826 pages.

Durako, M. J., M. O. Hall, M. Merello. 2002. Patterns of change in the seagrass dominated Florida Bay hydroscape. Pages 523-537 In Porter and Porter Eds. *The Everglades, Florida Bay and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook*. CRC Press Boca Raton. 1000 pp.

Eggleston, D.B. and R.N. Lipcius. 1992. Shelter selection by spiny lobster under variable predation risk, social conditions, and shelter size. Ecology 73:992-1011.

Eggleston, D.B., R.N. Lipcius, D.L. Miller, and L. Coba-Cetina. 1990. Shelter scaling regulates survival of juvenile Caribbean spiny lobster *Panulirus argus*. Marine Ecology Progress Series 62:70-88.

Egler, F. E. 1952. Southeast saline Everglades vegetation, Florida and its management. Vegetation 3: 213-265.

Ehrhardt, N. M., and C. M. Legault. 1999. Pink shrimp, *Farfantepenaeus duorarum*, Recruitment variability as an indicator of Florida Bay dynamics. Estuaries 22(2B): 471-483.

Eklund, A. M. Status of the stock of Nassau grouper, *Epinephelus striatus*, and jewfish, *Epinephelus itajara*. Final Report. SEFSC report. Miami Contribution No. MIA-94/95-15, Miami, FL, 170 pp.

Enos, P. 1989. Islands in the bay - a key habitat of Florida Bay. Bull. Mar. Sci. 44: 365-386.

Erdman, D. S. 1957. Recent fish records from Puerto Rico. Mar. Sci. Gulf and Carib. Vol. 6 (1956) No. 4: 315-340.

Evans, D. W. and P. J Crumley. *In review*. Mercury in Florida Bay Fish: Spatial distribution of elevated concentrations and possible linkages to Everglades restoration. Bull Mar. Research.

Evans, D.W., P.H. Crumley, D. Rumbold, and S. Niemczyk. 2003. Mercury in fish from eastern Florida Bay. Presented at the 2003 Joint Conference on the Science and Restoration of the Greater Everglades and Florida Bay Ecosystem. Palm Harbor, Florida. 13-18 April 2003. Available at: http://conference.ifas.ufl.edu/jc/FB.pdf [March 11, 2005].

Evans, L.H., J.B. Jones, and J.A. Brock. . 2000. Diseases of spiny lobsters, pp. 586 – 600 In Spiny Lobsters: Fisheries and Culture, B.F. Phillips and J. Kittaka (eds), Blackwell Science Press, Oxford.

Faunce, C. H., J. J. Lorenz, J. E. Ley, and J. E. Serafy. 2002. Size structure of gray snapper (*Lutjanus griseus*) within a mangrove "no-take" sancturay. Bull. Mar. Sci. 70: 211–216.

Fennema, R.J., C.J. Neidrauer, R.A. Johnson, T.K. MacVicar, and W.A. Perkins. 1994. A computer model to simulate natural Everglades Hydrology. Pages 249-290 In Davis, S.M. and J.C. Ogden Eds. *Everglades: The Ecosystem and its Restoration.* St. Lucie Press, Delray Beach, Florida. 826 pages.

Fish Habitat Assessment Program (FHAP). Florida Fish and Wildlife Conservation Commission. M. O. Hall, project manager.

Field, J.M. and M.J. Butler IV. 1994. The influence of temperature, salinity, and larval transport on the distribution of juvenile spiny lobsters, *Panulirus argus*, in Florida Bay. Crustaceana 67: 26-45.

Florida Department of Environmental Protection (FDEP). 2004. About the Florida Keys National Marine Sanctuary. 17 March 2005. http://www.dep.state.fl.us/coastal/sites/keys/info.htm.

Florida Department of Health (FDH). 2004. Fish Consumption Advisory: Eating Guidelines for Marine and Estuarine Fish from Florida Water. 17 March 2005. http://www.myfloridaeh.com/community/fishconsumptionadvisories/MEFG.htm.

Florida Fish and Wildlife Conservation Commission—Fish and Wildlife Research Institute (FFWCC). 2003. Website http://research.myfwc.com.

Florida Geographic Data Library (FGDL). 1999. University of Florida Geoplan Center, Florida Geographic Data Library. Available at: http://www.fgdl.org/. (accessed 14 February 2005).

Florida Marine Research Institute (FMRI). 2003a. Spiny Lobster, *Panulirus argus*, species account. Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute.

Florida Marine Research Institute (FMRI). 2003b. 2003 Spotted Seatrout Stock Assesment. Florida Marine Research Institute. Available at: http://www.floridamarine.org/features/view_article.asp?id=20805 (accessed 14 February 2005).

Florida Marine Research Institute (FMRI). 2003c. Gray Snapper (*Lutjanus griseus*). Florida Marine Research Institute. Available at: http://www.floridamarine.org/features/view_article.asp?id=5148 (accessed 14 February 2005).Florida Marine Research Institute (FMRI). Unpublished data. Florida Department of Natural Resoures. Annual Landings Summary. Florida Marine Research Institute, St. Petrersburg, FL.

Florida Marine Research Institute (FMRI). 2001. Florida Mangroves. Available at: http://www.fgdl.org/ (accessed 14 February 2005).

Florida Marine Research Institute (FMRI). 1990. Benthic resources for the state of Florida. Available at: http://www.fgdl.org/ (accessed 14 February 2005).

Florida Museum of Natural History (FMNH). 2005. Florida Keys: Threats. 17 March 2005. http://www.flmnh.ufl.edu/fish/southflorida/coral/threatskeys.html#urban.

Forcucci, D.M., M.J. Butler IV, and J.H. Hunt. 1994. Growth and population dynamics of juvenile Caribbean spiny lobster, *Panulirus argus*, in Florida Bay, FL (USA). Bulletin of Marine Science 54: 805-818

Fourqurean, J.W., M. J. Durako, M. O. Hall, and L. N. Hefty. 2002. Seagrass distribution in south Florida: A multi-agency coordinated monitoring program. Pages 497-522 In Porter and Porter Eds. *The Everglades, Florida Bay and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook*. CRC Press Boca Raton. 1000 pp.

Fourqurean, J.W., M.B. Robblee & L.A. Deegan (Eds.). 1999. Florida Bay: A dynamic subtropical estuary. Dedicated Issue, Estuaries, 22: 345-561.

Fourqurean, J.W. and M.B. Robblee. 1999. Florida Bay: A history of recent ecological changes. Estuaries 22(2B): 345-357.

Fourqurean, J. W., R. D. Jones, and J. C. Zieman. 1993. Processes influencing water column nutrient characteristics and phosphorus limitation of phytoplankton biomass in Florida Bay, FL USA: Inferences from spatial distributions. Estuarine, Coastal and Shelf Science 36:295-314.

Fourqurean, J. W., J. C. Zieman, and G. V. N. Powell. 1992. Relationships between porewater nutrients and seagrasses in a subtropical carbonate environment. Mar. Biol. 114:57-65.

Gilmore, R. G., L. H. Bullock, and F. H. Berry. 1978. Hypothermal mortality in marine fishes of south-central Florida January 1977. Northeast Gulf Sci. 2: 77–97.

Gleason, P.J. and P. Stone. 1994. Age, origin, and landscape evolution of the Everglades peatland. Pages 149-198 In Davis, S.M. and J.C. Ogden Eds. *Everglades: The Ecosystem and its Restoration.* St. Lucie Press, Delray Beach, Florida. 826 pages.

Grimes, C. B., C. S. Manooch, and G. R. Huntsman. 1982. Reef and rock outcropping fishes of the outer continental shelf of North Carolina and South Carolina, and ecological notes on the red porgy and vermilion snapper. Bull. Mar. Sci. 32: 277–289.

Grimes, S.A. and A. Sprunt. 1936. Aerial and surface inspection of the Florida Keys in connection with the status of the Great White Heron and spoonbill. Unpubl. Rept., National Audubon Society, Tavernier, Florida.

Gulf of Mexico Fishery Management Council (GMFMC). 2005. http://www.gulfcouncil.org/.

Hall, M. O., M. J. Durako, J. W. Fourqurean, and J. C. Zieman. 1999. Decadal changes in seagrass distribution and abundance in Florida Bay. Estuaries 22(2B):445-459.

Halley, R. B., and L. M. Roulier. 1999. Reconstructing the history of eastern and central Florida Bay using mollusk-shell isotope records. Estuaries 22(2B): 358-368.

Harper, D. E. 1995. Trends in the spiny lobster commercial fishery of Florida, 1960-1990. National Marine Fisheries Series Report No. MIA-94/95-47, Miami, FL, USA.

Harrigan, P., J. C. Zieman, and S. A. Macko. 1989. The base of nutritional support for the gray snapper (*Lutjanus griseus*): an evalulation based on a combined stomach content and stable isotope analsysis. Bull. Mar. Sci. 44: 65–77.

Heald, E. J. and W. E. Odum. 1970. The contribution of mangrove swamps to Florida fisheries. Proc. Gulf Carib. Fish. Inst. 22: 130-135.

Heald, E. J. 1969. The production of organic detritus in a south Florida estuary. Ph.D. dissertation. Univ. of Miami FL., 110 pp.

Helser, T. E., R. E. Condrey, and J. P. Geaghan. 1993. Spotted seatrout distribution in four coastal Louisiana estuaries. Trans. Am. Fish. Soc. 122: 99–111.

Herrnkind, W. M. Childress, and K. Lavalli. 2001. Cooperative defence and other benefits among exposed spiny lobsters: inferences from group size and behaviour. Marine Freshwater Research 52:1113-1124.

Herrnkind, W.H., M.J. Butler IV, J.H. Hunt, and M. Childress. 1997. The role of physical refugia: implications from a mass sponge die-off in a lobster nursery. Marine and Freshwater Research 48: 759-770.

Herrnkind, W.F., M.J. Butler IV, and J.H. Hunt. 1997. Can artificial habitats that mimic natural structures enhance recruitment of Caribbean spiny lobster? Fisheries 22: 24-27.

Herrnkind, W.F. and M.J. Butler IV. 1994. Settlement of spiny lobsters, Panulirus argus in Florida: pattern without predictability. Crustaceana 67: 46-64.

Herrnkind, W.F., M.J. Butler IV, and R.A. Tankersley. 1988. The effects of siltation on the recruitment of spiny lobsters (*Panulirus argus*) in south Florida. Fisheries Bulletin 86: 331-338.

