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# Status And Trends Of Emergent And Submerged Vegetated Habitats, Gulf Of Mexico, U.S.A.

U.S. ENVIRONMENTAL PROTECTION AGENCY
GULF OF MEXICO PROGRAM
HABITAT DEGRADATION SUBCOMMITTEE

# REPORT ON THE STATUS AND TRENDS OF EMERGENT AND SUBMERGED VEGETATED HABITATS OF GULF OF MEXICO COASTAL WATERS, U.S.A.

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## CONVERSION FACTORS

| METRIC | | ENGLISH |
|---|---|---|
| 1 meter (m) | = | 39.37 inches (in) |
| 1 meter (m) | = | 3.281 feet (ft) |
| 1 kilometer (km) | = | 0.62 mile (mi) |
| 1 hectare (ha) | = | 2.47 acres (ac) |
| 1 square kilometer ($km^2$) | = | 0.3861 square miles ($mi^2$) |
| 1 square kilometer ($km^2$) | = | 247 acres (ac) |
| 1 kilogram (kg) | = | 2.2046 pounds (lb) |
| 1 metric ton (mt) | = | 1.1 ton (t) |

# PREFACE

This report was prepared by the Habitat Degradation Subcommittee of the U.S. Environmental Protection Agency's Gulf of Mexico Program. Subcommittee members supplied pertinent data and references which were summarized and assembled by the editors of this document. The data and information contained in this review were obtained by dedicated scientists and resource managers who were aware of the ecological and social significance of wetlands, seagrasses, and other Gulf habitats long before their importance was generally recognized. This report is meant to summarize and highlight their contributions to the knowledge of Gulf of Mexico resources.

The authors appreciate the many useful recommendations given during review of the draft document. Those recommendations have helped in making this report on habitat losses in the Gulf of Mexico coastal area as complete a summary as possible. We gratefully acknowledge B. Albrecht, A. Leskowich, and E. Luna of TRI, Inc. for their assistance in reviewing, organizing, and summarizing the data used in this report. We also thank the many colleagues who supplied unpublished data and information, as well as encouragement. We are especially appreciative for the time and information given by W. White, Texas Bureau of Economic Geology, and W. Longley, Texas Water Development Board.

This report is not intended to be a complete literature review of all Gulf of Mexico habitats. Rather, it is a convenient summary of available information on the acreage of saline coastal emergent and submerged vegetated habitats. It is hoped that future reports will address the status and trends of other Gulf of Mexico coastal and nearshore habitats, such as swamps, pine flatwoods, savannahs, beaches and dunes, tidal flats, soft bottoms, hard bottoms, and coral reefs. The Bibliography contains many references not cited in the text and are included for those who wish to pursue a topic in more detail than given herein.

Finally, the authors wish to caution the readers that it is not advisable to compare acreages given in different inventories summarized in this report since different definitions, baselines, and criteria were used in preparation of each inventory. However, in Case Histories we have assembled historical data for selected estuarine systems which were analyzed by the same author for the purpose of identifying trends of habitat acreage.

## EXECUTIVE SUMMARY

The purpose of this report is to present a comprehensive summary of the current status and historical trends of saline vegetated habitats of the coastal region of the Gulf of Mexico, U.S.A. The inland boundary of this survey is based upon the U.S. Fish and Wildlife Service's (FWS) delineation of estuarine habitats and is generally an imaginary line across rivers, bays, sounds and the like where the ocean derived salts measure less than 0.5 ppt during periods of low flow and where there is little measurable tidal influence. The seaward limit of the survey is generally restricted to offshore areas of continually diluted seawater.

Data bases for habitat types within the coastal area of the Gulf were found to be dissimilar. Thus, it was not possible to discuss all habitat types in the same amount of detail. Most of the acreage inventories presented herein are for marsh habitats; however, available published information is summarized for seagrass and mangrove habitats.

Because of dissimilarities of definitions, baselines, and other factors, it is not advisable to compare acreages of habitat type given in different inventories. However, historical information given in Case Histories for selected waterbodies are compared when data from the same author was available for different time periods.

### Inventories of Coastal Habitats and Living Resources

Wetlands are highly productive habitats that are of great value to society and the environment. Wetlands are defined as: "Those areas which are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas" (Federal Interagency Committee for Wetland Delineation, 1989). Coastal wetlands of the Gulf of Mexico are of special interest because of their recognized importance in maintaining the production of the rich Gulf fisheries resources.

A review of historical inventories indicates that acreage of all emergent and submerged vegetated wetland communities in the nation is decreasing. Overall, approximately 54% of the original 215 million ac (87 x $10^6$ ha) which were present at the founding of the country have been lost. There were approximately 108 million ac (44 x $10^6$ ha) of wetlands in the 1950's and 99 million ac (40 x $10^6$ ha) in the 1970's, a loss of over 9 million ac (3.6 x $10^6$ ha). The majority of historic losses were due primarily to agricultural conversion of bottomland hardwoods and other wetland types and development of palustrine and estuarine wetlands. This rapid loss rate, in part, led to increased federal and state regulatory programs governing the conversion of wetlands.

The recently completed Gulf-wide inventory, prepared by the National Oceanographic and Atmospheric Administration (NOAA), is based on FWS National Wetland Inventory maps prepared from 1972 to 1984 aerial photographs. The NOAA report summarizes acreage of coastal wetland types by counties and by selected estuarine drainage areas. Because recent national trends indicate that the amounts of most wetland types are still declining, the acreages presented in the NOAA report may be greater than the actual current acreage of coastal wetlands. The following table summarizes the most recent information on the current status of five wetland categories in Gulf coast states as calculated from NOAA data:

**Totals (Acres) of Selected Wetlands by State for the Gulf of Mexico (from NOAA, 1991)[1,2]**

| State | Salt Marsh | Fresh Marsh (Tidal) | Forested Scrub-Shrub (Estuarine) | Forested Scrub-Shrub (Tidal Fresh) | Tidal Flats | Total | Percentage of Total |
|---|---|---|---|---|---|---|---|
| Texas | 432,000 | 22,500 | 2,600 | 7,400 | 275,100 | 739,600 | 20 |
| Louisiana | 1,722,800 | 65,000 | 10,200 | 4,800 | 31,800 | 1,834,600 | 49 |
| Mississippi | 58,800 | -[3] | 900 | - | 2,300 | 62,000 | 2 |
| Alabama | 25,500 | 100 | 2,800 | 2,000 | 4,100 | 34,500 | 0 |
| Florida | 257,200 | 9,800 | 613,700 | 18,400 | 192,900 | 1,092,000 | 29 |
| TOTALS | 2,496,300 | 97,400 | 630,200 | 32,600 | 506,200 | 3,762,700 | |

[1] Acreage originally reported as acres x 100
[2] Calculations based on FWS wetland inventory maps
[3] None recorded

Seagrass habitat was not included in the NOAA survey. Of the five major coastal wetland habitats included, 66% of the acreage was salt marsh, 17% forested scrub-shrub, 13% tidal flats, 3% tidal fresh marsh, and 1% forested. The distribution of these habitats is not equal across the Gulf states. Louisiana contains most of the Gulf's salt marshes with 69%, followed by Texas (17%), Florida (10%), Mississippi (2%), and Alabama (1%). Texas contains 54% of the tidal flats and Florida has 97% of the estuarine forested scrub-shrub habitats (mostly mangroves).

Other surveys have estimated acreage of seagrass meadows. There are an estimated 800,000 ac (323,887 ha) of seagrasses within Gulf estuaries, and 95% of these are found in Florida and Texas. Large meadows of seagrass are located near shore along the west coast of Florida; 2,123 $mi^2$ (5,500 ha) of beds are located within the boundaries of the Everglades National Park in Florida Bay. Seagrass meadows support diverse flora and fauna and are important nursery areas which provide both cover and food for many species of fish which are harvested commercially and recreationally. Unfortunately, human activities have resulted in extensive historic direct losses of seagrasses. Also, suspended particulate materials from dredging and other activities can block sunlight from seagrasses, and interfere with their growth and reproduction.

Mangrove forests occur mainly along Florida's coasts. Estimates of the total area of mangroves in Florida range from 430,000 to 650,000 ac (174,000 to 263,000 ha). Mangrove forests provide important habitats for young fish and other species and their elimination can result in a loss in recruitment of juveniles. Several human activities, including ditching or impounding for mosquito control, reduction of fresh-water input, and clearing and filling, have degraded the quantity and quality of mangrove habitats.

Wetlands and other coastal communities are important habitats for many threatened or endangered plants and animals. Marine turtles such as Kemp's Ridley, Hawksbill, Leatherback, Green, and Loggerhead, as well as other species of endangered or threatened vertebrates, are found along the Gulf coast. Gulf coastal habitats also provide important sites for bird rookeries.

**Functions and Values of Wetlands**

Society has been slow in recognizing the functional values of wetlands. Historically, wetland habitats have been treated as wastelands. Alternate "productive" uses of wetlands have disrupted the natural functions of many wetland ecosystems. Wetlands provide many important ecological and societal benefits. They are important habitats for a variety of plants and animals. Many species, including threatened and endangered ones, are dependent upon wetlands for food, protection from predators, resting areas, reproductive material or sites, molting grounds, and other requirements of life. Wetlands act as natural filters which remove organic pollutants, excess nutrients, and suspended particulates from water as it flows from uplands to the sea. Wetlands restrict the flow of flood waters and also serve as natural barriers that temper the effects of water surges from episodic events, such as hurricanes and other storms. They also may lessen wind damage to inland areas. Wetlands contribute to detrital food webs and are an important link in nutrient processing and cycling. Finally, wetland communities are aesthetically pleasing and are natural classrooms.

Estuaries and fringing wetlands provide critical habitats for many species of sport and commercial fishes. Estimates vary, but as many as 95% of commercial fish landed and 85% of the sport catch in the Gulf of Mexico spend at least a portion of their life in estuarine and wetland habitats. The production of waterfowl and other wildlife, such as fur-bearing mammals, is also determined by the abundance and quality of wetlands.

Often the values of natural systems are viewed in terms of the dollar value of the tangible and intangible benefits those systems provide. It is difficult to accurately quantify all of the benefits that wetlands provide to society. However, economic models, based upon the link between wetlands and commercial and sport fisheries, have quantified the value of specific wetland types. Values derived from these studies vary from approximately $1,000 to $82,000 per acre. The wide range of values reflects the disparity and accuracy of economic theories and estimates, as well as the variance of wetland types.

**Quality of Wetlands**

The quality of Gulf wetland habitats has been severely reduced through natural processes, such as subsidence, erosion, rising sea level, and by human activities, including construction of canals and channels, dredging, spoil disposal, draining, and filling. Subsidence is the fate of delta marshes where natural sediment deposition is precluded through construction of dams and other diversions. These lands can become compacted and sink beneath the water. Currents and surges from hurricanes and storms can accelerate erosion of sediment and other substrate, resulting in deterioration of wetland habitats. Sea level rise can hasten wetland subsidence and result in more open water acreage.

Construction and maintenance of canals has adversely impacted wetlands, particularly in coastal Louisiana. Channels convert emergent wetlands into open water and cause erosion of the remaining marsh. Saltwater intrusion brought about by altered circulation due to channelization has caused deterioration of large areas of coastal wetlands.

Historically, it was common practice in coastal areas to excavate and/or fill wetlands to create upland for real estate development. Material excavated from the bottom of adjacent waters or imported fill material was placed on submerged lands along the shoreline or on top of emergent wetlands to create uplands. Often, the fill material was contained or stabilized by bulkheads. In addition to direct loss of habitat, this process often resulted in high concentrations of suspended particulates and detrimental impacts to submerged plant communities in adjacent areas due to reduced light conditions. Also, changes in texture, particle size, depth and other physical/chemical changes resulting from dredging hampered recolonization by benthic flora and fauna.

One indication of the quality of estuarine habitats is the quality of indigenous shellfish. Shellfish waters are classified for the commercial harvest of oysters, clams, and mussels based upon public health concerns. Shellfish obtain nourishment by filtering large volumes of water and can bioconcentrate pollutants which are found in low concentrations in the water. Humans can ingest these pollutants, which include chemicals, bacteria, and other pathogens, by eating poorly prepared or raw shellfish meats. Therefore, the harvest of shellfish is strictly regulated and not permitted in waters which contain certain levels of bacteria and other pollutants.

Shellfish waters are classified by states and by the National Shellfish Sanitation Program (NSSP) guidelines. This classification reflects concentrations of pollutants, pollution sources, and concentration of fecal coliform bacteria. Waters are classified as approved, conditionally approved, restricted, or prohibited. A survey by NOAA in 1987 indicated that approximately 42% of waters suitable for raising shellfish in the Gulf of Mexico were approved for harvesting. The remaining acreage was contaminated sufficiently to result periodic closure or prohibition to shellfish harvesting.

**Case Histories**

Several detailed case histories are presented. These studies describe the modifications to habitats which have occurred in specific areas along the Gulf coast.

More than 70% of the total wetland loss from seven Texas river deltas (Colorado, Neches, Guadalupe, Lavaca, Trinity, Neches, and San Jacinto) from 1930 to 1980 occurred along the San Jacinto and Neches Rivers (White and Calnan, 1990). Major factors responsible for these losses include a rise in water level due to human-induced subsidence, natural compactional subsidence, global sea-level rise, and reduction of movement of sediment to marshes due to reservoir development in river drainage basins, channelization, and disposal of spoil on natural levees. The general trend in deltaic wetlands along the Texas coast is conversion of wetlands to open water and barren flats.

The impacts of human activities on habitats in Tampa Bay, Sarasota Bay, and Charlotte Harbor, Florida are well documented. Channel deepening, maintenance dredging, and dredging and filling to create uplands, have resulted in the loss of 44% of the original (100-year study) wetlands bordering Tampa Bay. In Sarasota Bay, changes in wetland habitat acres from 1948 to 1987 include losses of 35% of its seagrass beds, 45% of mangrove swamps, 85% of tidal marshes, and an increase of 16% of oyster beds. Wetland changes in Charlotte Harbor from 1945 to 1982 include losses of 29% of its seagrasses, 51% of salt marshes, 39% of oyster reefs, and an increase of 10% in mangroves. Several studies conclude that long-term effects of the release of fine sediment into the water column from dredging and filling may be even more damaging to plant communities than the initial habitat destruction.

In the Mississippi River Deltaic Plain, the major habitat issues include loss of salt marsh, salt water intrusion, and maintenance of habitat and water quality. In Louisiana, a significant change in acreage in coastal wetlands occurred between 1956 and 1978, when approximately 51% of the state's emergent marsh and 59% of forested wetlands were lost. During that time period, there was a concurrent 272% increase in acreage used for disposal of dredged material. Other studies have shown that approximately 34% of Louisiana marsh was changed to non-marsh from 1945 to 1980, and that currently there is a net wetland loss of approximately 39 $mi^2$ (101 $km^2$) annually in coastal Louisiana. Barrier island retreat in Louisiana is occurring at rates as high as 160 ac (65 ha) per year.

A study in Mobile Bay, Alabama revealed that emergent marsh habitat declined by more than 10,000 ac (4,000 ha), or 35%, between 1955 and 1979. Half of this loss resulted from industrial, commercial, and residential development; other losses were the result of erosion and/or subsidence. Another survey described a loss of 50% or more of submerged aquatic vegetation. The most significant impacts noted in these studies were the direct and indirect effect of dredging.

A decline of the area covered by seagrass and a decline of seagrass species has occurred in Mississippi Sound. Seagrass acreage in 1975 was approximately 60% of that found in 1969, and losses are continuing. Hurricane damage and seagrass destruction by fresh water discharged through a spillway account for approximately half of the loss. The causes of the remainder of the losses are undetermined. The seagrass beds remaining in Mississippi Sound are composed largely of *Halodule wrightii* and in this species leaf abundance is reduced compared to historic levels.

## Conclusions

- The Gulf coastal zone is characterized by a great diversity of habitats and natural resources, as well as industrial and recreational potential. The coastal area is richly endowed with productive wetland habitats whose ecological functions enhance viable wildlife and fishery resources of the Gulf. Wetland systems are in natural biogeochemical balance with contiguous terrestrial, freshwater, and marine systems from which the wetlands both receive and contribute biomass, nutrients, and other essential ingredients. The natural balance can be adversely impacted by society and nature and these impacts can change, irreversibly, the quantity and/or the character of the habitat. For example, when accretion of sediments by marshland is exceeded by subsidence, the result is a loss in marshland and an increase in open water.

- Although it is difficult and sometimes impossible to precisely compare acreages of wetlands at different time intervals, available comparisons indicate a major and continuing loss of emergent marsh, mangrove, and seagrass habitats along the Gulf coast. This loss often is due to a coupling of natural forces and human activities.

- Although economic evaluation of wetlands is still developmental, a recent study has placed the value of marsh in south Florida, based on sport and commercial fisheries landings alone, to be equivalent to some estimates of the commercial value of the land for development.

- In many instances, wetland habitats along the Gulf coast continue to be besieged by human activities such as dredging and filling for construction of canals, marinas, port facilities, real estate development, and conversion to other uses. Natural phenomena such as subsidence, sea level rise, and erosion are also adversely impacting wetlands. The primary causes of diminishing quality and losses of coastal emergent wetlands vary from state to state and include subsidence due to extraction of oil, gas, and fresh water in Texas; alterations of hydrodynamic flow due to construction of navigation channels, as well as subsidence, in Louisiana; and dredge and fill operations in Mississippi, Alabama, and Florida.

- Closure of vast areas of shellfish harvest waters is common to all Gulf states and is indicative of deterioration of water quality.

- Submerged vegetated communities in all Gulf coastal waters are susceptible to adverse effects of direct filling through deposition of dredged material and indirectly from the production of suspended material in the water column that blocks the sunlight necessary for growth and reproduction. Excess nutrients from sewage treatment plant discharges, septic tank systems, and drainage from agricultural fields contribute to seagrass declines by stimulating growth of periphyton on the plants and phytoplankton in the waters over the grass beds.

- The loss of quantity and quality of wetlands has slowed since the late 1970's due to passage and enforcement of federal and state laws and regulations. Also, the ecological and societal functions of wetlands are becoming better known to resource managers and to a more interested and active public. Better informed managers and public awareness have made a positive difference in management decisions concerning the disposition of wetland habitats.

- Interest in habitat management and research continues to grow. Many wetlands research and demonstration projects are underway or about to begin. For example, the U.S. FWS is digitizing information on wetland acreages in Gulf coastal states (some of the preliminary data are presented in this report), NOAA continues to collect data in the Status and Trends Program, EPA has initiated a monitoring program in near coastal waters and estuaries (Ecological Monitoring and Assessment Program), and Florida and Texas will soon publish up-to-date wetland acreage data. The U.S. Army Corps of Engineers is designing and testing several freshwater diversion projects to improve the condition of wetland habitats along the lower Mississippi River.

## Introduction

The purpose of this report is to summarize the current status and historical trends in quality and quantity of saline coastal wetland and seagrass habitats of the Gulf of Mexico. The paper is organized into the following sections: Inventories of Coastal Habitats, Functions and Values of Coastal Habitats, Quality of Coastal Habitats, Living Resources, Case Histories, and Conclusions. The boundaries of emergent coastal wetlands are based on the delineation of estuarine boundaries by Cowardin et al. (1979) and include areas: 1) upstream and landward where ocean-derived salts measure less than 0.5 ppt during periods of low flow, or where there is a significant tidal influence; 2) an imaginary line closing the mouth of a river, bay, or sound; and 3) to the seaward limit of wetland emergent vegetation where it is not included in 2). This system also includes offshore areas of continually diluted seawater.

Ecological habitats along the Gulf of Mexico coast are many and varied. Habitats discussed in this document are limited to the coastal plain and are generally delineated by the upper reaches of tidal action and include emergent herbaceous marshes, mangrove forests, and seagrass meadows. The intention of this document is not to summarize all available information on these habitats, but rather to summarize data given in surveys of historical and current acreages. Additional references are included in the Bibliography for those interested in pursuing other topics in more detail than presented in this report.

Data presented in published and unpublished surveys of habitat acreage are summarized and discussed. Many surveys of habitat quantity have been made for different purposes and the definitions of wetland types, baselines, and interpretation of boundaries may differ between surveys. Thus, it is not advisable to directly compare acreages of habitats measured at different times by different authors. We have summarized trends in habitat change in Case Histories for selected estuaries where measurements of habitat acreages were made using the same method for different time periods specifically for the purpose of calculating changes in habitat extent.

The function of coastal ecosystems is dependent upon the quantity and quality of coastal habitats. Thus, the amount of habitats lost may be a good indicator of estuarine "health." Other indicators of the health of coastal areas include the status of shellfish resources for human consumption and endangered species and their habitat. For this reason, summaries of available information on shellfish harvesting and endangered species are presented.

The Gulf of Mexico is bordered by 207 estuaries (Thayer and Ustach, 1981). The total open-water area of Gulf estuaries at mean high water is approximately $7.9 \times 10^6$ ac ($3.2 \times 10^6$ ha) with a volume of approximately $57.9 \times 10^6$ acre-ft (Lindall and Saloman, 1977. The estuarine acreage is not evenly distributed among Gulf states. Louisiana contains 43%, Florida 26%, Texas 19%, Mississippi 6%, and Alabama 5% of the estuarine acreage. The Gulf coastal region is dominated by warm subtropical waters and generally small tidal ranges. From Apalachee Bay, Florida, southward tides range from 1.97 to 3.94 ft (0.6 to 1.2 m), while to the west, tidal amplitude generally is < 2 ft (0.6 m) (Brooks, 1973; Christmas, 1973; Diener, 1975).

The shoreline of the Gulf coast of the U.S.A. is 15,500 mi (25,000 km) long and includes many diverse wetland habitats including tidal marshes, mangroves, and submerged seagrass beds. Coastal marshes and mangroves forests form the interface between marine and terrestrial habitats, while seagrass beds interface between emergent vegetation and unvegetated estuarine and coastal bottoms. The geography of these habitats vary throughout this broad zone. In some areas fringing wetlands occur as narrow bands, while in other areas marshes are very broad and cover large areas.

Coastal Gulf wetland habitats can consist of either monotypic stands or diverse plant communities. Wetlands are transitional zones between terrestrial and aquatic environments and there is a wide variety of wetland habitats. Coastal, estuarine wetlands occur in areas of reduced salinity and are transitional zones between fresh and salt water habitats. "Wetlands" is used as a collective term for habitats such as bogs, marshes, and swamps. A more scientific definition of wetlands has been offered by Cowardin et al. (1979): "Lands transitional between terrestrial systems where the water table is at or near the surface or the land is covered by shallow water. Wetlands must have one or more of the following attributes: 1) at least periodically, the land supports predominately hydrophytes; 2) the substrate is predominately undrained hydric soil; 3) the substrate is nonsoil and is saturated with water or covered with shallow water during the growing season." This definition includes submerged aquatic vegetation such as the productive seagrass beds found in coastal waters of the Gulf.

The U.S. Army Corps of Engineers (COE) and the U.S. Environmental Protection Agency (EPA) are responsible for making jurisdictional determinations of wetlands for regulatory purposes and have developed the following definition of wetlands: "Those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas." A standardized methodology for delineating wetland limits has been developed and includes this definition of wetlands (Federal Interagency Committee for Wetland Delineation, 1989). This method requires that hydric soils, hydrology, and prevalence of wetland plants be present at normal sites. Proposed changes to this methodology are currently under review.

Wetlands are highly productive habitats and have great value to people. The acreage of wetland habitats has been decreasing at an alarming rate for the past 40 years. In the past, wetlands were often viewed as wastelands to be converted to farmland or residential and industrial sites. More recently, the value of wetlands to coastal ecosystems and to human well-being has become recognized. Some now see wetlands as "wonderlands" as opposed to "wastelands" (U.S. EPA, 1988a). The gradual development of this positive attitude toward wetlands is in part due to an increased understanding of their role in general ecological processes and a better understanding and quantification of the tangible value of wetlands to society.

Wetlands provide shelter for many species of plants and animals which inhabit these unique areas. For example, coastal emergent wetlands, including salt marshes and mangrove forests, furnish protective cover for the food organisms needed by many larger animals. Coastal wetlands contribute organic material to the detrital food web and are integral to the processing and flow of nutrients. Wetlands are natural filters and remove organic pollutants, excess nutrients, and suspended particulates from water as it flows from the terrestrial environment to the sea. They also serve as natural barriers that temper the effects of water surges from episodic events, such as hurricanes and other storms, and may lessen wind damage to inland areas.

Wetlands contribute to human activities in many ways. For example, they provide important nursery habitat for many sport and commercial fish species. Estimates vary, but as much as 95% of the weight of commercial fish landed and 85% of the weight of the sport catch in the Gulf of Mexico comes from fish that spend at least a portion of their life in coastal wetland and estuarine habitats (Thayer and Ustach, 1981; Lindall and Thayer, 1982).

During the past few decades the Gulf estuaries have begun to show signs of increased deterioration of environmental quality. Serious deterioration is becoming evident in many locations. Loss of quality and acreage is occurring for marshes, seagrass beds, and mangroves, as well as in hard substrates, such as shellfish beds and coral habitats. Impacts are directly linked to increased human activities within estuarine watersheds (NOAA, 1990c).

Wetlands can be adversely impacted by both natural and human activities. Natural causes that result in loss of wetlands include subsidence and impact from hurricanes and other storm events. Subsidence is the natural fate of delta areas which no longer receive sediments from rivers or flood waters. Human activities which interrupt the flow of water to wetlands and deprive them of the required sediment can exacerbate the rate of subsidence. Hurricanes and other strong storm surges can flood freshwater marshes with saltwater which kills salt intolerant plants. Rising sea levels can accelerate the loss of wetlands by converting them to open water areas. Human activities that have been especially harmful to wetland habitats include: draining for crop and timber production;

draining or impounding for mosquito control; dredging for construction and maintenance of navigation and access channels, flood protection, or development; and discharging of sewage, toxic effluents, and other wastes. Increased state and federal regulatory control has begun to slow the rate of loss of several wetland types. For example, the passage of the "swampbuster" provisions of the 1985 Food Securities Act has slowed the rate of conversion of productive swamp wetlands to agriculture lands.

The Gulf of Mexico is one of the fastest growing regions in the nation. The population of the Gulf coast increased 33% between 1970 and 1980, and the population is projected to increase by 144% between 1960 and 2010 (Culliton et al., 1990). Concurrent with population growth has been a decline in environmental quality. Increased population has resulted in increased pressure for wetland development and in increased discharges of domestic and industrial wastes into coastal waters.

A good indicator of the ecological health of an estuarine system is the ability to harvest shellfish for human consumption. Approximately 30% of harvestable shellfish waters in the Gulf were closed to harvesting in 1985 (NOAA, 1988c). The closure of shellfish waters for harvesting due to excess coliform bacteria concentrations is one indication that society's pollution has degraded estuarine and wetland habitats. Recently, the occurrence of persistent marine debris such as plastics on coastal beaches has been a concern because this unsightly material accumulates and may have adverse effects on animals through ingestion or entanglement. Also, the aesthetic delight of a natural experience is degraded by the prevalence of human refuse which may be indicative of our disregard for natural habitats.

It is often difficult to precisely compare acreages of wetlands at one location measured at different time intervals because of possible differences in wetland classification, tidal and moisture conditions, and wetland mapping criteria. However, it has been documented that wetlands are continuing to be lost nationally as well as regionally in the Gulf of Mexico. Approximately 97 million ac ($39 \times 10^6$ ha) of wetlands remain of the original 215 million ac ($87 \times 10^6$ ha) in the contiguous U.S. (Tiner, 1984). According to Frayer et al. (1983), over 9 million ac ($3.6 \times 10^6$ ha) of the nation's wetlands were lost from the 1950's to the 1970's. Most of this decrease was in inland areas, but coastal habitats also experienced marked declines. Realization of these losses prompted Congress to include Section 404 in the 1972 reauthorization of the Federal Water Pollution Control Act. Section 404 requires a federal permit for dredging and filling of waters of the United States including wetlands. The history of the Section 404 program has been discussed by Lieberman (1984), Want (1984), Nagel (1985), Anadromous Fish Law Memo (1988), Brady (1990), and others. All agree that regardless of the federal regulatory program, wetlands continue to be lost by natural and human causes. This fact prompted the convening of the National Wetlands Policy Forum by the

Conservation Foundation at the request of EPA in 1987 which adopted as its interim goal to "achieve no overall net loss of the nation's remaining wetland base" (National Wetlands Policy Forum, 1988).

Recent wetland losses in some areas remain great. For example, Day and Craig (1981) estimated that 32,000 ac (50mi$^2$) (13,000 ha) of wetlands were being lost annually in coastal Louisiana. Coastal Louisiana wetland losses were recently estimated to be 31 mi$^2$ per year (Dunbar et al., 1990). Similar losses during this period have been documented in all Gulf coast states, particularly in Galveston Bay, Texas; delta habitats in Louisiana; the Mississippi Sound; Mobile Bay, Alabama; and Tampa and Sarasota Bays in Florida. Detailed analyses of historical and continuing wetland losses of these areas are summarized in Case Histories of this report.

The value of wetlands depends upon the beholder's perspective. Some view wetlands as potential sites for residential development, some as havens for fish and wildlife, and others as cultivatable farmland or prospecting grounds for oil and gas production. Because of the many competing uses of wetlands, it has become commonplace to attempt to quantify the market value of wetland acreage. This is often difficult because there are many kinds of habitats which produce varying amounts of tangible and intangible products. Several attempts have been made to place a monetary value on the tangible services provided by wetland habitats (e.g., fisheries productivity). However, these habitats are an integral part of the coastal ecosystem and provide many intangible goods and services to society, such as aesthetic values, teaching tools, and nutrient cycling. The monetary values of these intangible goods and services are very difficult to quantify.

Some progress has been made in estimating a market value of the capacity of wetlands to provide vital habitat for fishery organisms. Several biologists and economists have suggested a relationship between yields of commercial and sport fisheries and acreage of wetland habitats which are available to fish and shellfish (Turner, 1977; Deegan et al., 1986). Based on these relationships, economic models have been developed to establish the value of specific wetlands; estimated values have ranged from less than $1,000 to $82,000 per acre (Bell, 1989; Foster, 1978; Gosselink et al., 1974).

The nutrient-rich and highly productive emergent vegetated wetlands, such as marshes and mangroves, submerged aquatic habitats, such as seagrasses, coral reefs, and other live bottoms, and mud and sand flats provide important habitat for the living resources, including numerous endangered species of the Gulf of Mexico. Probably one of the most important values of wetlands is their use by many estuarine and marine animals for spawning, rearing, feeding, migration, and protection from predators. Thus, populations of these organisms can be limited by the quality and quantity of these habitats.

The coastal wetlands of the Gulf of Mexico are of special interest because the Gulf is an exceptionally productive sea that annually yields nearly 2 billion pounds (900,000 mt) of fish and shellfish and contains five of the top seven commercial fishery ports in the nation by weight (NOAA, 1990). The Gulf is one of the few areas of the U.S. Exclusive Economic Zone that possesses large quantities of unexploited fishery resources. The Gulf is well known for its coastal estuaries, wetlands, and barrier islands that provide important habitat for large populations of wildlife, including waterfowl, shore birds, and colonial nesting birds. For example, the Gulf provides critical habitat for 75% of the migratory waterfowl traversing the United States. The extensive emergent herbaceous coastal wetlands which remain in the Gulf make up about half of the Nation's remaining total wetland area (U.S. EPA, 1988c).

## Inventories of Coastal Habitats

The purpose of this section is to provide a summary of major inventories of coastal habitats. The reader is cautioned not to attempt to compare data between inventories since baselines and definitions are not consistent among them. All historical surveys document that the acreage of wetlands in the Nation, particularly the Gulf of Mexico, decreased dramatically between the 1950's and 1970's. According to Frayer et al. (1983), the total acreage of wetlands and deep-water habitat nationwide was 179.5 million ac (72 x $10^6$ ha) in the 1950's and 171.9 million ac (69.6 x $10^6$ ha) in the 1970's, a loss of 7.6 million ac (3.1 x $10^6$ ha) (Table 1). There were 108.1 million ac (43.8 x $10^6$ ha) of wetlands in the 1950's and 99.0 million ac (40.1 x $10^6$ ha) in the 1970's, a loss of over 9 million ac (3.6 x $10^6$ ha). The majority of this loss was in palustrine freshwater and estuarine wetlands. In general, these data are useful only at the national level because of difficulties in comparing acreage figures estimated at different times. Also, Frayer et al. did not estimate acreage of seagrass communities.

**Table 1. Wetlands Acreage (Acres) Changes from 1950's to 1970's (From Frayer et al., 1983)[1]**

| | Marine Intertidal | Estuarine Subtidal | Estuarine Intertidal Non-veg. | Estuarine Intertidal Emergent | Estuarine Intertidal Forested Scrub/shrub |
|---|---|---|---|---|---|
| Estimated Acreage in 1970's | 78,400 (14.0)[2] | 14,967,700 (1.5) | 746,500 (9.8) | 3,922,800 (4.3) | 573,000 (14.4) |
| Estimated Acreage lost or gained in 1950's to 1970's | -4,000 (57.5) | +200,200 (14.9) | +5,400 (*)[3] | -353,200 (8.3) | -19,100 (93.2) |

[1] Original acreage presented as acres x 1000
[2] Standard error (percentage of the entry)
[3] Standard error of estimate is equal to or larger than estimate

Tiner (1984) also reviewed the losses of wetlands from the 1950's to the 1970's and found that the greatest wetland losses nationally during that period occurred along the Gulf coast, particularly in Louisiana, Florida, and Texas. Most of Louisiana's losses were attributed to submergence by coastal waters, while dredge and fill and residential development in coastal areas were most significant in Florida and Texas.

Tiner divided the major causes of wetland loss and degradation into three categories: direct human effects, indirect human effects, and natural effects. Direct human effects included drainage for crop production, dredging and stream channelization, filling, construction of dikes, dams, levees, and seawalls, discharges of pollutants and other materials, and mining of wetland soils. Indirect human causes included sediment diversion by dams, deep channels, and other means, hydrologic alterations by canals, levees, and spoilbanks, and subsidence due to extraction of materials (ground water, oil, gas, etc.). Natural threats included subsidence, droughts, hurricanes and other episodic events, and grazing by nutria and muskrats.

A nationwide compilation of existing coastal wetlands data was conducted by the National Oceanic and Atmospheric Administration (NOAA) (Alexander et al., 1986) and was based on U.S. Fish and Wildlife Service (FWS) data and other existing sources of information. NOAA did not use the U.S. FWS classification system because some data were obtained from states, and many states had their own classification systems.

These data, gathered at different times between 1954 and 1980, indicate the presence of over 11 million ac (44,534 km$^2$) of wetlands along the coastal United States, of which 5,184,000 ac (20,988 km$^2$) are located in the Gulf of Mexico. Emergent wetland acreage, consolidated under three categories, is given in Table 2.

**Table 2. Wetland Acreage (Acres) in the Gulf of Mexico (from Alexander et al., 1986)[1]**

| State | Salt Marsh | Fresh Marsh | Forested | Total |
|---|---|---|---|---|
| Florida | 431,300 | 77,500 | 970,700 | 1,479,500 |
| Alabama | 14,600 | 10,600 | 151,300 | 176,500 |
| Mississippi | 64,000 | 4,000 | 76,000 | 144,000 |
| Louisiana | 1,748,600 | 688,800 | 437,200 | 2,874,600 |
| Texas | 390,400 | 896,500 | 40,300 | 509,400 |
| Subtotal | 2,648,900 | 1,676,500 | 1,675,500 | 5,184,000 |

[1] Aerial photography dates:

FL = 1972-1976
AL = 1976-1978
MS = 1979-1980
LA = 1976-1978
TX = 1950-1951

Alexander et al. (1986) also reported these data by county. The manner in which each state delineated wetlands and its classification methodology are found in an appendix of that report. A fundamental obstacle to consolidating these data is the lack of a unified system of classification and quantification. Another problem is that wetlands conform to drainage basins, not political boundaries. Also, acreages of seagrasses and tidal flats were not included in this survey.

The most recent Gulf-wide wetland inventory was prepared by NOAA (1991) which lists acreage and habitat types of coastal wetlands for the five Gulf coast states. The most recent NOAA report also summarizes coastal wetland acreage by estuarine drainage area (EDA) and by county. NOAA's research included a review of existing data on the areal extent and distribution of coastal wetlands, but much of this information was found to be incomplete or outdated. NOAA then evaluated an alternative source of information, the National Wetland Inventory (NWI) mapping program of FWS. The NWI program has summarized scientific information on the extent of the Nation's wetlands through creation of detailed maps and research on historical status and trends. The maps are developed from aerial photography and are based on 1:24,000 scale U.S. Geological Survey (USGS) maps. The NWI maps were quantified by NOAA using a grid sampling technique which provided a time and cost effective alternative to standard digitizing techniques. Methods and statistical reliability of this technique are given in the NOAA report.

In all, 56.2 million ac (227,530 km$^2$) were sampled by NOAA. Of this, 24% or 13.7 million ac (55,466 km$^2$) were identified as wetlands. Four major wetland habitat types are listed. The largest category is forested wetlands (8,100,000 ac; 32,794 km$^2$; 59% of total), followed by fresh marsh (2,600,000 ac; 10,526 km$^2$; 19%), salt marsh (2,500,000 ac; 10,121 km$^2$; 18%), and tidal flats (500,000 ac; 2,024 km$^2$; 4%). Open water and uplands are included as nonwetland habitat types in this survey.

The major habitat types are further subdivided into wetland subgroupings and nonwetlands and include: salt marsh - brackish, high, low, unspecified; fresh marsh - nontidal, tidal, unspecified; forested and shrub-scrub - estuarine, nontidal fresh, tidal fresh, unspecified fresh; tidal flats; open water - fresh and nonfresh; and uplands. Definitions of these habitat types and common plant communities are given in Table 3.

The dates of aerial photography for the maps used in the NOAA study ranged from 1972 to 1984, with 28% taken in 1979 and 42% taken after 1980. Because of the date of the photographs and the fact that national trends indicate that the amounts of most wetland types are still declining (Frayer et al., 1983), wetlands statistics presented by NOAA may over estimate the actual current amount of coastal wetlands.

**Table 3. Coastal Wetlands Classification for the Gulf of Mexico (from NOAA, 1991)**

| NOAA | FWS | Common Plant Community |
|---|---|---|
| **Salt Marsh** | | |
| Brackish | Estuarine intertidal emergent regularly and irregularly flooded salinity $\geq 0.5^{o}/_{oo}$ and $\leq 30^{o}/_{oo}$ | black needlerush (Juncus roemerianus), salt hay grass (Spartina patens), salt grass (Distichlis spicata) |
| High | Estuarine intertidal emergent irregularly flooded salinity $\geq 30^{o}/_{oo}$ | black needlerush (J. roemerianus), salt hay grass (S. patens), salt grass (D. spicata) |
| Low | Estuarine intertidal emergent regularly flooded or irregularly exposed salinity $>30^{o}/_{oo}$ | smooth cordgrass (S. alterniflora) |
| Unspecified | Estuarine intertidal emergent | see "brackish," "high," and "low" |
| **Fresh Marsh** | | |
| Tidal | Lacustrine littoral emergent tidal<br>Palustrine emergent tidal<br>Riverine tidal or lower perennial emergent tidal | spike-rush (Eleocharis spp.)<br>three-square rush (Scirpus americanus) |
| Unspecified | Lacustrine littoral emergent<br><br>Palustrine emergent<br>Riverine tidal or lower perennial | see "tidal" |
| **Forested and scrub-shrub** | | |
| Estuarine | Estuarine intertidal forested or scrub-shrub | black mangrove (Avicennia germinans)<br>marsh elder (Iva frutescens)<br>red mangrove (Rhizophora mangle) |
| Tidal fresh | Palustrine forested or scrub-shrub tidal | same as "nontidal" |
| **Tidal flats** | Estuarine intertidal/ Marine intertidal (includes aquatic beds, beach/bars, flats, reefs, rocky shores, streambeds, and unconsolidated shores) | saltwort (Batis maritima)<br>smooth cordgrass (S. alterniflora) |

The matrix presented in Table 4 is based on the most recent NOAA inventory (1991) and shows acreage of selected coastal habitats in Gulf states, by county. The term "unspecified" was used in some areas when information available from maps and aerial photographs was insufficient to determine whether the area was tidal or non-tidal. Therefore, NOAA's fresh marsh (unspecified) category was not included in total coastal acreage in this summary of status and trends.

