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South Florida Coastal Water Quality Characterization

Prepared for:

Water Quality Management Branch
U.S. Environmental Protection Agency
Region IV
345 Courtland Street, NE
Atlanta, GA 30365

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SOUTH FLORIDA COASTAL WATER QUALITY CHARACTERIZATION

Prepared for:

U.S. Environmental Protection Agency
Region IV
Water Quality Management Branch
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SUMMARY

The southeast coast of Florida is home to over 4 million year-round residents, with thousands more people visiting each year. Consequently, the area has experienced rapid urban growth and a corresponding increase in pollution. Several factors—physical, chemical, and biological —influence water quality off the southeast coast of Florida. This study evaluates the relative contributions of effluents from six sewage treatment plants (publicly owned treatment works, or POTWs) and nonpoint sources of concern to the changes in water quality observed along this coast. To better understand the potential impacts of point and nonpoint sources of pollution on the resources of southeast Florida and to guide the application of screening models, a conceptual model was developed to provide a graphical representation and descriptive summary of existing knowledge concerning key ecosystem resources that may be impacted by changes in turbidity, nutrients, and toxics. A deposition calculation (DECAL) model was then used to map the transport of POTW effluents off southeast Florida.

Physical factors influencing water quality in the study area include the northward-flowing Florida Current, located at the 24-meter (m) isobath along the continental shelf and transporting, on the average, 32 million cubic meters per second (m^3/s) at a mean flow velocity of 1.8 to 2.6 m/s. Cyclonic spin-off eddies transport Florida Current water into the coastal areas, produce strong current reversals, and advect heat and salt into the waters between the current and the shore. Water in this area is also influenced by wind and tidal forces, resulting in lateral meanders and cyclonic spinoff eddies.

A variety of environmental impacts, resulting from both human activities and natural processes, have been identified in the study area. The biological resources within the area include the mangroves and seagrass beds of Biscayne Bay and the coral hard-bottom communities of the bay and offshore. These communities are influenced by the physical, chemical, and biological processes occurring in the area. Large drainage canals serve as collection points for stormwater runoff, some of which is not treated, and deliver significant volumes of fresh water to coastal waters. These canals are the primary conduits for nonpoint source pollution to coastal areas. Contaminants of concern include heavy metals and nutrients. The primary sources of point pollution affecting the offshore water quality are the six POTWs discharging offshore within the study area. Biological oxygen demand, total suspended solids, fecal coliform bacteria, and residual chlorine are the primary pollutants in the effluent.

Comparisons of the relative contributions of point (POTW) and nonpoint sources of pollution show that the POTW contribution is greater than the nonpoint contribution, in terms of pounds of pollutant per million gallons of discharge. However, it should be noted that several assumptions are associated with the nonpoint source pollutant loading, including the fact that most nonpoint source discharges occur within the Intracoastal Waterway and Biscayne Bay and pollutants may not reach the coral communities offshore.

Because of data gaps, especially in the areas of nonpoint source pollution loading and nutrient loadings from point sources, this study should be considered preliminary in nature. It provides direction for future studies of the impacts on water quality and the biota off the coast of southeast Florida.

1. INTRODUCTION

A variety of direct and indirect environmental impacts resulting from human activities, as well as from natural processes, have been identified in the South Florida coastal area. These impacts threaten the health of various South Florida ecosystems. Of particular concern are the coral reefs, which are already under natural stresses at the northern limit of their range (Jaap, 1984). Direct impacts to the reefs resulting from human activities include boating impacts (e.g., groundings, propeller damage, anchor damage), diver impacts, overfishing, and dredging. Indirect impacts include water quality degradation due to sewage outfalls, deep well injection, septic tanks, spills, litter, and stormwater runoff (Grigg and Dollar, 1990). These indirect sources may be more serious in terms of long-term effects because of the difficulties encountered in reducing or eliminating excessive nutrients or toxic contaminants.

Currently, six sewage treatment plants (publicly owned treatment works, or POTWs) discharge secondary-treated effluents toward the edge of the narrow continental shelf in southeast Florida, where water depths begin to increase rapidly from 27 to 30 m (Figure 1-1). Although maximum coral reef development occurs south and west of Cape Florida off the Florida Keys, tropical reef biota also occur on octocoral-dominated hard-bottom communities from Miami north to Palm Beach. The hard bottom in this transition zone is characterized by transecting ridges and troughs and a large diversity of plant and animal life (also known as "live bottom"). Reef areas are found at between 12 and 18 m and between 27 and 40 m. Beyond this depth the bottom is mostly sandy and gently sloping.

Black band disease has been observed in corals in the vicinity of these discharges. This disease, caused by the cyanobacterium *Phormidium corallyticum*, has been linked to nutrient enrichment, high sedimentation rates, elevated temperatures, direct toxicity, and physical damage (Peters, 1992). Excessive nutrients may also stress coral reefs by promoting the growth of fleshy algae, outcompeting corals and other sessile benthic organisms. Increased phytoplankton and related eutrophication problems (deposition of suspended solids) may reduce light penetration (Tetra Tech, 1983). Because the corals derive a portion of their nutrition from mutualistic symbiotic algae, known as zooxanthellae, living within their tissues, reduced light levels that decrease photosynthesis can also affect the general health of the coral.

This report evaluates the relative contributions of effluents from the six POTWs and nonpoint sources of concern to changes in water quality observed off southeast Florida from Miami to Fort Lauderdale. Existing effluent data on nutrients, toxics, suspended solids, and other conventional pollutants were obtained from the six POTWs to characterize pollutant mass loadings. The most recent physical dispersion and environmental impact data from the Southeast Florida Outfalls Experiment (SEFLOE) were obtained from the National Oceanic and Atmospheric Administration's Atlantic Oceanographic and Meteorological Laboratory. Data were also collected to characterize and estimate mass loadings from nonpoint sources along the coast in this region. Oceanographic data were compiled and evaluated to characterize the extent and potential causes and sources of environmental impacts observed in nearshore marine communities.

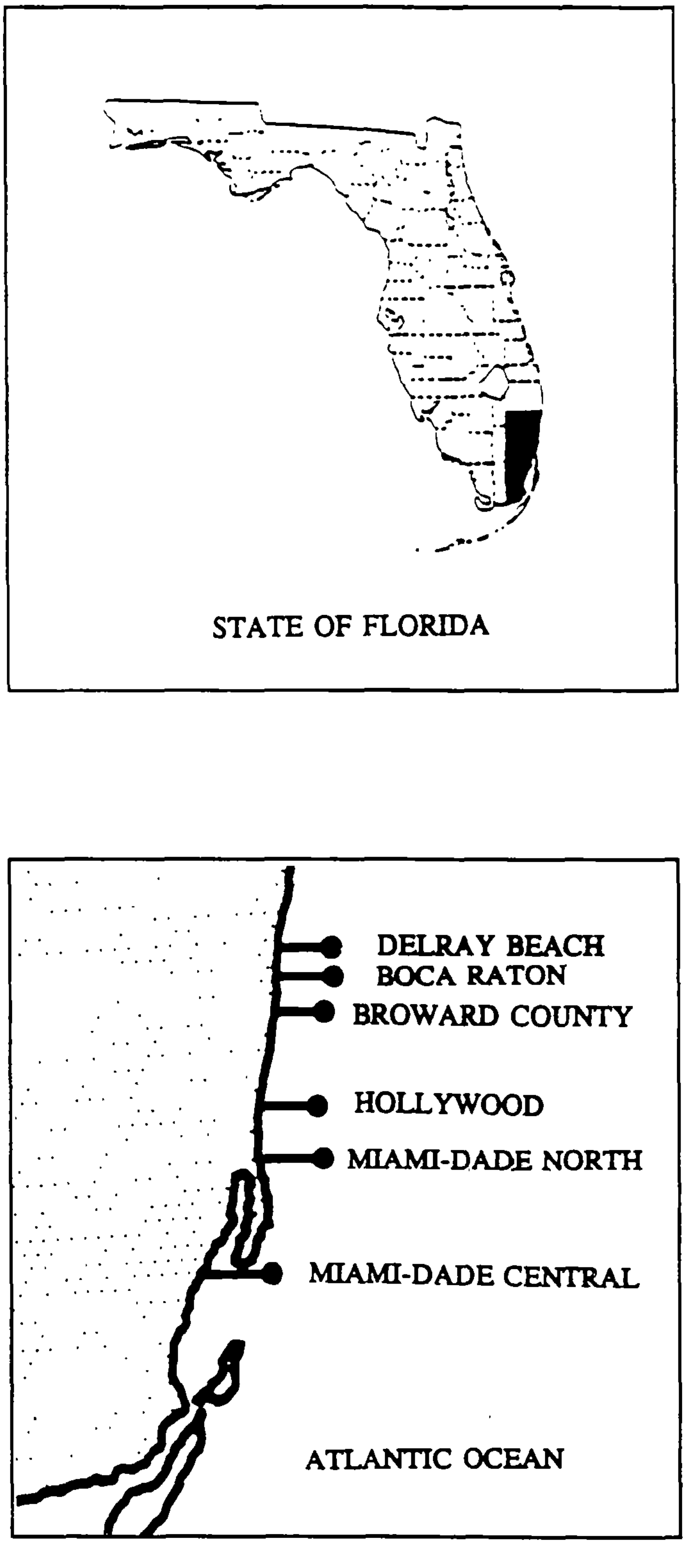


Figure 1-1. The southeast Florida study area and locations of the six POTWs.

A conceptual model was prepared to synthesize and integrate the information on ecosystem components, physical and chemical processes, and interactions between these processes and organisms in the nearshore communities. This conceptual model was used to guide the application of a screening-level model, DECAL (deposition calculation), to better quantify the relative contribution of pollutant loadings from the widely dispersed multiple point and nonpoint sources and to account for possible overlapping effects and oceanographic transport influences. The DECAL model was first used to estimate plume dispersion and the sediment deposition footprint around each POTW to better address possible impacts of organic sediments and toxics released from the point sources on the coral communities off southeast Florida. Most of the SEFLOE II data needed to conduct these model simulations were not available in time to be included in this report. As a result, only one DECAL run, based on information from the Broward outfall, is presented here. A DECAL model simulation was also planned to examine the nonpoint source contributions from coastal canals for comparison with the POTW contributions. However, important information necessary for the use of this model (suspended sediment concentration data) could not be located.

This report is divided into several chapters, which address the study area (Chapter 2), the circulation and hydrology of the Straits of Florida (Chapter 3), POTW contributions (Chapter 4), nonpoint source contributions (Chapter 5), the conceptual model (Chapter 6), results of the model simulation and analyses of available data (Chapter 7), and other related issues (Chapter 8). The appendices contain additional information on SEFLOE sampling stations, NPDES data for the POTWs, and a list of contact persons from whom information and data for this report were obtained.

2. CHARACTERIZATION AND DESCRIPTION OF THE STUDY AREA

This section describes the principal geographic and demographic features of the southeast coast of the Florida peninsula. Knowledge of these features is essential to an understanding of the water quality problems of this area.

2.1 Physical Location

The study area is bounded generally by Delray Beach, Florida, to the north (26°30′ north latitude) and Miami/Key Biscayne, Florida, to the south (25°40′ north latitude). The area covers approximately 85 km of the South Florida coast and extends to the approximate axis of the Florida Current (as noted in NOAA nautical chart 11460).

2.1.1 Major Urban Areas

The study area is a rapidly urbanizing area that includes the Fort Lauderdale and Miami metropolitan areas. This area sustains a population of approximately 4 million people. The primary industries in the area are service-related; tourism and trade play a key role in the economy. Within the study area are two seaports that are major cargo and cruise ship destinations: Port Everglades and the Port of Miami. The Fort Lauderdale/Hollywood and Miami International Airports reported 6.3 million air visitors in 1988 (Florida Department of Commerce, 1988). There is a concentration of seasonal visitors, especially in the coastal area. This tourism, along with the increase in the permanent population, has led to overbuilding along the shorelines, contributing to the degradation of nearshore water quality from urban runoff and other impacts.

2.1.2 General Topography

The land area is relatively flat, with elevations ranging from sea level to approximately 6 m above sea level. The highest point is in Broward County along the Pine Island Ridge. Another ridge, the Atlantic Coastal Ridge, is an ancient dune system that runs generally parallel to the shoreline through Palm Beach, Broward, and Dade Counties. This ridge was the area of initial development in South Florida because it provided a dry area on which to build.

2.1.3 Major Water Bodies

South Florida is historically part of the vast Everglades wetlands system. Most of the areas west of the Atlantic Coastal Ridge were once under water at least part of the year. Areas east of the coastal ridge were wetlands transitioning into a beach habitat. As the area grew, the need for flood control and potable water delivery to wellfields became apparent. During the first half of this century, the U.S. Army Corps of Engineers dug canals to drain the land for urban and agricultural development and to deliver water to coastal wellfields. All of these canals drain to tide, with a few also backpumping to the constructed Water Conservation Areas in western

Dade, Broward, and Palm Beach Counties. These canals, often referred to as the primary drainage system, are now maintained by the South Florida Water Management District and are the ultimate receiving waters for stormwater runoff not retained on site. Because of this role, they are a major contributor to near coastal and offshore nonpoint source pollution.

The Intracoastal Waterway (ICW) extends the entire length of the study area. In the Palm Beach County and Broward County portion of the study area, the ICW consists of a wide channel dredged on the landward side of the beach system. In Dade County it is part of Biscayne Bay. Because of the large volume of boat traffic in the ICW, it is a potential significant nonpoint source of pollution.

There are six inlets in the study area: Hillsboro Inlet, Port Everglades Inlet, Haulover Inlet, Norris Cut, Bear Cut, and Government Cut. These features play an important role in near shore circulation patterns and may be a conduit for nonpoint source pollution discharges to coastal waters; however, little information exists to confirm this possibility.

The study area includes the northern portion of Biscayne Bay, a shallow 71,225-hectare (ha) subtropical lagoon. The northern part of the Bay has experienced severe water quality problems over the years. The Miami River and Little River are major sources of poor-quality water in Biscayne Bay. The following paragraphs provide general information regarding the water quality of northern Biscayne Bay and its contributing tributaries.

- **Dumfoundling Bay** - Dumfoundling Bay receives runoff from surrounding urban and industrial areas. Water quality problems include stagnant areas associated with dead-end canals, sewage contamination, low dissolved oxygen, and contamination from these PCBs and other anthropogenic chemicals (SFWMD, 1988).

- **North Bay** - The primary water quality problem in this part of the bay is turbidity from scoured shorelines and dredged areas. High levels of phthalic acid esters (PAEs) are found in North Bay (SFWMD, 1988). This is an area of heavy marina activity, which includes the Port of Miami, a potential source of contamination from spills and leaching of antifouling paints. As mentioned previously, the Miami River and Little River discharge into northern Biscayne Bay. They have been identified as sources of sewage contamination and other pollutants from upland land uses. The Little River has a history of poor water and sediment quality with high levels of lead and other trace metals, nutrients, coliform bacteria, turbidity, and hydrocarbons that may indicate persistent sources (SFWMD, 1988).

  In spite of its water quality problems, northern Biscayne Bay also contains some unique environmental resources. Bird Key has one of the largest pelican rookeries along the southeast coast of Florida. The largest and healthiest seagrass bed in north Biscayne Bay is found between the Julia Tuttle and 79th Street Causeways. The preservation of these areas is critical to the health of this part of Biscayne Bay.

- **Miami River** - The Miami River has been used historically as a waterway for commercial and marine commerce. Water quality has been sampled in the river since 1984 and consistently violates coliform standards for Florida Class III water bodies. The water is turbid due to the effects of shipping, and it has low levels of dissolved oxygen (SFWMD, 1988). Acute pollution problems are the result of sewage pollution and stormwater discharges. Chronic problems include metals, tributyltin, and organic chemical contamination. Because of sediment accumulation in the river, many areas are often disturbed by ships traveling on the river, resuspending contaminated sediments. An additional source of contamination may be contaminated groundwater from Miami International Airport.

2.1.4 Habitats and Ecosystems

The following discussion is a general overview of the habitats and ecosystems in the study area. A more detailed discussion is included in the conceptual model. More information on the structure, function, and composition of these nearshore communities of the South Florida basin has been compiled by Continental Shelf Associates, Inc. (CSA, 1990).

Mangroves

Mangroves perform important environmental functions, including filtering upland runoff, protecting inland areas during storms, and providing habitat for marine organisms. Historically, South Florida was fringed with mangrove swamps. Within the study area these have been replaced by urban development. The Metro Dade County Planning Department (1986) has estimated that of the 18,615 ha of mangroves in Dade County in 1900, 4,249 ha exist today. Within the study area, there are only small pockets of mangroves, the largest of which are located in West Lake in Hollywood and in the Oleta River in North Miami Beach.

Seagrasses

Seagrass beds serve as nursery grounds and habitat to several hundred species of marine organisms. Their presence indicates good water quality with minimal turbidity. As indicated above, the most extensive seagrass bed in the study area is in north Biscayne Bay between the Julia Tuttle and 79th Street Causeways. North of Biscayne Bay, the coverage drops off rapidly. The shifting sand beaches that result from the high-energy coastal environment provide an unstable substrate. Throughout the study area seagrasses are usually found only in small pockets that are protected from wave energy (Zieman, 1982).

Coral and Other Benthic Communities

Within the study area, there is an extensive coral system, primarily soft corals and stony corals. These occur in ridges and patches and are not like the spur-and-groove formations that occur south of Miami and in the Florida Keys. In general, southeast Florida reefs are considered relict and not in an active growth stage, but they are veneered by a variety of living reef organisms

(Jaap, 1984; Dodge, 1987). From Palm Beach to Cape Florida, elements of tropical biota become increasingly dominant from north to south. *Acropora palmata* was once an important reef builder in this area, but ceased building reefs about 4,000 years ago. Three terraces have been recognized, with a back reef consisting of one terrace at 100 meters offshore in 4 to 5 m of water and a second terrace at approximately 800 m offshore at a 7- to 10-m depth. The third terrace reef platform is at a depth of 16 to 20 m. The forereef region is composed of a flat plain of rubble at a 30- to 50-m depth (Goldberg, 1973). The third terrace is biologically and geologically the most well-developed and occurs from 900 to 1200 m offshore. The appearance and location of the terraces vary along the coast. Today, only a few isolated reef communities appear north of Fort Lauderdale (Jaap, 1984). Coverage by hermatypic (reef-building) corals is low when compared to the Florida Keys and Caribbean region; however, the fauna form a valuable component of community structure and provide the principal means by which material is actively incorporated into the reef framework. The communities also provide relief, which in turn provides habitat for a variety of marine organisms (Dodge, 1987).

Coral communities are sensitive to increased turbidity and other perturbations. Because the study area is at the northern limits of their range, they are more susceptible to adverse impacts and stresses, both natural and human-induced. Natural stresses include temperature fluctuations and bacterial diseases. Examples of human-induced stresses include boat anchors, boat and ship groundings, physical contact from divers, and nutrient and pesticide loadings from upland land uses, as well as increased turbidity from offshore dredging, beach renourishment, and stormwater runoff.

2.2 Climate

The climate of South Florida is characterized as subtropical marine and is strongly influenced by the adjacent marine environment.

2.2.1 Annual Rainfall and Variation

Typically, South Florida receives 152 cm of rain annually. There are two distinct "seasons": the rainy season from June to November and the dry season from December to May. The rainy season is characterized by daily afternoon thunderstorms and an average monthly rainfall of 15.7 cm. This is the major period of freshwater flow to the coastal areas. During the dry season, there is minimal rainfall, with a monthly average of 5 cm (South Florida Regional Planning Council, 1991). The area is also within the zone subject to storms and hurricanes during the summer and fall months. These can have short-term impacts such as large flows of fresh water to the coastal environment and increased turbidity in coastal waters due to the movement of these large volumes of water and sediment resuspension by increased wave action. The construction of the primary drainage system has led to incidents of "slugs" of fresh water being released to the coastal environment instead of the historic sheet flow. These slugs can carry loads of pollutants from nonpoint sources and can cause a rapid change in the salinity of an area. This occurrence is discussed in greater detail in Chapter 5.

2.2.2 Temperature Range

The annual temperature range for South Florida is from approximately 70 °F in the winter to 90 °F in the summer. There are temperature variations, depending on meteorological occurrences.

2.3 Demographics

Land uses within the study area are primarily urban in nature. Since the early 1900s, with the extension of the Florida East Coast Railway to West Palm Beach and Miami, the area has experienced rapid growth. Favorable climate and plentiful natural resources have made the area very attractive not only to vacationers, but also to permanent residents.

2.3.1 Population and Trends

Table 2-1 illustrates the estimated increases in population in the study area since 1950 and the projected growth through the year 2000.

Urbanization started at the coast and along river banks and has moved west. Because a vast majority of the area was Everglades wetlands, initial development occurred on higher areas that were not prone to flooding. To minimize flooding and make land more suitable for agricultural and urban development, canals were dug to drain the wetlands to the west. Most of these canals drain to tide, carrying the pollutants associated with urban and agricultural development to the coastal areas.

2.3.2 Major Industries/Employment

Tourism is the major industry in South Florida. The warm climate and unique natural resources are a draw for both domestic and foreign tourists. Tourism, however, has led to the exploitation and degradation of natural resources.

The coastal waters off southeast Florida are one of the most densely used recreational bodies of water in the country and are adjacent to the only living coral reef in the continental United States. These two factors make environmental impacts to these coastal waters a major concern in terms of protecting human health and delicate biological communities. In an area where the beaches and the recreational fishing industry are the major draws for the tourist industry, contamination of the waters could have disastrous effects on local economies.

Table 2-1. Population Trends in the Three Counties in the Study Area

Year	Broward	Dade	Palm Beach	Total
1950	83,933	495,084	224,688	693,705
1960	333,946	935,047	228,106	2,497,099
1970	620,100	1,267,792	348,993	2,236,885
1980	1,018,257	1,625,509	576,663	3,220,629
1990	1,255,488	1,937,094	863,518	4,056,100
2000	1,467,554	2,183,841	1,104,136	4,755,531
% Increase 1950-2000	1,648	341	863	585

SOURCE: Florida Census Estimating Conference, Population and Demographic Forecast (Spring 1991) -compiled by the South Florida Regional Planning Council, 1992.

Most Significant Factors Affecting Water Quality

- Over 4 million people live in the Miami/Fort Lauderdale area, and thousands of others visit the coast each year. Tourism and the increase in the permanent population have led to rapid development and water quality impacts from urban runoff.

- Canals used to drain the land for urban and agricultural development earlier in this century receive stormwater runoff and contribute to the near coastal and offshore nonpoint source pollution.

- Other contributors of nonpoint source pollution include boat traffic in the Intracoastal Waterway and inlets. The Miami River and Little River are major sources of poor-quality water in Biscayne Bay.

- In the rainy season (June to November), rainfall averages 15.7 cm per month, with additional storms and hurricanes that cause increased turbidity and reduced salinity, as well as increased loads of nonpoint source pollutants.

3. CIRCULATION AND HYDROLOGY OF THE STRAITS OF FLORIDA

The North Equatorial Current, which represents the southern portion of the anticyclonic gyre of the North Atlantic Ocean, joins the Guiana Current (the northern section of the South Equatorial Current) to penetrate the Caribbean through the openings between the Lesser Antilles and the Greater Antilles (Tchernia, 1980). These openings consist of the Dominica Passage, the St. Lucie Channel, the Anegada and Virgin Islands Passages, and the Windward Passage. Shortly beyond the Yucatan Channel, the current turns sharply to the right, entering the Straits of Florida, to become the Florida Current (Figure 3-1). The Florida Current is joined by the Antilles Current at the northern end of the Florida Channel to form the western meridian segment of the North Atlantic anticyclonic gyre.