Herrnkind, W.F. and M.J. Butler IV. 1986. Factors regulating settlement and microhabitat use by juvenile spiny lobsters, *Panulirus argus*. Marine Ecology Progress Series 34: 23-30.

Hettler, W. F., Jr. and D. L. Barker. 1993. Distribution and abundance of larval fishes at two North Carolina inlets. Est. Coast. Shelf Sci. 37: 161–179.

Hettler, W. F., Jr. 1989. Food habits of juveniles of spotted seatrout and gray snapper in western Florida Bay. Bull. Mar. Sci. 44: 155–162.

Hittle, C. 2001. Quantity, timing, and distribution of fresh water flows into northeastern Florida Bay. Abstract, Florida Bay Science Conference. Key Largo, Florida, April 25-26, 2001.

Holmquist, J. G., G. V. N. Powell, and S. M. Sogard. 1989. Decapod and stomatopod communities of seagrass-covered mud banks in Florida Bay: inter- and intra-bank heterogeneity with special reference to isolated subenvironments. Bull. Mar. Sci. 44: 251–262.

Holt, G. J. and S. A. Holt. 2003. Effects of variable salinity on reproductive and early life stages of spotted seatrout, Biology of the spotted seatrout (S. A. Bortone, ed.), CRC Press, Boca Raton, FL, pp. 135–145. http://www.audubonofflorida.org/science/spoonbills.htm. (accessed 14 February 2005).

Husar, J.D. and R.B. Husar. 2002. Trends of anthropogenic mercury mass flows and emissions in Florida. Final Report, Lantern Corporation, Clayton, MO.

Ingebritsen, S.E., McVoy, C., Glaz, B., Park, W. 1999. Florida Everglades: Subsidence threatens agriculture and complicates ecosystem restoration, In Galloway, D., Jones, D.R., and Ingebritsen, S.E., Eds., *Land Subsidence in the United States: U.S. Geological Survey Circular 1182*, p. 95-106.

Iverson, E. S. and D. C. Tabb. 1962. Subpopulations based on growth and tagging studies of spotted sea-trout, *Cynoscion nebulosus*, in Florida. Copeia 1962: 544-548.

Iverson, R. L. and H. F. Bittaker. 1986. Seagrass distribution and abundance in eastern Gulf of Mexico coastal waters. Estuar. Coast. Shelf Sci. 1986: 577–602.

Jaffe, R. 1999. Characterization of dissolved organic matter (DOM) in the Everglades and coastal environments of southern Florida. Report to South Florida Water Management District. 20 pp.

Jannke, T. E. 1971. Abundance of young sciaenid fishes in Everglades National Park, Florida, in relation to season and other variables. Univ. Miami Sea Grant Program Sea Grant Tech. Bull. 11.

Johnson, A. G., L. A. Collins, and C. P. Keim. 1994. Age-size structure of gray snapper from the south-eastern United States: a comparison of two methods of back-calculating size at age from otolith data. Proc. Annu. Conf. Southeast. Assoc. Fish Wildl. Agencies 48: 592–600.

Johnson, A. G., T. D. Williams, and C. R. Arnold. 1977. Chlorine-induced mortality of eggs and larvae of spotted seatrout (*Cynoscion nebulosus*). Trans. Am. Fish. Soc. 106: 466–469.

Johnson, R.A. and R.A. Fennema. 1989. Conflicts over flood control and wetland preservation in the Taylor Slough and eastern panhandle basins of Everglades National Park, pages 451-462 In: Wetlands: Concerns and Successes, D.W. Fisk, Ed. American Water Resources Association. Bethesda, Maryland.

Jones, D. L., M. R. Lara, C. Yeung, T. L. Jackson, M. M. Criales, and W. J. Richards. 2001. Offshore larval supply of snapper larvae (Pisces: Lutjanidae) into Florida Bay. Proc. Florida Bay Science Conference, Florida Sea Grant, Key Largo, FL: 186–187.

Jones, R. D., and J. A. Amador. 1992. Removal of total phosphorus and phosphate by peat soils of the Florida Everglades. 1992. Canadian Journal of Fisheries and Aquatic Sciences 49:577-583.

Jordon, D. 1990. Mercury contamination: Another threat to the Florida panther. Endangered Species Technical Bulletin 15(2): 1,6. U.S. Department of the Interior, Fish and Wildlife Service. Washington, D.C.

Kemp, W. M., R. R. Twilley, J. C. Stevenson, W. R. Boynton, and J. C. Means. 1983. The decline of submerged vascular plants in upper Chesapeake Bay: Summary of results concerning possible causes. Mar. Tech. Soc. J. 17: 878-889.

Kennish, M. J. and B. E. Ruppel. 1998. Organochlorine contamination in selected estuarine and coastal marine finfish and shellfish of New Jersey. Water, Air, Soil Pollution 101: 123-136.

Kennish, M. J. and B. E. Ruppel. 1996. Chlordane contamination in selected estuarine and coastal marine finfish and shellfish of New Jersey. Environ. Pollut. 94: 75–81.

King, D. 2005. Personal Communication. Park Ranger, Everglades National Park. National Park Service. Homestead, Florida.

Koch, M. and M. J. Durako. 2004. High salinity and multiple stressor effects on seagrass communities of NE Florida Bay. Interim report to South Florida Water Management District, Contract C-12430.

Koch, M. 2004. unpublished videographic data of die-off event in Buchanan Bank in November 2004.

Koch, M.S., R.E Benz & D.T. Rudnick. 2001. Solid-phase phosphorus pools in highly organic carbonate sediments of northeastern Florida Bay. Estuarine, Coastal and Shelf Science 52: 279-291.

Koch, M. S. 1996. Resource availability and abiotic stress effects on Rhizophora mangle L. (red mangrove) development in South Florida. Ph.D. Dissertation, Univ. of Miami. Coral Gables, FL.

Koenig, C. C., F. C. Coleman, A. M. Eklund, J. Schull, and J. Ueland. *In review*. Mangrove as essential nursery habitat for goliath grouper (*Epinephelus itajara*). Mar. Ecol. Progr. Ser.

Koenig, C., F. Coleman, and A.M. Eklund. 1997. Studies of the jewfish (Epinephelus itajara) in the eastern Gulf of Mexico. Unpubl. report to The Curtis and Edith Munson Foundation, 26 pp.

Kucera, C. J., C. K. Faulk and G. J. Holt. 2002. The effect of spawning salinity on eggs of spotted seatrout (*Cynoscion nebulosus*, Cuvier) from two bays with historically different salinity regimes. J. Exp. Mar. Biol. Ecol. 272:147-158.

Lapointe, B. E., P. J. Barile, W. R. Matzie. 2004. Anthropogenic nutrient enrichment of seagrass and coral reef communities in the Lower Florida Keys: discrimination of local versus regional nitrogen sources. J. Exp. Mar. Biol. Ecol. 308:23-58.

Lapointe, B. E., W. R. Matzie, and P. J. Barile. 2002. Biotic phase shifts in Florida Bay and fore reef communities of the Florida Keys:Linkages with historical freshwater flows and nitrogen loading from Everglades runoff. Pages 629-648 In Porter and Porter Eds. *The Everglades, Florida Bay and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook.* CRC Press Boca Raton. 1000 pp.

Lapointe, B. E. D. A. Tamasko, and W. R. Matzie. 1994. Eutrophication and trophic state classification of seagrass communities in the Florida Keys. Bull. Mar. Sci. 54:696-717.

Lapointe, B. E. 1989. Macroalgal production and nutrient relations in oligotrophic areas of Florida Bay. Bull. Mar. Sci. 44:312-323.

Lara, M.R., D.L. Jones and J. T. Lamkin. *In review.* ICP-MS elemental analysis of juvenile snapper otoliths results in rare earth element based Florida Bay signature.

Lassuy, D. R. 1983. Species profiles: life histories and environmental requirements (Gulf of Mexico)—spotted seatrout. U.S. Fish and Wildl. Serv., Div. Biol. Serv. FWS/OBS-82/11.4. U.S. Army Corps of Engineers, TR EL-82-4.

Le Moullac, G. and P. Haffner. 2000. Environmental factors affecting immune responses in Crustacea. Aquaculture 191:121-131.

Lee, T. N., E. Williams, E. Johns, D. Wilson, and N. P. Smith. 2002. Transport processes linking South Florida coastal ecosystems, Pages 309–342 In J. W. Porter and K. G. Porter, Eds. *The Everglades, Florida Bay and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook.* CRC Press Boca Raton. 1000 pp.

Lee, T. N. and E. Williams. 1999. Mean distribution and seasonal variability of coastal currents and temperature in the Florida Keys with implications for larval recruitment. Bull. Mar. Sci. 64: 35–56.

Lee, T. N., M. E. Clarke, E. Williams, A. F. Szmant, and T. Berger. 1994. Evolution of the Tortugas Gyre and its influence on recruitment in the Florida Keys. Bull. Mar. Sci. 54: 621–646.

Lee, T. N., C. Rooth, E. Williams, M. F. McGowan, A. F. Szmant, and M. E. Clarke. 1992. Influence of Florida Current, gyres and wind-driven circulation on transport of larvae and recruitment in the Florida Keys coral reefs. Cont. Shelf Res. 12: 971–1002.

Ley, J. A., and C. C. McIvor. 2002. Linkages between estuarine and reef fish assemblages:Enhancement by the presence of well-developed mangrove shorelines. Pages 539-562 In Porter and Porter Eds. *The Everglades, Florida Bay and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook.* CRC Press Boca Raton. 1000 pp.

Ley, J. A., C.C. McIvor, and C.L. Montague. 1999. Fishes in mangrove prop root habitats of northeastern Florida Bay: distinct assemblages across an estuarine gradient. Estuarine, Coastal and Shelf Sciences Vol. 48: 710-723

Light, S.S. and J.W. Dineen. 1994. Water control in the Everglades: A historical perspective. Pages 47-84 In Davis, S.M. and J.C. Ogden Eds. *Everglades: The Ecosystem and its Restoration.* St. Lucie Press, Delray Beach, Florida. 826 pages.

Limouzy-Paris, C. B., H. C. Graber, D. L. Jones, A. W. Roepke, and W. J. Richards. 1997. Translocation of larval coral reef fishes via sub-mesoscale spin-off eddies from the Florida Current. Bull. Mar. Sci. 60: 966–983.

Limouzy-Paris, C., M. F. McGowan, W. J. Richards, J. P. Umaran, and S. S. Cha. 1994. Diversity of fish larvae in the Florida Keys: Results from SEFCAR. Bull. Mar. Sci. 54: 857–870.

