Courtesy C. McIvor

Coastal & Marine Ecosystems

by
Science Editor
Carole C. McIvor
National Biological Service
Arizona Cooperative Fish and Wildlife Research Unit
University of Arizona
Tucson, AZ 85721

Overview

The quantity and health of the nation's coastal and marine resources have declined over historical time at the species, community, and ecosystem levels. All articles in this section implicate human activities as contributing to these declines. Human impacts on the coastal and nearshore marine zone include urbanization (direct loss of habitat, lowered water quality), shoreline modification (dredging and filling, diking and impoundments), overfishing, and high-density recreational use.

Some portion of the overall downward trend is directly attributable to natural processes. Hurricanes and coastal storms can have significant negative impacts on both barrier islands (Williams and Johnston) and seagrass beds (Handley, Onuf). Rising sea level and coastal subsidence—natural processes that are likely being accelerated by anthropogenic (human-caused) activities—are responsible for coastal wetland loss in Louisiana (Johnston et al.). Rising sea level is also implicated in the erosion of barrier islands (Williams and Johnston). The inescapable conclusion is, however, that even where natural processes play a role, human impact is of equal or greater importance to the long-term health of these resources.

Despite overall declines in coastal and marine resources, there is some room for cautious optimism. Some coral reefs are far enough from human habitation that they are probably stable and not declining (Jameson). Despite changes in the relative abundances of native fish species and the introduction of exotic species in the tidal portion of the Hudson River, no native fish species have beeen extirpated within the period of record (1936 to 1990) (Daniels). The population trend for manatee (*Trichechus manatus*) in Florida appears stable and perhaps slightly increasing (Lefebvre and O'Shea). Recent local reversals in the decline of seagrasses have occurred in Chesapeake Bay (Pendleton) and in lower Tampa and Little Sarasota bays (Handley). These successes, however, are tempered by the realization that human populations in coastal states are projected to substantially increase soon.

It is clear from these articles that the quality and extent of our information bases for judging status and trends of our coastal and marine resources are often inadequate. Whereas the areal coverage of some ecosystems can be judged by comparison of remotely sensed data (e.g., coastal wetlands), gathering analogous information on other ecosystems or components (e.g., fishes on coral reefs) requires much smaller scale, more labor-intensive efforts. In their review of Florida Keys reef fishes, Smith-Vaniz et al. were forced to rely on a combination of

Contents ◄ Article ► ◄ Page ►

human population trends and the status of Keys' reefs combined with information collected from commercial fisheries' landings to infer the health of reef fishes: no single reef site has ever been repeatedly surveyed for fish abundance over time. This example clearly demonstrates that to better judge the status of our coastal and marine resources in the future, carefully chosen and designed long-term monitoring is required.

Nearshore Fish Assemblage of the Tidal Hudson River

by
Robert A. Daniels
New York State Museum
Biological Survey Laboratory

The Hudson River drains about 45,000 km^2 (17,370 mi^2), most of it in eastern New York. Although this is a young river with a relatively small watershed at higher latitudes, the Hudson and its tributaries support a rich fish fauna of more than 200 species (Smith and Lake 1990). This fauna is a diverse mixture of native and exotic freshwater species, diadromous (migratory between fresh and salt waters) fish, and marine strays (Barnhouse et al. 1988). More than 150 of these species are reported from the tidal portion of the river that extends 243 km (151 mi) from the Battery on Manhattan Island to the Troy Lock (Fig. 1); of these, about 80 species are freshwater or diadromous forms and 50 species occur regularly in nearshore areas (Smith 1985). During the last half-century, the nearshore fauna of the tidal portion of the river has undergone two types of changes: species have been added to and deleted from the fauna and the relative abundances of the dominant species have changed.

I explore differences in the nearshore fish assemblages of the Hudson River by comparing information on the distribution and abundance of fish collected between 1936 and 1990. This comparison offers only a coarse look at change in the fish assemblage and provides little information on trends. The nearshore fish assemblage of the Hudson River is dynamic and changes on a daily, seasonal, and annual basis.

Surveys of Fish Fauna, 1936-92

The study of Hudson River fish dates to Samuel Mitchill's publication on the fish of New York (Mitchill 1815). DeKay (1842) and Bean (1903) also provided information on fish in the Hudson River, but the first synoptic survey of the fish in the river system was not undertaken until 1936 (Greeley 1937). The watershed surveys of New York conducted between 1926 and 1939 included a detailed survey of fish distribution and abundance in the lower Hudson River drainage. Fish collected during these surveys were vouchered; specimens are housed at the New York State Museum (NYSM). Beginning in the early 1970's, interest in the fish of the Hudson River increased dramatically (Limburg et al. 1986), and several long- and short-term monitoring programs began. Data collection continues in many of the long-term programs.

Painting of pumpkinseed (*Lepomis gibbosus*), a persistent species of the Hudson River.

Courtesy NY Environmental Conservation Department

Courtesy R. Daniels, NY State Museum

Hudson River.

To examine change in the nearshore fish assemblage of the Hudson River, I used selected information from the 1936 watershed survey; NYSM surveys conducted between 1990 and 1992; intensive site surveys conducted between 1976 and 1979 by Lawler, Matusky and Skelly Engineers (LMS); and surveys supported by Con Edison between 1974 and 1989. Because techniques and equipment vary among the surveys, I have included in the analyses only information collected by workers using seines. Still, the size of the seines used, the mesh size, and the area sampled differ among the surveys and contribute a bias not easily quantified. Because this analysis is relatively coarse, any biases that may exist in the data should be masked. Furthermore, in most of the analyses I have made comparisons within data bases. Comparisons between data bases are used primarily with presence and absence applications.

The 1936 watershed survey collected information on fish from 112 sites in the tidal Hudson River (Fig. 1). All sampling was conducted during summer. Fish collected during this survey were identified and counted or ranked; the ranking system may have varied among the crews. To compare abundance, I assigned numbers to the ranks in the fieldbooks and compared my assigned number to the actual number of preserved fish. In 20 comparisons of each of the five ranks, the assigned number equaled or underrepresented the number preserved 73% of the time; therefore, the abundance estimates should be conservative.

Between 1990 and 1992, I collected information on fish abundance and distribution from several sites on the tidal portion of the Hudson River. Most information discussed here is from work done at four sites during the summer of 1990 (Fig. 1). These sites typified the nearshore habitats present along the entire main channel. LMS intensively collected fish from four sites between 1976 and 1979 (Fig. 1). Day and night sampling, using seines to collect fishes at weekly or biweekly intervals, began early in spring and continued until December. The data base from Con Edison includes information from 31,582 nearshore, shallow-water sites throughout the 243-km (151-mi) course of the lower Hudson River. These collections were made between 1974 and 1989.

Table 1. Freshwater and diadromous fishes collected from nearshore areas of the Hudson River, 1936-92. Records from 1936 are from the watershed survey of the lower Hudson River, with identifications verified, and specimens vouchered. Records from 1974-89 are from the Con Edison data base, no specimens vouchered. Records from 1990-92 are from New York State Museum (NYSM) surveys and other additional specimens, vouchered.

Species	Fish surveys			
	Watershed 1936	Con Ed 1974-80	Con Ed 1981-89	NYSM 1990-92
Longnose gar				x
American eel	x	x	x	x
Blueback herring	x	x	x	x
Alewife	x	x	x	x
American shad	x	x	x	x
Gizzard shad		x	x	x
Bay anchovy	x	x	x	x
White catfish	x	x	x	x
Brown bullhead	x	x	x	x
White sucker	x	x	x	x
Creek chubsucker	x			
Northern hog sucker	x	x	x	
Goldfish	x	x	x	
Grass carp				x
Common carp	x	x	x	x
Rudd				x
Golden shiner	x	x	x	x
Creek chub	x	x		
Fallfish	x	x	x	x
Eastern silvery minnow	x	x	x	x
Comely shiner	x	x	x	x
Emerald shiner	x	x	x	x
Bridle shiner	x	x	x	
Spottail shiner	x	x	x	x
Common shiner	x	x	x	
Spotfin shiner	x	x	x	
Fathead minnow		x	x	
Blacknose dace		x	x	
Longnose dace		x	x	
Rainbow smelt	x	x		
Redfin pickerel	x	x	x	
Northern pike		x	x	x
Chain pickerel	x	x	x	x
Banded killifish	x	x	x	x
Mummichog	x	x	x	x
Inland silverside	x	x	x	x
Fourspine stickleback	x	x	x	x
Threespine stickleback		x	x	
White perch	x	x	x	x
White bass			x	
Striped bass	x	x	x	x
Rock bass	x	x	x	x
Bluespotted sunfish	x			
Redbreast sunfish	x	x	x	x
Pumpkinseed	x	x	x	x
Bluegill	x	x	x	x
Smallmouth bass	x	x	x	x
Largemouth bass	x	x	x	x
White crappie		x	x	x
Black crappie	x	x	x	x
Tessellated darter	x	x	x	x
Yellow perch	x	x	x	x
Logperch	x	x		
Shield darter	x	x		
Freshwater drum				x
Total number of species	43	48	45	38

Changes, 1936-90

The changes in the nearshore fish assemblages of the Hudson River that have occurred during the past six decades are illustrated in several ways. First, the component species have changed, although species richness (number of species in the assemblage) has remained relatively constant (Table 1). During the 1936 survey, the assemblage had 43 freshwater and diadromous species. Based on recent NYSM collections, the assemblage consists of 38 species. Recently introduced to the river are gamefishes such as northern pike (*Esox lucius*) and white crappie (*Pomoxis annularis*) and exotic fishes such as rudd (*Scardinius erythrophthalmus*) and grass carp (*Ctenopharyngodon idella*). Two additions from undocumented sources also included the gizzard shad (*Dorosoma cepedianum*) and freshwater drum (*Aplodinotus grunniens*). Several species that remain common in tributary streams are now extirpated or extremely rare (e.g., bridle shiner).

In addition, the relative abundance of most resident species (excluding diadromous forms) has changed (Table 2). The two dominant resident species in 1936 (spottail shiner [*Notropis hudsonius*] and white perch [*Morone americana*]) made up 34% of the individuals in the assemblage. The same two species remained dominant in the 1990 survey, but have almost doubled their relative abundance to 64% of the individuals in the assemblage. The relative abundances of an additional five persistent species have declined between the two sampling events, but only slightly. Thus, declines in relative abundance were most noticeable in the remaining species (not dominant or persistent) of the freshwater component of the river fish assemblage. In 1936, 36 species made up 26% of the catch, while the remaining species accounted for 7% in 1990.

The diadromous fishes typically dominated, by number, the nearshore assemblage during summer (Fig. 2) because of the presence of young-of-year individuals. The most common species in all samples included blueback herring (*Alosa aestivalis*), alewife (*A. pseudoharengus*), American shad (*A. sapidissima*), bay anchovy (*Anchoa mitchilli*), American eel (*Anguilla rostrata*), and striped bass (*Morone saxatilis*). The difference between 1936 and all other years was the curious near-absence of

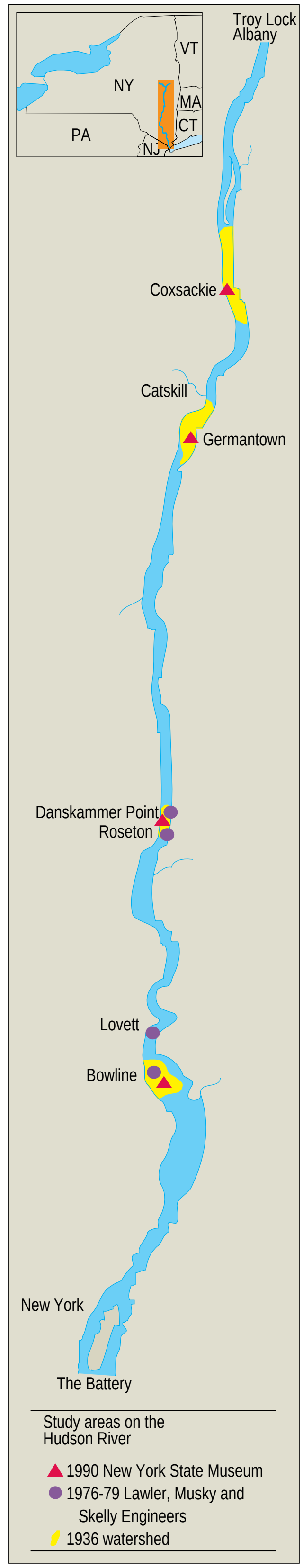


Fig. 1. The tidal portion of the Hudson River, New York, showing areas where some fish collections have been made over the last six decades.

Table 2. The relative abundance of resident fishes (percentage) in nearshore assemblages in the summers of 1936 and 1990, lower Hudson River, New York.

Species	Survey 1936	Survey 1990
Spottail shiner	20	33
White perch	14	31
Total for two dominant species	34	64
Banded killifish	14	12
Tessellated darter	10	6
Mummichog	7	4
Pumpkinseed	5	4
Redbreast sunfish	4	3
Total for five persistent species	40	29
Fourspine stickleback	6	
Eastern silvery minnow	4	
Goldfish	3	
Fallfish		2
Bridle shiner	2	
White sucker	2	
White catfish	2	
Golden shiner	2	
Gizzard shad		1
Others	5	4
All other species	26	7

blueback herring in 1936 when this species was taken at only 4 of the 112 sites sampled. In 1990, and during the last two decades, blueback herring dominated the summer catches at nearshore sites (Fig. 2). In the nearshore fish assemblage of the Hudson River in 1974-89, the five diadromous species dominated throughout the sample period (Fig. 2).

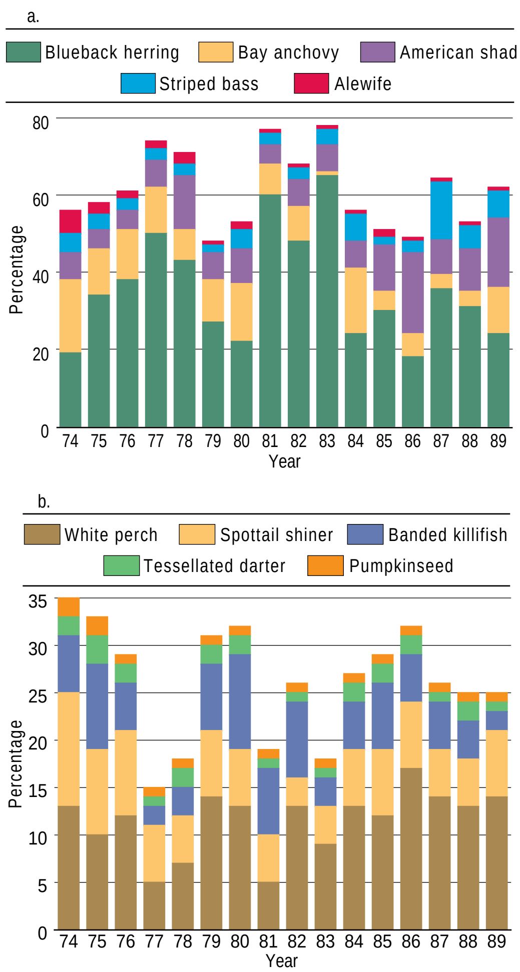


Fig. 2. Changes in the relative abundance of the 10 most numerous fishes in the nearshore fish assemblage of the Hudson River, New York, 1974-89 (data from Con Edison files). Changes in percentage abundance in (a) 5 diadromous fish species and (b) 5 resident freshwater species.

Despite fluctuations in each of the most abundant species, no obvious trends in relative abundance were apparent although the relative abundance of other species has changed. For example, the abundance of Atlantic silverside (*Menidia menidia*), a marine stray, has increased, while other species, such as two resident fish, emerald shiner (*Notropis atherinoides*) and goldfish (*Carassius auratus*), and the diadromous rainbow smelt (*Osmerus mordax*), have dramatically decreased in relative abundance (Fig. 3). Relative fish abundances exhibited site, diel, and seasonal variation.