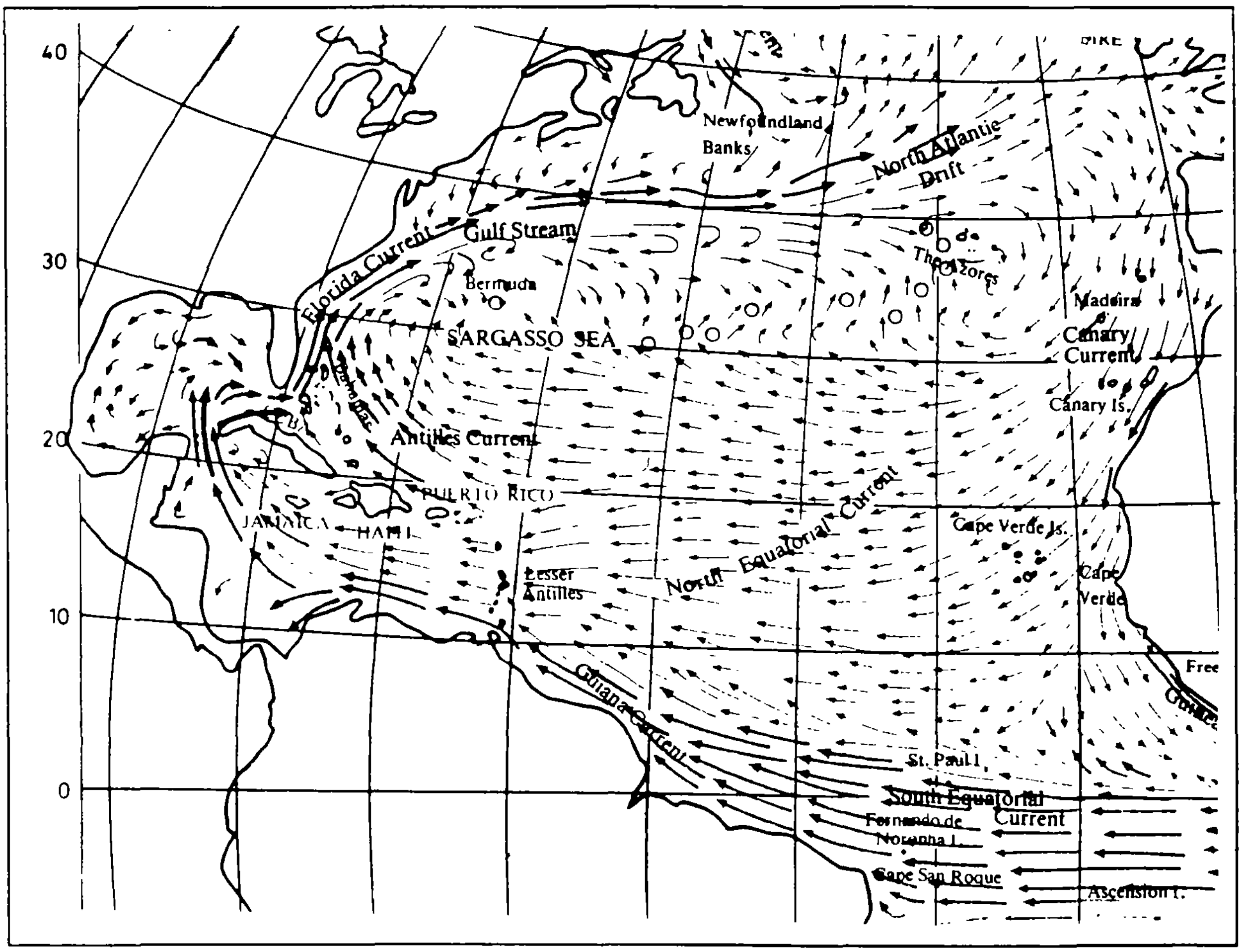


Figure 3-1. The prevailing currents affecting the southeast coast of Florida (adapted from Tchernia, 1980).

The combination of the east winds and narrowness of the Straits of Florida causes an accumulation of water in the northern Gulf of Mexico, resulting in a sea level on the west coast of Florida 19 cm higher than that on the east coast. The outflow caused by the geostrophic potential follows the Florida Channel, the only deep channel connecting the Gulf of Mexico to the Atlantic Ocean. This channel is 648 km long and runs east-west to its halfway point, where it turns south-north between Havana, Cuba, and the Dry Tortugas. The channel is 130 km wide at this point and narrows to 74 km, with a maximum depth of 800 m (Tchernia, 1980; King and O'Brien, 1971). The channel widens to 148 km at the northern exit, located between the east coast of Florida and Little Bahama Bank.

3.1 Velocity and Volume Transport of the Florida Current

The continental shelf off the southeast Florida coast is narrow, ranging from 1.9 km off Boca Raton to between 5.6 and 6.5 km off Miami (Lee and McGuire, 1972; Lee and Mayer, 1977). There is a 6-m ridge at the outer edge of the shelf, in 24 m of water (King and O'Brien, 1971; Figure 3-2), from which the bottom gently shoals at a ratio of 20:1 to the shore (Lee and

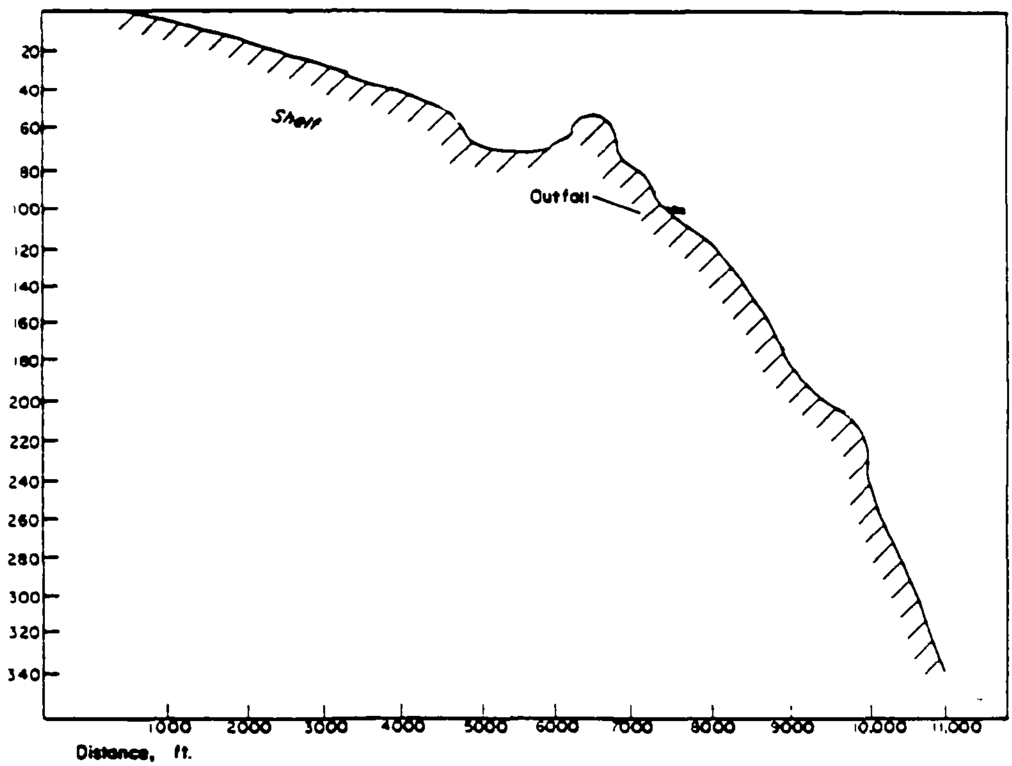


Figure 3-2. Cross-sectional view of the continental shelf off Pompano Beach, Florida (adapted from Lee and McGuire, 1972).

McGuire, 1972). The western edge of the Florida Current follows a sharp discontinuity near the ridge that separates the shelf from the depths. The strongest nearshore current between Delray and Miami is only 24 km offshore. Mean flow velocities for the Florida Current range from 1.8 to 2.6 m/s (King and O'Brien, 1971; Tchernia, 1980). Sharp horizontal gradients in salinity, current speed, and water color are characteristic of the western edge of the Florida Current.

Water volume transport of the Florida Current is known to fluctuate on tidal and annual time scales (Lee and Williams, 1988). Mayer et al. (1984) have reported tidal fluctuations on the order of ± 1.5 million cubic meters per second (m^3/s). Seasonal fluctuations of ± 4 million m^3/s are asymmetrically distributed about a mean northward volume transport of approximately 32 million m^3/s, with maximums in the summer and minimums in the fall (Schmitz and Richardson, 1968; Niiler and Richardson, 1973; Larsen and Sanford, 1985; Molinari et al., 1985; Leaman et al., 1987). Schmitz and Richardson (1968) estimated the net fluctuation bound for the steady-state volume transport to be ± 12 million m^3/s.

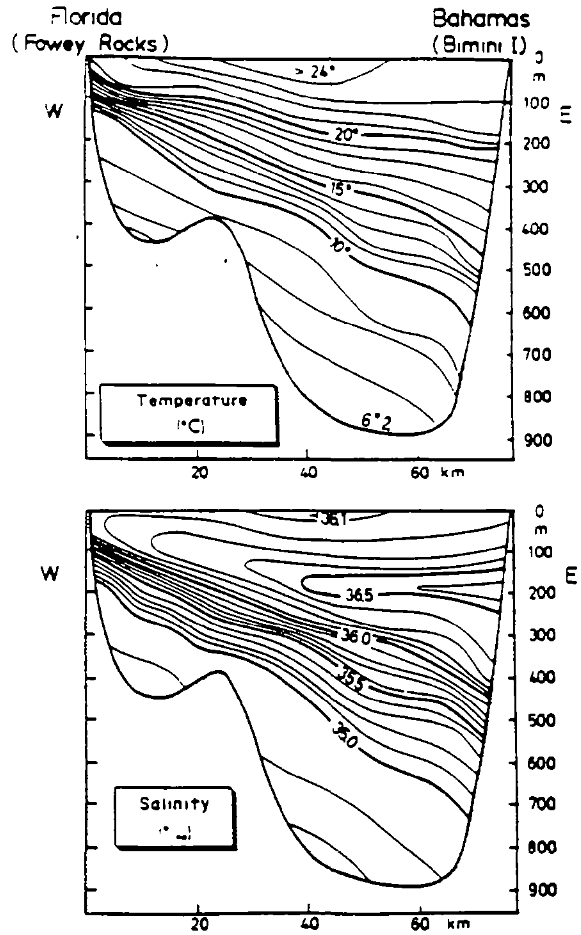


Figure 3-3. Cross-sectional view of the Florida Current illustrating salinity and temperature gradients (Tchernia, 1980).

Water volume transport can be significantly affected by local along-channel winds created by synoptic weather events (Lee and Williams, 1988). Northerly winds cause an increase in volume, and southerly winds cause a decrease. Northward winds create an easterly Ekman transport, which establishes a westward barotropic pressure gradient (the height of the water is greater on the westward side) that drives the northward geostrophic transport (Lee et al., 1985). The fluctuating wind stress can also produce surface onshore/offshore Ekman transport (downwelling/upwelling), which causes deep cross-shelf water movement over the steep bottom topography (Brooks, 1975). These movements advect warmer waters off the slope, acting as a mechanism for the redistribution of water masses and nutrients. The barotropic perturbations and cross-channel interior flows cause a steepening of the pycnocline, a distinct vertical density gradient (Figure 3-3). The combination of barotropic perturbations and steepened isopycnals is advected northward. This baroclinic instability (friction) converts mean potential energy to eddy kinetic energy and then back to the mean flow, thereby providing a mechanism for meander and eddy growth downstream (Lee and Mayer, 1977; Lee et al., 1985; Lee and Williams, 1988).

3.2 Florida Current Meanders and Spin-Off Eddies

Meanders and eddies are an integral part of the Florida Current and are known to occur all along the southeastern United States from the Straits of Florida off Miami to near Cape Hatteras (Von Arx et al., 1955; Webster, 1961; Duing, 1975; Lee, 1975; Duing et al., 1977; Lee and Mayer, 1977; Bane and Brooks, 1979; Brooks, 1979; Legeckis, 1979; Pietrafesa and Janowitz, 1980; Brooks and Bane, 1983; Lee and Atkinson, 1983; Lee and Waddell, 1983; McClain et al., 1984; Zantopp et al., 1987). Such changes in current patterns were described first by Pillsbury (1890) off Key West and Miami and were addressed later by Parr (1937).

3.2.1 Meanders

Meanders are low-frequency current fluctuations that appear as northward-traveling waves (Legeckis, 1979; Bane and Brooks, 1979; Lee and Mayer, 1977) where the westward displacement of the Florida current appears as the "crest" of an onshore meander. More specifically, a meander is a northward-moving barotropic wave superimposed on the mean baroclinic profile of the current (Duing, 1975), with periods ranging from 2 days to 2 weeks (Lee, 1975; Lee and Mayer, 1977; Lee et al., 1985).

Schmitz and Richardson (1968) reported horizontal meanders having a 1-week time scale and 5-km amplitudes. A 191-day record of transport collected east of Foley Rocks (just south of Miami) indicated periodicities ranging from 5 to 10 days (DeFerrari, 1970). Duing (1975) recorded current modulations on 4- to 6-day time scales, estimated wavelengths of 160 and 240 km, and a phase speed of approximately 50 cm/s. Webster (1961) located a 7-day meander of the Florida Current off Onslow Bay that had an amplitude of 10 km. Like the results documented by Lee and Mayer (1977) and Duing et al. (1977), this meander was correlated with onshore winds from north-south pressure variances that lagged by 4.5 days.

Fluctuations in physical parameters such as temperature and salinity may be used as indicators of current meanders. Temperature records from the waters off Miami show several-day oscillations with amplitudes of 2 to 3 C (Mooers and Brooks, 1974, 1977). Salinity transects conducted offshore from Pompano Beach indicate a subsurface core of high-salinity water that ranges from 36.2 to 36.6 parts per thousand (ppt.) (Lee and McGuire, 1972). This represents an intrusion of the Florida Current into coastal waters and an east-west meander of 1.9 to 3.7 km (Figure 3-4).

Meanders cause large fluctuations of current speed and direction of shelf waters, thereby creating instabilities in the lateral shear region of the Florida Current (Lee and McGuire, 1972). Upwelling occurs in the meander wave troughs between the offshore displaced front and the shelf break (Zantopp et al., 1987). The combination of these effects is thought to produce cold, cyclonic (anticlockwise) spin-off eddies that are entrained in the coastal waters (Figure 3-5) and move northward with the parent wave (Lee et al., 1981; Lee and Atkinson, 1983; Brooks and Bane, 1983; McClain et al., 1984). These cyclonic vortices are thought to occur on a continual basis, thereby creating a zone of divergence and upwelling along the eastward side of the current (Baranov, 1967).

3.2.2 Spin-Off Eddies

Lateral meanders moving northward can transfer momentum onto the shelf and produce current data that indicate the passage of an eddy (circular current) when measured from a fixed location. They cannot, however, transfer heat and salt (Stommel, 1965). There is no exchange of water mass across the front from a meander. However, cyclonic (counterclockwise) eddies transport Florida Current water into the coastal region and entrain coastal waters into the Florida Current. Strong southward flows and perturbations of high-salinity (≥ 36.4 ppt.) water are produced, as well as isotherms that either are horizontal or deepen toward the west (DeRycke and Rao, 1973). The characteristics of the Florida Current, most notably significant turbidity decreases and a deep blue coloration, accompany the southward-moving extrusion (Lee, 1975).

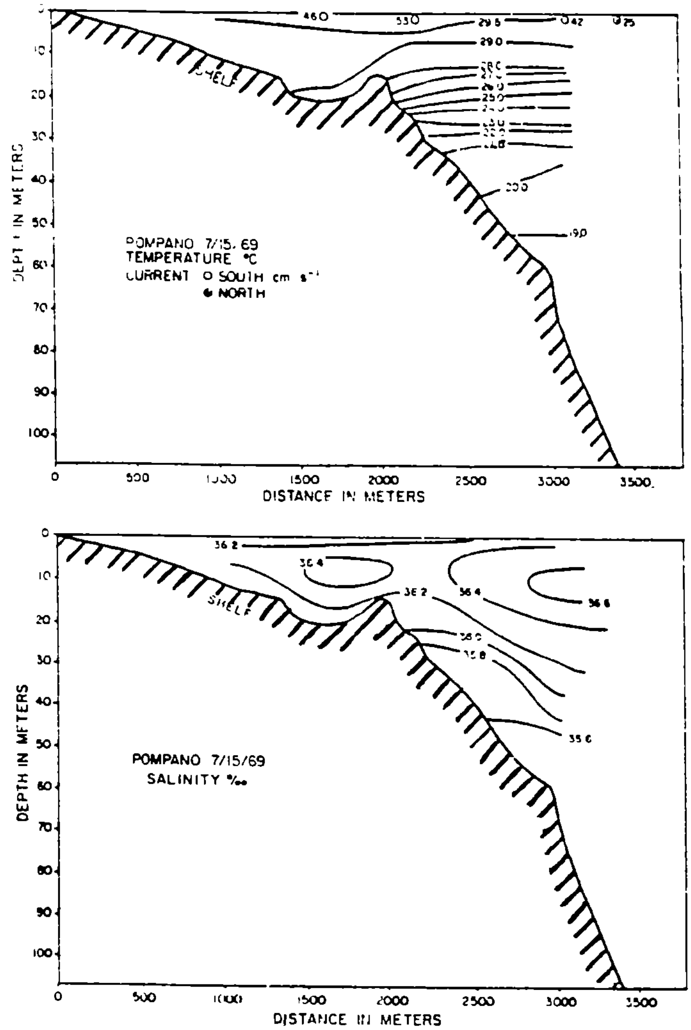


Figure 3-4. Cross-sectional view illustrating temperature and salinity gradients off Pompano Beach, Florida, during a strong southward coastal flow from a spin-off eddy (Lee, 1975).

Cyclonic eddies are typically detected from current data that indicate the occurrence of current reversals. Nine months of current data off Boca Raton indicated 163 reversals occurring in 270 days, or a reversal approximately four times a week (Lee and McGuire, 1972). The v component (north-south) series was to the south 31 percent of the time with a mean velocity of 20 cm/s and to the north 62 percent of the time with a mean velocity of 23 cm/s. The u component (east-west) series was to the east 47 percent and to the west 46 percent, and both directions had equivalent mean velocities of 8 cm/s. The variability of these data, which are summarized in Table 3-1, was extremely high, with standard deviations ranging from two to four times the mean. These reversals were concluded to be cyclonic eddies with currents moving southward at the rate of 10 to 77 cm/s, widths of 1.9 to 11.1 km, and downstream dimensions two to three times greater than the width. The duration of the current reversals varied from a few hours to 60 hours. Eddies can also be detected by strong current reversals accompanied by an advection of heat and salt into the "coastal strip," the thin band of coastal water between the Florida Current and the shore (Duing and Johnson, 1971).

The rate of northward movement of these eddies can be affected by seasonal events. Niiler and Richardson (1973) reported an average velocity in July (20 cm/s) that was three times the average velocity recorded in March and April (7 cm/s). The increased volume transport of the Florida Current during the summer is thought to be the cause of seasonal differences. Lee (1975) collected satellite imagery that suggests that the formation of edge eddies may be directly attributed to "Ford bands" (Ford et al., 1952) consisting of relatively fresh, cool water along the Gulf Stream (Florida Current) front. The intense lateral shear found at the boundary regions may create sufficient barotropic instability to produce short-lived eddies.

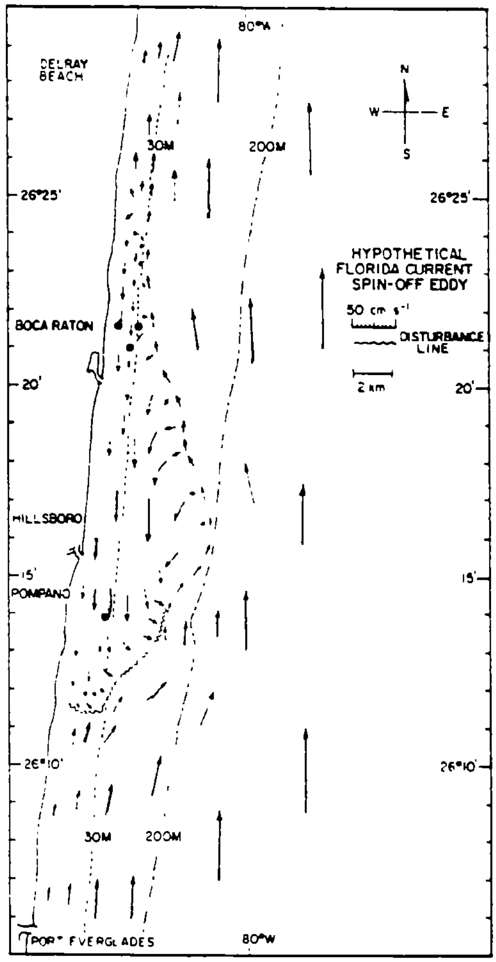


Figure 3-5. Qualitative model of a hypothetical spin-off eddy estimated from drogue trackings (Lee, 1975).

3.3 Tidal Influences

Typically, the movement of shelf waters is dominated by wind and tidal forces; however, the

Table 3-1. Summary of Pompano Beach Current Data (9/6/68 to 10/5/68)

Quadrant (Degrees)	Current Direction	Number of Observations	% of Total Observations	Mean Velocity for Quadrant (Knots)	Mean Standard Deviation for Quadrant
316-45	North	4939	59.1	0.44	0.216
46-135	East	494	5.9	0.34	0.192
136-225	South	2331	27.9	0.40	0.202
226-315	West	588	7.0	0.23	0.123

SOURCE: King and O'Brien, 1971.

coastal waters of the southeast coast of Florida are unique because of the extreme narrowness of the continental shelf (1 - 3 km) and the close proximity of the Florida Current (Lee and McGuire, 1972). The high current variability of the coastal waters off Boca Raton and Pompano Beach is governed primarily by small-diameter, cyclonic edge eddies (Lee, 1975), with semidiurnal and diurnal tidal currents contributing only 10 to 25 percent to the variability (Lee and Mooers, 1977; Kielman and Duing, 1974; Smith et al., 1969). Harmonic tide analysis of the 10 major tidal constituents showed the amplitude of the diurnal components to be equal in magnitude to that of the semidiurnal components, even though the tidal changes in sea level in the Straits of Florida are semidiurnal (Lee and McGuire, 1972). There is a longitudinal diurnal standing wave joining the Gulf of Mexico with the Atlantic Ocean, with a node located near Miami (Zetler, 1968). The current meters were located near the node, thereby recording diurnal currents without diurnal changes in sea level.

A tide prediction model was developed by recombining the amplitude and phase of the 10 major tidal constituents (Schureman, 1958). The difference between predicted and actual tidal currents was termed the residual. When 1-hour averages are graphed (Figure 3-6), the actual and residual curves are almost identical, indicating only a small tidal influence (Lee and McGuire, 1972). Average currents fluctuated around the mean by 40 to 70 percent, and only 5 percent of this fluctuation was attributed to the tides. It was concluded that tidal currents play an insignificant role in the circulation of coastal waters off southeast Florida.

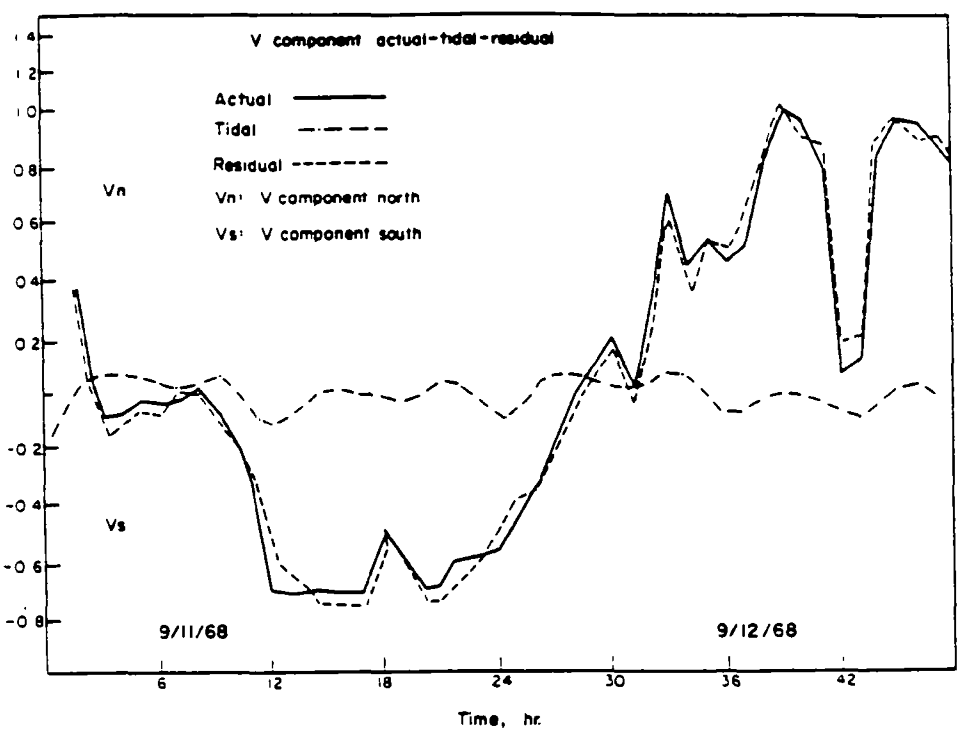


Figure 3-6. Current meter data off the coast from Boca Raton, Florida (Lee and McGuire, 1972).