Lindall, W. N., Jr., W. A. Fable Jr., and L. A. Collins. 1975. Additional studies of the fishes, macroinvertebrates, and hydrological conditions of upland canals in Tampa Bay, Florida. Fish. Bull. (US) 73: 81–85.

Lindeman, K. C. 1997. Development of grunts and snappers of southeast Florida: cross-shelf distributions and effects of beach management alternatives, University of Miami, University of Miami, Coral Gables, FL, 419 pp.

Lodge, T.E. 2005. *The Everglades Handbook: Understanding the Ecosystem*. 2nd Edition. CRC Press. Boca Raton, Florida. 302 pages.

Loftus, W.F. 2000. Inventory of fishes of Everglades National Park. Florida Scientist 63(1): 27-47.

Loftus, W. and O. Bass, Jr. 1992. Mercury threatens wildlife resources. Park Science 24(4): 18-20.

Longley, W. H. and S. F. Hildebrand. 1941. Systematic catalogue of the fishes of Tortugas, Florida, with observations on color, habits and local distribution. Papers Tortugas Lab. 34:.

Lorenz, J.J., J.C. Odgen, R.D. Bjork, G.V.N. Powell. 2002. Nesting patterns of Roseate Spoonbills in Florida Bay 1935-1999: Implications of landscape scale anthropogenic impacts. In J.W. Porter, K.G. Porter Eds: *The Everglades, Florida Bay, and Coral Reefs of the Florida Keys: An Ecosystem Sourebook.* CRC Press, Boca Raton, FL. 1000 pp.

Lorenz, J.J. 1999. The response of fishes to physiochemical changes in the mangroves of northeast Florida Bay. Estuaries 22: 500-517.

Lugo, A. E. R. R. Twilley and C. Patterson-Zucca. 1980. The role of black mangrove forests in the productivity of coastal ecosystems in south Florida. Report to the EPA Corvallis Research Laboratory, Corvallis, OR. 281 pp.

Lugo, A. E., M. Sell, and S. C. Snedaker. 1976. Mangrove ecosystem analysis. Pages 113-145 In B. C. Patten Ed. *Systems analysis and simulation in ecology*. Academic Press NY.

Lugo, A. E. and S. C. Snedaker. 1975. Properties of a mangrove forest in southern Florida. Pages 170-211 In G. Walsh, S. Snedaker, and H. Teas Eds. *Proceedings of the international symposium on the biology and management of mangroves*. Univ. of Fl. Gainesville.

Lugo, A. E. and S. C. Snedaker. 1974. The ecology of mangroves. Ann Rev. Ecol. Syst. 5: 39-64.

Madden, C. J. and D. H. Grossman. 2004. A framework for a coastal/marine ecological classification standard. NatureServe, Arlington VA. 169 pp.

Madden, C. J. 2004. Unpublished in-situ SCUBA observations of die-off event in Barnes Key Sound in November 2004.

Madden, C. J., A. M. McDonald, M .J. Hunt, W. M. Kemp, and D. Gruber. 2003. Summary: Ecosystem process models of seagrass communities in Florida Bay. Report to Florida Bay Seagrass Modeling Program. USGS Biological Resources Division.

Madden, C. J. 2002. Unpublished Data. Observations of ponding behind the Buttonwood ridge in a red mangrove (Rhizophora mangle) swamp during large precipitation event in Florida Bay during 2002.

Madden, C. J. 2001. Seagrass habitat recovery and Everglades restoration: Use of an ecological model to assess management strategies in Florida Bay. Florida Bay Science Conference abstract, Apr 23-26, 2001 Key Largo, FL.

Madden, C. J. 1999. Applications of a simulation model of seagrasses for Florida Bay. Florida Bay Science Conference, 1999. Abstract.

Madden, C. J. and J. W. Day, Jr. 1992. An instrument system for high-speed mapping of chlorophyll a and physico-chemical variables in surface waters. Estuaries 15(3): 421-427.

Madden, C. J. Unpublished Data. Observations of condition of red mangroves (Rhizophora mangle) during a hypersalinity event in Florida Bay during 2001.

Maltby, E. and P.J. Dugan. 1994. Wetland ecosystem protection, management, and restoration: An international perspective. Pages 29-46 In Davis, S.M. and J.C. Ogden Eds. *Everglades: The Ecosystem and its Restoration.* St. Lucie Press, Delray Beach, Florida. 826 pages.

Manooch, C. S., III. 1984. Fisherman's guide to the fishes of the southeastern United States, North Carolina State Museum of Natural History, Raleigh, NC, 362 pp.

Manooch, C. S., III and R. H. Matheson III. 1981. Age, growth and mortality of gray snapper collected from Florida waters. Proc. Ann. Conf. S.E. Assoc. Fish & Wildl. Agencies 35: 331–344.

Marx, J.M. and Herrnkind, W.F. 1985. Macroalgae (Rhodophyta: *Laurencia* spp.) as habitat for juvenile spiny lobsters, *Panulirus argus*. Bulletin Marine Science 36:423-431.

Matheson, R. E., Jr., D. K. Camp, S. M. Sogard, and K. Bjorgo. 1999. Changes in seagrass-associated fish and crustacean communities on Florida Bay mud banks: The effects of recent ecosystem changes? Estuaries 22(2B):534-551.

Maul. G.A. and D.M. Martin. 1993. Sea level rise at Key West, Florida, 1846-1992: America's longest instrument record? Geophysical Research Letters 20(18): 1955-1958.

Mazzotti, F. J. 1989. Factors affecting the nesting success of the American crocodile Crocodylus acutus in Florida Bay. Bull. Mar. Sci. 44(1): 220-228.

Mazzotti, F. J. 1999. The American crocodile in Florida Bay. Estuaries 22(2B): 552-561.

McIvor, C.C., J.A. Ley, and R.D. Bjork. 1994. Changes in freshwater inflow from the Everglades to Florida Bay including effects on biota and biotic processes: A review. Pages 117-145 In Davis, S.M. and J.C. Ogden Eds. *Everglades: The Ecosystem and its Restoration.* St. Lucie Press, Delray Beach, Florida. 826 pages.

McMichael, R. H. J. and K. M. Peters. 1989. Early life history of spotted seatrout, *Cynoscion nebulosus* (Pisces: Sciaenidae) in Tampa Bay, Florida, U.S.A. Estuaries 12: 98-110.

Meyers, T. R. and J. D. Hendricks. 1982. A summary of tissue lesions in aquatic animals induced by controlled exposures to environmental contaminants, chemotherapuetic agents, and potential carcinogens. Mar. Fish. Rev. 44: 1–17.

Mintz, J.D., R.N. Lipcius, D.B. Eggleston, and M.S. Seebo. 1994. Survival of juvenile Caribbean spiny lobster: effects of shelter size, geographic location, and conspecific abundance. Marine Ecology Progess Series 112:255-266.

Moody, W. D. 1950. A study of the natural history of spotted seatrout, *Cynoscion nebulosus*, in the Cedar Key, Florida area. Q. J. Fla. Acad. Sci. 12: 147–171.

Murphy, M. D. 2003. A stock assessment of spotted seatrout, *Cynoscion nebulosus*, in Florida: status of the stocks through 2001, Florida Fish and Wildlife Conservation Commission Florida Marine Research Institute In-House Report Number IHR2003-0, Florida Fish and Wildlife Conservation Commission Florida Marine Research Institute In-House Report Number IHR2003-0.

Nagelkerken, W. P. 1981. Distribution and ecology of the groupers (Serranidae) and snappers (Lutjanidae) of the Netherlands Antilles. Found. Scientif. Research Surinam and the Netherland Antilles 107: 71 pp.

National Atlas. 2005. National Atlas of the United States of America. 17 March 2005. U.S. Department of Interior. Available at: http://www.nationalatlas.gov/ (accessed 14 February 2005).

National Atmospheric Deposition Program: Mercury Depostion Program (NADP/MDP). 2003. National Atmospheric Deposition Program. Champaign, Illinois. http://nadp.sws.uiuc.edu/mdn/.

National Marine Fisheries Service (NMFS). 1995. Our living oceans: report of the status of U.S. living marine resources. NOAA Tech. Mem. NMFS-F/SPO-19.

National Oceanic and Atmospheric Administration (NOAA). 2005. March 14, 2005 last update. NOAA's South Florida Ecosystem Research and Monitoring Program – Research: Archived Data. Available: http://www.aoml.noaa.gov/sfp/data_archives.shtml. (accessed 14 February 2005).

National Oceanic and Atmospheric Administration (NOAA). 2004. Species of concern and candidate species. NOAA National Marine Fisheries Service. http://www.nmfs.noaa.gov/pr/species/concern/profiles/goliath_grouper.pdf. (accessed 14 February 2005).

National Park Service (NPS). 2005a. NR-GIS Metadata and Data Store: NPS Natural Resource and GIS Programs. 17 March 2005. National Park Service. Fort Collins, Colorado. http://science.nature.nps.gov/nrdata/.

National Park Service (NPS). 2005b. NPSpecies database. Version 2.2. Unpublished data. 2004. National Park Service. Fort Collins, Colorado.

National Park Service (NPS). 2005c. NPS ESA database. Unpublished data. 2004. National Park Service. Fort Collins, Colorado.

National Park Service (NPS). 2003. Everglades National Park: General Management Plan – Newsletter #2, September 2003. 17 March 2005. http://www.nps.gov/ever/gmp/ern2.pdf/.

National Park Service (NPS). 2002. Park Statistics. Everglades National Park. 1 February 2005. http://www.nps.gov/ever/presskit/stats.htm.

National Park Service (NPS). 1999. Park Establishment. Everglades National Park. 17 March 2005. National Park Service. http://www.nps.gov/ever/eco/nordeen.htm.

National Park Service (NPS). 1997. Wading Birds and the Water Cycle. Everglades National Park. 17 March 2005. National Park Service. http://www.nps.gov/ever/eco/wading.htm.

Nelson, G. A. and D. Leffler. 2001. Abundance, spatial distribution, and mortality of young-ofthe-year spotted seatrout (*Cynoscion nebulosus*) along the Gulf Coast of Florida. Gulf Mex. Sci. 19: 30–42.