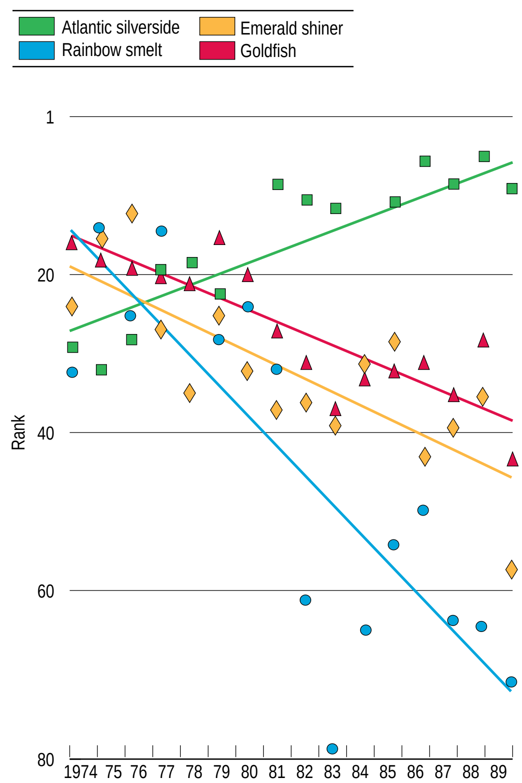


Fig. 3. Annual trends in the abundance of four fish species from the nearshore Hudson River assemblage. Rank is based on total number of fish caught during the year; the most abundant fish caught received a rank of 1, the least abundant a rank of 82.

Implications of Changes

Change in the nearshore fish assemblage of the tidal portion of the Hudson River is continuous. To identify trends in the abundance of an assemblage made up of resident freshwater and estuarine species, diadromous fishes, and marine strays, data must be collected in ways that account for the dynamic qualities of the species involved. Although the Hudson River is among the most-studied aquatic systems in North America, data necessary to confirm population trends in its fish assemblage are scant. Abundance data are best for some commercially important and protected fishes. Data on other species are often inadequate, rare, or nonexistent. Early or baseline data are often incompatible with modern surveys, and long-term data bases, although growing, are still in their early years.

Some changes appear to be trends. First, the number of fish species in the Hudson River appears to be increasing. The presence of recent entrants into the river—such as gizzard shad, rudd, grass carp, central mudminnow (*Umbra limi*), white bass (*Morone chrysops*), and freshwater drum—may create management concerns in the future.

Second, another group of fish appears to be declining, although it seems that only a few species, if any, have been extirpated. This group consists of fish that were common in the 1936 survey of the river but rare in all recent collections, including the bridle shiner (*Notropis bifrenatus*), common shiner (*Luxilus cornutus*), comely shiner (*Notropis amoenus*), spotfin shiner (*Cyprinella spiloptera*), creek chub (*Semotilus atromaculatus*), northern hog sucker (*Hypentelium nigricans*), and creek chubsucker (*Erimyzon oblongus*). These fish remain common, or at least present, in tributaries to the lower Hudson River. Their absence from the main channel may result from increasing development and loss of riparian vegetation at the mouths of many tributaries, which may isolate tributary populations from those of the main channel and lead to the creation of sub- or new populations.

The third apparent trend is that, although richness is increasing, diversity (an expression that includes the number of species and their relative abundance) in the nearshore fish assemblage has declined because of the increase in population size of the dominant species.

Studies that allow a better assessment of trends in the Hudson River fish assemblage will provide broad-based benefits. Management agencies, commercial fishing operations, and individual anglers, for example, all have an interest in the fisheries and fish of the river. Other river users, such as municipal planners and utility companies, also will gain from increased knowledge of the population trends of river-dwelling organisms because the trends reflect changes in water-quality conditions.

References

Barnhouse, L.W., R.J. Klauda, D.S. Vaughan, and R.L. Kendall, eds. 1988. Science, law, and Hudson River power plants. A case study in environmental impact assessment. American Fisheries Society Monograph 4. 347 pp.

Bean, T.H. 1903. Catalogue of the fishes of New York. New York State Museum Bull. 60, Zoology 9. 784 pp.

DeKay, J.E. 1842. Zoology of New York or the New York fauna, Part IV. Fishes. W. and A. White and J. Visscher, Albany, NY. 415 pp.

Greeley, J.R. 1937. Fishes of the area with annotated list. Pages 45-103 *in* E. Moore, ed. A biological survey of the lower Hudson watershed. Supplemental to 26th Annual Report, State of New York Conservation Department, Albany.

Limburg, K.E., M.A. Moran, and W.H. McDowell. 1986. The Hudson River ecosystem. Springer-Verlag, New York. 331 pp.

Mitchill, S.L. 1815. The fishes of New York, described and arranged. Transactions of the Literary and Philosophical Society (1814) 1:355-492.

Smith, C.L. 1985. Inland fishes of New York State. Department of Environmental Conservation, Albany. 522 pp.

Smith, C.L., and T.R. Lake. 1990. Documentation of the Hudson River fish fauna. American Museum Novitates 2981. 17 pp.

For further information:

Robert A. Daniels
New York State Museum
145 Jordan Rd.
Troy, NY 12180

Natural Resources in the Chesapeake Bay Watershed

by
Edward Pendleton
National Biological Service

The Chesapeake Bay is the nation's largest estuary; its watershed covers 165,760 km^2 (64,000 mi^2) and is occupied by 13 million people. By the 1980's, the bay's waters were enriched with nutrients from agriculture and loaded with pollutants from urban and suburban areas. The bay's submersed grasses were disappearing, fisheries 2 centuries old were in serious decline, and wetlands and other natural habitats were under continuing threats of development (Flemer et al. 1983).

In 1983 the federal government, Virginia, Maryland, Pennsylvania, the District of Columbia, and the Chesapeake Bay Commission formally declared their intent to work cooperatively to restore the natural resources of the bay. Their partnership, known as the Chesapeake Bay Program, attacked water-quality problems by adopting measures to reduce inputs of nitrogen and phosphorus from urban, industrial, and agricultural sources and to increase levels of dissolved oxygen in bay waters. Simultaneously, scientists and managers determined the status of bay species and natural habitats and began to track historical and ongoing trends.

Status and trends assumed special relevance as they were incorporated into managerial objectives and goals, or as indices of the success of programs and policies (Chesapeake Bay Implementation Committee 1988). Trends for three habitats—submersed aquatic vegetation beds, wetlands, and forests; four key aquatic species—oysters (*Crassostrea virginica*), blue crabs (*Callinectes sapidus*), striped bass (*Morone saxatilis*), and American shad (*Alosa sapidissima*); and waterfowl are summarized below. These trends represent a mixture of moderate successes and continuing challenges for managers of the bay.

Status and Trends

The status and trends of key habitats and species in the Chesapeake Bay are based on multiple annual surveys conducted by state and federal agencies. Perhaps the most comprehensive is a survey of the bay's submersed aquatic plant community; each year, the extent of submersed aquatic vegetation (SAV) is estimated by aerial photography of the entire bay and the tidal portions of its major tributaries (Orth and Moore 1983). Wetland areas are likewise estimated from aerial photographs and have been extrapolated for the watershed from a finite number of sites in various geographic strata for three time periods (Tiner and Finn 1986; Tiner et al., U.S. Fish and Wildlife Service, unpublished data). Approximately every 8 years, forested areas are estimated for each state in the bay watershed by the U.S. Department of Agriculture Forest Service's Forestland Inventory from satellite imagery (Chesapeake Bay Program 1993).

Many aquatic animal species that are surveyed annually (including those addressed here) support commercial and recreational fisheries or hunting and bird watching. Oyster, blue crab, striped bass, and American shad populations are estimated from commercial landings, and are augmented at times with surveys that are independent of fishery statistics, such as numbers of oyster spat that set each year, estimates of the biomass of spawning striped bass, or numbers of juvenile striped bass per seine haul (the young-of-year index). Waterfowl have been counted during their wintering season on the bay by the aerial Midwinter Surveys since the 1940's.

Submersed Aquatic Vegetation (SAV)

Beginning in the late 1960's and continuing into the 1970's, the distribution and abundance of a community of 20 species of submersed grasses declined throughout the bay because of nutrient enrichment, increased loads of suspended sediments, and other factors (Stevenson and Confer 1978; Orth and Moore 1983). In 1978 the first aerial survey estimated 16,500 ha (40,700 acres) of SAV in the bay (Anderson and Macomber 1980). The next year, 15,400 ha (38,000 acres) were documented (Orth et al. 1985); since that time, annual surveys have shown modest but continual increases in SAV coverage to an estimated 28,600 ha (70,600 acres; Orth et al. 1993; Fig. 1). Recent increases represent gains in brackish mid-bay regions and are tempered somewhat by slow or no SAV recovery in freshwater areas in the upper bay and by the spread of the exotic species hydrilla (*Hydrilla verticillata*) in the tidal freshwater portions of the Potomac River.

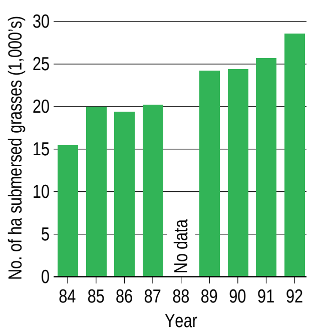


Fig. 1. Area of submersed grasses (submersed aquatic vegetation or SAV) in the Chesapeake Bay, 1984-92.

Wetlands

The status and trends for more than a million acres of wetlands in the Chesapeake Bay watershed have been estimated over two time periods, from the mid-1950's to the late 1970's and early 1980's (Tiner and Finn 1986), and from this period to 1989 (Tiner et al., USFWS, unpublished data). Dominant wetland types include nontidal forested wetlands (60% of total wetlands), nontidal shrub-scrub wetlands (10%), and salt and freshwater marshes (10% each).

Losses occurred in all of these wetland types during the period from the mid-1950's to late 1970's and early 1980's. About 9% of the watershed's salt marshes were lost to dredging, impoundment, and filling. Nontidal wetlands declined by nearly 6% as a result of being drained and converted to agriculture or impounded to form ponds, lakes, and reservoirs. During the 1980's, losses continued; the rate of marsh loss declined, while forested wetland losses increased. Overall, there was an estimated net loss of 0.5% of estuarine wetlands and a net loss of 2.0% of palustrine wetlands (roughly equal to tidal and nontidal wetlands) during the 1980's. These trends mirror historical losses over the past 200 years (Dahl 1990).

Forests

An estimated 95% of the Chesapeake Bay watershed was forested before European settlement; around 58% remains today (Chesapeake Bay Program 1993; Fig. 2). This percentage is declining for the first time in over a century because of recent forest clearing for urban and suburban development. Forest clearing has proceeded unevenly over the watershed, with some drainages intact and others as much as 85% cleared.

Oysters

Oyster landings in Chesapeake Bay have experienced a 95% decline since 1980 and are estimated to be at their lowest recorded level (Kennedy 1991; National Marine Fisheries Service, Annapolis, Maryland, unpublished data; Fig. 3a). Although reproductive success of the oyster remains high (as measured by larval oyster, or spat, set on oyster reefs and other suitable substrates; Maryland Department of Natural Resources, Oxford, Maryland, unpublished data), populations have suffered from harvest to low levels, two parasitic diseases (Dermo and MSX), habitat loss (including decreased water quality), and predation.

Blue Crabs

Blue crab populations in the Chesapeake Bay, as indicated by commercial landings data, vary from year to year, making trends less apparent than those of other bay species (Lipcius and Van Engel 1990; National Marine Fisheries Service, Annapolis, Maryland, unpublished data; Fig. 3b). Populations appear to follow a 7-12 year cycle and may be in the "trough" of this cycle at present. This perception and increasing annual harvests as fishery efforts shift to crabs from other species have prompted Maryland and Virginia to begin to regulate the blue crab fishery.

Striped Bass

Probably the most monitored fish species in the bay, striped bass populations have increased about 25% a year since 1984, after falling to low levels in the early 1980's (Gibson 1993; Fig. 3c). Increases are at least partially attributed to a moratorium on harvest from 1985 to 1989 to allow improvement of the age and sex structure of the spawning stock. The 1993 young-of-the-year index, a measure of numbers of juvenile fish entering the population, is the highest on record (National Marine Fisheries Service, Annapolis, Maryland, unpublished data) and may be related to the timing of high freshwater flows, nutrient inputs, and increases in planktonic prey (Blankenship 1994), which may interact to allow large numbers of young fish to survive after hatching.

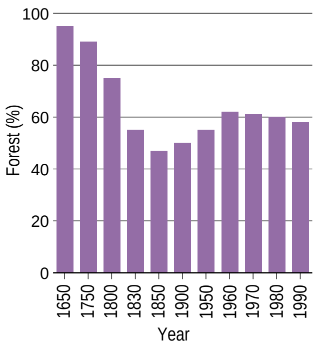


Fig. 2. Forest cover in the Chesapeake Bay watershed, 1650-1990.

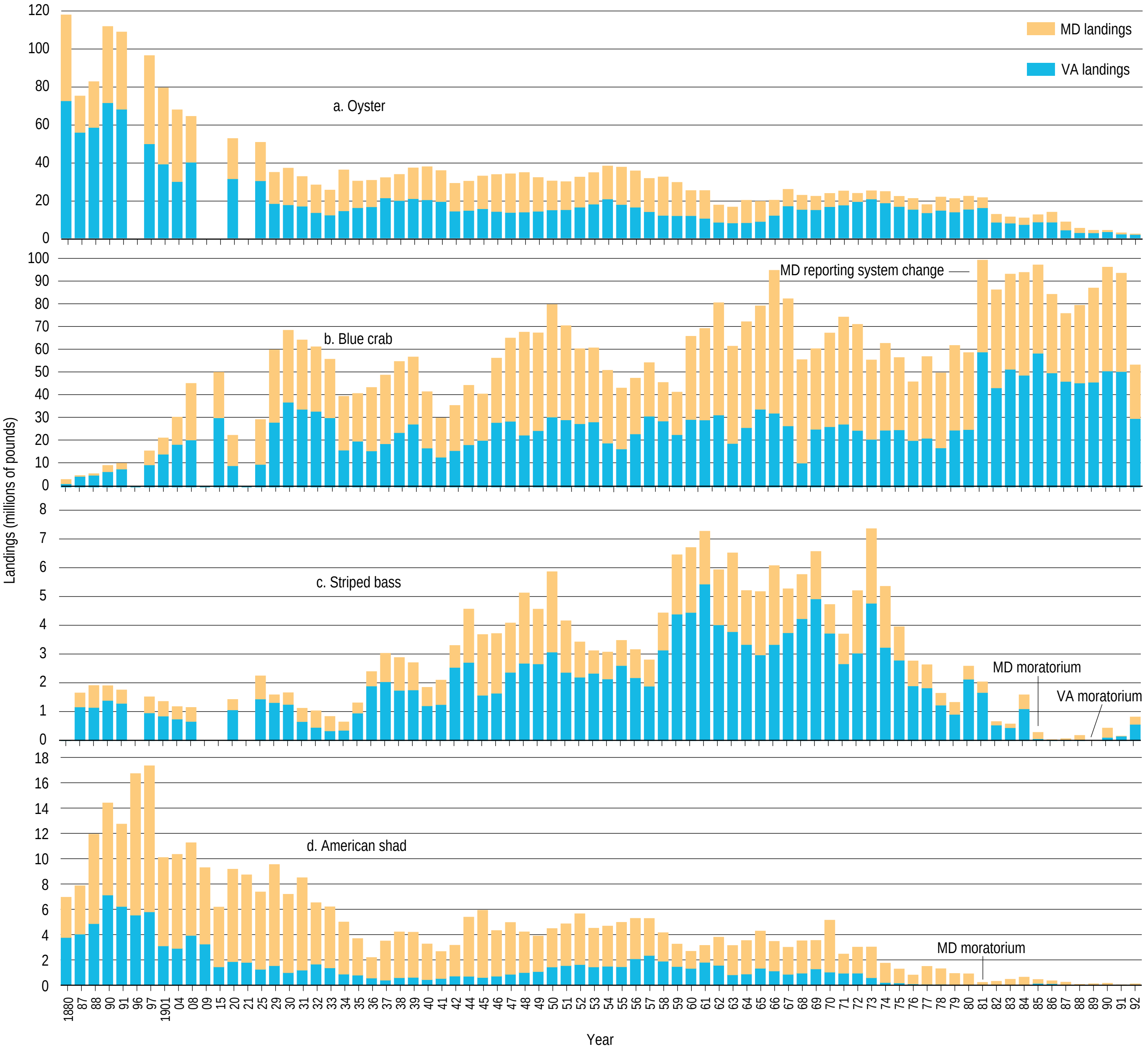


Fig. 3a-d. Commercial landings in the Chesapeake Bay of (a) oysters, 1880-1992; (b) blue crab, 1880-1992 (population increases in the 1980's are partially due to mandatory catch reporting requirements); (c) striped bass, 1887-1992; and (d) American shad, 1880-1992.