The six ocean outfalls in this study discharge into a narrow strip of coastal water located at or just beyond the 24-m isobath. The circulation of the coastal waters off southeast Florida is dominated by lateral meanders of the Florida Current and cyclonic spin-off eddies. Because of these dominating forces, the effluents from the six POTW facilities are discharged into the coastal waters, the western edge of the Florida Current, or the western portion of a cyclonic eddy. The direction of these currents is primarily north-south; however, a sizable western component, with a mean velocity of 8.2 cm/s, does occur (Lee and McGuire, 1972). The residence time of coastal waters is on the order of 1 week (Lee and McGuire, 1972; Lee, 1975).

Most Significant Factors Affecting Water Quality

- The Florida Current flows northward along a ridge situated at the 24-m isobath on the edge of the continental shelf. The strongest nearshore current between Delray Beach and Miami is only 24 km offshore.

- The mean flow velocity of the Florida Current ranges from 1.8 to 2.6 m /s and has a mean volume transport of 32 million m^3/s. Maximum transport occurs in the summer and minimum values are seen in the fall.

- Low-frequency horizontal current fluctuations appear as northward-traveling waves with periodicities ranging from 5 to 10 days, wavelengths of 160 to 240 km, and amplitudes ranging from 5 to 10 km.

- These lateral meanders produce cyclonic (counterclockwise) spin-off eddies, which transport Florida Current water into the coastal region and entrain coastal waters into the Florida Current. Eddies produce strong current reversals and advect heat and salt into the "coastal strip," the thin band of coastal water between the Florida Current and the shore.

- Typically, shelf waters are dominated by wind and tidal forces; however, the continental shelf off the southeast coast of Florida is very narrow, thereby producing a unique situation. The circulation of the coastal waters off southeast Florida is dominated by lateral meanders and cyclonic spin-off eddies.

- The residence time of the coastal waters off southeast Florida is approximately 1 week.

4. POTW CONTRIBUTIONS TO THE COASTAL WATERS OF SOUTHEAST FLORIDA

Six municipal wastewater treatment facilities discharge primarily secondary-treated residential wastewater into the coastal waters of southeast Florida. All six outfalls are located at the 27-m isobath or deeper and discharge into the western edge of the Florida Current, the western edge of a northward-moving cyclonic eddy, or coastal waters (residence time is approximately 1 week). The coastal waters off southeast Florida are one of the most densely used recreational bodies of water in the country and contain octocoral-dominated hard-bottom communities that attract a wide array of tropical coral reef organisms despite being located at the northernmost limit of the range for reef biota. These two factors make these coastal waters a major concern in terms of protecting human health and delicate biological communities. In an area where the beaches and the recreational fishing industry are the major draws for the tourist industry, contamination of the waters could have disastrous effects on local economies.

The associated nutrients, pathogens, and toxicants from the point source discharge of sewage and septic tank seepage pose a serious threat to coral reef ecosystems, which depend on low-nutrient, high-clarity water (De Freese, 1991). Coral reefs are stenotropic ecosystems, having very narrow physiological tolerance ranges for physicochemical parameters (Johannes, 1975; Endean, 1976). Chronic, low-level water quality degradation may seriously impact sensitive life stages, resulting in long-term changes in community structure and stability. The structure and function of plant-herbivore relations, algae-coral competition, and reef fish communities are altered by pollution stress (Johannes and Betzer, 1975; Brock et al., 1979). Environmental perturbations can be detrimental to the growth and survival of corals (Johannes and Betzer, 1975; Endean, 1976; Pearson, 1981) and will eventually affect those organisms that depend on corals for food and shelter (Johannes, 1975). Also, tropical temperatures increase the solubility, biotic uptake, and toxicity of potential toxics discharged in the effluent, further compounding the environmental impact (Johannes and Betzer, 1975).

A cooperative study plan (Southeast Florida Outfalls Experiment, or SEFLOE), which pooled the resources of Federal, State, and local governmental agencies, was initiated in 1988. This joint venture was designed to characterize the effluent and determine the toxicity of discharges from the six publicly-owned treatment works (POTWs) located along the southeast Florida coastline. Methods never before used to determine plume dynamics were tested in conjunction with conventional methods. The new methodology was based on the detection of the acoustic backscatter of sound waves transmitted through the water column. This method had previously been used to discern areas of high particulate concentrations, but it had never been used to illustrate effluent plumes. The results of this study had some shortcomings, which fueled the effort to create a second study that is similar in scope but is expected to provide more comprehensive and more conclusive data. Both SEFLOE studies are discussed in greater detail in Section 4.1.

Section 4.2 describes the treatment process, the results of the bacteriological analyses, and toxicity testing from the 1988 SEFLOE, the National Pollution Discharge Elimination System (NPDES) permit requirements, and a summary of the monthly discharge data for the six POTWs of interest. This section provides a quantitative perspective of the point source discharges located within the study area.

Sewage pollution impacts on delicate biological communities have been placed into three broad, interacting categories: nutrient enrichment, sedimentation, and toxicity (Pastorok and Bilyard, 1985). Section 4.3 characterizes the effluent discharged by the six wastewater treatment facilities and discusses the potential impact those discharges may have on marine environments located along the southeast coast of Florida.

4.1 Southeast Florida Outfalls Experiments

4.1.1 SEFLOE I

In 1988, the Southeast Florida Outfalls Experiment (SEFLOE) was initiated by a cooperative agreement between the Miami-Dade Water and Sewer Authority Department, Broward County Utilities Division, South Central Regional Wastewater Treatment and Disposal Board of Palm Beach County, City of Hollywood Utilities Department, City of Boca Raton Public Utilities Department, Florida Department of Environmental Regulation (FDER), and National Oceanic and Atmospheric Administration. The objective was to characterize the physical dispersal conditions and the environmental impacts caused by the following six POTWs: Delray, Broward County, Boca Raton, Hollywood, Miami-Dade North District, and Miami-Dade Central District.
The SEFLOE project had four principal objectives:

- To characterize the oceanographic conditions associated with the southeast Florida coast that encompasses the six outfalls;
- To define and characterize the zone of initial dilution (ZID), as well as the farfield plume boundaries;
- To determine the toxicity of the whole effluent and of the diluted effluent found within the mixing zone, which would be done with and without chlorination; and
- To determine the natural capability of open ocean water to disinfect effluents treated to secondary treatment levels.

Two-day surveys were conducted between October 1987 and June 1988 for each of the six outfalls. On the first day of sampling, all treatment and chlorination activities proceeded as usual; on the second day of sampling, however, chlorination ceased and Rhodamine-WT dye was injected into the effluent prior to its entrance into the discharge pipe. The 2 days of sampling per facility were not necessarily conducted on consecutive days. Plume dispersion and mixing characteristics were examined using Rhodamine-WT dye tracer studies and acoustic backscatter techniques. Acoustic backscatter equipment had been previously employed for high-particulate wastewater projects, but had never been used to determine effluent dilution in ocean waters. Prior to its use in the SEFLOE project, the equipment was calibrated and the results validated against traditional methods of measuring plume dilution. The plume was acoustically scanned, and the results were used to create characteristic dilution curves that describe the spatial and temporal characteristics of plume dispersion.

Open ocean outfalls have two primary mixing zones. The zone of initial dilution (ZID) occurs where the freshwater effluent rises towards the surface, forming a surface plume or boil. From this point, the effluent flows horizontally with the prevailing current, forming the farfield mixing zone. The initial dilution is used to determine appropriate effluent concentrations for acute toxicity testing, and the farfield dilutions are used to determine chronic toxicity testing concentrations and compliance with Class III water quality criteria (e.g., the criterion for fecal coliform bacteria is 800 counts/100 mL with a geometric mean of 200 counts/100 mL at the edge of the mixing zone). A Class III surface water is classified according to the State of Florida water quality criteria as designated for use in recreation, propagation, and maintenance of a healthy, well-balanced population of fish and wildlife. The largest amount of mixing occurs in the ZID. The minimum dilution values (same as the initial dilution values) for the six outfalls ranged from 36:1 to 71:1 (seawater:effluent). Dilution curves were developed from data obtained from the farfield mixing zone.

All six farfield mixing zones conformed to the 4/3 rule (Brooks, 1960; USEPA, 1982), a method developed for predicting farfield dilution in open coastal waters. The method states that the lateral diffusion coefficient increases as the 4/3 power of the wastefield width. The equation is as follows (USEPA, 1982):

$$\epsilon = \epsilon_o \left(\frac{L}{b}\right)^{4/3}$$

where

ε = lateral diffusion coefficient, ft /sec;
ε_o = diffusion coefficient when L = b;
L = width of sewage field at any distance from the ZID, ft; and
b = initial width of sewage field, ft.

The initial diffusion coefficient is determined by

$$\epsilon_o = 0.0001\ b^{4/3}$$

The centerline dilution, D_s, can now be calculated by

$$D_s = \left[erf \left(\frac{1.5}{\left(1 + \frac{8\, \epsilon_o t}{b^2}\right)^3 - 1} \right)^{1/2} \right]^{-1}$$

where
t = travel time, hr, and
erf = the error function.

The use of this mathematical procedure for determining dilutions can be time-consuming. Table 4-17 in *Revised Section 301(h) Technical Support Document* (USEPA, 1982) lists the dilutions that correlate to various initial field widths (10 to 5000 ft) and travel times (0.5 to 96 hr). This table is not always applicable for outfalls located in shallow waters and plumes with long travel times. In those instances, EPA recommends that the subsequent dilution be determined using a constant lateral diffusion coefficient rather than the 4/3 law (USEPA, 1982).

A worst-case plume configuration was developed to correlate the plume area with the downcurrent distance from the ZID. This resulted in an isosceles triangle composed of the angles 20 degrees (vortex), 80 degrees, and 80 degrees. The vortex was located over the outfall, and the triangle was oriented in the direction of the current. The area of the plume (Figure 4-1) for any given range, *r*, can be calculated using the equation

$$A = r^2 \tan \Theta/2$$

where *r* is the range and Θ is the vortex angle. In this instance, Θ = 20 degrees. Florida Administrative Code (FAC) 17-4 allows a maximum mixing zone of 502,655 m^2 and requires that a minimum dilution of 500:1 be achieved within that mixing zone. Using the above triangle dimensions, this allows a plume to extend 1,700 m from the outfall. The dilution curves from the six facilities were used to determine whether the respective effluents achieved the minimum dilution of 500:1 within this distance.

The data from the SEFLOE I study consisted of three principal components: in-plant measurements, at-sea field measurements, and laboratory analyses.

In-Plant Measurements

Rhodamine-WT dye was injected into the effluent upstream of the outfall pipe at the rate required to maintain an effluent dye concentration of 1 part per million (ppm). On each day of the study, water samples were collected every 30 minutes at a location between the injection point and the outfall and analyzed for dye concentration, fecal coliform, fecal streptococcus, total coliform, chlorides, and suspended solids.

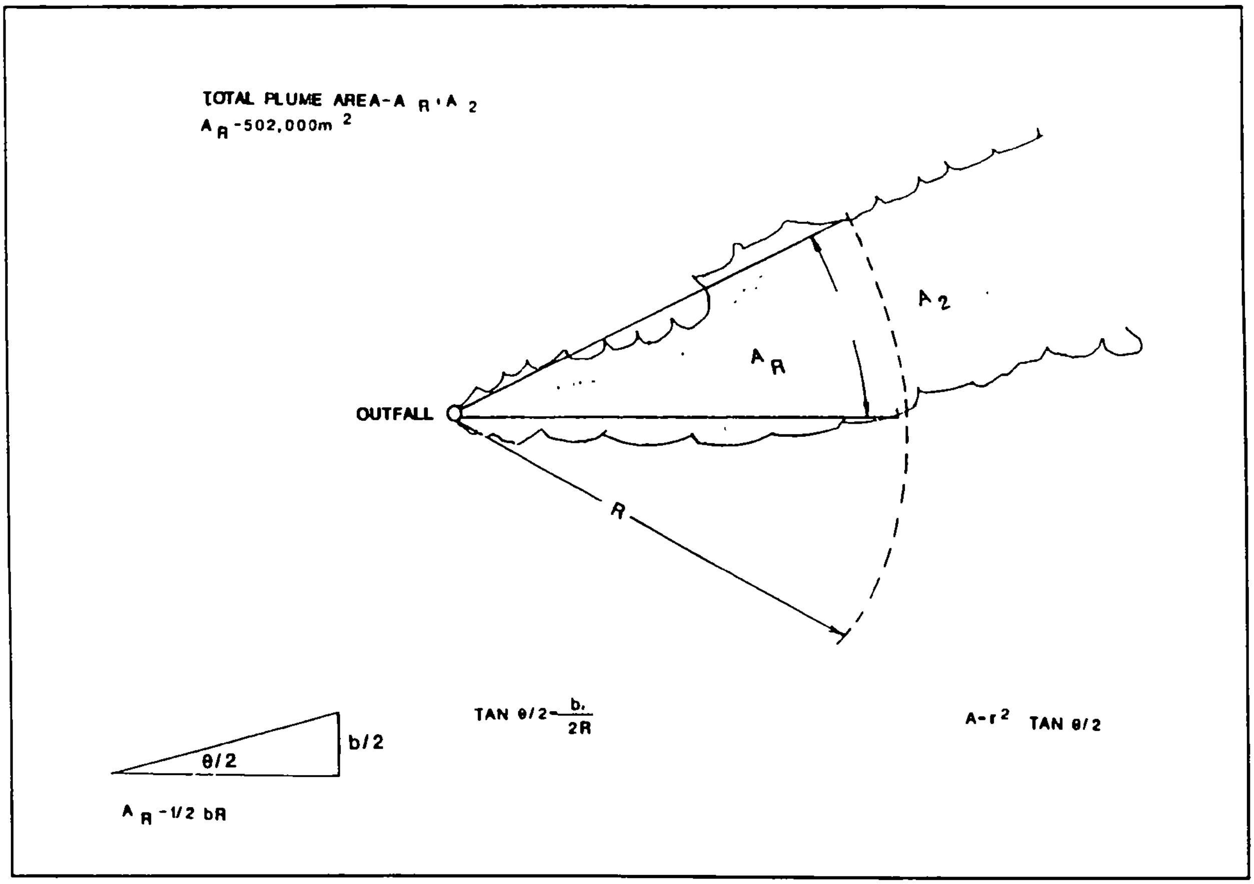


Figure 4-1. Typical surficial plume configuration and triangle sector approximation (Proni and Damman, 1989).

At-Sea Field Measurements

The at-sea field operations collected acoustic backscatter data along transects while the research vessel was under way, and water samples were collected while the vessel was on-station. Vessel location was maintained using a Loran C with an accuracy of ±15 m. Mean surface currents were measured using a drifting spar buoy with a drogue placed in the water for a period of time consistent with the collection of the acoustic backscatter data. (See Table 4-1.) The distance between the drop-off point and the retrieval point was calculated by the Loran C and divided by the drift time. An electromagnetic current meter was moored to the outfall at a mid-water position and programmed to record the vector at 30-second intervals. Temperature and conductivity profiles (as a function of depth) were also recorded at the outfall. The T-C profiles collected during the fall and winter months did not indicate vertical stratification; however, the summer T-C profiles from the Hollywood and Miami-Dade North District outfalls showed some minor stratification. Vertical stratification tends to sheer the plume and aid the mixing process.

Table 4-1. Surface Current Velocities and Direction for the six POTWs in the SEFLOE I Study

Facility	Sampling Dates	Direction (degrees)	Velocity (knots)
Delray Beach	01/13/88	165	0.25
Boca Raton	10/07/87	110	0.19
	01/12/88	130	0.16
Broward County	10/08/87	30	0.66
	01/11/88	135	0.22
Hollywood	06/06/89	358	0.87
	06/07/89	348	0.24
Miami-Dade North	10/18/87	92	0.32
	10/20/87	80	0.34
	06/10/88	0	1.0
Miami-Dade Central	10/27/87	19	0.24
	01/24/8	335	1.1

The seawater samples were gathered at 1-m, 7-m, and 15-m depth intervals using a 10-L Niskin bottle. Subsamples were used for biotoxicity testing, water quality, and dye concentration analyses. Microbiology samples were taken at 1-m intervals using sterile bottles.

Acoustic backscatter (high-frequency echo sounding) was the primary tool used to determine the spatial variability and distribution of the effluent plume. The transducer was towed across the plume, sending a sound pulse vertically downward every 0.24 second. The pulse, which traveled at 1,500 meters per second, was scattered back to the surface by discontinuities in the water column. (The discontinuities, which consist of differences in particulate concentrations, alter the intensity of the scatter, thereby providing a measure of the concentration.) By towing the transducer across the plume, a profile of the particulate concentration, as a function of depth and position, was created. This image represented a cross-sectional view of the particulate scattering field along the vessel's heading.

Laboratory and Sample Analysis

In-plant and at-sea samples were subdivided for the analyses. The analyses consisted of tests for dye concentration; bacteriological analyses; and suspended solids, chlorides, and toxicity testing. The bacteriological samples were further divided into fecal coliform, fecal streptococcus, and total coliform.

organism (USEPA, 1991). Acute toxicity tests determine the LC_{50} for the effluent. The LC_{50} is defined as the concentration of effluent required to induce 50 percent mortality of the test organisms within the designated time period. Chronic toxicity tests measure the no-observed-effect concentration (NOEC), which indicates the effects of an effluent on larval growth, reproduction (fertilization and fecundity), and embryo/larval survival (USEPA, 1991). The NOEC is defined as the highest tested concentration of an effluent at which no adverse effects are observed on the test organisms at a specific time of observation (USEPA, 1991).

Screening toxicity tests were performed on water samples collected from the treatment plant prior to discharge, from within the ZID, and from the farfield mixing zone. Definitive toxicity tests were conducted using plant effluent at test concentrations of 1, 10, 30, 60, and 100 percent. LC_{50} values for each of the facilities were determined from the definitive tests. All tests used the acute 96-hr methodology using the sheepshead minnow, *Cyprinodon variegatus*, and the mysid shrimp, *Mysidopsis bahia*, as test organisms.

Data Analysis

The primary product of the SEFLOE project was a characteristic dilution curve for each of the six outfalls (Figure 4-2). The characteristic dilution curve can be determined using one or a combination of the existing subfields, F_i, of the wastewater plume field, *FF*. This curve can be used to determine effluent dilution and compliance with Class III water quality criteria.

Initial dilution is the process undergone by the effluent as it travels through the discharge pipe and the water column to the zone where it makes its nearest approach to the surface. This zone, which may be near or at the ocean surface, is commonly referred to as the boil. For all six outfalls examined in the SEFLOE project, the effluent plume was confined to the upper 8 m of the water column.

As previously mentioned, the wastewater plume field, $FF\ (r, \Theta, z, t)$, is composed of several subfields, $F_i\ (r, \Theta, z, t)$, where $FF = F_1, F_2 \ldots F_b \ldots F_N$. These subfields, which may include the fecal coliform field, fecal streptococcus field, temperature field, kinetic energy field, acoustic backscatter strength field, total coliform field, and dye tracer field, can be used to calculate the initial dilution. The initial dilution for any particular field is expressed as the ratio of the concentration measured in the boil to the effluent insertion concentration. The equation is as follows:

$$\textit{Initial Dilution} = \frac{F_i\ (r, \theta, z, t)\ \ \textit{Boil}}{F_i\ (t-\Delta t)\ \ \textit{Effluent}}$$

where Δt is the time required for a unit of effluent to go from the in-plant measurement site to the boil measurement site. This is commonly referred to as the detention time.

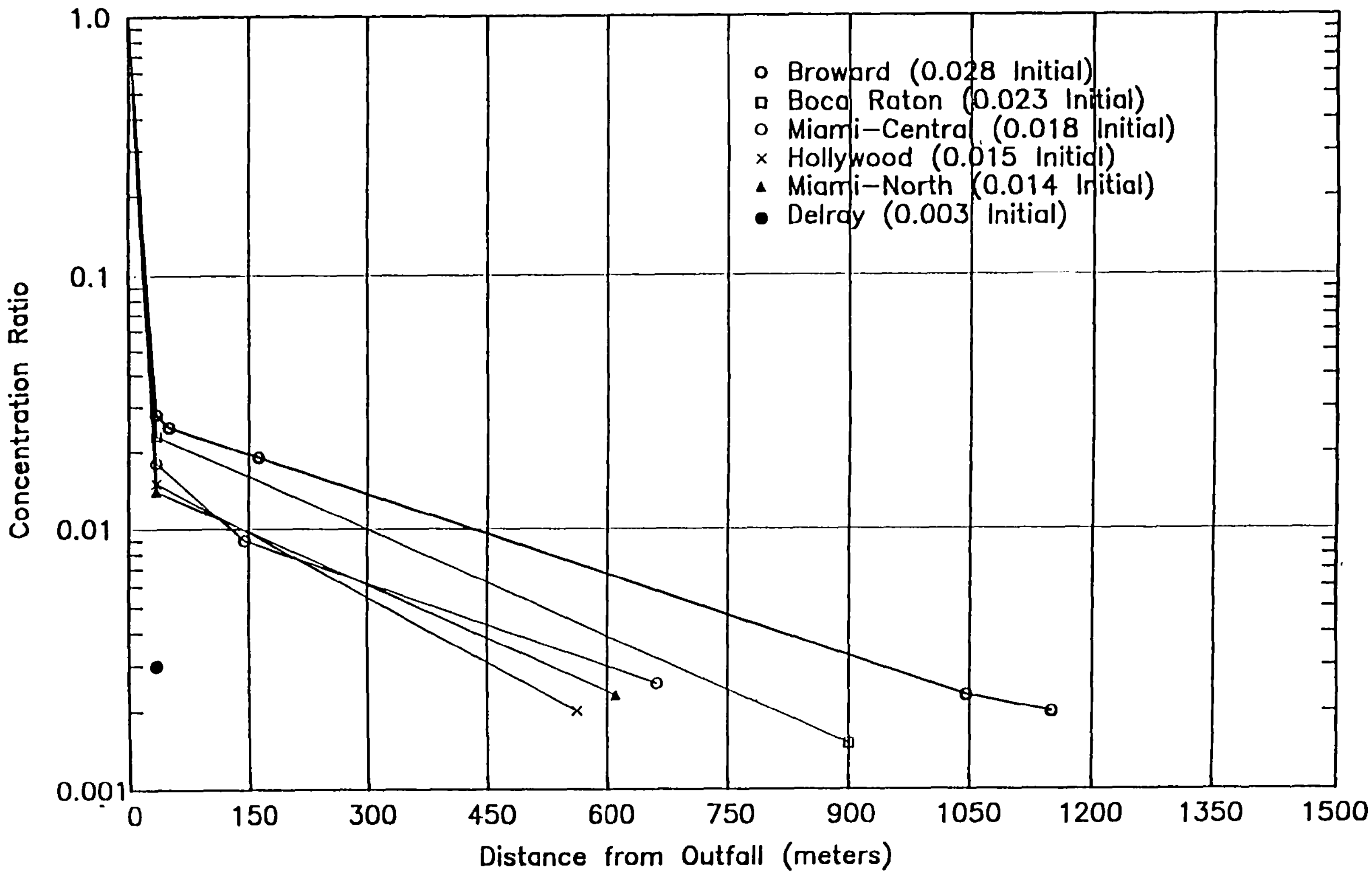


Figure 4-2. Characteristic dilution curve for South Florida outfalls.

Once the effluent reaches the farfield zone, the dilution dynamics are determined by oceanographic processes rather than by effluent plume characteristics. As mentioned above, a single subfield or a combination of subfields may be used to derive the characteristic dilution curve (CDC). The concentration ratio, $\delta(r)$, of the CDC at any given range, r, is calculated by:

$$\delta(r) = \frac{(F_D)_b}{(F_D)_e} \times \frac{(F_D(r))}{(F_D)_b}$$

for the dye (D) subfield where b = boil and e = effluent. The validity of the acoustic data can be tested by calculating the $\delta(r)$ for the acoustic scatter subfield and comparing it with the $\delta(r)$ for the dye subfield. Once developed, the characteristic dilution curve can be used to calculate the concentrations of parameters, such as fecal coliform, at the edge of the mixing zone. Of course, in the case of fecal coliform, this value would be the result of physical dilution only and would not indicate any deviation due to die-off. The characteristic dilution values for the six POTWs used in the SEFLOE I study are listed in Table 4-2.