Summaries of acreages presented in Table 4 are presented in Table 5 and Figures 1 and 2 by state and habitat type. Even though total wetland acreage has diminished greatly during the last 30 years, the Gulf of Mexico still contains large areas of these valuable habitats (Table 5). Louisiana and Florida contain approximately 80% of the total 3,762,800 ac (15,234 km$^2$) in the five wetland categories. Louisiana has the greatest acreage of coastal emergent wetlands with 49% of the total, followed by Florida (29%), Texas (20%), and Mississippi (2%). Texas contains 54% of the regional tidal flats, and Florida has 97% of the estuarine forested scrub-shrub, most of which is mangrove (Figure 1). Of the five major coastal wetland habitats, 66% of the acreage is salt marsh, 17% forested scrub-shrub, 13% tidal flats, 3% tidal fresh marsh, and 1% forested (Figure 2).

Acreage of wetland habitats along the Gulf coast in selected estuarine drainage areas (EDA) is given in Table 6. Wetland inventory maps are not yet complete for all EDA's, and the NOAA document describes the location, dominant wetland type, dimensions, and other characteristics for twenty-three of the thirty-one major EDA's. This analysis by natural ecological units, rather than by state or county delineation, is an especially useful baseline in evaluating the potential impacts of natural and man-induced degradation of specific ecosystems.

Another inventory of wetlands and other habitats is presented by Watzin and Baumann (1988), who list acreages of the Coastal Barrier Resource System (CBRS) within the Gulf states (Table 7). The coastal barrier resource system (CBRS) is a network of coastal barriers protected under the Coastal Barrier Resource Act (CBRA). There are 186 units (islands) in the CBRS, ranging in size from less than 20 ac (8.1 ha) to more than 49,000 ac (19,838 ha). Barrier islands, barrier spits and peninsulas, bay barriers, and tombolos (sand or gravel bar that connects an island with the mainland or another island) are all included in the system. Watzin and Baumann (1988) summarized the results of a 1982 inventory of the CBRS and a study of the shoreline change and wetland loss which occurred in selected units of the system. A discussion of their findings on losses of wetlands from the coastal barrier system is presented later in this document.

Other habitat inventories are available on a statewide basis. For example, the Environmental Geologic Atlas project of Texas was initiated in 1969 to produce a thorough regional analysis of natural resources and processes, lands, water bodies, and other factors for coastal counties. The

## Table 4. Emergent Wetlands (Acres) in Coastal Counties (from NOAA, 1991)[1]

| State/County | Salt Marsh | Fresh Marsh (Tidal) | Fresh Marsh[2] (Unspecified) | Forested Scrub-Shrub (Estuarine) | Forested Scrub-Shrub (Tidal Fresh) | Tidal Flats | Total |
|---|---|---|---|---|---|---|---|
| **TEXAS** | | | | | | | |
| Aransas | 25,400 | 300 | | 600 | 100 | 10,900 | 37,300 |
| Brazoria | 53,600 | 2,800 | | <500[3] | 1,900 | 4,100 | 62,400 |
| Calhoun | 33,800 | 4,000 | | 100 | 400 | 5,000 | 43,300 |
| Cameron | 31,200 | | | 600 | | 53,100 | 84,900 |
| Chambers | 45,900 | 7,100 | | | 3,300 | 2,800 | 59,100 |
| Galveston | 35,100 | | | 200 | <100 | 8,000 | 43,300 |
| Harris | 1,200 | 200 | | | <100 | 800 | 2,200 |
| Jackson | 10,500 | 100 | | | 100 | 400 | 11,100 |
| Jefferson | 63,000 | 3,500 | | | 100 | 4,400 | 71,000 |
| Jim Wells | | | <100 | | | | |
| Kenedy | 24,400 | | | 100 | | 112,000 | 136,500 |
| Kleberg | 4,700 | 100 | | | | 14,800 | 19,600 |
| Liberty | | <100 | | | | | |
| Matagorda | 56,300 | 400 | | <100 | 100 | 8,700 | 65,500 |
| Nueces | 5,200 | <100 | | 1,000 | <100 | 8,100 | 14,300 |
| Organge | 3,100 | 900 | | <100 | <100 | <100 | 4,000 |
| Refugio | 11,200 | 2,200 | | | 1,200 | 2,100 | 16,700 |
| San Patricio | 12,000 | 900 | | | 200 | 4,000 | 17,100 |
| Victoria | 1,500 | | | | | 100 | 1,600 |
| Willacy | 13,900 | | | <100 | | 35,800 | 49,700 |
| STATE TOTAL | 432,000 | 22,500 | <100 | 2,600 | 7,400 | 275,100 | 739,600 |
| **LOUISIANA** | | | | | | | |
| Calcasieu | 7,800 | | 10,900 | | | 600 | 8,400 |
| Cameron | 355,600 | | 188,500 | | | 15,600 | 371,200 |
| Iberia | 89,500 | | 4,800 | | | <100 | 89,500 |
| Jefferson | 60,100 | | 16,300 | 500 | | 800 | 61,400 |
| Lafourche | 223,800 | | 26,300 | 2,300 | | 3,400 | 229,500 |
| Orleans | 40,900 | | 1,100 | | | <100 | 40,900 |
| Plaquemines | 258,700 | 65,000 | 4,000 | 500 | 4,800 | 1,900 | 330,900 |
| St. Bernard | 208,700 | | 100 | 300 | | 1,800 | 210,800 |
| St. Charles | 6,700 | | 300 | | | | 6,700 |
| St. James | | | 500 | | | | |

## Table 4. Emergent Wetlands (Acres) in Coastal Counties (from NOAA, 1991)[1] (Continued)

| State/County | Salt Marsh | (Tidal) | Fresh Marsh (Unspecified) | Fresh Marsh Shrub (Estuarine) | Forested Scrub-Shrub (Tidal Fresh) | Forested Scrub-Tidal Flats | Total |
|---|---|---|---|---|---|---|---|
| **LOUISIANA (Cont.)** | | | | | | | |
| St. Jn. Baptist | 6,800 | | 3,700 | | | | 6,800 |
| St. Mary | 18,400 | | 82,900 | | | 100 | 18,500 |
| St. Tammany | 31,300 | | 14,100 | 100 | | 100 | 31,500 |
| Terrebonne | 312,400 | | 147,400 | 6,500 | | 6,900 | 325,800 |
| Vermillion | 102,100 | | 21,900 | | | 600 | 102,700 |
| STATE TOTAL | 1,722,800 | 65,000 | 535,400 | 10,200 | 4,800 | 31,800 | 1,834,600 |
| **MISSISSIPPI** | | | | | | | |
| Hancock | 22,900 | | 900 | 100 | | | 23,000 |
| Harrison | 7,700 | | | 300 | | 1,200 | 9,200 |
| Jackson | 28,200 | | | 500 | | 1,100 | 29,800 |
| Pearl River | <100 | | | | | | |
| STATE TOTAL | 58,800 | | 900 | 900 | | 2,300 | 62,000 |
| **ALABAMA** | | | | | | | |
| Baldwin | 13,800 | <100 | <100 | 2,400 | 1,900 | 2,100 | 20,200 |
| Mobile | 11,700 | 100 | 300 | 400 | 100 | 2,000 | 14,300 |
| STATE TOTAL | 25,500 | 100 | 300 | 2,800 | 2,000 | 4,100 | 34,500 |
| **FLORIDA** | | | | | | | |
| Bay | 8,200 | | | <100 | 600 | 4,800 | 13,600 |
| Calhoun | | | <100 | | | | |
| Charlotte | 5,100 | 100 | | 22,600 | | 18,400 | 46,200 |
| Citrus | 27,500 | 3,500 | | 6,600 | 2,100 | 500 | 40,200 |
| Collier | 15,000 | | | 95,900 | | 9,100 | 120,000 |
| Columbia | | | | | | | |
| Dade | 8,400 | | <100 | 72,600 | | 1,100 | 82,100 |
| De Soto | | 100 | | | 300 | | 400 |
| Dixie | 20,300 | 300 | | 800 | 100 | 2,700 | 24,200 |
| Escambia | 1,300 | | | 600 | 200 | 2,900 | 5,000 |
| Franklin | 18,400 | 1,500 | | 100 | 7,700 | 6,000 | 33,700 |
| Gadsden | <100 | | | | | <100 | |
| Gulf | 2,800 | 3,200 | | <100 | 4,800 | 2,000 | 12,800 |
| Hamilton | | | | | | | |

**Table 4. Emergent Wetlands (Acres) in Coastal Counties (from NOAA, 1991)[1] (Continued)**

| State/County | Salt Marsh | (Tidal) | Fresh Marsh (Unspecified) | Fresh Marsh Shrub (Estuarine) | Forested Scrub-Shrub (Tidal Fresh) | Forested Scrub-Tidal Flats | Total |
|---|---|---|---|---|---|---|---|
| **FLORIDA (Cont.)** | | | | | | | |
| Hardee | | | <100 | | | | |
| Hernando | 9,200 | 500 | | 400 | 300 | 1,700 | 12,100 |
| Hillsborough | 12,000 | <100 | | 7,400 | 300 | 6,800 | 15,700 |
| Jefferson | 3,800 | | | 100 | 100 | 400 | 4,400 |
| Lee | 3,200 | 200 | | 41,700 | | 49,900 | 95,000 |
| Levy | 35,400 | 300 | | 2,600 | | 1,300 | 39,600 |
| Manatee | 1,700 | | | 6,000 | 200 | 11,200 | 19,100 |
| Monroe | 39,400 | | | 345,000 | | 41,300 | 425,700 |
| Okaloosa | 300 | | | 100 | | 2,700 | 3,100 |
| Pasco | 4,600 | 100 | | 700 | 200 | 1,000 | 6,600 |
| Pinellas | 900 | | | 5,500 | 100 | 11,900 | 18,400 |
| Santa Rosa | 6,500 | | | 300 | 400 | 2,400 | 9,600 |
| Sarasota | 900 | | | 1,400 | | 4,500 | 6,800 |
| Taylor | 21,800 | | | 2,800 | | 1,600 | 26,200 |
| Wakulla | 18,600 | | <100 | 200 | 600 | 5,300 | 24,700 |
| Walton | 2,700 | | | 300 | 400 | 3,400 | 6,800 |
| STATE TOTAL | 257,200 | 9,800 | | 613,700 | | 192,900 | 1,092,000 |

1 Acreage originally reported as acres x 100
2 Total does not include fresh water unspecified
3 < values not included in totals

**Table 5. Totals (Acres) of Selected Wetlands by State for the Gulf of Mexico (from NOAA, 1991)[1,2]**

| State | Salt Marsh | Fresh Marsh (Tidal) | Forested Scrub-Shrub (Estuarine) | Forested Scrub-Shrub (Tidal Fresh) | Tidal Flats | Total | Percentage of Total |
|---|---|---|---|---|---|---|---|
| Texas | 432,000 | 22,500 | 2,600 | 7,400 | 275,100 | 739,600 | 20 |
| Louisiana | 1,722,800 | 65,000 | 10,200 | 4,800 | 31,800 | 1,834,600 | 49 |
| Mississippi | 58,800 | -[3] | 900 | - | 2,300 | 62,000 | 2 |
| Alabama | 25,500 | 100 | 2,800 | 2,000 | 4,100 | 34,500 | 0 |
| Florida | 257,200 | 9,800 | 613,700 | 18,400 | 192,900 | 1,092,000 | 29 |
| TOTALS | 2,496,300 | 97,400 | 630,200 | 32,600 | 506,200 | 3,762,700 | |

[1] Acreage originally reported as acres x 100
[2] Calculations based on FWS wetland inventory maps
[3] None recorded

## Table 6. Coastal Wetlands by Estuarine Drainage Area (Acres) (NOAA, 1991)

| | Salt Marsh | | | Fresh Marsh | | | | Forested and Scrub-Shrub | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Brack | Unsp. | Subtotal | Non-Tidal | Tidal | Unsp. | Subtotal | Est. | Fresh (Unsp.) | Non-Tidal Fresh | Tidal Fresh | Subtotal | Tidal Flats | Total |
| Ten Thousand Islands (100) | 0 | 54,800 | 54,800 | 807,600 | 0 | 0 | 807,700 | 400,300 | 0 | 861,300 | 0 | 1,261,600 | 40,900 | 2,165,000 |
| Charlotte Harbor (100) | 0 | 6,800 | 6,800 | 289,200 | 400 | 0 | 289,700 | 52,600 | 0 | 218,400 | 300 | 271,300 | 56,300 | 624,000 |
| Tampa Bay (100) | 0 | 3,100 | 31,00 | 46,600 | 0 | 0 | 46,700 | 16,000 | 0 | 148,200 | 500 | 164,800 | 37,600 | 252,200 |
| Suwanee Bay (100) | 0 | 20,900 | 20,900 | 17,400 | 200 | 0 | 17,500 | 1,000 | 0 | 189,100 | 100 | 190,300 | 300 | 229,000 |
| Apalachee Bay (87) | 0 | 24,400 | 24,400 | 25,200 | 100 | 0 | 25,400 | 500 | 0 | 635,600 | 700 | 636,900 | 8,800 | 695,400 |
| Apalachicola Bay (95) | 0 | 17,000 | 17,000 | 4,100 | 4,600 | 0 | 8,700 | 0 | 0 | 547,800 | 10,700 | 558,500 | 7,500 | 591,600 |
| St. Andrew Bay (100) | 0 | 8,500 | 8,500 | 2,800 | 0 | 0 | 2,800 | 0 | 0 | 235,700 | 600 | 236,300 | 3,500 | 251,100 |
| Choctawhatchee Bay (87) | 0 | 2,700 | 2,700 | 3,700 | 0 | 0 | 3,700 | 300 | 0 | 267,200 | 500 | 267,900 | 5,800 | 280,100 |
| Pensacola Bay (54) | 0 | 6,700 | 6,700 | 6,100 | 0 | 0 | 6,100 | 300 | 0 | 229,100 | 400 | 229,700 | 2,000 | 244,500 |
| Perdido Bay (100) | 0 | 1,900 | 1,900 | 1,800 | 0 | 0 | 1,800 | 1,000 | 0 | 164,100 | 600 | 165,700 | 700 | 170,200 |
| Mobile Bay (89) | 0 | 17,000 | 17,000 | 7,000 | 100 | 1 | 7,200 | 1,900 | 800 | 623,300 | 1,300 | 627,200 | 3,000 | 654,500 |
| Mississippi Sound (63) | 97,600 | 73,000 | 170,600 | 10,400 | 0 | 328 | 43,200 | 1,200 | 164,900 | 681,600 | 77 | 847,700 | 7,500 | 1,068,900 |
| Miss. Delta Region (73) | 513,700 | 529,200 | 1,042,900 | 71,700 | 64,400 | 1,964 | 332,400 | 8,500 | 125,300 | 240,200 | 4,800 | 378,800 | 15,100 | 1,769,300 |
| Atchafalaya/Vermillion (41) | 126,300 | 200 | 126,500 | 2,700 | 0 | 99,900 | 102,700 | 0 | 108,900 | 121,500 | 0 | 230,400 | 1,900 | 461,500 |
| Calcasieu Lake (55) | 80,500 | 2,100 | 82,600 | 0 | 0 | 32,800 | 32,800 | 0 | 0 | 0 | 500 | 0 | 6,500 | 122,000 |
| Sabine Lake (67) | 38,800 | 71,200 | 110,000 | 80,200 | 4,400 | 600 | 85,200 | 0 | 0 | 186,900 | 100 | 187,100 | 11,400 | 239,300 |
| Galveston Bay (82) | 0 | 94,900 | 94,900 | 51,500 | 7,400 | 0 | 58,900 | 200 | 0 | 70,400 | 3,800 | 74,400 | 1,100 | 393,700 |
| Brazos River (57) | 0 | 300 | 300 | 6,700 | 0 | 0 | 6,800 | 0 | 0 | 12,400 | 200 | 12,600 | 0 | 19,600 |
| Matagorda Bay (54) | 0 | 43,500 | 43,500 | 28,700 | 200 | 0 | 28,900 | 0 | 0 | 6,900 | 100 | 7,100 | 6,400 | 85,900 |
| San Antonio Bay (100) | 0 | 32,900 | 32,900 | 24,100 | 4,200 | 0 | 28,300 | 100 | 0 | 1,500 | 700 | 2,300 | 3,500 | 67,000 |
| Aransas Bay (92) | 0 | 30,700 | 30,700 | 39,600 | 2,400 | 0 | 42,000 | 1,700 | 0 | 7,900 | 1,200 | 10,700 | 13,900 | 97,400 |
| Corpus Christi Bay (59) | 0 | 12,200 | 12,200 | 6,500 | 800 | 0 | 7,300 | 0 | 0 | 1,300 | 100 | 1,400 | 8,700 | 29,500 |
| Laguna Madre (100) | 0 | 67,800 | 67,800 | 193,200 | 100 | 0 | 193,400 | 800 | 0 | 21,800 | 0 | 22,700 | 166,800 | 450,600 |
| Gulf of Mexico Total | 856,700 | 1,122,000 | 1,978,800 | 1,726,900 | 89,500 | 362,800 | 2,179,300 | 486,500 | 399,900 | 5,472,100 | 26,700 | 6,385,200 | 419,200 | 10,962,500 |

**Table 7. Total Acreage (Acres) of the Coastal Barrier Resource System within the Gulf States Area (Adapted from Watzin and Baumann, 1988)[1]**

| State | FL | WL | IOW | OW | D | Total | Total w/o water | Str. (no.) |
|---|---|---|---|---|---|---|---|---|
| Florida | 14,090.6 | 6,725.5 | 534.1 | 19,839.3 | 181.6 | 41,371.1 | 20,997.7 | 313 |
| Alabama | 2,930.6 | 2,050.3 | 3.7 | 5,683.6 | 9.8 | 10,678.0 | 4,990.7 | 16 |
| Mississippi | 553.3 | 1,151.1 | 0.5 | 2,600.9 | 3.6 | 4,309.4 | 1,708.0 | 44 |
| Louisiana | 4,456.8 | 27,259.3 | 1,014.3 | 25,451.8 | 61.1 | 59,243.3 | 31,777.2 | 59 |
| Texas | 46,692.8 | 126,315.8 | 4,969.1 | 3,297.6 | 289.6 | 181,564.9 | 173,298.2 | 276 |
| TOTAL | 68,724.1 | 163,502.0 | 6,521.7 | 56,873.2 | 545.7 | 297,166.7 | 232,771.8 | 708 |

[1] From areal photographs taken in 1982 and 1983
FL = Fastland (nonwetlands)
WL = Wetland
IOW = Interior Open Water
OW = Open Water
D = Developed
Str = Structures (Any walled or roofed building other than gas or liquid storage tank)

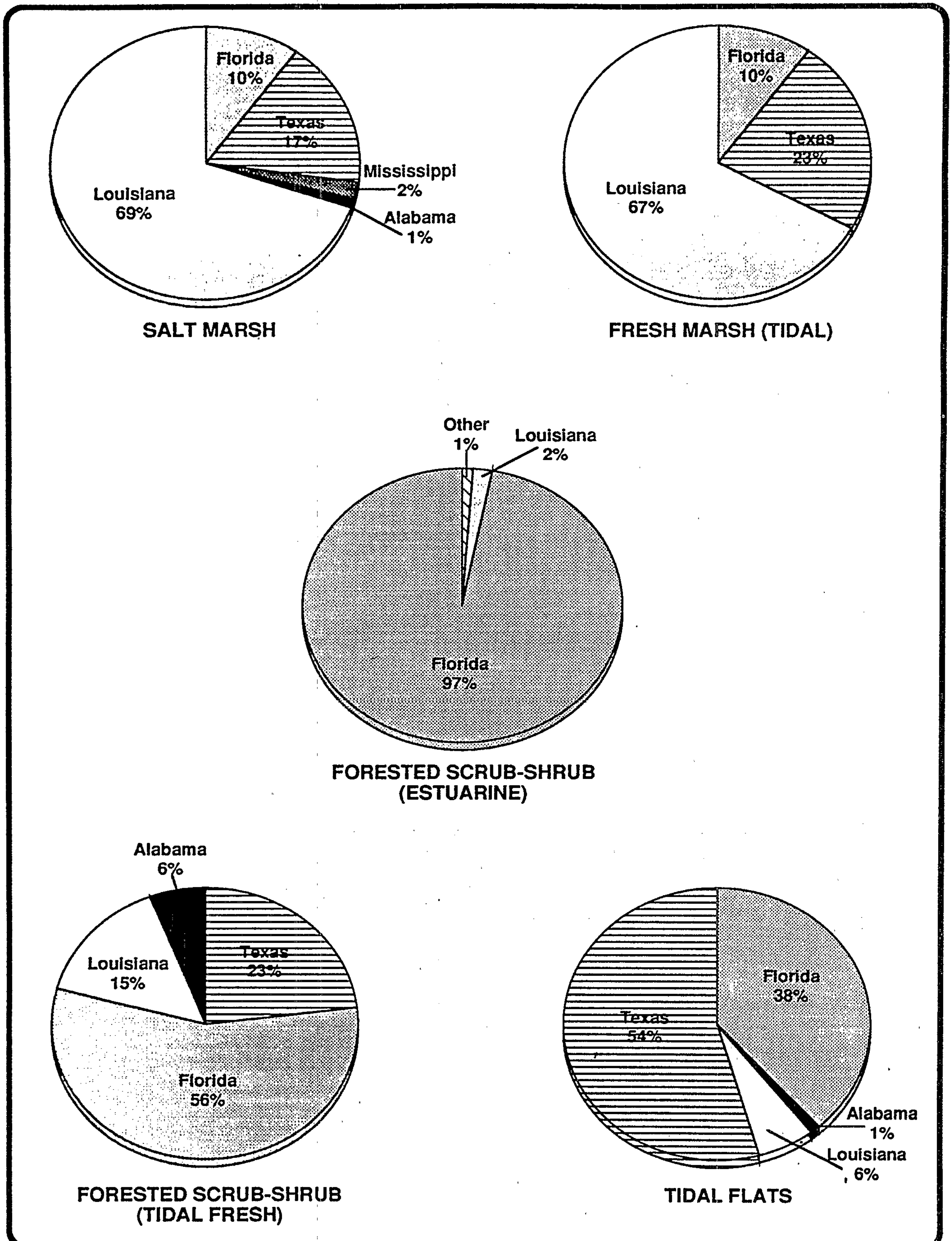


**Figure 1. Acreage Percentages of Coastal Emergent Wetlands by State (from NOAA, In Press).**

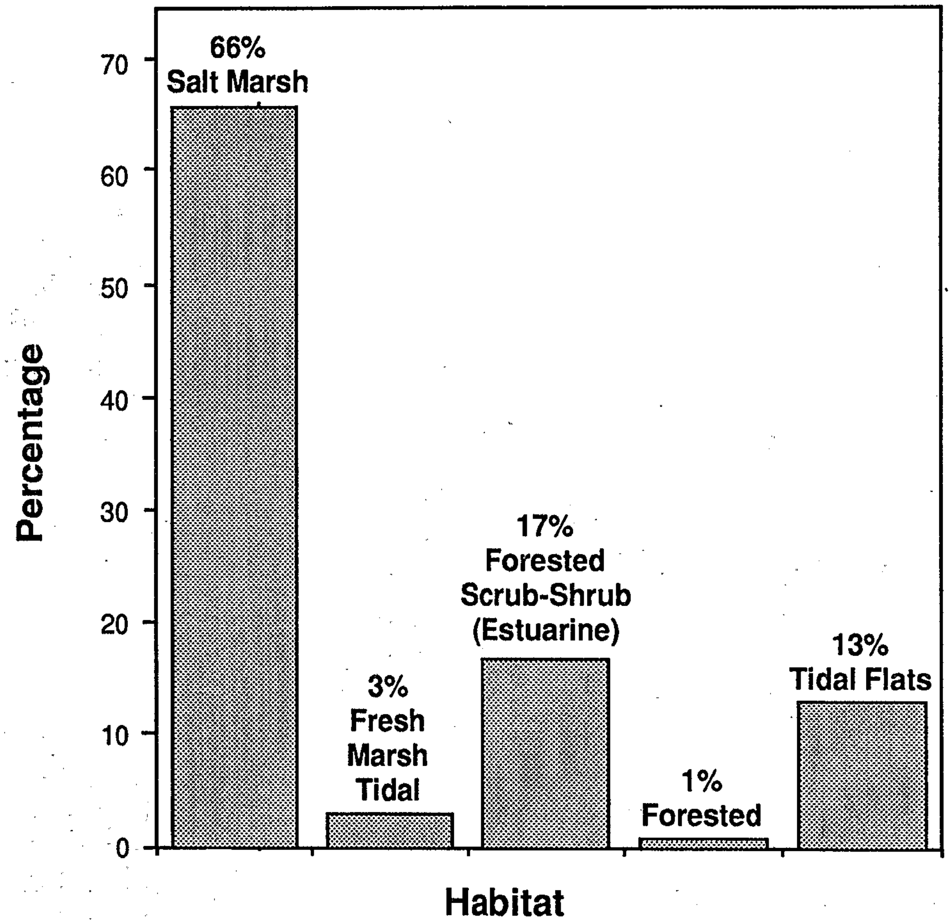


Figure 2. Percentage of Specified Habitat Gulf-Wide (From NOAA National Estuarine Inventory, 1991).

Texas coastal zone was defined as the area from the inner Continental Shelf to approximately 40 mi (64.5 km) inland and includes all estuaries and tidally influenced streams and adjacent wetlands. The zone was divided into seven areas from the Texas-Louisiana boundary southwestward to the Rio Grande: 1) Beaumont-Port Arthur (Fisher et al., 1973); 2) Galveston-Houston (Fisher et al., 1972), 3) Bay City-Freeport (McGowen et al., 1976a); 4) Port Lavaca (McGowen et al., 1976b); 5) Corpus Christi (Brown et al., 1976); 6) Kingsville, (Brown et al., 1977); and 7) Brownsville-Harlington (Brown et al., 1980). The regional resources of each of these seven areas is included in a separate Environmental Geologic Atlas. The seven atlases cover about 20,000 mi$^2$ (51,800 km$^2$), including 2,100 mi$^2$ (5,440 km$^2$) of bays and estuaries, 367 mi (592 km) of Gulf coastline, and 1,425 mi (2,298 km) of bay, estuary, and lagoon shoreline. A composite was made from the seven atlases to illustrate acreages of wetland habitats and associated areas in coastal counties of Texas (Table 8).

The Florida Department of Environmental Regulation (1978) analyzed a number of biophysical, social, economic, and environmental quality factors for coastal habitats. This inventory included acreages of marine sea grasses, coastal marshes, beaches, mangroves, and estuarine shoreline (Table 9), but usually did not include areas less than 40 ac (16.2 ha) in size. The potential applications of these data include: 1) an aid in making land-use projections; 2) an indicator of potential land-use hazards, problems, and opportunities; and, 3) a tool in developing policies to protect and utilize coastal resources wisely. A statewide inventory of marine habitats is currently being conducted by the Florida Department of Natural Resources, and reports for several estuaries are completed (e.g., Haddad and Hoffman, 1985).

Hundreds of square miles of valuable wetlands have been lost in the Mississippi River Deltaic Plain (MRDP). The causes of this loss include human activities and natural events. May and Britsch (1987) present data on maps which document the changes that have occurred in land area within the MRDP from approximately 1932 to 1983 (Table 10). Land loss and land accretion were mapped by comparing U.S. Coast and Geodetic Survey Air Photo Compilation sheets dated 1931, 1932, and 1933 (T-sheets), U.S. Geological Survey 15-minute quadrangle maps, and 1983 National High Altitude Program photography. The T-sheets were the oldest suitable data source containing the level of detail necessary for mapping. The T-sheets were produced before wetland loss had become a significant problem and provide an excellent temporal baseline. T-sheets were not available for some quadrangles; for those areas the oldest 15-minute USGS quadrangles were used to establish the baseline. Land loss and accretion data were delineated on the most recent 15-minute quadrangle. The 1983 photography was the most recent color infrared photography available at the time of the study.

TABLE 8. Texas Coastal Habitats (Acres) (from Fisher et al., 1972, 1973; McGowen et al., 1976a,b; Brown et al., 1980)[1]

| County | Beach | Unvegetated Coastal Mud Flat | Vegetated Flat | Salt Water Marsh | Brackish Water Marsh | Brackish to Fresh Water Marsh | Sand Flats | Oysters | Grass Flats |
|---|---|---|---|---|---|---|---|---|---|
| Aransas | 960 | | 11,136 | 9,984 | | 3,776 | 15,744 | 4,224 | 4,160 |
| Bee | | | | | | | | 448 | |
| Brazoria | 1,408 | | 1,280 | 9,984 | | 35,200 | 1,344 | 64 | 640 |
| Calhoun | 3,840 | | 23,808 | 18,560 | | 1,280 | 7,168 | 3,840 | 6,080 |
| Cameron | 1,152 | | 3,392 | | | | 56,704 | | 66,112 |
| Chambers | 64 | | 512 | 7,680 | 18,752 | 35,200 | | 3,840 | 640 |
| Fort Bend | | | | | | 512 | | | |
| Galveston | 2,176 | 128 | 26,240 | 19,200 | 3,520 | 16,000 | | 52,480 | 3,840 |
| Harris | | | | 128 | | 4,480 | | | |
| Jackson | | | | 3,200 | | 320 | | | |
| Jefferson | 640 | 1,280 | 5,760 | 6,400 | 35,840 | 76,224 | | | |
| Kenedy | 1,024 | | 10,560 | | | | 112,320 | | 10,048 |
| Kleberg | 832 | | 11,328 | | | | 22,208 | | 23,784 |
| Matagorda | 3,264 | | 3,584 | 17,280 | 17,920 | | 7,680 | 32,128 | 4,288 |
| Nueces | 768 | | 11,200 | 960 | | | 9,920 | 3,136 | 19,776 |
| Orange | | | | 3,840 | | 20,480 | | | |
| Refugio | | | | 3,840 | | 512 | 1,728 | 1,408 | 128 |
| San Patricio | | | 128 | 3,264 | | 4,032 | 2,368 | | |
| Starr | | | | | | | | | |
| Victoria | | | | 1,280 | | | | | |
| Willacy | 512 | | | 960 | | | 40,256 | | 30,720 |
| TOTAL | 16,640 | 1,408 | 108,928 | 106,560 | 76,032 | 198,016 | 277,440 | 101,568 | 169,216 |

[1] Compiled from areal photographs taken in 1956 and 1960 and U.S. Coast and Geodetic Survey Nautical Charts 1960-1972

Table 9. Acreage (Acres) of Selected Habitats in Florida (from Florida Department of Environmental Regulation, 1978)[1]

| County | Marine Grass Bed | Selected Coastal Marshes | Beaches and Mangroves | Beaches and Dunes | Spoil Islands | Selected Beaches Estuarine |
|---|---|---|---|---|---|---|
| Escambia | 3,769 | 2,720 | 0 | 1,715 | 0 | 1,363 |
| Santa Rosa | 3,756 | 7,942 | 0 | 332 | 0 | 1,420 |
| Okaloosa | 2,828 | 652 | 0 | 1,408 | 0 | 1,638 |
| Walton | 1,004 | 3,673 | 0 | 2,374 | 0 | 1,068 |
| Bay | 8,294 | 6,624 | 0 | 3,814 | 0 | 2,649 |
| Gulf | 6,604 | 633 | 0 | 2,400 | 0 | 723 |
| Franklin | 9,081 | 20,518 | 0 | 3,308 | 0 | 1,619 |
| Wakulla | 19,609 | 19,596 | 0 | 19 | 0 | 0 |
| Jefferson | 8,729 | 4,563 | 0 | 0 | 0 | 0 |
| Taylor | 4,300 | 23,916 | 0 | 0 | 0 | 0 |
| Dixie | 4,780 | 23,532 | 0 | 70 | 44 | 0 |
| Levy | 33,248 | 39,212 | 0 | 0 | 0 | 0 |
| Citrus | 51,622 | 30,643 | 3,020 | 0 | 108 | 0 |
| Hernando | 7,225 | 11,270 | 76 | 172 | 0 | 0 |
| Pasco | 6,361 | 3,705 | 294 | 44 | 307 | 19 |
| Pinellas | 9,785 | 0 | 1,728 | 678 | 0 | 96 |
| Hillsborough | 4,876 | 2,451 | 4,723 | 0 | 64 | 0 |
| Manatee | 8,415 | 1,081 | 4,006 | 742 | 0 | 160 |
| Sarasota | 1,657 | 895 | 831 | 1,414 | 0 | 492 |
| Desoto | 0 | 115 | 76 | 0 | 0 | 0 |
| Charlotte | 14,335 | 12,166 | 16,883 | 0 | 0 | 0 |
| Lee | 38,284 | 14,201 | 34,271 | 1,900 | 0 | 454 |
| Collier | | 159,539 | 81,663 | 1,696 | 0 | 0 |

[1] From maps and photographs made in 1972-1975

**Table 10. Calculations of Total Loss/Accretion (Acres) in the Mississippi River Deltaic Plain Recorded on Quadrangle Maps (from May and Britsch, 1987)[1]**

| Quadrangle | Loss | Accretion |
|---|---|---|
| Barataria | 30,609 | <100 |
| Bay Dogris | 34,154 | <100 |
| Bayou Du Large | 19,790 | <100 |
| Bayou Sale | 6,262 | 271 |
| Belle Isle | 7,926 | 3,325 |
| Bonnet Carre | 6,212 | 1,072 |
| Black Bay | 8,827 | 303 |
| Breton Island | 7,534 | 3,888 |
| Caillou Bay | 10,480 | 284 |
| Cat Island | 2,858 | <100 |
| Centerville | 22 | <100 |
| Chef Menteur | 12,360 | 613 |
| Covington | 1,360 | <100 |
| Cutoff | 11,577 | 537 |
| Drouen | 6,033 | 897 |
| Dulac | 32,911 | 564 |
| East Delta | 37,433 | 12,501 |
| Empire | 28,414 | 563 |
| Fort Livingston | 16,810 | 651 |
| Gibson | 12,785 | 447 |
| Hahnville | 7,714 | <100 |
| Houma | 3,958 | <100 |
| Jeanerette | 1,609 | 604 |
| Lac Des Allemands | 2,584 | <100 |
| Lake Decade | 15,849 | <100 |
| Lake Felicity | 28,028 | <100 |
| Leeville | 15,869 | 511 |
| Mitchell Key | 1,422 | <100 |
| Morgan City | 19,677 | 2,321 |
| Morgan Harbor | 8,689 | <100 |
| Mount Airy | 1,111 | <100 |
| New Orleans | 5,937 | <100 |
| Oyster Bayou | 3,091 | <100 |
| Point Chicot | 2,697 | 1,273 |
| Pointe a La Hache | 15,646 | 2,166 |
| Point Au Fer | 4,702 | 6,392 |
| Ponchatoula | 22,687 | <100 |
| Rigolets | 5,246 | 273 |
| Slidell | 2,336 | <100 |
| Southwest Pass | 2,929 | 697 |
| Spanish Fort | 377 | 285 |

(Continued)

**Table 10. Continued**

| Quadrangle | Loss | Accretion |
|---|---|---|
| Springfield | 436 | <100 |
| Terrebonne | 10,126 | 822 |
| Thibodeaux | 365 | <100 |
| Timbalier | 9,036 | 1,026 |
| Three Mile Bay | 3,960 | <100 |
| Venice | 24,990 | 7,214 |
| West Delta | 47,475 | 7,797 |
| Yscloskey | 12,793 | 1,338 |
| TOTALS | 583,696 | 59,813 |

[1] From photographs made in 1931-33 and 1983

The accuracy of this mapping is related to the quality of the photography and the skill of the photo interpreter. Overall, the photography was excellent but occasionally poor photo quality made interpretation difficult. Another problem was the short-term water fluctuations along the coast. Differences in winds and tides at various locations when the photographs were taken had an effect on delineation of habitats. The complexity and amount of detail made perfect alignment between overlays somewhat difficult. However, these factors are a minor source of error considering the entire scope of the project. The results of this study document that land loss has far exceeded land accretion for all areas measured.

## Functions and Values of Gulf Coastal Habitats

Gulf coast habitats consist of a wide range of diverse ecological settings which provide many important direct and indirect benefits to society. An obvious benefit provided by these diverse habitat types is the availability of food and shelter for many coastal species, including many threatened and endangered species.

Coastal wetlands and seagrass meadows are very productive systems (Table 11). In general, net primary productivity of tidal marsh systems varies from 200-3000 g dry weight/$m^2$/yr. This primary productivity supports a rich and varied wetland flora and fauna. For example, Subrahmanyam et al. (1976) reported over 55 species of benthic macroinvertebrates in an intertidal *Juncus* marsh of northwest Florida; benthic organisms are important components in feeding strategies of marsh resident and estuarine fishes (Subrahmanyam and Coultas, 1980). Seagrass meadows produce approximately 300 g C/$m^2$/yr and support an equally diverse fauna (Zieman, 1982; Zieman and Zieman, 1989).

Not all of the primary productivity of coastal wetlands is utilized as an energy source within the wetland. Much of the organic matter produced in tidal marshes is exported to adjacent estuaries as detritus (Odum and de la Cruz, 1967). Seagrasses (Zieman, 1982; Zieman and Zieman, 1989), and mangroves (Odum, and Heald, 1972) also produce copious quantities of detritus which is an important energy source in estuarine communities. The role of detritus produced in coastal habitats and other sources of organic matter in the energy dynamics of estuarine systems is reviewed by Odum, Zieman, and Heald (1973), and Nixon (1980).

The function of coastal wetlands and seagrass meadows in primary and secondary production, cycling of nutrients, harvest of fiber, fur, commercial, sport fish and shellfish, as well as aesthetic considerations, is linked to the presence, movement, quality, and quantity of water. Clark and Clark (1979) and Sather and Smith (1984) discuss the manner in which wetland functions are related to the hydrologic cycles. Wetlands have the potential to store water and therefore play an important role in the control of flood water. Wetlands associated with streams can provide flood water storage, decrease stream water velocities, reduce peaks during flooding, and influence the duration of floods. Also, wetlands serve to temper the impacts of droughts by holding and slowly releasing water. Many wetland systems can recharge the groundwater and act as a freshwater discharge point. Several factors control the magnitude of the ground water recharge function including the nature of the substrate, confining layers, and surface outlets, as well as the amount of edge, and type and amount of vegetation. Some scientists believe that wetlands act more as a freshwater discharge point than a recharge area, and that they are sometimes good indicators of potential water supplies for municipalities (Sather and Smith, 1984).