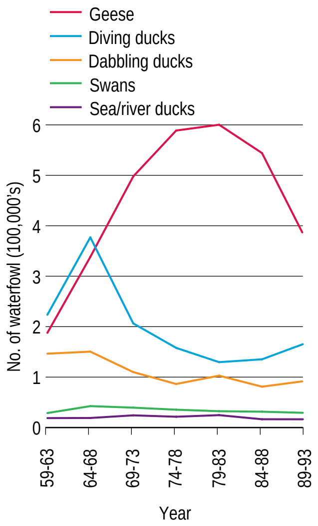


Fig. 4. Trends in waterfowl abundance in the Chesapeake Bay, based on 5-year running means, 1959-93.

American Shad

Like striped bass, American shad have declined in Chesapeake Bay in recent decades; unlike the stripers, this species has not shown a strongly positive population response despite moratoria on fishing in Maryland and Virginia. Long-term trends show a drastic decline in fishery landings to the point of almost total disappearance in the bay (National Marine Fisheries Service, Annapolis, Maryland, unpublished data; Fig. 3d). This decline has been related to blockages of spawning streams by dams, overharvest, and pollution (Blankenship 1993). Population estimates in 1992 and 1993 for the upper bay, where shad are counted during their upstream migration to the Susquehanna River, show a reversal of a recent positive trend, for reasons yet unknown.

Waterfowl

Midwinter surveys estimate an average of more than one million waterfowl along the Atlantic Flyway winter in Chesapeake Bay each year (USFWS, Chesapeake Bay Field Office, Annapolis, Maryland, unpublished data). Of the 28 species of ducks, geese, and swans represented in this total, some are declining in abundance, whereas others show increasing or variable trends in abundance (Fig. 4; Table). In general, duck numbers declined and goose populations increased since the late 1950's as submersed aquatic vegetation and other duck foods dwindled and changing farming practices left more grain in fields for geese. Recently, geese have also declined as excessive harvest and poor production on northern breeding grounds reduced their numbers. Their distribution along the Atlantic Flyway has also shifted to the north. Mallards (*Anas platyrhynchos*) and introduced mute swans (*Cygnus olor*) have shown moderate increases, but many other species, including American black duck (*Anas rubripes*), wigeon (*A. americana*), northern pintail (*A. acuta*), canvasback (*Aythya valisineria*), and redhead (*A. americana*), have declined or stabilized at population levels substantially lower than in the 1950's.

Table. Trends for waterfowl in Chesapeake Bay, based on 5-year running means from 1959 to 1993 (USFWS, Chesapeake Bay Field Office, Annapolis, Maryland, unpublished data).

Group	Trend
Swans and geese	
Tundra swan	Variable
Mute swan	Increasing
Snow goose	Increasing
Canada goose	Decreasing
Brant	Variable
Dabbling ducks	
Mallard	Increasing
Black duck	Decreasing
Gadwall	Variable
Teal (blue- and green-winged)	Variable
American wigeon	Decreasing
Northern pintail	Decreasing
Northern shoveler	Variable
Bay (diving) ducks	
Canvasback	Decreasing
Redhead	Decreasing
Ring-necked duck	Variable
Scaup	Variable
River and sea ducks	
Goldeneye	Decreasing
Bufflehead	Increasing
Ruddy duck	Decreasing
Scoter	Decreasing
Oldsquaw	Decreasing
Mergansers	Increasing

References

Anderson, R.R., and R.T. Macomber. 1980. Distribution of submerged vascular plants, Chesapeake Bay, Maryland. Environmental Protection Agency, Chesapeake Bay Program, Annapolis, MD. 126 pp.

Blankenship, K. 1994. Bay bounces back from record-setting spring "freshet." Bay Journal 3(10):1,7.

Blankenship, K. 1993. Biologists puzzled by sudden decline of East Coast shad. Bay Journal 3(9):1,6.

Chesapeake Bay Implementation Committee. 1988. The Chesapeake Bay Program: a commitment renewed. U.S. Environmental Protection Agency, Chesapeake Bay Program Office, Annapolis, MD. 88 pp.

Chesapeake Bay Program. 1993. Environmental indicators: measuring our progress. U.S. Environmental Protection Agency, Chesapeake Bay Program Office, Annapolis, MD.

Dahl, T.E. 1990. Wetlands losses in the United States 1780's to 1980's. U.S. Department of the Interior, Fish and Wildlife Service, Washington, DC. 21 pp.

Flemer, D.A., G. Mackiernan, W. Nehlsen, and V. Tippe. 1983. Chesapeake Bay: a profile of environmental change. U.S. Environmental Protection Agency, Chesapeake Bay Program Office, Annapolis, MD. 200 pp.

Gibson, M.R. 1993. Historical estimates of fishing mortality on the Chesapeake Bay striped bass stock using separable virtual population analysis applied to market class catch data. Report to the Atlantic States Marine Fisheries Commission Striped Bass Technical Committee. Atlantic States Marine Fisheries Commission, Washington, DC. 20 pp.

Kennedy, V.S. 1991. Eastern oyster. Pages 3-1 to 3-20 *in* Habitat requirements for Chesapeake Bay living resources. 2nd ed. Living Resources Subcommittee, Chesapeake Bay Program. U.S. Fish and Wildlife Service, Annapolis, MD.

Lipcius, R.M., and W.A. Van Engel. 1990. Blue crab population dynamics in Chesapeake Bay: variation in abundance (York River, 1972-1988) and stock-recruit functions. Bull. of Marine Science 46:180-194.

Orth, R.J., and K.A. Moore. 1983. Chesapeake Bay: an unprecedented decline in submerged aquatic vegetation. Science 222:51-53.

Orth, R.J., J.F. Nowak, G.F. Anderson, and J.R. Whiting. 1993. Distribution of submerged aquatic vegetation in the Chesapeake Bay and tributaries and Chincoteague Bay—1992. U.S. Environmental Protection Agency, Chesapeake Bay Program Office, Annapolis, MD. 161 pp.

Orth, R.J., J. Simons, R. Allaire, V. Carter, L. Hindman, K. Moore, and N. Rybick. 1985. Distribution of submerged aquatic vegetation in the Chesapeake Bay and tributaries—1984. EPA Final Report. Chesapeake Bay Program, Annapolis, MD. 155 pp.

Stevenson, J.C., and N.M. Confer. 1978. Summary of available information on Chesapeake Bay submerged vegetation. U.S. Fish and Wildlife Service Biological Rep. FWS/OBS-78/66. 335 pp.
Tiner, R.W., and J.T. Finn. 1986. Status and recent trends of wetlands in five mid-Atlantic states: Delaware, Maryland, Pennsylvania, Virginia, and West Virginia. U.S. Fish and Wildlife Service, Region 5, Newton Corner, MA, and U.S. Environmental Protection Agency, Region 3, Philadelphia, PA. Cooperative Tech. Publ. 40 pp.

For further information:

Edward Pendleton
National Biological Service
Leetown Science Center
1700 Leetown Rd.
Kearneysville, WV 25430

Florida Manatees

by
Lynn W. Lefebvre
Thomas J. O'Shea
National Biological Service

The endangered Florida manatee (*Trichechus manatus latirostris*) is a survivor. It is one of only three living species of manatees which, along with their closest living relative, the dugong (*Dugong dugon*), make up the Order Sirenia. This taxonomic distinctiveness reflects their evolutionary and genetic uniqueness. Sirenians are the only herbivorous marine mammals; manatees feed on seagrasses; freshwater plants, including nuisance species such as hydrilla and water hyacinth; and even some shoreline vegetation. Because manatees depend on marine, estuarine, and freshwater ecosystems, our efforts to protect them necessitate protection of aquatic resources.

Life-history Research

Major efforts have concentrated on better quantification of Florida manatee populations, emphasizing reproduction, population size, and mortality. Most of the information on manatee reproduction (Table) comes from long-term studies based on recognizable individuals at winter aggregation sites (e.g., Rathbun et al. in press). Florida manatees are at the northern limit of the species' range and must seek warmer waters when water temperatures drop below about 20 ºC. Natural springs, such as those found in Crystal River on the west coast and Blue Spring on the St. Johns River, and discharges from industrial plants provide warmwater refuges for hundreds of manatees during cold periods.

Individual manatees are recognized at these sites largely through their unique scar patterns, caused by boat strikes (Figs. 1a and 1b). National Biological Service personnel have cataloged almost 1,000 recognizable manatees and maintained their sighting histories in a computer-based system (Beck and Reid in press).

Table. Estimated population traits of the Florida manatee based on long-term life-history research (data are from the National Biological Service and the Florida Department of Environmental Protection).

Life-history trait	Data
Maximum life expectancy	60 years
Gestation	11-13 months
Litter size	1
% twins	
Blue Spring	1.79%
Crystal River	1.40%
Sex ratio at birth	1:1
Calf survival	
Blue Spring	60%
Crystal River	67%
Annual adult survival	
Atlantic coast	90%
Blue Spring	96%
Crystal River	96%
Age of first reproduction (female)	3-4 years
Mean age first reproduction (female).	5 years
Spermatogenesis (male).	2-3 years
Proportion pregnant (female)	33% salvaged carcasses
Blue Spring	41%
Proportion nursing 1st-year calves during winter season	36% (mean)
Blue Spring	30%
Crystal River	36%
Atlantic coast	38%
Calf dependency	1.2 years
Interbirth interval	2.5 years
Highest number of births	May-September
Highest frequency in mating herds	February-July
No. salvaged carcasses	2,219 (1974-93)
No. documented in ID catalog	> 950 (1975-February 1994)
Highest count (aerial surveys)	1,856 in January 1992


Courtesy J.P. Reid, NBS

Fig. 1a. Female manatee and calf. Individuals can be identified by their unique scar patterns; scars are usually the result of collisions with boats.


Courtesy R.K. Bonde, NBS

Fig. 1b. A manatee often bears scars from multiple boat collisions.

Estimates of manatee reproductive traits are similar across study sites (Table), despite large habitat differences among study areas. There is also agreement in reproductive estimates obtained from salvaged carcasses (Marmontel in press), indicating that Florida manatees have probably achieved a maximum level of reproduction (O'Shea and Hartley in press).

Aerial Surveys

The population of Florida manatees cannot be directly estimated because they are often difficult to see. They occupy waters that may be turbid or obscured by overhanging branches; they can move long distances between counting areas over a short time; and many environmental factors, particularly temperature, influence their distribution and behavior (Lefebvre et al. in press).

Three statewide aerial surveys, coinciding with maximum manatee use of winter aggregation sites, resulted in counts of 1,268 (January 1991), 1,465 (February 1991), and 1,856 (January 1992; Ackerman in press). The differences in these counts are thought to reflect the influence of different environmental conditions, not changes in population size. Manatee presence at winter aggregation sites varies within and between winters, depending upon the pattern and severity of winter cold fronts.

Garrott et al. (1994) developed a population index by using a temperature covariate to model a simple linear trend in annual aerial survey data from the winters of 1977-78 through 1991-92. Their analyses showed an increasing trend in the temperature-adjusted counts of 7%-12% annually on the Atlantic coast, but the degree to which these increases are related to true population growth is unknown. No pronounced temporal trend was detected at the largest aggregation site on the southwest coast.

While this result seems promising because it shows no evidence for major declines, it is tempered by other factors. The number of human-related manatee deaths on the Atlantic coast is more than twice as high as on the gulf coast (Ackerman et al. in press). This fact is reflected in the lower survival rate of adult manatees on the Atlantic coast than at Crystal River and Blue Spring (O'Shea and Langtimm in press). Reynolds and Wilcox (1994) found that the number of calves sighted at winter aggregation sites has decreased since 1982, and that in three recent winters, the percentage of manatees sighted that are calves has also decreased. They note that mortality of calves at or near time of birth is the fastest-growing type of manatee mortality, thus the downward trend in aerial survey calf counts is a cause for concern and further investigation.

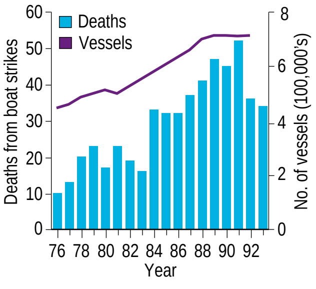


Fig. 2. Number of manatee deaths from watercraft collisions and number of Florida registered vessels from 1976-93 (data from National Biological Service and Florida Department of Environmental Protection).

Recovery Criteria

Species recovery criteria for the Florida manatee are three-fold: the population trend must be stable or increasing; mortality must be stable or declining; and threats to manatee habitat must be under control (USFWS 1989). Better population and life-history data suggest a greater potential for increase and higher numbers than previously recognized, and strong steps taken by local, state, and federal governments are increasing the number and area of sanctuaries and slow boat-speed zones. These steps may reduce mortality if they are continued and expanded, allowing the population to recover more quickly.

Management has focused on ways to reduce human-related mortality. Of greatest concern has been an increase over the years in the number of human-caused deaths, particularly those caused by collisions with boats (Fig. 2). Boat strikes account for 78% of human-related manatee mortality and 25% of all documented deaths (Wright et al. in press). A moderate reduction in the number of boat-related deaths in the last 2 years caused optimism; however, watercraft collisions accounted for 49 manatee deaths in 1994, almost matching the record number of 51 in 1991 (Fig. 2).

Habitat Threats

Habitat threats are far from under control, however. Florida has one of the fastest-growing human populations in the nation, with an estimated net gain of close to 1,000 people per day (Fernald et al. 1992). Much growth has occurred along the coast, with inevitable consequences for coastal habitats. For example, about a third of the 600,000 ha (1.5 million acres) of seagrass meadows present in coastal Florida in the 1940's no longer exist (Lewis 1987). One of the most important regions for manatees on the Atlantic coast is the Indian River Lagoon. Over the past 20 years, losses of submerged aquatic vegetation in some areas of the lagoon have exceeded 95% (Busby and Virnstein 1993). Submerged freshwater plants have also been affected adversely by increases in turbidity and nutrients.

Debris, particularly monofilament line, plastics, and unattended fishing nets and ropes, directly threatens manatees, who may ingest or become entangled in these materials (Beck and Barros 1991). Manatees are also vulnerable to natural and human-caused catastrophes, such as disease and oil spills, particularly when the animals are concentrated at winter aggregation sites.

Future

Population and life-history information suggests that the potential long-term viability of the Florida manatee population is good, provided that strong efforts are continued to curtail mortality, habitat quality is maintained or improved, and steps are taken to offset potential catastrophes.

References

Ackerman, B.B. Ongoing manatee aerial survey programs—a progress report. *In* T.J. O'Shea, B.B. Ackerman, and H.F. Percival, eds. Population biology of the Florida manatee (*Trichechus manatus latirostris*). National Biological Service, Biological Rep. Series. In press.

Ackerman, B.B., S.D. Wright, R.K. Bonde, D.K. Odell, and D.J. Banowetz. Trends and patterns in manatee mortality in Florida, 1974-1992. *In* T.J. O'Shea, B.B. Ackerman, and H.F. Percival, eds. Population biology of the Florida manatee (*Trichechus manatus latirostris*). National Biological Service, Biological Rep. Series. In press.

Beck, C.A., and N.B. Barros. 1991. The impact of debris on the Florida manatee. Marine Pollution Bull. 22(10):508-510.

Beck, C.A., and J.P. Reid. An automated photo-identification catalog for manatee life history studies. *In* T.J. O'Shea, B.B. Ackerman, and H.F. Percival, eds. Population biology of the Florida manatee (*Trichechus manatus latirostris*). National Biological Service, Biological Rep. Series. In press.

Busby, D.S., and R.W. Virnstein. 1993. Executive summary. Pages iii-viii *in* L.J. Morris and D.A. Tomasko, eds. Submerged aquatic vegetation and photosynthetically active radiation. Special Publ. SJ93-SP13, St. Johns River Water Management District, Palatka, FL.

Fernald, E.A., E.D. Purdum, J.R. Anderson, Jr., and P.A. Krafft. 1992. Atlas of Florida. University Press of Florida, Gainesville, FL. 280 pp.

Garrott, R.A., B.B. Ackerman, J.R. Cary, D.M. Heisey, J.E. Reynolds III, P.M. Rose, and J.R. Wilcox. 1994. Trends in counts of Florida manatees at winter aggregation sites. Journal of Wildlife Management 58(4):642-654.

Lefebvre, L.W., B.B. Ackerman, K.M. Portier, and K.H. Pollock. Aerial surveys for estimating manatee population size and trend—problems and prospects. *In* T.J. O'Shea, B.B. Ackerman, and H.F. Percival, eds. Population biology of the Florida manatee (*Trichechus manatus latirostris*). National Biological Service, Biological Rep. Series. In press.