The characteristic dilution curve serves two useful purposes: (1) it determines the dilution concentrations applicable to chronic toxicity testing, and (2) it determines compliance with State water quality standards. These values are also useful for determining the exposure of an organism to an oceanic wastewater plume.

Unfortunately, the data collected from the monitoring program were insufficient to conclude that "no unreasonable degradation" was occurring in the marine environment off southeast Florida. The sampling period of only 2 days did not provide enough data points, and the total suspended solids and the acoustic backscatter data did not correlate. The plume dynamics, particularly

Table 4-2. Characteristic Dilution Values with Range for the Six POTWs in the SEFLOE I Study

Range (meters)	Delray Beach	Boca Raton	Broward County	Hollywood	Miami-Dade North	Miami-Dade Central
0	333:1*	43:1	36:1	67:1	71:1	56:1
100	NA	53:1	45:1	87:1	87:1	85:1
200	NA	73:1	56:1	116:1	119:1	128:1
300	NA	100:1	71:1	185:1	165:1	164:1
400	NA	141:1	91:1	270:1	227:1	208:1
500	NA	182:1	111:1	385:1	308:1	263:1

* - Values were determined during extremely rough weather and are not considered to be valid.
NA - Not Available

subsurface effluent entrainment and dilution patterns, proved to be far more complex than was originally believed. The main concern was that plume dynamics were not fully identified from the study and possibly pockets of undiluted water were leading to the increased algal growth along the shoreline or were transporting high concentrations of fecal coliform bacteria. These uncertainties led EPA Region IV to require more intensive monitoring of the POTWs' effluents.

4.1.2 SEFLOE II

Currently, the National Oceanic and Atmospheric Administration (NOAA), in cooperation with the wastewater treatment facilities, the U.S. Environmental Protection Agency (EPA), and the Florida Department of Environmental Resources, is conducting a follow-up investigation known as SEFLOE II. The objective of SEFLOE II is to provide a comprehensive and more conclusive database related to plume dynamics and the impact of the POTWs' effluents on the marine environment. Small boat operations will collect twice-monthly water samples for a period of 1 year. The water samples are to be analyzed for microbial (fecal coliform, total coliform, and enterococcus), nutrient (TKN, nitrates, nitrites, ammonia, and total phosphorus), and oil and grease content. More intensive shipborne operations will be conducted two times during the year for water sampling, dye studies, acoustic data collection, and temperature and conductivity profiles. Current, temperature, and conductivity measurements will also be collected continuously for a year from instruments moored to the bottom.

4.2 Facility Description and Effluent Characterization

This section describes the wastewater treatment facilities and the location and design of the outfalls of the six POTWs examined in the SEFLOE I study. Following these descriptions are the results from the bacteriological analyses and the toxicity testing from the SEFLOE I study. These results were not available for the Delray Beach facility.

4.2.1 Delray Beach

The municipal wastewater treatment facility located in Delray Beach has a design capacity of 24 MGD. The discharge pipe has a diameter of 46.2 cm, extends 1,609 m into the Atlantic Ocean, and terminates at a depth of 29.2 m. Treatment processes consist of coarse screening, grit removal, activated sludge biological treatment, secondary clarification, and chlorination. The sludge is thickened by air flotation or centrifuge and then lime-stabilized. The final sludge is disposed of by means of land spreading.

4.2.2 City of Boca Raton

The wastewater treatment facility for the City of Boca Raton has a design capacity of 20 MGD and an open ocean outfall. Treated wastewater is discharged into the Atlantic Ocean 1,524 m from the shoreline at a depth of 27.4 m. The discharge pipe undergoes two expansions along its length, starting with a diameter of 76.2 cm, expanding to 91.4 cm, and expanding again to 106.7 cm.

The raw influent is treated by the screening of solids, grit separation, primary clarification, conventional stabilized activated sludge in three aeration basins, and secondary clarification. The sludge is anaerobically digested, dewatered to 12 percent, and disposed of in a landfill.

4.2.3 Broward County North District

The Broward County North District wastewater treatment facility, located in Pompano Beach, has a design capacity of 66 MGD. The treatment processes consist of screening and grit removal, activated sludge biological treatment, secondary clarification, and chlorination. Treated wastewater travels through a 137-cm-diameter pipe prior to its discharge into the Atlantic Ocean. The terminal end of the pipe is 2,134 m from the shoreline and at a depth of 33.5 m. Sludge is thickened by flotation before being anaerobically digested and filter pressed. The final sludge is disposed of in a sanitary landfill.

Bacteriological Data

Bacterial counts (colonies/100 mL) during chlorination (October 8, 1987) were consistently low and ranged from 0 to 10 for total coliform, from 0 to 10 for fecal coliform, and from 0 to 150 for fecal streptococcus. The bacterial counts from January 11, 1988, when there was no chlorination, are reported as being inconsistent. Within the zone of initial dilution (ZID) they ranged from <100 to 4500 for total coliform, <10 to 530 for fecal coliform, and 40 to 260 for fecal streptococcus. Ranges recorded within the farfield mixing zone were <100 to 1100 for total coliform, <10 to 240 for fecal coliform, and 30 to 490 for fecal streptococcus.

Bioassay Data

Acute toxicity screening tests were conducted on 43 water samples taken from the ZID and the farfield mixing zone. The sheepshead minnow, *Cyprinodon variegatus*, was used in 24 of the tests and the mysid shrimp, *Mysidopsis bahia*, was used in 19 of the tests. None of the water samples proved to be toxic to either organism.

Definitive acute toxicity tests were performed using effluent dilutions of 6, 12, 25, 50, and 100 percent. Twelve tests used the sheepshead minnow and resulted in LC_{50} values (the effluent concentration that is toxic to 50 percent of the test organisms) of 100+ percent. The mysid shrimp was used for eight tests, and the resulting LC_{50} values were 100+, 100, and 85 percent. Presently, the State requires that effluent be nontoxic at a concentration of 30 percent. The LC_{50} values obtained from the definitive tests indicate that the Broward County effluent is well within the FDER requirement.

4.2.4 City of Hollywood

The Hollywood municipal wastewater treatment facility discharges primarily residential wastewater at a rate of 38 MGD into the Atlantic Ocean. The outfall pipe has a 152.4 cm diameter and extends 3,048 m offshore, terminating at a depth of 27.4 m. The facility's treatment processes consist of screening and grit removal, two parallel pure oxygen activated sludge reactors, six secondary clarifiers, and postchlorination. The sludge is thickened by gravity beds and stabilized with lime. The resulting dewatered sludge is disposed of via land application in nearby agricultural areas.

Bacteriological Data

Water samples were collected at the treatment plant, within the ZID, and in the farfield mixing zone, with and without chlorination. The samples were tested for total coliform, fecal coliform, and fecal streptococcus. During chlorination, bacteria levels (colonies/100 mL) within the mixing zone ranged from 0 to 48 for total coliform, 0 to 18 for fecal coliform, and 0 to 95 for fecal streptococcus. Bacterial data for the unchlorinated samples were incomplete and were not included in the final report. The report noted that fecal coliform samples collected at 800 m from the outfall were either at or above acceptable levels, whereas samples collected 400 m from the outfall were at completely acceptable levels. Laboratory error was speculated to be responsible for the confounding results.

Bioassay Results

Fifteen 96-hr acute toxicity tests were performed using samples of effluent that were collected prior to dilution at the end of the pipe. The mortality of the invertebrates (*Ceriodaphnia dubia*, *Daphnia pulex*, and *Mysidopsis bahia*) was 100 percent, and the survival of the vertebrates (*Pimephales promelas* and *Cyprinodon variegatus*) was 100 percent. Previous water samples have contained the organophosphate pesticide Diazinon, and toxicity tests proved Diazinon to be the source of the invertebrate mortality. Seventy-two toxicity tests were performed on water samples collected within the ZID and the farfield mixing zone. None of the samples proved to be toxic to the test organisms.

4.2.5 Miami-Dade North District

The Miami-Dade North District municipal wastewater treatment plant has a design capacity of 80 MGD and a discharge outfall located in the Atlantic Ocean. The discharge pipe extends 3,353 m offshore, has a 228.6-cm diameter, and terminates at a depth of 32.9 m. The T-shaped diffuser (Figure 4-3) comprises twelve 61-cm ports (six on each side of the T) and has a total length of 112.5 m. Treatment at this facility consists of prescreening, primary settling, degritting, activated sludge, secondary clarification, and disinfection. The sludge is pumped to the Miami-Dade Central District facility, where it is anaerobically digested, air dried, and used as fertilizer.

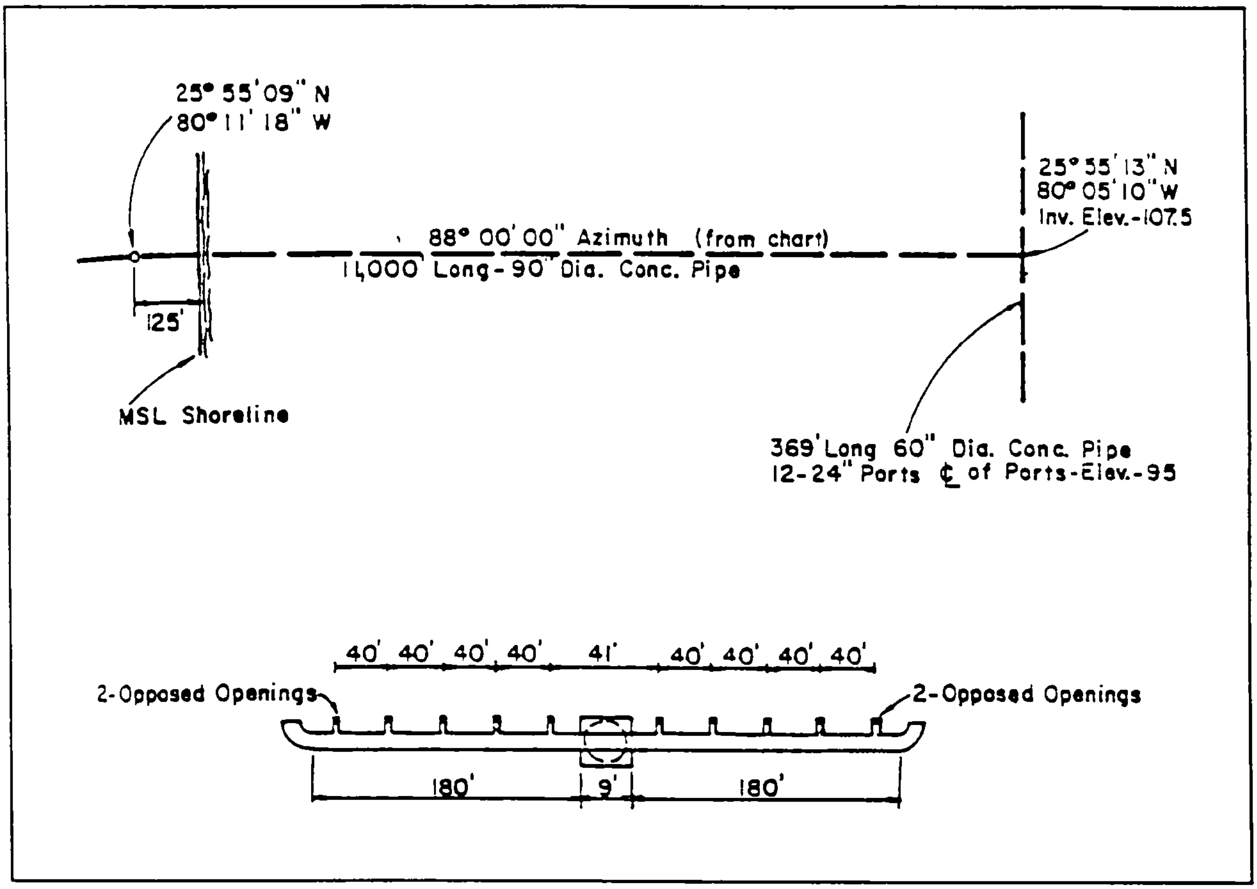


Figure 4-3. Schematic of the Miami-Dade North District discharge pipe and diffuser.

Bacteriological Data

During chlorination, total coliform counts ranged from 0 to 10 per 100 mL, fecal coliform ranged from 0 to 10 per 100 mL, and fecal streptococcus ranged from 0 to 45 per 100 mL. These values fall well within the requirements established by the FDER. Data from the 2 days when chlorination did not occur are inconsistent and do not correlate with previous sampling efforts.

Bioassay Data

Screening toxicity tests were performed on water samples collected from the treatment plant prior to discharge, from within the ZID (29 samples), and from the farfield mixing zone (26 samples). Definitive toxicity tests were conducted using plant effluent at test concentrations of 1, 10, 30, 60, and 100 percent. Both types of tests (screening and definitive) were acute 96-hr toxicity tests using the sheepshead minnow, *Cyprinodon variegatus*, and the mysid shrimp, *Mysidopsis bahia*.

The 100 percent effluent used in the screening tests proved to be nontoxic to the sheepshead minnow; however, four of the eight samples tested with mysids proved to be acutely toxic. None of the samples from within the ZID were acutely toxic to either organism, and only one sample from the farfield mixing zone was toxic. This sample caused 60 percent mortality to mysid shrimp, and its toxicity was thought to be due to the unusually high salinity of the test solution.

The LC_{50} values from all of the definitive toxicity tests using the sheepshead minnows were 100+ percent. The four tests using the mysid shrimp resulted in LC_{50} values of 100+, 100, 63, and 77 percent. All of these results are well above the 30 percent limit imposed by the State.

An EPA-approved pretreatment program was implemented for all the industrial facilities that discharge to the Miami-Dade North District treatment plant. Pretreatment programs require industrial dischargers to treat their wastewater prior to discharging it to the POTW. The pretreated wastewater is subject to compliance with EPA-approved pretreatment standards. Once properly treated, the industrial wastewater can be discharged to the POTW. Table 4-3 is a summarization of the results for the Miami-Dade North District treatment facility following the implementation of the pretreatment program.

4.2.6 Miami-Dade Central District

The Miami-Dade Central District municipal treatment facility discharges primarily residential wastewater at a rate of 133 MGD into the Atlantic Ocean. The discharge pipe is 5,740 m long and has two sections. The first section has a 228.6-cm diameter and extends 1,347 m. It is coupled to a 304.8-cm diameter pipe that extends another 4,393 m (Figure 4-4). The terminal end has five 122-cm ports and is located in 33.5 m of water. Wastewater from the Central District is proportioned to two treatment plants located on Virginia Key. Treatment capabilities at Plant #1 consist of an aerated channel for grit removal, high-rate activated sludge treatment, and digestion of the sludge upon removal from the final clarification tanks. The facilities at Plant #2 are similar; however, pure oxygen is used for the activated sludge process. Digested sludge is dewatered by centrifugation and drying beds. Processed sludge is used for fertilizer by the nearby agricultural community.

Until the late 1970s, the Miami-Dade Central District outfall extended only 1,347 m from shore. D'Amato and Lee (1977) developed a kinematic model that illustrated plume behavior for the Miami-Dade Central District facility. The outfall was located in waters only 5 m deep and dominated by prevailing winds. The model indicated that the effluent plume entered Biscayne Bay through Government Cut, Norris Cut, or Bear Cut, depending on wind direction. The effluent was rarely, if ever, carried offshore. In 1978, plans were developed to add an additional 4,393 m of pipe, creating a discharge pipe that extended 5,740 m offshore and terminated at a depth of 33.5 m. The relocation of the outfall was one step toward decreasing the impact of wastewater on areas designated for uses involving human contact.

Bacteriological Data

The October 27 (chlorination day) samples from the farfield mixing zone ranged from 1 to 4 counts per 100 mL for total coliform. Fecal coliform values were undetected, and fecal

Table 4-3. Summary of Priority Pollutants Detected After Implementation of an EPA-Approved Pretreatment Program at the Miami-Dade North District POTW

Detected Inorganic Priority Pollutants (mg/L)	2/86	2/87	2/88	DER Effluent Maximum Concentration (EMC)	EPA Marine Acute Toxicity (MAC)
Lead	0.048	ND	0.025	0.500	N/A
Nickel	0.050	ND	0.030	1.000	N/A
Total Cyanide	ND	0.005	0.01 **	NE	N/A
Thallium	0.014	ND	0.00006	0.480	N/A
Antimony	ND	ND	0.027	NE	N/A
Cadmium	ND	ND	0.0039	0.1	N/A
Detected Organic Priority Pollutants (μg/L)					
Chloroform*	16	17	13	1,570	N/A
Tetrachloroethane 1,1,1-	27	5.9	ND	1,000	9,020
Trichloroethane	4.1	7.6	8.2	NE	31,200
Trichlorethene	1	1.4	ND	8,000	N/A
Bis (2-ethylhexyl) Phthalate	13	ND	ND	NE	2,944
Tetrachloroethene	ND	ND	5.1	885	N/A
Phenols	ND	ND	.0034	NE	N/A

* Chlorination by-product
** Test detection limit
ND - Not Detected
NA - Not available
NE - Not established

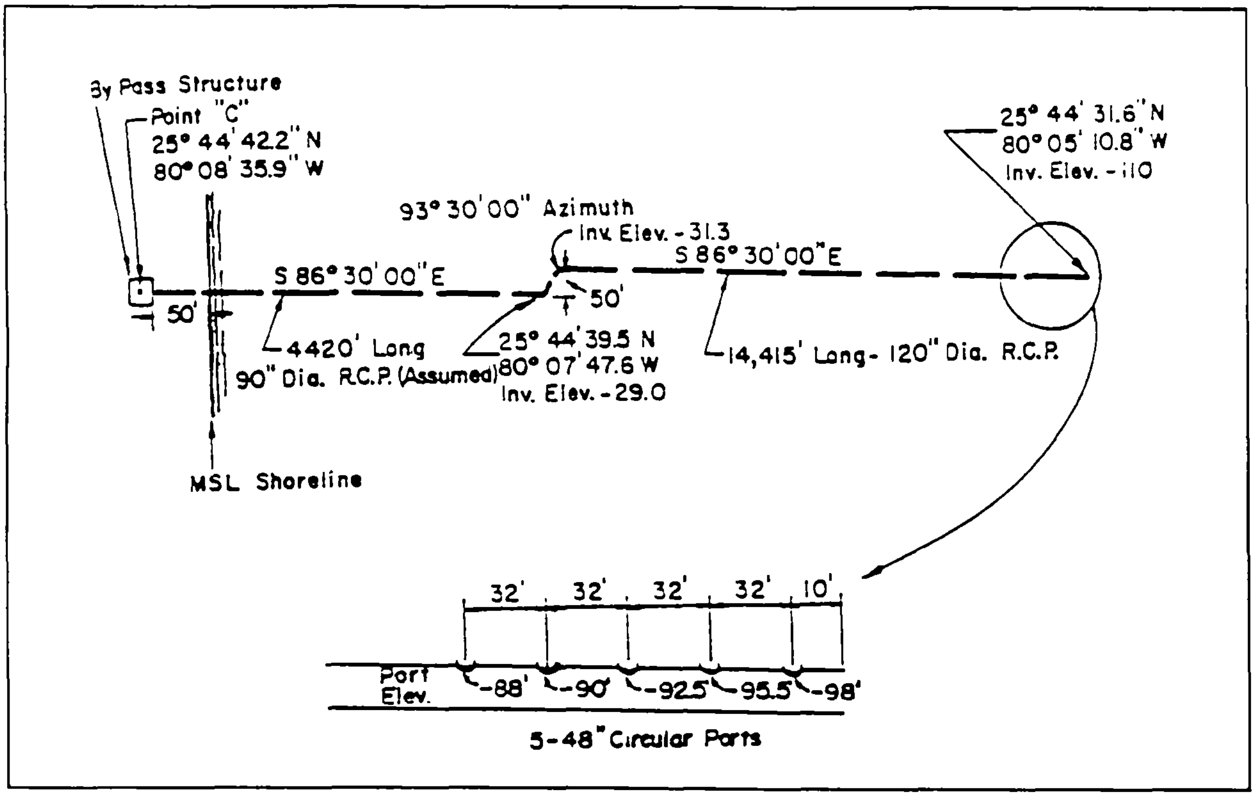


Figure 4-4. Schematic of the Miami-Dade Central District discharge pipe and diffuser.

streptococcus ranged from 1 to 37 counts per 100 mL. All values were well below the FDER limits. The report indicates that the bacteriological data collected on the day of no chlorination were inconsistent and did not correlate with previous sampling efforts. The data were not reported, and laboratory and/or sampling error was given as the cause of the inconsistency.

Bioassay Data

Acute 96-hr toxicity tests were conducted on water samples collected from treatment plant effluent prior to discharge, within the rising plume at the edge of the ZID (16 samples), and in the farfield mixing zone (24 samples). Both screening (single-dilution) and definitive (multiple-dilution) tests were performed, using the sheepshead minnow, *Cyprinodon variegatus*, and the mysid shrimp, *Mysidopsis bahia*, as test organisms.

The screening tests using 100 percent effluent proved to be nontoxic to the sheepshead minnow; however, three of four samples were toxic to the mysids. None of the samples from the ZID or the farfield mixing zone were acutely toxic.

The definitive toxicity tests used effluent concentrations of 1, 10, 30, 60, and 100 percent to determine the LC_{50} concentration limit. Ten definitive tests were done at the Miami-Dade Central

District plant—four with sheepshead minnows and six with mysid shrimp. All the tests with the sheepshead minnows had an LC_{50} value of 100+ percent, whereas only three of the six mysid tests had LC_{50} values of 100+ percent, with the remaining values being 88, 55, and 55 percent. All tests met the FDER toxicity test requirement of 30 percent.

An EPA-approved pretreatment program was implemented for all the industrial facilities that discharge to the Miami-Dade Central District treatment plant. Table 4-4 is a summarization of the results for the Miami-Dade Central District treatment facility following the implementation of the pretreatment program.

4.2.7 Summary

Yearly average discharge values for each of the six POTWs in the southeast Florida study area are compiled in Table 4-5. These values were calculated from the monthly data collected from July 1990 to July 1991 for the NPDES monitoring program.

4.3 Impacts from POTW Discharges

4.3.1 Effluent Plume Dynamics

The characteristics of the terminal end of the discharge pipe (i.e., open-ended or diffuser), the properties and volume of the effluent, the receiving water, the diffuser design, and the depth of the discharge pipe determine the possible level of dilution for each of the six POTW effluent discharges (USEPA, 1982). Perhaps the most significant characteristics are the use of multiport diffusers and treatment processes that remove large particulates. Properly designed multiport diffusers can generate sufficient dispersion and dilution to significantly reduce the detrimental effects of the effluent on the marine environment (Grigg and Dollar, 1990). Nonsaline effluent rises rapidly upon discharge from the pipe, and as it rises the effluent entrains ambient saline waters (Figure 4-5). This dilution causes the density to increase and the buoyancy to decrease. If the water column is stratified, by either a thermocline or pycnocline, the plume will level out and move horizontally at the point of neutral buoyancy. If the density gradient is insufficient or nonexistent, the plume will rise to the surface before flowing horizontally with the prevailing surface current. This completes the process referred to as the "initial dilution." The vertical distance from the discharge point to the point of neutral buoyancy is known as the "height of rise" (USEPA, 1982). The volume of water and the underlying seabed are known as the "zone of initial dilution" or ZID. It is within the ZID that compliance with acute water quality standards and acute toxicity requirements must be maintained. The benthos underlying the ZID may be subjected to chronic levels of pollutants, although typically chronic toxicity is measured in the farfield mixing zone and not in the ZID. Figure 4-6 shows ZID configurations for various types of diffusers.