Table 11. Production Rate for Various Wetland Habitats

| Habitat/Species | Type of Measurement | Productivity | Citation |
|---|---|---|---|
| **SALT MARSH** | | | |
| Salt Marsh | Primary Production | 200-2,000 gms dry wt/$m^2$/yr | Turner, 1976 |
| Louisiana Marshes | Primary Production | 700-3,000 gms dry wt/$m^2$/yr | Hopkins and Day, 1989 |
| Mississippi Marshes | Primary Production | 500 gms C/$m^2$/yr | Gabriel and de la Cruz, 1974 |
| Salt Marsh | Net Primary Production | 0.8-8.2 gms C/$m^2$/day | Bell, F.W., 1989 |
| Marsh Zone "Low" | Primary Production | 1,840 gms dry wt/$m^2$/yr | Kruczynski, 1978a |
| Marsh Zone "Middle" | Primary Production | 1,289 gms dry wt/$m^2$/yr | Kruczynski, 1978a |
| Marsh Zone "High" | Primary Production | 3,130 gms dry wt/$m^2$/yr | Kruczynski, 1978a |
| Salt Marsh | Organic Production | 2 kg dry wt/$m^2$/yr | Eleuterius, 1972 |
| Marshes | Average Value | 1,300 gms C/$m^2$/yr | Thayer, et al., 1981 |
| Fresh Marsh | Organic Production | 400 gms dry wt C/$m^2$/yr | Eleuterius, 1972 |
| Tidal Marshes | Primary Production | 300 gms C/$m^2$/yr | Lewis, 1989 |
| Needle Rush | Primary Production | 390-1,140 gms dry wt/$m^2$/yr | Lewis, 1989 |
| Smooth Cordgrass | Primary Production | 130-700 gms dry wt/$m^2$/yr | Lewis, 1989 |
| Mississippi *Juncus* | Exported as Detritus | 28 gms C/$m^2$/yr | Hackney, 1977 |
| Mississippi *Juncus* | Exported as Nitrogen | 0.2 gms N/$m^2$/yr | Hackney, 1977 |
| **SEAGRASS** | | | |
| Seagrass | Net Primary Production | 0.5-16.0 gms C/$m^2$/day | Bell, 1989 |
| Seagrass | Primary Production | 500 gms C/$m^2$/yr | Thayer and Ustach, 1981 |

Table 11. (Continued)

| Habitat | Type of Measurement | Productivity | Citation |
|---|---|---|---|
| Seagrass | Primary Production | 100-900 gms $C/m^2/yr$ | Thayer, et al., 1981 |
| Seagrass, N/W Florida (Northwest) | Primary Production | 500 gms $C/m^2/yr$ | Kruczynski, 1978 |
| Seagrass, Tampa to Everglades | Primary Production | 600 gms $C/m^2/yr$ | Kruczynski, 1978 |
| Seagrass, Florida Bay | Primary Production | 700 gms $C/m^2/yr$ | Kruczynski, 1978 |
| Seagrass, Dry Tortugas | Primary Production | 700 gms $C/m^2/yr$ | Kruczynski, 1978 |
| Seagrass and Epiphytes | Primary Production | 730 gms $C/m^2/yr$ | Lewis, 1989 |
| Seagrass, Turtlegrass | Primary Production | 125-4,000 gms $C/m^2/yr$ | Phillips, 1980b |
| Seagrass, Turtlegrass, South Florida | Primary Production | 0.9-16 gms $C/m^2/day$ | Phillips, 1980b |
| Seagrass, Eelgrass | Primary Production | 50-960 gms $C/m^2/yr$ | Phillips, 1980b |
| Seagrass, Subtropical Zone, Average Total | Primary Production | 1,000 gms $C/m^2/yr$ | Phillips, 1980b |
| Seagrass, Temperate Zone, Average Total | Primary Production | 480 gms $C/m^2/yr$ | Phillips, 1980b |
| Seagrass, *Thalassia* | Average Production | 580-900 gms $C/m^2/yr$ | Brylinsky, 1967 |
| Seagrass | Average Production | 300 gms $C/m^2/yr$ | Thayer, et al., 1981 |
| Submerged Macrophytes Florida Gulf Shelf | Primary Production | $5.59 \times 10^6$ mt/yr | Kruczynski, 1978 |
| Seagrass, *Thalassia* | Biomass Values | 1,500 gms $C/m^2/yr$ | McRoy and McMillan, 1977 |
| Seagrass, *Halodule* | Biomass Values | 93 gms $C/m^2/yr$ | McRoy and McMillan, 1977 |
| Seagrass, *Ruppia* (S. Texas) | Biomass Values | 100-150 gms dry $wt/m^2/yr$ | Pulich, 1985 |

Table 11. (Continued)

| Habitat | Type of Measurement | Productivity | Citation |
|---|---|---|---|
| Seagrass, *Halodule* (S. Texas) | Biomass Values | 200-600 gms dry wt/$m^2$/yr | Pulich, 1982 |
| Seagrass, *Halodule* (S. Texas) | Primary Production | 15 gms C/$m^2$/yr | Dunton, 1989 |
| Seagrass, *Syringodium* | Biomass Values | 23 gms C/$m^2$/yr | McRoy and McMillan, 1977 |
| Seagrass, *Ruppia* (S. Texas) | Primary Production | 38 gms C/$m^2$/yr | Dunton, 1989 |
| **MANGROVES** | | | |
| Mangroves | Primary Production | 400 gms C/$m^2$/yr | Lugo and Snedaker, 1974 |
| Mangroves Forests | Primary Production | 1,132 gms C/$m^2$/yr | Lewis, 1989 |
| Mangroves Biomass | Primary Production | 390-8,700 gms dry wt C/$m^2$/yr | Lugo and Snedaker, 1974 |
| Mangroves Litter | Primary Production | 15-4,900 gms C/$m^2$/yr | Lugo and Snedaker, 1974 |
| Mangroves | Net Primary Production | 1.0-12.6 gms C/$m^2$/day | Bell, 1989 |
| Mangroves, Red, Black & White | Production Values | 400 gms C/$m^2$/yr | Lugo and Snedaker, 1974 |
| **PHYTOPLANKTON** | | | |
| Phytoplankton | Primary Production | 350 g C/$m^2$/yr | Thayer and Ustach, 1981 |
| Phytoplankton N/W, FL, Tampa to Everglades | Primary Production | 57 gms C/$m^2$/yr | Kruczynski, 1978 |
| Phytoplankton over Grassbeds Florida Gulf Shelf | Primary Production | 0.57 x $10^6$ mt/yr | Kruczynski, 1978 |
| Phytoplankton Bays deeper than 2 M | Primary Production | 340 gms C/$m^2$/yr | Lewis, 1989 |

Table 11. (Continued)

| Habitat | Type of Measurement | Productivity | Citation |
|---|---|---|---|
| Phytoplankton Bays shallower than 2 M | Primary Production | 50 gms $C/m^2/yr$ | Lewis, 1989 |
| Water Column | Net Primary Production | 0.9 gms $C/m^2/day$ | Bell, 1989 |
| Mud Flat | Net Primary Production | 0.5 gms $C/m^2/day$ | Bell, 1989 |
| Micro Algae | Primary Production | 70 gms $C/m^2/yr$ | Lewis, 1989 |
| Benthic Micro Algae | Primary Production | 150 gms $C/m^2/yr$ | Lewis, 1989 |
| Benthic Algae Florida Gulf Shelf | Primary Production | 0.10-0.16 x $10^6$ mt/yr | Kruczynski, 1978 |
| Benthic Algae | Net Production | 45 mg $C/m^2/hr$ | Kruczynski, 1978 |
| Phytoplankton | Chlorophyll a Standing Crop | 7 mg Chl $a/m^2$ | Thayer and Ustach, 1981 |
| Phytoplankton | Net Production | 300 gms $C/m^2/yr$ | Thayer and Ustach, 1981 |
| **WATER COLUMN/COMMUNITY PRODUCTION** | | | |
| Lake Cataouatche, LA | Gross Production | 696 gms $C/m^2/yr$ | Hopkinson and Day, 1979 |
| Lake Cataouatche, LA | Chlorophyll a | 11-55 $mg/m^3$ | Hopkinson and Day, 1979 |
| Lake Cataouatche, LA | Gross Production | 876 gms $O_2/m^2/yr$ | Hopkinson and Day, 1979 |
| Lake Cataouatche, LA | Night Respiration | 1,205 gms $O_2/m^2/yr$ | Hopkinson and Day, 1979 |
| Lake Cataouatche, LA | Net Community Production | -350 gms $O_2/m^2/yr$ | Hopkinson and Day, 1979 |
| Little Lake, LA | Gross Production | 0-15.2 gms $O_2/m^2/day$ | Hopkinson and Day, 1979 |
| Little Lake, LA | Night Respiration | 1,205 gms $O_2/m^2/yr$ | Hopkinson and Day, 1979 |
| Little Lake, LA | Net Community Production | -117 gms $O_2/m^2/yr$ | Hopkinson and Day, 1979 |
| Lake Salvador, LA | Gross Production | 402 gms $O_2/m^2/yr$ | Hopkinson and Day, 1979 |

Table 11. (Continued)

| Habitat | Type of Measurement | Productivity | Citation |
|---|---|---|---|
| Lake Salvador, LA | Night Respiration | 602 gms $O_2/m^2/yr$ | Hopkinson and Day, 1979 |
| Bayou Chevrevil | Gross Production | 888 gms $O_2/m^2/yr$ | Hopkinson and Day, 1979 |
| Lac des Allemands | Gross Production | 3,284 gms $O_2/m^2/yr$ | Hopkinson and Day, 1979 |
| Lac des Allemands | Upper Basin Gross Production | -450 gms $O_2/m^2/yr$ | Hopkinson and Day, 1979 |
| Lac des Allemands | Lower Basin Gross Production | 0-54 gms $O_2/m^2/yr$ | Hopkinson and Day, 1979 |
| Barataria Basin | Gross Production | 1,850 gms $O_2/m^2/yr$ | Hopkinson and Day, 1979 |
| Louisiana, 4 stations | Chlorophyll a | 55-230 $mg/m^3$ | Hopkinson and Day, 1979 |
| Louisiana, other stations | Chlorophyll a | 11 $mg/m^3$ | Hopkinson and Day, 1979 |
| Corpus Christi Bay, TX | Gross Production | 440 g $C/m^2/yr$ | Stockwell, 1989 |
| Corpus Christi Bay, TX | Gross Production | 175 g $C/m^2/yr$ | Flint and Kalke, 1985 |
| San Antonio Bay, TX | Gross Production | 450 g $C/m^2/yr$ | MacIntyre and Cullen, 1988 |
| Southern, TX | Primary Production (Blue-green algae mat) | 0.6 g $C/m^2/day$ - 2.4 g $C/m^2/day$ | Pulich and Rabalais, 1986 |

Wetlands have the capacity to improve water quality by filtering pollutants and sediments from flowing waters. As water flows through wetlands it undergoes certain changes, primarily as a result of: 1) a reduction in the velocity of flowing water as it enters or passes through a wetland, 2) decomposition of organic substances, 3) metabolic activities of plants and animals, 4) photosynthesis, and 5) sediment binding of ions and particles. Because of this capacity to improve the quality of water, natural and artificial wetlands have been used for the treatment of waste waters (Sather and Smith, 1984).

The ability of wetland systems to bind nitrogen and phosphorus is reviewed by Adamus (1983). This function depends upon several attributes, including the capacity of the vegetation, on a net annual basis, to assimilate and transfer to deep sediments more nutrients than are released through leaching and decay. The substrate of the wetland system accumulates organic matter on a net annual basis. Generally, sediments accrete faster than they are removed, and the rate of denitrification in wetland soils consistently exceeds the rate of nitrogen fixation.

In an attempt to counter achievement of economic gains through development of wetlands, several studies have been performed to place a defensible monetary value on wetlands based upon the tangible and intangible benefits which wetlands provide to society. However, no standard method for estimating the value of wetlands has been established. Also, it is difficult to place a monetary value on many wetland functions. As a result, available estimates vary greatly.

Several studies have been conducted to quantify the role of wetlands in fisheries production. Turner (1977) hypothesized that the abundance and type of commercially valuable quantities of penaeid shrimp are directly related to the area and type of estuarine intertidal vegetation. Numerous factors influence the distribution and abundance of shrimp, including the nature of the substrate, area of estuarine lagoons or wetlands, and other physical and biological factors (Kutkuhn, 1966). Turner demonstrated that there is a direct relationship between the volume of commercial penaeid shrimp landings and the area of intertidal land over a wide geographical area. He also presented a discussion of the limitations of such a study, which include the fact that:

1. Fishermen are taxed based on pounds of shrimp landed; therefore, actual landing amounts may be under reported.

2. Species other than commercial penaeid shrimp may be included.

3. In some instances, shrimp are "de-headed" at sea, which is not always easily determined.

4. Shrimp landed in one area may have been caught in another.

5. Estimates of intertidal area are not uniformly accurate. However, the author believes that if a statistically significant relationship between land and yield for a variety of locations

were demonstrated, the relationship probably would be real even with these sampling inaccuracies.

In addition, poundage landed is based on both effort and the size of the population. In Turner's study, the data from more than 27 locations, representing a stabilized and developed fishing effort, were averaged for several years. Several efforts were made to assure the data quality, and data were normalized when necessary. Intertidal areas were defined as coastal areas vegetated by the salt marsh macrophytes, *Spartina* spp., *Juncus* spp., or mangroves.

The annual shrimp yields per area of intertidal land ranged from 8.9 to 178 lb/ac (10 to 200 kg/ha), and a highly significant ($P = 0.02$) relationship was found between yield and degree latitude (Turner, 1977). Also, there was a good relationship ($r = 0.69$) between the area of intertidal land and yields of shrimp caught in inshore Louisiana waters. No statistically significant relationship was found between the average landings (inshore) and estimates of estuarine water surface, average of estuarine water surface, or average depth or volume. Of the total amount of shrimp caught (inshore), the percentage of brown shrimp was highly correlated with the percentage of saline marsh vegetation ($r = 0.92$).

These analyses lacked data on the amount of submerged macrophytes in the estuarine waters. Inclusion of such data could improve the regression equation relating the weight of shrimp to vegetated estuarine habitats. Also, there are insufficient data to determine the impact of mangrove litter production with changes in latitude. Given these restrictions, a positive relationship was demonstrated between commercial yields of penaeid shrimp with area, intertidal vegetation, and latitude for 27 locations. Thus, on a regional basis, the inshore yields of shrimp are directly related to areas of estuarine vegetation (Turner, 1979).

The association between shrimp life cycles and estuarine and wetland acreage is discussed by Turner and Boesch (1988). The complex stock-recruitment relationships for penaeid shrimp are presented. Adult stock is dependent upon juvenile and postlarvae abundance. Causal relationships exist, given reasonable assumptions, between wetland acreage and juvenile abundances, which affect subsequent adult densities. Recruitment is dependent upon acceptable climatic factors and habitat quality; conversely, adverse changes in habitat quality and quantity can be detrimental to shrimp production.

Long-term yields of shrimp are linearly related to both the quantity and quality of intertidal habitats. Positive relationships were found between penaeid stock sizes (reflected in annual harvests) and coastal wetland areas within specific regions around the world and in the Gulf of Mexico (Turner and Boesch, 1988). Correlation coefficients between amounts of shrimp harvested and amounts of wetlands (total length of mangrove-lined rivers) were: 0.76 for Australia, 0.74 for Malaysia, and 0.62 for the Philippines. A correlation of 0.97 was reported between shrimp landing and intertidal vegetation in the Gulf of Mexico. Turner and Boesch (1988) also documented changes in penaeid shrimp stock following changes in intertidal wetland habitats. Changes in quantity of vegetation resulted in changes in quantity of shrimp stock in Louisiana, Kuwait, Saudi Arabia, El Salvador, and Vietnam. Decline of shrimp yields in Japan were related to the cumulative losses of intertidal lands ($r = 0.85$).

Several experimental studies have demonstrated a gain in shrimp stocks following gains in wetland acreage or quality. A marsh rehabilitated by planting with *Spartina alterniflora* (smooth cordgrass) accumulated higher densities of juvenile and postlarvae penaeid shrimp than control sites which were not planted (Turner and Boesch, 1988). Thus, vegetative structure positively affected habitat selection. However, the presence of vegetation, in itself, does not define a healthy productive marsh for fisheries organisms. Evidence suggests that a large amount of edge is beneficial and that creeks and channels which connect the interior of the marsh with the open bay provide flushing necessary to maintain moderate soil salinities required for plant health and access to more of the marsh surface (Minello et al. 1986). Edge has also been increased with creation of "brush-parks"- which are areas of brush strategically placed in estuaries. Brush-parks accumulated higher densities of fish or stimulated growth/production in West Africa and other countries (Turner and Boesch, 1988).

Data presented by Turner and Boesch (1988) support the hypothesis that habitat quality and quantity control adult stocks of penaeid shrimp, and they conclude that conservation of habitat quality is of high significance to success of sustained recruitment. Because other aquatic animals inhabit these ecosystems and have life histories similar to shrimp, it is probable that the quality of wetland habitats can directly limit the productivity of other fisheries as well.

Turner (1982) suggests that where there are large areas of wetlands and estuaries, there are likely to be substantial fishing industries nearby. This relationship has been quantified in Louisiana, where coastal wetlands area is positively correlated with commercial landings of shrimp caught in inshore waters (Turner, 1977). The relationship between landings and area of open water is not statistically significant and appears to be negative. Data also suggest that species of shrimp landed are directly related to the kind of vegetation present (Zimmerman, Minello, and Zamora, Jr., 1984; Minello and Zimmerman, 1985).

Total animal production in various ecosystems is ranked in Table 12. In areas where plant production is high, animal production is high. Animal production in swamps and marshes (9.0 $gC/m^2/yr$) and estuaries (17.8 $gC/m^2/yr$) is higher than production in terrestrial ecosystems (2-3 $gC/m^2/yr$). Total animal production in estuaries is among the highest measured (17.8 $gC/m^2/yr$).

Turner (1982) concludes that although the exact mechanism coupling fisheries productivity and coastal habitat is not always clear, a strong correlation is clearly indicated. He further states that if the reported figures of a 1% loss of wetlands per year is equivalent to a 1% decline in the potential fishing yield, then the impact of wetland loss on cumulative loss in dockside dollar value over a 20-year period (1982-2002) would be $380 million (1982 dollars). The actual economic value could be three times higher than the dockside value as a result of value added during processing and delivery (Jones et al., 1974). Harris (1983) supports that conclusion. Harris compared the landings of spotted seatrout (*Cynoscion regalis*) and red drum (*Sciaenops ocellata*) from two geographically separate but similar bays (Tampa Bay and Charlotte Harbor, Florida). Because these two species spend so much time in estuaries, their catch and landings are an indication of the health of the estuaries in which they live. Tampa Bay yielded a much lower amount of these species than Charlotte Harbor, presumably because of greater cumulative habitat losses, changes in circulation, and pollution due to human sources.

Deegan et al. (1986) evaluated the relationship of physical factors and vegetation to fishery harvest for 64 estuaries in the Gulf of Mexico. Many estuaries along the coast were formed by combinations of tectonics, coastal processes, and riverine deposition. These three factors, acting together but on different time scales, have formed the present intertidal and inshore open water areas bordering the Gulf of Mexico. A listing of major estuaries in the five U.S. Gulf coastal states is given in Table 13. Gulf estuaries are generally shallow systems, averaging from 3.9 to 19.3 ft (1.2 to 5.9 m) in depth, and contain relatively large acreages of submerged and emergent vegetation. Except for estuaries associated with the Mississippi River, most Gulf coast estuaries have small volumes of riverine discharges compared to estuaries along the Atlantic coast.

The areal development of different vegetation types is related to the water budget of the estuary. The two principal sources of fresh water to most estuaries are rivers and streams and local rainfall. The percentage of the total fresh water input into estuaries from rivers and streams averages 42% over the Gulf and ranges from 0 to 99% in different locations. The area of emergent vegetation, including marsh or mangrove, was directly dependent upon intertidal area and rainfall. The area of emergent vegetation was not significantly related to river discharge through stepwise linear regression. However, all areas with high river discharge also have high rainfall amounts, which complicates the statistical analyses. An analysis of fishery catch statistics in estuaries from three

**Table 12. Preliminary Estimates of Animal Secondary Production, Consumption, Standing Stock and Turnover for Different Ecosystems (Adapted from Whittaker and Likens (1973) and Turner (1982))**

| Ecosystem Type | $gC/m^2$ Animal Biomass | % Animal Consumption Plant Production | $gC/m^2/yr$ Animal Production | Days Turnover of Animal Biomass |
|---|---|---|---|---|
| Rock, ice, and sand | 0.01 | 2 | 0.0004 | 9,125 |
| Desert scrub | 0.02 | 3 | 0.15 | 486 |
| Tundra and alpine meadow | 0.02 | 3 | 0.2 | 365 |
| Cultivated land | 0.02 | 1 | 0.3 | 243 |
| Boreal forest | 2.2 | 4 | 1.4 | 573 |
| Woodland and shrubland | 2.2 | 5 | 1.4 | 573 |
| Temperate evergreen forest | 4.5 | 4 | 2.4 | 684 |
| Temperate deciduous forest | 7 | 5 | 2.7 | 946 |
| Temperate grassland | 3.1 | 10 | 3.3 | 343 |
| Open ocean | 1.1 | 40 | 3.4 | 118 |
| Tropical seasonal forest | 5.4 | 6 | 4.0 | 493 |
| Lake and stream | 2.2 | 20 | 5.5 | 146 |
| Tropical rain forest | 9.0 | 7 | 6.5 | 505 |
| Savanna | 6.8 | 15 | 7.0 | 354 |
| Continental shelf | 9.0 | 30 | 7.3 | 450 |
| Swamp and marsh | 4.5 | 8 | 9.0 | 183 |
| Upwelling zones | 4.5 | 35 | 12.5 | 131 |
| Estuaries | 6.8 | 15 | 17.8 | 139 |
| Algal bed and reef | 9.0 | 15 | 18.3 | 180 |
| TOTAL CONTINENTAL | 3.1 | 7 | 2.5 | 452 |

Table 13. Characteristics of Some Gulf of Mexico Estuaries (Adapted from Deegan et al., 1986)[1]

| Estuary | Submerged Vegetation Areas (acres) | Emergent Vegetation Areas (acres) | Open Waters (acres) | Mean Depth (m) | Mean Annual River Discharge (CMS) |
|---|---|---|---|---|---|
| Florida Bay | 259,712 | 213,675 | 606,675 | 1.3 | 283.1 |
| Ten Thousand Is. | 4,831 | 178,147 | 103,782 | 1.4 | 9.5 |
| Charlotte Harbor | 53,270 | 64,693 | 277,896 | 2.3 | 86.0 |
| Sarasota Bay | 7,611 | 4,070 | 34,760 | 1.7 | 2.3 |
| Tampa Bay | 29,615 | 21,046 | 306,046 | 3.3 | 43.8 |
| Apalachicola Bay | 9,380 | 21,307 | 170,039 | 2.9 | 763.6 |
| Pensacola Bay | 7,912 | 10,418 | 151,472 | 5.9 | 268.0 |
| Perdido Bay | 0 | 1,070 | 17,270 | 2.6 | 26.5 |
| Mobile Bay | 5,001 | 21,480 | 284,795 | 2.5 | 1,664.0 |
| Mississippi Sound | 29,652 | 66,932 | 434,454 | 3.0 | 715.0 |
| Deltaic Plain | 247 | 1,905,618 | 3,720,866 | 2.0 | 22,897.7 |
| Calcasieu Lake | 0 | 252,222 | 231,817 | 1.5 | 157.8 |
| Sabine Lake | 0 | 42,499 | 55,857 | 1.4 | 474.0 |
| Galveston Bay | 18,105 | 231,493 | 353,872 | 2.3 | 73.1 |
| Corpus Christi Bay | 12,753 | 45,017 | 109,839 | 1.2 | 24.5 |

[1] Acreage originally reported in hectares

states in Mexico demonstrated that the numbers of fish captured per unit area are positively related to river discharge. Deegan et al. (1986) state that while the mechanisms remain uncertain, there is little doubt that both estuarine area and fresh water input are related to fishery harvest.

Other studies have attempted to place a market value on wetland acreage based on the ecological links between wetlands and fishery landings or value. For example, Batie and Wilson (1978) attempted to correlate economic values from oyster production with acreage of adjacent wetlands and developed a bioeconomic model to define the contribution of wetlands to oyster production. Using this model, they calculated the marginal value products and capitalized values from oyster harvesting accruing to wetlands in seven Virginia counties. The values reflected in that study do not compare favorably with development values. For example, the discounted marginal value associated with residential development of Virginia Beach wetlands is approximately $17,650 per acre (1978 value). The model suggests a wetlands marginal value product of $47.11 (±$1,402.33) per acre if the wetlands are preserved for oyster production. When other possible values of the wetlands such as erosion control, wildfowl habitat, or fishery nursery are added, the amount may or may not exceed the average $17,650 per acre estimated marginal value of development. The important conclusions to be reached from this study are not the estimated values per se, but rather that refined estimates of value are possible if appropriate data are collected over a period of time on all parameters including wetlands acreage, property rights, fishing effort variables, biological variables, and prices.

Bell (1989) recently studied the importance of estuarine wetlands to commercial and recreational marine fisheries in Florida. Wetlands were found to be linked to approximately 80% of the total weight of fish landed by recreational fishermen, and nearly 92% of Florida's commercial landings (Bell, 1984). These data are probably applicable to the Gulf coast in general. According to Bell (1989), one approach to determining the tangible value of wetlands is to assign a monetary value to harvestable products and divide the market value of products by total wetland acreage to establish a dollar value per acre. Some disagree with dollar per acre values when used as the only evaluation, because this approach does not include true values of ecological and intangible products (e.g., aesthetics). The method also assumes that reduction in wetlands acreage directly affects the amount of a product that is harvested; however, harvest can be affected by other factors. The cost of harvesting the product is added to the value assigned when the total monetary value of the product is allocated to wetland value. Other methods assume that the actual dollar value of harvestable products includes the increasing dollar value generated as the product is processed, wholesaled, and retailed (Gosselink et al., 1974).

Bell (1989) employed marginal productivity theory in estimating the value of estuarine wetlands. This approach utilizes population dynamics, as employed by ecologists and fishery biologists, and marginal analysis, as used by economists. The theory is based upon a direct linkage between wetlands and marine fishery catch. The marginal product of an acre of estuarine wetland, holding all other factors constant, is valued at how much the public is willing to pay for the fishery product. Bell estimates 84% of the value of commercial fishery landings (by weight) on Florida's east coast and 95% of the value on Florida's Gulf coast are estuarine dependent. The degree to which they are dependent is established by estimation of a marine fisheries product function that includes fishing effort and salt marsh or estuarine acreage. A production function was estimated for eight estuarine-dependent species: blue crab, stone crab, grouper, red drum, oyster, spiny lobster, shrimp, and black mullet. The results indicated a strong statistical correlation between catch, fishing effort, and salt marsh acreage. The marginal product model calculates the retail value of an acre of salt marsh to the commercial fisheries of Florida at $1,355 on the west coast and $8,811 on the east coast. The recreational estuarine-dependent species generated a value of $1,222 on the west coast and $8,073 on the east coast. Bell believes that the marginal productivity approach is a possible method to integrate and summarize economic and biologic principles toward a common goal of wetland valuation. Nontangible wetland products were not included in this analysis.

Gosselink et al. (1974) estimated the value of unaltered wetlands to be as high as $82,000 per acre (1974 dollars) using a "life-support" valuation. These scientists calculated the economic value of the products of wetlands including the market value of estuarine-dependent species as well as sales of fur-bearing animals. Criticisms of this technique include: 1) the methodology assumes that marketable fish harvest is directly linked to wetland acreage, whereas fish catch also depends on biomass and fishing effort; 2) the methodology implies that all wetland acreage is equally productive; 3) value gained or lost cannot be approximately measured by comparison to total market; and 4) the whole market value of the fisheries is attributed to the wetlands. This method also assumed that value of capital and effort of the fishing industry was zero.

Odum (1977) proposed that the value of wetlands can be determined by establishing energy units as the common denominators of economic and environmental exchanges. The general formula is as follows:

$$\text{\$/acre/year} = \text{kcal/acre/year} \times \frac{\text{Gross National Product}}{\text{National Energy Consumption}}$$

Critics maintain that this method assumes that all goods and services, as well as inputs such as labor, machinery, and raw materials, are transformed energy. In addition, this method does not consider consumer demand and energy use.

Foster (1978) observed that the environmental benefits of wetland areas are largely in the public domain while alternative uses for wetlands are primarily for private benefit only. This fact has led to widespread destruction of wetlands. While society has exerted some governmental control to maintain the public benefits of wetland resources, many difficulties are encountered in calculating the social value of wetland benefits due to the nonmarket, intangible nature of many benefits. All benefits derived from wetlands cannot be measured in the same way. Decisions affecting public benefit are made daily by private and governmental bodies. These decisions result in giving up some benefits to secure others. The private decision-maker, in general, has little difficulty making the decisions, as intangible wetland benefits are perceived to have little value, while development may be of considerable value. On the other hand, the public decision-maker must weigh the social values of wetland benefits against their loss (Foster, 1978).

If more information were available, it would be easier to make these decisions. However, in most circumstances regulatory decisions must be made in a timely manner regardless of information gaps; decisions of this nature are made daily. In measuring the social value of wetlands, several of their characteristics must be considered (Foster, 1978):

1. The benefits of wetlands vary from place to place.
2. The level of benefits also vary.
3. The intangibility of some wetland benefits cannot be measured by market demand.
4. The value of wetlands vary from place to place and time to time.

Given these assumptions, Foster suggests that it is possible for the loss of wetlands to be more beneficial than their preservation. If the policy is to preserve all wetlands, valuations of productivity become unnecessary. Foster (1978) concluded that the public condones the draining and filling of wetlands for real estate and agricultural uses and construction of canals for navigation purposes because the value of wetlands to commercial and recreational fisheries has not been fully appreciated.

Not all of the proposed evaluations of wetlands are based on their value to commercial and recreational fisheries. Farber (1987) estimated the value of coastal wetlands for protection of coastal communities from hurricane winds. The basis for this protection is that wetlands help to weaken storms and provide a vegetated buffer between the storm and populated areas. As a hurricane makes landfall, the central core weakens as it passes over flat coastal terrain because the core no longer is fueled by the ocean heat source. Also, it is possible for wetlands to diminish the inland tidal surge that results from a hurricane impacting the coastline because of the effects of marshlands on wave energy and the reservoir capacity of the intervening area. A simple model of damages suggests a structural equation in which wind velocity is a function of the distance of location inland, the distance from the path of the storm, the intensity of the storm at landfall, and the nature of the intervening terrain. The terrain variable would reflect topographical features such as wetland areas.

Farber used this model to estimate effects of wind damage of coastal Louisiana. The value of the loss of a one mile (1.6 km) depth of coastal wetlands along the Louisiana Gulf coast was estimated for each Louisiana parish. The total incremental annual damages from the loss of one mile (1.6 km) of wetlands along the 250 mi (403 km) coast was $63,676 per year based on 1988 costs. When the annual incremental damage is capitalized, the present value (in 1980 dollars) of increased wind damage is between $1.1 and $3.7 million. On a per-acre basis, this amounts to between $7 and $23 per acre. The current market value (1987) of Louisiana wetlands is approximately $200 per acre; this value is derived primarily from mineral and hunting rights.

Farber suggests that hurricane protection is only one of several functions that wetlands produce as "public goods." To achieve an estimate of the full value of wetlands, their value for wind protection must be added to other values such as flood protection, recreation, and fishery values. All these goods and services provided by wetlands must be examined and considered when decisions are made on costly projects to revegetate or otherwise protect wetlands. Also, the same concept applies when decisions are made to destroy wetlands through canal construction and other developments. Flood damage is probably more damaging to low-lying coastal areas than wind, and wetland areas may be more useful for flood protection than wind protection. That relationship remains to be quantified.

## Quality of Coastal Habitats

By virtue of their location in the coastal zone, coastal habitats are extremely vulnerable to natural and man-induced destructive forces. The effects of the destructive force can be categorized into reversible and irreversible impacts. Obviously, impacts which irreversibly alter the character or quality of a habitat are most destructive. The severity of an adverse impact is directly proportional to the length of time that it takes for the system to recover from the impact.

It is important to understand that ecosystems change in time, and attempts by man to stabilize inherently unstable systems are very expensive and ultimately may be doomed to failure. For example, the westward drift of barrier islands along the central portion of the Gulf Coast is inevitable, given prevailing currents and sediment transport systems. Man's activities can hasten or exacerbate loss of barrier island habitat by upsetting the equilibria which maintain those systems or control their patterned development. Thus, dredging a channel on the western margin of a barrier island may result in loss of island habitat by precluding island accretion. Likewise, dredging of channels through seagrass meadows or creating channels by propeller scouring can result in increased seagrass losses due to changes in hydrological patterns and increased erosion.

Naturally occurring physical and biological processes can adversely affect the quality of coastal wetland habitats, including subsidence, erosion, and rising sea level, phytoplankton blooms, and plant diseases. Common human activities which adversely impact wetlands, include construction of canals and channels, dredging, spoil disposal, impounding, draining, and filling.

The impacts of natural and man-induced activities on habitats of Mobile Bay, Alabama, were summarized in conceptual models by Watzin et al. (in preparation). Those models consist of a problem statement, a list of causes, and an evaluation of effects. This approach is useful since it produces a summary and evaluation of habitat issues. Sources of habitat degradation and effects for Gulf of Mexico habitats, including vegetated habitats, are summarized in this manner in Figures 3, 4, and 5. Although there are not enough quantitative data to rank the causes of habitat loss, changes in sedimentation patterns, water movement, subsidence, and dredging and filling activities are the causes of the majority of losses of all Gulf habitats. Structure and function of habitats are inextricably related. Forces which impact structure always detrimentally impact the ecological functions provided by that habitat (e.g., filling of wetlands eliminates fish foraging sites).