Lewis, R.R., III. 1987. The restoration and creation of seagrass meadows in the Southeast United States. Pages 153-173 *in* M.J. Durako, R.C. Phillips, and R.R. Lewis III, eds. Proceedings of the Symposium on Subtropical-tropical Seagrasses of the Southeastern United States, 12 August 1985. Florida Marine Research Publications Number 42, St. Petersburg, FL.

Marmontel, M. Age and reproductive parameter estimates in female Florida manatees. *In* T.J. O'Shea, B.B. Ackerman, and H.F. Percival, eds. Population biology of the Florida manatee (*Trichechus manatus latirostris*). National Biological Service, Biological Rep. Series. In press.

O'Shea, T.J., and W.C. Hartley. Longitudinal studies of manatee reproduction and early age survival at Blue Spring, upper St. Johns River, Florida. *In* T.J. O'Shea, B.B. Ackerman, and H.F. Percival, eds. Population biology of the Florida manatee (*Trichechus manatus latirostris*). National Biological Service, Biological Rep. Series. In press.

O'Shea, T.J., and C.A. Langtimm. Adult survival estimates for Florida manatees at Crystal River, Blue Spring, and the Atlantic coast. *In* T.J. O'Shea, B.B. Ackerman, and H.F. Percival, eds. Population biology of the Florida manatee (*Trichechus manatus latirostris*). National Biological Service, Biological Rep. Series. In press.

Rathbun, G.B., J.P. Reid, R.K. Bonde, and J.A. Powell. Reproduction in free-ranging Florida manatees (*Trichechus manatus latirostris*). *In* T.J. O'Shea, B.B. Ackerman, and H.F. Percival, eds. Population biology of the Florida manatee (*Trichechus manatus latirostris*). National Biological Service, Biological Rep. Series. In press.

Reynolds, J.E., III, and J.R. Wilcox. 1994. Observations of Florida manatees (*Trichechus manatus latirostris*) around selected power plants in winter. Marine Mammal Science 10(2):163-177.

USFWS. 1989. Florida Manatee (*Trichechus manatus latirostris*) Recovery Plan. Prepared by the Florida Manatee Recovery Team for the U.S. Fish and Wildlife Service, Atlanta, GA. 98 pp.

Wright, S.D., B.B. Ackerman, R.K. Bonde, C.A. Beck, and D.J. Banowetz. Analysis of watercraft-related mortality of manatees in Florida, 1979-1991. *In* T.J. O'Shea, B.B. Ackerman, and H.F. Percival, eds. Population biology of the Florida manatee (*Trichechus manatus latirostris*). National Biological Service, Biological Rep. Series. In press.

For further information:

Lynn W. Lefebvre
National Biological Service
Sirenia Project
412 NE 16th Ave., Rm 250
Gainesville, FL 32601

Gulf of Mexico Coastal Wetlands: Case Studies of Loss Trends

by
James B. Johnston
Mary C. Watzin
John A. Barras
Lawrence R. Handley
National Biological Service

The Gulf of Mexico's coastal wetlands are of special interest because the gulf is an exceptionally productive sea that yields more than 1.1 billion kg (2.5 billion lb) of fish and shellfish annually and contains four of the top five fishery ports in the nation by weight (U.S. Environmental Protection Agency 1988). The volume of commercial shrimp landings in the gulf has been statistically related to the areal coverage of gulf coastal wetlands (and seagrass beds) that provide crucial nursery habitat to the young (Turner 1977). Coastal wetlands (particularly salt marshes and mangroves) and associated shallow waters function similarly in support of many fish species of commercial interest (Seaman 1985). The gulf wetlands are also well known for their large populations of wildlife, including shorebirds, colonial nesting birds, and 75% of the migratory waterfowl traversing the United States (Duke and Kruczynski 1992). The extensive coastal wetlands that remain along the gulf make up about half of the nation's total wetland area (NOAA 1991).

General Trends

The National Oceanic and Atmospheric Administration (NOAA 1991) examined the areal extent and distribution of gulf coast coastal wetlands in the mid-1980's by using aerial photographs and maps from 1972 to 1984 (28% from 1979 and 42% from 1980 or later). Summaries of NOAA's data are shown in the Table for three wetland categories: marshes (fresh, brackish, and salt marshes), estuarine shrub-scrub (mangroves), and freshwater forested/shrub-scrub wetlands. Even though wetland area has diminished greatly along the gulf coast during the last 30 years, about 1.3 million ha (3.3 million acres) still remain in these three categories.

Louisiana has the greatest area of coastal wetlands with 55% of the total, followed by Florida (18%), Texas (14%), and Mississippi (2%). Louisiana contains 69% of the marshes, while Florida has 97% of the estuarine scrub-shrub, most of which is mangrove. Of the three wetland types, 80% is marsh, 19% estuarine scrub-shrub, and 1% forested wetland.

Table. Total area (hectares) of selected vegetated wetlands by state for the Gulf of Mexico (from NOAA 1991).*

State	Marsh	Estuarine scrub-shrub	Fresh forested and scrub-shrub	Total	% of total**
Texas	183,900	1,100	3,000	188,000	14
Louisiana	723,500	4,100	1,900	729,500	55
Mississippi	23,800	400	-	24,200	2
Alabama	10,400	1,100	800	12,300	1
Florida	108,100	248,400	7,400	363,900	28
Total	1,049,700	255,100	13,100	1,317,900	-

* Calculated based on Fish and Wildlife Service's National Wetlands Inventory maps. Area originally reported as acres x 100; hectare = 2.471 acres.
** Fractions of percent rounded to the next highest whole percent (1.6 = 2.0%).

Because of the age of the photographs used by NOAA and because national trends suggest that the area of most wetland types is still declining (Frayer et al. 1983), the wetland statistics presented by NOAA may be overestimates. No current studies summarize coastal wetland area or loss rates for the entire Gulf of Mexico region; therefore, four case studies conducted by the National Biological Service's Southern Science Center, the U.S. Fish and Wildlife Service's National Wetland Inventory, and their partners are presented to depict status and trends from the 1950's to 1970's and the 1970's to the late 1980's. The areas chosen (Fig. 1) represent a cross-section of current trends.

Coastal Wetland Loss: Gulf of Mexico Case Studies

Galveston Bay

White et al. (1993) reported both gains and losses in Galveston Bay wetlands from the 1950's to 1989, but the net trend was one of wetland loss, going from 69,800 ha (171,000 acres) in the 1950's to 56,100 ha (138,600 acres) in 1989. The rate of loss decreased over time from about 405 ha (1,000 acres) per year between 1953 and 1979 to about 283 ha (700 acres) per year between 1979 and 1989. The rate of loss from 1979 to 1989 would probably be lower if inaccuracies in wetland interpretation of the 1979 photographs could be taken into account. In general, freshwater scrub-shrub habitats decreased in area from the 1950's to 1979 and 1989, while forested wetlands increased. Marshes (fresh and non-fresh) decreased from about 67,000 ha (165,500 acres) in the 1950's to about 52,800 ha (130,400 acres) in 1989, producing a total net marsh loss of about 21% of that resource.

The five key factors contributing most to wetlands decline in the Galveston Bay since the 1950's are (1) industrial development; (2) urbanization; (3) navigation channels; (4) flood control and multipurpose water projects to meet Houston's future water demand, especially upstream impoundments on the Trinity and San Jacinto rivers; and (5) pollution due to agricultural runoff despite the diminished acreage lost to agricultural expansion. It should be noted that human-induced subsidence due to industrial development (oil and gas activities) and urbanization (groundwater withdrawals) are considered in this analysis (D. Whitehead, U.S. Fish and Wildlife Service, personal communication).

Coastal Louisiana

Coastal wetland loss for Louisiana represents 67% of the nation's total loss. For the time period 1978-90, the loss was 177,625 ha (290,432 acres), representing an annual loss rate of 9,802 ha/yr (24,203 acres/yr) for this 12-year period; that is equal to 97.9 km^2 or 37.8 m^2/yr. For the time period 1956-78, net wetland loss was even greater, 267,800 ha (661,700 acres), representing a loss rate of 12,170 ha/yr (30,000 acres/yr); that is equal to 121.7 km^2 or 47 mi^2/yr.

Although much of this loss is only indirectly linked to human activities, most of the net current, catastrophic wetland loss is primarily the result of altered hydrology stemming from navigation, flood control, and mineral extraction and transport projects (Sasser et al. 1986; Louisiana Wetland Protection Panel 1987; Turner and Cahoon 1988). These operations do not always destroy wetlands directly, but they do amplify tidal forces in historically low-energy systems, which upsets the balance of subsidence and accretion, reduces nutrient and sediment influx, decreases freshwater retention, and increases the levels of salt, sulfate, and other substances potentially toxic to indigenous plant species (Good 1993).

Current wetland losses are concentrated in the southern Deltaic Plain (78%; Fig. 1). In this region, losses are especially severe in the fringing marshes of the Terrebonne and Barataria basins (Figs. 2 and 3). Previous losses in the Deltaic Plain occurred primarily in large areas of interior lands. In the Chenier Plain (Fig. 1), loss rates were more constant (22%); many of the larger areas of loss there seem related to impounded areas with managed water levels.

The Barataria and Terrebonne basins suffer the highest land loss rates (all land but mostly wetlands) in Louisiana (2,880 ha/yr [7,120 acres/yr] and 2,630 ha/yr [6,500 acres/yr], respectively), accounting for 64% of all land loss in the 1978-90 period. In contrast, this area accounted for only 43% of all loss in the 1956-78 period. The Mermentau and Sabine basins (Fig. 2) have the next highest loss rates (1,080 ha/yr [2,670 acres/yr] and 660 ha/yr [1,630 acres/yr]), with losses largely confined to the northern and central portions, except for shoreline erosion along the Mermentau Basin's coastline. Loss rates within the Teche-Vermilion, Mississippi, Breton Sound, and Pontchartrain basins (Fig. 2) are all less than 930 ha/yr (2,300 acres/yr), which seems to indicate more stable environments. The Atchafalaya and Pearl River basins (Fig. 2) experienced losses of less than 130 ha/yr (321 acres/yr). In summary, land loss rates in coastal Louisiana, although decreasing, remain high for the 1978-90 period.

The National Biological Service is providing future land loss updates for coastal Louisiana by using Landsat Thematic Mapper satellite imagery on a 3-year basis.

Mobile Bay

Non-freshwater marshes surrounding Mobile Bay declined by more than 4,047 ha (10,000 acres) from 1955 to 1979, representing a loss of 35% (Roach et al. 1987). Freshwater marshes in all of coastal Alabama declined by about 69% from 1955 to 1979. More than 2,500 ha (6,200 acres) were lost during that time (Roach et al. 1987).

When comparing these data to 1988 wetland habitat maps prepared for upper Mobile Bay, it appears that in this portion of the bay no additional net loss of non-freshwater marsh has occurred since 1979. Some marsh has obviously continued to be lost in certain areas, primarily because of dredge disposal associated with navigation and industry. These losses, though, seem to have been offset by the growth of emergent marsh in existing spoil sites (Watzin et al. 1994).

The Southern Science Center's 1988 areal estimates show a substantial increase of 189 ha (467 acres) in freshwater marsh from 1979 to 1988 in upper Mobile Bay. Further investigation revealed that some of this gain was the result of the growth of emergent vegetation in existing disposal areas and in ditches along railroads and highways. Because of disparities in photointerpretation between dates, it is also quite likely that some of these differences are simply due to mapping errors and differences in mapping technique (Watzin et al. 1994).

As a result of mapping errors associated with interpreting forested and scrub-shrub wetlands in the 1956 photographs, Roach et al. (1987) had little faith in the quantitative estimate of change between 1956 and 1979 for these wetland types. The Southern Science Center's 1988 wetland area figures for forested wetlands appear relatively accurate; they indicate that about 486 ha (1,201 acres) of forested wetlands (2.7%) were lost in upper Mobile Bay between 1979 and 1988. These losses can be attributed to conversion of forested habitats to scrub-shrub areas (e.g., clearcutting associated with timber harvest), small impoundments, and commercial and residential development (Watzin et al. 1994).

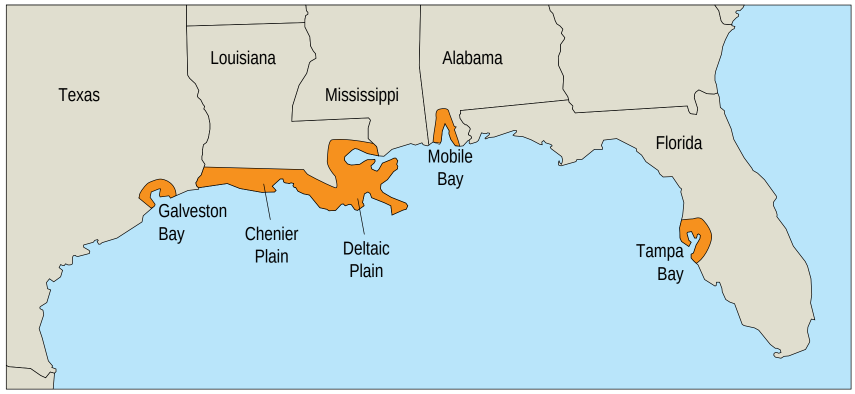
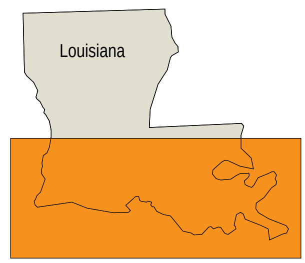


Fig. 1. Locations of wetland loss study sites along the Gulf of Mexico region.

Tampa Bay

Haddad (1989) reported emergent wetlands decreased from 29,000 ha (71,700 acres) in the 1950's to 23,900 ha (59,100 acres) in 1982, about an 18% loss. Mangroves decreased from 8,629 ha (21,320 acres) to 8,032 ha (19,847 acres), a decline of about 7%. Salt marshes declined from 2,063 ha (5,097 acres) to 1,423 ha (3,538 acres), or a loss of 30%. Freshwater wetlands decreased 21% from 18,335 ha (45,305 acres) to 14,440 ha (35,681 acres).

Lewis et al. (1985) estimate that 44% of the salt marsh and mangrove has been lost in Tampa Bay since the late 1800's. Although their numbers and those of Haddad (1989) are not readily



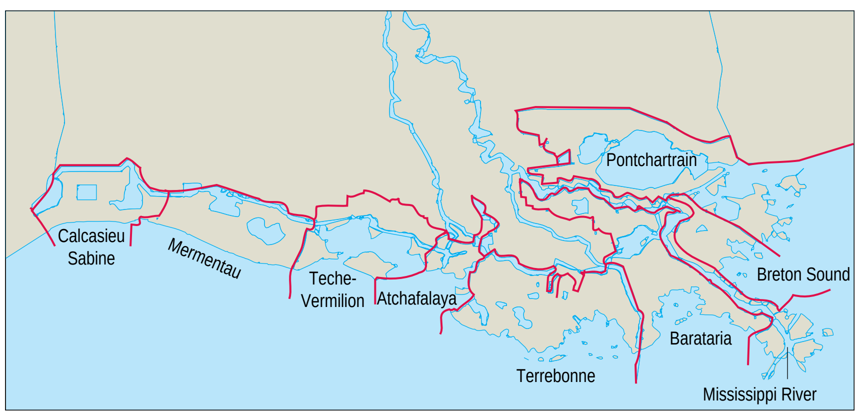


Fig. 2. Coastal Louisiana basins as defined in the Coastal Wetlands Planning, Protection, and Restoration Act Plan.