Table 4-4. Summary of Priority Pollutants Detected After Implementation of an EPA-Approved Pretreatment Program at the Miami-Dade Central District POTW

Detected Inorganic Priority Pollutants (mg/L)	2/86	2/87	2/88	DER Effluent Maximum Concentration (EMC)	EPA Marine Acute Toxicity (MAC)
Lead	0.064	ND	ND	0.500	N/A
Nickel	ND	ND	0.011	1.000	N/A
Total Cyanide	ND	0.005	ND	NE	N/A
Thallium	0.027	ND	0.00015	0.480	N/A
Antimony	ND	ND	0.025	NE	N/A
Detected Organic Priority Pollutants (µg/L)					
Chloroform*	7.1	5.6	7.4	1,570	N/A
Tetrachloroethane	27	5.9	ND	1,000	9,020
1,1,1-Trichloroethane	ND	1.0	4.4	NE	31,200
Trichlorethene	2.2	3.2	2.0	8,000	N/A
Methylene Chloride	ND	ND	24	NE	2,944
Tetrachloroethane	ND	8.5	4.5	885	N/A
Phenols	ND	ND	.0031	NE	N/A
Trans-1,2-Dichloroethene	4.7	1.7	2.0	NE	

* Chlorination by-product.
ND - Not detected
NA - Not available
NE - Not established

Table 4-5. Yearly Average Effluent Concentrations for Each POTW of Concern

	Flow	BOD (mg/L)	BOD (lb/d)	TSS (mg/L)	TSS (lb/d)	Fecal Coliform (counts/ 100 mL)	Total Residual Chlorine (mg/L)
Delray Beach	14.5	10.0	1196.6	11.5	1416.8	100.4	2.4
Broward County	62.4	12.0	6200.7	6.6	3446.8	4.5	N/A
Boca Raton	11.8	8.0	N/A	4.7	N/A	2.3	0.9
Hollywood	36.6	10.3	3125.6	17.2	5272	6.9	N/A
Miami-Dade North	86.6	13.1	9576.5	19.4	14,138	19.2	0.6
Miami-Dade Central	124.7	16.8	17,576	14.8	12,065	2.2	2.9

4.3.2 Impacts on Water Quality

The level of impact on the receiving water depends on the quantity and composition of the effluent, the receiving water conditions, and the level of dilution achieved (USEPA, 1982). Water quality parameters of the most concern are dissolved oxygen (and biochemical oxygen demand), suspended solids, nutrients (i.e., nitrogenous compounds and phosphates), toxics, and coliform bacteria. Pathogenic human enteric viruses also were detected in the vicinity of non-treated as well as secondarily treated, chlorinated sewage effluents off southeast Florida in the late 1970s (Edmond et al., 1978).

Increased levels of organic matter from sewage effluent and the subsequent oxidation of that organic matter may depress dissolved oxygen concentrations in areas that are already existing at the upper threshold of tolerance (Bathen, 1968; Kinsey, 1973; Johannes, 1975). The ambient dissolved oxygen concentration is most critical at night (Pastorok and Bilyard, 1985). Reef communities maintain a constant rate of respiration (Kinsey, 1973), and nighttime oxygen concentrations may fall to near zero. An increase in the biological oxygen demand resulting from the oxidation of organic matter by bacteria may depress oxygen concentrations below critical levels, significantly stressing the reef community. Suspended solids impact the environment by increasing turbidity, thereby lowering the transmission of light through the water column. This, in turn, affects the biological communities in a number of ways. The accumulation rate of solids is dependent on effluent discharge rates, the concentration of suspended solids in the effluent, settling characteristics, current conditions, and the presence of density stratifications (USEPA, 1982). Hydrogen sulfide generation and toxic by-products of pesticides, herbicides, chlorine, and heavy metals can produce deleterious effects (Grigg and Dollar, 1990), as can increased

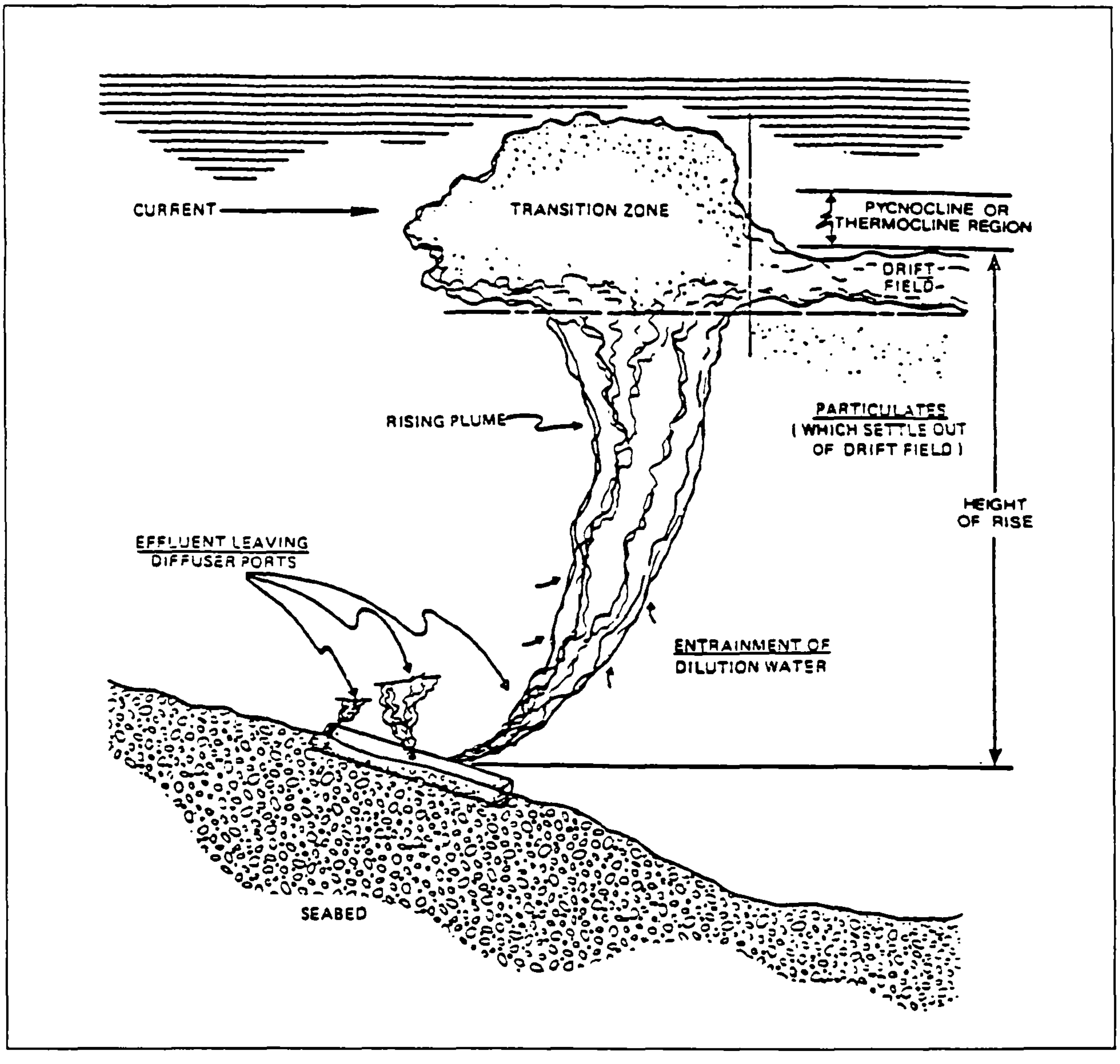


Figure 4-5. Schematic of wastefield from an open ocean outfall (USEPA, 1982).

nutrient (e.g., nitrates and phosphates) loadings. Simkiss (1969) found that inorganic orthophosphates, pyrophosphates, and organic phosphates (e.g., glycero-phosphate or adenosine triphosphate) strongly inhibit the calcification process by acting as crystal poisons.

Effluent from municipal treatment facilities has the potential to adversely impact recreational and commercial fisheries through the bioaccumulation of toxic organics or the induction of diseases that lower or eliminate the marketability of the catch (USEPA, 1982). Fish embryos are also sensitive to sewage effluents, resulting in toxic effects on embryo viability, time of hatching, mortality during hatching, post-hatch larval survival, and larval feeding (Costello and Gamble, 1992).

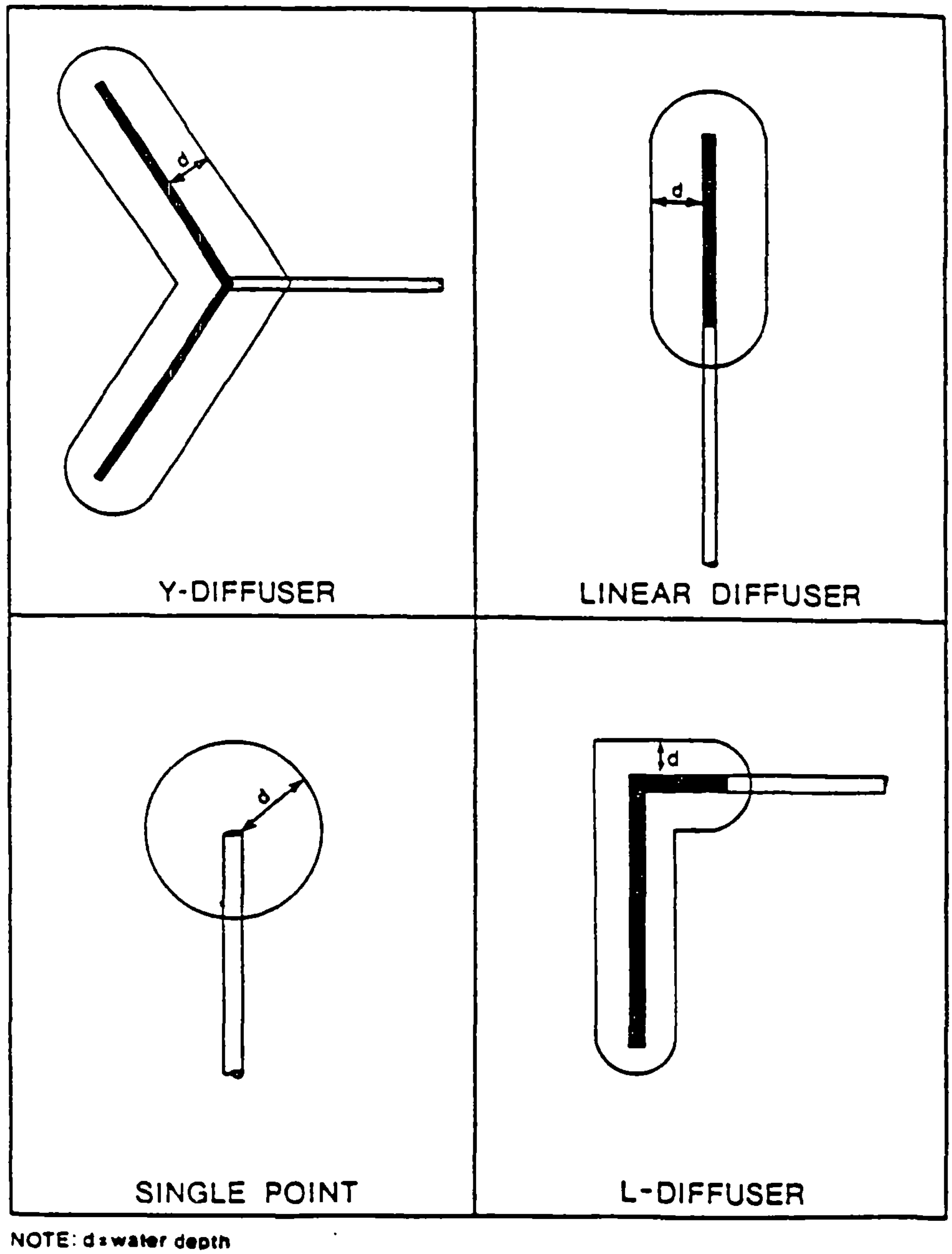


Figure 4-6. Types of diffusers and their corresponding ZID configurations (USEPA, 1982).

Most Significant Factors Affecting Water Quality

- The Miami-Dade North, the Miami-Dade Central, and the Broward County facilities discharge the largest quantities of effluent.

- The Miami-Dade North, the Miami-Dade Central, and the Broward County facilities also discharge the highest concentrations and loadings of BOD.

- The highest loadings of TSS are from the Miami-Dade North, Miami-Dade Central, and Hollywood facilities. Although Miami-Dade Central has a higher loading than the Hollywood facility, the Hollywood plant has the higher TSS discharge concentration.

- The Delray Beach facility discharges the highest concentration of fecal coliform despite the fact that it is second only to Miami-Dade Central for total residual chlorine.

- All six of the POTWs complied with the Florida Department of Environmental Resources' requirements for toxicity. None of the effluents proved to be acutely toxic.

5. NONPOINT SOURCE CONTRIBUTIONS TO WATER QUALITY

Nonpoint sources of pollution are those that cannot be attributed to one specific source. The primary contributor to nonpoint source (NPS) pollution in coastal areas is urban runoff. As discussed later in this section, the nonpoint source loadings associated with urban areas are usually a function of the land use and land activity.

5.1 General Land Use Patterns

The development of land is a major contributor to nonpoint source pollution, primarily because of stormwater runoff from both urban and agricultural land uses. As the area is developed, the amount of impervious area and actively maintained landscape areas increases, leading to increased runoff and pollutant loadings. Figures 5-1 through 5-3 illustrate the general land use distribution throughout the counties in the study area. Table 5-1 shows general land uses for the three counties in the study area.

5.2 Sources of Nonpoint Source Pollution

There are a variety of contributors to the nonpoint source loading of pollution, including urban stormwater runoff, agricultural runoff, marinas, and mismanagement of household toxics.

5.2.1 Urban Stormwater Runoff

As an area becomes developed, the amount of impervious surface increases, thereby increasing the volume and velocity of runoff. In addition, the amount of actively maintained landscape increases as an area is developed. Landscope maintenance may lead to increases in nutrient and pesticide loadings from excess application. Urban runoff contains a variety of nonpoint source pollutants, including heavy metals, hydrocarbons, fertilizers, pesticides, and oils and greases, depending on the land use. Unless stormwater runoff is treated using best management practices

Table 5-1. General Land Uses for Counties in the Study Area (Acres)

County	Urban	Agriculture	Range Land	Forested Upland	Wetlands[a]	Water	Barren Land	Total
Broward	181,633	44,071	25,093	9,013	502,119	14,956	4,575	781,460
Dade	223,702	93,470	10,774	32,902	872,398	198,078	5,790	1,437,114
Palm Beach	224,423	587,823	6,579	30,209	401,931	168,181	4,047	1,416,157

SOURCE: South Florida Water Management District data, 1991, unpublished.
[a] The majority of these wetlands are within the Water Conservation Areas or Everglades National Park.

See legend on p. 5-9.

Source: SFWMD, 1992.

Figure 5-1. General land use for Broward County.

See legend on p. 5-9.

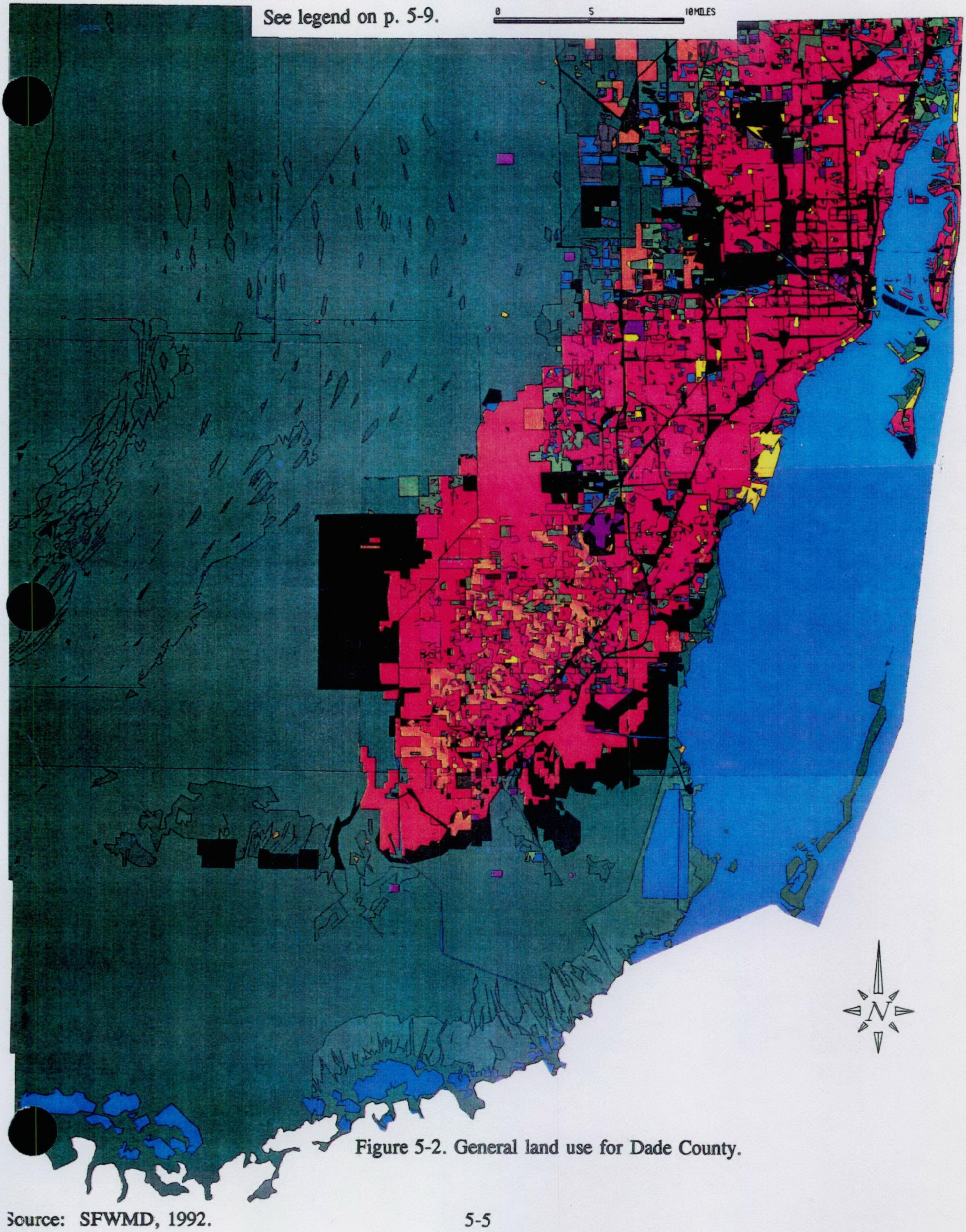
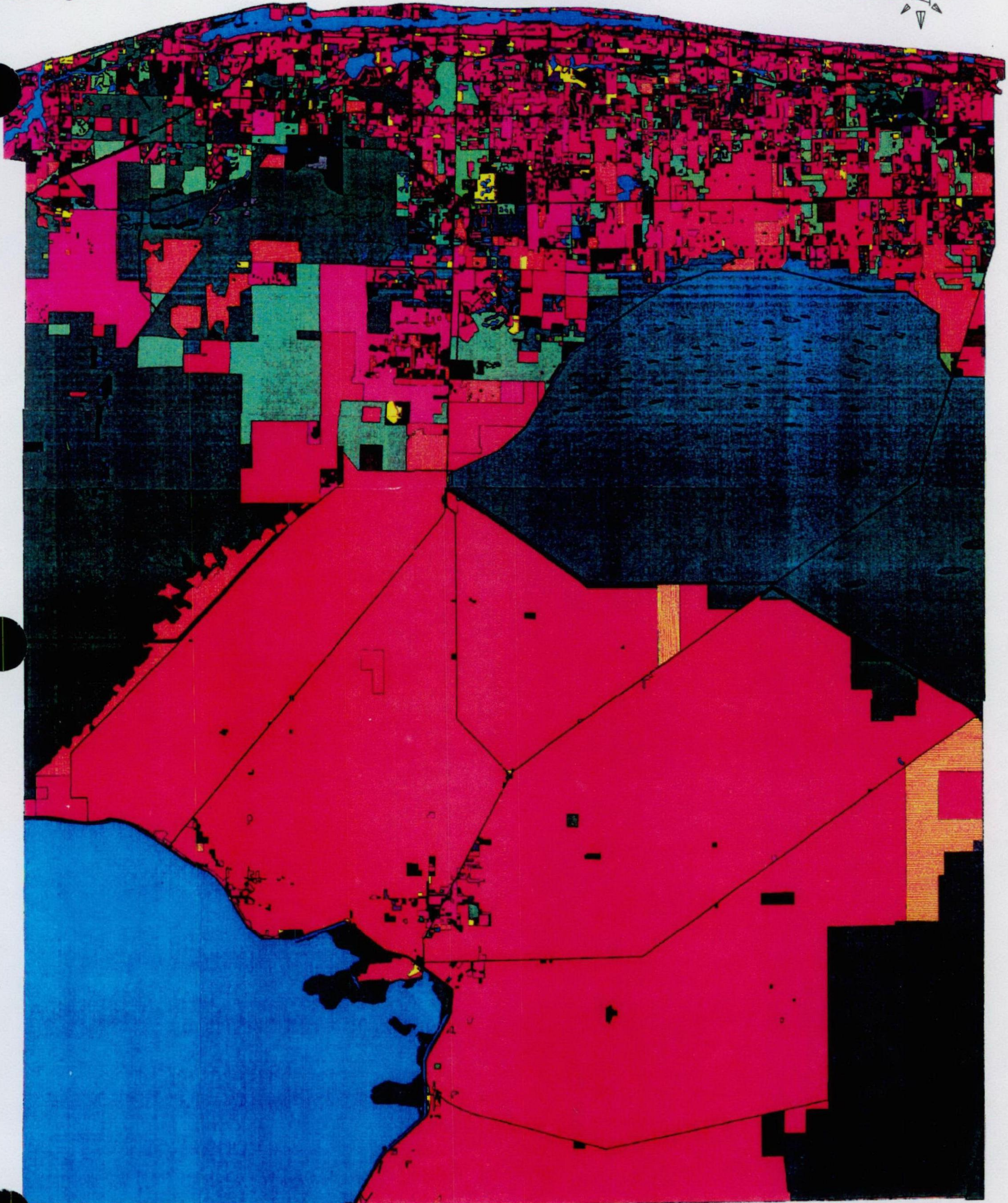


Figure 5-2. General land use for Dade County.

Source: SFWMD, 1992.

See legend on p. 5-9.



Source: SFWMD, 1992.

Figure 5-3. General land use for Palm Beach County.

Legend for Figures 5-1 through 5-3.

- Urban
- Commercial/Industrial
- Recreation and Open Space
- Water
- Agriculture
- Wetlands

depending on the land use. Unless stormwater runoff is treated using best management practices (BMPs) prior to discharge to coastal waters or their tributaries, these pollutants may end up offshore. Estimates of various pollutant loadings from three urban land uses are presented in Table 5-2.

Table 5-2. Estimated Stormwater Pollutant Loads for South Florida (lb/ac/yr)

Land Use	TOC	BOD	COD	TN	TP	Pb	TSS	Zn
Highway	0.41	n/a	0.94	0.02	0.002	0.007	14.0	0.002
Commercial	0.40	n/a	1.8	0.02	0.002	0.010	9.9	0.003
Residential	0.42	0.46	0.70	0.046	0.009	0.003	n/a	0.003

SOURCE: Adapted from Wanaliesta and Yousef, 1985.

Pollutant loadings vary depending on several factors, including volume of rainfall, amount of impervious surface, land use, and effectiveness of best management practices (BMPs) in use. Practices such as allowing open space and buffer areas to filter stormwater runoff, minimizing use of pesticides and fertilizers, street sweeping, and public education will minimize the contributions of urban areas to nonpoint source pollution.

5.2.2 Agricultural Runoff

Agricultural land uses are very chemically intensive. Application of pesticides and fertilizers is a necessary component. Land tilling and erosion account for large quantities of sediment being carried away by runoff. Because of the relatively flat topography of the area, erosion from agricultural areas is not as critical a concern in South Florida as it is in other parts of the country; however, because of the geology of the area (limestone substrate containing an unconfined aquifer), pollutants can be introduced directly into the water table and flow to the secondary and primary canals and ultimately to the coast.

5.2.3 Marinas and Other Nearshore Industries

Commercial marinas are another source of nonpoint pollution. In addition to sewage released from on-board septic systems, metals and metal-containing compounds have many functions in boat operations, maintenance, and repair. Copper and tin are found in biocides used to kill marine fouling organisms that attach themselves to boats and pilings. Lead is used as a fuel additive and may be released through incomplete combustion and boat bilge discharge. Zinc anodes are used to deter corrosion of metal hulls and engine parts.

Because they do not dissolve in water and are readily bound to sediment, many of the pollutants associated with marina activity do not cause problems in the water column but accumulate in the bottom sediments. They become a water quality issue when resuspension of sediment occurs, as has occurred in the Miami River.

5.2.4 Mismanagement of Household Toxics

Improper management of household hazardous materials and waste can be a significant contributor to nonpoint source pollution. Sources of nonpoint pollution from households include improper lawn and garden care, improper disposal of household hazardous wastes (such as backyard and storm drain dumping), use of phosphate detergents, and failing septic systems. All of these household activities have the potential to degrade water quality.

Sources of household hazardous waste include household cleaners, automotive products, and lawn and garden products. These can pose a threat to the environment if disposed of improperly. The usual undesirable methods of disposal include storm and sanitary sewers, landfilling, illegal dumping, and long-term storage leading to container deterioration.