Subsidence or "sinking" is the natural fate of delta marshes which undergo cyclic periods of construction and deterioration. The construction period is initiated when a river changes course and sediment deposition begins. Natural subsidence occurs as a result of dewatering, compaction of

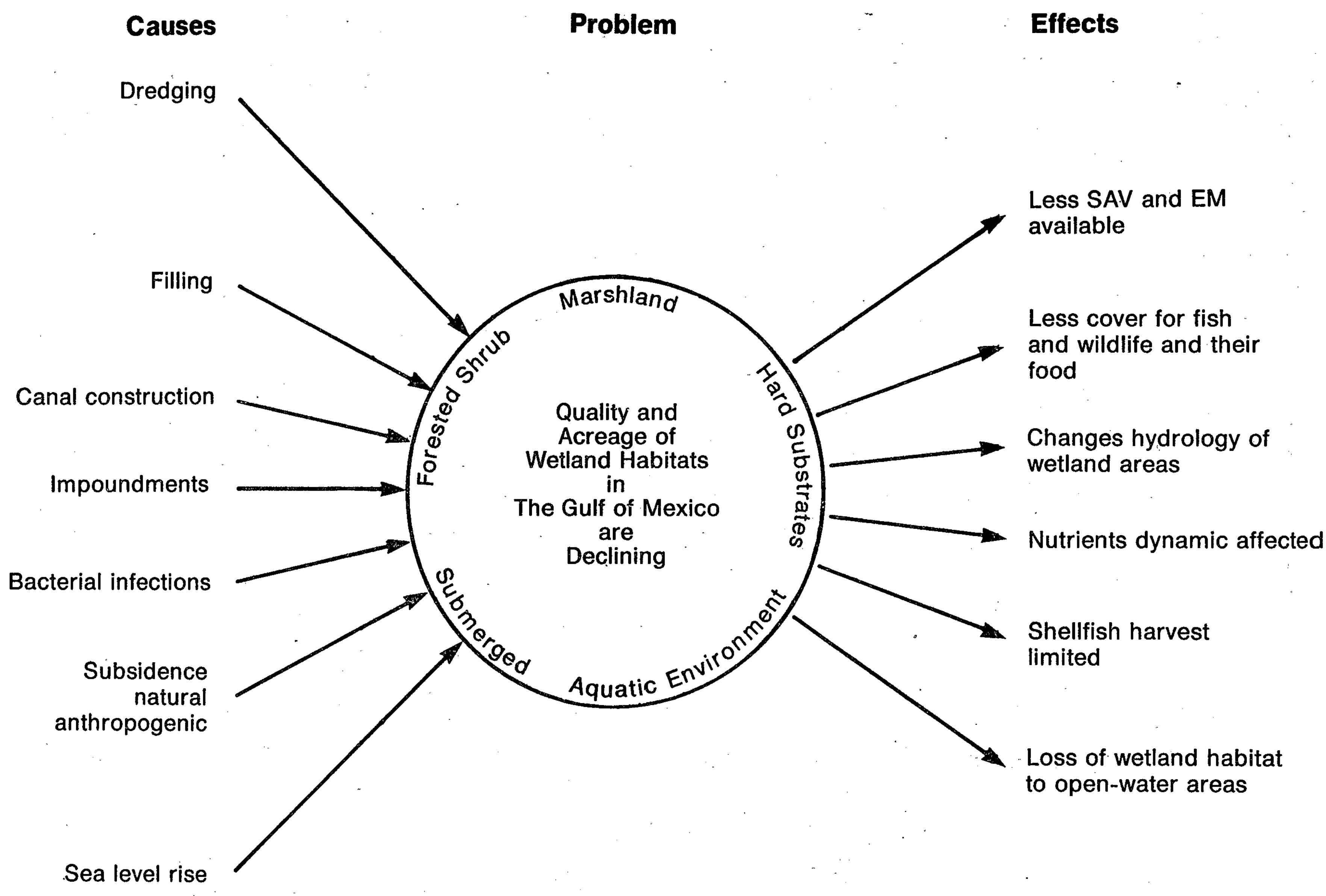


**Figure 3. Impacts on Quality and Acreage of Wetland Habitats in the Gulf of Mexico. (Modified from Watzin et al., in preparation)**

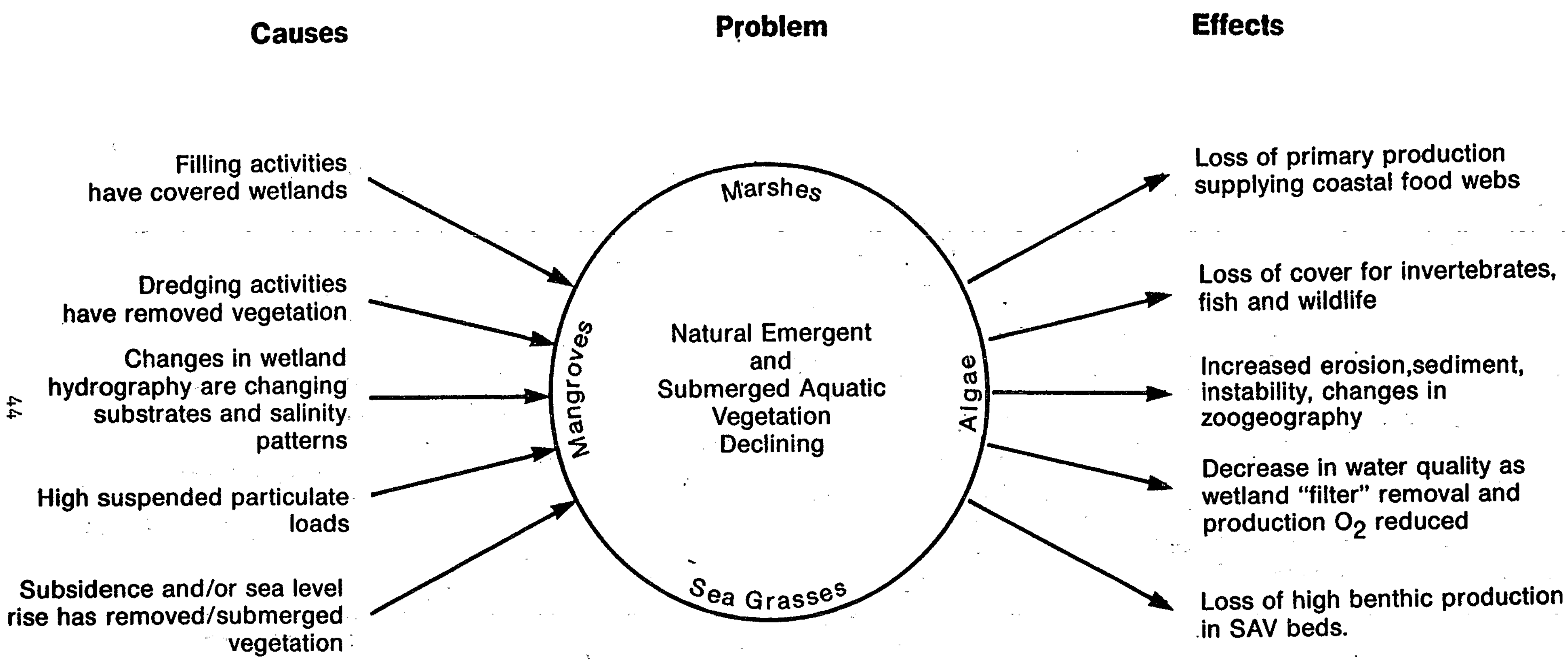


**Figure 4. Impacts on Natural Emerged and Submerged Aquatic Vegetation. (Modified from Watzin et al., in preparation)**

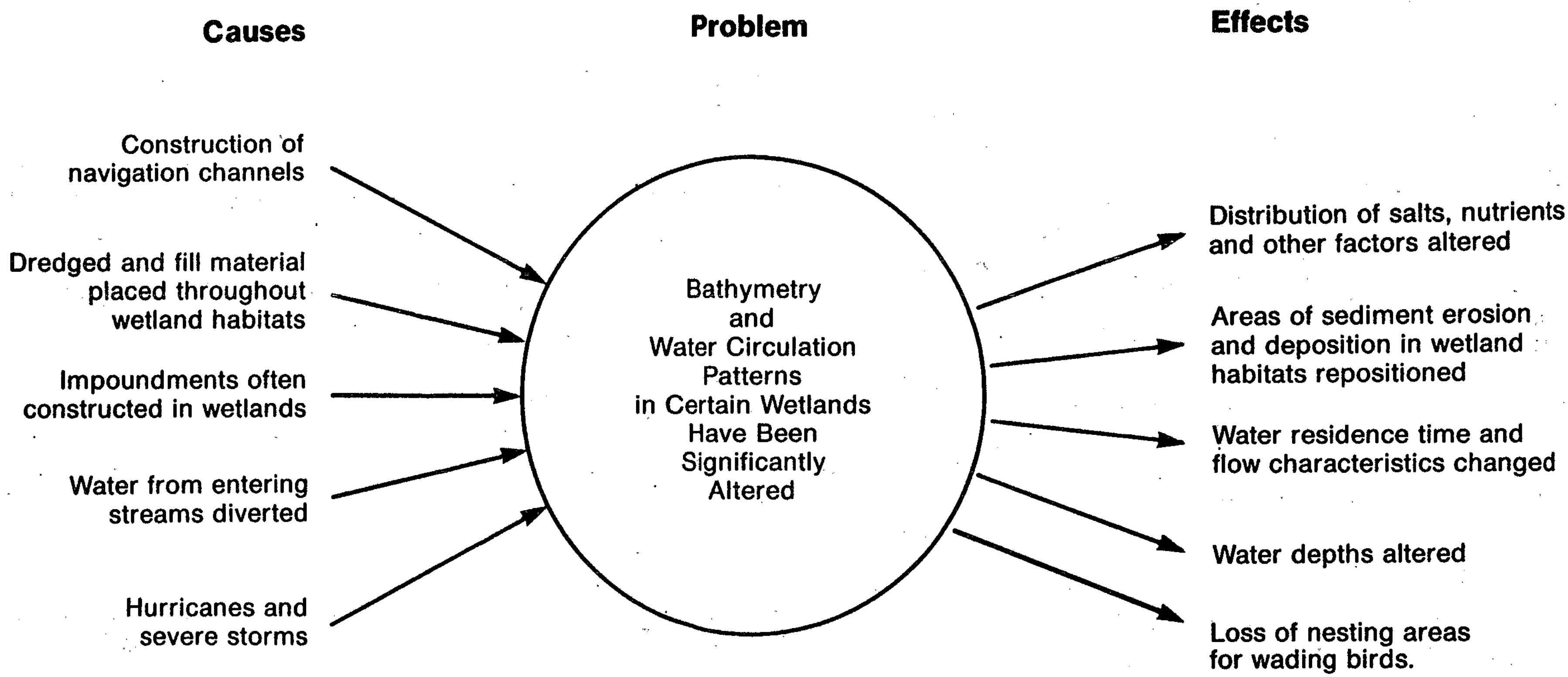


**Figure 5. Impacts on Bathymetry and Water Patterns on Wetlands. (Modified from Watzin et al., in preparation)**

sediments, and sinking due to weight of sedimentary deposits. During the construction period, deposition exceeds subsidence and the delta enlarges. As delta construction continues, the river begins to favor a shorter pathway and deposition is reduced while subsidence continues. Land levels may be maintained by an accumulation of organic materials and remobilization of mineral sediments. However, subsidence rates eventually exceed organic accumulation, which initiates the destruction phase which is characterized by the losses of large areas of wetlands (Leibowitz and Hill, in preparation).

Disruption of hydrologic and sediment patterns by man's activities or natural processes can accelerate this cycle. Swift currents and surges from hurricanes and storms can enhance the erosion of sediment and other substrate and result in deterioration of wetland habitats. Sea level rise can hasten the flooding of wetlands and result in more open water acreage.

River deltas along the Louisiana coast evolve through alternating periods of land building and land loss. The frequency and duration of these periods are determined by the balance between sediment supply and variations in relative sea level. Ongoing subsidence resulting from compaction and the sinking of Mississippi delta sediments is the major symptom of the relative sea-level rise. The construction and maintenance of dams and canals, especially navigation canals, is a major cause of changes in sedimentation patterns and wetland loss, especially in Louisiana (Johnson and Gosselink, 1982; Turner, 1987; Turner et al., 1982; Chabreck, 1982; Lindall et al., 1979; Baumann and Turner, 1990). Construction of canals contributes directly and indirectly to the disappearance of Louisiana's coastal wetlands (Johnson and Gosselink, 1982). Dredging results in the conversion of wetlands into open water. Also, as canals become older and are used by boats, the banks erode and the canals becomes wider, resulting in additional wetland loss. Increased open water from canal construction can result in changes of water flow patterns and subsequent saltwater intrusion. Saltwater intrusion related to canal construction is responsible for the loss of many acres of Louisiana coastal marshes.

Oil and gas extraction activities can result in detrimental impacts to onshore, nearshore, and offshore wetlands and other aquatic habitats. Onshore oil and gas activities which can affect habitats include construction of pipelines and support facilities; nearshore activities include construction and use of navigation channels. The detrimental effects of Outer Continental Shelf (OCS) activities on Louisiana coastal habitats are discussed by Turner and Cahoon (1987). While it is difficult to quantify a direct cause and effect relationship between OCS activities and habitat loss, it is noteworthy that the rapid wetlands loss in the central area of the Gulf coast is confined to the most concentrated development of onshore and offshore oil and gas recovery efforts. Over 40% of the total OCS lease blocks sold in the United States are in the central Gulf region. Dredge and fill activities have been responsible for the majority of conversions of wetland habitats to open water activities have been responsible for the majority of conversions of wetland habitats to open water habitats in this region (Turner and Cahoon, 1987). Turner and Cahoon discuss the importance of direct impact on wetland loss (habitat change), the significance of direct conversion of wetland habitats to open water and spoil sites by oil and gas activities, and the relevance of construction of navigation channels to continuing wetland losses. The extraction of oil and gas on the OCS has contributed approximately 4-5 percent of wetlands losses from pipeline canals and navigation channels crossing wetlands. Another 4-13 percent of wetlands loss was estimated to be due to indirect impacts from these activities.

Johnson and Gosselink (1982) quantified the effects of canal construction on wetland habitats. Their study areas included: oil field navigation channels on the Rockefeller Wildlife Refuge at Grand Chenier, Louisiana; the Southwestern Louisiana Canal in southern Lafourche Parish; and the Leeville oil fields near Leeville, Louisiana. Johnson and Gosselink (1982) observed that newly dredged canals were constructed wider than the widths specified on permit applications and concluded that few, if any, regulatory observations were made during canal construction. Also, boat traffic greatly increased the permitted canal widths. Canals near navigation routes with increased traffic widened from 4.8 to 5.3 ft (1.46 to 1.63 m) per year faster than those located some distance from boat traffic. They demonstrated that once spoil banks eroded away, a dramatic increase in canal widening occurred.

Turner et al. (1982) also studied the effects of canals on south Louisiana wetlands. Historically, some canals which were constructed for oil and gas exploration have been filled, but many more have been abandoned after commercial use and are utilized recreationally. Canals are constructed in straight lines, whereas natural creeks in the delta are serpentine. The linear structure of canals results in major differences in water and sediment movement between canals and natural drainage systems. The direct and indirect effects of canal development have greatly accelerated the rate and geographic extent of wetland loss in Louisiana; wave attack and the deficit of sediment accretion compared to subsidence and sea level rise result in localized land loss. Some have postulated that canals probably caused at least a majority and perhaps as much as 90% of the present land loss in coastal Louisiana (Penland and Boyd, 1982).

Hopkinson and Day (1979) also found that two canals, Bayou Segnette and Barataria Waterway, have drastically modified the hydrology of the area. In the past, all water from the upper portion of Barataria Bay (mainly a fresh water area) passed through Lake Salvador. Now, only that portion originating from the Des Allemands area passes through. Thus, the canals are causing saltwater intrusion to occur farther north and stormwater runoff from the West Bank is flowing directly into Barataria Bay.

Lindall et al. (1979) concluded that construction and maintenance of navigation channels was probably the largest form of estuarine alteration in the southeast. Increased rates of canal construction are directly related to increases in human population. More than 4,400 mi (7,097 km) of navigation channels existed, were under construction, or were planned at the time of the Lindall study. Two years earlier (1977), Lindall and Solomon reported that an average of 151.5 million cubic yards of sediment were removed each year from existing channels and the total yearly disposal from Federal maintenance dredging was 200 million cubic yards. More recently, Mager (1990) documented that marsh management projects are a major cause of habitat alteration in the southeast. Typically, diked and managed marshes convert to open water.

Changes in habitat area in coastal Louisiana which occurred between 1955-1956 and 1978 are summarized in Table 14. There has been a dramatic decrease in the area of marsh and swamp and a concurrent increase in areas of open water, mudflats, canals, and spoil disposal sites.

**Table 14. Changes in Area by Habitat in Louisiana's Coastal Zone from 1955/56-1978 (from Bauman and Turner, 1990)[1]**

| Habitat Description | Area (acres) 1955/56 | Area (acres) 1978 | Changes from 1955/56-78 |
|---|---|---|---|
| Marsh | 3,125,825 | 2,494,030 | -631,795 |
| Swamp | 518,436 | 437,560 | -80,876 |
| Forest/Upland | 119,890 | 142,206 | +22,318 |
| Aquatic Grass Bed/Mud Flat | 16,173 | 66,193 | +50,020 |
| Canal and Spoil | 84,189 | 198,733 | +114,543 |
| Open Water | 4,881,211 | 5,344,219 | +462,910 |
| Urban/Agriculture | 455,400 | 523,476 | +68,076 |
| Beaches and Dunes | 11,757 | 7,613 | -4,144 |

[1] Acreage originally reported in hectares

Although acreage of habitat loss caused by oil and gas activities is significant, these losses only represent a small portion of total wetland losses experienced between 1955 and 1978 (Table 14). All sources of direct detrimental impacts to coastal wetlands accounted for an estimated 25.6% of the total net wetland loss within the Louisiana portion of the study area from 1955/56 to 1978. Of the total direct impacts of 182,619 ac (73,935 ha), OCS-related activities accounted for 28,636 to 33,682 ac (11,593 to 13,636 ha) of the wetland loss. Although this is a substantial areal loss, it represents only 4.0 to 4.7% of the total Louisiana wetland loss from 1955/56 to 1978, and 15.7 to 18.4% of direct impacts.

Direct impacts from OCS pipelines averaged 1.6 ac/mi (0.4 ha/km) and totalled 29,682 ac (12,017 ha). Direct impacts are variable and are related to construction technique, geologic region, habitat type, age and diameter of pipeline, and other factors that were not examined (Bauman and Turner, 1990).. The current guidelines published by Minerals Management Service on the impacts of pipelines on Gulf of Mexico coastal marshes estimate that construction of a single oil and gas pipeline destroys approximately 25 ac of wetlands per mile of pipeline (6.28 ha/km).

The direct impacts per unit length of navigation channels were found on the average to be approximately twenty times greater than pipelines. Navigation channels accounted for a minimum of 41,765 ac (16,909 ha) of habitat change. Of the total change, 33,643 ac (13,620 ha) represented the losses of wetland and beach habitats. The maximum amount of habitat change attributable to OCS activities was 7,126 ac (2,885 ha; 17%) of which 5,666 ac (2,293 ha; 16.8%) was from the loss of wetland and beach habitats. The impact of commercial traffic using navigational channels is due mostly to activities other than those associated with OCS activities. Of the total direct wetland loss resulting from navigation channels, 33,729 ac (13,655 ha; 81%) occurred in the Mississippi River Gulf Outlet, Calcasieu Ship Channel, and Beaumont Channel/Sabine Pass. All of these areas have very low OCS destination usage.

An example of the manner in which nature and human activities can adversely impact wetland habitats is presented by Chabreck (1982). He reported that there are four vegetative types that typically occur in bands generally paralleling the coastline: saline, brackish, intermediate, and fresh. Each band contains characteristic water salinity levels and plant communities. Human activities such as construction of levees, canal dredging, and stream channelization along with natural processes including subsidence and erosion have removed many natural tidewater barriers and reduced fresh water flow through the marshes. As a result, saltwater intrusion from the Gulf of Mexico has greatly increased and has resulted in alteration of vegetation types. Saline vegetation has increased while brackish and intermediate types have retreated inland. This has caused a drastic reduction in the acreage and diversity of freshwater vegetated communities.

The definitions and acreages of plant communities which are discussed in Chabreck's Louisiana study are:

**Saline Vegetative Type.** This type of vegetation borders the shoreline of the Gulf of Mexico and is subjected to tidal fluctuations. It forms a narrow band in the Chenier Plain but is quite extensive in the deltaic plain and occupies about 667,170 ac (270,000 ha). Water salinities average 18.0 ppt and soils have less organic content than those further inland. Species are salt-tolerant and dominated by *S. alterniflora* (cordgrass), *Distichlis spicata* (saltgrass), and *Juncus roemerianus* (black needlerush).

**Brackish Vegetative Type.** Although this type is further removed from the shoreline, it is still subject to tidal fluctuations. This is a major vegetative type and occupies about 1,284,400 ac (520,000 ha) in coastal Louisiana. Water salinities average 8.2 ppt and the soils contain more organics than more saline soils. The brackish vegetative type is dominated by *S. patens* (salt hay grass) and *D. spicata.*

**Intermediate Vegetative Type.** This vegetative type is found inland of the brackish vegetation and occupies an area of about 691,600 ac (280,000 ha). Some tidal influence is present and salinities average 3.3 ppt. Plant diversity in this zone is high. *S. patens* dominates this zone; other common plants include *Phragmites communis* (phragmites) and *Sagittaria falcata* (duck potato).

**Fresh Vegetative Type.** The fresh vegetative type occupies the zone inland from the intermediate type and south of the Prairie formation and Mississippi River alluvial plain. It encompasses an area of 1,309,100 ac (530,000 ha) and is normally free of tidal influence. This community supports the greatest diversity of plants, and organic matter in the soil can exceed 80%. Dominant plants include *Panicum hemitomon* (maidencane) and *Eleocharis* spp.

Changes in the location of saline and brackish vegetative types over a period of approximately 25 years were determined by comparing maps of 1949 with those in 1968. Saline types in the deltaic plain on the earlier map extended inland for an average of 5.8 mi (9.3 km) from the Gulf shoreline, while the 1968 map shows an intrusion of 7.9 mi (12.7 km) inland (an encroachment averaging 2.1 mi (3.4 km) over the 25-year period). In the period from 1968 to 1978, the fresh vegetation type was reduced by 6.8% while the saline type increased 8.9% (Chabreck, 1970).

Dredging and filling of wetland habitats has significantly altered the ecological balance of coastal ecosystems throughout the Gulf Coast. For example, one study demonstrated that dredging and filling for development in Florida resulted in the loss of an estimated 1,100 t (1,000 mt) of seagrass, 1,800 t (1,636 mt) of invertebrates, and 73 t (66 mt) of fishery products (Taylor and Solomon, 1968). Even though our knowledge and appreciation of coastal wetlands has increased substantially since the time of that study, the pressure to develop coastal habitats has not abated. Indeed, the numbers of people moving to the Sun Belt to live on or near the water has continued to increase (U.S. EPA, 1988c; Culliton et al., 1990). This has resulted in increased pressure to convert natural coastal habitats into residential areas, marines, and other facilities. Creation of waterfront development can adversely affect coastal habitats by directly obliterating (filling) coastal vegetated communities, redistributing sediments, interrupting the normal access of tides and currents, and replacing numerous natural zones of gradation (ecotones) with open water or developed upland habitats.

Quantification of current losses through dredge and fill activities along the Gulf Coast are summarized in a data base collected by the National Marine Fisheries Service's (NMFS) Habitat Conservation Program (Mager and Thayer, 1986; Mager and Keppner, 1987; and Mager and Ruebsamen, 1988, Mager, 1990). This data base summarizes the effectiveness of the U.S. Army COE regulation program in controlling coastal wetland habitat loss. The COE has the responsibility and authority to issue federal permits for dredging and filling activities in waters of the United States including wetlands. Other federal and state resource agencies, such as the U.S. Fish and Wildlife Service and U.S. Environmental Protection Agency, advise the COE on the potential environmental impacts of proposed projects. The NMFS provides comments and recommendations to the COE concerning the environmental impacts which could result with completion of a proposed action on the management and development of marine fisheries resources. In the NMFS Southeast Region (North Carolina to Texas including the Virgin Islands and Puerto Rico), most of commercial fishery landings and most of the recreational fish caught consist of species that inhabit estuaries at some part of their life cycle (Thayer and Astach, 1981; Lindall and Thayer, 1982; Mager and Thayer, 1986). Yet, there is continuing competition for multiple use of estuaries and marshes that result in the loss of the valuable habitat. Some modifications resulting in adverse effects on the habitat include physical modification, addition of biological and chemical pollutants, and alteration of fresh water inflow patterns (Lindall et al., 1979; Tiner, 1984).

A cooperative agreement between the NOAA and the U.S. Department of Army Corps of Engineers established pilot studies from 1985 to 1988 in the Southeast to evaluate the feasibility of establishing a national program for creating and restoring fishery habitat. The results of the pilot studies are summarized in Thayer et al. (1989) and Pullen and Thayer (1989) and a national restoration program was established in 1991. As part of the pilot study, NOAA (NMFS) designed and implemented a computerized system to track agency recommendations concerning permit requests for proposed habitat alteration (dredging and filling) in the Southeast. Data from 1981-

1987 were analyzed by Mager and Thayer (1986), Mager and Keppner (1987), and Mager and Ruebasem (1988) and are summarized in Table 15.

The average values associated with permit actions reflect projected losses due to direct removal, burial, impounding, and draining. However, the impact on habitats of point and non-point sources of pollutants associated with development are not addressed by this program; thus, losses of certain habitats, particularly submerged aquatic vegetation reported through the permit process, are conservative. Of the sample of 5,122 permit applications evaluated, approximately 40% of the acres proposed for dredging by applicants, 46% of the fill applications, and 2% of the drain applications were "not objected to" by NMFS. During this time period, NMFS recommended against issuance of Corps permits which would have resulted in the loss of 158,603 ac (64,212 ha) of coastal wetlands.

Dredging and filling activities not only result in creation of dry land from wetlands (U.S. Department of the Interior, 1974), they can result in creation of canals which pose other environmental threats to coastal ecosystems. Canal developments have been constructed throughout the Gulf coast by excavating sediments from shallow estuarine bottoms adjacent to emergent wetlands and pumping or placing this excavated material on marshlands or mangroves. Often, bulkheads are built to contain the excavated materials. Canals which exist between filled fingers of wetlands generally exhibit poor water quality due to poor circulation and nutrient loading from adjacent development (U.S. EPA, 1973). As homesites on canals are developed, the sandy soil is usually covered with grass that may be fertilized, watered, and treated with pesticides and herbicides. Runoff from lawns transports the nutrients and pesticides, as well as particulate material, to the poorly flushed canals (U.S. EPA, 1973). Oil, gasoline, polychlorinated biphenyls from tire wear, detergents, and other pollutants may reach the water in the canals. Water quality in canals and adjacent waters can be further reduced due to failed septic tank systems located along canal developments (U.S. EPA, 1982).

Creation of finger-filled canal projects not only can destroy wetlands through filling and creation of waters which may not conform to water quality standards, but it can also destroy estuarine benthic organisms and vegetation. Suspended material from dredging activities can reduce light penetration and retard the growth of submerged vegetation. Also, changes in the texture and particle size of excavated materials may inhibit recolonization by invertebrates (Johnson, 1971). Thus, this form of development can be disruptive to submerged and emergent habitats.

The indirect effects of urban development on habitats in Barataria Basin, Louisiana are discussed by Hopkinson and Day (1979), who evaluated changes that occurred at the upper estuary interface as a result of urban development. Their work suggests that processes occurring at the

TABLE 15. Proposed Projects and Acres of Habitat Involved in NMFS Habitat Conservation Efforts from 1981 through 1987 (from Mager and Thayer, 1986; Mager and Keppner, 1987; and Mager and Ruebsamen, 1988)

| State | No. of permit appli-cations | Acreage proposed by applicants | | | | Acreage NMFS accepted or did not object to | | | | Potential acreage conserved | | | | Mitigation recommended by NMFS[1] | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Dredge | Fill | Drain | Impound | Dredge | Fill | Drain | Impound | Dredge | Fill | Drain | Impound | Restore acreage | Generate acreage | Enhance |
| LA | 1,391 | 71,926.8 | 38,619.7 | 6,442.9 | 103,811.2 | 25,056.8 | 11,184.4 | 160.5 | 54,492.8 | 46,876 | 30,100.2 | 6,282.4 | 49,318.4 | 93,976 | 15,169.5 | 47,414.1 |
| TX | 882 | 5,751.9 | 8,339.2 | 0.1 | 10,793.1 | 2,795.8 | 867.8 | 0 | 989.4 | 2,956.1 | 7,471.4 | 0.1 | 9,803.7 | 5,007.3 | 1,993.1 | 43.2 |
| MS | 122 | 396.8 | 670.8 | 5 | 6.3 | 152.5 | 249.9 | 0 | 0 | 244.3 | 420.9 | 5.0 | 6.3 | 6.7 | 42.6 | 0 |
| AL | 244 | 2,978.0 | 20,732.6 | 0 | 2.6 | 2,694.5 | 20,092.8 | 0 | 0 | 283.5 | 639.7 | 0 | 2.6 | 151 | 99.8 | 0 |
| FL | 2,483 | 3,812.5 | 4,081.8 | 0.5 | 279.7 | 2,721.6 | 1,249.8 | 0.2 | 10.4 | 1,090.9 | 2,832.1 | 0.3 | 269.3 | 730.6 | 862.6 | 51.2 |
| Total | 5,122 | 84,866.0 | 72,444.1 | 6,448.5 | 114,892.9 | 33,421.2 | 33,644.7 | 160.7 | 55,492.6 | 51,450.8 | 41,464.3 | 6,287.8 | 59,400.3 | 99,871.6 | 18,167.6 | 47,508.5 |

[1] Restore: restoration of temporary alterations
Generate: creation of new wetlands
Enhance: primarily for marsh management areas or for minor activities to reestablish preproject patterns

uplands-estuary interface can have direct ecological effects, such as nutrient runoff and eutrophication. In addition, these processes can cause indirect effects from a distance, such as changes in hydrology. As previously discussed, hydrology changes may alter salinity distribution in an estuary and in coastal waters.

Nutrient addition to coastal waters can severely disrupt ecological equilibria. Hopkinson and Day (1979) studied three fresh to brackish water coastal lakes (Cataouatche, Salvador, and Little) for one year. Measurements of community production and metabolism, chlorophyll, and water column nitrogen and phosphorus were used to evaluate functional relationships between the uplands and the estuary. Community heterotrophy decreased from upland to the lower estuary and chlorophyll, nitrogen, and phosphorus concentrations were highest in the streams and lakes adjacent to upland areas. This study concluded that Lake Cataouatche and Lake Salvador serve as efficient nutrient processing traps. Lake Cataouatche is highly eutrophic and has experienced fish kills following storm runoff from the West Bank area of New Orleans. These fish kills indicate that the lake is losing its value as a prime nursery area for recreationally and commercially important fish species. Thus, excess nutrients have reduced the quality of this lake and presumably the surrounding wetlands.

One critical indicator of the quality of estuarine habitats is the quality of indigenous shellfish and the water in which they grow (Broutman and Leonard, 1988). Shellfish waters are classified for the commercial harvest of oysters, clams, and mussels in relation to public health concerns. Shellfish obtain nourishment by filtering large volumes of water and can accumulate (bioconcentrate) pollutants contained in the water. Humans can acquire these pollutants, which include bacteria and other pathogens, by eating poorly cooked or raw shellfish meats. Therefore, the harvest of shellfish is strictly regulated and is not permitted in waters containing high levels of fecal coliform bacteria. In 1985, Gulf of Mexico estuaries produced 60% of the weight (25,509,000 lb; 11,571,000 kg) valued at over 40% of the total U.S. shellfish catch. According to the NMFS (NOAA, 1987), over 30 million lb (13,600,000 kg) of oyster meats were landed in the Gulf of Mexico in 1986. The American oyster (*Crassostrea virginica*) is the major species harvested, with some commercial clam harvest occurring in south Florida.

Shellfish waters are classified by states and by the National Shellfish Sanitation Program (NSSP) guidelines. These classifications reflect levels of pollutants, pollution sources, and fecal coliform bacteria levels in surface waters (Broutman and Leonard, 1988). Waters are classified by states into four categories:

**Approved Waters:** Shellfish may be harvested for the direct marketing of shellfish at all times;

**Conditionally Approved Waters:** Waters do not meet the criteria for approved waters at all times, but may be harvested when criteria are met;

**Restricted Waters:** Shellfish may be harvested from restricted waters if subjected to a suitable purification process; or,

**Prohibited Water:** Harvest cannot occur at any time.

The NSSP standard for approved waters is a total coliform bacteria concentration of less than 70 Most Probable Number (MPN) per 100 ml, with no more than 10% of the samples exceeding 230 MPN per 100 ml. As the basis for closing waters to shellfish harvesting, most states, including all the Gulf of Mexico states, use a fecal coliform standard of 14 MPN per 100 ml, with no more than 10% of the samples exceeding 43 MPN per 100 ml.

Most of the oyster reefs which are harvested in the Gulf of Mexico are classified as "approved", "conditionally approved", or "conditional." The conditionally approved and conditional classification is most used in waters that are affected by nonpoint sources. The classification requires the development of a management plan that is explicit about the conditions for opening the waters for harvest. Approximately 30% of coastal Gulf waters which are suitable for raising shellfish are closed for harvest, and 58% are conditionally approved or prohibited (Table 16). A more recent analysis of shellfish harvest classification in Alabama (Perkins, 1991) demonstrates a similar trend: Approved- 73,919 ac (29,900 ha); Conditional- 193,774 ac (78,450 ha); Restricted- 84,313 ac (34,134 ha); Prohibited- 1,179 ac (477 ha); Unclassified- 17,452 ac (7,000 ha). An indication of the occurrence of temporary closures of approved waters in the Gulf of Mexico is given in Table 17.

**Table 16. Shellfish Harvest Classification by States in 1985 (Adapted from Broutman and Leonard, 1988)**

| | Area (Acres) | | | | |
|---|---|---|---|---|---|
| State | Approved | Approved Conditional | Conditional | Prohibited | Closures[1] |
| Florida | 95,092 | 153,595 | 138,622 | 199,212 | 17,452 |
| Alabama | 0 | 0 | 175,487 | 84,860 | 0 |
| Mississippi | 76,888 | 120,083 | 189,958 | 96,749 | 0 |
| Louisiana | 1,620,458 | 0 | 283,242 | 1,042,157 | 0 |
| Texas | 727,941 | 570,045 | 0 | 328,500 | 0 |
| Totals | 2,520,379 | 843,723 | 787,309 | 1,751,478 | 17,452 |
| Percentage of Total | 42 | 14 | 13 | 29 | 1 |

[1] For reasons other than a sanitary survey

Table 17. **Examples of Periods of Temporary Closures of Approved Waters in the Gulf of Mexico (from NOAA, 1988c.)**

| State and Estuary | Area | Dates of Closure |
|---|---|---|
| **Florida** | | |
| Tampa Bay | Cockroach Bay | Aug-84, Feb-83, Feb-80 |
| | Lower Tampa Bay | Nov-83, Aug-83 |
| Apalachee Bay | Wakulla Cty Waters | Dec-85, Nov-85, Apr-84, Mar-83, Feb-82, Jan-82, Feb-81, Nov-80, May-80, Mar-80, Feb-80, Nov-79 |
| St. Andrew Bay | East Bay | Dec-85, Apr-84, Mar-84 |
| | West Bay | Dec-85, Mar-84, Apr-83 |
| | All | Oct-85 |
| Choctawhatchee | All | Dec-85, Mar-84 |
| Pensacola Bay | All | Dec-85 |
| | Escambia & East Bay | Apr-84, Mar-84 |
| **Mississippi** | | |
| Mississippi Sound | All Mississippi Waters | Dec-87, Feb-83 |
| | W. Mississippi Snd. | Dec-82 |
| **Texas** | | |
| Galveston Bay | All | Nov-86, Mar-85, Nov-84, Oct-84, Jul-79, Apr-73, Mar-73 |
| | Galveston and Trinity Bays | Feb-87, Nov-86, Jul-81, Jun-81, Jun-76, Jan-74 |
| | East Bay | Jan-87, Nov-86, Jul-81, Jun-76, Jan-74 |
| | West Bay | Jan-83, Jun-81 |

(Continued)

Table 17. (Continued)

| State and Estuary | Area | Dates of Closure |
|---|---|---|
| Matagorda Bay | All | Mar-85, Nov-84 |
| | Lavaca, Cox, Keller Bays | Dec-85, Nov-85, Nov-84, Mar-84, Nov-83, Feb-83, Nov-82, Mar-82, Nov-81 |
| | Lavaca Bay | Feb-87, Jan-85 |
| | East Matagorda Bay | Feb-87, Jan-87, Nov-84 |
| | Tres Palacios, Carancahua Bay | Feb-87, Apr-85, Nov-84, Feb-83 |
| | Oyster and Powder-horn Lakes | Mar-85, Mar-82 |
| San Antonio Bay | All | Dec-86, Dec-85, Apr-85, Mar-85, Nov-81 |
| | Portion | Apr-85, Jan-79 |
| Aransas Bay | Copano Bay | Feb-87 |
| | Mission, Copano, Port Bays | Apr-85, Mar-85 |

The NSSP standards are based upon concentration of pathogens in sewage; however, these standards do not include biotoxins, *Vibrio* bacteria, or diseases that are not necessarily associated with contaminated waters. Therefore, waters closed to shellfish harvest based on fecal coliform count do not include waters closed to harvest due to biotoxins and *Vibrio*. In the Gulf of Mexico, the dinoflagellate *Ptychodiscus brevis* produces toxins that cause fish kills and neurotoxic shellfish poisoning from consumption of contaminated shellfish. Consumption of shellfish contaminated with *Vibrio* causes gastroenteritis; this malady resulted in seven deaths in 1987 (Broutman and Leonard,1988). Studies indicate no apparent correlation between coliform-bacteria levels and *Vibrio*.

One function of wetlands is to filter pollutants and nutrients from flood or tidal waters. Thus, the cumulative loss of wetlands may significantly impact the availability of approved shellfish waters. Sources of water quality impacts in coastal waters have been summarized by Broutman and Leonard (1988). They found that sources of fecal coliform pollution that contribute to the permanent or temporary closure of shellfish areas include: sewage treatment plants (STP) that discharge inadequately treated effluent or bypass raw sewage through an outfall pipe during overload periods; straight pipes where sewage is discharged directly from units not connected to collection systems; industrial discharges containing fecal coliform from seafood-processing plants, pulp and paper mills, or human sewage; septic systems that leach improperly treated materials to surface waters; raw sewage from boats; urban runoff from storm sewers, drainage ditches, or overland runoff from urban areas containing fecal materials; agricultural and feedlot runoffs carrying fecal coliform; and wildlife areas where runoff contains fecal coliform from waterfowl, rodents, rabbits, deer, etc.

The main sources of coliform contamination and pollution of Gulf estuarine waters were summarized by Broutman and Leonard (1988). Gulf coast estuaries predominantly affected by STP's and urban runoff include Caloosahatchee River, Tampa Bay, Pensacola Bay, Lakes Pontchartrain and Borgne, Brazos River, and Corpus Christi Bay. Estuaries impacted by combined urban and nonurban sources are St. Andrew Bay, Mississippi Sound, Galveston Bay, and Laguna Madre. Estuaries impacted by upstream sources are Apalachicola Bay, Mobile Bay, Mississippi Sound, Mississippi Delta, Atchafalaya and Vermilion Bays, and San Antonio Bay. Septic tank failure is the main source of pollution in Aransas Bay; septic and straight pipes in Chandeleur/Breton Sounds, Terrebonne/ Timbalier Bays, and Caillou Lake; septics and boating activities in Ten Thousand Islands and Charlotte Harbor; septics and wildlife in Apalachee and Choctawhatchee Bays; septics and agricultural runoff in Matagorda Bay. Wildlife sources were listed as the main source of contamination of the Suwannee River and agricultural runoff in Barataria Bay.

Coastal barrier islands and other natural structures are dynamic systems that provide a plethora of ecological habitats. Shoreline changes and wetland losses of the Coastal Barrier Resource System were studied by Watzin and Baumann (1988). Data from selected barrier islands presented in that study are summarized in Table 18. Under natural conditions, erosion along one shoreline of a barrier structure is usually compensated by gains in habitat along the opposite shoreline. This process results in a east to west drift of barrier islands along the northern Gulf coastline. However, the activities of man have disrupted this delicate balance in many areas. Catastrophic natural forces can also disrupt the balance that maintains barrier resources.

The general trend is recession of shoreline of all coastal barrier islands along the Gulf coast. Many of the observed changes that have taken place are due to erosion of the shoreline. There are many causes of increased rates of erosion and it is difficult to separate and quantify impacts from the various sources. Erosion is generally accelerated through human activities such as construction of coastal stabilization features, including jetties and groins, and results in an alteration of the dynamic equilibrium of coastal processes. Dredging interrupts the littoral drift system and traps sediment in areas where it is no longer available for beach nourishment. Nature can also have a detrimental effect when hurricanes and storms erode beaches and wash over dunes and marshes.

Recession of barrier structures can also result from reduction in the amount of sediment supplied to the coast due to damming of rivers and other factors. Also, sea level rise, due to melting of polar glaciers, has resulted in flooding of barrier habitats. Reduction of onshore transport of sediment from offshore sources is an additional cause of recession of barrier resources.

Table 18. **Areal Changes (in Acres) in Selected Coastal Barrier Resource Systems Units, ca. 1940's to 1982 (from Watzin and Baumann, 1988)**

| Unit Name | Land Area[a] 1982 | Land Area[b] Pre-1982 | Land Area Change | Percentage Land Area Change | Annual Percentage Land Area Change |
|---|---|---|---|---|---|
| Florida | | | | | |
| Cape San Blas | 3,002.43 | 2,915.70 (1943) | +86.73 | +2.97 | +0.08 |
| Moreno Point (Destin) | 3,147.90 | 3,267.81 (1955) | -119.91 | -3.66 | -0.13 |
| Alabama | | | | | |
| Mobile Point | 3,769.79[c] | 3,945.49[d] (1979) | -175.70 | -4.45 | -1.48 |
| | 3,945.49 (1979) | 3,969.65 (1956) | 0 | 0 (within error margin) | 0 |
| Mississippi | | | | | |
| Deer Island | 505.18 | 544.05 (1952) | -38.86 | -7.14 | -0.23 |
| Cat Island | 549.04 | 804.77 (1956) | -255.73 | -31.77 | -1.22 |
| Louisiana | | | | | |
| Isles Dernieres | 2,381.55 | 4,616.67 (1956) | -2,235.12 | -48.41 | -1.86 |
| Texas | | | | | |
| Bolivar Peninsula | 6,667.19 | 6,759.65 (1956) | -92.46 | -1.37 | -0.05 |
| Follets Island | 3,005.59[e] | 2,914.18 (1956) | +91.41 | +3.13 | +0.12 |
| Boca Chica | 2,519.6 | 2,580.97 (1948,1950) | -61.32 | -2.37 | -0.07 |

[a] Land area includes the following habitats: fastlands, wetlands, and developed land. Excluded habitats include: open water and inland open water. These numbers differ slightly from the total without water reported in earlier tables because different computational procedures were used.

[b] The numbers(s) in parentheses indicate the year of the earlier map base. Multiple dates indicate more than one quadrangle map was required to cover the entire CBRS unit and these different maps had different data dates.

[c] After Hurricane Frederick.

[d] Before Hurricane Frederick.

[e] Does not include extensive erosion caused by Hurricane Alicia (1983).

## Living Resources

Several publications give a broad overview of Gulf habitats. The living marine resources that occupy Gulf coastal habitats have been described in a comprehensive atlas of the coastal and oceanic resources (NOAA, 1985) which summarizes: 1) physical environments, 2) biotic environments, 3) living marine resources, 4) economic activities, 5) marine environmental quality, and 6) jurisdictions. Maps contained in the NOAA atlas are at a scale of 1:4,000,000. The distribution of forested and nonforested wetland, seagrass, and benthic algal communities are summarized on maps; however, acreages are not given in this large-scale, regional overview.

The occurrence and distribution of soft-bottom communities of the northwest Gulf of Mexico is contained in an atlas prepared by Minerals Management Service (1983b). Distribution patterns of the biota are presented. Species are listed and numbers of individuals, percentage of the total catch, and seasonal distributions of selected biota are presented.