Nelson, T.A., G. Garte, C. Featherstone, H.R. Wanless, J.H. Trefry, W. Kang, S. Metz, C. Alvarez-Zarikian, T. Hood, P. Swart, G. Ellis, P. Blackwelder, L. Tedesco, C. Slouch, J.F. Pachut, and M. O'Neil. 2002. Linkages between the south Florida peninsula and coastal zone: A sediment-based history of natural and anthropogenic influences. Pages 415-449 In: J.W. Porter, K.G. Porter Eds: *The Everglades, Florida Bay, and Coral Reefs of the Florida Keys: An Ecosystem Sourebook*. CRC Press, Boca Raton, FL, 1000 pp.

Nova 2005. The Everglades Restoration. The Oceantic Center of Nova Southeastern University. http://www.nova.edu/ocean/eglades/. (accessed 14 February 2005).

Nuttle, W., J. Hunt, and M. Robblee. 2003. Florida Bay Science Program: A synthesis of research on Florida Bay. Florida Caribbean Science Center, Gainesville, Florida.

Nuttle, W.K., J.W. Fourqurean, B.J. Crosby, J.C. Zieman, and M.B. Robblee. 2000. Influence of net freshwater supply on salinity in Florida Bay. Water Resources Research 36(7): 1805-1822.

Odum, W. E., C. McIvor, and T. J. Smith III. 1982. The ecology of mangroves in south Florida: A community profile. FWS/OBS-82/24. 144 pp.

Odum, W. E. and E. J. Heald. 1975. The detritus-based food web of an estuarine mangrove community. Pp 265-286 In: *Estuarine Research*. Academic Press New York.

Odum, W. E. and E. J. Heald. 1972. Trophic analysis of an estuarine mangrove community. Bull Mar Sci 22: 671-738.

Odum, W. E. 1971. Pathways of energy flow in a southern Florida estuary. Univ. Miami Sea GrantProgr. Sea Grant Tech. Bull. 7.

Odum, W. E. 1970. Pathways of energy flow in a south Florida estuary. Ph.D. dissertation. Univ. of Miami FL., 162 pp.

Ogden, J.C. 1994. A comparison of wading bird nesting colony dynamics (1931-1946 and 1974-1989) as an indication of ecosystem conditions in the southern Everglades. Pages 533-570 In Davis, S.M. and J.C. Ogden Eds. *Everglades: The Ecosystem and its Restoration.* St. Lucie Press, Delray Beach, Florida. 826 pages.

Orlando, S. P., Jr., M. B. Robblee, and C. J. Klein. 1997. Salinity characteristics of Florida Bay: a review of the archived data set (1955–1995), Office of Ocean Resources Conservation and Assessments, Office of Ocean Resources Conservation and Assessments, NOAA, Silver Spring, MD, 33 pp.

Ortner, P. R., M. J. Dagg, J. Rabelais, and G. Thayer. 2001. Mesozooplankton abundance variability within Florida Bay (1994–2000). 2001 Florida Bay Science Conference.

Owens, M. and J. C. Cornwell. 2001. Nitrogen cycling in Florida Bay mangrove environments: sediment-water exchange and denitrification. Abstract from 2001 Florida Bay Science Conference, Key Largo, Florida.

Pedersen, O., T. Binzer and J. Borum. 2004. Sulfide intrusion in eelgrass (Zostera marina L.). Plant, Cell and Environment, 27, 595–602.

Perry, W.B. 2005. Personal Communication. Ecologist, Everglades National Park. National Park Service. Homestead, Florida.

Phlips, E.J., S. Badylak, and T.C. Lynch. 1999. Blooms of the picoplanktonic cyanobacterium *Synechococcus* in Florida Bay, a subtropical inner-shelf lagoon. Limnology and Oceanography 44(4): 1166-1175.

Phlips, E. J. and S. Badylak. 1996. Spatial variability in phytoplankton standing crop and composition in a shallow inner-shelf lagoon, Florida Bay, Florida. Bull. Mar. Sci. 58:203-216.

Polovina, J. J. and S. Ralston. 1987, Polovina, J. J. and S. Ralston Eds. *Tropical snappers and groupers: biology and fisheries management.* Westview Press, Boulder, CO, 659 pp.

Porter, J.W. and K.G. Porter. 2002. *The Everglades, Florida Bay, and Coral Reefs of the Florida Keys: an Ecosystem Sourcebook.* CRC Press, New York.

Porter, L., M. Butler, and R.H. Reeves. 2001. Normal bacterial flora of the spiny lobster *Panulirus argus* and its possible role in shell disease. Marine and Freshwater Research 52: 1401-1405.

Powell, A. B., R. T. Cheshire and E.H. Laban. 2004. Growth, mortality and hatchdate distributions of larval and juvenile spotted seatrout (*Cynoscion nebulosus*) in Florida Bay, Everglades National Park. Fish. Bull. 102:142-155.

Powell, A. B., M. W. Lacroix, and R. T. Cheshire. 2002. An evaluation of northern Florida Bay as a nursery area for red drum, *Scianeops ocellatus*, and other juvenile and small resident fishes. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-485. 29 pp.

Powell, A. B. 2002. Distribution and abundance of ichthyoplankton in Florida Bay. NOAA Tech. Mem. NMFS-SEFSC-488: 22 pp.

Powell, A. B, G. W. Thayer, M. Lacroix, and R. Cheshire. 2001. Interannual changes in juvenile and small resident fish assemblages, and seagrass densities in Florida Bay. 2001 Florida Bay Science Conference.

Powell, A. B., D. E. Hoss, W. F. Hettler, D. S. Peters, and S. Wagner. 1989b. Abundance and distribution of ichthyoplankton in Florida Bay and adjacent waters. Bull. Mar. Sci. 44: 35–48.

Powell, G.V.N., R.D. Bjork, J.C. Ogden, R.T. Paul, A. Harriett Powell, W.B. Robertson, Jr. 1989a. Population trends in some Florida Bay wading birds. Wilson Bull. 101(3): 436-357.

Prakash, A and M. Rashid. 1968. Influence of humic substances on the growth of marine phytoplankton: Dinoflagellates. Limnol Oceanogr. 13: 598-606.

Program Management Committee (PMC). 2004. The strategic science plan for Florida Bay. August 2004.

Qualls, R. G. and C. J. Richardson. 2003. Factors controlling concentration, export and decomposition of dissolved organic nutrients in the Everglades of Florida. Biogeochemistry 62(2):197-229.

Randall, J. E. 1996. *Caribbean reef fishes*, 3rd ed., T. F. H. Publications, Neptune City, NJ. Reid, S. K., Jr. 1954. An ecological study of the Gulf of Mexico fishes, in the vicinity of Cedar Key, Florida. Bull. Mar. Sci. Gulf Caribb. 4: 1–94.

Rasmussen, E. 1977. The wasting disease of eelgrass (Zostera marina) and its effect on environmental factors and fauna. Pp. 1-52 In: C. P. McRoy and C. Helfferich [eds] Seagrass ecosystems: A scientific perspective. M. Dekker, New York. 314 pp.

Rays, W. 2005. Personal Communication. Park Ranger, Florida Bay District of Everglades National Park. 1-June-2005.

Reid, G. K. Jr. 1954. An ecological study of the Gulf of Mexico fishes, in the vicinity of Cedar Key, Florida. Bull. Mar. Sci. Gulf and Caribb. 4:1-94.

Restoration Coordination and Verification (RECOVER). 2005. The Recover team's recommendations for interim goals and interim targets for the Comprehensive Everglades Restoration Plan. February 17, 2005. Restoration Coordination and Verification. Comprehensive Everglades Restoration Plan. Central and Southern Florida Project.

Richards, W. J., K. C. Lindeman, J. L. -Schults, J. M. Leis, A. Ropke, M. E. Clarke, and B. Comyns. 1994. Preliminary guide to the identification of the early life history stages of lutjanid fishes of the western central Atlantic. NOAA Tech. Mem. NMFS-SEFSC-345: 49 pp.

Richards, W. J. and V. P. Saksena. 1980. Description of larvae and early juveniles of laboratory reared gray snapper, *Lutjanus griseus* (Linnaeus) (Pisces, Lutjanidae). Bull. Mar. Sci. 30: 515–521.

Rivera-Monroy, V. H., C. J. Madden, J. W. Day, Jr., R. R. Twilley, F. Vera-Herrera, and H. Alvarez-Guillen. 1998. Seasonal coupling of a tropical mangrove forest and an estuarine water column: enhancement of aquatic primary productivity. Hydrobiol. 279:41-53.

RMB Consulting & Research, Inc. 2002. Atmospheric mercury emissions from major point sources – Broward, Dade, and Palm Beach Counties, 1980-2000. Draft Final Report, RMB Consulting & Research, Inc., Raleigh, NC.

Robbins. 1963. The state of research in the National Park System. A report to the National Academy of Sciences Advisory Committee on Research in the National Parks.

Robblee, M.B., G. Clement, D. Smith, and R. Halley. 2001. Salinity Pattern in Florida Bay: A synthesis: 1900-2000. Abstracts 2001 Florida Bay Science Conference. Key Largo, FL.

Robblee., M. B., T. R. Barber, P. R. Carlson, M. J. Durako, J. W. Fourqurean, L. K. Muehlstein, D. Porter, L. A. Yarbro, R. T. Zieman, and J. C. Zieman. 1991. Mass mortality of the tropical seagrass *Thalassia testudinum* in Florida Bay (USA). Mar. Ecol. Prog. Ser. 71:297-299.

Robins, C. R. and G. C. Ray. 1986. *A field guide to Atlantic coast fishes of North America*, Houghton Mifflin Company, 354 p.

Rood, B.E., J.F. Gottgens, JJ. Delfino, C.D. Earle and T.L. Crisman. 1995. Mercury accumulation trends in Florida Everglades and savannas marsh flooded soils. Water, Air & Soil Pollution 80: 981-990.

Rooker JR, Dennis GD. 1991. Diel, lunar and seasonal changes in a mangrove fish assemblage off southwestern Puerto Rico. Bull Mar Sci 49: 684-698.

Ross, M.S., J.F. Meeder, J.P. Sah, P.L. Ruiz, and G.J. Telesnicki. 2000. The southeast saline Everglades revisited: 50 years of coastal vegetation change. Journal of Vegetation Science 11: 101-112.

Rudnick, D. T., Z. Chen, D. L. Childers, J. N. Boyer, and T. D. Fontaine III. 1999. Phosphorus and nitrogen inputs to Florida Bay: The importance of the Everglades watershed. Estuaries 22(2b): 398-416.