It is difficult to determine the volume of hazardous materials that are improperly disposed of and become nonpoint source pollution. To illustrate the potential impacts of improperly disposing of household hazardous waste and waste oil, the Washington Department of Ecology estimates that of the more than 4.5 million gallons of used oil dumped in Washington each year, 2 million gallons end up in Puget Sound (USEPA, 1988). This is significant given that 1 quart of oil can contaminate up to 2 million gallons of drinking water. The 4 quarts of oil from a car engine can form an oil slick approximately 8 acres in size (University of Maryland Cooperative Extension Service, 1987).

5.3 Runoff Characterization/Estimates of Mass Loadings

As discussed in Section 5.2.1, urban runoff is a major contributor to nonpoint source pollution in the study area. Because of the extensive drainage system in the area and the porous limestone, most urban runoff ultimately flows to the canals of the primary drainage system. Best management practices (BMPs) treat a portion of the urban stormwater runoff; however, much of the coastal area in south Florida was developed prior to current requirements for treating stormwater runoff. Therefore, runoff flows untreated from impervious surfaces or maintained landscapes to surface waters. Agricultural runoff may also have an impact in this area. The primary system drains from south of Lake Okeechobee, through the Everglades Agricultural Area, to the more developed portions of Dade, Broward, and Palm Beach counties. In addition, there is agricultural development, primarily grazing, citrus groves, and ornamental horticulture, in the eastern portions of the counties.

Table 5-3 provides estimates of loadings for three nonpoint pollutants within the study area. Flow and water quality data from certain primary canals maintained by the South Florida Water Management District were evaluated to estimate loadings for certain pollutants. The sites are identified in Figure 5-4. This information was derived from data collected by the South Florida

Table 5-3. Estimated Loadings for Nonpoint Source Pollution from Select Points in the Study Area

SITE	Flow (mgd)	BOD (lb/day)	BOD (lb/10^6 gal)	Total P (lb/day)	Total P (lb/10^6 gal)	Total n (lb/day)	Total N (lb/10^6 gal)
1	0.064	0.97	15.2	0.04	0.62	0.8	12.5
2	2,761	30,802	11.2	2,228	0.82	2,383	8.5
3	1,603	2,970	1.9	543	0.34	2,038	1.3
4	924	11,272	12.2	267	0.29	9,907	10.7
5	2,562	28,659	11.2	2,785	1.09	21,020	8.2
6	750	9,211	12.3	749	1.5	6,764	9.0

SOURCE: Adapted from data from the South Florida Water Management District, 1985-1989.

Water Management District, and collection periods and frequencies varied by site. Data were collected approximately quarterly at each canal site and included flow, BOD, total phosphorus, and total nitrogen. Other data were collected at some sites, but data collection was not consistent from site to site. Data in Table 5-3 represent quarterly loadings for canal sites that were averaged over a 5-year period (1985 to 1989). Missing data were treated accordingly, and times of zero flow were treated as zero loadings.

Table 5-3 reports pollutant contributions as a loading in pounds per day to indicate an estimate of the total (or absolute) contribution of a pollutant from the canals. Additionally, data are presented as a concentration in pounds per million gallons per day to provide a way to compare estimates of the relative contribution of pollutants.

The data presented here are intended only to illustrate the potential of nonpoint sources of pollution to affect offshore water quality. The lack of continuous, long-term data collected from the canals does not allow for cause-and-effect relationships to be evaluated. Additionally, a linkage of nearshore to offshore pollutant transport has not been established for this study area. Thus conclusions that nonpoint sources, point sources, or a combination of both causes or contributes to adverse impacts to offshore communities can not be made from the available data.

The available data are, however, useful for providing an indication of the potential relative and absolute contributions of point and nonpoint sources of pollution to declines in offshore water quality and impacts on the coral ecosystems. Trends in the available data are also useful for targeting pollutant sources and defining cost-effective ways to reduce both point and nonpoint pollution.

Most Significant Factors Affecting NPS Contributions

- The type of land use and the control measures in place to minimize nonpoint source pollution impacts are important considerations in the evaluation of the relative contribution of nonpoint source pollution.
- Heavy metals and nutrients are the main constituents of urban nonpoint source pollution. Their impacts can be controlled through public education and other nonstructural controls.

6. CONCEPTUAL MODEL OF ECOSYSTEM AND WATER QUALITY INTERACTIONS

To better understand the potential impacts of point and nonpoint sources of pollution on southeast Florida coastal communities and to guide the application of the screening model in the next section, a conceptual model was developed to provide a graphical representation and descriptive summary of existing knowledge concerning key ecosystem resources that may be affected by changes in turbidity, nutrients, and toxics from anthropogenic pollutants found in this area.

6.1 Ecosystem Components and Processes

The key resources included in this assessment are the major coastal aquatic habitats of southeast Florida. These ecosystems include mangroves (Odum et al. 1982; Odum and McIvor, 1990), seagrass beds (Zieman, 1982), and coral communities (Jaap and Hallock, 1990). Although other types of benthic communities that may be impacted by point and nonpoint source pollution also occur in this area, such as macroalgal beds, algal turfs, soft-bottom sand, and mud flats (see Alongi, 1989), they will not be considered specifically here. The mangrove forests and seagrass beds are primarily confined to estuaries and coastal lagoons.

Coral communities within the study area are poorly developed, but do form an encrusting veneer over relict reefs that lie offshore (Lighty, 1977; Dodge et al., 1991). The northernmost true coral reef on the southeast coast of Florida is generally agreed to be Fowey Rocks, approximately 16 km southeast of Miami (Dodge et al., 1991). Coral communities have received the most attention presumably because of their more sensitive nature, but possibly because of their aesthetic appeal as a research subject (Hatcher et al., 1989).

The discussion herein of the effects of stress is more extensive for coral communities than for the other major habitat types as a result of the larger quantity of information available for synthesis. The conceptual model focuses primarily on the key components of each habitat (i.e., coral colonies, mangrove trees, and seagrass plants) while attempting to address impacts to other components (e.g., sessile invertebrates, epiphytic algae, fish, zooplankton, and phytoplankton) where information is available. These key components form the primary structural and functional support for each habitat.

The productivity, biomass, species composition, and areal extent of coral, mangrove, and seagrass communities are controlled by complex interactions among chemical, physical, and biological factors, as well as human-induced sources of stress. The issues relating to the effects of point and nonpoint source pollutants on these habitats are also complex. More than one conceptual model may need to be developed to adequately address the issues and ecosystems of concern (USEPA, 1991). For example, an issue-specific model may include:

- Perturbation sources - source characterization: the kinds, amounts, locations, and sources of contaminant entry into the system;

- Affected ecosystem resources - receptor characterization: types, abundance, locations, and sensitivity of potentially affected valued resources;

- Processes affecting perturbation exposure - exposure assessment: processes affecting how pollutants get from the source to valued ecosystem resources; e.g., bioaccumulation of toxics; nutrient removal due to uptake by micro/macroalgae, sedimentation, advection, and dispersion; and

- Processes affecting resource stress - toxicity assessment: processes affecting how pollutants affect valued resources; e.g., light decreases due to increased turbidity or increased phytoplankton populations reduce benthic macroalgae, seagrass, and coral growth; toxics may be transformed and transferred through the food chain.

Appropriate issue-specific models of the complex processes occurring in the ecosystem may need to be subdivided into categories designed to fully understand each of the factors affecting the habitats and organisms. For example, sewage effluent plume dispersion results in increased particulate loading (turbidity), increased nutrient availability (eutrophication), and the release of contaminants that can be taken up by organisms in the water column and sediments (bioaccumulation). The last process may be subdivided to address contaminant transport (physical processes), contaminant transformations (chemical processes), and contaminant bioaccumulation (biological processes). Contaminant transformations could be further subdivided into those transformations occurring in the water column, biochemical transformations, and transformations in the sediments (USEPA, 1991).

The conceptual model presented here provides a preliminary look at the sources of ecosystem stress and the variables that influence the behavior of the ecosystem. These variables include light, temperature, material inputs, and other influences not under control of the system (Dahl et al., 1974). Additional subdivisions and modeling of the elements and processes would be required to provide the information needed for the development of appropriate management decisions regarding the mitigation of such stresses.

6.2 Status of the Major Ecosystems

The three key habitats of concern have been stressed by recent expansion of the human population in the South Florida area. It has been estimated that only 23 percent of the mangrove coverage existing in 1900 still remains in Dade County (Metro Dade County Planning Department, 1986). Seagrass beds in North Biscayne Bay, north of the Rickenbacker Causeway, were degraded due to thermal and municipal effluents, dredging and filling operations for construction of the Intracoastal Waterway, causeway construction, urban runoff, fish processing and ship construction industry effluents, inland drainage for flood control canals, and opening of two artificial channels between the bay and the ocean (Thorhaug, 1980). Impacts to offshore coral communities due to human activities remain less certain (Dodge, 1987; Dodge et al., 1991), although localized impacts due to dredging activity related to beach renourishment projects have been reported (Marszalek, 1980, 1981). The following subsections present a summary of recent assessments of stresses and impacts on the coral community off southeast Florida.

6.2.1 Ecological Studies

Goldberg (1973) examined the geomorphology, species composition, and zonation of the three submarine terraces off southern Palm Beach County (approximately the central portion of the area). He reported finding 27 species of scleractinian or stony corals, with 15 occuring on the flat patch reefs on the second reef terrace at a 9-meter depth. The most abundant organisms at this depth, however, were the octocorals (gorgonians or soft corals), with maximum diversity of 19 species. A total of 39 species of octocorals were found on transects at this site. These numbers compared favorably with other studies of coral/octocoral fauna in the Caribbean and the Bahamas. *Montastrea cavernosa* accounted for nearly 20 percent of all coral colonies, more common than its congener *M. annularis*, which usually dominates Caribbean and Florida Keys reefs. The large polyped *M. cavernosa* has been found to survive conditions of sedimentation and turbidity better than smaller polyped species (Lewis, 1960; Loya, 1976). Goldberg (1973) speculated that this phenomenon was the result of increased turbidity off southeast Florida. He noted that the terraces were a mile or less from shore "with its associated runoff and canal effluents" (p. 485), and that several coastal communities were operating sewage outfall pipes that terminated directly on the outer reef. The elimination or reduction of zooxanthellate (photophilic) reef corals could be attributed to these factors, Goldberg (1984) stated, since reduced temperatures and siliceous sand flows that inhibit coral growth farther north did not appear to be a problem in this study area.

6.2.2 Beach Renourishment Impacts

Several studies have evaluated the impact of activities associated with beach renourishment on the coral communities of South Florida (Figure 6-1). Unless sand is obtained from an upland source, beach renourishment most often involves dredging sand from a submerged borrow area near the renourishment site; therefore, impacts to offshore communities are from two sources—dredging of fill material and deposition of sand. Three categories of impacts on organisms result from dredging:

- Mechanical damage;
- Sediment loading from the dredge plume; and
- Increases in turbidity (Marszalek, 1981).

Mechanical damage occurs when machinery associated with dredging inadvertently comes into contact with the reef. Such contact can cause permanent or temporary damage, depending on the

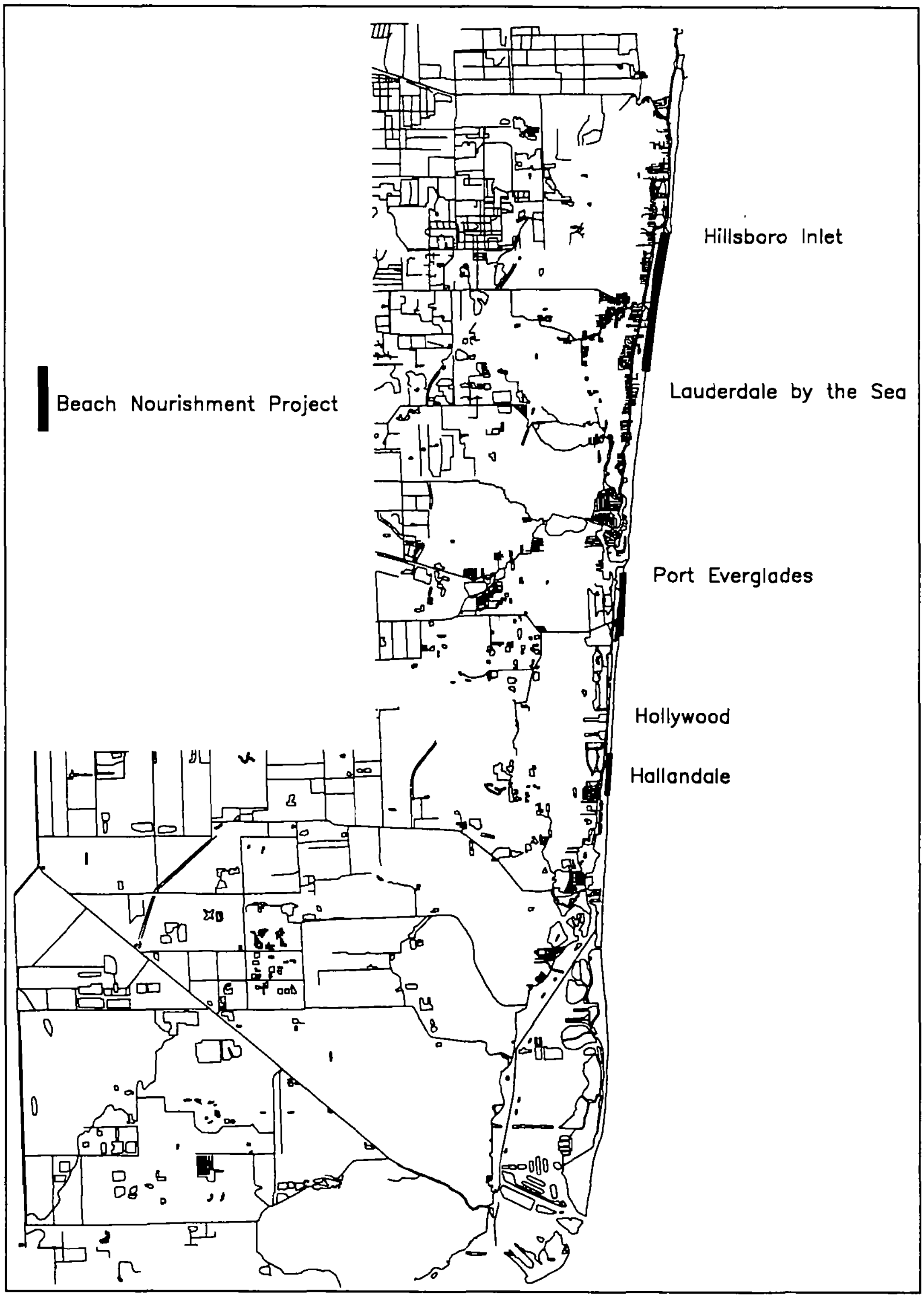


Figure 6-1. Beach renourishment projects in southeast Florida.

extent of the incident. Sources of turbidity can be dredging operations or natural, seasonal resuspension of sediment as a function of wave climate. Natural resuspension can also be compounded by the presence of fill on the beach and the quality of the fill (Marzalek, 1981).

The Metro-Dade County Department of Environmental Resources Management conducted a study from September 18 to November 12, 1980, to correlate coral morbidity and mortality associated with dredging activities. Dredging ceased on October 19, 1980. The results of the survey indicate that dredging activities at Miami Beach had affected hard-bottom communities and resulted in the deposition of a silt layer on portions of the reefs. The most severely impacted area showed evidence of direct exposure to the dredge turbidity plume. A silt layer was recognizable over about one mile of reef and then thinned to the north where the reef surface was patchily exposed. Stony corals appeared to be the most impacted by siltation and turbidity. Bleaching was observed throughout the survey area, including in 5 to 10 percent of the corals in the control transects. It was assumed that this finding was also due to natural causes, and not only the dredging. While it is difficult to distinguish between natural and dredge-induced bleaching, the bleaching was more severe in areas impacted by silt deposition. The most severe impacts observed, tissue loss and the presence of silt on the corals, were, in most cases, attributed to dredging (Marszalek, 1980).

In 1980, the Broward County Environmental Quality Control Board (now the Office of Natural Resources Protection) surveyed and monitored 12 sites in the vicinity of a beach renourishment project and its associated borrow area. Fifteen months later, the board resurveyed 10 of the 12 sites in an effort to document changes in the flora and fauna that may have occurred as a result of conditions related to beach restoration (Goldberg, 1984). Reductions in the coral population were noted. Scleractinians showed signs of tissue reduction or were missing at three stations. Reductions of 15 to 50 percent were found at five stations. Two stations showed reduced gorgonian populations and scleractinian losses. Sponge populations did not appear to vary in any observed pattern. The locations of the affected stations did not correspond to any geographic pattern that might be associated with dredging (Goldberg, 1984).

The Broward County Office of Natural Resources Protection monitored another beach renourishment project from 1989 to 1991. Monitoring occurred prior to, directly after, and one year after the project at 11 sites (4 in the vicinity of the borrow area, 6 in the vicinity of the beach fill areas, and 1 reference site). In addition, four core sampling stations (two near the borrow site and two near the fill area) were established to monitor impacts on infaunal communities. The macroinvertebrates and macroalgae showed no variations in pattern in organism diversity and abundance relative to dredge or fill activities at the sites; however, major faunal shifts were observed in the areas where the cores were taken. This was most likely due to the altered sedimentary environment. The benthic community in the borrow area was strongly modified immediately after dredging, and recovery was incomplete 1 year later (Dodge et al., 1991).

The difference in impacts from the 1980 and 1989 beach nourishment projects can be attributed to several factors. Several permit requirements for dredging were required during the 1989 project that had not been required during the 1980 project. These include the following:

- There is no anchoring or chaining near the reef;
- County staff have access to the site for monitoring at all times; and
- There is a specified buffer area between reef and borrow areas.

In addition, dredgers are using more sophisticated equipment to position dredge machinery to help avoid impacts to the reefs. Thus, although the hard grounds off the southeast Florida coast have been altered in recent years due to the physical damage caused by these dredging projects, future impacts from beach renourishment projects should be minimized.

6.2.3 Fish Community Observations

Shinn and Wicklund (1989) reported on observations made at 16 artificial reef structures off southeast Florida, including wrecks, abandoned oil rigs, and the Miami-Dade Central District sewage outfall using a manned submersible. Of all the artificial and natural reefs observed, the greatest diversity and numbers of fish were at the sewer outfall. Some of the fishes observed included French grunts (*Haemulon flavolineatum*), pork fish (*Anistostremus virginicus*), spade fish (*Chaetodipterus faber*), amberjack (*Seriola dumerili*), horse eye jack (*Caranx latus*), blue runners (*C. fusus*), barracuda (*Sphyraena barracuda*), Bermuda chubs (*Kyphosus sectatrix*), yellowtail (*Ocyurus chrysurus*), and gray hogfish (*Lachnolaimus maximus*). Two species, the sting ray (*Dasyatis americana*) and the cobia (*Rachycentron canadum*), were observed at the outfall and at no other site.

The effluent, with over 90 percent of the solids removed by the treatment plant, had a brown translucent appearance and did not mix with the water column until it neared the surface. The fish avoided entering the plume and appeared to be in an excited state, rapidly swimming back and forth. It was speculated that perhaps the fish were attracted to the sound of the effluent rushing out of the outfall. The southern area of the outfall is a popular fishing spot; however, the boil, and the north and west sides of the boil, are avoided. The area is never used by divers.

Effects of other environmental perturbations related to point and nonpoint source pollution have not been adequately examined in the southeast Florida coastal communities.

6.3 Sources of Stress and Interactions Between Sources

A conceptual model of the sources of stress and their interactions is outlined in Figure 6-2. The types of stress that affect coral, mangrove, and seagrass communities may be divided into water quality-related effects and direct physical impacts to these communities. This section will briefly

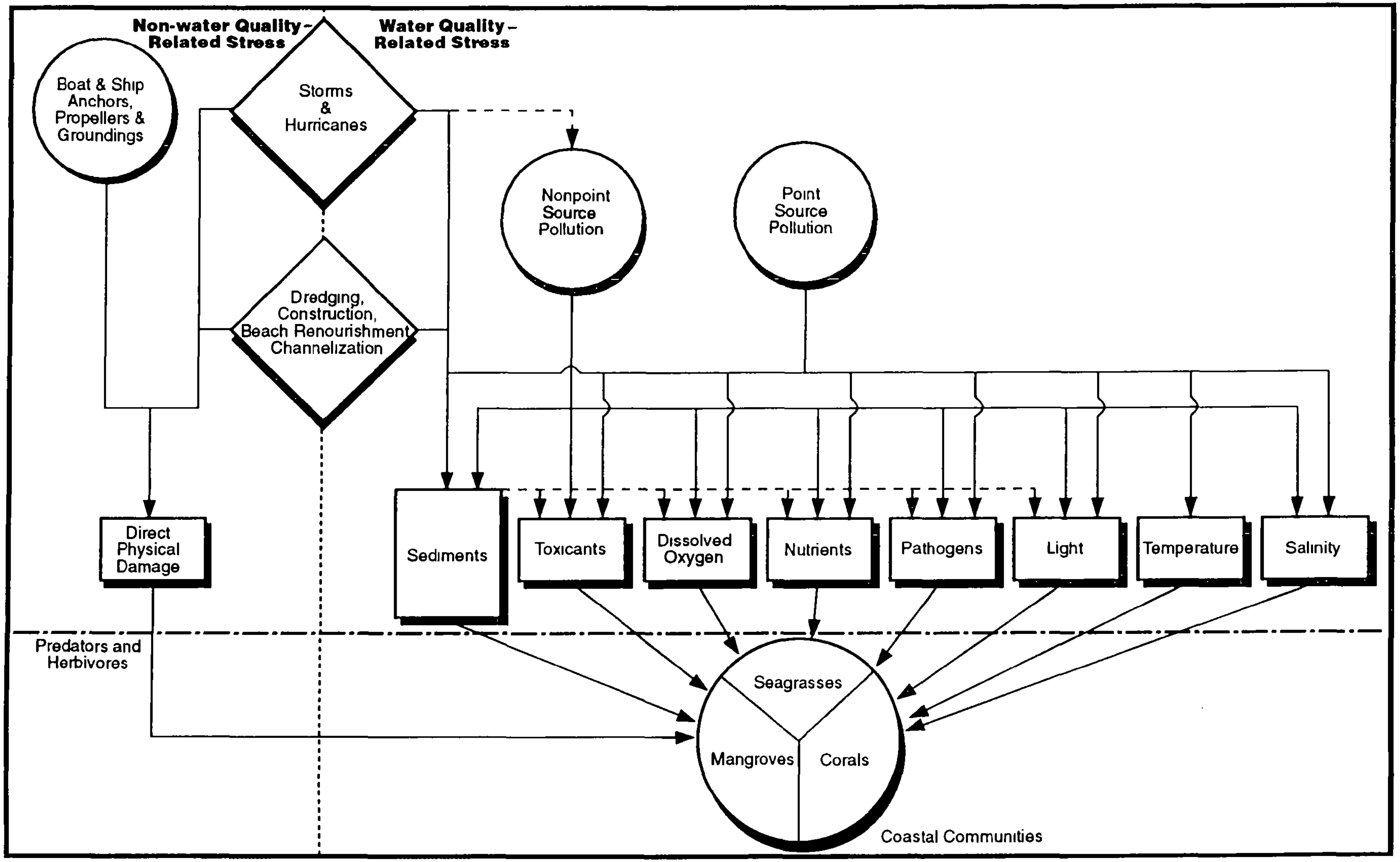


Figure 6-2. Conceptual model of types and sources of stress to coral, mangrove, and seagrass communities.

describe the types of stress and interactions that occur between these stresses. The types of stress resulting in changes in water quality include sediments, nutrients, toxicants, temperature variations, oxygen depletion, light reduction, salinity changes, and pathogens. The sources of these types of stress include point source discharges of treated wastewater from POTWs or other facilities; nonpoint discharges of stormwater from agricultural, industrial, residential, and urban areas; discharges from marinas and other nearshore industries; accidental releases of chemicals (e.g., oil spills); and sediments produced during construction and dredging activities.

As mentioned above, sediment loads in the coastal waters off southeast Florida may increase through land-clearing activities associated with agriculture and construction, dredging operations, beach renourishment activities, and the discharge of sediment-rich effluents from industrial and municipal point sources. Storms and hurricanes are a significant nonhuman source of stress, producing rainfall, which contributes to nonpoint discharges of sediments. The sediments washed off during these storms also transport toxicants, oxygen-demanding substances, nutrients, and pathogens into the marine environment. Increased turbulence due to storms and hurricanes can also resuspend and transport bottom sediments, which can then be redeposited, resulting in additional stress. Natural sediment transport along coastlines may also be a source of sediment.