The following is a summary of the acreages and functions of the major vegetated Gulf habitats. This analysis is not intended to be a comprehensive literature review, but rather as a summary of data pertinent to the status and trends of each habitat type. The major vegetated habitats, marshes, seagrasses, and mangroves, have several common features and functions. Thayer and Ustach (1981) found that regardless of community type, organic material exported from these systems is an important energy source to adjoining estuarine and coastal systems. Some technical questions remain concerning the relative importance of the various sources of organic matter in the nutrient and energy dynamics of estuaries (Nixon, 1980). However, it is well documented that marshes, mangroves, and seagrasses serve as feeding, reproductive, and nursery habitats for many species of aquatic organisms and their existence is critical to important Gulf fishery and recreational resources (Durako et al., 1985). A comparison of the relative importance of marsh and seagrass systems to fauna has been conducted by Orth et al. (1984) and Orth and Montfrans (1990). The relative importance of these habitats to selected species may vary regionally and latitudinally.

### Marshes

The total area of tidal marsh along the Gulf coast was estimated by Lindall and Saloman (1977) at 6 million ac ($2.43 \times 10^6$ ha), which represented 63% of the tidal marsh area in the United States. At the time of that study, Louisiana contained most of the Gulf's tidal marsh (64%), followed by Texas (19%), Florida (15%), Mississippi (1%), and Alabama (0.6%). The most recent estimate of Gulf wetland acreage is given by NOAA (1991) to be 3.8 million acres ($1.54 \times 10^6$ hectares) (Table 5). The NOAA report summarizes the distribution of Gulf tidal marsh habitat as follows: Louisiana 69%; Texas 17%; Florida 10%; Mississippi 2%; and Alabama 1%. The differences in acreage and percentages between these two studies probably represents a combination of delineation of habitat types, sampling differences, and actual changes in acreage.

*Spartina alterniflora* is the predominant halophyte of Atlantic coast marshes. However, the general environmental conditions of Gulf coast tidal marshes are quite different from tidal marshes found along the Atlantic coast. Also, these conditions vary geographically along the Gulf coast. The northeastern Gulf marshes are characterized by irregular flooding, low energy wave and tidal action, and long periods of exposure (Stout, 1984). This area is also characterized by large discharges of fresh water and raised marsh elevations (Crance, 1971; Eleuterius, 1973). These environmental conditions are thought to be responsible for the predominance of *J. roemerianus*, in salt marshes of the northeastern Gulf of Mexico. Kruczynski (1978) estimated that approximately 28% of intertidal habitats along the entire Gulf coast of Florida are dominated by *J. roemerianus*, and that species was the dominant herbaceous halophyte from Florida Bay to Pensacola, Florida. Eleuterius (1976) observed that *J. roemerianus* was the dominant plant in 92% of the tidal marsh acreage in Mississippi and 52% of tidal marsh acreage of Alabama. Approximately 8% of the U.S. tidal marsh area is found in Florida, Alabama and Mississippi (627,249 acres; 253,947 hectares) and much of this acreage is vegetated by *J. roemerianus* (Stout, 1984).

The zonation of Gulf tidal marsh plant species within a marsh is dependent upon several ecological factors including salinity, tidal range, and soils (Kurz and Wagner, 1957). Thayer and Ustach (1981) found that *S. alterniflora* occurs from mean sea level up to the level of the highest predicted tide, and that *J. roemerianus* occurs landward from the zone occupied by *Spartina. Distichlis spicata* and *Spartina patens* occur above the mean high tide line. *Spartina cynosuroides* is also found in low tide areas of many Gulf coast marshes (Eleuterius, 1976). *Spartina patens* and *S. alterniflora* are the dominate species in tidal marshes of Louisiana, where marshes are dominated by *S. alterniflora* at low elevations, and *Salicornia* spp. and *Juncus* at higher elevations. These species are dominant in approximately 30% of the total Louisiana marsh area (Perret et al., 1971). The acreage and species composition of the four major plant communities as described by Chabreck (1982) are discussed in Quality of Coastal Habitats.

Durako et al. (1985) reviewed the structure and function of tidal marshes of Florida. Geologically, Florida marshes are a transitional community between mangroves and fresh water communities. Grasses (Graminae), sedges (Cyperaceae), and rushes (Juncaceae) are the plant groups most frequently found in Florida salt marshes. *S. alterniflora* is found in high salinities and *S. patens* is more common at lower salinities. *Cladium jamaicense* (sawgrass) is also found in low-salinity areas. *J. roemerianus* is found in association with mangroves south of Homestead and predominates low energy tidal marshes from Tarpon Springs to Pensacola Bay (the Florida panhandle). There is usually a fringe of *S. alterniflora* along tidal creeks and the waterward margin of *Juncus* marshes.

Intertidal areas of Texas marshes present harsher conditions for development of tidal marsh systems compared to other parts of the Gulf. Pulich (1990) reviewed the occurrence of marsh plant species and the environmental regimes affecting their production in Texas estuaries. A gradient in marsh types results from the decreasing precipitation gradient occurring from Sabine Lake southward to Corpus Christi and Laguna Madre. The corresponding decrease in fresh water inflows, combined with low tidal range, produces marsh vegetation adapted to moisture deficits, high groundwater salinities, and low inputs of nutrients due to infrequent freshwater discharges. Along the northern Texas coast, from Sabine Lake to Galveston Bay, extensive marshes are dominated by *S. alterniflora* or *S. patens-D. spicata.* From Matagorda Bay south to Aransas Bay, limited amounts of *S. alterniflora* occur in fringing bands near the water's edge, while extensive interior marshes of *Salicornia-Distichlis* occur at slightly higher elevations. From Corpus Christi Bay south to lower Laguna Madre and the Rio Grande Delta, both fringe and interior marshes are dominated by succulent halophytes such as *Batis maritima* (saltwort), *Salicornia vinginicus* (glasswort), and *Borrichia frutescens* (sea-oxeye). Changes in distribution and abundance of marsh indicator species were attributed to moisture and nutrient requirements, rather than salinity tolerances.

Although tidal flats and sand flats are not vegetated, these intertidal areas constitute a distinct habitat and major resource of the middle and lower Texas coastal zone. Because of low tidal range and arid conditions, hypersaline flats develop instead of marsh wetlands. These areas are covered by algae mats (diatom and cyanobacterial mats) and are very productive. Important functions of these areas are the export of nutrients and organic matter to other estuarine habitats and feeding habitats for foraging fish and decopods (Pulich and Rabalais, 1986; Pulich and Scalan, 1989). Because of their superficial "wasteland" appearance, pressure to develop and destroy this habitat has persisted.

Nationally, most of the research on the ecology of tidal marshlands has been confined to systems dominated by *Spartina* spp. The biogeochemical cycling of *Spartina* marshes may not be applicable to *Juncus*, *Salicornia*, or *Batis* marshes. The life cycle, metabolic pathways, and nutritive value of these halophytes are different (Pulich, 1990; Kruczynski et al., 1978a,b). The primary production of *Juncus* marshes on the west coast of Florida is similar to the productivity of some east coast marshes, but the standing crops and morphological features of *Juncus* differ within a marsh depending upon soils, tidal flooding, and elevation. Based upon these factors, Kruczynski et al. (1978a) divided *Juncus* marshes into lower, middle, and upper marsh zones.

The primary ecological functions of salt marshes include primary productivity, detrital export, nutrient export, sediment trapping, pollutant removal, and feeding grounds and protected habitat for juvenile fish and shellfish. Durako et al. (1985) reported that over 90% of net primary productivity of a salt marsh may form plant detritus. The diverse microbial and algal communities of tidal marshes contribute to the nutritive value of tidal marsh detritus, and the rate of detrital formation is dependent upon flooding frequency within a marsh (Kruczynski et al., 1978; Stout, 1984). Animal production is very high at the marsh-estuarine interface due to the abundant food sources found there; plant detritus forms the base of the food chain for many marsh and estuarine species (Odum and de la Cruz, 1967; Stout, 1984).

There is some scientific disagreement over the extent to which nutrients, including those derived from salt marsh detritus, are exported from marsh areas and utilized by coastal fisheries (Nixon, 1980). While the balance of energy sources driving these dynamic systems has not yet been fully understood, the nursery role that marshes play for many species of commercial and recreational importance is well known (Hackney, 1978). More than 1,100 species of vertebrates and invertebrates depend upon salt marsh-estuarine habitats for at least one stage of their life cycle (Durako et al., 1985). Production of commercial and sports fisheries depends upon the ecological viability of tidal marsh habitats. Distribution and abundance of juvenile fishes in tidal marshes are dependent upon salinity, food availability, water quality and depth, bottom type, and vegetative cover (Durako et al., 1985; Subrahmanyam and Coultas, 1980).

Estimates of the numbers of commercial and recreational fish that spend at least a portion of their history in estuarine habitats vary (McHugh, 1976; Lindall and Saloman, 1977, Thayer and Ustach, 1981; and Comp and Seaman, 1985). However, most authors agree that species which make up in excess of 90% of the Gulf's commercial or recreational catch spend a portion of their life cycle in tidal creeks and marshes. Several species, such as penaeid shrimp and blue crab, may spend the majority of their life cycle in wetlands and shallow estuarine habitats.

A diverse fauna has adapted to the unique environment of tidal marshes. Zooplankton is composed chiefly of the larvae of fiddler crabs and other decapods; meiofauna consist primarily of nematodes and harpacticoid copepods; and macroinvertebrates include crustaceans (especially crabs), molluscs, annelids, and insects. Several crustaceans such as grass shrimp, penaeid shrimp, blue crabs, as well as resident and migratory fishes, are seasonally abundant in marsh creeks. Birds are one of the larger carnivorous groups in this system and can bioconcentrate persistent pollutants which accumulate through the food chain (Stout, 1984).

Subrahmanyam et al. (1976) observed 55 species of benthic invertebrates in *Juncus* marshes of north Florida. Zonation of benthic fauna was correlated with soil type, flooding frequency, and plant zonation. While the feeding strategies among this diverse group vary, detritivores were the most common group. Detritivores in turn provide food for many commercially important species which invade tidal creeks and marshes at high tide (Subrahmanyam and Coultas, 1980) Minello et al. (1986) demonstrated that waterways into vegetated marshlands are essential for their maximum use by fisheries organisms.

Marshlands also provide habitat for many species of waterfowl and mammals. Of the five furbearing species economically important in the south, the muskrat was the most abundant in three of the four vegetative types studied by Palmisano (1973) (Table 19). The fur industry has changed dramatically since the early 1970's, and these data may not be representative of the current conditions.

**Table 19. Estimated Fur Catch per 1,000 Acres of Coastal Marsh (from Palmisano, 1973)**

| | Vegetative Type | | | | | |
|---|---|---|---|---|---|---|
| | Brackish | | Intermediate | | Fresh | |
| Species | Mean | Maximum | Mean | Maximum | Mean | Maximum |
| Muskrat | 84.4 | 6,477.7 | 97.5 | 513.9 | 78.5 | 646.8 |
| Nutria | 86.4 | 191.1 | 284.9 | 499.6 | 512.7 | 884.4 |
| Mink | 1.1 | 12.8 | .9 | 11.9 | 2.1 | 14.2 |
| Raccoon | [b] | 15.6 | [b] | 6.3 | [b] | 31.0 |
| Otter | .2 | .7 | .4 | 1.3 | .5 | 1.3 |

[a] Mean values are determined from recent records. Maximum values are an average of long-term maximum-catch figures.

[b] Inadequate records.

Mississippi tidal marshes were sampled during 1968-1969 by Eleuterius (1972) to determine species composition, area, zonation, production, and regulating factors. That study area covered three of the four major estuarine systems in Mississippi: the Pascagoula River system, Biloxi Bay, and Bay St. Louis. The total area of Mississippi tidal marshes was 66,931 ac (27,098 ha).

In addition to salinity-induced zonation along Mississippi estuaries, there was also a zonation from low marsh to uplands. Of the total 66,931 ac (27,098 ha) of marshlands, 63,982 ac (25,904 ha) were mainland salt marsh, 823 ac (333 ha) were freshwater marsh, and 2,126 ac (861 ha) of salt marsh were found on offshore barrier islands. *J. roemerianus* dominated 96% (61,398 ac; 24,857 ha) of the mainland salt marsh. *S. alterniflora* dominated approximately 30% (2,028 ac; 821 ha), *S. patens* 0.7% (460 ac; 186 ha), and *Scirpus olneyi* 0.1% (96 ac; 39 ha) (Eleuterius, 1972).

Three hundred species of vascular plants were identified in Eleuterius' survey. Eleuterius (1972) delineated two major divisions of tidal marshes: the lower region (saline and brackish) and upper region (intermediate and freshwater). The lower region exhibited higher and more stable salinities. The upper region exhibited lower salinities with greater fluctuations in salinity and greater diversity of plant species. High plant diversity was the result of changes of freshwater input. Most lower marshes were dominated by *J. roemerianus* with no definite sharp delineation between zones within the marsh. *S. alterniflora* was predominant along creeks in saline marshes. There was a reduction of *S. alterniflora* and increased density of *J. roemerianus* in the lower brackish marsh.

Seagrasses

Seagrass meadows cover approximately 7,440 $mi^2$ (19,275 $km^2$) of the Gulf of Mexico, including Mexico and Cuba. Seagrasses grow in shallow, clear waters in protected estuaries and nearshore waters (e.g. Florida Big Bend Area). Iverson and Bittaker (1986) and Orth and Montfrans (1990) estimated total seagrass coverage in Gulf states to be 2.47 million ac (1 million ha). Florida has the most acreage of seagrasses among the Gulf states-2.2 million ac (890,000 ha). Acreage in other Gulf states is: Alabama-30,381 ac (12,300 ha); Mississippi-4,940 ac (2,000 ha); Louisiana-10,127 ac (4,100 ha); Texas-169,195 ac (68,500 ha).

Areal coverage of selected seagrass beds is given in Table 20. There are approximately 800,000 ac (323,887 ha) of sea grasses within Gulf estuaries. Approximately 95 percent estuarine acreage is in Florida and Texas, where seagrasses occupy approximately 20 percent of bay bottoms (Thayer and Ustuach, 1981). Although often considered continuous around the entire periphery of the Gulf, a combination of low salinity and high turbidity results in only narrow bands or scattered patches of seagrass communities, mostly in bays, from Louisiana to Copano-Aransas, Texas. Discontinuous nearshore beds are found from Apalachicola, Florida, to Brownsville, Texas. Beds are particularly well developed around the Chandeleur and Breton Islands, Louisiana, and in Laguna Madre, Texas.

Table 20. Areal Coverage (Acres) of Selected Seagrass Beds.

| Location | Size | Reference |
|---|---|---|
| <u>Gulf of Mexico</u> | | |
| Florida embayments | 157,403 | McNulty et al. (1972) |
| Big Bend area | 741,300 | Iverson and Bittaker (1986) |
| Outer Florida Bay | 716,590 | Iverson and Bittaker (1986) |
| Inner Florida Bay | 624,000 | Zieman et al. (1989) |
| Florida Keys Reef Tract | 197,680 | Enos and Perkins (1977) |
| Alabama coast | 30,393 | Minerals Management Service (1983a) |
| Mississippi coast | 4,942 | Eleuterius and Miller (1976) |
| Louisiana coast | 10,131 | Minerals Management Service (1983a) |
| Texas lagoons | 169,264 | Bureau of Economic Geology (1980) |

Five species of seagrasses occur in the Gulf: *Thalassia testudinum* (turtle grass), *Halodule wrightii* (shoal grass), *Syringodium filiforme* (manatee grass), *Halophila engelmannii*, and *Halophila decipiens* (Zieman and Zieman, 1989). In addition, *Ruppia maritima* (widgeon grass), which is generally not considered as a true seagrass, is commonly reported in coastal zones of all Gulf coast states.

Seagrasses display vertical zonation. *Halodule* can occur into the low intertidal zone. *T. testudinum, S. filiforme*, and *Halophila* are found below the mean low tidal level. While *T. testudinum* is the most abundant Gulf submergent, *Halodule* is dominant in Mississippi and Alabama, and *R. maritima* is dominant in some areas of coastal Louisiana. Light, salinity, temperature, substrate type, and currents are locally important factors that affect distributional patterns (Ferguson et al., 1981).

In Texas, *Halodule* is the dominant species along the northern and middle coast, south to Aransas Bay. *Halodule* and *Syringodium* are most abundant in lower Laguna Madre. *Halodule* dominates in upper Laguna Madre. Annual changes between *Halodule* and *Ruppia* populations often occur at northern and middle Texas coastal locations (Pulich and White, 1990).

Eleuterius (1977) reviewed four local species of seagrasses of Mississippi, including information on productivity and local distribution. In that report, Eleuterius summarized most of the existing literature on the following species: *T. testudinum, H. wrightii, S. filiforme*, and *H. engelmannii*. All are found in estuarine beds north of the barrier islands off the Mississippi coast. These four species are considered to be primarily tropical flora, and the Mississippi coast represents their northern limit. The productivity for *T. testudinum* for Mississippi was determined to be at 1,848.5 gm dry weight/$m^2$/yr, and yearly production is about twice the standing crop.

The effects of a large-scale environmental perturbation, Hurricane Camille in 1969, on seagrasses and macrophytic algae in Mississippi waters were investigated by Eleuterius and Miller (1976). They concluded that considerably smaller areas of coastal bottoms were vegetated with seagrasses in the early 1970's compared to pre-1969 surveys. Physical disturbance of bay bottoms due to heavy wave action was thought to have destroyed large areas of seagrass beds. Approximately 20,000 ac (8,097 ha) of submerged vegetation in Mississippi Sound were located and mapped in 1969. Approximately 11,676 ac (4,727 ha) were lost as a result of Hurricane Camille. Losses due to low salinity in the winters of 1971 through 1975 reduced the remaining acreage to 4,866 ac (1,970 ha).

Zieman and Zieman (1989) discussed the distribution, abundance, and productivity of dominant species in seagrass systems on the west coast of Florida, from Florida Bay to Apalachicola Bay. These seagrass beds were dominated by three species, *T. testudinum, S. filiforme*, and *H. wrightii.*

Seagrasses occurred both on the shallow, zero-energy continental shelf and in inshore bays and estuaries. The largest seagrass meadow is located within the boundaries of the Everglades National Park in Florida Bay and is 1,359,050 ac (5,500 $km^2$). The second largest is located between Tarpon Springs and St. Marks and is 741,300 ac (3,000 $km^2$). Tampa Bay once had 76,601 ac (31,000 ha) of seagrass meadows, but as of 1982, the grassbeds have declined to approximately 20,000 ac (8,097 ha) (Lewis et al., 1985; Lewis et al., 1991). Zieman and Zieman (1989) observed that grass meadows in Florida estuaries were noticeably more stressed and impacted by human activities than the more pristine nearshore beds. They observed that seagrasses within west Florida estuaries must rank alongside the seagrasses of Chesapeake Bay as some of the most devastated and degraded in the entire country. Urban development and dredging and filling are the major threats to seagrass beds in this region. Motor boat propeller scouring is also a major threat to seagrass meadows in shallow waters.

Zieman and Zieman (1989) concluded that the importance of the loss of seagrasses to both the ecology and economy of Florida are far out of proportion to the total acreage lost due to the critical nature of estuarine seagrass meadows as nursing areas. While measures must be taken to ensure the continued productivity of the nearshore beds, it is most critical that the water quality degradation that has caused extensive losses of estuarine grass beds be arrested and reserved. Kenworthy and Haunerdt (1991) summarized the dependency of seagrasses on adequate light and water quality conditions and presented technically sound arguments for revising water quality standards and initiating monitoring programs to protect and restore this dwindling resource. The major man-induced causes of seagrass declines worldwide are reviewed by Thayer et al. (1975) and Shepherd et al. (1989). They list the main reasons for the "cultural" decline of seagrass meadows to be increases in turbidity and sediment mobilization resulting in light reduction, increased epiphytism due to nutrient enrichment, and increased grazing due to community imbalances. Once seagrass meadows are totally destroyed, they are likely to remain lost forever, along with the myriad of organisms that they feed and shelter.

The distribution of seagrasses on the broad, shallow Continental Shelf in the Florida Big Bend area was also studied by the MMS (1985) which mapped and inventoried seagrass beds using a combination of aerial photography (remote sensing) and shipboard/diver observations. The seaward extent of major seagrass meadows was determined, and major benthic habitats were classified and delineated. These mapping activities delineated 575,479 ac (232,987 ha) of dense seagrasses, 1,230,643 ac (498,236 ha) of sparse seagrasses, and 691,195 ac (279,836 ha) of patchy seagrasses between Ochlockonee Bay and Tarpon Springs.

Three major species associations were found in the MMS (1985) study. An inner or nearshore association consisted of *T. testudinum*, *S. filiforme*, and *H. wrightii* that occurred in waters less than 29.5 ft (9 m). Seaward of that group, but still in waters less than 29.5 ft (9 m) was a community vegetated by five seagrass species: turtle grass, manatee grass, shoal grass, *H. decipiens*, and *H. engelmannii*. Farther offshore, between the 32.8 and 65.6 ft (10- and 20-m) depth, a mixed macroalgal/seagrass assemblage, in which the two *Halophila* species were the only seagrasses present, was common. One of the unique aspects of the seagrass beds in the Big Bend area is the extended nature of the deeper fringing zone of seagrasses dominated by *H. decipiens* and *H. englemannii* and macroalgae. The macroalgae account for 21% of total blade density in this location. These macroalgae communities have been relatively little studied, but are thought to provide habitat and a food source for many species of the marine ecosystem.

Iverson and Bittaker (1986) investigated seagrass species composition, biomass, and areal coverage on the seabottom along the West Florida Continental Shelf. They observed that *T. testudinum, S. filiforme*, and *H. wrightii* form two large seagrass meadows along the northwest and southern coasts of Florida. They found approximately 741,000 ac (3,000 $km^2$) of seagrasses in the Big Bend area and 1,358,500 ac (5,500 $km^2$) in Florida Bay and concluded that the cross-shelf limits of these two major seagrass beds are controlled nearshore by increased water turbidity and lower salinity and offshore by light penetration to depths receiving 10% or more of sea surface photosynthetically active radiation. Similar to the findings of the MMS study (1985), Iverson and Bittaker observed that *T. testudinum* and *S. filiforme* were most abundant in shallow depths and *H. wrightii* grew in shallow waters near the inner edges of the beds, while *H. decipiens* and *H. wrightii* grew in deeper water outside the beds. *Ruppia maritima* grew chiefly in brackish water near the mouths of rivers.

Many studies have enumerated the fauna inhabiting seagrass meadows (Zieman, 1982; Zieman and Zieman, 1989; Lewis et al., 1985a; Sheridan and Livingston; 1983; and others). For example, Sheridan and Livingston (1983) quantified the infauna and epifauna inhabiting a *H. wrightii* meadow in Apalachicola Bay, Florida. Fifty-eight infaunal species were recorded, an average of 35 species per monthly sample. Maximum faunal abundance was 104,338 organisms per $m^2$ in April. Sixteen species accounted for 84% of the total numbers and 80% of the total biomass over the study period. Numerical dominants were *Hargeria rapax, Heteromastus filiformis, Ampelisca vadorum, Aricidea fragilis*, and oligochaetes. Biomass dominants were *Tagelus plebeius, Neritina reclivata, Ensis minor*, and *Haploscoloplos fragilis*. Epibenthic fishes and macroinvertebrates were sampled by monthly trawling. Twenty-three species of fishes (mostly juveniles) were collected. Diversity and abundance were greatest during the months May through September. *Bairdiella chrysoura* (silver perch), *Orthopristis chrysoptera* (pigfish), and *Lagodon rhombodies* (pinfish) made up 76% of the total fish numbers. Eleven species of macroinvertebrates were collected in trawls, most abundantly in June and July. *Callinectes sapidus* (blue crab) made up 61% of the total invertebrate numbers. Sheridan and Livingston postulated that the influx of juvenile fishes and crabs into the *Halodule* meadow in summer months caused a coincident decline of infaunal population densities (number per $m^2$) through predation. Infaunal biomasses were largely unaffected by these predators because the biomass dominants were large or deep-burrowing species.

Mangroves

Wood et al. (1969) described the functional roles of mangroves. These include the following factors:

1. Organic productivity is relatively high, and for some species it rivals or often exceeds that of subsidized agricultural crops.

2. Standing crops are high but few organisms feed directly on the plant. As a result, mangrove systems produce large quantities of detritus, which plays a major role in the dynamics of the particular system and the estuary of which they are a part.

3. Leaves, stems, and prop roots present surfaces for epibiotic organisms. This increases both the primary and secondary productivity of the habitat and the epibiota may be significant food sources for fish and invertebrates.

4. Roots, stems, and leaves reduce current velocity, thus promoting sedimentation of both inorganic and organic matter. Entrained allochthonous and autochthonous material decomposes, thus recycling nutrients within the system.

5. The root system generally binds sediments and retards erosion.

6. The presence of above-substrate vegetation and lateral zonation presents a wide variety of habitats for protection and growth of fish, birds, and invertebrates.

Four species of mangroves are native to the United States: red mangrove (*Rhizophora mangle*), black mangrove (*Avicennia germinans*), white mangrove (*Laguncularia racemosa*), and buttonwood (*Conocarpus erectus*). Only black mangroves occur more or less continuously from the Dry Tortugas to the islands off the Mississippi Delta (Humm, 1973). Black and red mangroves also occur in Texas (Duncan, 1974; Sherrod and McMillan, 1985). The distribution of mangrove wetlands is limited by temperature, and permanent communities generally do not occur north of 25° north latitude or in areas where temperatures routinely fall below -4° degrees C (Thayer and Ustach, 1981). Approximately 450,000 ac (182,186 ha) of mangroves occur along the Gulf coast, with the largest proportion occurring in Florida. The Ten Thousand Islands area of Florida has the greatest aereal extent of mangroves along the Gulf coast.

The different species of mangroves have different ecological requirements and exhibit zonation along environmental gradients. Red mangroves are generally found in the deepest waters along shorelines; landward are black mangroves, white mangroves, and buttonwoods (McNulty et al., 1972).

Factors involved in zonation are disputed, but are generally considered to be the degree and duration of inundation and soil and water salinity (Lugo and Snedaker, 1974).

Lewis et al. (1985b) described the three mangrove species to have the following requirements:

1. The red mangrove, *R. mangle*, is the most saline tolerant. Arching prop roots act as a tree stabilizer and brace, as well as aid in gas exchange under anoxic conditions, as these trees often grow in oxygen-depleted areas.

2. The black mangrove, *A. germinans*, is found landward of the red mangrove. It reaches heights of 60 feet and is supported by a network of horizontal cable roots that give rise to vertical aerating roots called pneumatophores, which appear as finger-like emergents in the soil.

3. The white mangrove, *L. racemosa*, is the most landward species of the three. When its habitat is flooded for extended periods of time it will produce adventitious roots.

Lewis et al. (1985b) state that mangrove forests thrive under strict environmental parameters. Mangroves grow best in 10-20 ppt saltwater. These plants require a certain amount of fresh water for adequate growth, and the salinity factor is important because it enables the mangroves to better compete with other salt-tolerant plants. Fluctuating tides are necessary because they act to remove toxic substances from the root system. Lewis et al. (1985b) described five types of mangrove communities:

1. The <u>basin forests</u>, which are dominated by red mangroves, are located near terrestrial runoff sites and associated with regular tidal flushing. In those areas where tidal flushing rarely occurs, low salinities exist and white and black mangroves dominate.

2. The <u>riverine forests</u> are associated with rivers that are apt to flood. These forests frequently are inhabited by the tallest red mangrove stands.

3. The <u>fringe forests</u> are found on the edge of the mainland or surrounding islands and act as the buffer zone in high-energy situations.

4. The <u>overwash forests</u> are characterized by low organic accumulations that are frequently washed away by tidal flushes.

5. The <u>dwarf or scrub forests</u> are stands of mangroves that are stunted in growth due to low organic productivity, making few nutrients available to them.

Distribution and productivity of mangroves may also be related to soil parameters. Odum et al. (1982) found that:

1. Mangroves can grow on a wide variety of substrates including mud, sand, peat, and limestone crevices.

2. Mangrove ecosystems appear to flourish on fine-grained sediments that are usually anaerobic and may have a high organic content.

3. Mangrove ecosystems that persist for some time may modify the underlying substrate through peat formation. This appears to occur only in the absence of strong physical forces.

4. Mangrove peat is formed primarily by red mangroves and consists predominantly of root material.

5. Red mangrove peats may reach thicknesses of several meters, have a relatively low pH, and are capable of dissolving underlying layers of limestone.

6. When drained, dried, and aerated, mangrove soils usually experience dramatic increases in acidity due to the oxidation of reduced sulfur compounds. This greatly complicates their conversion to agriculture and other uses.

Several studies have described the occurrence and importance of mangrove communities in the coastal system of Florida (Lewis et al., 1985b; Odum et al., 1982; Patterson, 1986; Thayer and Ustach, 1981; Thayer et al., 1987; and Yokel, 1975a and b). Ninety percent of the approximately 500,000 ac (202,429 ha) of mangrove forests located in Florida are found in four counties: Lee, Collier, Monroe, and Dade. These mangrove forests serve as "land stabilizers," because the trees act as a barrier, protecting the inland ecological systems from storm surges.

Areal estimates of mangrove communities in Florida vary from 430,000 to 650,000 ac (174,089 to 263,158 ha) due to: 1) the inclusion or exclusion of bays, ponds, and creeks, or 2) the incorrect identification of mangroves and marshes from aerial photography (Lewis et al., 1985b). Clear examples of differences in these estimates may be observed by comparing acreages given by the Coastal Coordinating Council (CCC) with those given by the National Wetland Inventory (NWI) (Lewis et al. 1985b). The CCC estimates 469,000 ac (190,000 ha) with a 15% error (400,000-540,000 ac; 162,000-219,000 ha). The NWI estimates 674,241 ac (272,973 ha) of mangroves (and 383,317 ac (155,190 ha) of tidal marshes). Lewis et al. (1985b) theorize that the difference in acreages may be the result of misidentification between marshes and mangroves on aerial photographs. A summary of the acreages of emergent wetlands of coastal counties in Florida according to the NWI are given in Table 21.

**Table 21. Estimates of Emergent Wetland Acreages (Acres) of Gulf Coast Counties in Florida (Source: National Wetland Inventory, St. Petersburg, 1982, modified from Lewis et al., 1985b).**

| County | Mangroves (acres) | Tidal Marshes (acres) |
|---|---|---|
| Bay | - | 7,207 |
| Charlotte | 22,431 | 3,831 |
| Citrus | 3,394 | 36,273 |
| Collier | 85,513 | 14,177 |
| Dixie | 243 | 19,259 |
| Escambia | 78 | 2,075 |
| Franklin | - | 16,538 |
| Gulf | - | 5,257 |
| Hernando | 136 | 11,792 |
| Hillsborough | 10,095 | 1,675 |
| Indian River | 4,133 | 910 |
| Jefferson | - | 4,865 |
| Lee | 40,164 | 2,832 |
| Levy | - | 35,703 |
| Manatee | 5,754 | 1,029 |
| Monroe | 361,036 | 11,834 |
| Okaloosa | - | 257 |
| Pasco | 10,588 | 12,228 |
| Pinellas | 7,216 | 423 |
| Santa Rosa | 148 | 7,125 |
| Sarasota | 1,115 | 1,128 |
| Taylor | 154 | 23,740 |
| Wakulla | - | 19,658 |
| Walton | 14 | 2,938 |
| TOTAL | 548,114 | 238,994 |

Mangrove forests are composed of a "habitat mosaic," and there is much interaction between the different habitats within this mosaic. Adverse impacts on part of the mosaic can result in loss of production within the entire forest (Lewis et al., 1985b). This complex community is particularly important to maintenance of fish populations. There is a positive correlation between the acreage of mangrove communities and the recruitment of juvenile fishes (Lewis et al., 1985b).

Mangrove forests are subject to degradation due to human activities. These activities include: 1) impounding or ditching for mosquito control, 2) reduction of fresh water input, 3) clearing, and 4) filling. Since 1972, 23,521 ac (9,523 ha) of wetlands (mostly mangroves) and open water have been filled along the west coast of Florida and another 1,792 mi (2,890 km) of navigational channels have been dredged. Of the approximately 650,000 ac (263,158 ha) of mangroves that existed before 1972, 23% have been lost and only approximately 500,000 ac (202,429 ha) remain. If the remaining mangrove forests and marsh ecosystems are not protected, irreparable declines in fisheries production will occur (Lewis et al., 1985b).

A number of studies have been conducted on the impact of pollutants on mangrove communities. Mangroves do not actively take up organic pollutants but can concentrate heavy metals. Concentrations of up to six times the background level of heavy metals have been found in mangrove tissues. It is not certain whether heavy metal contamination is transferred to higher organisms through the detrital food web. A mangrove tree may drop 217-1,082 gm dry weight/$m^2$/yr litter and this organic matter forms the base of the detritus food web in systems where mangroves predominate (Lewis et al., 1985b).

The Rookery Bay Project (Yokel, 1975a and b) was initiated as a result of public concern over the fate of coastal mangrove communities of Florida. In 1964, citizens of Naples, Florida became alarmed at the loss of coastal wetlands and fisheries resources in Collier County. By 1966, the Collier County Conservancy purchased 4,000 ac (1,619 ha) of bays, islands, and mangrove shorelines that became the Rookery Bay Sanctuary. To date, over 5,000 ac (2,024 ha) of shoreline are included in the Rookery Bay Sanctuary representing an investment of over $1 million of private capital. Located 5 mi (8 km) south of Naples, Florida, the Sanctuary consists of 5,038 ac (2,040 ha) of uplands, marshes, mangrove forests, tidal creeks, and open water areas. Mangroves dominate this system with 2,368 ac (959 ha; 47%), while bays and tidal creeks account for 1,746 ac (707 ha; 35.1%), and tidal marshes 688 ac (278 ha; 13.7%). These three submerged and intertidal habitats account for 95% of the surface area. The results of the Rookery Bay study have provided some invaluable insights into the requirements and vegetative characteristics of mangroves. Studies of preserved, pristine mangrove systems, such as Rookery Bay, are critical to defining and quantifying the detrimental impacts on these communities which have occurred due to coastal development.

Odum et al. (1982) discussed the inhabitants of mangrove forests. Many animals utilize mangrove forest habitats during various stages in their lives. However, very few species are dependent directly upon growing mangrove tissue as a food source; these include the white-tailed deer, the mangrove tree crab, a wood-boring isopod, and a variety of insects. Many species of invertebrates and fishes depend upon mangrove detritus as a major source of energy.

The structurally diverse habitats provided by mangroves harbor a greater variety of bird life than salt marshes, mud flats, and beaches. Wading birds, probing shore birds, floating and diving water birds, aerially searching birds, birds of prey, arboreal birds, and migrating North American land birds depend on the mangrove forests for food and shelter (Odum et al., 1982). Economically important species associated with mangroves include the spiny lobster and grey snapper.

Endangered Species and Habitats

Coastal habitats are occupied by many plants and animals considered threatened or endangered by extinction. According to the U.S. FWS (1979), "endangered" species are those that are in danger of extinction throughout all or a significant portion of their range. A "threatened" species is defined as one that is likely to become endangered in the foreseeable future. The U.S. FWS (1979, 1980, 1983, and 1987b), U.S. Department of the Interior, MMS (1986), heritage programs within the various states (e.g., Texas Natural Heritage Program, 1989b,c), and various individuals (e.g. Eley, 1989) have summarized information on these species. The following discussion is not meant to be a definitive summary of the ecology and life histories of endangered Gulf coast species, but rather to emphasize their specific habitat requirements, where known.

The U.S. FWS (1980) published a series of species accounts to provide resource managers and the public with information about Federally listed endangered and/or threatened vertebrate species that occur along or within 62 mi (100 km) of the seacoast of the United States. Information on life history, distribution, requirements, and conservation is included for each species. This series is intended to complement the computerized Sensitive Wildlife Information System (SWIS) developed by the U.S. Army COE in coordination with the Offices of Endangered Species and Biological Services of the U.S. FWS.

Figures 6 through 9 give the distribution of selected endangered and threatened vertebrates which occur along the Gulf coasts as reported by FWS (1980). Although four species of sea turtles are generally found along the entire Gulf coast (Figure 6), their numbers have severely declined in recent years. The future of many of these species is threatened due to encroachment of development on critical habitats.

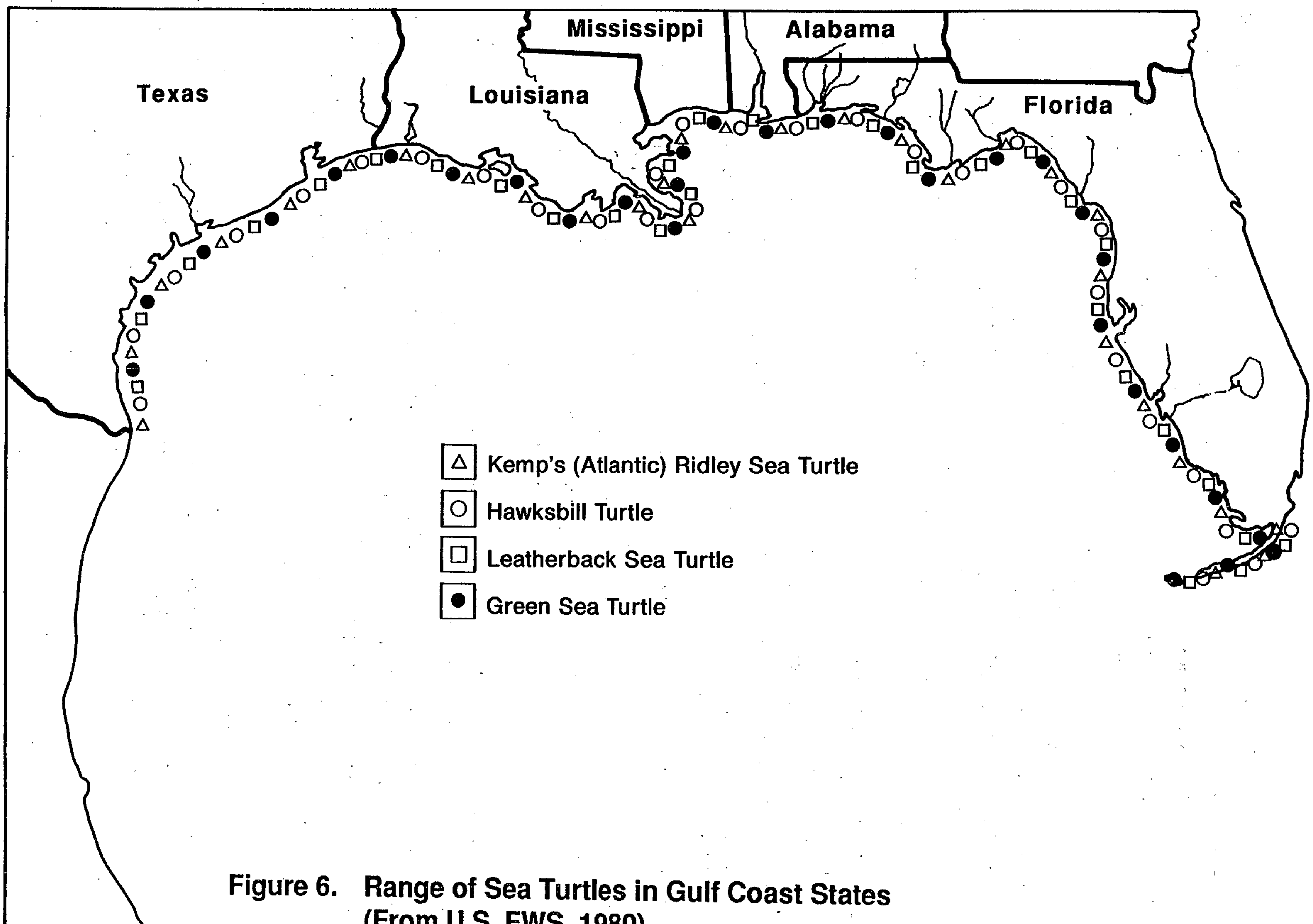


Figure 6. Range of Sea Turtles in Gulf Coast States (From U.S. FWS, 1980).