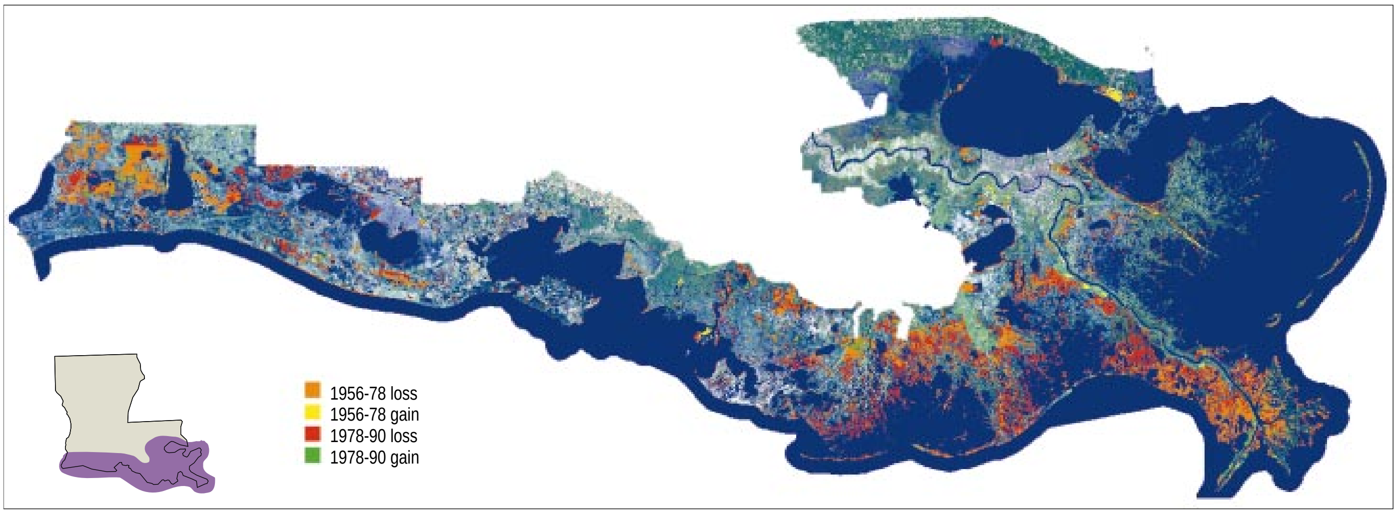


Fig. 3. Coastal landloss in Louisiana and elsewhere is analyzed by using computerized geographic information systems that produce graphics such as this map.

comparable because of differences in time frame, methodology, vegetation classification, and area mapped, the results taken together confirm that significant losses of wetland habitat have occurred. Marsh and mangrove losses are the product of dredge and fill activities that are now under strict regulatory control; although permitted dredging continues, protective measures exist to minimize loss that is not for public benefit.

Future Concerns

To protect the future of gulf coastal wetlands, status and trends over time must be continually recorded and noted in the scientific and public literature. Preliminary data from selected coastal areas studied in the 1980's show a reduced rate of wetland loss compared with earlier decades. While this is good news, the pressures of a continuously expanding human population make it unclear whether this trend will continue into the 21st century. Only additional monitoring data can answer this question.

For further information:

James B. Johnston
National Biological Service
Southern Science Center
700 Cajundome Blvd.
Lafayette, LA 70506

References

Duke, T.W., and W.L. Kruczynski, eds. 1992. Status and trends of emergent and submerged vegetated habitats of the Gulf of Mexico, USA. Gulf of Mexico Program, U.S. Environmental Protection Agency, John C. Stennis Space Center, MS. 161 pp.

Frayer, W.E., T.J. Monahan, D.C. Bowden, and F.A. Graybill. 1983. Status and trends of wetlands and deepwater habitats in the coterminous United States, 1950's to 1970's. Colorado State University, Department of Forest and Wood Sciences, Fort Collins. 32 pp.

Good, B. 1993. Louisiana's wetlands: combatting erosion and revitalizing native ecosystems. Restoration and Management Notes. 11:125-133.

Haddad, K.D. 1989. Habitat trends and fisheries in Tampa and Sarasota bays. Pages 113-128 *in* Tampa and Sarasota bays: issues, resources, status, and management. National Oceanic and Atmospheric Administration, Estuary-of-the-Month Seminar Series 11.

Lewis, R.R., III, M.J. Durako, M.D. Moffler, and R.C. Phillips. 1985. Seagrass meadows of Tampa Bay. Pages 210-246 *in* S. Treat, J. Simon, R. Lewis III, and R. Whitman, Jr., eds. Proceedings Tampa Bay Area Scientific Information Symposium. Florida Sea Grant Rep. 65.

Louisiana Wetland Protection Panel. 1987. Saving Louisiana's coastal wetlands: The need for a long-term plan of action. U.S. Environmental Protection Agency, EPA-230-02-87-026. 102 pp.

NOAA. 1991. Coastal wetlands of the United States: an accounting of a national resource base. National Oceanic and Atmospheric Administration Rep. 91-3. 59 pp.

Roach, E.R., M.C. Watzin, and J.D. Scurry. 1987. Wetland changes in coastal Alabama. Pages 92-101 *in* T.A. Lowery, ed. Symposium on the Natural Resources of the Mobile Bay Estuary. Alabama Sea Grant Extension Service, Mobile, AL. MASGP-87-007.

Sasser, C.E., M.D. Dozier, J.G. Gosselink, and J.M. Hill. 1986. Spatial and temporal changes in Louisiana's Barataria Basin marshes, 1945-1980. Environmental Management 10:671-680.

Seaman, W., Jr. 1985. Florida aquatic habitat and fishery resources. Florida Chapter of the American Fisheries Society, Kissimmee. 543 pp.

Turner, R.E. 1977. Intertidal vegetation and commercial yields of penaeid shrimp. Transactions of the American Fisheries Society, Kissimmee, FL. 543 pp.

Turner, R.E., and D. Cahoon, eds. 1988. Causes of wetlands loss in the coastal Gulf of Mexico. Vol 1. Executive summary. Minerals Management Service OCS Study/MMS 87-0119.

U.S. Environmental Protection Agency. 1988. The gulf initiative: protecting the Gulf of Mexico. John C. Stennis Space Center, MS.

Watzin, M.C., S. Tucker, and C. South. 1994. Environmental problems in the Mobile Bay ecosystem: the cumulative effects of human activities. U.S. Environmental Protection Agency Tech. Rep. In press.

White, W.A., T.A. Tremblay, E.G. Wermund, and L.R. Handley. 1993. Trends and status of wetland and aquatic habitats in the Galveston Bay system, Texas. The Galveston Bay National Estuary Program. GBNEP-31. 225 pp.

Seagrass Distribution in the Northern Gulf of Mexico

by
Lawrence R. Handley
National Biological Service

Seagrass ecosystems are widely recognized as some of the most productive benthic habitats in estuarine and nearshore waters of the gulf coast. Seagrass meadows provide food for wintering waterfowl and important spawning and foraging habitat for several species of commercially important finfish and shellfish. Physical structure provided by seagrasses affords juveniles refuge from predation and allows for attachment of epiphytes and benthic organisms. Seagrass communities also support several endangered and threatened species, including some sea turtles and manatees. Changes in seagrass distribution can reflect the health of a water body, and losses of seagrasses may signal water-quality problems in coastal waters. Losses of seagrasses in the northern Gulf of Mexico over the last five decades have been extensive—from 20% to 100% for most estuaries, with only a few areas experiencing increases in seagrasses.

Although often considered continuous around the entire periphery of the gulf, seagrasses exist only in isolated patches and narrow bands from Mobile Bay, Alabama, to Aransas Bay, Texas (Figure). This pattern of occurrence results from a combination of low salinities, high turbidity, and high wave energy in shallow waters. Seagrasses are more extensively developed from Mobile Bay to Florida Bay (Figure). Although freshwater submerged aquatic vegetation also occurs throughout gulf coast estuaries and river deltas, its distribution is not considered in this article.

Seagrass habitats in the Gulf of Mexico have declined dramatically during the past 50 years, mostly because of coastal population growth and accompanying municipal, industrial, and agricultural development. Although proximate causes of local declines can sometimes be identified, most habitat loss has resulted from widespread deterioration of water quality (Neckles 1993).

The total seagrass coverage in the shallow, clear waters in protected estuaries and nearshore waters of the Gulf of Mexico coastal states is estimated to be 1.02 million ha (2.52 million acres; Duke and Kruczynski 1992). About 693,000 ha (1.71 million acres) of seagrasses occur in waters of the Florida Big Bend and Florida Bay (Figure). The remaining 324,000 ha (800,000 acres) are within gulf estuaries, with about 95% in the estuarine areas of Florida and Texas. Florida Bay seagrass meadows occupy about 550,000 ha (1.36 million acres), while the seagrass meadows of the Florida Big Bend area cover about 300,000 ha (740,000 acres; Zieman and Zieman 1989).

Six species of seagrasses occur in the Gulf of Mexico: turtle grass (*Thalassia testudinum*), shoal grass (*Halodule wrightii*), manatee grass (*Syringodium filiforme*), star grass (*Halophila engelmanni*), *Halophila decipiens*, and widgeon grass (*Ruppia maritima*). The latter has a distribution in water with lower salinity, but is commonly reported in association with the seagrasses throughout the gulf coast.

Case Histories

Sarasota Bay

Between 1948 and 1974, South Sarasota and Roberts bays lost 193 ha (477 acres) or 25%; Dryman, Blackburn, Dona, and Roberts (a different Roberts Bay) bays lost 31 ha (77 acres) or 29%; and Lemon Bay lost 55 ha (136 acres) or 21% of seagrasses (Evans and Brungardt 1978). Losses have been attributed mainly to dredge-and-fill activities and decline in water quality (Wolfe and Drew 1990). Improved water quality in Little Sarasota Bay caused seagrasses to increase between 1948 and 1974 by 14 ha (34 acres) or 9%.

Tampa Bay

In Tampa Bay (Figure), turtle grass and shoalgrass are dominant, and widgeon grass, manatee grass, and star grass are also found. A historical estimate places 30,970 ha (76,527 acres) occurring within the shallow- water margins of Tampa Bay before human influence (ca. 1876; Lewis et al. 1985). Based on 1981 estimates of seagrass coverage, a reduction of 81% of seagrasses has occurred in Tampa Bay; 5,750 ha (14,208 acres) were present in 1981. The most striking decrease occurred between 1940 and 1963, when about 50% of the grass beds were lost (Lewis et al. 1985). During this period, Hillsborough Bay alone lost 94% of its grass beds, Old Tampa Bay lost 45%, and Tampa Bay proper lost 35%. These losses have been attributed primarily to direct dredging of grassbeds and major shoreline modifications through filling and siltation (Wolfe and Drew

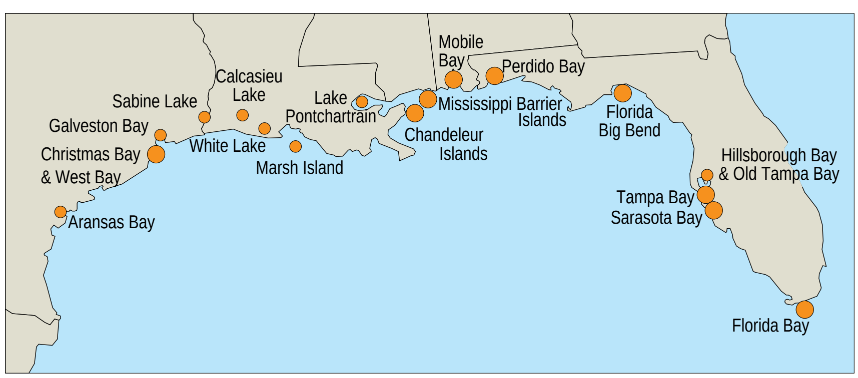


Figure. Study sites along the Gulf of Mexico region.

1990), which reduced light penetration and produced bottom sediments that are not conducive to seagrass growth and development.

Since 1963, grass beds have continued to decline in the upper bays of Tampa Bay to a point where Hillsborough Bay has lost the remaining 139 ha (343 acres) and Old Tampa Bay has lost nearly 60% (Figure). In lower Tampa Bay, grass beds have regained some area, increasing about 14% or 435 ha (1,075 acres). Tampa Bay as a whole has lost 5,984 ha (14,786 acres), or 51% of seagrasses between 1940 and 1983.

Perdido Bay

In Perdido Bay (Figure), widgeon grass and shoal grass are the predominant species. They grow in large and small beds, in numerous patches along shallow sandy reaches of the shoreline, and in large shallow flats in the lower bay and outlet. From 1940 to 1987, changes in the upper and middle parts of the bay consisted mainly of shifts in the locations of small meadows, with only minor changes in density. In the lower bay, some shifting of locations and changes in density occurred, and the coverage of seagrasses declined from 486 ha (1,201 acres) in 1940-41 to 251 ha (619 acres) in 1987. While the loss of seagrasses for the whole area was nearly 48%, some areas in U.S. Geological Survey quadrangles lost as much as 82% of the seagrasses delineated between 1940-41 and 1987. The changes in the extent of seagrasses are due to increased turbidity caused primarily by channel dredging and boat traffic; shoreline modifications; decreasing water quality and sedimentation from increasing farmlands and residential, commercial, and industrial development; and the high wave energy, overwash, sedimentation, erosion, and runoff from Hurricane Frederick in 1979.

Mississippi Gulf Coast

Along the Mississippi gulf coast, the Gulf Islands National Seashore includes most of the state's barrier islands (Figure). Manatee grass and shoal grass are the dominant seagrasses found in the shallow water on the northern side of the barrier islands, where they are protected from the high wave energy of the open gulf. Between 1956 and 1987, 416 ha (1,029 acres) of seagrasses declined to 140 ha (345 acres), a loss of 66%. The largest concentration of seagrasses was found on the north side of Horn Island, where 169 ha (417 acres) in 1956 declined to 56 ha (138 acres) by 1987, and to 6 ha (14 acres) by 1992.

Coastal Louisiana

Coastal Louisiana has a large amount of submerged aquatic vegetation but only a small portion is seagrasses (5,657 ha [13,974 acres] in 1988). Since the mid-1950's Louisiana has lost all of its seagrass in Lake Pontchartrain, in the Mississippi River Delta, behind the south coast barrier islands and Marsh Island, and in the coastal lakes (White, Calcasieu, and Sabine). The only remaining seagrass beds in coastal Louisiana exist in Chandeleur Sound behind the Chandeleur Islands. Turtle grass, shoal grass, manatee grass, widgeon grass, and star grass are present in the sandy sediments of the shallow backbarrier lagoon. These seagrass beds are virtually unaffected by human impacts because of their distance from the mainland, and they are controlled by high waves from chronic frontal passages and hurricanes causing overwash, erosion, sedimentation, changes in water depth, and turbidity. For example, Hurricane Camille in August 1969, with a storm surge of nearly 11 m (36 ft) on the Mississippi mainland, caused a loss of 530 ha (1,310 acres), or 22% of the seagrasses, on the North Islands (USGS 1:24,000 quadrangle), and a loss of 303 ha (749 acres) or 54% of the seagrasses, on Chandeleur Light (USGS 1:24,000 quadrangle).

The Chandeleur Islands (Figure) have been intensively mapped for wetland and seagrass habitats for 1978, 1982, 1987, and 1989. The areal extent of seagrasses for the Chandeleur Islands has remained relatively constant over the 11-year period, from 6,409 ha (15,831 acres) in 1978 to 5,657 ha (13,974 acres) in 1989. This constitutes a loss of only 12% of the seagrasses from 1978 to 1989, a period that had two hurricanes, two tropical storms, and countless cold fronts that influenced these islands.

Galveston Bay

In the Galveston Bay estuary (Figure), the distribution of seagrasses, predominantly shoal grass and widgeon grass, decreased in areal extent from more than 2,024 ha (5,000 acres) in the mid-1950's to about 283 ha (700 acres) in 1989, a loss of 1,471 ha (3,635 acres) or about 85% (White et al. 1993). The most significant losses were along the margins of western Galveston Bay and were related to the effects of subsidence and Hurricane Carla in 1970. In West Bay nearly 890 ha (2,200 acres) of seagrasses were completely lost, primarily through human activities including industrial, residential, and commercial development; wastewater discharges; chemical spills; and increased turbidity from boat traffic and dredging (Pulich and White 1991). In Christmas Bay, which has the largest concentration of seagrass beds in the

Galveston Bay estuarine system, seagrass areal extent declined from 121 ha (300 acres) in 1975 to 81 ha (200 acres) in 1987, but increased to 156 ha (385 acres) by 1989.

Conclusions

Losses of seagrasses in the northern Gulf of Mexico have been extensive over the last five decades, with losses varying 20%-100% for most estuaries of the northern Gulf of Mexico. Only a few locales have experienced increases in seagrasses. The high productivity of the Gulf of Mexico seagrass beds as spawning, nursery, food, and shelter areas increases the importance of the loss of this valuable habitat far beyond the areal extent of the resource. Regionwide, the loss of seagrasses is attributable to natural causes (hurricanes, cold-front storms, and increased or decreased salinities) and human-induced effects (increased turbidity and decreases in water quality resulting from dredging, boating activities, and other development pressures), which work in concert to deteriorate the environmental quality of the habitat.