Nutrients are contributed by both point and nonpoint sources, particularly nonpoint pollution from agricultural watersheds (Dierberg, 1991) and point source pollution from municipal wastewater treatment plants. Nutrient input (especially that of phosphorus) is closely tied to discharges of sediments. Hence, storms and hurricanes may have a significant influence on nutrient input. Approximately 33 percent of the annual total nitrogen and 50 percent of the annual total phosphorus load to drainage canal waters that discharge to the Indian River Lagoon occurred during a 6-week period when three major storms occurred (Dierberg, 1991). Sediments from dredging and construction activity are an additional source of nutrients as a result of the association of nutrients with sediments. Another, often overlooked source of nutrients is in the form of nutrient-rich groundwater. Lewis (1985, 1987) investigated the contribution of nutrients from groundwater and concluded that groundwater was the chief source of nutrients for enhanced coastal production along the west coast of Barbados.

Sources of toxic pollutants also include point and nonpoint sources. Nonpoint pollution includes pesticides from agricultural activity, metals from mining activities, and pollutants from urban and residential runoff. Urban and residential runoff can be a source of oils, metals, household wastes, and pesticides from landscaping activities. Point sources include metals and chemical toxicants discharged from domestic and industrial sources. An additional source of toxic pollutants to these coastal communities includes accidental spills of toxic chemicals, particularly oil and fuel.

Coastal communities off southeast Florida may also be affected by changes in water temperature. Seasonal sources of thermal stress include winter cooling of nearshore surface waters and upwelling of deep, cooler water from offshore. Air temperatures may fluctuate dramatically on a seasonal basis, bringing occasional periods of frost during winter and prolonged heat waves in summer. Warm water stress is generally the result of summer warming of shallow water during warm, calm periods although larger scale ocean-warming events (e.g., El Niño) may also bring unseasonably warm water into the vicinity of these communities. Human-induced temperature stress is supplied mainly in the form of heated effluents from power and water desalination plants, but these are not a concern in this study area. Johannes and Betzer (1975) discussed the effect of changing temperature on the susceptibility of marine communities to stress. Although a temperature increase can induce an increase in production, respiration increases as well. If respiration increases more rapidly than production, the P/R ratio will approach 1.0. Thus, although tropical marine organisms have high rates of production, much of this production is respired. Therefore, additional stress (e.g., that due to sublethal chemical toxicity) that increases respiration can easily result in little or no net productivity, leaving little metabolic reserve for reproduction and maintenance. Higher rates of metabolism, coupled with the generally greater solubility of pollutants in warmer water, result in rapid uptake of toxicants and hence greater sensitivity of warm water organisms to pollutants. However, greater uptake rates may be balanced by equally rapid toxicant elimination rates.

Temperature increase in warm waters also results in lower oxygen concentrations at saturation levels, placing warm water organisms closer to their oxygen limits (Johannes and Betzer, 1975). Therefore, slight increases in temperature and minor changes in the amounts of oxygen-demanding substances discharged to these environments can push oxygen levels to critical levels. Community respiration may exceed available oxygen reserves at night, resulting in the most sensitive conditions, especially for coral communities in poorly flushed areas such as lagoons. Human sources of such oxygen-demanding substances include direct discharges of municipal, food processing, and paper-making process wastewater. Oxygen-demanding substances from dredging operations may be an additional source of stress, as well as nonpoint pollution runoff. Oil spilled in the marine environment also exerts an oxygen demand on the surrounding water as do a variety of organic chemicals that may be accidentally spilled into coastal areas. Indirectly, increased eutrophication due to human-caused increases in nutrient loading to coastal waters may result in periods of oxygen depletion as the result of the bacterial degradation of phytoplankton biomass following algal blooms. These sources of oxygen-demanding substances are in addition to the normal respiration (community oxygen consumption) that occurs in these communities. Oxygen stress is most critical at night due to the absence of oxygen production by the photosynthetic organisms of these communities.

Photosynthesis is dependent on adequate levels and appropriate wavelengths of light. Light quality may be altered by the discharge of solids from both point and nonpoint sources of pollution. Major storms have a significant influence on sediment input to coastal marine systems, resulting in elevated turbidity and reduction in light intensity. Human activities such as land clearing, urbanization, coastal construction activities, and dredging all contribute to elevated suspended sediment levels and turbidity in coastal waters. Eutrophication of coastal waters also results in increased turbidity and light reduction due to increased phytoplankton biomass. Stratospheric ozone depletion, on the other hand, may be a source of increased ultraviolet light intensity. This stress is restricted to shallow, clear water areas under calm conditions, which allows maximum penetration of light.

Salinity changes in the coastal area off southeast Florida most likely are influenced by freshwater runoff (and hence low salinity stress) due to rainfall. The quantity of freshwater runoff discharged is increased through the removal of vegetation that retains runoff; paving of large areas, which prevents infiltration of runoff; and the channelization of runoff, which routes stormwater more quickly to coastal areas. Road construction, channelization, drainage alteration, and dredging activities can also alter the balance of salinity in nearshore mangrove, seagrass, and coral communities. Although the effect of storms and hurricanes on runoff is short-lived, changes in the natural drainage system caused by the storm can cause long-term changes in water exchange, and hence salinity of these communities. These physical changes may result in diversion of fresh water (e.g., drainage construction) from these communities and may result in increased salinity in one location and possibly reduced salinity in the area that receives the diverted water. Alterations of coastal barriers (channel construction) or channels of tidal exchange (causeways without adequate culverts) may result in increased or decreased exchange of seawater, resulting in increased or decreased salinity of the affected areas. Salinity reductions may also occur in the vicinity of the POTW outfalls. At the Hollywood facility, minimum salinity values recorded were 27.1 and 24.8 ppt.

Naturally occurring biological factors such as pathogens, parasites, predators, and herbivores are also important modifiers of species composition, biomass, areal coverage, and productivity of these coral hard grounds, seagrass beds, and mangroves. While periodic outbreaks of disease and predation can have widespread and devastating impacts on the structure and function of these ecosystems, the factors influencing such outbreaks, the reasons why they occur, and how they interact with complex environmental factors are poorly understood. Some specific examples for each community type will be discussed further below.

Two additional types of stress that are not directly related to water quality include direct physical damage and physical alterations of the environment. Physical damage may be human induced (e.g., direct impact during construction and/or dredging activities), or it may be due to non-human causes (e.g., tropical storms and hurricanes). Storms and hurricanes have a very significant impact on these communities directly through physical impacts and indirectly through physical processes associated with heavy rainfall runoff. The sediments washed off during these storms transport nonpoint source pollutants such as toxics, oxygen-demanding substances, nutrients, and pathogens. Finally, changes in the hydric regime (i.e., frequency and extent of exchange of fresh and/or saline water) of coastal areas due to physical disturbances of the environment are more subtle. Sources of change in the hydric regime are both human and nonhuman. Human interference in drainage patterns may cause changes in coastal sediments, nutrients, and salinity. Storms and hurricanes can produce changes in the original drainage of the mangrove forest, resulting in either increased or decreased seawater exchange. The effect of these changes will depend on the input rate of fresh water, evaporation, and the magnitude of the increase or reduction in the tidal exchange of seawater. Longshore transport of sand may also gradually enclose mangrove-lined bays and prevent tidal exchange.

6.4 Potential Effects of Water Quality Alterations on Coastal Communities

This section presents a discussion of the potential impacts of each type of stress on the coral, seagrass, and mangrove ecosystems along the southeast Florida coast due to the presence of point and nonpoint sources of pollution. Examples of important processes and effects are also illustrated graphically. The impacts of sewage pollution can be placed into three broad categories: nutrient enrichment, sedimentation, and toxicity. Degradation of the marine environment may result in the following changes in the local biological communities (USEPA, 1982):

- Modification of the structure of benthic communities as a result of the accumulation of discharged solids on the seabed;
- Stimulation of phytoplankton and/or macroalgal growth due to nutrient enrichment;
- Reduction of phytoplankton and/or macroalgal growth due to turbidity increases;
- Reduction of dissolved oxygen due to phytoplankton blooms and subsequent die-offs, causing mass mortalities of fish and invertebrates;
- Bioaccumulation of toxic pollutants due to direct contact or ingestion of sediment, direct uptake from the effluent, or ingestion of contaminated organisms; and
- Induction of diseases from contact with sediments, ingestion of contaminated organisms, or exposure to the effluent.

Increased levels of organic matter in sewage effluent and the subsequent oxidation of that organic matter may depress dissolved oxygen concentrations in areas that are already existing at the upper threshold of tolerance (Bathen, 1968; Kinsey, 1973; Johannes, 1975). The ambient dissolved oxygen concentration is most critical at night (Pastorok and Bilyard, 1985). Reef communities maintain a constant rate of respiration (Kinsey, 1973), and nighttime oxygen concentrations may fall to near zero. An increase in the biological oxygen demand resulting from the oxidation of organic matter by bacteria may depress oxygen concentrations below critical levels, significantly stressing the reef community. Hydrogen sulfide generation and toxic by-products of pesticides, herbicides, chlorine, and heavy metals can produce deleterious effects (Grigg and Dollar, 1990), as can increased nutrient (e.g., nitrates and phosphates) loadings. Simkiss (1969) found that inorganic orthophosphates, pyrophosphates, and organic phosphates (e.g., glycero-phosphate or adenosine triphosphate) strongly inhibit the calcification process by acting as crystal poisons. The impact on the local biota will be indicative of the pollutants to which the biota are exposed, their concentrations, and the period of exposure.

6.4.1 Sediments

Excessive amounts of sediments and sedimentation result in the death of reef-forming coral organisms and degradation of the reef framework. Loss of the reef framework and its associated structural complexity will result in habitat loss and reduction of coral reef fish (Rogers, 1990). Heavy sedimentation has been associated with fewer coral species, less live coral, lower coral growth rates, decreased calcification rates, decreased net productivity, decreased reef accretion rates, and reduced coral recruitment (Rogers, 1990). Heavy sedimentation will also favor a shift from the coral benthic community to a benthic community dominated by benthic filter feeders and detritivores (Banner, 1974; Birkeland, 1977).

Sediments or particulate organic and inorganic matter discharged to coral reef environments may directly smother corals, result in increased metabolic costs to corals as a result of sediment removal by the coral organism, reduce the amount of suitable hard substrate on which coral larvae can settle and attach, or result in abrasion and scour of coral tissue during sediment resuspension and transport (Figure 6-3). Indirectly, suspended sediments increase turbidity, which in turn can decrease the amount of light available for coral production associated with their symbiotic algae (zooxanthellae). Sediments also transport particulate-associated toxic pollutants, nutrients, oxygen-demanding organic matter, and pathogens. These types of stress will be discussed further below.

Although it is generally recognized that sedimentation is detrimental to corals, the types and degree of sedimentation where impairment of coral reef growth, reproduction, and recruitment occurs have not been determined (Hubbard, 1987). The effects due to sedimentation depend on a number of factors, which include the type of sediment (grain size distribution, carbonate content, organic content, toxic pollutant levels); the amount of sediment (i.e., sedimentation rate); and the duration (e.g., chronic vs. acute effects) and timing of coral exposure (e.g., night vs. day, reproductive or recruitment period). However, some generalizations regarding the impact of various sedimentation levels have been made. Rogers (1990) indicated that reefs not subjected to human-induced stress had average sedimentation rates less than or equal to 10 mg dry-wt/cm^2 per day. Tetra Tech (1983) estimated that sedimentation rates of 1 to 10, 10 to 50, and greater than 50 mg dry-wt/cm^2 per day would result in slight to moderate impacts, moderate to severe impacts, and severe to catastrophic impacts, respectively. Hudson (1981) observed decreased growth rates of *Montastrea annularis* colonies in the Key Largo National Marine Sanctuary that appeared to coincide with the increased dredge and fill operations in the Florida Keys area from 1953 to 1968. Eutrophication along the west coast of Barbados resulted in decreased settlement rates for coral planulae, thus altering the composition of the reef communities (Tomascik, 1991). These reduced recruitment rates were speculated to be the result of either increased suspended particulate matter or toxic effects of the effluents.

Generally, seagrass beds occur in sedimentary environments ranging from clean sand to fine silty sediments (McRoy, 1983). Additionally, seagrasses act as stabilizers of sediment and may actually enhance sedimentation within the seagrass bed (Zieman, 1975, 1982). Excessive sediments, however, can impact the seagrasses through direct sedimentation and smothering,

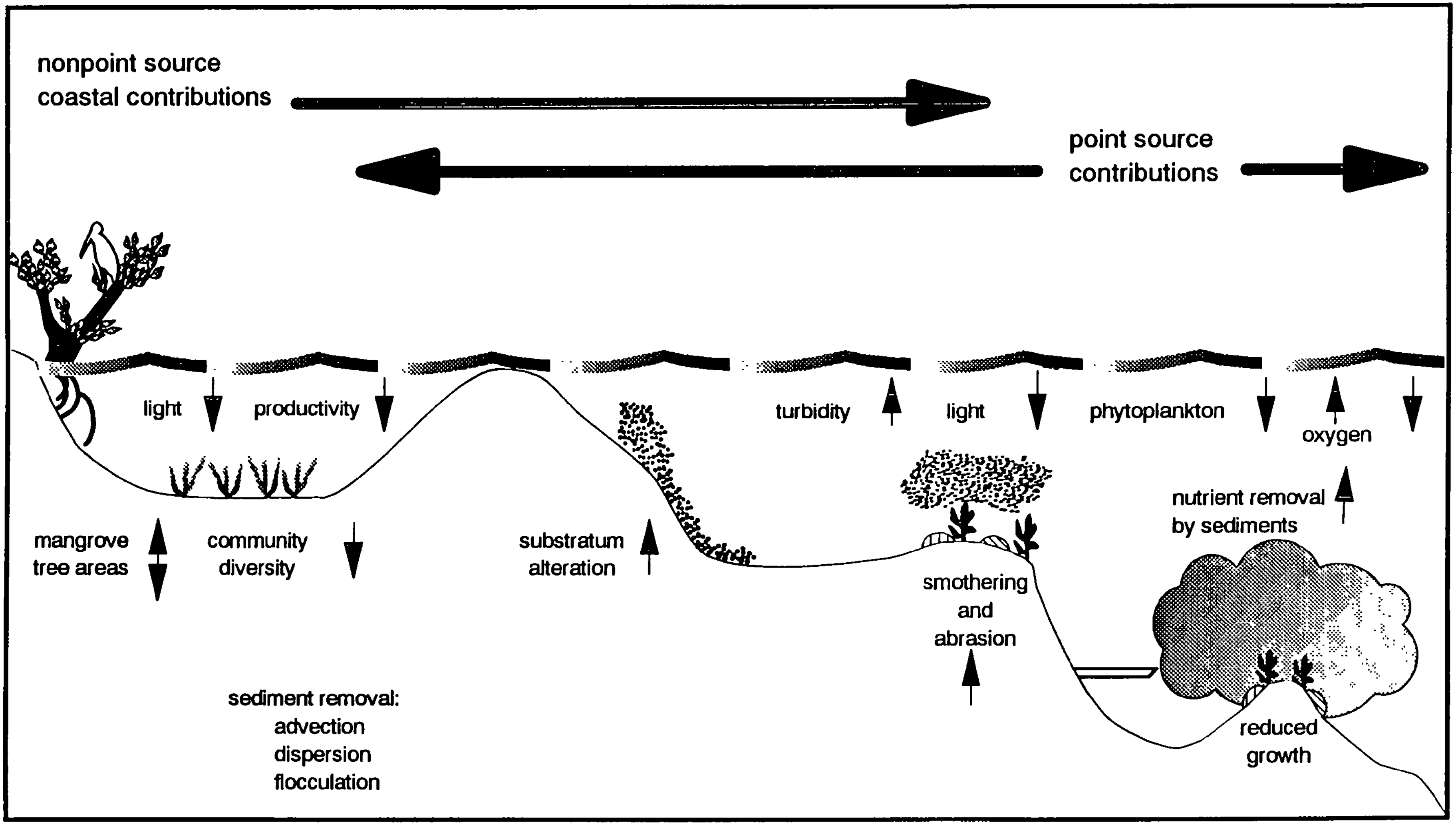


Figure 6-3. Conceptual model of processes and effects relating to sedimentation in southeast Florida coastal communities.

depletion of oxygen due to oxygen-demanding substances in the sediment, and increased turbidity resulting in decreased light penetration and subsequent impairment of seagrass photosynthetic production. Sediments not only affect the seagrasses themselves, but they also influence the species composition and productivity of the associated seagrass community, which includes benthic and epiphytic algae, crustaceans, and fish.

Mangroves generally thrive in areas of high sediment loading (e.g., river deltas), but catastrophic sediment deposition that smothers the mangrove aerial root system can cause tree mortality (Odum and Johannes, 1975; Cintrón and Schaeffer-Novelli, 1983). Mangrove seedlings are also sensitive (Odum and Johannes, 1975). Fresh deposits of sand associated with sand extraction and storm waves killed mangroves when the sand deposited was 30 cm deep, and partial mortality was observed where sediment depth was 20 to 30 cm.

The loss of mangroves due to acute sediment stress would typically be accompanied by losses of the biotic community associated with the mangrove (e.g., crustaceans, reptiles, amphibians, and birds). It is also possible that chronic sublethal sediment stress would not cause mangrove tree mortality, but that biotic communities associated with the mangrove root system would be affected. On the other hand, increased sediment input, if not catastrophic, could allow for the gradual expansion of the areal extent of the mangrove forest (Odum and Johannes, 1975).

Sediment diversion should also be considered because of the special requirements of mangrove forests. Sediments may be diverted from these communities through natural changes in drainage patterns or through channelization and ditch construction. The prevention of the natural filtration of stormwater runoff by mangroves allows for increased discharge of sediments to offshore coral and seagrass communities.

6.4.2 Nutrients

The effect of increased nutrient input to coral reef areas is not well understood, and conflicting evidence has been presented in the literature. The association of nutrients with increased sediment input often complicates interpretation of field data. Nutrients may directly affect coral skeletal growth by inhibiting skeletal calcification (phosphate) (Simkiss, 1964) and may indirectly affect the coral community through the enhancement of attached algal growth, which may overgrow living corals or interfere with coral larval recruitment (Birkeland, 1977). Based on evidence from field studies in the Great Barrier Reef, Kinsey and Davies (1979) suggested that phosphate (PO_4) levels greater than 62 μg PO_4/L could suppress reef calcification by more than 50 percent. Kinsey and Davies (1979) determined that nitrogen enrichment had no effect on calcification activity and that calcification may actually be enhanced by elevated levels of ammonium. However, Stambler et al. (1991) reported no significant differences in the growth rate of *Pocillopora damicornis* (collected from Kaneohe Bay, HI) treated with up to 10 times the ambient phosphate phosphorus levels (61.9 μg PO_4/L). Furthermore, Stambler et al. (1991) found reduced skeletal growth rates due to enrichment with ammonium (210 μg NH_4/L) and with enrichment with ammonium and phosphate together. This reduction in growth was attributed to a reduction in translocation of zooxanthellae photosynthetic products due to the rapid growth of the nitrogen-limited zooxanthellae and/or due to competition with the coral animal for the carbon dioxide required for calcification.

The more likely effect of nutrient input is the enhancement of benthic algal growth and subsequent competition with corals for the limited space within the community. Johannes et al. (1983) suggested that nutrient supply in high-latitude reefs played a significant role (along with temperature) in controlling coral community structure, explaining the dominance of benthic algae in these habitat. Littler and Littler (1985) went further to include the role of herbivores in the control of algal biomass in coral communities. They concluded that nutrient levels determine the potential size of algal standing stocks, but that herbivores can maintain macroalgal biomass well below the limits set by nutrient supply. The growth of other suspension feeding and bioeroding animals stimulated by nutrient inputs, however, may lead to increased erosion of the carbonate substratum (Rose and Risk, 1985; Hallock and Schlager, 1986). Additionally, nutrient input could indirectly affect the coral community if a significant fraction of the input is converted to phytoplankton production. Increased phytoplankton biomass would reduce light levels and shade coral and seagrass communities, with subsequent detrimental effects (Figure 6-4).

Seagrasses growing with adequate light appear to be nutrient-limited (Williams, 1987, 1990), although light could become limiting with increased turbidity (Vicente and Rivera, 1982) and/or with increased water depth (Williams, 1988a). Eutrophication, or the increased rate of input of nutrients, could enhance the production of phytoplankton and epiphytic algae, which could shade the seagrasses and cause a decrease in productivity of seagrasses. This could lead to eventual replacement of the seagrass community with algae (i.e., phytoplankton and/or benthic algae and periphyton) (Zieman, 1975, 1982).

Generally, the growth and biomass of mangrove forests are considered to be limited by nutrient input, with the mangrove ecosystem acting as a sink for nutrients, including nitrogen and phosphorus (Odum and Johannes, 1975; Odum et al., 1982). Therefore, eutrophication of mangroves has not generally been considered a problem. On the contrary, serious attention has been focused on using mangrove forests for domestic wastewater treatment (Odum and Johannes, 1975, Clough et al., 1983). Since mangrove productivity and biomass are generally limited by the nutrient supply, changes in water drainage patterns may have a negative effect on mangroves when they depend on drainage water for essential nutrients. Nutrient diversion should also be considered as a stress, particularly in the case of mangrove forests. Nutrient diversion occurs through sediment diversion techniques, which create drainage systems that prevent the natural filtration of upland runoff by mangrove forests.

6.4.3 Toxicants

Toxicants, such as chlorine, metals, pesticides, petroleum hydrocarbons, or other organic pollutants, may directly affect various life stages of corals (gametes, planulae, or larval settlement) or the various life stages of animals and plants that make up the coral community. Coral colonies themselves may be fairly resistant to toxic pollutants (Marszalek, 1987), but certain other coral life stages may be sensitive. Corals exhibit a variety of reproductive

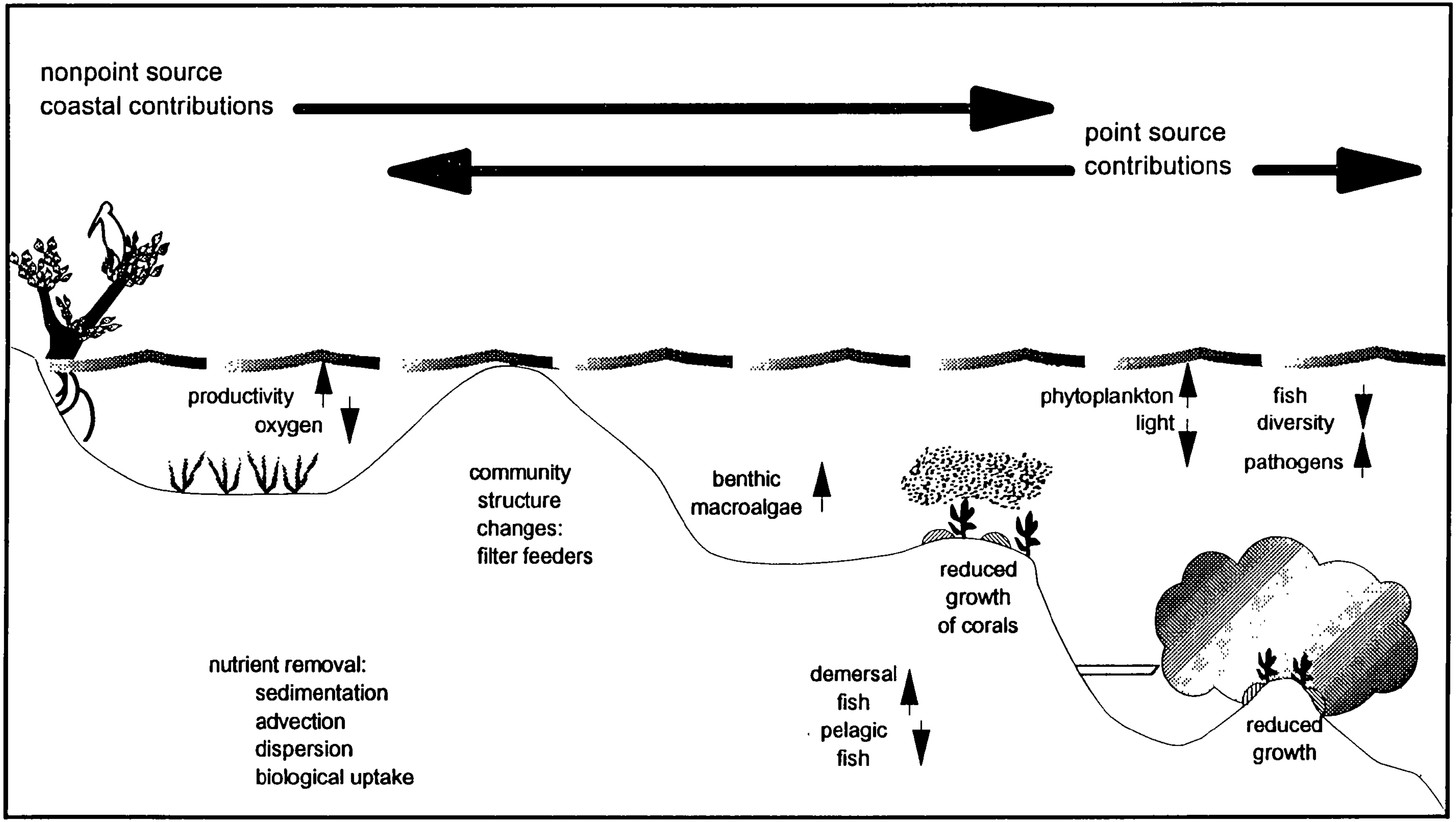


Figure 6-4. Conceptual model of processes and effects relating to nutrient inputs in southeast Florida coastal communities.

strategies, including spawning of gametes and external fertilization, timing of spawning, and spawning synchrony among species (Richmond and Hunter, 1990). Therefore, the timing of pollutant input may play an important role in toxic effects on coral life stages. Although several toxicity studies have been performed on various life stages of corals, these studies have generally been performed on tropical Pacific species (Evans, 1977; Acevedo, 1991; Goh, 1991; Te, 1991). How the toxic pollutants interact with other stressors (e.g., sediments, nutrients, and abnormal temperature variations) has not been adequately studied.