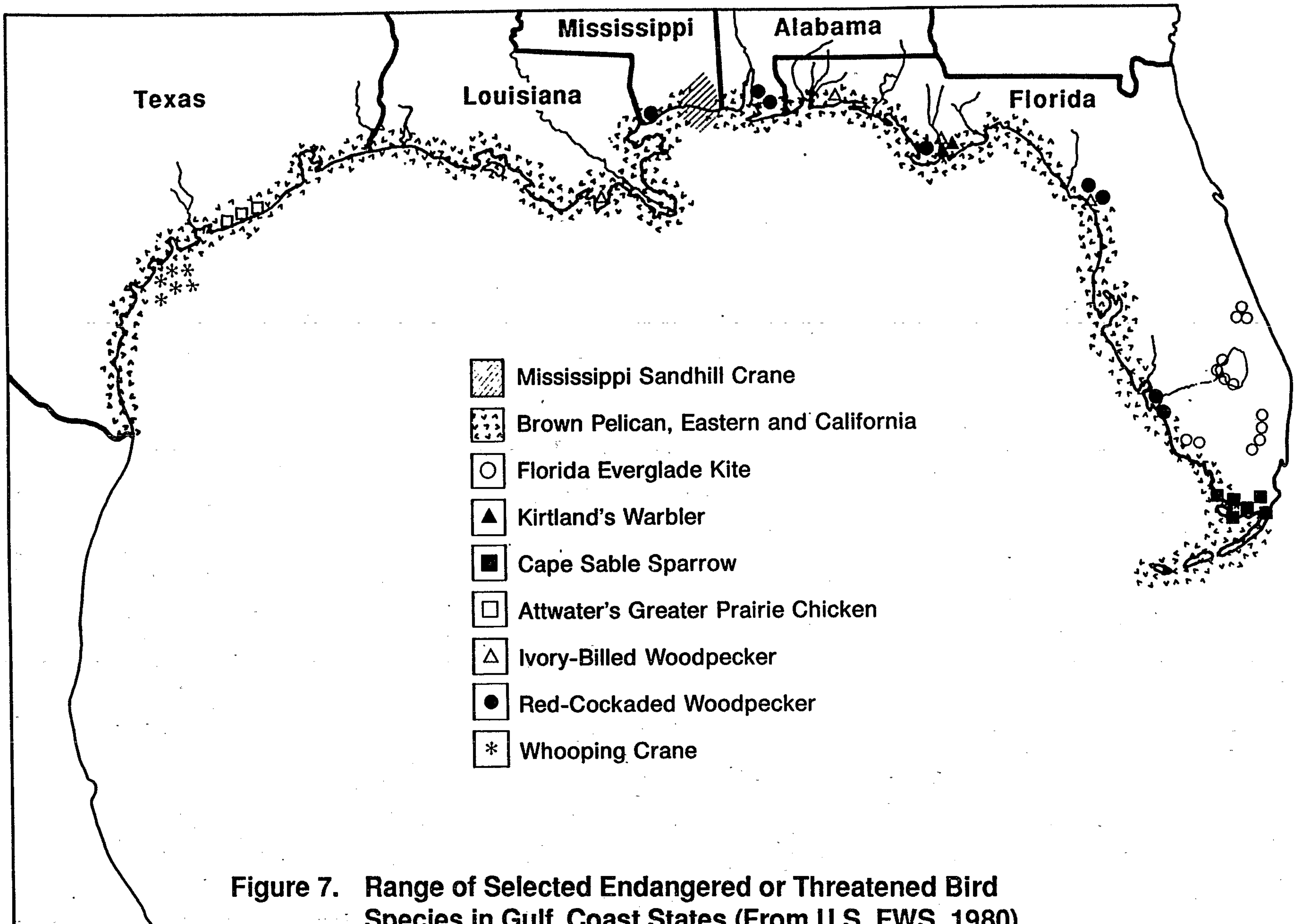


Figure 7. Range of Selected Endangered or Threatened Bird Species in Gulf Coast States (From U.S. FWS, 1980).

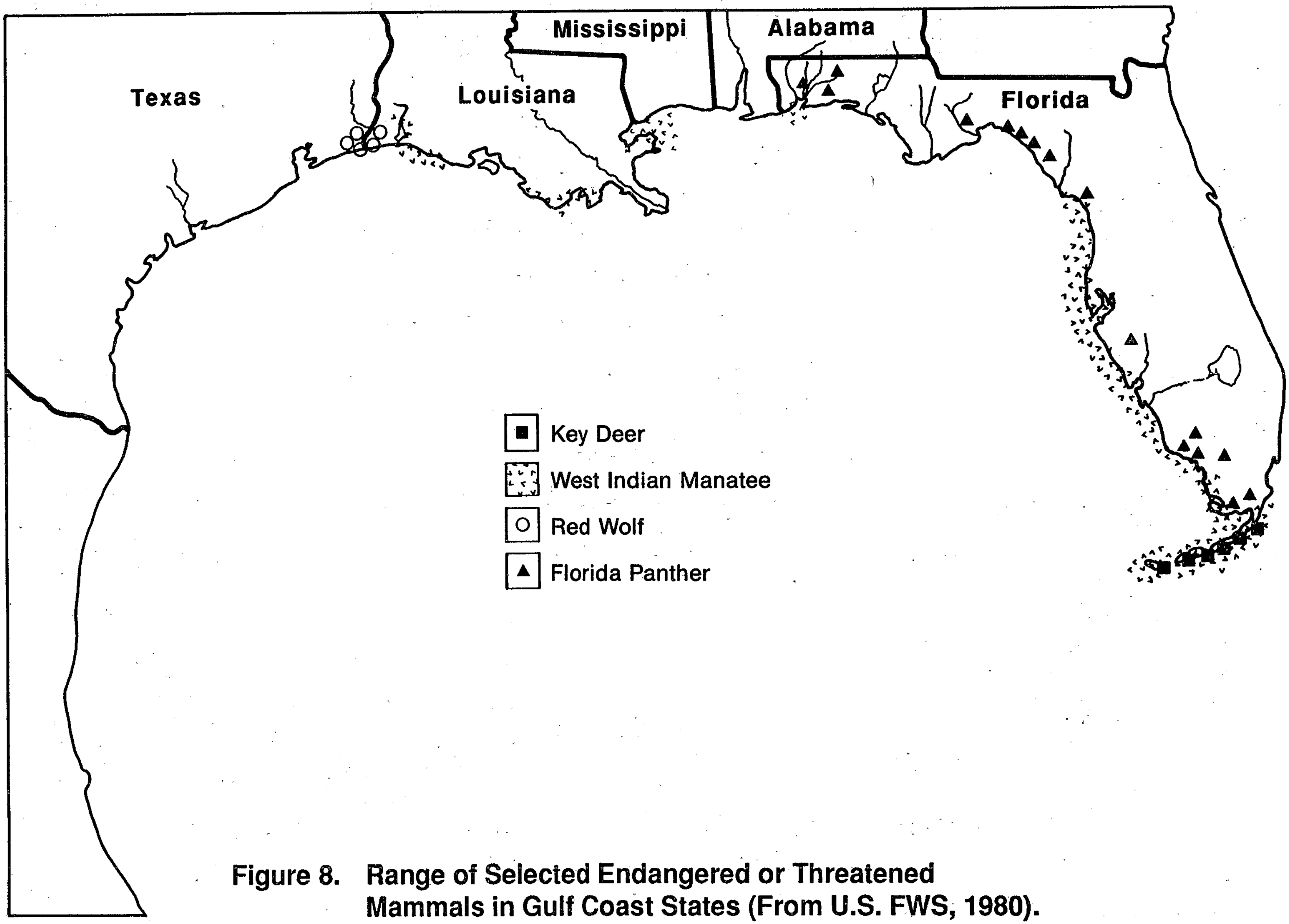


Figure 8. Range of Selected Endangered or Threatened Mammals in Gulf Coast States (From U.S. FWS, 1980).

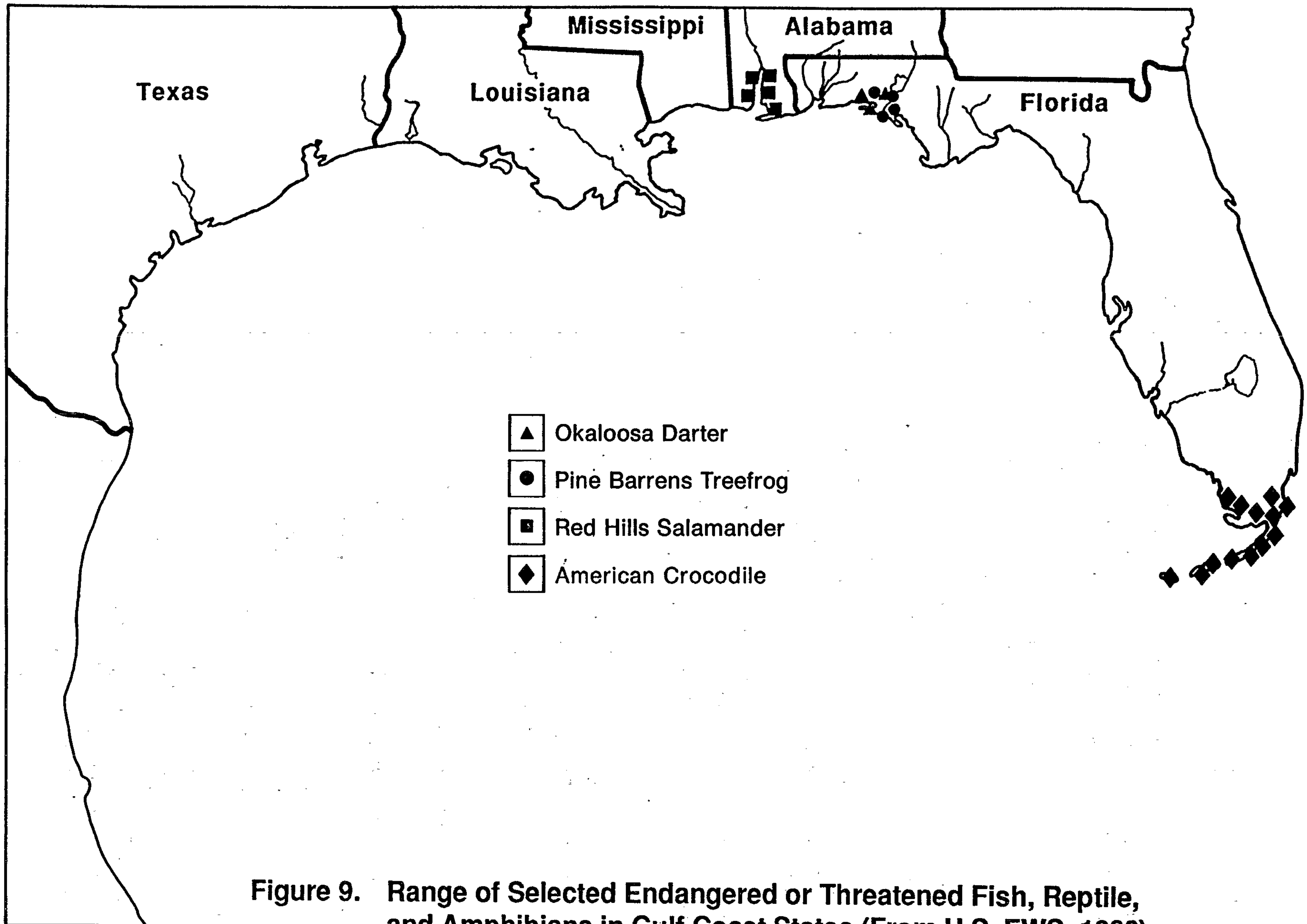


Figure 9. Range of Selected Endangered or Threatened Fish, Reptile, and Amphibians in Gulf Coast States (From U.S. FWS, 1980).

Many states have programs which identify and list "sensitive" species and habitats. For example, the Louisiana Natural Heritage Program (LNHP, 1988) is a comprehensive, computerized, ecological inventory of the sensitive plants and animals and outstanding natural areas of Louisiana. The program was initiated by the Nature Conservancy, a private conservation organization specializing in the preservation of ecologically significant lands, including endangered species habitat, outstanding examples of native natural communities, environmentally critical areas, and sites for scientific research. The program is currently an interagency effort of the Louisiana Department of Wildlife and Fisheries and the Louisiana Department of Natural Resources (LDNR). The data base contains 545 elements in three classes: special plants, special animals (including waterbird nesting colonies), and natural communities. Forty-two percent of these elements have been recorded in the Coastal Zone. Of 719 known occurrences of these elements in the Coastal Zone, 200 (28%) are special animals, 222 (31%) are special plants, 131 (18%) are natural communities, and 166 (23%) are unique features such as waterbird nesting colonies (162 occurrences), salt domes (3 occurrences), and "champion" (i.e. large, old) trees (1 occurrence).

Because of the large number of occurrences of sensitive plants and animals in the Louisiana Coastal Zone and the high level of natural and human threats associated with this area, the LNHP and LDNR have published information to aid field investigators in identifying sensitive species and habitats (Louisiana National Heritage Program, 1988).

A special list on sensitive plant and animal species occurring in 13 coastal Texas counties was prepared by the Texas Natural Heritage Program (1989a). Also, the Texas Colonial Waterbird Census Summary (1988) contains information on rookery locations by county and latitude/longitude coordinates.

Special interest has recently been given to the survival of endangered and threatened species of sea turtles in the Gulf of Mexico. The literature on sea turtles is too voluminous to summarize in this document and the interested reader should contact the National Marine Fisheries Service for more detailed information. Eley (1989) suggests that due to rapidly diminishing populations, marine turtles are receiving much attention at this time. All seven of the world's recognized species of marine turtles are classified as either endangered, threatened, or rare. Very little ecological or behavioral information has been recorded for the five species of marine turtles which exist in the Gulf of Mexico.

Most turtle research has focused on the Caribbean, Pacific, and Indian Ocean populations; the northern Gulf of Mexico has received relatively little attention. Eley (1989) censused nesting females in order to develop a population index for the Gulf Islands National Seashore and adjacent areas in Florida, Alabama, and Mississippi. Aerial surveillance, the primary assessment method, was supported by ground surveys to ascertain the correction factor (number of nests missed). Approximately 58 hours of aerial surveys and about 300 hours of ground surveys yielded 133 loggerhead turtle nests and two green turtle nests along this portion of the Gulf coast (Table 22). It is not known whether suitability of beach nesting sites limits the populations of these species. Accidental killing of sea turtles in fishing nets is thought to be a major source of mortality of all species of sea turtles in the Gulf of Mexico.

Collard (1987) hypothesized that Kemp's Ridley turtle undergoes an extended pelagic planktonic developmental stage. The young hatchlings appear to vanish shortly after entering the ocean and only reappear in the subadult stage. The hatchlings frequently take refuge in floating aggregations of *Sargassum* (a macroalga often found floating in large mats throughout the Gulf of Mexico) probably as a predator-avoidance or resting tactic. Collard cited evidence that between 200 to 1500 adult female Kemp's Ridley turtles nest each year on a 17 km stretch of beach in Tamaulipas State, Mexico. It is estimated that the number of females nesting each year will decrease by 3%.

Collard states that even though survivorship rates of Kemp's Ridley turtles are unknown at this time, the odds are against the survival of hatchlings. Hatchlings apparently live off their yolk sac reserves until reaching the *Sargassum* mats. The mats may furnish habitat, food, and transportation, as well as protection, but it is not known how long they remain on the mats. Encrusting organisms indigenous to *Sargassum* mats have been found growing on young turtles, indicating extended periods of time spent there. Carr (1986) suggests that the turtles rely on the mats more as a concentration point for food than for transportation or other functions. DeSola and Abrams (1933) suggest that the Kemp's Ridley gut seems to be characteristic of herbivores, and it is not known whether gut morphology changes with age in this species. Forage preference changes when these turtles arrive in the demersal and inshore environment and exhibit an increased efficiency in searching for food. Their diet primarily consists of portunid crabs and other crustaceans, gastropods, bivalves, echinoderms, jellyfish, and squids.

Carr and Meylan (1980) examined the stomach contents of 15 green turtle hatchlings that were washed ashore during Hurricane David (1979) and found pieces of *Sargassum*, sargassum snails (*Litiopa melanostoma*), pelagic snails (*Diacria trispinosa*), and the isopod, *Idotea metallica*. This indicates that green turtle hatchlings feed on a variety of food organisms during their pelagic stage.

Rookeries are concentrations of colonial bird species and there are many bird rookeries along the Gulf coast. Information on rookeries is maintained by the U.S. FWS, state agencies, and heritage programs. Keller et al. (1984) summarized ground and air surveys of the coastal areas of Louisiana,

Table 22. **Number of Turtle Nests Observed Along Sections of Coast, Summer 1989 (from Eley, 1989).**

| Location[1] | Number of Nests | Length of Beach (km) | Density (nests/km) |
|---|---|---|---|
| Fort Walton Beach (D) | 9 | 28.8 | 0.31 |
| Eglin AFB (U) | 33 | 55.4 | 0.60 |
| Navarre (D) | 13 | 7.5 | 1.73 |
| Santa Rosa Area (U) | 13 | 43.2 | 0.30 |
| Pensacola Beach (D) | 11 | 8.6 | 1.28 |
| Fort Pickens (U) | 8 | 12.2 | 0.66 |
| Johnson Beach (U) | 8 | 15.0 | 0.53 |
| Perdido Key (D) | 9 | 12.6 | 0.71 |
| Gulf Shores-Ft. Morgan (D&U) | 13 | 46.7 | 0.28 |
| Dauphin Island (D&U) | 12 | 23.0 | 0.52 |
| Petit Bois (U) | 1 | 10.6 | 0.09 |
| Horn Island (U) | 2 | 20.1 | 0.10 |
| East Ship (U) | 1 | 6.1 | 0.16 |
| West Ship (U) | 0 | 6.1 | 0.00 |
| TOTAL | 133 | 295.9 | 0.45 |

[1](D)= Developed Coastline
(U)= Undeveloped Coastline

Mississippi, and Alabama to locate active and abandoned nesting sites of wading birds and seabird colonies. When these surveys were first conducted in 1976 (Portney, 1977), 847,000 birds representing 26 species were reported nesting in the area's coastal swamps, marshes, and on barrier islands. The census was carried out via ground and aerial surveys by teams who determined species composition, nesting habitat, and colony sizes. The teams also plotted colonies on 1:250,000-scale maps and on 1:24,000-scale USGS topographic or wetland habitat maps.

Two surveys were conducted one month apart for each census year. The first was performed during the nesting period of wading birds (herons, egrets, ibises) and Forster's terns, and the second during nesting periods of other birds of interest. Individual colonies were defined as groups of birds nesting together and separated by at least 0.62 mi (1 km). If nesting birds were not found within 0.62 mi (1 km) of a historic site, that colony was recorded as deserted.

The manual contains an atlas section which consists of a series of photographically reduced 1:250,000-scale maps that locate individual colonies by a six-digit number. Also included in this manual are 98 maps at 1:24,000 scale which depict active rookery sites, historic sites, and new rookeries.

Numbers of wading birds and rookeries are directly correlated with wetland acreage. Nesbitt et al. (1982) documented that wading birds are dependent upon wetland habitats for nesting and feeding sites. Most waders nest between February and August in colonies that may contain several species and anywhere from a few dozen to thousands of nests. Colony sites may be used in other seasons as roosting sites. The birds fan out daily from the colonies, flying as far as 20 mi (32 km) to feed in a variety of wetland habitats. Most species eat small fish, frogs, aquatic insects, crayfish, or fresh water prawns, while the ubiquitous cattle egret eats insects captured in fields, pastures, and along roadsides.

The location and size of nesting colonies depend directly on the presence of suitable nesting habitat and adequate amounts of food. Most colonies in Florida are located in natural wetland habitats characterized by woody vegetation over standing water or on islands. An increasing number of colonies are now being found in artificial habitats such as dredged-material islands and fresh water impoundments. This latter situation may suggest that sufficient undisturbed natural sites are no longer available (Nesbit et al., 1982).

It is very difficult to accurately determine the number of wading birds nesting in any locality. Such an inventory requires a well organized effort, because wading birds move freely over large areas and colony sites are frequently abandoned in one area in preference for another site the next year.

Thus, long-term records of birds in a single colony may not accurately reflect statewide population trends.

Although absolute numbers of the entire population of wading birds in Florida was never measured, some historical information does exist on the trends in wading bird populations in south Florida. A historical review of population trends of waders in southern Florida (Lake Okeechobee south into Florida Bay) indicates that the estimated total number has changed drastically over the past century due to human activities, primarily hunting and habitat alteration (Robertson and Kushlan, 1974). The estimated number of south Florida waders dropped from 2,500,000 birds in 1870 to 500,000 in 1910, primarily due to plume hunting, and then increased to 1,200,000 by 1953 after plume hunting was made illegal. Increased rates of wetland destruction since 1935 have resulted in a second acute decline to an estimated 150,000 waders in 1970 (Robertson and Kushlan, 1974) and 130,000 in 1975 (Kushlan and White, 1977). This recent decline is more pernicious since it may be due primarily to habitat destruction and appears to be continuing. A review of regional wading bird trends in the southeastern coastal plain shows that with the exception of cattle egrets and a few estuarine species, wading birds have experienced greater declines in Florida than in most other coastal states (Ogden, 1978). The most serious declines in Florida have occurred among those species inhabiting primarily freshwater habitats and nesting in interior colonies. It has become increasingly apparent that more quantitative information on wading birds and their nesting sites is needed to understand their problems and how their habitats may be protected.

## Case Histories

The following is a summary of the status and trends of vegetated habitats in specific regions, bays, and estuaries. These case histories provide more detailed accounts of specific wetland and seagrass modifications that have taken place along the Gulf coast.

### Texas

Matlock and Osborn (1982) provide a detailed summary of the shallow water habitats of estuaries in Texas. Approximately 601,000 acres (243,000 ha) of Texas bays are 4 ft ($\leq$ 1.2 m deep) (6 ft; $\leq$ 1.8 m in Corpus Christi Bay system). Approximately 24% of this shallow water habitat is found in lower Laguna Madre. The Galveston and Matagorda Bay systems each have about 99,000 acres (40,000 ha); the San Antonio Bay, Aransas Bay, and upper Laguna Madre systems have 57,000 ac (23,000 ha); East Matagorda has 37,000 ac (15,000 ha); and Corpus Christi Bay has 32,000 ac (13,000 ha).

The length of shoreline in Texas is 2,351 mi (3,792 km), including 365 mi (588 km) of Gulf beaches. The Galveston Bay system has the longest shoreline, 411 mi (662 km), followed by Matagorda, 285 mi (458 km); Aransas, 264 mi (426 km); and the lower Laguna Madre, 252 mi (406 km). The San Antonio Bay and upper Laguna Madre systems each have approximately 224 mi (361 km) of shoreline and the Corpus Christi Bay system has 171 mi (276 km) of shoreline. The shorelines of Sabine Lake, East Matagorda Bay, and Cedar Lake are less than 78 mi (126 km) each.

Texas estuaries support large commercial and sport fisheries. Dominant fishes found in Texas estuaries are species able to tolerate a wide range of salinity (Armstrong, 1987). The dominant fish species represent many different feeding strategies and include planktivores, detritivores, and predators. Abundant planktivorous species include: *Anchoa mitchilli* (bay anchovy), *Brevoortia patronus* (Gulf mendhaden), *Menidia beryllina* (tidewater silverside), *Mugil cephalus* (striped mullet), and *Lagodon rhomboides* (pinfish). Penaeid shrimp are common and are considered detritivores and predators. Other common predators are the crustaceans, *Callinectes* spp. (crabs), and *Squilla empusa* (mantis shrimp). Common vertebrate predators include *Micropogonias undulatus* (croaker), *Leiostomas xanthurus* (spot), *Paralichthys lethostigma* (southern flounder), *Pogonias cromis* (black drum), and *Cynoscion arenarius* (sand trout) (Armstrong, 1987). Many of these species are dependent upon tidal creek and marsh habitats for feeding and shelter during their life histories.

Volume, depth, and acreage of wetlands for the mid 1970's in the major Texas river delta estuaries is given in Table 23. In general, there has been a dramatic decrease in emergent wetland habitats during recent years. However, there have been some localized increases in wetland acreage.

White and Calnan (1990) analyzed the wetland losses in seven river deltas in Texas: Colorado, Nueces, Guadalupe, Lavaca, Trinity, Neches, and San Jacinto. Distribution of emergent vegetation was mapped from sequential aerial photographs, dating from the 1930's to the late 1970's and 1980's. Field studies were conducted in the Colorado River delta and the Trinity River delta to document changes in quantity and quality of marsh habitats.

**Table 23. Water Volume, Average Depth, and Marsh Area of River Deltas (from Armstrong, 1987)**

| Estuary | Volume ($km^3$) | Average Depth (m) | Marsh (acres) |
|---|---|---|---|
| Sabine Lake | 0.326 | 1.8 | 33,987 |
| Galveston Bay | 2.911 | 2.1 | 13,387 |
| Matagorda Bay | 2.134 | 2.3 | 28,232 |
| San Antonio Bay | 0.754 | 1.4 | 11,937 |
| Copano-Aransas Bays | 0.925 | 2.0 | ---- |
| Corpus Christi Bay | 1.147 | 2.4 | 13,214 |
| Laguna Madre | 2.574 | 1.2 | ---- |
| TOTAL | 10.771 | | |

A total of 52,758 ac (21,359 ha) of wetlands occurred in all areas studied in the 1930's (Table 24). The acreage of vegetated wetlands in the Colorado River delta has increased since the mid 1950's. However, there were marked declines in wetland habitat in all other Texas river estuaries. White and Calnan (1990) concluded that hurricanes can result in major deposition of sediments which can later result in marsh accretion which has occurred in the Colorado River delta. However, the general trend in the deltaic wetlands along the Texas coast is replacement of vegetated areas by shallow, open water habitat or barren intertidal flats.

Extensive losses of vegetated wetlands occurred from 1930-1980 along the San Jacinto and Neches Rivers. Those losses accounted for greater than 70% of the total wetland losses of Texas river deltas. Two factors were responsible for those losses: 1) rise of water level from human-induced subsidence, natural compactional subsidence, and global sea-level rise, and 2) processes which impede movement of sediment to marshes, such as reservoir development in river drainage basins and channelization and disposal of spoil on natural levees.

**Table 24. Areal Extent (Acres) of Vegetated Wetlands by River System and Year (Adapted from White and Calnan, 1990).**

| Fluvial-Deltaic Area and Year | Vegetated Area (Acres) |
|---|---|
| Colorado River | |
| 1937 | 799 |
| 1956 | 5,117 |
| 1974 | 5,061 |
| 1982 | 5,222 |
| Guadalupe River | |
| 1930 | 8,667 |
| 1957 | 7,903 |
| 1974 | 7,526 |
| 1979 | 7,428 |
| Lavaca River | |
| 1930 | 7,690 |
| 1958 | 7,313 |
| 1979 | 6,252 |
| Nueces River | |
| 1930 | 5,582 |
| 1959 | 5,449 |
| 1975 | 5,126 |
| 1979 | 5,262 |
| Neches River | |
| 1938 | 9,027 |
| 1956 | 8,182 |
| 1987 | 4,958 |
| San Jacinto River | |
| 1930 | 9,404 |
| 1956 | 7,948 |
| 1986 | 4,839 |
| Trinity River | |
| 1930 | 11,589 |
| 1956 | 13,061 |
| 1974 | 9,495 |
| 1988 | 9,276 |

The dynamics of submerged lands and associated coastal wetlands in Texas were recently studied during two projects, namely the Texas Barrier Island Project (Longely and Wright, 1989) and the Submerged Lands of Texas Project (White et al., 1983, 1985, 1986, 1987, 1989).

## Barrier Islands

The final report of the Texas Barrier Island Project presents a detailed description of each Texas coastal basin with emphases on habitats and hydrology. Information is presented on the area, physical appearance, flora and fauna, and human impacts on each region (Longley and Wright, 1989). This Project was sponsored by the MMS and was a cooperative venture between the U.S. FWS and Texas General Land Office. The Texas Barrier Island Region was examined through systems analysis utilizing models to integrate diverse information about the region. A summary of information on the ten coastal basins that make up the barrier region is presented in Table 25. Human activities have adversely affected this region. Subsidence due to withdrawal of subsurface water, oil, and gas has resulted in substantive increases in open water. Deposition of spoil and construction of roads, railroads, dams, and drainage canals have resulted in changes in historic hydrologic patterns. Disposal of sewage has resulted in restrictions on shellfish harvesting. Pesticides historically applied to crops have reached coastal waters due to erosion processes and have resulted in mortality of seafood and other estuarine organisms (Longley and Wright, 1989).

## Estuaries

The Submerged Lands of Texas Project was an extensive sampling program which was initiated in 1975. Surficial bottom sediments were collected at regularly spaced distances across the submerged lands and the project resulted in detailed sedimentological, geochemical, and biological characterization of shallow water habitats and tidal marshes. Samples were analyzed to characterize: (1) sediment distribution; (2) selected trace metals and nutrient concentrations; and (3) benthic macroinvertebrate populations.

Data from the Submerged Lands Project are presented in seven atlases and comprehensive texts for the following areas of the Texas coast: Brownsville-Harlingen (White et al., 1986), Kingsville (White et al., 1989), Corpus Christi (White et al., 1983), Port Lavaca (White et al., 1989), Bay City-Freeport (White et al., 1988), Galveston-Houston (White et al., 1985), and Beaumont-Port Arthur (White et al., 1987). General changes in the distribution of wetlands can be estimated by comparing aerial photographs from the 1950's with photographs taken in 1979. The authors recommend caution in making comparisons because: 1) wetland map units used in the two projects were similar but not identical; (2) moisture and tidal conditions were at higher levels during 1979 than during the 1950's,

**Table 25. Human Impacts on Selected Texas Barrier Island Regions and a Summary of Current Acreage (Acres) by Habitat Type (from Lonley and White, 1989).**

| | Habitat | | | | | |
|---|---|---|---|---|---|---|
| Basin | Barrier | Bay | Grassflat | Salt/Brackish Marsh | Wind-Tidal Flat | Human Impacts |
| Rio Grande | 13,640 | 4,547 | 9,662 | 74 | 8,253 | Land surface hydrology altered by roads, railway beds, drainage canals and the Brownsville ship channel. Wetlands adversely impacted by dredging and maintenance of ship channel. |
| Laguna Madre | 120,066 | 183,348 | 130,419 | 1,853 | 245,617 | Moderate level of industrial and commercial development, 57 waste discharges into bay. Spoil piles cause alteration in flow. |
| Corpus Christi | 139,612 | 657,286 | 103,041 | 29,405 | 60,045 | Major changes in hydrologic patterns from dredging and filling. Moderate problem with heavy metals in Nueces Bay. Some subsidence due to oil and gas extraction. |
| Aransas-Copano | 29,183 | 100,792 | 4,102 | 19,274 | 17,915 | Twenty-one permitted dischargers of which 12 are from domestic waste treatment plants. Water quality generally good. |
| San Antonio | 16,062 | 89,154 | 2,496 | 9,340 | 6,153 | Moderate water quality problems from runoff. Shellfish harvest restricted. Low quantity of waste materials added to basin. |

**Table 25. Continued**

| | Habitat | | | | | |
|---|---|---|---|---|---|---|
| Basin | Barrier | Bay | Grassflat | Salt/Brackish Marsh | Wind-Tidal Flat | Human Impacts |
| Matagorda | 34,248 | 233,139 | 4,225 | 30,986 | 9,217 | Shellfish harvest limited in some areas. Subsidence in vicinity of oil and gas wells. Large return of water through drainage of agricultural fields. |
| Colorado | 4,423 | 30,196 | 840 | 6,474 | 1,285 | Receives effluent from five permitted discharges, one of which involves domestic sewage. Subsidence due to extraction of fresh ground water, oil and gas. |
| Brazos-Colorado | 9,587 | 61,503 | 1,656 | 17,149 | 2,298 | Subsidence around old oil and gas fields. Some restrictions on shellfish harvest. |
| Brazos | 1,927 | 7,611 | 198 | 7,092 | 395 | Entire basin impacted by subsidence to some degree. Hydrologic changes due to flood control dikes and canals. |
| Galveston | 44,404 | 269,561 | 4,473 | 61,058 | 3,879 | Widespread subsidence. Approximately 50% of estuarine waters of Galveston Bay closed to shellfishing. Many hydrologic changes due to alteration of flow from roads, railroads and ditches. |
| Totals | 413,152 | 1,637,137 | 261,112 | 182,705 | 355,057 | |

which was a period of drought; (3) more refined interpretations were possible due to the high quality, color-infrared photographs taken in 1979; and (4) wetland mapping criteria varied between the two projects. The information contained in these and other studies for specific Texas estuaries are summarized below.

Matagorda Bay

McGowen and Brewton (1975) reported a net loss of marsh area between 1856 and 1957 of about 5,000 ac (2,024 ha) in Matagorda Bay (Port Lavaca). These losses were due to natural processes and human activities. Natural processes included shoreline erosion or accretion, land-surface subsidence and burial beneath sediment, and burial loss during drought. Human activities included burial of marshes by spoil disposal along bay margins, erosion of spoil creating conditions favorable for marsh growth, and submergence of marshes by construction of dams across tidal creeks.

A visual comparison of wetlands on maps from the 1950's with those of 1979 show the following changes in the Port Lavaca area: 1) conversion of intertidal and subtidal unvegetated shallows to grassflats; 2) reduction of grassflats along some margins of the bay; 3) spread of saltwater marshes over wind-tidal flats; 4) expansion of shallow subaqueous flats in areas of saltwater and brackish water marshes; 5) landward expansion of existing marshes; 6) loss of marshes due to erosion of bay shorelines; 7) erosion of spoil and encroachment of marsh vegetation along spoil island margins; and 8) reduction of wetlands as a result of human activities.

Between 1956-57 and 1979, many vegetated areas occurring along the margins of the Bay were replaced by subaqueous flats or wind-tidal flats. This change was particularly evident in the marsh between Blackjack Peninsula and Sundown Bay, the interior of the marshes of St. Charles Bay, and the southwest-projecting marsh on the eastern side of the mouth of Lavaca Bay.

Human activities which produced this habitat loss included: 1) dredging and spoil disposal along the Intracoastal Waterway; 2) construction of holding ponds northeast of Mission Lake along Victoria Channel, and 3) dam construction at the mouth of the small inlet along the northern shore of Matagorda Bay east of Carancahua Bay. Local gains in marsh vegetation occurred as a result of colonization of eroded spoil mounds and drainage of Burgentine Lake at the head of St. Charles Bay.

In 1979, the wetlands area had increased along the Pleistocene barrier strandplain, particularly on Blackjack Peninsula, from the area shown in 1956-57. This was attributed to wetter climatic conditions in 1979. The photographs in 1979 demonstrated obvious differences in the amount of standing water and moisture levels in depressions on the Pleistocene barrier strandplain. The higher quality infrared photographs of 1979 also allowed a more detailed delineation of habitats. In addition, two new map units were used in 1979, undifferentiated wetland/upland areas and transitional areas, which allowed depiction of complex upland and wetland mosaics. The numerous small marshes, water bodies, and transitional areas on the coastal plain, such as those west of the San Antonio River, were within regions mapped as frequently flooded fluvial areas in the earlier study. The increase in wetlands in these areas in 1979 was a result of mapping criteria and map unit designations. Some of the additional wetlands noted in 1979 were also attributed to higher levels of precipitation.

Brownsville

Near the Brownsville ship channel, obvious differences in the distribution of salt and brackish water marshes were identified from comparison of 1960 photographs with those of 1979. In 1979, marshes occurred in areas previously mapped as saline grasslands; this difference may have been due to higher precipitation of the 1970's. Mangroves, salt marshes, and brackish water marshes along San Martin Lake appeared to have spread into previously barren wind-tidal flats. Changes in municipal and agricultural discharges into San Martin Lake may have contributed to the spread of marshes. Little change was observed for inland fluvial-deltaic system wetlands. The resacas (former, often marshy, stream channels) generally were mapped as fresh and saline water bodies and frequently flooded fluvial areas.

Visual comparison of photographs from 1960 with those of 1979 and 1983 (White et al., 1986) show that the following changes have occurred in the Brownsville-Harlingen area: 1) local increases in marshes, including mangrove marshes along the margins of wind-tidal flats; 2) erosion of spoil and encroachment of marsh vegetation along spoil-island margins and flats; 3) local expansion of marine grassflats near the mainland shore of Laguna Madre; and 4) displacement of marine grassflats along the lagoonward margin of Padre Island by dredged channels and local storm-washover deposits.

Neches River

The most extensive changes in the Beaumont-Port Arthur area have occurred along the river valleys. The Neches River is a dramatic example. Between Port Neches and Bridge City, extensive marshes existing in 1956 were completely replaced by open water by 1978. Dredged canals, sediment reduction, subsidence, and sea-level rise contributed to this habitat loss.

The dredging of the Neches River for deep-draft navigation and disposal of spoil on the banks created an upland levee between the river and adjacent marsh areas. This levee deprived the adjacent marsh of sediment which was historically supplied by flooding events. In addition, natural subsidence in the chenier plain of about 0.7 inches (1.7 cm) per year, coupled with the rise in sea level, contributed to the drowning of these marshes.

West of State Highway 87, an increase in vegetated wetlands from 50 acres (20 ha) to 630 acres (255 ha) was observed (Weirsema et al., 1973) between 1957 and 1971. This gain was attributed to the intrusion of water from a dredged discharge canal created by the Sabine Power Station.

Human-induced subsidence was a major cause of marsh losses. Solution mining for sulfur in salt dome cap rock, extraction of oil and gas, and extraction of ground water have resulted in subsidence. White et al. (1987) presented evidence of subsidence occurring in the Neches River Valley along a fault line. The fault is more easily identified in the 1978 photographs compared to 1956 photographs. Changes from marsh to open water from all causes in the Neches River basin are summarized in Table 26.

**Table 26. Net Changes in Area from Marsh to Open Water in the Neches River Basin, Texas (From White et al., 1987)**

| Map Unit | 1956 Acres | 1978 Acres | Net Change* Acres |
|---|---|---|---|
| Water | 2,560 | 11,070 | +8,510 |
| Marsh | 15,740 | 6,330 | -9,410 |

* Does not include loss of an additional 900 acres (365 hectares) of marsh due to spoil disposal.

## Galveston Bay

Galveston Bay is the seventh largest estuary in the United States and the largest in Texas (Sea Grant College Program, 1989a). It is an irregularly shaped, shallow body of water, roughly 30 mi (48 km) long and 17 mi (27 km) wide at its widest point. Water depth is generally between 7 and 9 ft (2 to 3 m). It is nearly separated into two parts by Red Fish Bar, a chain of small islets and shoals. The part of the bay north of Red Fish Bar is called the Upper Bay, and the area to the south is known as Lower Bay. The northeastern section of Upper Bay is known as Trinity Bay. This bay system is unique to Texas since it occurs in an area with high rainfall and humidity, as opposed to the more arid climate found further south along the Texas coast. High rainfall aids in maintaining low overall salinity of Galveston Bay. Also, relatively low temperatures are especially important during the summer fish and shellfish reproduction periods.