References

Duke, T.W., and W.L. Kruczynski, eds. 1992. Status and trends of emergent and submerged vegetated habitats of the Gulf of Mexico. Gulf of Mexico Program, U.S. Environmental Protection Agency, Stennis Space Center, MS. 161 pp.

Evans, M., and T. Brungardt. 1978. Shoreline analysis of Sarasota County Bay systems with regard to revegetation activities. Pages 193-206 *in* D.P. Cole, ed. Proceedings of the Fifth Annual Conference on Restoration of Coastal Vegetation in Florida. Environmental Studies Center, Hillsborough Community College, Tampa.

Lewis, R.R., M.J. Durako, M.D. Moffler, and R.C. Phillips. 1985. Seagrass meadows of Tampa Bay—a review. Pages 210-246 *in* S.F. Treat, J.L. Simon, R.R. Lewis III, and R.L. Whitman, Jr., eds. Proceedings of the Tampa Bay Area Scientific Information Symposium, May 1982. University of South Florida, Tampa.

Neckles, H.A., ed. 1993. Seagrass monitoring and research in the Gulf of Mexico: draft report of a workshop held at Mote Marine Laboratory in Sarasota, Florida, January 28-29, 1992. National Biological Survey, National Wetlands Research Center, Lafayette, LA. 75 pp.

Pulich, W.M., Jr., and W.A. White. 1991. Decline of submerged vegetation in the Galveston Bay system: chronology and relationship to physical processes. Journal of Coastal Res. (4):1125-1138.

White, W.A., T.A. Tremblay, E.G. Wermund, Jr., and L.R. Handley. 1993. Trends and status of wetland and aquatic habitats in the Galveston Bay system, Texas. The Galveston Bay National Estuary Program, Publ. GBNEP-31. 225 pp.

Wolfe, S.H., and R.D. Drew, eds. 1990. An ecological characterization of the Tampa Bay watershed. U.S. Fish and Wildlife Service Biological Rep. 90(20). 334 pp.

Zieman, J.C., and R.T. Zieman. 1989. The ecology of the seagrass meadows of the west coast of Florida: a community profile. U.S. Fish and Wildlife Service Biological Rep. 85(7.25). 155 pp.

For further information:

Lawrence R. Handley
National Biological Service
Southern Science Center
700 Cajundome Blvd.
Lafayette, LA 70506

Seagrass Meadows of the Laguna Madre of Texas

by
Christopher P. Onuf
National Biological Service

A series of lagoons forms an almost continuous fringe of water behind coastal barriers for 500 km (310 mi) from Galveston Bay, Texas, to the Mexican border (Fig. 1). At the northeast end, where river discharge and precipitation greatly exceed evaporation from the embayments, fringing marshes are the dominant wetland type. Toward the southwest, freshwater inputs decrease, fringing marshes are replaced by wind-tidal flats that support highly productive algal mats during periodic inundation, and seagrasses dominate the shallow waters of the embayments (Table).

Seagrasses are so prevalent in Laguna Madre that they define the structure of the physical environment, as well as being the source of biological production for the ecosystem. Consequently, seagrass meadows serve a critical nursery function in support of the region's rich fisheries, and one waterfowl species has established an exclusive dependence on Laguna Madre and its most common seagrass. More than 75% of the world population of redhead ducks (*Aythya americana*) winters in the greater Laguna Madre ecosystem (inclusive of the

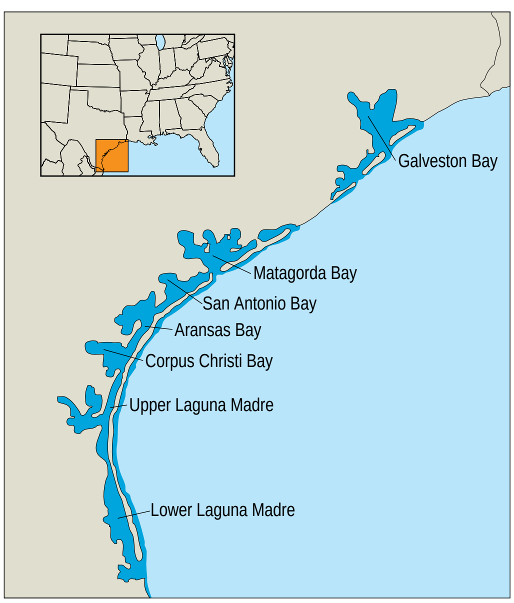


Fig. 1. Major bay systems along the Texas coast.

Table. Seagrass cover in bays of the Texas coast.

Bay system	Bottom vegetated (%)
Galveston Bay System*	0.3
Matagorda Bay System*	1.1
San Antonio Bay System**	5.0
Aransas-Copano Bay System**	5.2
Corpus Christi Bay System**	12.1
Upper Laguna Madre***	75.2
Lower Laguna Madre***	70.5

* Adair et al. 1994.
** Adair and Moore 1990.
*** Quammen and Onuf 1993.

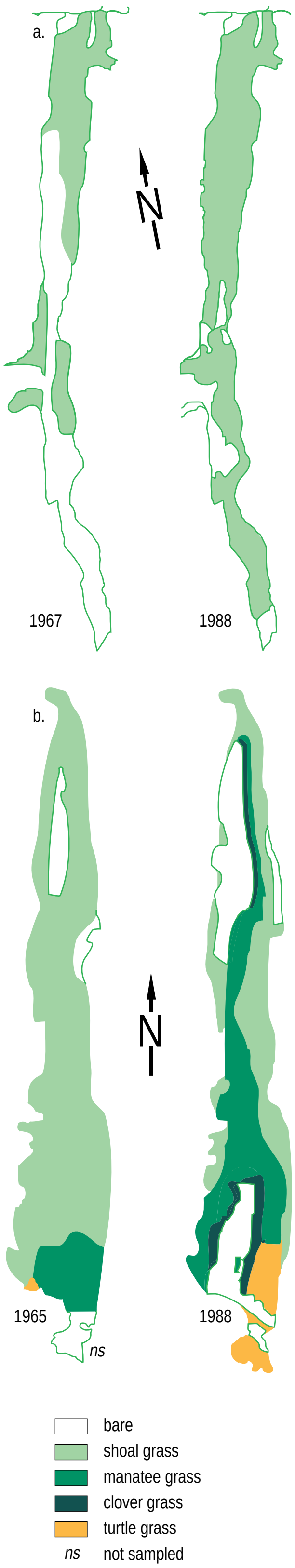


Fig. 2. Dominant cover types in the continuously submerged portions of upper (a) and lower (b) Laguna Madre.

Laguna Madre de Tamaulipas, immediately south of the delta of the Rio Grande in Mexico; Weller [1964]) and feeds almost exclusively on one species of seagrass while in residence (shoal grass, *Halodule wrightii*). Because of the degree of dependence of the redhead population on the laguna and reports of major disruptions to the laguna's seagrass community, the National Biological Service began a research program in coastal Texas.

The Texas Parks and Wildlife Department inventoried aquatic vegetation in Laguna Madre in the 1960's (Texas Parks and Wildlife Department 1965-67). In 1988 the National Wetlands Research Center (now the Souhern Science Center) resurveyed the laguna (Quammen and Onuf 1993).

Distributional Patterns

Seagrass meadows are undergoing profound change in Laguna Madre. The area of vegetated bottom in upper Laguna Madre has increased 130 km^2 (50 mi^2), from 120 km^2 (46 mi^2) to 250 km^2 (97 mi^2) between 1967 and 1988 (Quammen and Onuf 1993), an amount exceeding the total area of seagrass meadows in bays of the middle and upper Texas coast (Adair and Moore 1990; Adair et al. 1994). Concurrently, seagrass cover in lower Laguna Madre decreased by an even larger amount, 140 km^2 (54 mi^2), from 620 km^2 (239 mi^2) to 480 km^2 (185 mi^2), confined to deeper areas (Quammen and Onuf 1993).

Changes in the species composition of seagrass meadows affected even larger areas of the lower laguna (Fig. 2). Shoal grass covered 82% of the bay bottom in 1965 compared to 33% in 1988. Over the same period, cover of bay bottom by manatee grass (*Syringodium filiforme*) increased from 9% to 27% and by turtle grass (*Thalassia testudinum*) from 1% to 7%.

Factors Responsible

Processes responsible for the loss of seagrass from deep areas are different from those for the other changes. The loss of seagrass has resulted from reduced light reaching the bottom in deep areas near navigation channels because of increased turbidity caused by maintenance dredging. In 1988-89, waves generated by frequent episodes of high winds resuspended fine materials from dredge deposits and increased light attenuation for more than a year after a dredging project was completed (Onuf 1994). Since the interval between dredging projects is 2 years, the reduction in available light is essentially permanent.

Hydrological modifications of the laguna are most likely the primary cause of the expansion of seagrass cover in upper Laguna Madre and the shift in the composition of surviving seagrass meadows in lower Laguna Madre. Historically, a 20-km (12.4-mi) expanse of usually emergent flats separated the two sections of the laguna. Salinities greater than 60 ppt in the lower laguna and greater than 100 ppt in the southern part of the upper laguna were not unusual.

In 1949 the Gulf Intracoastal Waterway was completed, providing a continuous water connection between the two parts of the laguna, improving exchange with the Gulf of Mexico and moderating the salinity regime of the laguna. Since completion of the waterway, salinities have seldom reached 50 ppt in the lower laguna and 60 ppt in the upper laguna, even during extreme drought (Quammen and Onuf 1993).

Isolation from source populations of seagrass probably accounts for the slower colonization of the upper laguna than the lower laguna, after the environment became tolerable. The displacement of shoal grass by manatee grass and turtle grass after salinity moderation is consistent with the relative intolerance of those species to hypersalinity (high salinity) and their superior competitive capabilities under benign conditions. The current distributions of the three species are consistent with their relative colonizing abilities since salinity moderation: shoal grass is most widespread, manatee grass is intermediate, and turtle grass is most closely confined to its point of origin at the south end of the laguna (Quammen and Onuf 1993).

Management Implications

The dramatic decrease of shoal grass in the lower laguna is a particular concern to natural resource managers because redheads feed almost exclusively on shoal grass while in winter residence. Historically, there were several other important wintering areas for these ducks, such as Chesapeake Bay, Pamlico Sound, and Galveston Bay. The possibility existed that other areas could absorb additional birds if habitat quality in Laguna Madre deteriorated. Now, none of the alternative areas support significant winter populations of redheads, and few others do either, making the condition of Laguna Madre all the more critical for redheads.

Changes in the upper laguna since 1988 are almost certain to worsen the problem of redhead habitat deterioration. Whereas increases in the upper laguna compensated for about 40% of the losses of shoal grass in the lower laguna over the period of this analysis, a persistent phytoplankton bloom known as the brown tide has been resident in the upper laguna since 1990.

The bloom is so dense in some locations that it reduces light penetrating 1 m (3.3 ft) by more than 50% (Dunton 1994). This light reduction is leading to loss of shoal grass in the deep areas most influenced by the brown tide.

Displacement of shoal grass by manatee grass was not evident in the upper laguna in 1988 but is now. In all likelihood, the same processes responsible for the profound changes in the composition of seagrass meadows in the lower laguna will now take hold in the upper laguna. The greater isolation of the upper laguna from a source population of the invader probably accounts for the much later initiation of the replacement process than in the lower laguna.

A final factor further magnifies the importance to redheads of these changes in seagrasses of the Laguna Madre of Texas. The Laguna Madre de Tamaulipas, just south of the delta of the Rio Grande, is an integral part of the winter life-support system of redheads. In most years, more redheads overwinter in Texas than Mexico; however, in years of drought in Texas, more ducks continue south into Mexico. The large geographic extent of available habitat apparently buffers the population by increasing the probability that suitable conditions prevail somewhere in the system every year. The governor of the State of Tamaulipas, however, is now promoting the extension of the Gulf Intracoastal Waterway through the Laguna Madre in Mexico. In all likelihood, this development will reduce the support capacity of the laguna in Mexico for redheads, further increasing the reliance of the ducks on the laguna in Texas.

Modification of dredging practices in Texas and planning of waterway construction in Mexico hold the most promise for sustaining seagrasses and habitat for redheads to the maximum extent possible. At present, most dredge disposal is to submerged receiving areas along the channel, where bay resources are directly affected and wave-caused resuspension sometimes impairs water clarity for long periods after dredging. Land-based or deep-sea disposal would alleviate these problems. In Mexico, conducting an inventory of key resources, prominently including seagrasses and redheads, routing the waterway to avoid concentration areas, and implementing environmentally sound construction and disposal practices will ensure the greatest security for the wintering habitat of redheads and other resources linked to seagrass meadows.

References

Adair, S.E., and J.L. Moore. 1990. A survey of seagrass distribution of the middle Texas coast. Chapter 11 *in* S.E. Adair, J.L. Moore, W.H. Kiel, Jr., and M.W. Weller, eds. Winter ecology of redhead ducks in the gulf coast region. Final Rep., Cooperative Agreement 14-16-0009-87-909 between Texas A & M University and the U.S. Fish and Wildlife Service, National Wetlands Research Center.

Adair, S.E., J.L. Moore, and C.P. Onuf. 1994. Distributional ecology of submerged aquatic vegetation in estuaries of the upper Texas coast. Wetlands 14(2):110-121.

Dunton, K.H. 1994. Seasonal growth and biomass of the subtropical seagrass *Halodule wrightii* in relation to continuous measurements of underwater irradiance. Marine Biology 120:479-489.

Onuf, C.P. 1994. Seagrasses, dredging and light in Laguna Madre, Texas, USA. Estuarine, Coastal and Shelf Science 39:75-91.

Quammen, M.L., and C.P. Onuf. 1993. Laguna Madre: seagrass changes continue decades after salinity reduction. Estuaries 16:302-310.

Texas Parks and Wildlife Department. 1965-67. Coastal waterfowl project. Federal Aid Projects W-29-R-18 to 24. Austin, TX.

Weller, M.W. 1964. Distribution and migration of the redhead. Journal of Wildlife Management 28:64-103.

For further information:

Christopher P. Onuf
National Biological Service
Southern Science Center
Campus Box 339
6300 Ocean Dr.
Corpus Christi, TX 78412

Coastal Barrier Erosion: Loss of Valuable Coastal Ecosystems

by

S. Jeffress Williams
U.S. Geological Survey

James B. Johnston
National Biological Service

The conterminous United States has nearly 142,000 km (88,182 mi) of tidal shoreline that exists in a delicate balance with the forces of nature (Culliton et al. 1990). Much of this shoreline and the coastal barriers in particular are experiencing greatly increased pressures as a result of rapid population growth and accompanying development. Although coastal areas are highly desirable for their abundant natural resources and habitability, they are also extremely dynamic environments in which conditions hazardous to humans (e.g., erosion, flooding, pollution) may be present. In many regions, these hazards, which threaten not only humans but also valuable marine resources and even entire ecosystems, are increasing at alarming rates as coastal development, recreation, and waste disposal increase, often in direct conflict with long-term natural coastal processes. This article defines coastal barriers, summarizes their changes, and discusses the U.S. Department of the Interior's (DOI) Coastal Barrier Resources System (CBRS).

Coastal Barriers Defined

Coastal barriers are geologically recent depositional sand bodies that are highly variable in shape, size, and their response to natural processes and human alterations. They may stretch many kilometers in length and contain high sand dunes—such as the Outer Banks of North Carolina—or they may be small and isolated islands, so low in relief that they are routinely overwashed by spring tides and minor storms. Their dynamic nature means coastal barriers are constantly shifting and being modified by winds and waves, but scientific field investigations over the past several decades are revealing some disturbing trends.

Long-term survey data by the U.S. Geological Survey and others, based on analyses of archive maps, reports, and aerial photographs, demonstrate that coastal erosion is affecting each of the 30 coastal states (Figure; Williams et al. 1991a). About 80% of U.S. coastal barriers are undergoing net long-term erosion at rates of less than 1 m (3.3 ft) to as much as 20 m (65.6 ft) per year. Natural processes such as storms, rise in relative sea level, and sediment starvation (a reduction in volume of sediment transported by rivers reaching the coast), which may also be a result of human interference, are responsible for most of this erosion; but human factors such as mineral extraction, emplacement of hard coastal-engineering structures, and dredging of sand from navigation channels are now recognized as having major effects on shoreline stability (Table 1).