Rumbold, D., L. Fink, N. Niemeyer, D. Evans, D. Krabbenhoft, and M. Olson. 2003. Source identification of Florida Bay's methylmercury problem: Mainland runoff versus atmospheric deposition and *in situ* production. Presented at the 2003 Joint Conference on the Science and Restoration of the Greater Everglades and Florida Bay Ecosystem. Palm Harbor, Florida. 13-18 April 2003. Available at: http://conference.ifas.ufl.edu/jc/FB.pdf [March 11, 2005].

Rutherford, E. S., T. W. Schmidt, and J. T. Tilmant. 1989. Early life history of spotted seatrout (*Cynoscion nebulosus* and gray snapper (*Lutjanus griseus*) in Florida Bay, Everglades National Park, Florida. Bull. Mar. Sci. 44: 49–64.

Rutherford, E. S., E. B. Thue, and D. G. Buker. 1983. Population structure, food habitats, and spawning activity of gray snapper, Lutjanus griseus, in Everglades National Park. South Florida Research Center Report SFRC–82/02, National Park Service, Homestead, FL, 41 pp.

Sadovy, Y. and A. Eklund. 1999. Synopsis of biological data on the Nassau Grouper, *Epinephelus striatus* (Bloch, 1792), and the Jewfish, *E. itajara* (Lichtenstein, 1822). NOAA Technical Report NMFS–146: 65 pp.

Schmidt, T.W. 2005. Personal Communication. Marine Biologist, Everglades National Park. National Park Service. Homestead, Florida.

Schmidt, T. W., J. Obsborne, J. Kalafarski, C. Greene, G. Delgado, and M. Alvarado. 2002. Year 2001 Annual Fisheries Report Everglades National Park. South Florida Natural Resources Center, Homestead, FL.

Schmidt, T. W., G. Delgado, and M. Alvarado. 2001. Assessment of the recreational sport fisheries of Florida Bay and adjacent waters from 1985–1998. Proceedings of the 52nd Gulf and Caribbean Fishery Institute 52: 385–401.

Schmidt, T. W., J. S. Ault, and J. A. Bohnsack. 1999. Site characterization for the Dry Tortugas region: fisheries and essential habitat, Florida Keys National Marine Sanctuary and National Park Service. NOAA Tech. Mem. NMFS-SEFSC-425: 95 pp.

Schmidt, T. W. and M. B. Robblee. 1994. Causes of fish kills in the Flamingo area of Everglades National Park. Bull. Mar. Sci. 54: 1083.

Schmidt, T. W. 1993. Report on fish kill investigations in the Flamingo Area of Everglades National Park. South Florida Research Center Project Report. Everglades National Park, Homestead.

Schmidt, T.W. 1979. Ecological study of fishes and the water quality characteristics of Florida Bay, Everglades National Park, Florida. Everglades National Park Technical report RSP-EVER-N36. Republished on NOAA/AOML website http://www.aoml.noaa.gov./general/lib/cedar79.pdf.

Schmidt, T. W. 1977. Effects of hypersalinity on estuarine and marine fishes in Florida Bay, Everglades National Park, Florida. Proceedings of the 57th annual meeting of the American Society of Ichthyologists and Herpetologists.

Science Coordinating Team (SCT), South Florida Ecosystem Restoration Working Group. 2003. The role of flow in the Everglades ridge and slough landscape. http://sofia.er.usgs.gov/publications/papers/sct_flows/.

Serafy, J. E., K. C. Lindeman, T. E. Hopkins, and J. S. Ault. 1997. Effects of freshwater canal discharge on fish assemblages in a subtropical bay: field and laboratory observations. Mar. Ecol. Prog. Ser. 160: 161–172.

Settle, L. 2003. Personal communication. June 2003. Fishery Biologist. NOAA Fishery Service, Beaufort, NC.

Sharp, W.C., W.A. Lellis, M.J. Butler, W.F. Herrnkind, J.H. Hunt, M. Pardee-Woodring, and T.R. Matthews. 2000. The use of coded microwire tags for mark-recapture studies of juvenile Caribbean spiny lobster, Panulirus argus. Journal of Crustacean Biology 20: 510-521

Sheriden, P. F. 1992. Comparative habitat utilization by estuarine macrofauna within the mangrove ecosystem of Rookery Bay, Florida. Bull. Mar. Sci. 50(1): 21-39.

Shields, J. and D. Behringer. 2004. New pathogenic virus in Caribbean spiny lobster *Panulirus argus* from the Florida Keys. Diseases Aquatic Organisms 59: 109-118.

Short, F.T., B. W. Ibelings, and C. denHartog. 1988. Comparison of a current eelgrass disease to the wasting disease in the 1930s, Aquatic Bot. 30:295-304.

Silberman, J. D., S. K. Sarver, and P. J. Walsh. 1994. Mitochondrial DNA variation and population structure in the spiny lobster *Panulirus argus.* Mar. Biol. 120:601-608.

Silberman, J. D., and P. J. Walsh. 1994. Population genetics of the spiny lobster *Panulirus argus.* Bull. Mar. Sci. 54:1084.

Sklar, F., C. McVoy, R. VanZee, D.E. Gawlik, K. Tarboton, D. Rudnick, ShiLi Miao, and T. Armentano. 2002. The effects of altered hydrology on the ecology of the Everglades. Pages 39-82 in J.W. Porter and K.G. Porter Eds. *The Everglades, Florida Bay, and Coral Reefs of the Florida Keys: An ecosystem sourcebook.* CRC Press. Boca Raton, Florida. 1000 pp.

Sklar, F., C. McVoy, R. Van Zee, D. Gawlik, D. Swift, W. Park, C. Fitz, Y. Wu, D. Rudnick, T. Fontaine, S Miao, A. Ferriter, S. Krupa, T. Armentano, K. Tarboton, K. Rutchey, Q. Dong, and S. Newman. 1999. Hydrologic Needs: The effects of altered hydrology on the Everglades. Pages 2-1/2-70 In: Everglades Interim Report. South Florida Water Management District, West Palm Beach, Florida.

Smith, C. L. 1997. National Audubon Society field guide to tropical marine fishes: Caribbean, Gulf of Mexico, Florida, Bahamas, Bermuda, Chanticleer Press, New York, 718 p.

Smith, C. L. 1971. A revision of the American groupers: *Epinephelus* and allied genera. Bull. Amer. Mus. Nat. Hist. 146: 69–241.

Smith, G. G. 1976. Ecology and distribution of eastern Gulf of Mexico reef fishes. Fla. Mar. Res. Publ. No. 19: 78 pp.

Smith, K.N. and W.F. Herrnkind. 1992. Predation on early juvenile spiny lobsters *Panulirus argus* (Latreille): influence of size and shelter. Journal Experimental Marine Biology and Ecology 157: 3-18

Smith, N. P., and P. A. Pitts. 2002. Regional-scale and long-term transport patterns in the Florida Keys. Pages 343-360 In Porter and Porter Eds. *The Everglades, Florida Bay and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook.* CRC Press Boca Raton. 1000 pp.

Smith, N.P. 1998. Tidal and long-term exhanges through channels in the middle and upper Florida Keys. Bulletin of Marine Science 62: 199-211.

Smith, N. P. 1994. Long-term Gulf-to-Atlantic transport through tidal channels in the Florida Keys. Bull. Mar. Sci. 54:602-609.

Smith, T.J, III. 1992. Forest structure. Pages 101-136 in A. I. Robertson and D. M. Alongi, eds. Tropical mangrove ecosystems. American Geophysical Union, Washington, DC.

Smith, T.J. III, J.H. Hudson, M.B. Robblee, G.V.N. Powell, and P.J. Isdale. 1989. Freshwater flow from the Everglades to Florida Bay: A historical reconstruction based on fluorescent banding in the coral *Solenastrea bournoni*. Bulletin of Marine Science 44(1): 274-282.

Snedaker, S. 1989. Overview of the ecology of mangroves and information needs for Florida Bay. Bull. Mar. Sci. 44(1): 341-347.

Sogard, S. M., G. V. N. Powell, and J. G. Holmquist. 1989. Spatial distribution and trends in abundance of fishes residing in seagrass meadows on Florida Bay mudbanks. Bull. Mar. Sci. 44(1):179-199.

South Florida Water Management District (SFWMD). 2005a. DBHYDRO browser. South Florida Water Management District. http://www.sfwmd.gov/org/ema/dbhydro/index.html (accessed 14 February 2005).

South Florida Water Management District (SFWMD). 2005b. Everglades/Florida Bay: Photos. South Florida Water Management District. http://www.sfwmd.gov/org/wrp/wrp_evg/2_wrp_evg_info/photos_seagrass.html. (accessed 14 February 2005).

Springer, V. G. and K. D. Woodburn. 1960. An ecological study of the fishes of the Tampa Bay area. Fla. Board Conserv. Mar. Lab. Prof. Pap. Ser. No. 1.

Starck, W. A., II. 1970. Investigations on the gray snapper, *Lutjanus griseus*. Stud. Trop. Oceanogr. Miami 10: 11–150.

Steidinger, K., W. Richardson, M.B. Neely, G. McRae, S. Richards, R. Bray, T.H. Perkins, and C.R. Tomas. 2001. Florida Bay microalgal blooms. Abstracts 2001 Florida Bay Science Conference. Key Largo, FL.

Stevely, J.M. and D.E. Sweat. 1999. The recovery of sponge populations in Florida Bay and the Upper Keys following a widespread sponge mortality. Abstract. 1999 Florida Bay and Adjacent Marine Systems Science Conference, Key Largo, FL.

Strober, Q.J., K. Thorton, R. Jones, J. Richards, C. Ivey, R. Welch, M. Madden, J. Trexler, E. Gaiser, D. Scheidt, and S. Rathbun. 2001. South Florida Assessment: Phase I/II (Summary) – Everglades stressor interactions: Hydropatterns, eutrophication, habitat alteration, and mercury contamination. United States Environmental Protection Agency, EPA 904-R-01-002.

Strom, D.G. and G.A. Graves. 2001. A comparison of mercury in estuarine fish between Florida Bay and Indian River Lagoon, Florida, U.S.A. Estuaries 24: 597-609.

Stumpf, R. P., M. L. Frayer, M. J. Durako, and J. C. Brock. 1999. Variations in water clarity and bottom albedo in Florida Bay from 1985-1997. Estuaries 22(2B): 431-444.

Sumner, F. B., R. C. Osburn, and L. J. Cole. 1911. A biological survey of the waters of Woods Hole. Bull. U.S. Bur. Fish. 31: 549–794.