Galveston Bay provides the nursery and spawning grounds for approximately 30% of the total fisheries harvest from the Texas coast. While turbid, water quality in the bay is typically described as good, although it often has unacceptable levels of coliform bacteria. Large portions of the bay are often closed to shellfishing, primarily as a result of bacteria introduced by runoff from surrounding lands. The turbidity is a result of the bay's shallow nature and local wind patterns.

The greatest problem regarding maintenance of Galveston Bay habitats has been human manipulation and utilization of estuarine resources, including coastal and freshwater wetlands and other coastal habitats. The total area of marsh vegetation in Galveston Bay declined by approximately 16 % from 1956 to 1979 (Table 27). Remaining salt marshes are vegetated predominantly by *S. alterniflora* and cover approximately 34,580 ac (14,000 ha). Brackish marshes occur in areas of moderate salinity and cover 56,810 ac (23,000 ha). There are 9,880 ac (4,000 ha) of fresh marsh confined to the northern portions of Galveston Bay. A comparison of the 1956 and 1979 wetland surveys by the U.S. National Wetland Inventory indicates losses of 15,561 ac (6,300 ha) of fresh marsh and 10,374 (4,200 ha) of salt and brackish marshes during that period (NOAA, 1989c). The aerial extent of submerged vegetation in Galveston Bay declined from 5,187 ac (2,100 ha) in 1960 to less than 247 ac (100 ha) in 1979 (NOAA, 1989c).

**Table 27. Galveston Bay Wetland Habitat Changes, 1956-1979 (Adapted from NOAA, 1989c)[1]**

| Wetland Habitat Type | 1956 Acres | 1979 Acres | Change Acres | Change %age |
|---|---|---|---|---|
| Estuarine Marsh | 154,588 | 130,139 | -24,449 | -15.8 |
| Estuarine Open Water | 363,213 | 388,397 | +25,184 | +6.9 |
| Beach | 3,015 | 1,413 | -1,602 | -53.1 |

[1] Changes in acres calculated from U.S. FWS National Wetland Inventory Maps

Galveston Bay supports a large commercial and recreational fishery. Landings for 1986 are given in Table 28. The percent of total Texas inshore fish landings, which were harvested by recreational fishermen, has remained relatively constant between 1983-1986 (Table 29). However, there appears to be an overall decline in landings from all Texas bays.

**Table 28. Landings of Commercial and Recreational Fisheries in Galveston Bay in 1986. (NOAA 1989c).**

| | Commercial (x 1000) | | Recreational (x 1000) | |
|---|---|---|---|---|
| | Kg | $ | Kg | $ |
| Atlantic Croaker | 18 | 9 | 37 | 11 |
| Flounder | 73 | 157 | 39 | 52 |
| Blue Crab | 1,375 | 1,043 | -[1] | - |
| Shrimp (3 species) | 6,820 | 18,135 | - | - |
| Oyster | 1,610 | 6,950 | - | - |

[1] Not available.

**Table 29. Annual Weight of Finfish Landed by Recreational Boat Fishermen from Galveston Bay during 1984, 1985, and 1986 (Osbourn and Fergason, 1987)**

| Year | Galveston Bay | All Texas Bays | % of total from Galveston Bay |
|---|---|---|---|
| 1983-1984 | 1,391,100 | 4,316,900 | 32 |
| 1984-1985 | 940,700 | 2,922,000 | 32 |
| 1985-1986 | 1,121,400 | 3,205,400 | 35 |

A description of the biological components of Galveston Bay is given by NOAA (1989c). Oyster reef assemblages occur primarily in central Galveston Bay and divide Galveston Bay into upper and lower sections. Public oyster reefs within the estuary are densest in the mid- bay region and across the mouth of East Bay. Settlement of oyster spat generally occurs from April to November. Oysters reach market size in 13 to 18 months. Since 1975, the areal distribution of oyster beds has been fairly stable.

NOAA (1989c) found that the following six species of fish accounted for 91% of the total number of fish collected during a 2-year synoptic trawl study: *Micropogonias undulatus* (Atlantic croaker, 51%), *Anchoa mitchilli* (bay anchovy, 22%), *Stellifer lanceolatus* (star drum, 8%), *Leiostomus xanthurus* (spot, 4%), *Cynoscion arenarius* (sand seatrout, 3%), and *Arius felis* (hardhead catfish, 3%). These six species, plus *Mugil cephalus* (striped mullet), were responsible for 74% of the biomass collected (NOAA, 1989c).

Approximately 139 species of birds associated with wetlands and bay habitats have been reported to use the Galveston Bay area. The Galveston Bay system is important breeding habitat for three species of waterfowl: *Dendrocygna bicolor* (fulvous whistling duck), *Anas fulvigula* (mottled duck), and *Aix sponsa* (wood duck). Another 29 species of waterfowl use the bay as an overwintering site or migration flyway. Large populations of migrating and overwintering shore birds make use of the Galveston Bay area, especially during winter and spring months. Of the 35 species of shorebirds reported to use the area, several are known to nest in the bay complex, including: *Charadrius wilsonia* (Wilson's plover), *Charadrius vociferus* (killdeer), *Catoptrophorus semipalmatus* (willet), *Haematopus palliatus* (American oystercatcher), and *Himantopus mexicanus* (black-necked stilt). About 22 species of colonial waterbirds have been reported nesting during the 21 years of surveys. The three most common species during the 1986 season were: *Larus atricilla* (laughing gull), *Sterna maxima* (royal tern), and *Bubulcus ibis* (cattle egret). From 1973 to 1987, numbers of pairs of colonial nesting waterbirds varied from lows of approximately 39,000 in 1978 and 1985 to a high of 71,700 in 1982, with a mean of 52,136. Active colony numbers have increased from 20 in 1973 to 42 in 1987 (NOAA, 1989c).

Pulich and White (1990) estimated acreage of seagrass beds in Galveston Bay system from 1987 NASA aerial photographs. Those measurements were corroborated with field surveys in 1988-1989. Historic black and white photomosaics from 1956 Edgar Tobin aerial surveys, 1965 black and white photographs from the U.S. Coast and Geodetic Survey, and 1975 color-infrared photographs from NASA were compared with recent photographs to assess changes in acreage. Both upper and lower Galveston Bay were studied. Since 1979, total acreage of seagrasses in Galveston Bay has declined approximately 90%. The extent of seagrass decline was especially noticeable in the western half of the Bay (West Bay) where there were 1,131 ac (458 ha) in 1956, 288 ac (117 ha) in 1965, 91 ac (37 ha) in 1975, and 0 ac in 1987. Several physical and hydrographic processes contributed to these changes, including geomorphic impacts from Hurricane Carla, subsidence in the upper Bay, and effects from industrial and urban shoreline development in the lower Bay.

Shew et al. (1981) listed rare and endangered vertebrates of the Galveston Bay area. These include: *Canis rufus* (red wolf), which is considered extinct in Texas and Louisiana (other than for a few isolated sightings); *Lutra canadensis texensis* (river otter) which is threatened; *Ursus americanus* (black bear) which is endangered; *Trichechus manatus latirostris* (West Indian manatee) which is endangered; and *Lynx rufus texensis* (bobcat) which is under consideration for an endangered status. Endangered bird species found in Texas coastal areas include *Haliaetus leucocephalus leucocephalus* (southern bald eagle), *Grus americana* (whooping crane), *Pandion haliaetus carolinensis* (osprey), *Falco peregrinus tundrius* (Artic peregrine falcon), *Pelecanus occidentalis carolinensis* (brown pelican), *Tympanuchus cupido attwateri* (Attwater's greater prairie chicken), *Picoides borealis* (red-cockaded woodpecker), *Chen rossii* (Ross's goose), *Campephilus principalis* (ivory-billed woodpecker), and *Numenius borealis* (Eskimo curlew).

*Alligator mississippiensis* (American alligator) is listed as an endangered species in Galveston Bay; however, recent reports indicate that populations of this species are increasing. *Malaclemys terrapin littoralis* (Texas diamondback terrapin) and *Opheodrys vernalis blanchardi* (smooth green snake) are both listed as threatened on the list prepared by the Texas Organization for Endangered Species.

Florida

The Florida coastal zone has experienced significant historic losses of natural habitats. Approximately 75% of Florida's population lives in coastal counties and the desire to live on or near the coast has resulted in the loss of fringing wetland habitats through filling activities, loss of seagrass communities through pollution, dredging for boat channels and marinas, increased use of coastal waters, and a deterioration of water quality from point and nonpoint discharges. Coastal development has exacerbated beach erosion and has resulted in loss of sand dune habitat.

Although the sensitivity of coastal habitats is becoming better understood, the demands on Florida's coastal resources continue to increase. The movement of people to the Sun Belt is well documented (Culliton et al., 1990). It is estimated that approximately one thousand people move to Florida daily to establish permanent residence (Shoemyen et al., 1989). This rapidly growing population places increasing demands on shipping, agriculture, transportation, and other industries. Growth of these industries in response to population demands has resulted in increasing encroachment on natural habitats and resources. Approximately 31% of Florida's Gulf coast estuaries are severely affected by pollution. That amount is slightly less than the overall national average of estuarine pollution (Comp and Seaman, 1985). More and more developments are appearing on Florida's Gulf coast because the east coast of Florida is largely developed. Thus, the demands on Gulf coast estuaries will continue to increase with increasing development.

The marine coastline of Florida is 1,300 mi long (2,097 km), and the tidal shoreline is 8,500 mi (13,709 km) long. Approximately 22% of the U.S. Gulf coast acreage of estuaries and tidal wetlands occur on the west coast of Florida (3 million ac; 1.2 million ha) (Comp and Seaman, 1985). Florida's Gulf coast estuaries tend to be well mixed with broad salinity gradients caused by fresh water inflow. The Gulf coast has low relief, and many rivers, creeks, and springs discharge into estuarine habitats. The continental shelf along the Big Bend portion of Florida's Gulf coast (Tarpon Springs to Apalachicola) is wide and shallow which results in a low-energy shoreline in that portion of the coast. In general, the remainder of the coast is characterized by high energy beaches on barrier islands separated from the mainland by estuarine lagoons.

Tampa Bay

Tampa Bay is the largest estuary in Florida, covering an area of approximately 254,410 ac (1,030 $km^2$). It receives drainage from nine rivers and streams in a watershed that covers approximately 1,407,900 ac (5,700 $km^2$). Tampa Bay is subdivided into seven subunits: Old Tampa Bay, Hillsborough Bay, Middle Tampa Bay, Lower Tampa Bay, Boca Ciega Bay, Terra Ceia Bay, and the Manatee River. The origins of Tampa Bay are not clearly known; however, it is believed that it is a large, drowned floodplain of a subtropical river flowing from the Florida peninsula to the Gulf of Mexico. Tampa Bay is located in a transition zone between temperate and tropical climates. The Peninsula Arch to the east and the broad continental shelf to the west tend to protect the bay from oceanic influences (Lewis and Estevez, 1988).

Lewis and Estevez (1988) reviewed the physical and chemical properties of bay waters and concluded: 1) Tampa Bay is not grossly polluted; 2) some parts of the bay are cleaner than others for natural as well as cultural reasons; 3) the levels of some pollutants have declined over the past decade while others have increased; and, 4) the overall quality of bay zones is the same whether judged by ecological or human-use criteria. The significant environmental problems in Tampa Bay include low dissolved oxygen which often results in fish kills, reduction of light to lower levels of the water column, which decreases primary production from phytoplankton and seagrasses, and excessive nutrients in runoff and discharges from sewage treatment plants, other point sources, and nonpoint sources which can cause blooms of phytoplankton and algae.

According to Lewis et al. (1985a), four species of seagrass are found in Tampa Bay: *T. testudinum*, *S. filiforme*, *H. wrightii*, and *H. engelmannii*. *Ruppia maritima* is also reported in Tampa Bay. These species make up seagrass meadows covering 20,000 ac (8,097 ha) of the bay bottoms (Lewis et al., 1985; Lewis et al., 1991). Estimations from comparing historical aerial photographs and maps suggest seagrasses at one time covered 76,527 ac (30,980 ha) of Tampa Bay. It is probable that this loss of 73% (56,527 ac; 22,885 ha) of seagrasses has adversely affected the fisheries resources of Tampa Bay. There is recent evidence that seagrass systems are recovering in Tampa Bay. However, it is still uncertain whether the observed expansion of seagrass meadows is short-term, due to drought condition, or long-term, due to improved water quality.

Lewis et al. (1985) found that the seagrass meadows are generally monospecific stands. Approximately 40% of the remaining beds are vegetated by *T. testudinum*, 35% by *H. wrightii*, 15% by *S. filiforme*, and 10% by *R. maritima*. Five types of seagrass meadows occur in Tampa Bay: 1) mid-bay shoal perennial; 2) healthy fringe perennial; 3) stressed fringe perennial; 4) ephemeral; and 5) colonizing perennial. It is hypothesized that types 3 and 4 are stages that lead to the eventual disappearance of a seagrass meadow due to human activities.

Approximately 17,791 ac (7,200 ha) of emergent wetlands border Tampa Bay and consist of five major plant species: *J. roemerianus, S. alterniflora, L. racemosa, R. mangle*, and *A. germinans* (Lewis and Whitman, 1985). Mangrove forests suffered significant losses during three recent freezes: January 1977, December 1983, and December 1989. Because of these freezes, the total area of tidal marsh may have increased as more temperature-tolerant marsh plants colonized the dead mangrove forests.

An estimation of the annual production of primary producers in the Tampa Bay area is given in Table 30. Phytoplankton are responsible for the majority of primary production in this system.

**Table 30. Annual Production of Primary Producers in Tampa Bay (from Lewis and Estevez, 1988)[1]**

| Primary Producer | Production ($gC/m^2/yr$) | Area (Acres) | Total Production ($gC/yr \times 10^6$) | Percentage of Total |
|---|---|---|---|---|
| Seagrasses & Epiphytes | 730 | 14,208 | 42.0 | 8.5 |
| Macroalgae | 70 | 24,710 | 7.0 | 1.4 |
| Benthic Microalgae | 150 | 49,568 | 30.0 | 6.0 |
| Mangrove Forests | 1,132[2] | 15,938[3] | 73.0 | 14.7 |
| Tidal Marshes | 300 | 2,595 | 3.2 | 0.6 |
| Phytoplankton[4] | 340 | 213,494 | 293.8 | 59.1 |
| Phytoplankton[5] | 50 | 23,722 | 48.0 | 9.7 |

[1] Acreage originally recorded in $km^2$
[2] Estevez and Mosura, 1985.
[3] Assumes 14% of the bay's emergent wetlands are tidal marsh.
[4] For areas deeper than 2 m.
[5] For areas shallower than 2 m.

Unvegetated bottoms are a major component of the Tampa Bay ecosystem. Approximately 74% (228,067 ac; 92,335 ha) of the subtidal bottom of Tampa Bay is unvegetated (Haddad, 1989). This habitat includes artificial reefs, natural rock reefs, sand, and mud bottoms.

Changes in the community structure of Tampa Bay between 1950 and 1982 were estimated during a cooperative study between the U.S. FWS and the Florida Department of Natural Resources (Table 31). A significant decline in vegetated habitats, particularly seagrasses, was documented.

Increased urban and agricultural runoff probably led to this decline due to increased turbidity and nutrient loading.

**Table 31. Summary of Major Habitat Trends In Acres for the Tampa Bay Region (from Haddad, 1989)**

| Habitat | 1950 | 1982 | Percentage of Change |
|---|---|---|---|
| Mangrove | 21,314 | 19,839 | -7 |
| Salt marsh | 5,096 | 3,537 | -30 |
| Seagrasses | 63,728 | 32,030 | -50 |
| Mudflats | 16,825 | 23,190 | +37 |
| Agriculture | 62,607 | 111,626 | +78 |
| Urban | 80,843 | 236,097 | +192 |

Lewis (1977) and Lewis et al. (1985a) estimated that 44% of the salt marsh and 81% of the seagrass meadows in Tampa Bay have been lost since the 1800's. The recent calculations of Haddad (1989; Table 31) are not readily comparable with those of Lewis et al. because of differences in time, methods of calculation, vegetation classification, and other factors. However, the results of both studies confirm that significant losses in habitat have occurred. Haddad suggests that the small percentage of change in mangroves listed in Table 31 is an artifact of the classification system used in his study which is currently being corrected.

Significant losses of habitats have occurred in the Tampa Bay area. Dredge and fill activities have caused direct losses of mangroves, marshes, and seagrasses by uprooting or covering the habitat and indirect losses in seagrasses by raising the turbidity and lowering the quality of the water. Seagrass losses have occurred throughout the Tampa Bay, particularly in the upper portion where shallow seagrass meadows were dredged and filled for residential and commercial development. Although historic loss of seagrasses was partially due to direct mechanical destruction from dredging and filling, most loss was due to deterioration of water quality. The causes of this deterioration include: 1) loss of adjacent rangeland, forests, swamps, and marshes that normally filter runoff water; 2) increases in agricultural area that may increase sedimentation and suspended particulates; 3) increase in urbanization and industrialization that introduces wastewaters and stormwater into the bay; and 4) dredging that causes long-term release of fine sediments into the bay (Haddad, 1989).

Fisheries have historically been an important part of the Tampa Bay ecosystem (Lewis and Estevez, 1988). The production of mullet, blue crabs, hard shell clams, tarpon, snook, and spotted seatrout are an important contribution to the local economy. Declines in habitat such as seagrass meadows, mangrove forests, and tidal marshes that furnish food and protection for many of these species have apparently resulted in declines in fishery production. Total emergent wetlands (tidal marshes and mangrove forests) have declined from 24,831 ac (10,053 ha) (ca. 1876) to 13,906 ac (5,630 ha) (ca 1976), a 44% loss (Lewis, 1977). This loss was mainly due to dredge and fill operations. Commercial fishery landings of species that depend on wetland habitats have decreased concurrently with wetland losses in Tampa Bay. Fishery landings in Tampa Bay are low compared to those in Charlotte Harbor, a nearby estuary of similar size but with much less habitat loss.

U.S. FWS and Tampa Port Authority recently completed a three-year cooperative project to establish a data base available for management of wildlife habitat and port development in Tampa Bay (Fehring, 1986). This project identified management and mitigation options, developed an information base, and defined long-range management scenarios for port development and mitigation. The study produced a list of acreages of wetland habitats for 21 U.S. Geological Survey quadrangles in the Tampa Bay region. Estuarine vascular aquatics (seagrasses) suffered the greatest losses, but salt marshes and mangroves also declined significantly (Table 32).

**Table 32. Area (Acres) of Various Wetland Habitats for 21 Quadrangles in the Tampa Bay Region for Three Time Periods (Modified from Fehring, 1986)**

| Habitat Type | 1950 Acres | 1972 Acres | 1982 Acres | 1948-1982 Change | % Change |
|---|---|---|---|---|---|
| Salt marsh | 4,672 | 3,610 | 3,238 | -1,434 | -30.7 |
| Estuarine vascular aquatics | 40,324 | 25,576 | 22,526 | -17,798 | -44.1 |
| Mangroves | 18,645 | 18,656 | 18,030 | -615 | -3.3 |
| Estuarine open water | 206,740 | 214,043 | 212,623 | +5,883 | +2.8 |
| Beaches, bars and flats | 18,771 | 16,369 | 19,825 | +1,654 | +8.8 |

The impact of approximately 100 years of dredging activities on the Tampa Bay estuary is discussed by Lewis (1977). Channel deepening, maintenance dredging, shell dredging, and dredging for fill material for construction are the four major types of dredging activities which have impacted the Bay. Over 12,355 ac (5,000 ha) of Tampa Bay have been converted to uplands for residential, commercial, and dredged material disposal use. According to Lewis (1977), disposal of dredged material resulted in most of the loss of 44 % of the original marine wetlands which bordered Tampa Bay. Little research has been performed on the effects of channel dredging and construction of open water spoil disposal sites on the hydrological patterns in Tampa Bay. However, it is known that recovery of disturbed seagrass meadows is extremely slow after dredging (Godcharles, 1971).

Open water spoil disposal can result in changes in hydrological patterns in enclosed water bodies. The Federal Water Pollution Control Administration recognized this problem and recommended in 1969 that a master plan be developed by pertinent Federal, state, and local operating and regulating agencies for dredging and filling in Hillsborough Bay (a part of Tampa Bay). However, there is no master plan for the entire Bay, and historic dredging and filling in Hillsborough Bay has occurred with little regard for the master plan. Studies undertaken by the U.S. Geological Survey, funded by the Tampa Port Authority and the Corps of Engineers, evaluated a proposed Harbor Deepening Project in 1970. Results of that modeling effort concluded that improved flushing in Hillsborough Bay is probably not possible due to size and orientation of existing channels and spoil disposal sites (Lewis, 1977).

In 1967, filled areas in Tampa Bay totaled 10,537 ac (4,266 ha). Filling created uplands where there were once mangroves, tidal marshes, or seagrass communities (McNulty, Lindall, and Sykes, 1972). The filling of 3,458 ac (1,400 ha) of bay bottom in Boca Ciega Bay since 1950 has reduced the total area of that bay by 20%. That study estimated a loss of 28,425 t (25,841 mt) of infauna and an annual loss of $1.4 million (1972 dollars) in fisheries production due to filling.

Lewis (1977) suggested that a general increase in public environmental awareness and the declining availability of marine resources make it unlikely that massive dredge and fill projects, such as those which have historically despoiled Tampa Bay, will occur in the future. Unfortunately, loss of wetland habitat and pollution in Tampa Bay have already decreased commercial harvest of fish and shellfish and adversely affected populations of other organisms dependent upon the Bay for survival.

Habitat losses, similar to those which have occurred in Tampa Bay, have been observed in Sarasota Bay (Table 33). These losses are a symptom of the rapid development of coastal Florida.

Charlotte Harbor

Haddad (1986) compared the loss of habitat in Charlotte Harbor with that observed in Tampa Bay. Charlotte Harbor is on the southwest coast of Florida and is one of the State's least modified and least polluted estuaries. Its dimensions are approximately 35 mi (56 km) from north to south, encompassing 227,332 ac (92,000 ha) of water area, with an average depth of 10.5 ft (3.2 m). The average tidal range is 5.9 ft (1.8 m). The harbor is used extensively by sport and commercial

**Table 33. Changes in Habitat Acreages in Sarasota Bay (Gail McGarry, Florida Department of Natural Resources, Personal Communication)**

| Category | 1948 | 1972 | 1987 | % Change (1948-1987) |
|---|---|---|---|---|
| Seagrass (sparse-med. density) | 930 | 860 | 845 | -9 |
| Seagrass (dense) | 4,814 | 4,286 | 2,650 | -45 |
| Seagrass (patchy) | 441 | 71 | 544 | +23 |
| TOTAL SEAGRASS | 6,185 | 521 | 4,039 | -35 |
| Mangrove swamp | 1,546 | 1,032 | 843 | -45 |
| Tidal marshes | 271 | 44 | 48 | -83 |
| Oyster beds | 109 | 115 | 129 | +16 |
| Bay waters | 12,220 | 1,707 | 13,802 | +12 |

fishermen and provides over 50% of Florida's west coast commercial landings of red drum and spotted seatrout. The Myakka, Peace, and Caloosahatchee Rivers, with a combined drainage basin of approximately 2,668,680 ac (1.08 million ha), drain into Charlotte Harbor. These watersheds include pasture, farmland, and citrus groves. Industrial effluents from phosphate mining areas, light industry, and domestic wastes are discharged into these rivers.

Table 34 summarizes the cumulative effects of various perturbations on marine wetland habitats in Charlotte Harbor. Except for an increase in mangrove habitat, there have been declines in all natural habitat types. This trend is similar to the trends observed in Tampa Bay and Sarasota Bay. Losses of salt marshes and mudflat habitats are particularly significant in Charlotte Harbor.

**Table 34. Wetland Acreage Change in Charlotte Harbor from 1945-1982 (data from Harris et al., 1983, as presented by Haddad, 1986)**

| | Acreage | | |
|---|---|---|---|
| Wetland Habitat | 1945 | 1982 | % Change |
| Seagrasses | 82,959 | 58,495 | -29 |
| Mangroves | 51,524 | 56,631 | +10 |
| Salt marsh | 7,251 | 3,547 | -51 |
| Mudflats | 11,206 | 2,723 | -76 |
| Oyster reefs | 806 | 488 | -39 |

The acreage of mangrove habitat increased by 10% during the study period. A portion of this increase may be due to differences in photographic resolution and interpretation, but the majority of this increase is probably due to management practices that began in the late 1960's. An important part of this plan was the establishment of a wetland "buffer" zone around the harbor, and existing mangroves were protected as part of the estuarine system. Thus, mangroves were allowed to grow into mudflats and oyster reefs, and very few were lost due to direct removal for development. The concept of establishing a preserved wetland fringe around the harbor was successful in maintaining the mangrove community, but the concurrent loss of seagrass beds and salt marshes substantiates the need for managing the entire ecosystem, including the drainage basin. Salt marshes decreased by 51%, probably due to dredge and fill activities, as well as from secondary impacts of extensive upland development. This development included construction of canals that may have permitted intrusion of saltwater, which resulted in increased growth of mangroves. Seagrasses declined almost 30%. Most of this decline was in an area where a coastal highway was constructed which required extensive dredging and channelization of a river. Some of the decline was probably due to the decrease in light penetration from turbidity produced by dredging and channelization. Loss of seagrass habitat adversely affected fisheries resources, particularly bay scallops which historically inhabited dense grass beds (Haddad, 1986).

During the period of this study (1945-1982), the urban area around Charlotte Harbor increased by approximately 92,395 ac (37,400 ha). Approximately 50% of this increase was due to development of large tracts of agriculture and rangeland.

In summary, although Charlotte Harbor is relatively unimpacted compared to other estuaries of Florida, the area is undergoing rapid growth, and water and sediment quality are showing signs of deterioration. Haddad (1986) listed the major lessons learned from monitoring the deterioration of Charlotte Harbor:

1. Although dredging and filling can have a catastrophic direct effect, the long-term effects from the release of fine sediment into the water column may be even more damaging.
2. The densities of phytoplankton may be significantly increased through the introduction of nutrients in runoff and effluent discharges.
3. Alterations in natural drainage patterns can have an effect on ambient water quality and the life cycle of biota in the estuary.
4. Altered circulation patterns resulting from the construction of roads, dikes, channels, etc., and the channelization of a contributing river also interact with water quality and cycles of biota.
5. Alterations of drainage affect the habitat and survival of scallops within an estuary.

## Louisiana

Gosselink (1984) reviewed the ecological information on the extensive marshes of the Mississippi River Deltaic Plain (MRDP). Over the last 6,000 years, the Mississippi River has built a delta onto the continental shelf of the Gulf of Mexico which measures approximately 5,903,300 ac (23,900 $km^2$). The MRDP is the largest continuous wetland system in the United States with approximately 1,790,750 ac (725,000 ha) of wetlands, not including forested wetlands. These marshes represent about 22% of the total coastal wetland area of the 48 continental United States. The Mississippi River system is the largest in the United States, draining an area of over 826,440,776 ac (3,346,000 $km^2$).

A community profile of the area was developed using the 1978 FWS habitat mapping data. At that time, the MRDP consisted of 9,000,000 ac ($3.6 \times 10^6$ ha) in the following categories: 58% open water, 5.6% urban and agricultural, 32% wetlands and 2.1% dredged canals and spoil banks. Wetlands were categorized into salt marsh (15%), brackish marsh (47%), intermediate marsh (23%), and seagrasses (1%).

The MRDP is composed of nine drainage basins, which are the result of shifts in the major tributaries over time. The youngest basin is the Atchafalaya, which is actively prograding out through the shallow Atchafalaya Bay. It receives approximately one-third of the flow of the combined Mississippi and Red River systems; this interior basin is predominantly fresh all year long.

The next youngest basin in the active Mississippi River delta is the Balize Delta. It receives almost two-thirds of the flow of the Mississippi River, emptying into deep water at the edge of the continental shelf. The majority of these areas are fresh water with some brackish marsh. In succession, Barataria, Terrebonne, Vermilion-Cote Blanche, and the Pontchartrain-Lake Borgne basins are of increasing age. They all have extensive marshes with well-developed salt and brackish zones.

Wetland losses in the Mississippi River Deltaic Plain have been studied by Turner (1990), Gosselink (1984), Bahr et al., (1983), and Leibowitz and Hill (in preparation). Turner (1990) found that Louisiana coastal zone habitats are converted to open waters and at an annual rate of 0.86% Gosselink (1984) estimated the annual loss of delta marshes to be 1.5% Leibowitz and Hill (in preparation) reported that annual land loss rates at Cameron, Terrebonn, and Lafourch study sites were 0.77, 0.66, and 0.95%, respectively. Regardless of the discrepancies in rate, all authors agree that the rate of marsh loss to open water has accelerated over the past 50 years and that human activities have interfered with the cycle of delta formation and accretion. Dunbar et al. (1990) report that the current land loss rate is approximately 31 square miles (80 $km^2$) per year.

Causes and mechanisms of changes in wetland acreage include: direct loss to dredging, construction, filling, erosion and machinery (marsh buggies); subsidence from loss of accretion, oil and gas withdrawal, and soil drying; changes in quantity and quality of vegetation from changes in organic deposition and sediment trapping; and death of plants from pollutants. Turner (1990) suggested that the majority of wetland losses are due to drowning of plants under increased water depth.

The detrimental impacts of human activities on the MRDP have been cumulative. Evidence of prehistoric Indian use of the area is indicated by the numerous elevated shell mounds that currently harbor groves of trees. Permanent settlements in the delta appeared in the early 1700's. The bases of the area's economy were agriculture, fishing, and trapping. Since the 1940's, man's impact on the delta has greatly accelerated. For example, thousands of miles of canals have been dredged throughout the delta to facilitate activities of the oil and gas industry.

Gosselink (1984) also listed reasons for the loss of marshes in the MRDP. The development of the river delta is a dynamic process that involves both accretion and erosion of sediments. Subsidence in marsh areas is usually not a significant concern because of continued deposition of new sediments which result in a net landbuilding process. However, up-river dams selectively remove the coarser sediments resulting in less material available for the natural delta-building processes. Gosselink argues that extensive navigation canals have allowed intrusion of saltwater deep into the estuary, which often results in the death of salt-intolerant vegetation. Plant death results in erosion of land before more tolerant plant species can be established. Dredged canals also act as a direct route out of the marsh for finer sediments which historically were deposited and accreted in wetland areas.

Many of the remaining marshes in the delta have been managed in an attempt to increase fisheries production. Impoundments yield excellent crops of fish and shellfish. Juveniles are trapped within an impounded marsh and grow within the sheltered waters (Davidson and Chabreck, 1983). However, impoundments disrupt normal hydrological cycles and sediment deposition patterns and can lead to conversion of emergent marsh to open water.

Leibowitz and Hill (in preparation) observed that marsh loss is not a spatially uniform process that takes place at the shoreline and advances landward. Rather, losses are greatest within the marshes proper, and areas of greatest loss are highly clustered. They concluded that the pattern and causes of coastal marsh losses in Louisiana are both complex and heterogeneous.

Bahr et al. (1983) described 20 habitat types in the MRDP and grouped them into three broad categories: (1) highland or terrestrial habitats, (2) intertidal or periodically flooded habitats, and (3) aquatic or continuously flooded habitats. Bahr et al. (1983) estimated the area of estuarine open-water habitats in 1978 in the MRDP to be 4.7 x $10^6$ ac (1.9 X $10^6$ ha) or 56% of the region. The area of estuarine open water has been increasing because sediments from the Mississippi River, through the Atchafalaya River, are deposited into the Atchafalaya basin rather than on fringes of the marsh. Thus, there is a net reduction of MRDP land. Brackish marsh is the second largest habitat in areal extent in the MRDP, covering 997,880 ac (404,000 ha) or 11.7% of the entire area. Salt marsh makes up a significant portion of the MRDP: 449,540 ac (182,000 ha), or more than 5% of the total area in 1978. There are 17 other distinct habitat types in the MRDP that are discussed by Bahr et al. that each cover less than 5% of the area.

Morgan and Morgan (1977) determined the rate of shoreline changes throughout coastal Louisiana through a comparison of 1969 aerial photographs with those of two earlier periods (1954 and 1932). The comparison revealed that areas of land loss have far exceeded areas of new land formation. They concluded that the dominant cause of land loss was the erosion by waves of soft, unconsolidated sediments that compose the low-lying coastal zone, especially in the MRDP.

Linear shoreline retreat rates and areal land changes were evaluated to establish trends which document continuing and accelerating rates of land loss along most of coastal Louisiana. For example, Louisiana's shoreline has been retreating at an average linear rate of approximately 10 ft/yr (3 m) for the past 37 years. However, that average reflects an increase from about 6.5 ft/yr (2 m) in the interval between 1932 and 1954 to 17 ft/yr (5 m) during the period from 1954 to 1969. Similarly, areal change measurements for 75% of the coast, where they could be made, show a loss of about 374 ac (151 ha) per year (0.58 $mi^2$/yr; 1.5 $km^2$) during the 37-year period 1932-1969. However, there has been an increase in rate of land loss from 348 ac (141 ha) per year during the period 1932-1954 to 413 ac (167 ha) per year from 1954 to 1969.

Atchafalaya Basin

The U.S. Army COE is evaluating the potential ecological impact of flood waters in the Atchafalaya Basin Floodway system that extends from the proximity of Old River, at the juncture of the Red and Mississippi Rivers, to the Gulf of Mexico (Mathies, U.S. Army COE, personal communication). The main purpose of the Atchafalaya Basin project is to safely convey one-half of flood waters to the Gulf of Mexico. Operation of the Federal flood control project has induced flooding in areas to the east and northeast of the end of the existing levee system due to backwater flooding. Increased and prolonged flooding is destroying extensive areas of wetland habitats. Two alternative plans are being evaluated to provide protection against this increased flooding. Potentially, the most environmentally damaging alternative is construction of a 5.5 mi (8.9 km) extension of the existing Avoca Island levee. The levee extension would adversely impact the Terrebone marsh complex, a 600,000 ac (242,915 ha) wetland area located to the east of the existing levee, by restricting sediment-laden river flows from entering the area. By limiting river flows, the vital nutrients and sediments carried by those flows and essential to nourish the marsh complex would be limited.

To assess the ecological consequences of the levee extension alternative, the U.S. Army COE recently provided funds to Louisiana State University, through the U.S. FWS, to develop a spatial simulation model for the study area to predict ecological changes. The resulting Coastal Ecological Landscape Spatial Simulation model (Costanza, et al., 1990) is comprised of nearly 2,500 interconnecting 1-square kilometer cells. Each cell is characterized by its own set of variables and is connected to its four adjacent cells by the exchange of water and suspended materials. During the model verification process, predicted conditions in 1978 and 1983 were evaluated against actual conditions at those time periods. Degree of fit was calculated relative to each of the simulations. Numerous management scenarios, both with and without the levee extension, were evaluated and future conditions predicted. The resulting 50-year predictions indicate that the levee extension would induce the conversion of approximately 1,200 ac (486 ha) of existing marsh to open water habitat. Several mitigation options have been evaluated to offset environmental impacts and provide environmental compensation. However, selection of the flood control alternative has not been made at this time. Information on historic acreage of various habitats in Atchafalaya Basin is shown in Table 35.

**Table 35. Acres (x 100) of Specific Habitats in the Atchafalaya Basin (After Costanzo, et al., 1990)[1].**

| Year | Swamp | Brackish Marsh | Saline Marsh | Open Water |
|---|---|---|---|---|
| 1956 | 321.23 | 1561.67 | 242.16 | 1833.48 |
| 1978 | 279.22 | 1368.93 | 370.65 | 2169.54 |
| 1983 | 286.64 | 857.44 | 383.01 | 2466.06 |

[1] Calculations from FWS Natural Wetland Inventory Maps and Photographs.

## Barataria Basin

The Barataria Basin is roughly triangular in shape, and is approximately 68 mi (110 km) long and 31 mi (50 km) wide where it meets the Gulf of Mexico. The basin has been closed to river flow since the leveeing of the Mississippi River in the 1930's and 1940's and the closing of Bayou Lafourche, the Mississippi River connection, in 1902. Precipitation is the main source of fresh water for the basin (Conner and Day, 1987).

Information on wetland losses in Barataria Bay is quantified in a computerized Geographical Information System (GIS) (Johnston et al., 1986). They conclude that maintenance of marsh areas continues to depend upon availability of nutrients and sediments from riverine inputs and that human activities have greatly modified the input of both to Barataria Bay. Construction of flood control levees along the Mississippi River and Bayou Lafourche has severely reduced fresh water inflow and riverborne sediment into Barataria Bay. The GIS aided in identification of three problem areas: Cut-off Golden Meadow Area, Lafitte oil field, and Adams Bay. Based on National Wetland Inventory wetland maps digitized from 1956 and 1978 photographs, the loss rate of marsh and other wetland habitat was different for each of the problem areas. However, for the entire Bay, from 1956 to 1978, wetland area decreased an average 12% and open water increased by 11% Marsh acreage alone decreased from 548,356 ac (222,000 ha) in 1956 to 410,586 ac (166,229 ha) in 1978, a decrease of 25%.

In another study, computer analysis was used to compare land and water areas of Barataria Bay by comparing aerial photographs taken in 1945, 1956, 1969, and 1980. This study concluded that the rate of marsh loss has increased from 0.36% per year in the 1945-56 period, to 1.03% per year in 1956-69, and to 1.96% per year in 1969-80 (Sasser et al., 1986).

Patterns of marsh loss do not occur uniformly. Marsh loss rates have been highest where fresh water marshes have been subjected to saltwater intrusion. The increase in wetland loss rates corresponds to accelerated rates of subsidence and canal dredging and a cumulative increase in area of canals and spoil deposits. The rate of change of marsh acreage from 1945 to 1980 is summarized in Tables 36 and 37.