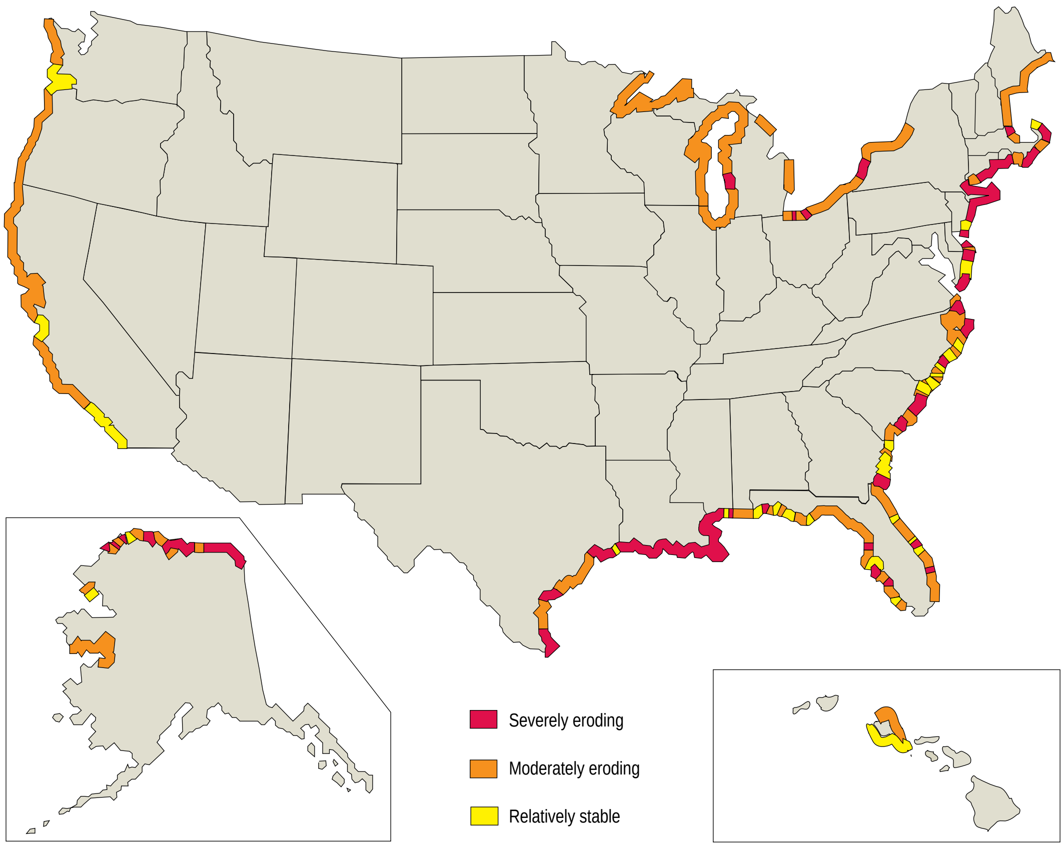


Figure. Classification of annual shoreline change around the United States (modified from U.S. Geological Survey 1985).

Table 1. Primary factors (geologic and human) affecting coastal areas ranked by decreasing relative importance (modified from Williams et al. 1991b).

Primary factors affecting coastal areas
Land subsidence (sediment compaction)
Storm impacts
Coastal processes (waves, winds, tides)
Eustatic sea-level change
Sand supply at the coast
Human activities: dredging, dams, mining, engineering structures, withdrawal of fluids (e.g., oil, gas, and water)
Regional tectonic movements

Undeveloped Coastal Barriers

Since 1982 the U.S. Fish and Wildlife Service (and now the National Biological Service) has been conducting inventories of the CBRS along the Atlantic and Gulf of Mexico coasts and the Great Lakes, as defined by the Coastal Barrier Resources Act of 1982 (Public Law 97-384) and the Coastal Barrier Improvement Act of 1990 (Public Law 101-591). (The Pacific undeveloped coastal barriers are under review by DOI as required by Section 6 of Public Law 101-591.)

The photographic inventories from aerial color infrared photographs (scales 1:12,000 to 1:65,000) provide a precise visual identification for each unit within the CBRS. Undeveloped coastal barriers are defined as areas that have less than one structure per 2 ha (4.9 acres) of fastland (areas suitable for building structures). Additionally, there are no areas in CBRS that are less than 0.4 km (0.25 mi) long. The entire barrier coastline was reviewed for inclusion into the CBRS system; inclusion into the CBRS means that the areas were ineligible for direct or indirect federal financial assistance that might support or encourage development.

The total shoreline length of the CBRS system for the United States is 2,055 km (1,276 mi), encompassing an area of about 537,000 ha (1.3 million acres; Table 2).

Table 2. Summary of undeveloped coastal barriers (F. McGilvery, U.S. Fish and Wildlife Service, Washington, DC, and National Oceanic and Atmospheric Administration, unpublished report).

State	Shoreline lengths (km)	No. CBRS units	CBRS shoreline lengths (km)	CBRS area (ha)
Maine	5,565	31	37.7	1,949
Massachusetts	2,430	79	197.0	27,301
Rhode Island	614	25	53.1	4,502
Connecticut	989	28	36.6	3,718
New York	2,960	90	167.4	26,216
New Jersey	2,867	16	16.7	3,279
Delaware	610	8	28.2	2,813
Maryland	5,104	48	45.1	2,901
Virginia	5,304	62	124.0	19,412
North Carolina	5,400	16	69.2	14,268
South Carolina	4,602	21	97.0	39,765
Georgia	3,750	11	31.6	26,085
Florida	18,811	105	304.8	115,484
Alabama	971	8	31.6	4,609
Mississippi	574	7	20.6	2,422
Louisiana	11,394	21	286.6	142,454
Texas	5,360	23	283.2	79,377
Puerto Rico	1,120	62	82.3	8,179
Virgin Islands	280	35	23.5	1,536
Ohio	320	10	13.0	1,941
Michigan	2,368	46	88.9	7,569
Wisconsin	960	7	12.2	793
Minnesota	368	1	4.8	381
Total	82,721	760	2,055.1	536,954

Comparing the CBRS maps after three surveys were conducted since 1982 shows that there have been no significant changes of CBRS unit boundaries in the United States (Frank McGilvery, USFWS, personal communication). Quite significant changes have occurred, however, in the size, shape, and character of many barriers because of natural processes.

Future

As the coastal population grows and barriers become urbanized, valuable habitats are being destroyed, and associated negative impacts such as waste disposal, pollution, and changes in freshwater and fine-grained sediment dispersal are altering entire coastal marine and maritime ecosystems. Protecting all remaining undeveloped coastal barriers should be a national priority. Some protection occurs through the Coastal Barrier Resources System, as well as other local, state, and federal programs, including acquisition, restoration, protection, and management programs.

References

Culliton, T.J., M.A. Warren, T.R. Goodspeed, D.G. Remer, C.M. Blackwell, and J.J. McDonough III. 1990. Fifty years of population change along the nation's coasts, 1960-2010. National Oceanic and Atmospheric Administration Coastal Trend Series, 2nd rep. 41 pp.

Williams, S.J., K. Dodd, and K.K. Gohn. 1991a. Coasts in crisis. U.S. Geological Survey Circular 1075. Reston, VA. 32 pp.

Williams, S.J., S. Penland, A.H. Sallenger, R.A. McBride, and J.L. Kindinger. 1991b. Geologic controls on the formation and evolution of Quaternary coastal deposits of the northern Gulf of Mexico. Pages 1082-1095 *in* N.C. Kraus, K.C. Gingrich, and D.L. Kriebel, eds. Coastal sediments '91. Vol. 1. American Society of Civil Engineers, New York.

U.S. Geological Survey. 1985. The national atlas, shoreline erosion and accretion map. U.S. Government Printing Office, Washington, DC.

For further information:

S. Jeffress Williams
U.S. Geological Survey
914 National Center
Reston, VA 22092

Reef Fishes of the Florida Keys

by

William F. Smith-Vaniz
National Biological Service

James A. Bohnsack
National Marine Fisheries Service

James D. Williams
National Biological Service

The Florida Keys are a chain of islands extending 320 km (199 mi) along the southern edge of the Florida Plateau from Biscayne Bay to the Dry Tortugas (101 km [63 mi] west of Key West). The Florida Reef Tract, a band of living coral reefs paralleling the Keys, extends from Fowey Rocks to the Marquesas and includes about 130 km (81 mi) of bank reefs and 6,000 patch reefs. For convenience, the Keys can be divided into the upper, middle, and lower Keys (Fig. 1).

The environmental and economic importance of the Florida Keys is indicated by the many protected or regulated areas, which include several national wildlife refuges, national parks, marine sanctuaries, and state-protected areas (Fig. 1). Because many recreational and commercial activities occur in nearshore habitats, these areas have high potential for environmental damage.

Relatively high rates of human population increase (28%-44%) are predicted over the next 20 years in some parts of the Keys; Monroe County, which includes all of the Keys, had a population growth of 160% during the past 40 years. Human activities associated with increased population growth may well ultimately disrupt the Florida Keys marine ecosystem and damage the area's overall economy. In recognition of this possibility, the Florida Keys National Marine Sanctuary was designated in 1990 under the Marine Protection, Research, and Sanctuaries Act, U.S. Public Law 101-605. The sanctuary includes 9,515 km^2 (3,673 mi^2) of coastal waters around the Florida Keys. The Sanctuaries and Reserves Division of the National Oceanic and Atmospheric Administration was charged with developing a comprehensive management plan and regulations to protect sanctuary resources (NOAA 1995). We focus on the current status of Florida Keys reef fishes and areas where research is needed immediately.

The Fish Fauna

The diversity and richness of fishes in the Florida Keys are unparalleled in shelf waters of the continental United States and reflect the mixing of dissimilar faunal components

Coral Reef Ecosystems

by
Stephen C. Jameson
National Oceanic and Atmospheric Administration

Coral reefs are one of the most diverse, complex, and beautiful ecosystems on earth. Coral reef ecosystems benefit humans commercially, recreationally, and environmentally (Laist et al. 1986). The abundant biological diversity of the coral reef ecosystem not only includes coral and the commercially important species associated with the reef but also thousands of other plant and animal species. Thus, the status and trends of this ecosystem are not easily evaluated.

Historically, most coral reef surveys have been limited to discrete reefs or species or have been time-limited (Rogers 1985; Dustin and Halas 1987; Bythell et al. 1992; Porter and Meier 1992; Ginsburg 1994). The status and trends of complete coral reef ecosystems around entire islands or reef tracts (e.g., the entire Florida reef tract) have never been comprehensively evaluated because of the complexity, length of time, and cost of such endeavors. Because of this lack of a comprehensive understanding of the status and trends of coral reef ecosystems under U.S. jurisdiction, this article looks at broad patterns in the status and trends of these ecosystems today with the hope of providing a useful focus for future ecosystem-based National Biological Service (NBS) coral reef endeavors.

Status and Trends

Coral reef ecosystems under U.S. jurisdiction are located in waters throughout the world (Figs. 1 and 2). These reefs can be divided into two broad categories, pristine and at risk. For references on specific areas, please contact the author.

Pristine Coral Reef Ecosystems

Pristine coral reef ecosystems are in remote locations with little or no human threats to ecosystem health. By definition, the status of these ecosystems is good and the trend in health is steady. Areas under U.S. jurisdiction with pristine coral reef ecosystems include the Flower Garden Banks in the Gulf of Mexico; the northwest Hawaiian Islands (uninhabited); Wake Island; the Northern Mariana Islands (excluding Saipan); Palmyra Island and Kingman Reef; Howland Island; Baker Island; and Jarvis Island in the Pacific Ocean (Figs. 1 and 2).

Coral Reef Ecosystems at Risk

Coral reef ecosystems at risk are near human population centers with some or all reefs experiencing local anthropogenic stress. Some important sources of stress include nutrient enrichment from sewage and agriculture, overfishing, and stress from high sedimentation caused by deforestation, agriculture, vessel traffic, and coastal runoff. The status and trends of many coral reef resources within these areas are poor (D'Elia et al. 1991; Ginsburg 1994). In addition, it is impossible to know the status and trends of these ecosystems on an island-wide or reef tract basis because of our lack of understanding of these ecosystems in any comprehensive way. Within U.S. jurisdiction, the coral reef ecosystems at risk include the Florida Reef tract, Puerto Rico, and the U.S. Virgin Islands in the western Atlantic and Caribbean; and the main Hawaiian Islands (inhabited), Johnston Atoll, Saipan (Northern Mariana Islands), and American Samoa in the Pacific Ocean (Figs. 1 and 2).

Future

The United States has abundant coral reef ecosystems. Pristine coral reef ecosystems are especially valuable as "natural" laboratories and control sites that can help us eventually understand the evolution and function of healthy coral reef ecosystems. We will not be able to clearly evaluate the status and trends of unhealthy ecosystems until we better understand pristine coral reef ecosystems. It is vital that adverse effects to these pristine areas are avoided.

Figs. 1 and 2 show that over half of all U.S. coral reef ecosystems are at risk, and some are nearly dead because of human perturbations. Swift legislative efforts and public works programs to reduce nutrients and

Fig. 1. Coral reef ecosystems under U.S. jurisdiction in the western Atlantic Ocean, Gulf of Mexico, and Caribbean Sea. Coral reef ecosystems are found on or around the Florida Reef tract, Flower Garden Banks, Puerto Rico, and U.S. Virgin Islands. Coral reef ecosystems at risk are indicated by an asterisk.

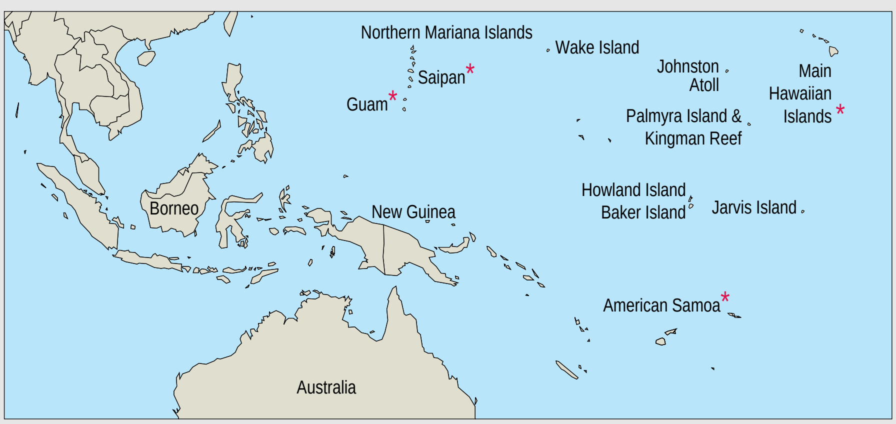


Fig. 2. Coral reef ecosystems under U.S. jurisdiction in the Pacific Ocean. Coral reef ecosystems are found around the northwest and main Hawaiian Islands, Wake Island, Johnston Atoll, Northern Mariana, Guam, Palmyra Island and Kingman Reef, Howland Island, Baker Island, Jarvis Island, and American Samoa. Coral reef ecosystems at risk are indicated by a red asterisk.

sediments may be the only way to save many of these national treasures.

References

Bythell, J.C., B. Gladfelter, and M. Bythell. 1992. Ecological studies of Buck Island Reef National Monument, St. Croix, U.S. Virgin Islands: a quantitative assessment of selected components of the coral reef ecosystem and establishment of long-term monitoring sites. Island Resource Foundation, St. Thomas. 72 pp.

D'Elia, C.F., R.W. Buddemeier, and S.V. Smith, eds. 1991. Workshop on Coral Bleaching, Coral Reef Ecosystems and Global Change: Report of Proceedings. Maryland Sea Grant College UM-SG-TS-91-03. 49 pp.

Dustin, P., and J.C. Halas. 1987. Changes in the reef-coral community of Carysfort Reef, Key Largo, Florida: 1974 to 1982. Coral Reefs 6:91-106.

Ginsburg, R.N., compiler. 1994. Proceedings of the Colloquium on Global Aspects of Coral Reefs: Health, Hazards and History, 1993. Rosenstiel School of Marine and Atmospheric Science, University of Miami. 420 pp.

Laist, D.W., T.E. Bigford, G.W. Robertson, and D.R. Gordon. 1986. Management of corals and coral ecosystems in the United States. Coastal Zone Management Journal 13(3/4):203-239.

Porter, J.W., and O.W. Meier. 1992. Quantification of loss and change in Floridian reef coral populations. American Zoologist 32:625-640.

Rogers, C.S. 1985. Degradation of Caribbean and western Atlantic coral reefs and decline of associated fisheries. Proceedings of the 7th International Coral Reef Symposium 6:491-496.