Swain, E. D. 2000. Two-Dimensional Simulation of Flow and Transport to Florida Bay Through the Southern Inland and Coastal Systems (SICS). WRIR Report.

Swart, P.K., G. Healy, L. Greer, M. Lutz, A. Saied, D. Anderegg, R.E. Dodge, and D. Rudnick. 1999. The use of proxy chemical records in coral skeletons to ascertain past environmental conditions in Florida Bay. Estuaries 22(2B): 384-397.

Swart, P.K., G.F. Healy, R.E. Dodge, P. Kramer, J.H. Hudson, R.B. Halley, and M Robblee. 1996. The stable oxygen and carbon isotopic record from a coral growing in Florida Bay: A 160 year old record of climatic and anthropogenic influence. Palaeogeography, Palaeoclimatology, Palaeoecology 123: 219-237.

Tabb, D. C. and Roessler. 1989. History of studies on juvenile fishes of coastal waters of Everglades National Park. Bull. Mar. Sci. 44(1): 23-34.

Tabb, D. C. 1966. Treasure those estuaries! Proceedings of the Gulf and Caribbean Fisheries Institute 18: 47–50.

Tabb, D. C., D. L. Dubrow, and R. B. Manning. 1962. The ecology of northern Florida Bay an dadjacent estuaries. Technical Series #39 State of Florida Board of Conservation, Miami FL.

Tabb, D. C., and R. B. Manning. 1961. A checklist of the flora and fauna of northern Florida Bay and adjacent brackish waters of the Florida mainland collected during the period July 1957 through September 1960. Bull. Mar. Sci. Gulf and Caribb. 11: 552-649.

Taniguchi, A. K. 1980. Effects of salinity, temperature, and food abundance upon survival of spotted seatrout eggs and larvae. Abstract. P 16 in Proc. Colloquium on the Biology and Management of Red Drum and Seatrout. Gulf States Marine Fishery Commission.

Thayer, G. W., A. B. Powell, and D. E. Hoss. 1999. Composition of larval, juvenile, and small adult fishes relative to changes in environmental conditions in Florida Bay. 22(2B):518-533.

Thayer, G. W. and A. J. Chester. 1989. Distribution and abundance of fishes among basin and channel habitats in Florida Bay. Bull. Mar. Sci. 44: 200–219.

Thayer, G. W., D. R. Colby, and W. F. Hettler Jr. 1987. Utilization of the red mangrove prop root habitat by fishes in south Florida. Mar. Ecol. Prog. Ser. 35: 25–38.

Thompson, R. and J. L. Monro. 1978. Aspects of the biology and ecology of Caribbean reef fishes: Serranidae (hinds and groupers). J. Fish Biol. 12: 115–146.

Tilmant, J.T. 2005. Personal Communication. Fishery Biologist. Water Resources Division. National Park Service. Fort Collins, CO.

Tilmant, J. T. 1989. A history and an overview of recent trends in the fisheries of Florida Bay. Bull Mar. Sci. 44(1):3-22.

Tomas, C.R., B. Bendis, and K. Johns. 1999. Role of nutrients in regulating plankton blooms in Florida Bay. Pages 323-337 In H. Kumpf, K. Steidinger, and K. Sherman Eds. *The Gulf of Mexico Large Marine Ecosystem: Assessment, Sustainability, and Management.* Blackwell Science. Malden, MA. 704 pp.

United States Army Corps of Engineers (USACE). 2005. Modified Water Deliveries to Everglades National Park and South Dade Canals (C-111) Projects. United States Army Corps of Engineers. http://www.saj.usace.army.mil/dp/MWDC111.htm. (accessed 14 March 2005).

United States Department of Commerce (USDOC). 2005. NOAA–NMFS website http://www.nmfs.noaa.gov/.

United States Department of Commerce (USDOC). 2003. NOAA–NMFS website http://www.nmfs.noaa.gov/.

University of Florida (UF). 2005. Florida Museum of Natural History Website http://www.flmnh.ufl.edu/.

United States Census Bureau (USCB). 1995. Florida: Population of Counties by Decennial Census: 1900 to 1990. U.S. Census Bureau. http://www.census.gov/population/cencounts/fl190090.txt. (accessed 14 February 2005).

United States Geologic Survey (USGS). 2001. R. W. Schaffranek, H. L. Jenter, E. D. Swain, C. D. Langevin, and Kevin Kotun. Tides and inflows in the mangrove ecotone (TIME) model development. Work Plan Document.

Valiela, I., K.H. Foreman, M. LaMontagne, D. Hersh, J. Costa, C.D. D'Avanzo, M. Babione, Chi-Ho Sham, J. Brawley, P. Peckol, B. DeMeo-Anderson, K. Lajtha. 1991. Coupling of watersheds and coastal waters: Sources and consequences of nutrient enrichment in Waquoit Bay. Estuaries 15:443-457.

Wakeman, J. M. and D. E. Wolschlag. 1977. Salinity stress and swimming performance in spotted seatrout. Proceedings Annual Conference Southeast Association of Fish and Wildlife Agencies 31: 357–361.

Wanless, H. R., R. W. Parkinson, and L. P. Tedesco. 1994. Sea level control on stability of Everglades wetlands. Pages 199-224 In Davis, S.M. & J.C. Ogden Eds. *Everglades: The ecosystem and its restoration*. St. Lucie Press, Delray Beach, Florida. 826 pp.

Warren, C. E. 1971. Biology and water pollution control, W. B. Saunders Company, Philadelphia, PA, 434 pp.

Weinstein, M. P., C. M. Courtney, and J. C. Kinch. 1977. The Marco Island estuary: A summary of physiochemical and biological parameters. Fla. Sci 40(2):98-124.

Wuenschel, M. 2002. Bioenergetics of larval and juvenile spotted seatrout (*Cynoscion nebulosus*). Unpublished Ph.D. Dissertation, State University of New York College of Environmental Science and Forestry, State University of New York College of Environmental Science and Forestry, Syracuse, NY.

Wuenschel, M.M., R. G.Werner, D. E. Hoss, and A. B. Powell. 2001. Bioenergetics of larval spotted seatrout (*Cynoscion nebulosus*) in Florida Bay. 2001 Florida Bay Science Conference:.

Yeung, C., D. L. Jones, M. M. Criales, T. L. Jackson, and W. J. Richards. 2001. Influence of coastal eddies and countercurrents on the influx of spiny lobster, *Panulirus argus*, postlarvae into Florida Bay. Mar. Freshwater Res. 52: 1217–1232.

Yeung, C., D.L. Jones, M.M. Criales, T.L. Jackson, and W.J. Richards. 2000. Influence of coastal eddies and counter-currents on the influx of spiny lobster, Panulirus argus, postlarvae into Florida Bay. Marine and Freshwater Research 52: 1217-1232.

Yeung, C., M. M. Criales, and T. N. Lee. 2000. Unusual larval abundance of *Scyllarides nodifer* and *Albunea* sp. during an intrusion of low-salinity Mississippi flood water in the Florida Keys in September 1993: Insight into larval transport from upstream. J. Geophys. Res. 105: 28741–28758.

Yeung, C. 1996. Transport and retention of lobster phyllosoma larvae in the Florida Keys, University of Miami, University of Miami, Coral Gables, FL, 216 pp.

Yeung, C. and M. F. McGowan. 1991. Differences in inshore-offshore and vertical distribution of phyllosoma larvae of *Panulirus*, *Scyllarus* and *Scyllarides* in the Florida Keys in May-June, 1989. Bull. Mar. Sci. 49: 699–714.

Yokel, B. J. 1975. Rookery Bay land use studies: Environmental planning strategies for the development of a mangrove shoreline No 5. Estuarine Biology. Conservation Foundation, Washington DC. 112 pp.

Zieman, J. C., Univ. of VA. 2002. Personal communication. On-site at Barnes Key during a die-off event, describing the phenomenon in 2002 as being similar to during the large die-off event in 1987.

Zieman, J. C., J. W. Fourqurean and T. A. Frankovich. 1999. Seagrass die-off in Florida Bay: Long-term trends in abundance and growth of turtle grass, Thalassia testudinum. Estuaries 22(2B):460-470.

Zieman, J.C. 1999. Seagrass dieoff in Florida Bay 1989-1999: decadal trends in abundance and growth of *Thalassia testudinum*. 1999 Florida Bay and Adjacent Marine Systems Science Conference Abstract.

Zieman, J.C., Fourqurean, J.W. & Iverson, R.L. 1989. Distribution, abundance and productivity of seagrasses and macroalgae in Florida Bay. Bulletin of Marine Science 44: 292-311.

Zieman, J. C. Jr. 1982. A community profile: The ecology of the seagrass ecosystem of south Florida. US Fish and Wildlife Service, Office of Biological Services. Washington, DC. FWS/OBS-82/25. 123 pp.

Zinkowski, N. 1971. Julia the jewfish. Skin Diver June 1971: 46–50.

Appendix: Everglades Fish Species

(Source: National Park Service 2005b).