Conner and Day (1987) described the ecology and loss of wetlands in the Barataria Basin. This area is noted for its network of interconnecting water bodies that allow transport of water and migrating organisms throughout the basin. High ground within the wetlands is occupied by natural and artificial levees that are surrounded by extensive swamp forests and fresh, brackish, and salt marshes. These productive wetlands provide valuable nursery habitat for fish, shellfish, wintering

Table 36. **Net Areal Change (Acres) and Average Annual Rates of Change of Marsh to Nonmarsh in Barataria Basin, Louisiana (from Sasser et al., 1986).**

| Type of Change | Area (Acres) | Rate (Acres/yr) | Change (% marsh) | Change Rate (%/yr) |
|---|---|---|---|---|
| Marsh to nonmarsh (total change) | | | | |
| 1945-1956[a] | 6,175 | 563 | 3.92 | 0.36 |
| 1956-1969[b] | 20,299 | 1,562 | 13.39 | 1.03 |
| 1969-1980[c] | 28,315 | 2,575 | 21.57 | 1.96 |
| 1945-1980[a] | 54,083 | - | 34.29 | - |
| Marsh to canal/spoil | | | | |
| 1945-1956[a] | 2,335 | 213 | 1.48 | 0.13 |
| 1956-1969[b] | 8,784 | 675 | 5.80 | 0.45 |
| 1969-1980[c] | 1,520 | 138 | -1.16 | -0.11 |
| 1945-1980[a] | 9,600 | - | 6.09 | - |
| Marsh to developed[d] | | | | |
| 1945-1956[a] | 378 | 35 | 0.24 | 0.02 |
| 1956-1969[b] | 4,599 | 353 | 3.03 | 0.23 |
| 1969-1980[c] | 7,976 | 724 | 6.08 | 0.55 |
| 1945-1980[a] | 12,953 | - | 8.21 | - |
| Marsh to water (undetermined causes) | | | | |
| 1945-1956[a] | 3,462 | 314 | 2.20 | 0.20 |
| 1956-1969[b] | 6,281 | 484 | 4.14 | 0.32 |
| 1969-1980[c] | 21,537 | 1,957 | 16.41 | 1.49 |
| 1945-1980[a] | 31,530 | - | 19.99 | - |

$^{a}\,\% = \dfrac{\text{area of change}}{\text{area of marsh 1945 (157,719 acres)}}$

$^{b}\,\% = \dfrac{\text{area of change}}{\text{area of marsh 1956 (151,537 acres)}}$

$^{c}\,\% = \dfrac{\text{area of change}}{\text{area of marsh 1969 (131,255 acres)}}$

[d] Developed includes agricultural, industrial, and residential areas

Table 37. **Marsh Area (Acres) in 1945, 1956, 1969, and 1980 and Rates of Change During 1945-56, 1956-69, and 1969-80 in Barataria Basin Marshes (from Sasser et al., 1986)[1].**

| Category | Marsh area by year | | | | 1945-56 rate of change (Acre/yr) | 1956-69 rate of change (Acre/yr) | 1969-80 rate of change (Acre/yr) |
|---|---|---|---|---|---|---|---|
| | 1945 (Acres) | 1956 (Acres) | 1969 (Acres) | 1980 (Acres) | | | |
| Marsh (<10% water) | 13,055 | 116,844 | 60,873 | 29,444 | -1,245 | -4,304 | -2,856 |
| Marsh (10%-25% water) | 11,327 | 18,266 | 44,359 | 36,331 | 306 | 2,006 | -729 |
| Marsh (25%-40% water) | 2,656 | 5,691 | 9,182 | 18,873 | 274 | 30 | 882 |
| Marsh (40%-60% water) | 282 | 628 | 5,970 | 10,561 | 32 | 410 | 418 |
| Marsh (60%-80% water) | 655 | 843 | 378 | 6,783 | 17 | -35 | 583 |
| Water | 69,430 | 72,699 | 79,786 | 94,565 | 297 | | 1344 |
| Developed | 4,762 | 5,140 | 9,738 | 1,775 | 32 | 546 | 724 |
| Natural levee | 12,901 | 10,111 | 10,870 | 7,732 | -257 | 59 | 284 |
| Canal[2] | 974 | 3,311 | 13,093 | 10,573 | 210 | 675 | -138 |
| Other[3] | 1,819 | 1,567 | 2,286 | 2,226[4] | ---[4] | ---[4] | 27[5] |

1 Acreage reported as $km^2$ by Sasser et al., 1986 ($km^2$ x 247.1 = acres).
2 Because canals and their associated spoil deposits are narrow, linear features, these numbers contain an undetermined error (see *Materials and Methods, Classification Scheme*).
3 This category includes Bayou Lafourche, the beach, and Louisiana Highway 1. Because these are narrow, linear features, the numbers may not closely approximate the actual areas of the features and are included only as an index of relative value.
4 The 1980 datum for Louisiana Highway 1 was not generated and is not included in this entry.
5 This entry represents only Bayou Lafourche since the beach and Louisiana Highway 1 represent <0.01% of the study area.

waterfowl, and fur-bearers. The basin is a dynamic system and is undergoing changes in the amount of open water and wetland habitats due to subsidence, erosion, and human activities. The latter have altered the natural hydrologic patterns in the basin, and this may cause long-term modifications of wetland habitat.

Conner and Day (1987) divided the Barataria Basin into five environmental zones: levee and developed lands, swamp forest, fresh marsh, brackish marsh, and salt marsh. The brackish marsh is found between the lower swamp and upper marine environments and covers 338,390 ac (137,000 ha) (22% of the basin). This is the upper limit of the effects of tides and storm surges. Salinity ranges from 2 to 10 ppt. It is the inland flow of saltwater that determines the kinds of vegetation that grow, aids in nutrient recycling, and allows the inland migration of larval estuarine species. Conner and Day (1987) found that *S. patens* is the dominant plant in the brackish marsh zone. Chabreck (1972) observed the predominant composition of brackish marsh plants in the Basin was *S. patens* (44%), *Distichlis spicata* (16%), *Bacopa monnieri* (12%), *Pluchea camphorata* (8.4%), and *S. alterniflora* (4.5%).

Barataria Bay has approximately 145,000 ac (58,704 ha) of salt marsh habitat. Salinity ranges from 6 to 22 ppt. The major plant species is *S. alterniflora*, which covers 63% of the salt marsh area (Conner and Day, 1987). Chabreck (1972) observed the salt marsh community to consist of the following species: *S. alterniflora* (63%), *J. roemerianus* (15%), *D. spicata* (10.1%), *S. patens* (7.8%), and *Batis maritima* (3.1%).

Several studies have demonstrated that alteration of normal hydrologic conditions can affect structure and productivity of herbaceous and wooded wetlands. Mendelssohn et al. (1982) report that hydrologic modifications of salt marshes cause increased waterlogging which may affect plant productivity. For example, the Leeville oil field lies on the western boundary of the Barataria Basin in a *S. alterniflora* marsh and consists of a dense network of canals dug for access to drilling sites. Spoil disposal levees line many of the canals. Allen (1975) estimated that standing live *Spartina* biomass was 50% lower in marshes surrounded by spoil banks than in a comparable control site. Spoil banks along the canals also contain canal flow and restrict the natural lateral transport of water necessary for the removal of wastes, the replenishment of marsh sediments, and the import of new nutrients to the swamp forest. Obstruction of overbank flooding by levees can result in nutrient starvation and blocking of swamp substrate accretion.

Northern Barataria Bay has experienced an annual increase in salinity of 0.108 ppt from 1961-1974. Ultimately, the entire basin may become an open-water brackish bay or sound unless Mississippi River flow is reintroduced to the area. Intrusion of saltwater as a result of diversion of fresh water has resulted in a decline of diverse swamp forests and an increase in monotypic marshes and open water habitat (Van Sickle et al., 1976). The total number of plant species identified for each wetland type is: swamp, 200+; fresh marsh, 154; brackish marsh, 23; and saline marsh, 25 (Conner et al., 1986). Loss of biodiversity has affected ecosystem functions of the area.

There are distinct differences between the primary aquatic production of the lower, more saline part of the basin and the upper freshwater zone. Water bodies in the upper basin have high levels of primary productivity that increase significantly in the summer and possess heterotrophic characteristics. The aquatic community in the lower basin is less productive and lacks a consistent seasonal trend (Conner and Day, 1987). The major categories of environmental impact within the Barataria Basin are wetland loss, eutrophication, saltwater intrusion, reduction of nursery grounds for fisheries, and introduction of toxic substances into wetlands.

There were 12,103 ac (4,900 ha) of natural oyster reefs in the basin in 1976 (Van Sickle et al., 1976). These reefs experience high mortality in the summer months due to the protozoan *Perkinsus marinus*, which is referred to colloquially as "Dermo." Increased salinity and temperature increase the vulnerability of oysters to this disease.

The fish community in the Barataria Basin exhibits the most diversity of any water body in Louisiana (186 species; 65 families). Because of the lack of freshwater input from river systems, conditions in lower Barataria Bay closely resemble those of the nearshore Gulf of Mexico. Thus, the fish community in that portion of the Basin is dominated by marine species. Studies of trawl-caught fishes by Barrett et al. (1978) and Chambers (1980) revealed the following 10 most abundant species: *A. mitchilli* (bay anchovy); *M. undulatus* (Atlantic croaker); *Chloroscombrus chrysurus* (Atlantic bumper); *B. patronus* (Gulf menhaden); *L. xanthurus* (spot); *Arius felis* (hardhead catfish); *C. arenarius* (sand seatrout); *Polydactylus octonemus* (Atlantic treadfin); *Anchoa hepsetus* (striped anchovy); and *Bagre marinus* (gafftopsail catfish).

Several species of macroinvertebrates that inhabit Barataria Bay are commercially important. Brown shrimp (*Penaeus aztecus*) and white shrimp (*Penaeus setiferus*) are harvested commercially in the Bay. Pink shrimp (*Penaeus duorarum*), seabobs (*Xiphopenaeus kroyeri*), and two species of *Trachypenaeus* can be significant components of the community during certain times of the year, but generally are not taken in sufficient quantity to be of commercial importance. The blue crab (*Callinectes sapidus*) is harvested commercially and recreationally. In many of the lower bay areas, a related species, *Callinectes similis*, is found in greater abundance than *C. sapidus*; however, *C. Similis* does not achieve a long enough size for commercial harvesting. The brief squid (*Lolliguncula brevis*) is a nektonic mollusc that is abundant within the Barataria basin (Conner and Day, 1987).

Craig and Day (1987) suggested that wetland losses in this system occur in three major ways: 1) wetlands become open water by erosion, subsidence, or dredging, 2) wetlands are covered and made into a terrestrial habitat, or 3) wetlands can be wholly or partly isolated by spoil banks. Predictions of net wetland losses include 39.4 $mi^2$ (102 $km^2$) (25,212 acres) annually in coastal Louisiana (Gagliano, 1981), with some estimates as high as 59.8 $mi^2$ (155 $km^2$) (38,265 ac) (Paul Templet, LSU Environmental Studies, personal comm., as cited by Craig and Day, 1987). Barrier island retreat in Louisiana represents a serious problem indicated by a rate of loss estimated as high as 160 ac (65 ha) per year. Shoreline retreat rates of 164 ft (50 m) per year have been reported by Mendelssohn et al. (1982).

An example of wetland loss directly related to human activities occurred in the 12-year period between 1962 and 1974, when 44,800 ac (18,138 ha) of wetlands in the Barataria basin were drained or converted to water. Agricultural impoundments and oil access canals accounted for the largest acreages (Adams et al., 1976). Craig and Day (1987) reported:

> "...that between 40% and 90% of the total land loss in coastal Louisiana can be attributed to canal construction, including canal-spoil area and cumulative losses (Craig et al., 1979b; Scaife et al., 1983). In the deltaic plain of Louisiana, canals and spoil banks are currently 8% of the marsh area compared to 2% in 1955; there was an increase of 35,943 ac (14,552 ha) of canals between 1955 and 1978 (Scaife et al., 1983). Barataria Basin had a 0.93%/yr direct loss of marsh due to canals for the period of 1955-78 (Scaife et al., 1983). Canals indirectly influence land loss rates by changing the hydrologic pattern of a marsh, such as blockage of sheet flow, which in turn lessens marsh productivity, quality, and the rate of accretion. Canals widen with time because of wave action and altered hydrologic patterns, and apparently the larger the canal, the faster it widens. Annual increases in canal width of 2% to 14% in Barataria basin have been documented, indicating doubling rates of 5 to 60 years (Craig et al., 1979b)."

Calcasieu Basin/Chenier Plain

Bahr et al. (1977) developed a conceptual model of the Chenier Plain coastal ecosystems in southeastern Texas and southwestern Louisiana. The model took into account spatial heterogeneity, ecological or functional complexities, time scale of events, and management needs. Use of this model determined that annual primary production in the Calcasieu Basin is $2.5 \times 10^6$ t ($2.3 \times 10^6$ mt). This produces an estimated 1,100 t (1,000 mt) of shrimp, 8,470 t (7,700 mt) menhaden, 5 million sport-fishing efforts, 222,000 dabbling ducks at peak density, and 49,000 fur pelts.

According to Bahr et al. (1977), 18,038 ac (7,300 ha) of natural marsh have been lost since 1953, 70% of it to open water and the rest to impounded marshes, agriculture, and urban habitats. This figure represents a loss of 26% of the total natural marsh area of the basin. The consequence in natural resources is estimated to be a loss of 286,000 t (260,000 mt) of organic production, 77,000 birds per year, 19,000 fur pelts, and a significant impact on the shrimp and menhaden fisheries (Table 38).

**Table 38. Change in Area (Acres) of Habitats in the Calcasieu Basin, Louisiana between 1953 and 1975 (Adapted from Bahr et al., 1977)**

| Habitat | Area (Acres) 1953 | Area (Acres) 1975 | Net Area Net % Change |
|---|---|---|---|
| Open Water | 68,813 | 101,203 | +32.0 |
| Nearshore Gulf | 99,840 | 99,443 | -0.4 |
| Natural marsh | 181,952 | 120,800 | -33.6 |

Other studies on the Chenier Plain were conducted by Gosselink et al. (1979), who observed much of the impact is a result of canals that allow saltwater intrusion to change the habitat from fresh swamp to brackish open water (Table 39).

**Table 39. Wetland Losses (Acres) by Category in the Chenier Plain Between 1952 and 1974 (after Gosselink et al., 1979)**

| Converted To | Area Acres | Area Percentage |
|---|---|---|
| Urban | 4,762 | 0.3 |
| Agricultural | 11,300 | 1.1 |
| Spoil | 13,269 | 1.3 |
| Impounded marsh | 99,438 | 10.0 |
| Aquatic or open water | 75,627 | 7.6 |
| TOTAL | 204,396 | 20.3 |

The major nonrenewable resources of the Chenier Plain are oil and gas (value of $438 million/year). Major renewable resources (1977 dollars) are agricultural production of rice and cattle ($26 million/year); commercial fishing and trapping ($12 million/year), and recreational fishing and hunting ($21 million/year). The adverse effects of the direct pressure on the recovery of oil and gas on renewable resources may have been too severe for the basins to recover naturally (Bahr et al., 1977).

The Louisiana Geological Survey and U.S. EPA (1987) discussed the need for a long-term plan to manage Louisiana's wetlands from losses due to the maintenance of shipping lanes, dredging of canals, flood control levees, and withdrawal of oil and gas. Wetlands of coastal Louisiana are being converted to open water at a rate of approximately 35 to 60 $mi^2$/yr (91-155 $km^2$/yr). If current trends continue, an ecosystem that supports 33% of the nation's fishing industry and North America's largest fur-producing area will become extinct. An acceleration of this trend (Table 40) is possible if the predicted global warming from the greenhouse effect results in significant sea level rise.

**Table 40. Change in Acreage (Acres x 100) in Louisiana Wetlands (from LGS and EPA, 1987)[1]**

| Habitat Type | 1956 Acreage | 1978 Acreage | Acreage Change | Percentage Change |
|---|---|---|---|---|
| Marsh | 182,838 | 89,381 | -93,457 | -51% |
| Forested wetland | 7,894 | 3,233 | -4,661 | -59% |
| Upland | 3,362 | 6,915 | +3,553 | +106% |
| Dredge deposit | 3,057 | 11,369 | +8,313 | +272% |

[1] Calculations of acreage based in FWS National Wetlands Inventory documentation.

## Alabama

### Mobile-Tensaw River Delta

Stout et al. (1982) inventoried the extent and composition of wetlands and submerged aquatic vegetation grassbeds in the Mobile-Tensaw River Delta. This delta, designated as a National Natural Landmark in 1974, consists of approximately 115,103 ac (46,600 ha) of wetland habitats. The study area was bordered to the north by the confluence of the Alabama and Tombigbee Rivers and extended southward 45 miles (72.5 km) to the head of Mobile Bay.

Interpretation of color-infrared photography and field verification were used to delineate wetland areas. Three general categories and nine wetland habitats were delineated, and acreage, location, topography, and major plant species of each habitat were determined. The three general categories of wetlands were forested wetlands, marshes, and submerged grassbeds. Of these three, forested wetlands were the most extensive, covering 100,014 ac (40,490 ha) (86.8%) of the total wetland acreage. Marshes covered 10,589 ac (4,287 ha) (9.2%) and submerged grassbeds covered 3,696 ac (1,496 ha) (3.2%) (Table 41). Forested wetland types and corresponding acreages were: bay forest, 3,291 ac (1,332 ha); alluvial swamp, 33,966 ac (13,751 ha); deep alluvial swamp, 35,301 ac (14,292 ha); natural levees, 26,564 ac (10,755 ha); and moist pine savannah, 60 ac (24.3 ha). Many rare and endangered plant species were found in the moist pine savannah community (Stout et al., 1982). Freshwater marsh habitat was divided into low marsh, 4,354 ac (1,763 ha) and high marsh, 6,235 ac (2,524 ha). In addition to wetland habitats, 808 ac (327 ha) of upland pine-oak community were delineated.

**Table 41. Habitat Acreages in Alabama's Mobile-Tensaw River Delta (from Stout, 1982)**

| Wetland Category | Acres | % of Total |
|---|---|---|
| Forested wetlands | 100,014 | 86.8 |
| Marshes | 10,589 | 9.2 |
| Submerged grassbeds | 3,696 | 3.2 |

Summary of Wetland Habitats and Acreage from the Inventory:

| Wetland Habitat | Acres |
|---|---|
| Alluvial swamp | 33,966 |
| Deep alluvial swamp | 35,301 |
| Natural levee | 26,564 |
| Bay forest | 3,291 |
| Moist pine forest | 832 |
| Moist pine savannah | 60 |
| Freshwater marshes | |
| High marsh | 6,235 |
| Low marsh | 4,354 |
| Submerged grassbeds | 3,696 |
| Upland pine-oak | 808 |

There are no other detailed studies to which the results of this survey can be compared. However, in recent years there has been increased oil and gas exploration in the area. Dredged canals have the potential to modify the hydrology and community structure of the delta.

Mobile Bay

Mobile Bay has received much attention during recent years because of declining natural resources. The Alabama Sea Grant Extension Service (1987) summarized the knowledge of chemical, physical, and biological oceanography of Mobile Bay. Stout (1979), Stout and Lelong (1981), and Roach et al. (1987) surveyed the historic changes in wetlands, and Stout (1987b) summarized the status of seagrass communities in coastal Alabama. Baldwin (1987) summarized waterfowl habitats in Mobile Bay.

Watzin et al. (unpublished) are preparing a summary of the cumulative impacts on the Mobile Bay ecosystem. This assessment proposes a strategy for effective environmental management of Mobile Bay and includes: environmental problem analysis, status and trends analysis, goal setting for bay resources, and implementation of goals. Information in that report is organized through the use of cause and effect models. The first step in construction of the model is to state the problem. Next, the causes of the problem are listed, and the effects of the problem defined. For example, one problem discussed is the decline of natural emergent and submerged aquatic vegetation. Causes of decline include filling activities that cover wetlands, dredging activities that remove vegetation, and high concentrations of suspended particulate loads in bay waters that limit light penetration. Effects include loss of primary production, cover for fish and wildlife, and benthic production. A total of 13 cause and effect models are being prepared which summarize the relationship between actions and their effects on components of the Mobile Bay ecosystem.

Data from status and trends analyses in the Watzin et al. report were used to verify some of the conceptual processes in the models and to quantify some impacts observed in the Mobile Bay ecosystem. A decline of over 10,000 ac (4,049 ha) of emergent marsh and probable loss of 50% or more of submerged aquatic vegetation occurred between the 1940's and 1979. In addition, the hydrology of the Mobile Bay has been markedly altered through excavation of a deep channel through the center of the Bay and the profusion of spoil areas in open water.

Watzin et al. report several animals which require large habitat ranges which once lived in this area, but are no longer found, including red wolf (*Canis n. niger*), Florida panther (*Felis concolor coryi*), and Florida black bear (*Ursus americanus floridanus*). Rare, but still occasionally found are swallowtail kite (*Elanoides forficatus*), sandhill crane (*Grus canadensis*), and gopher tortoise (*Gopherus polyphemus*).

The most significant human impact noted in Mobile Bay by Stout (1979) was the direct and indirect effects of dredging. Some of the effects of major dredging projects in Mobile Bay are given in Table 42. The impacts of small dredging projects are not included in Table 42 because they were difficult to quantify. Approximately 6,000 ac (2,429 ha) of marshland have been destroyed and approximately 2,200 ac (891 ha) created by spoil deposition. Assuming that a newly created marsh is functionally equivalent to natural marshland (according to Stout, this is a dangerous assumption), Mobile Bay has experienced a net loss of approximately 3,800 ac (1,538 ha) of marshland through filling for spoil disposal.

The distribution of the remaining marshes in Mobile Bay shows the influence of freshwater inflow (Stout, 1979). Salt marshes dominated by *S. alterniflora* and *J. roemerianus* occur only in lower Mobile Bay. The locations and marsh types in Mobile Bay are shown in Table 43.

Wetland losses due to erosion along the shoreline of Mississippi Sound in Alabama, including adjacent islands, was approximately 21 ac (8.5 ha) per year, or a total of 630 ac (255 ha) from 1955-1985. Much of this loss was marshland (Smith, 1989). This loss was not compensated by accretionary gains which were negligible. Continued loss of coastal wetlands is expected under the prevailing natural system and this progressive loss will be due, primarily, to action of natural forces including wind-generated waves, tides, currents, and predicted drowning effect of sea level rise. Smith presented a series of nine maps, representing the 1955 and 1988 shoreline positions of a portion of this shoreline. Erosion rates as great as 12 ft (3.66 m) per year are reported (Smith, 1989).

Mississippi

Mississippi Sound is a large coastal body of water bounded seaward by a series of barrier islands. The main estuaries of Mississippi Sound include the Pascagoula River, Biloxi Bay, Bay St. Louis, and Pearl River. Approximately 70 mi (112.9 km) of coast occurs in Mississippi. The tidal shoreline included in this distance is 369 mi (595 km). The total wet surface area at mean low water (MLW) is 433,447 ac (175,485 ha) and at mean high water (MHW) is 500,379 ac (202,583 ha). Approximately 21% of the area has a MHW depth of less than 3 ft (0.9 m), and about 6% has a depth over 18 ft (5.5 m). The average depth at MLW is 11.7 ft (3.6 m) (Christmas, 1973).

Relatively little recent information is available on the status and trends of Mississippi's wetlands and seagrass habitats. Mississippi marshes occur as an ecotone between saline and fresh water habitats. Most of the coastal mainland marsh area (65,805 ac; 26,642 ha) is dominated by *J. roemerianus* (61,398 ac; 24,857 ha). *S. alterniflora* (2,028 ac; 821 ha), *S. patens* (460 ac; 186 ha), and *Scirpus olneyi* (96 ac; 39 ha) are the major plants that occur in the saline regions of the marsh. The freshwater habitats are very diverse communities possessing more than 30 plant species (Christmas, 1973).

Table 42. **Impact of Dredging Activities on Mobile Bay Estuarine Marshes (from Stout, 1979).**

| | Location | Acres |
|---|---|---|
| I. | Loss to Spoil Deposition | |
| | Bon Secour River | 95 |
| | Blakeley Island | 3,000 |
| | East Fowl River | 172 |
| | Little Dauphin Island | 10 |
| | Dog River | 81 |
| | I-10 Highway | 180 |
| | I-10 Twin Tunnels | 13 |
| | Alcoa-Blakeley Island | 300 |
| | Scott Paper Company | |
| | 3 Mile Creek | 150 |
| | Private Projects | 1,000 |
| | TOTAL | 6,002 |
| II. | Loss to Canal Dredging | |
| | I-10 | 34 |
| | I-65 | 8 |
| | Theodore Industrial | 50 |
| | Private Projects | 46 |
| | TOTAL | 138 |
| III. | Creation by Spoil Deposition | |
| | Blakeley Island | 900 |
| | Polecat Bay | 900 |
| | Pinto Island | 387 |
| | Theodore Spoil Island | 7 |
| | TOTAL | 2,194 |

Total Loss 6,140 ha - Total Creation 2,194 ha = Net Loss 22% Total Marshlands

Table 43. **Estimates of Marsh Area by Type and Geographic Area for Mobile Bay and the Mobile Delta (from Stout, 1979).**

| Geographic Area | Marsh Area (Acres) Salt/Brackish[a] | Fresh[b] | Total |
|---|---|---|---|
| Mobile Bay | 3,291[c] | 214 | 3,505 |
| 1. Dog River | | 186 | |
| 2. Northwest Mobile Bay, Dog River to Deer River | 20 | | |
| 3. Deer River Complex | 215 | | |
| 4. East Fowl River | 358 | | |
| 5. Southwest Mobile Bay, East Fowl River to Cedar Point | 300 | | |
| 6. East Dauphin Island Little Dauphin Island | 182 | 28 | |
| 7. Weeks Bay | 191 | | |
| 8. Oyster Bay/Bon Secour River | 1,086 | | |
| 9. Fort Morgan Peninsula | 939 | | |
| Mobile Delta (Hwy, 90 North to Mobile & Baldwin County Lines) | | 10,450 | 10,450 |
| TOTALS | 3,291 | 10,664 | 13,955 |

[a] From Stout, 1977
[b] From Vittor and Stout, 1975

In 1968, there were 66,931 ac (27,098 ha) of coastal marsh in southern Mississippi. The 64,805 ac (26,237 ha) of coastal mainland marsh consisted of 823 ac (333 ha) of freshwater marsh and 63,982 ac (25,904 ha) of salt or brackish marsh. There were 2,126 ac (860 ha) of salt marsh found offshore on the northern shores of the barrier islands. In addition, there were approximately 17,000 ac (6,883 ha) of submerged vegetation. The most abundant species were shoal grass, manatee grass, turtle grass, and widgeon grass (Christmas, 1973).

Prior to 1930, approximately 1,000 ac (405 ha) of marsh were filled for road development. Since 1930, another 8,170 ac (3,308 ha) have been filled for industrial or urban development, and 85 ac (34 ha) for landfills. As of 1973, 12% of Mississippi marshes had been filled for development (Christmas, 1973).

The importance of Mississippi estuaries as a breeding and nursery ground for a multitude of species has been well documented. The commercial catch of fish and shellfish in 1989 for all of the Gulf states was 1.8 billion lb (800,000 mt), with a dockside value of $649 million (NOAA, 1990a). These landings represented 21% of the total United States landing for that year. Mississippi's contribution was 298,206,000 lb (135,000 mt), which was 17% of the total from Gulf states. Of that catch, most were menhaden, industrial bottom fish, shrimp, oysters, and crabs. The value of the 1989 Mississippi catch was $43,949,000. In 1989, the port of Pascagoula-Moss Point was third of all U.S. ports in volume of landings with a catch of 282,100,000 lb (128,000 mt). According to Power (1963), 97% of the species caught by volume and 88% by value in the 1961 Mississippi landings were estuarine dependent at one time or during all stages of their life histories.

Christmas (1973) listed 180 species of invertebrates found in trawl samples in Mississippi Sound. Thirty species of crustaceans and molluscs made up 97% of the total number of species. White and brown shrimp made up 29% and 14% of the catch, respectively. Crabs and oysters were collected in amounts lower than expected because the sampling gear was not designed to capture them.

Bay anchovies were the predominant fish in samples taken in Mississippi Sound, and made up 70% of the catch by number. The small size of the bay anchovy adult has precluded commercial exploitation of this fish. However, the anchovy was found to be very important as a forage fish, but further study was needed to quantify its value to the commercial fishing industry. The next five species in order of dominance were menhaden (10.7%), Atlantic croaker (6.6%), spot (2.3%), butterfish (2.1%), and seatrout (1.6%). These species and the bay anchovy made up 93% of the total number of fish collected (Christmas, 1973).

In 1880, 62,000 lb (28 mt) of oysters were included in the first records of landings. This amount represented 3% of the Gulf Coast harvest at that time. The maximum landings of oysters occurred in 1937 with 12,894,000 lb (5,860 mt) (53% of the Gulf harvest). The minimum harvest of oysters in Mississippi (23,000 lb; 10.4 mt) occurred in 1952 (0.2% of the Gulf harvest). More efficient management resulted in increased landings. In 1964, 4,829,000 pounds (2,195 mt) (21%) were harvested in Mississippi. Since that time, the State has produced approximately 13% to 17% of the annual Gulf harvest.

Oyster reefs are located along the entire coast of Mississippi, with the largest reefs near the western boundary. In 1962, approximately 100 oystermen were harvesting oysters from the reefs at the mouth of the Pascagoula River, but that year heavy rains virtually wiped out the harvest. As of 1969, there were still very few oysters on the Pascagoula beds.

In 1966, according to a survey by W.J. Demoran (Gulf Coast Research Laboratory, Ocean Springs, Mississippi), there were 9,934 ac (4,022 ha) of oysters; 582 ac (236 ha) were planted beds. According to the National Register of Shellfish Production Areas (Houser and Silava, 1966), there were 35,000 ac (14,170 ha) of approved shellfishing area. This is the highest classification available to estuarine areas. A total of 87,300 ac (35,344 ha) of estuaries were closed to shellfishing. In 1961, 350 ac (142 ha) of a highly productive oyster reef in Back Biloxi Bay were closed. Also, 540 ac (219 ha) of oyster bottom at the Pascagoula oyster reef were closed due to pollution. The Escatawpa River estuary is heavily contaminated for 7 $mi^2$ (18 $km^2$) and is "dead" in comparison with other local rivers (Demoran, personal communication).

In a recent study, Eleuterius (in press) noted a decline of the area covered by seagrass and a decline in occurrence of seagrass species in Mississippi Sound. Acreage of seagrasses in 1975 was about 60% of that found in 1969, and losses are continuing. Hurricane damage and destruction by fresh water discharged through a spillway accounted for approximately half of the observed loss. The cause of the remaining loss is not known, but may be related to sediment quality (Eleuterius, personal communication). The seagrass beds remaining in Mississippi Sound are composed almost entirely of *H. wrightii*. However, even this relatively robust seagrass species exhibits sparse leaves.

## Conclusions

The Gulf coastal zone is characterized by a great diversity of habitats, flora and fauna, natural resources, industry, and recreational potential. The coast is richly endowed with productive wetland habitats whose ecological functions enhance viable wildlife and fishery resources of the Gulf. Coastal wetland systems are in natural biogeochemical balance with contiguous terrestrial, freshwater, and marine systems. The natural balance can be adversely impacted by natural and human activities, and these impacts can irreversibly change the character of the wetlands. For example, when accretion of sediments by marshland is exceeded by subsidence, the result is a loss in marshland and an increase in open water habitat.

- Although economic evaluation of wetlands is still developmental, the value of some marshes, based on sport and commercial fisheries landings alone, may be nearly equivalent to their real estate value.

- In many instances, wetland habitats along the Gulf coast continue to be besieged by human activities, such as dredging and filling for construction of canals, real estate development, and conversion to other uses. Natural phenomena such as subsidence, sea level rise, impact of storms, and erosion, also adversely impact wetlands. The primary causes of diminishing quality and acreage of coastal emergent wetlands vary from state to state: subsidence due to extraction of oil, gas, and fresh water in Texas; alterations of hydrodynamic flow due to construction of navigation channels, as well as subsidence in Louisiana; dredge and fill operations in Mississippi and Alabama; and conversion of wetlands to commercial and private developments in Florida.

- Although it is difficult, and sometimes impossible, to precisely compare acreages of wetlands at different time intervals, reputable comparisons indicate a trend of major losses of wetland habitats, particularly marsh, mangroves, and submerged aquatic vegetation. These losses are summarized in Table 44. Some recent data suggest that the rate of loss may be declining in specific areas (Table 45).

- Elevated levels of bacteria, which result in closure of waters to shellfish harvesting, is a common problem in all Gulf states.

- Submerged aquatic vegetation in all Gulf coastal waters is especially susceptible to adverse effects of dredging and filling activities, both from deposition of spoil directly on the plant communities and from increased turbidity which reduces light penetration.

Table 44. Summary of Studies Documenting Loss of Coastal Habitats in Gulf States.

| Type of Habitat | Location | Time Frame | Loss (Acres) | % Loss | Citation |
|---|---|---|---|---|---|
| Marine intertidal | USA | Mid-1950's to mid-1970's | 4,000 | - | Frayer et al., 1983 |
| Estuarine intertidal emergent | USA | Mid-1950's to mid-1970's | 353,200 | - | Frayer et al., 1983 |
| Estuarine intertidal forested | USA | Mid-1950's to mid-1970's | 19,100 | - | Frayer et al., 1983 |
| Wetlands | Chenier Plain, Louisiana | 1952-1974 | 204,313 | - | Gosselink et al., 1979 |
| Coastal Louisiana wetlands | Coastal Louisiana | 1970's | 25,194/yr | - | Gagliano et al., 1981 |
| Coastal wetlands | Mississippi Deltaic Plain | 1932-1987 | 583,696 | - | May and Britsch, 1987 |
| Nearshore gulf | Calcasieu Basin, LA | 1953-1975 | 397 | - | Bahr et al., 1977 |
| Harvestable shellfish waters | Gulf coast | 1985 | 1,751,478 (closed for harvest) | - | Broutman and Leonard, 1988 |
| Beaches and dunes | Louisiana coastal zone | 1955/56-1978 | 4,142 | - | Baumann and Turner, in press |
| Beaches and bars | Upper Mobile Bay | 1955-1979 | 54 | -89% | Roach et al., 1987 |
| Mud flats | Charlotte Harbor, FL | 1945-1982 | 8,483 | -76% | Harris et al., 1983 |
| Forested wetlands | Coastal Louisiana | 1956-1978 | 4,661 | -59% | Louisiana Geological Survey and EPA, 1987 |
| Seagrasses | Tampa Bay Region, FL | 1950-1982 | 31,698 | -50% | Haddad, 1989 |
| Seagrasses | Sarasota Bay, FL | 1948-1987 | 2,146 | -35% | McGarry, personal communication |
| Seagrasses | Charlotte Harbor, FL | 1945-1982 | 24,464 | -29% | Harris et al., 1983 |
| Seagrasses | Tampa Bay, FL | 1879-1981 | 56,527 | -73% | Lewis et al., 1985<br>Lewis et al., 1991 |
| Seagrasses | Mississippi Sound | 1969-1975 | 15,134 | -76% | Eleuterius and Miller, 1976 |
| Seagrasses | Tampa Bay, FL | Historical-1989 | 62,367 | -79% | Zieman and Zieman, 1989 |

* Loss defined as the substantial removal of land from its ecologic role under natural conditions (Craig et al., 1979b).

Table 44. (Continued)

| Type of Habitat | Location | Time Frame | Loss (Acres) | % Loss | Citation |
|---|---|---|---|---|---|
| Mangrove | Sarasota Bay, FL | 1948-1987 | 703 | -45% | McGarry, personal communication |
| Mangrove | Tampa Bay Region | 1950-1982 | 1,475 | -7% | Haddad, 1989 |
| Marsh | Louisiana Coastal Zone | 1955/56-1978 | 631,539 | - | Baumann and Turner, in press |
| Marsh | Beaumont-Port Arthur, TX | 1956-1978 | 9,410 | - | White et al., 1985 |
| Marsh | Coastal Louisiana | 1956-1978 | 93,457 | -51% | Louisiana Geological Survey and EPA, 1987 |
| Marsh | Barataria Bay, LA | 1945-1980 | 107,906 | - | Sassar et al., 1986 |
| Marsh | Galveston Bay, TX | 1956-1979 | 24,449 | - | NOAA, 1989c |
| Marsh | Tampa Bay Region, FL | 1950-1982 | 1,559 | -30% | Haddad, 1989 |
| Marsh | Charlotte Harbor, FL | 1945-1982 | 3,704 | -51% | Harris et al., 1983 |
| Marshland | Mobile Bay, Al | 1979 | 6,135 | - | Stout, 1979 |
| Natural marsh | Calcasieu Basin, LA | 1953-1975 | 392 | - | Bahr et al., 1977 |
| Non-fresh marsh | Coastal Alabama | 1955-1979 | 12,027 | -29% | Roach et al., 1987 |
| Non-fresh marsh | Upper Mobile Bay | 1955-1979 | 6,680 | -35% | Roach et al., 1987 |
| Swamp | Louisiana Coastal Zone | 1955/56-1978 | 80,843 | - | Baumann and Turner, in press |
| Oyster reef | Charlotte Harbor, FL | 1945-1982 | 318 | -39% | Harris et al., 1983 |

Table 45. Selected Wetland Changes (Acres) Along the Gulf Coast (Calculations based on maps supplied by FWS-NWRC, 1990, Personal Communication)

| Site/Habitat | 1956 | 1978 | 1979 | 1983 | Acreage Change | | Percent Change | |
|---|---|---|---|---|---|---|---|---|
| Galveston Bay | | | | | 1956-1979 | | 1956-1979 | |
| Marsh | 117,603 | | 105,730 | | -11,873 | | -10% | |
| Beach Bar | 2,697 | | 1,521 | | -1,176 | | -43% | |
| Flats (mud and sand) | 16,461 | | 13,748 | | -2,813 | | -16.5% | |
| Fresh Water Ponds and Lakes | 11,475 | | 19,395 | | +7,920 | | +69% | |
| Atchafalaya River Basin (LA) | | | | | | | | |
| Marsh | 149,034 | 39,452 | | 97,123 | (not comparable) | | | |
| Non-Fresh Open Water | 107,152 | 120,908 | | 124,497 | (not comparable) | | | |
| Benches | 439 | 472 | | 542 | (not comparable) | | | |
| Lower Mississippi River Delta (LA) | | | | | 1956-1978 | 1978-1983 | 1956-1979 | 1978-1983 |
| Marsh | 183,361 | 97,408 | | 95,632 | -85,953 | -1,776 | -46% | -18% |
| Open Water | 468,867 | 550,883 | | 553,329 | +81,916 | +2,496 | +17.3% | +0.5% |
| Horn Island | | | | | 1956-1978 | 1978-1983 | 1956-1979 | 1978-1983 |
| Marsh/Grassland | 1,444 | 1,085 | | 1,155 | -359 | +70 | -25% | +6% |
| Vegetated Dunes | 0 | 338 | | 1,243 | -338 | +905 | +300% | +63% |
| Benches and Bars | 1,778 | 1,229 | | 559 | -549 | -600 | -31% | -18% |
| Mississippi Sound and Mobile Bay | | | | | | | | |
| Salt Marsh | 98,325 | | | | | | | |
| Benches, Flats, Shores | 12,528 | | | | | | | |
| Non-Fresh Open Water | 1,110,620 | | | | | | | |

• Excess nutrients from septic tank systems, sewage treatment plant discharges, and drainage from agricultural fields contribute to seagrass loss by increasing turbidity and stimulating growth of periphyton on the plants and phytoplankton (additional suspended particulates) in the waters over the grass beds.

• The loss of quality and acreage of wetlands has slowed from the middle to late 1970's due to passage and enforcement of Federal and state laws and regulations. Also, the ecological and productive functions of wetlands are becoming better known to resource managers and to a more interested and active public. Better informed managers and public awareness have made a positive difference in management decisions concerning the disposition of wetland habitats.

• Interest in wetland management and research continues to grow, and many wetland-oriented projects are underway. For example, the U.S. FWS is digitizing information on wetland acreages in Gulf coastal states (some preliminary data are presented in this report), NOAA continues to collect data in their Status and Trends Program, MMS in cooperation with the state of Louisiana is completing a study of marsh management practices, and the states of Florida and Texas will soon publish updated data on wetland acreage. The U.S. Army Corps of Engineers is designing and testing several freshwater diversion projects to improve the condition of wetland habitats along the lower Mississippi River. Finally, the U.S. EPA Gulf of Mexico Program is developing a database on a GIS system which will be instrumental in establishing a ecologically balanced management plan for Mobile Bay.

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