For further information:

Stephen C. Jameson
National Oceanic and Atmospheric Administration
Office of Ocean Resources Conservation and Assessment
1305 East-West Hwy.
Silver Spring, MD 20910

(Gilbert 1973) and the variety of habitats. More fish species have been reported from Alligator Reef in the upper Keys than at any single location in the Western Hemisphere (Starck 1968). These fishes consist primarily of continental, warm-temperate species characteristic of the northern Gulf of Mexico, and tropical Caribbean species, especially on the Atlantic side of the Florida Keys. Mixing of the warm-temperate and tropical Caribbean species occurs from north to south with distribution limits of individual species determined by seasonal temperature variations and the exchange of Gulf of Mexico and Atlantic Ocean waters in nearshore habitats in the middle to lower Keys. The key silverside (*Menidia conchorum*) is the only fish confined to the Florida Keys. It is not as rare as had previously been thought, and a recommendation has been made to change its official state listing from "threatened" to "special concern" (Gilbert 1992).

Two studies of single sites indicate the total diversity of Florida Keys fishes. Longley and Hildebrand (1941) listed 442 species from the Dry Tortugas, 300 of which are closely associated with coral reefs. Starck (1968) recorded 517 fish species from Alligator Reef, including 389 considered members of the reef community. The category "coral reef fish" is arbitrary, however, because a continuum exists from obligate (*see* glossary) species that spend their entire adult lives largely hidden within recesses of the reef, to opportunistic species that use many habitats. Also, most economically important reef fish are dependent on seagrasses and mangroves along the Keys and in Florida Bay for critically important nursery habitat. The availability of such habitats permits a higher density of organisms and a more complex reef community (Parrish 1989).

As more researchers, anglers, recreational scuba divers, and snorkelers have visited the Keys, an appreciation of the complex nature of reef fish communities has increased (Sale 1991). Research that uses visual census techniques has focused on the more common and readily observable reef fish. The most comprehensive census study to date (Bohnsack et al. 1987) provided a detailed quantitative description of the fish fauna of Looe Key National Marine Sanctuary for depths less than 13 m (43 ft). Quantitative studies of this kind serve as essential baseline references required for monitoring and detecting future changes in reef fish abundances and distributions. That study, with additional data from Key Largo, showed that fish faunas of the outer reefs in the Keys are diverse and complex, and their community structures are similar to well-developed reefs throughout the Caribbean.

Influences and Trends

As one of the most heavily fished areas in Florida, the Keys support extensive commercial and recreational fisheries for food, sport, and the marine aquarium trade (Bohnsack et al. 1994). A major management goal is ensuring continued sustainability of limited resources and traditional activities under rapidly increasing human population growth and exploitation of the reef fisheries. Excessive use and fishing may cause long-term harm to individual species, disrupt reef ecosystems, and damage the area's overall economy.

Demand and use of resources have increased (Fig. 2) with the growing number of residents and tourists (White 1991; Bohnsack et al. 1994). The number of registered boats has increased more than sixfold since 1965 while the number of commercial and partyboat vessels has remained stable (Bohnsack et al. 1994). Fishing success has increased, however, because of more accurate navigational aids, inexpensive electronic fish-finding equipment,

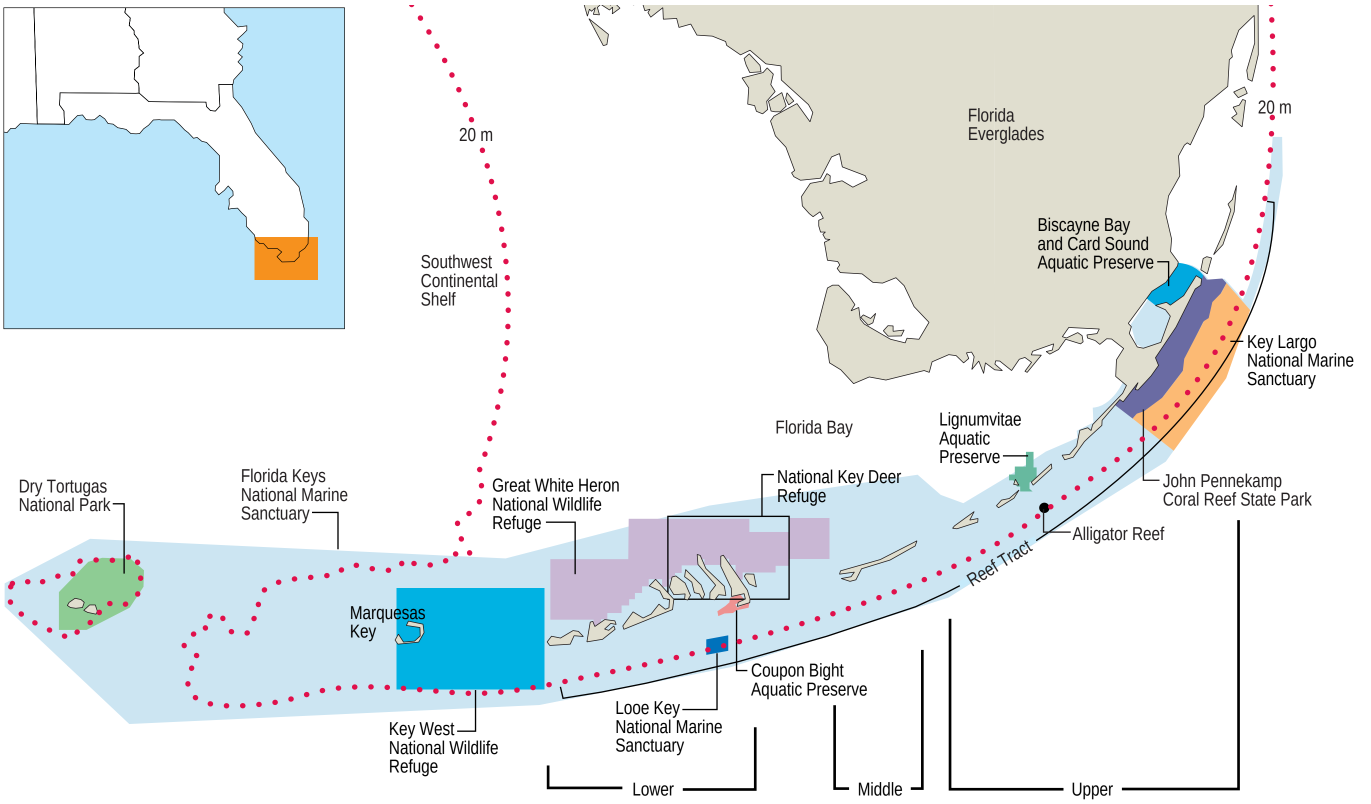


Fig. 1. The upper, middle, and lower Florida Keys. National marine sanctuaries, national parks, national wildlife refuges, and aquatic preserves are also shown (modified from maps provided by James A. Reed II, Florida Marine Research Institute). Various colors used simply to delineate designated areas.

and improved fishing gear and vessel technology. Although fishing can directly reduce stocks, other human activities also can damage resources and affect fish, including pollution, sedimentation, habitat loss from land-use practices, and vessel groundings. For example, habitat changes in Florida Bay have been attributed to water management and land-use practices in southern Florida (McIvor et al. 1994).

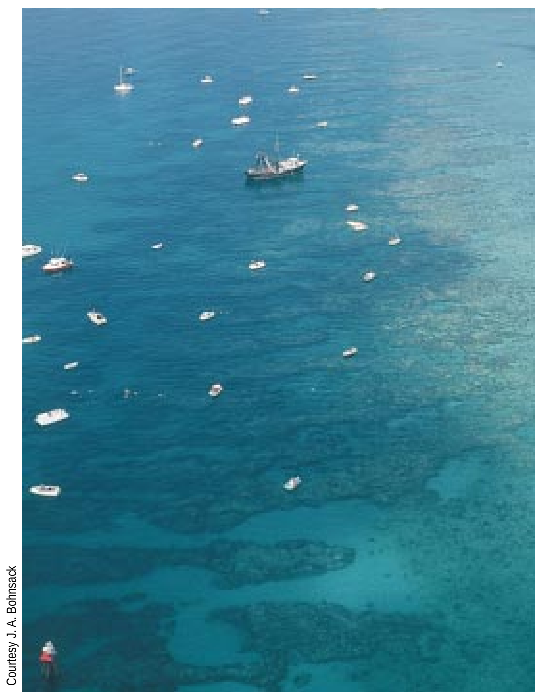


Fig. 2. Extensive use of resources in the Florida Keys: Looe Key (spur and groove zone).

Because of insufficient data, population trends and stock condition are impossible to determine for many species. Few fishery-independent data exist and fishery-dependent data have been limited to a relatively few years, to certain species, or to specific fishery components. Analyses are complicated because of the many species targeted, the large number of fisheries operating out of different ports, the number of different fishing methods used, and the many different fishing objectives, especially within the recreational fishery.

Some fishery trends are apparent despite data limitations. King mackerel (*Scomberomorus cavalla*) stocks collapsed in the early 1980's, but recovered somewhat after management measures were implemented. Pink shrimp (*Penaeus duorarum*) and grouper (Serranidae) landings have declined, and fisheries for queen conch (*Strombus gigas*), Nassau grouper (*Epinephelus striatus*), and jewfish (*Epinephelus itajara*) were closed because of reduced stock size. Increased landings reported for greater amberjack (*Seriola dumerili*), stone crab (*Menippe mercenaria*), blue crab (*Callinectes sapidus*), and yellowtail snapper (*Ocyurus chrysurus*) mostly reflect increased or redirected fishing efforts. For example, amberjack became commercially targeted only in the mid-1980's after king mackerel and red snapper (*Lutjanus campechanus*) landings declined. Landings of some species such as mutton snapper (*L. analis*), gray snapper (*L. griseus*), and

Courtesy J.A.Bohnsack

Florida Keys habitat showing representative reef fishes.

West Indies spiny lobster (*Panulirus argus*) have generally remained stable, despite large increases in effort (Bohnsack et al. 1994).

There is no guarantee, however, that any of these trends will continue, especially if fishing efforts increase or habitats become further impaired. For example, annual pink shrimp landings from the Dry Tortugas fluctuated around 4.5 million kg (9.9 million lb) for about 40 years before plummeting to less than half that level in the mid-1980's. Some of this decline may be a result of environmental changes caused by reduced freshwater inflow to Florida Bay (McIvor et al. 1994). Sponge and seagrass die-offs in Florida Bay may eventually reduce lobster and other fishery landings because of lost juvenile habitat. Fishery landing data will not necessarily reveal the full impact of those removals on the ecosystem or its sustainability. This is particularly true in complex tropical ecosystems such as the Florida Keys (Knowlton 1992). The annual removal of millions of kilograms of shrimp and spiny lobster is expected to affect their fish predators, while the removal of large numbers of predators may affect abundances and interactions of their prey. Fishing is a particular concern because it tends to target top predators, which are often the keystone species important for maintaining community structure (Knowlton 1992).

The widespread ecosystem changes documented in the Florida Keys and elsewhere in the Caribbean are of special concern to the long-term status of coral reef fish communities (Richards and Bohnsack 1990; Hallock et al. 1992). These changes include unexplained sea urchin mass mortalities, major coral loss and coral bleaching, shifts from coral- to algal-dominated substrates, extensive algae blooms, and numerous fish kills. Porter and Meier (1992) reported a loss of coral diversity between 1984 and 1991 at six locations and a decrease in abundance at five locations in protected areas between Miami and Key West. Although Porter and Meier (1992) could not determine the specific causes responsible for the changes, they noted that continued equal rates of loss over long periods would not allow the historical coral reef community structure of the Florida Keys to be sustained.

Algal fouling that may be related to leaching of nutrient-enriched groundwater (NOAA 1995) has recently caused severe damage to Algae Reef off Key Largo, and may be spreading to nearby Horseshoe Reef. Whether caused by increased nutrient enrichment, human alteration of historically freshwater runoff from the Everglades, reduced natural flushing effects associated with hurricanes during the last 20 years, or a combination of factors, continued deterioration of Florida Bay water quality ultimately will seriously alter the fish community structure of the bay and affect the Florida Keys ecosystem as well.

Recommendations

Realistic goals and objectives must be established to protect and restore Florida Keys ecosystems and their fish resources to allow optimal sustainable economic use while preserving biodiversity. Research efforts should focus on obtaining a better understanding of ecosystem dynamics and the effects of human interactions in order to generate and test predictive management models. Marine sanctuaries should have scientific reference sites and be used to develop strategies to reduce user conflicts. To be effective, management efforts must be international and must include cooperation between all levels of government and users. Because it is possible to love a reef to death (Fishman 1991), increased public education, understanding, awareness, and appreciation of the complex nature of reef fish communities and the effects of human activities within the Florida Keys ecosystem are especially important. Although efforts are needed to restore habitats, primary emphasis should be to prevent further habitat degradation from human activities.

Objective measures of fish populations, habitat conditions, and ecosystem function should be developed and monitored. Standard measures are needed to compare ecological impacts of different fisheries (Bohnsack et al.

1994), including better fishery and habitat data and more precise stock assessments. There is also an urgent need to develop nondestructive methods of collecting fishery-independent data. Cryptic, obligate reef fish, which have received the least attention, are likely among the best indicator species of environmental degradation because they are more sensitive to environmental changes. A comprehensive inventory of the cryptic reef fauna of the Florida Keys is also needed for baseline data in conjunction with establishment of long-term monitoring stations throughout the Keys.

References

Bohnsack, J.A., D.E. Harper, D.B. McClellan, D.L. Sutherland, and M.W. White. 1987. Resource survey of fishes within Looe Key National Marine Sanctuary. National Oceanic and Atmospheric Administration Technical Memorandum NOS MEMD 5:1-108.

Bohnsack, J.A., D.E. Harper, and D.B. McClellan. 1994. Fisheries trends from Monroe County, Florida. Bull. of Marine Science 54(3):982-1018.

Fishman, D.J. 1991. Loving the reef to death. Sea Frontiers 37(2):14-21.

Gilbert, C.R. 1973. Characteristics of the western Atlantic reef-fish fauna. Quarterly Journal of the Florida Academy of Sciences [1972] 35(2-3):130-144.

Gilbert, C.R. 1992. Key silverside, *Menidia conchorum* Family Atherinidae. Pages 213-217 *in* C.R. Gilbert, ed. Rare and endangered biota of Florida. Vol. 2. Fishes. University Press of Florida, Gainesville. 247 pp.

Hallock, P., F.E. Müller-Karger, and J.C. Halas. 1992. Coral reef decline. National Geographic Res. and Exploration 9(3):358-378.

Knowlton, N. 1992. Thresholds and multiple stable states in coral reef community dynamics. American Zoologist 32: 674-682.

Longley, W.H., and S.F. Hildebrand. 1941. Systematic catalogue of the fishes of Tortugas, Florida, with observations on color, habits, and local distribution. Papers from Tortugas Laboratory No. 34 (Carnegie Institution of Washington Publ. 535). 331 pp.

McIvor, C.C., J.A. Ley, and R.D. Bjork. 1994. Changes in freshwater inflow from the Everglades to Florida Bay including effects on biota and biotic processes: a review. Pages 117-146 *in* S.M. Davis and J.C. Ogden, eds. Everglades: the ecosystem and its restoration. St. Lucie Press, Delray Beach, FL. 826 pp.

NOAA. 1995. Florida Keys National Marine Sanctuary draft management plan/environmental impact statement. National Oceanic and Atmospheric Administration, Silver Spring, MD. Vols 1-3.

Parrish, J.D. 1989. Fish communities of interacting shallow-water habitats in tropical oceanic regions. Marine Ecology Progress Series 58:143-160.

Porter, J.W., and O.W. Meier. 1992. Quantification of loss and change in Floridian reef coral populations. American Zoologist 32:625-640.

Richards, W.J., and J.A. Bohnsack. 1990. The Caribbean Sea: a large marine ecosystem in crisis. Pages 44-53 *in* K. Sherman, L.M. Alexander, and B.D. Gold, eds. Large marine ecosystems: patterns, processes and yields. American Association for the Advancement of Science, Washington, DC. 242 pp.

Sale, P.F., ed. 1991. The ecology of fishes on coral reefs. Academic Press, Inc., New York. 754 pp.

Starck, W.A., II. 1968. A list of fishes of Alligator Reef, Florida with comments on the nature of the Florida reef fish fauna. Undersea Biology 1(1):5-36.

White, B., ed. 1991. Monroe County statistical abstract, 1991. NCS Corporation, Key West, FL. 295 pp.

For further information:

William F. Smith-Vaniz
National Biological Service
Southeastern Biological Science Center
7920 NW 71st St.
Gainesville, FL 32653