Scientific Name	Abundance	Residency	Nativity	Cultivation
Aphrododerus sayanus	Rare	Resident	Native	
Albula vulpes	Rare	Resident	Native	Not cultivated
Anguilla rostrata	Uncommon	Migratory	Native	Not cultivated
Hypoatherina harringtonensis	Unknown	Unknown	Native	Not cultivated
Labidesthes sicculus	Uncommon	Breeder	Native	Not cultivated
Membras martinica	Common	Breeder	Native	Not cultivated
Menidia peninsulae	Uncommon	Breeder	Native	Not cultivated
Synodus intermedius	Unknown	Unknown	Native	Not cultivated
Strongylura marina	Common	Breeder	Native	Not cultivated
Strongylura notata	Common	Breeder	Native	Not cultivated
Strongylura timucu	Common	Breeder	Native	Not cultivated
Tylosurus crocodilus	Rare	Unknown	Native	Not cultivated
Carcharhinus leucas	Rare	Breeder	Native	Not cultivated
Carcharhinus plumbeus	Rare	Unknown	Native	Not cultivated
Galeocerdo cuvier	Rare	Unknown	Native	Not cultivated
Negaprion brevirostris	Rare	Resident	Native	Not cultivated
Rhizoprionodon terraenovae	Unknown	Unknown	Native	Not cultivated
Brevoortia patronus	Rare	Unknown	Native	Not cultivated
Brevoortia smithi	Uncommon	Unknown	Native	Not cultivated
Harengula humeralis	Uncommon	Resident	Native	Not cultivated
Jenkinsia lamprotaenia	Uncommon	Unknown	Native	Not cultivated
Opisthonema oglinum	Common	Breeder	Native	Not cultivated
Sardinella aurita	Uncommon	Unknown	Native	Not cultivated
Sardinella brasiliensis	Rare	Unknown	Native	Not cultivated
Notemigonus crysoleucas	Uncommon	Breeder	Native	Cultivated
Notropis maculatus	Rare	Breeder	Native	Not cultivated
Notropis petersoni	Uncommon	Breeder	Native	Not cultivated
Histrio histrio	Rare	Vagrant	Native	Not cultivated
Dasyatis americana	Uncommon	Unknown	Native	Not cultivated
Acanthurus chirurgus	Rare	Unknown	Native	Not cultivated
Astrapogon alutus	Uncommon	Unknown	Native	Not cultivated
Chasmodes saburrae	Common	Resident	Native	Not cultivated
Hypsoblennius hentz	Unknown	Resident	Native	Not cultivated
Parablennius marmoreus	Rare	Unknown	Native	Not cultivated
Diplogrammus pauciradiatus	Uncommon	Resident	Native	Not cultivated
Caranx bartholomaei	Rare	Unknown	Native	Not cultivated
Caranx crysos	Uncommon	Vagrant	Native	Not cultivated
Caranx latus	Rare	Unknown	Native	Not cultivated
Caranx ruber	Rare	Unknown	Native	Not cultivated
Chloroscombrus chrysurus	Uncommon	Unknown	Native	Not cultivated
Hemicaranx amblyrhynchus	Rare	Unknown	Native	Not cultivated

Scientific Name	Abundance	Residency	Nativity	Cultivation
Oligoplites saurus	Common	Resident	Native	Not cultivated
Selene setapinnis	Occasional	Unknown	Native	Not cultivated
Selene vomer	Uncommon	Unknown	Native	Not cultivated
Seriola zonata	Rare	Vagrant	Native	Not cultivated
Trachinotus falcatus	Uncommon	Resident	Native	Not cultivated
Enneacanthus gloriosus	Uncommon	Breeder	Native	Not cultivated
Lepomis macrochirus	Common	Breeder	Native	Not cultivated
Lepomis marginatus	Abundant	Breeder	Native	Not cultivated
Centropomus parallelus	Rare	Unknown	Native	Not cultivated
Centropomus pectinatus	Rare	Unknown	Native	Not cultivated
Astronotus ocellatus	Uncommon	Breeder	Non-Native	Cultivated
Cichlasoma bimaculatum	Uncommon	Breeder	Non-Native	Cultivated
Eucinostomus harengulus	Unknown	Unknown	Native	Not cultivated
Paraclinus fasciatus	Uncommon	Unknown	Native	Not cultivated
Paraclinus marmoratus	Uncommon	Unknown	Native	Not cultivated
Microdesmus	Unknown	Unknown	Native	Not cultivated
Calamus leucosteus	Occasional	Unknown	Native	Not cultivated
Achirus lineatus	Common	Resident	Native	Not cultivated
Trinectes inscriptus	Rare	Unknown	Native	Not cultivated
Trinectes maculatus	Common	Resident	Native	Not cultivated
Bothus ocellatus	Rare	Unknown	Native	Not cultivated
Symphurus diomedianus	Uncommon	Unknown	Native	Not cultivated
Symphurus plagiusa	Common	Resident	Native	Not cultivated
Ancylopsetta quadrocellata	Uncommon	Unknown	Native	Not cultivated
Etropus crossotus	Uncommon	Unknown	Native	Not cultivated
Syacium gunteri	Uncommon	Unknown	Native	Not cultivated
Bellator egretta	Occasional	Unknown	Native	Not cultivated
Bagre marinus	Common	Breeder	Native	Not cultivated
Hoplosternum littoralis	Occasional	Breeder	Non-Native	Unknown
Balistes capriscus	Rare	Vagrant	Native	Not cultivated
Balistes vetula	Unknown	Unknown	Native	Not cultivated
Canthidermis maculata	Rare	Vagrant	Native	Not cultivated
Aluterus schoepfi	Uncommon	Vagrant	Native	Not cultivated
Aluterus scriptus	Rare	Vagrant	Native	Not cultivated
Monacanthus ciliatus	Common	Resident	Native	Not cultivated
Monacanthus hispidus	Common	Resident	Native	Not cultivated
Amia calva	Rare	Breeder	Native	Not cultivated
Gymnothorax moringa	Occasional	Vagrant	Native	Not cultivated
Gymnothorax nigromarginatus	Rare	Unknown	Native	Not cultivated
Ahlia egmontis	Uncommon	Breeder	Native	Not cultivated
Bascanichthys bascanium	Uncommon	Unknown	Native	Not cultivated
Bascanichthys scuticaris	Uncommon	Unknown	Native	Not cultivated
Myrophis punctatus	Uncommon	Resident	Native	Not cultivated
Ophichthus gomesi	Uncommon	Resident	Native	Not cultivated

Scientific Name	**Abundance**	**Residency**	**Nativity**	**Cultivation**
Atherinomorus stipes	Common	Breeder	Native	Not cultivated
Menidia beryllina	Common	Breeder	Native	Not cultivated
Synodus foetens	Uncommon	Unknown	Native	Not cultivated
Trachinocephalus myops	Unknown	Unknown	Native	Not cultivated
Opsanus beta	Common	Breeder	Native	Not cultivated
Porichthys plectrodon	Uncommon	Unknown	Native	Not cultivated
Prognichthys gibbifrons	Occasional	Unknown	Native	Not cultivated
Chriodorus atherinoides	Uncommon	Breeder	Native	Not cultivated
Hemiramphus balao	Unknown	Unknown	Native	Not cultivated
Hemiramphus brasiliensis	Uncommon	Unknown	Native	Not cultivated
Hyporhamphus unifasciatus	Common	Unknown	Native	Not cultivated
Carcharhinus limbatus	Uncommon	Breeder	Native	Not cultivated
Sphyrna lewini	Unknown	Unknown	Native	Not cultivated
Sphyrna mokarran	Unknown	Unknown	Native	Not cultivated
Sphyrna tiburo	Uncommon	Breeder	Native	Not cultivated
Harengula jaguana	Common	Resident	Native	Not cultivated
Anchoa cubana	Unknown	Unknown	Native	Not cultivated
Anchoa hepsetus	Common	Breeder	Native	Not cultivated
Anchoa lamprotaenia	Rare	Unknown	Native	Not cultivated
Anchoa mitchilli	Abundant	Breeder	Native	Not cultivated
Anchoviella perfasciata	Rare	Unknown	Native	Not cultivated
Erimyzon sucetta	Common	Breeder	Native	Not cultivated
Rivulus marmoratus	Uncommon	Breeder	Native	Not cultivated
Cyprinodon variegatus	Abundant	Breeder	Native	Not cultivated
Floridichthys carpio	Common	Breeder	Native	Not cultivated
Jordanella floridae	Common	Breeder	Native	Cultivated
Adinia xenica	Uncommon	Breeder	Native	Not cultivated
Fundulus chrysotus	Common	Breeder	Native	Cultivated
Fundulus confluentus	Common	Breeder	Native	Not cultivated
Fundulus grandis	Common	Breeder	Native	Not cultivated
Fundulus seminolis	Uncommon	Breeder	Native	Not cultivated
Fundulus similis	Uncommon	Breeder	Native	Not cultivated
Lucania goodei	Abundant	Breeder	Native	Not cultivated
Lucania parva	Abundant	Breeder	Native	Not cultivated
Belonesox belizanus	Common	Breeder	Non-Native	Cultivated
Gambusia holbrooki	Abundant	Breeder	Native	Cultivated
Gambusia rhizophorae	Rare	Breeder	Native	Not cultivated
Heterandria formosa	Abundant	Breeder	Native	Not cultivated
Poecilia latipinna	Abundant	Breeder	Native	Cultivated
Elops saurus	Common	Breeder	Native	Not cultivated
Megalops atlanticus	Common	Resident	Native	Not cultivated
Esox americanus	Occasional	Unknown	Native	Not cultivated
Esox niger	Rare	Unknown	Native	Not cultivated
Anarchopterus criniger	Unknown	Unknown	Native	Not cultivated

Scientific Name	Abundance	Residency	Nativity	Cultivation
Bryx dunckeri	Unknown	Unknown	Native	Not cultivated
Cosmocampus albirostris	Unknown	Unknown	Native	Not cultivated
Hippocampus erectus	Unknown	Unknown	Native	Not cultivated
Hippocampus zosterae	Uncommon	Breeder	Native	Not cultivated
Micrognathus crinitus	Unknown	Unknown	Native	Not cultivated
Syngnathus floridae	Uncommon	Breeder	Native	Not cultivated
Syngnathus louisianae	Uncommon	Breeder	Native	Not cultivated
Syngnathus scovelli	Uncommon	Breeder	Native	Not cultivated
Ogcocephalus nasutus	Rare	Unknown	Native	Not cultivated
Ogcocephalus radiatus	Rare	Unknown	Native	Not cultivated
Agonostomus monticola	Occasional	Migratory	Native	Not cultivated
Mugil cephalus	Abundant	Resident	Native	Not cultivated
Mugil curema	Uncommon	Resident	Native	Not cultivated
Mugil gyrans	Uncommon	Unknown	Native	Not cultivated
Dasyatis sabina	Uncommon	Unknown	Native	Not cultivated
Gymnura micrura	Rare	Unknown	Native	Not cultivated
Gunterichthys longipenis	Occasional	Unknown	Native	Not cultivated
Ogilbia cayorum	Rare	Resident	Native	Not cultivated
Ophidion grayi	Rare	Unknown	Native	Not cultivated
Ophidion holbrooki	Occasional	Unknown	Native	Not cultivated
Ophidion marginatum	Occasional	Unknown	Native	Not cultivated
Ophidion welshi	Occasional	Unknown	Native	Not cultivated
Ginglymostoma cirratum	Rare	Unknown	Native	Not cultivated
Rhincodon typus	Occasional	Unknown	Native	Not cultivated
Caranx hippos	Common	Resident	Native	Not cultivated
Seriola dumerili	Rare	Vagrant	Native	Not cultivated
Trachinotus carolinus	Rare	Resident	Native	Not cultivated
Lepomis gulosus	Common	Breeder	Native	Not cultivated
Lepomis microlophus	Uncommon	Breeder	Native	Not cultivated
Lepomis punctatus	Common	Breeder	Native	Not cultivated
Micropterus salmoides	Uncommon	Breeder	Native	Not cultivated
Pomoxis nigromaculatus	Occasional	Unknown	Native	Not cultivated
Centropomus undecimalis	Uncommon	Breeder	Native	Not cultivated
Cichla ocellaris	Occasional	Unknown	Non-Native	Not cultivated
Cichlasoma managuense	Rare	Breeder	Non-Native	Cultivated
Cichlasoma urophthalmus	Abundant	Breeder	Non-Native	Not cultivated
Hemichromis letourneuxi	Uncommon	Breeder	Non-Native	Cultivated
Oreochromis aureus	Common	Breeder	Non-Native	Cultivated
Oreochromis mossambicus	Occasional	Unknown	Non-Native	Cultivated
Tilapia mariae	Uncommon	Breeder	Non-Native	Cultivated
Coryphaena hippurus	Rare	Vagrant	Native	Not cultivated
Echeneis naucrates	Uncommon	Unknown	Native	Not cultivated
Echeneis neucratoides	Uncommon	Unknown	Native	Not cultivated
Elassoma evergladei	Uncommon	Breeder	Native	Not cultivated

Scientific Name	Abundance	Residency	Nativity	Cultivation
Dormitator maculatus	Rare	Unknown	Native	Not cultivated
Eleotris pisonis	Occasional	Unknown	Native	Not cultivated
Gobiomorus dormitor	Occasional	Unknown	Native	Not cultivated
Chaetodipterus faber	Uncommon	Unknown	Native	Not cultivated
Diapterus plumieri	Uncommon	Resident	Native	Not cultivated
Eucinostomus argenteus	Common	Resident	Native	Not cultivated
Eucinostomus gula	Common	Resident	Native	Not cultivated
Eucinostomus lefroyi	Unknown	Unknown	Native	Not cultivated
Gerres cinereus	Uncommon	Unknown	Native	Not cultivated
Gobiesox strumosus	Unknown	Unknown	Native	Not cultivated
Barbulifer ceuthoecus	Occasional	Unknown	Native	Not cultivated
Bathygobius soporator	Uncommon	Breeder	Native	Not cultivated
Coryphopterus glaucofraenum	Rare	Unknown	Native	Not cultivated
Gobionellus oceanicus	Uncommon	Resident	Native	Not cultivated
Gobionellus saepepallens	Rare	Unknown	Native	Not cultivated
Gobionellus shufeldti	Rare	Unknown	Native	Not cultivated
Gobionellus smaragdus	Uncommon	Breeder	Native	Not cultivated
Gobiosoma bosc	Common	Breeder	Native	Not cultivated
Gobiosoma robustum	Common	Breeder	Native	Not cultivated
Lophogobius cyprinoides	Common	Breeder	Native	Not cultivated
Microgobius gulosus	Common	Breeder	Native	Not cultivated
Microgobius microlepis	Unknown	Unknown	Native	Not cultivated
Microgobius thalassinus	Rare	Resident	Native	Not cultivated
Anisotremus virginicus	Occasional	Unknown	Native	Not cultivated
Haemulon aurolineatum	Uncommon	Resident	Native	Not cultivated
Haemulon chrysargyreum	Occasional	Unknown	Native	Not cultivated
Haemulon flavolineatum	Rare	Unknown	Native	Not cultivated
Haemulon parra	Rare	Unknown	Native	Not cultivated
Haemulon plumieri	Common	Resident	Native	Not cultivated
Haemulon sciurus	Uncommon	Resident	Native	Not cultivated
Orthopristis chrysoptera	Common	Resident	Native	Not cultivated
Bodianus rufus	Occasional	Unknown	Native	Not cultivated
Halichoeres bivittatus	Occasional	Unknown	Native	Not cultivated
Lachnolaimus maximus	Occasional	Unknown	Native	Not cultivated
Lobotes surinamensis	Uncommon	Resident	Native	Not cultivated
Lutjanus apodus	Occasional	Unknown	Native	Not cultivated
Lutjanus griseus	Common	Resident	Native	Not cultivated
Lutjanus jocu	Rare	Unknown	Native	Not cultivated
Lutjanus synagris	Uncommon	Unknown	Native	Not cultivated
Ocyurus chrysurus	Occasional	Vagrant	Native	Not cultivated
Nomeus gronovii	Occasional	Vagrant	Native	Not cultivated
Etheostoma fusiforme	Rare	Breeder	Native	Not cultivated
Polydactylus octonemus	Rare	Unknown	Native	Not cultivated
Holacanthus ciliaris	Occasional	Vagrant	Native	Not cultivated

Scientific Name	Abundance	Residency	Nativity	Cultivation
Abudefduf saxatilis	Occasional	Unknown	Native	Not cultivated
Pomacentrus leucostictus	Occasional	Unknown	Native	Not cultivated
Pomatomus saltatrix	Rare	Migratory	Native	Not cultivated
Priacanthus arenatus	Occasional	Unknown	Native	Not cultivated
Rachycentron canadum	Uncommon	Resident	Native	Not cultivated
Cryptotomus roseus	Occasional	Vagrant	Native	Not cultivated
Nicholsina usta	Uncommon	Unknown	Native	Not cultivated
Scarus guacamaia	Occasional	Vagrant	Native	Not cultivated
Scarus taeniopterus	Unknown	Unknown	Native	Not cultivated
Sparisoma aurofrenatum	Unknown	Unknown	Native	Not cultivated
Sparisoma chrysopterum	Occasional	Unknown	Native	Not cultivated
Sparisoma radians	Unknown	Unknown	Native	Not cultivated
Sparisoma rubripinne	Rare	Unknown	Native	Not cultivated
Sparisoma viride	Unknown	Unknown	Native	Not cultivated
Bairdiella batabana	Uncommon	Unknown	Native	Not cultivated
Bairdiella chrysoura	Common	Breeder	Native	Not cultivated
Cynoscion arenarius	Rare	Unknown	Native	Not cultivated
Cynoscion nebulosus	Common	Breeder	Native	Not cultivated
Equetus acuminatus	Unknown	Unknown	Native	Not cultivated
Equetus lanceolatus	Rare	Unknown	Native	Not cultivated
Equetus umbrosus	Unknown	Unknown	Native	Not cultivated
Larimus fasciatus	Occasional	Unknown	Native	Not cultivated
Leiostomus xanthurus	Uncommon	Unknown	Native	Not cultivated
Menticirrhus americanus	Uncommon	Unknown	Native	Not cultivated
Menticirrhus littoralis	Rare	Unknown	Native	Not cultivated
Menticirrhus saxatilis	Occasional	Unknown	Native	Not cultivated
Micropogonias undulatus	Rare	Unknown	Native	Not cultivated
Pogonias cromis	Rare	Unknown	Native	Not cultivated
Sciaenops ocellatus	Common	Migratory	Native	Not cultivated
Stellifer lanceolatus	Occasional	Unknown	Native	Not cultivated
Scomberomorus cavalla	Occasional	Vagrant	Native	Not cultivated
Scomberomorus maculatus	Uncommon	Migratory	Native	Not cultivated
Scomberomorus regalis	Occasional	Vagrant	Native	Not cultivated
Centropristis striata	Unknown	Unknown	Native	Not cultivated
Diplectrum bivittatum	Unknown	Unknown	Native	Not cultivated
Diplectrum formosum	Rare	Unknown	Native	Not cultivated
Epinephelus itajara	Rare	Resident	Native	Not cultivated
Epinephelus morio	Rare	Unknown	Native	Not cultivated
Epinephelus striatus	Occasional	Vagrant	Native	Not cultivated
Mycteroperca bonaci	Occasional	Resident	Native	Not cultivated
Mycteroperca microlepis	Occasional	Unknown	Native	Not cultivated
Rypticus maculatus	Occasional	Unknown	Native	Not cultivated
Archosargus probatocephalus	Uncommon	Resident	Native	Not cultivated
Archosargus rhomboidalis	Occasional	Unknown	Native	Not cultivated

Scientific Name	Abundance	Residency	Nativity	Cultivation
Calamus arctifrons	Uncommon	Resident	Native	Not cultivated
Calamus penna	Occasional	Unknown	Native	Not cultivated
Lagodon rhomboides	Common	Breeder	Native	Not cultivated
Sphyraena barracuda	Uncommon	Resident	Native	Not cultivated
Sphyraena guachancho	Occasional	Unknown	Native	Not cultivated
Peprilus alepidotus	Unknown	Unknown	Native	Not cultivated
Citharichthys macrops	Occasional	Unknown	Native	Not cultivated
Citharichthys spilopterus	Rare	Unknown	Native	Not cultivated
Paralichthys albigutta	Uncommon	Unknown	Native	Not cultivated
Paralichthys lethostigma	Rare	Unknown	Native	Not cultivated
Paralichthys squamilentus	Occasional	Unknown	Native	Not cultivated
Pristis pectinata	Rare	Breeder	Native	Not cultivated
Rhinobatos lentiginosus	Occasional	Unknown	Native	Not cultivated
Scorpaena albifimbria	Unknown	Unknown	Native	Not cultivated
Scorpaena brasiliensis	Rare	Unknown	Native	Not cultivated
Prionotus rubio	Unknown	Unknown	Native	Not cultivated
Prionotus scitulus	Unknown	Unknown	Native	Not cultivated
Prionotus tribulus	Uncommon	Unknown	Native	Not cultivated
Lepisosteus platyrhincus	Abundant	Breeder	Native	Not cultivated
Arius felis	Common	Breeder	Native	Not cultivated
Clarias batrachus	Uncommon	Breeder	Non-Native	Cultivated
Ameiurus natalis	Common	Breeder	Native	Not cultivated
Ameiurus nebulosus	Occasional	Vagrant	Native	Not cultivated
Noturus gyrinus	Uncommon	Breeder	Native	Not cultivated
Chilomycterus antillarum	Rare	Vagrant	Native	Not cultivated
Chilomycterus schoepfi	Common	Unknown	Native	Not cultivated
Diodon hystrix	Rare	Vagrant	Native	Not cultivated
Stephanolepis hispidus				
Lactophrys quadricornis	Uncommon	Resident	Native	Not cultivated
Lactophrys trigonus	Rare	Unknown	Native	Not cultivated
Sphoeroides nephelus	Uncommon	Breeder	Native	Not cultivated
Sphoeroides spengleri	Rare	Unknown	Native	Not cultivated
Narcine brasiliensis	Occasional	Unknown	Native	Not cultivated