Coral mortality and coral community alterations due to oil pollution have been noted in several areas of the Caribbean (e.g., Bak, 1987; Jackson et al., 1989). Loya and Rinkevich (1980) reviewed research on the effects of petroleum hydrocarbons on corals and concluded that oil pollution in direct contact with corals can impair coral growth, reproductive systems, and coral larvae and can cause mortality of corals. Peters et al. (1981) demonstrated that a shallow-water Caribbean coral, *Manicina areolata*, could bioaccumulate petroleum hydrocarbons (water-accommodated No. 2 fuel oil). The highest mean hydrocarbon concentration over the 3-month-long flow-through test, 150 µg/L, did not induce coral mortality, but sublethal effects were noted, primarily impaired development of reproductive tissues, loss of zooxanthellae, and atrophy of mucous secretory cells and muscle bundles. Te (1991) investigated the effect of two petroleum products (benzene and an oil and gasoline mixture) on *Pocillopora damicornis* larvae survival and settlement. Mortality was minimal in treatments of up to 100 mg/L, but treatments did have an effect on corallite formation.

McCloskey and Chesher (1971) did not observe any mortality of specimens of *Montastrea annularis, Acropora cervicornis*, or *Madracis mirabilis* to exposure of 10-, 100-, and 1,000-µg/L mixtures of p,p'-DDT, dieldrin, and Arochlor 1254, but they did note an increase in respiration (R) and a decrease in photosynthesis (P) in all three species, which occasionally lowered the P/R ratio to below 1.0. Exposure of *P. damicornis* larvae to the pesticides carbaryl and 1-naphthol of 10 mg/L resulted in greater than 50 percent mortality (Acevedo, 1991). An investigation by Solbakken et al. (1985) demonstrated that several Bermudian corals could bioaccumulate naphthalene, phenanthrene, a PCB congener, and octachlorostyrene. Elimination of naphthalene was rapid, but the PCB congener was depurated very slowly with tissue concentrations of the contaminant still well above detectable levels up to a year following exposure. Glynn et al. (1989) detected high levels of organochlorine pesticides and heavy metals in corals and octocorals collected from Biscayne National Park off southeast Florida, similar to those levels used in toxicity tests that had led to bleaching and mortality of reef-building corals in the laboratory (Glynn et al., 1984).

Evans (1977) exposed the Pacific corals *P. damicornis* and *Montipora verrucosa* to solutions of copper (Cu) sulfate in flow-through systems. Exposures of 100 µg Cu/L resulted in 100 percent mortality after 24 hours. After 48 hours of exposure to 10 µg Cu/L, corals that had been observed to be stressed were dead by the sixth day of exposure (Evans, 1977). Exposure of *P. damicornis* planula larvae to nickel (9 mg Ni/L) over a 12-hour period caused 50 percent mortality almost 40 hours after discontinuance of exposure (Goh, 1991). Settlement of the larvae was more sensitive, with effects observed with exposure to 1 mg Ni/L for 12 to 96 hours (Goh, 1991).

Chlorine is typically used in disinfection of municipal wastewater prior to discharge. Johannes (1975) reviewed the effect of free residual chlorine on tropical marine organisms. Toxic effects were observed in marine fish at levels as low as 24 μg/L, while coral planulae of three Hawaiian corals tolerated exposure to 490 μg/L for up to 7 hours. Marszalek (1987) concluded that corals were relatively resistant to chlorinated effluents compared to fish, based on tests performed at an experimental sewage treatment plant in Miami, Florida.

Hydrogen sulfide has also been implicated as a toxicant to corals, partially explaining coral decline in Kaneohe Bay (Banner, 1974). Hydrogen sulfide production is the result of reduction of organic matter by bacteria using sulfate as the electron acceptor under anaerobic conditions. Therefore, hydrogen sulfide production is linked to discharges of oxygen-demanding substances and oxygen depletion in bottom sediments.

Although some toxicity testing has been performed, these tests have focused on very few coral species and life stages and have generally exposed the organisms to concentrations greater than would typically occur in even a heavily polluted environment. Exposure has also generally been short-term, and the sublethal effects of long-term exposure have been poorly characterized. Exposure to toxicant-contaminated sediments has also not been considered. These sediments may be discharged to coral areas by point and nonpoint sources. Through resuspension of contaminated sediments deposited in coral areas, the coral community may be continually exposed to sediment toxicants.

The effects of toxicants on seagrass ecosystems, with the exception of oil, have received little attention (Thorhaug, 1981). Oil damage to seagrasses that has been observed includes extensive damage to beds on the south shore of Puerto Rico following the discharge of approximately 10,000 tons of crude oil from a grounded vessel. Seagrass mortality and a change of benthic algal species composition to blue-green algal types was observed over a period of a few months (Díaz-Piferrer, 1964). Jackson et al. (1989) reported the mortality of intertidal seagrass beds and a reduction in biomass and species of animals of the seagrass community due to an extensive oil spill in the Caribbean along the coast of Panama. However, Jackson et al. (1989) observed that subtidal seagrasses survived, although sublethal effects such as browning of grass blades and fouling by epiphytic algae occurred.

Thorhaug (1988) reported on the effects of mixtures of oil with oil dispersants. She concluded that the toxicity response of seagrasses varied with the dispersant used, but that seagrass species responded predictably to exposure. *Thalassia* (turtle grass) was found to be more tolerant than *Halodule* (shoal grass), which was more tolerant than *Syringodium* (manatee grass) (Thorhaug, 1988).

The toxicity of organic compounds and metals to mangroves is dependent on (1) mass loading of the toxicant, (2) the duration of input, (3) the susceptibility of the organism, and (4) the physical-chemical factors in the receiving environment, which include temperature, salinity, and flushing rate (Lugo et al., 1981). Since mangroves occur in a variety of environments (e.g., basins, riverine estuaries, and shoreline fringes), their susceptibility to toxicant stress depends heavily on the above-mentioned factors. Therefore, basin mangroves may be more susceptible to inland sources of pollution (e.g., agricultural herbicides), and fringe mangroves may be more susceptible to seaward sources of pollution (e.g., oil spills) (Cintrón et al., 1978; Lugo et al., 1981).

Mangroves have been noted to be very sensitive to herbicides (Odum and Johannes, 1975; Odum et al., 1982; Culic, 1984). This conclusion is based on observations of limited mangrove recovery following large-scale (100,000 ha) defoliation of mangroves with herbicides in Vietnam. Culic (1984) reported that effects due to soil and foliage applications of 2,4-D applied to *Rizophora stylosa* were exhibited at dosages of 0.0125 and 0.3125 kg/ha, respectively. Walsh et al. (1973) reported that applications of 2,4-D of 4.4 kg/ha were lethal to *R. mangle*; however, levels of 0.44 kg/ha had no permanent effect.

Oil pollution has caused mangrove mortality throughout the world (reviewed by Lewis, 1983 and Hoi-Chaw, 1984). The impacts of oil spills on macroinvertebrate populations of the mangrove have also been noted (Díaz-Piferrer, 1964; Lewis, 1983; Jackson et al., 1989). Oil dispersants may also be toxic, and the mixture of dispersed oil may be more harmful than oil alone. While Hoi-Chaw et al. (1984) showed that dispersed oil was less toxic to mangrove trees and seedlings than oil alone, the reverse was true for mangrove invertebrates.

Effluent from municipal treatment facilities has the potential to adversely impact recreational and commercial fisheries through the bioaccumulation of toxic organics (Spies, 1984) or the induction of diseases that lower or eliminate the marketability of the catch (USEPA, 1982). Young (1964) produced perhaps the earliest report of the effects of sewer discharges on fishes that were collected near sewer outfalls on the California coast. He found changes in the consistency of the flesh, weight reductions, external lesions, exophthalmia (protruding eyes), and papillomas (tumors). Several other studies have implicated POTW discharges in the proliferation of diseases such as exophthalmia in spotfin croaker, *Roncador stearnsil*, and white seabass, *Cynoscion nobilis*; lip papilloma in white croakers, *Genyonemus lineatus*; and fin erosion in fishes of the New York Bight (Mahoney et al., 1973) and in the Dover sole, *Microstomus pacificus* (Mearns and Sherwood, 1974; McDermott-Ehrlich et al., 1977). Marine organisms collected near sewage outfalls have been known to bioaccumulate chlorinated hydrocarbons and trace metals in their tissues. These include the Dover sole, *Microstomus pacificus*; the rock crab, *Cancer anthonyi*; the mussel, *Mytilus californianus*; and the rock scallop, *Hinnites multirugosus* (Young et al., 1976, 1978; McDermott et al., 1976; McDermott-Ehrlich et al., 1978).

Fin erosion, ulcerations, papillomas, gill hyperplasia, and lymphocystis are characteristic diseases of fishes living in degraded habitats (Sindermann, 1990). Large populations of the heterotrophic bacteria *Vibrio*, *Pseudomonas*, and *Aeromonas* may occur as a result of heavy organic loadings from domestic sewage or agricultural runoff into estuarine and coastal waters (Sindermann, 1990). These bacteria can produce integumentary or penetrating ulcers on fish that are stressed from low dissolved oxygen levels, abnormal temperatures, or the presence of other pollutants. Environmental pollutants may reduce the ability to resist infection. It is believed that mucous secretion, the principal external defense, is suppressed, leaving the fish vulnerable to bacterial infection and fin erosion (Hodgins et al., 1977). In fact, the occurrence of fin erosion has been correlated to high coliform counts (Mahoney et al., 1973) and high heavy metal concentrations (Carmody et al., 1973) in sediments. In a study of fishes collected near a municipal sewer outfall near Los Angeles from 1971 to 1982, the decline in the prevalence of fin erosion was correlated to a decline in surficial sediment contaminants (Cross, 1985).

Several studies have indicated that the waters of Biscayne Bay are having adverse effects on the fish populations. These effects include the disruption of normal scale patterns, scale reversals (Sindermann, 1990), and fibroma-like tumors in the mullet, *Mugil cephalus* (Sindermann, 1976). Fishes with severe infestations and gill lesions were taken from the heavily-polluted canal waters that enter Biscayne Bay (Skinner, 1982). Among the pollutants found in the canal waters emptying into Biscayne Bay are organic pesticides such as diazinon, silvex, and parathion; heavy metals such as mercury, lead, and zinc; and ammonia. Figure 6-5 summarizes some of the processes and effects that need to be considered when coastal ecosystems are exposed to toxic substances.

6.4.4 Oxygen

Oxygen is necessary for the maintenance of marine life. Low oxygen concentrations or the lack of oxygen can stress coral communities. During the day, phytoplankton and attached algae consume carbon dioxide (CO_2) and produce oxygen (O_2) during photosynthesis. Animals (including corals) consume phytoplankton and other organisms and consume oxygen in order to oxidize the ingested organic matter and convert it into energy for maintenance, growth, and reproduction. Aerobic bacteria also consume oxygen in order to decompose the organic matter shed by the coral community. Thus, during the day the ambient oxygen level of the coral community is balanced by the production of oxygen through photosynthesis, the consumption of oxygen by animals and aerobic bacterial degradation, and the input of oxygen through diffusion from the atmosphere and advection from offshore waters. At night, photosynthetic oxygen production no longer occurs, due to the lack of sunlight, and respiration (oxygen consumption) by the coral community can reduce the ambient oxygen concentration if the community is not well supplied with fresh oxygenated water (e.g., as in a backreef lagoon). Also, because the respiration rate is typically higher at higher temperatures and the ambient oxygen concentration is lower at higher temperatures, coral communities in poorly flushed areas may be stressed by lack of adequate oxygen (Johannes and Betzer, 1975). Therefore, the input of oxygen-demanding organic matter to poorly flushed coral reefs can have a detrimental effect on coral communities.

Low oxygen levels in seagrass beds can be detrimental to seagrasses and their associated plants and animals. However, since seagrass productivity (i.e., oxygen production due to photosynthesis) is high, oxygen levels are generally adequate, although low oxygen levels have been observed in warm, calm waters during the night, when community respiration and additional oxygen-demanding matter may deplete oxygen reserves (Zieman, 1982). Anaerobic conditions in the sediments may actually be beneficial to seagrasses (Zieman, 1982); however, extremely low oxygen concentrations due to excessive loading of oxygen-demanding organic matter will have a detrimental effect on the seagrass community.

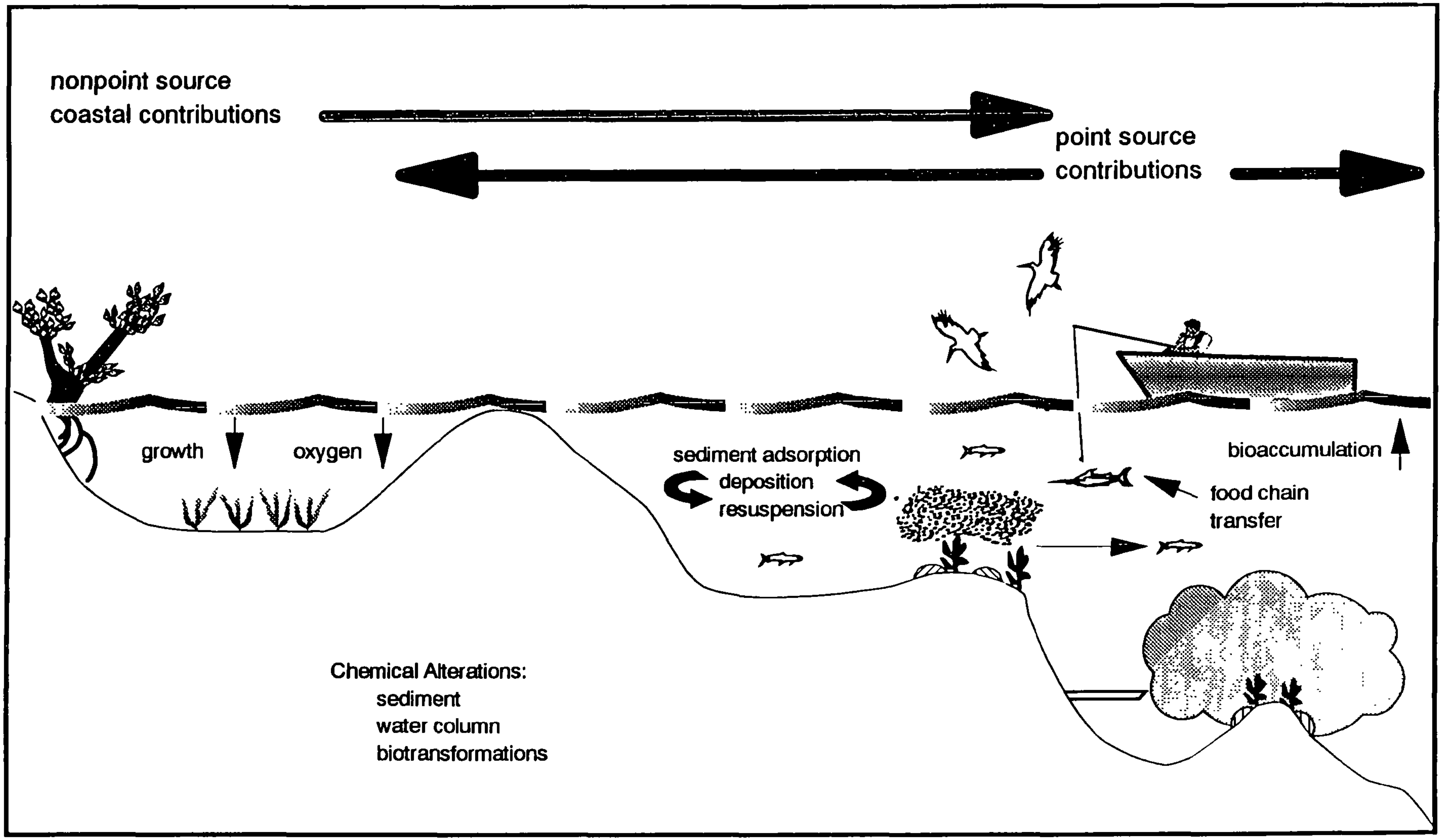


Figure 6-5. Conceptual model of processes and effects relating to toxic inputs in southeast Florida coastal communities.

The waters of mangrove forests are typified by low oxygen concentrations or even anaerobic conditions. Mangrove trees have adapted to low-oxygen environments by developing aboveground root systems to bring oxygen to the roots embedded in the anaerobic sediments (Odum et al., 1982). However, the biotic communities associated with the mangrove root system can change markedly as a result of changes in oxygen concentrations within the forest.

6.4.5 Light

Corals that possess symbiotic algae (zooxanthellae) respond to both the intensity and spectral quality of light. Goenaga et al. (1989) and Williams and Bunkley-Williams (1990) suggested that unprecedented coral bleaching events that occurred in the Caribbean in 1987 may have been due in part to increases in ultraviolet radiation related to stratospheric ozone depletion. However, warmer-than-normal seawater temperatures were also suggested as an additional factor. Because of the light dependence of zooxanthellate corals, these corals are restricted to areas above the lower limit of the photic zone. The depth of the photic zone depends on the reflectance, scattering, and absorption of the water column. Suspended sediments and phytoplankton are two sources of turbidity that can reduce the depth of the photic zone. Although corals and their zooxanthellae may be photoadapted to existing light regimes (Dustan, 1979), reduction in light due to turbidity lowers the coral growth rate and complete shading can cause bleaching and death (Rogers, 1979).

Coral species show varying susceptibility to light reduction (Rogers, 1979). Susceptibility appears to be related to the ability of the coral polyps to feed on zooplankton and particulate matter during periods of low light intensity (Rogers, 1979; Peters and Pilson, 1985). Rogers (1979) hypothesized that *Acropora cervicornis* was most susceptible to shading because of its relatively smaller polyp size (i.e., mouth size), which restricted its ability to obtain external sources of nutrition. However, *A. cervicornis* was a relatively more efficient sediment remover in sediment experiments (Rogers, 1979). In environments where turbidity is associated with a high sedimentation rate, the metabolic cost of sediment removal will cause an increase in respiration that will be compounded during the night when no light is available for photosynthetic production (Abdel-Salam et al., 1988).

Light penetration may limit seagrass growth when adequate nutrients are available in shallow areas or in turbid or deep-water areas (Williams, 1987, 1990). The maximum depth of seagrass growth in an area may be an indication of the depth of penetration of light, but other factors, such as herbivore grazing pressure, may be involved (Vicente and Rivera, 1982). The stress due to changes in light levels in mangrove areas is not considered to be important since the mangrove tree is an emergent plant and is not limited by light levels in the water column. In general, light levels in mangrove forests are low due to elevated turbidity and color from humic substances (Odum et al., 1982). However, mangrove community organisms that inhabit the mangrove root system may be altered as a result of changes in ambient light levels (e.g., attached algae).

6.4.6 Salinity

Both low and high salinity may stress coral communities. Low salinity may occur as the result of freshwater runoff following storms. Brief episodes of low salinity due to stormwater runoff have been reported to cause loss of zooxanthellae and coral mortality (e.g., Goreau, 1964). Since stormwater runoff is simultaneously the source of sediment and fresh water, the effects of salinity and sediment stress (as well as light reduction) are difficult to separate (Johannes, 1975). Elevated salinity has been observed to cause coral mortality at levels only slightly above normal (Johannes, 1975). Several Hawaiian species tested could not tolerate salinities greater than 110 percent of normal for more than 2 weeks. Most species tested died within 24 hours when exposed to salinities 150 percent of normal (Edmondson, 1928, cited in Johannes, 1975).

The common tropical seagrasses tolerate large salinity fluctuations. Zieman (1982) reported acute salinity tolerances for *Thalassia* from 3.5 to 60 ppt, but sublethal effects (e.g., grass blade loss and reduced productivity) at 20 ppt. Zieman (1982) considered *Halodule* as the most tolerant to salinity variation, *Thalassia* as intermediate, and *Syringodium* and *Halophila* as least tolerant. Zieman (1975, 1982) also noted that seagrass beds are sensitive to temperature and salinity changes. When salinity is low and temperature is high, seagrass productivity and biomass may decline dramatically.

Mangrove trees are adapted to accommodate fluctuations in salinity through a variety of mechanisms of salt excretion and salt exclusion (Odum et al., 1982). Although some mangrove trees may be able to grow in fresh water, they are easily outcompeted by freshwater plants and trees (Odum and Johannes, 1975). Elevated salinities may restrict mangrove tree growth and reproduction. Red mangroves (*Rizophora*) may be limited to soil salinities below 60 to 65 ppt (Odum et al., 1982). White mangrove (*Laguncularia*) and black mangrove (*Avicennia*) appear to be more tolerant of salinity, with stressed stands observed in areas with soil salinities of 80 ppt (Odum et al., 1982). Salinity stress has been observed to affect mangrove canopy height, leaf area index, leaf litter fall, and tree mortality (Cintrón et al., 1978). The biotic community associated with the mangrove root system is likely to be sensitive to smaller salinity fluctuations.

6.4.7 Pathogens, Herbivores, and Predators

Bacteria (Mitchell and Chet, 1975; Peters et al., 1983; Hodgson, 1990) and filamentous blue-green algae (Antonius, 1981b) have been identified as pathogenic agents in corals. Antonius (1981a) and Peters (1984) have reviewed pathogen effects on corals. Mitchell and Chet (1975) observed that corals stressed by elevated concentrations of crude oil, copper sulfate, potassium phosphate, or dextrose produced copious amounts of mucous and died. However, with the addition of antibiotics, mortality was not observed, implicating bacteria as the agents of mortality. Hodgson (1990) extended this interpretation to sediment stress by demonstrating that an antibiotic could prevent mortality due to sediment deposition. The loss of corals due to pathogenic effects may result in dramatic changes in coral community structure including fish populations (Gladfelter, 1982; Goenaga et al., 1989). Based on the laboratory investigations of Mitchell and Chet (1975) and Hodgson (1990), there appears to be a direct relationship between environmental stress, bacterial infection, and coral mortality. Field observations suggest that environmental links to pathogenic effects may exist (Peters, 1984), especially for black band disease (Antonius, 1981a, 1981b).

Community interactions are also important in determining community structure. Effects on coral community structure have been reported due to algal grazing by the black long-spined sea urchin (*Diadema antillarum*); the territorial behavior of the threespot damselfish (*Pomacentrus planifrons*); herbivorous reef fish (species of parrotfish and surgeonfish); and predation on corals by fish (species of parrotfish and surgeonfish), a polychaete worm (*Hermodice caruncculata*), and the flamingo tongue snail (*Cyphoma gibbosum*) (Jaap, 1984). The best example of the effects of these complex interactions involves the large-scale mass mortality of the long-spined sea urchin. This urchin was abundant on Caribbean reefs before 1983, but the abundance in particular areas may have depended on fishing intensity on the reefs (Hay, 1984). Areas where reefs were heavily fished presumably resulted in the reduction of herbivorous fish and fish predators of the sea urchin (Hay, 1984). However, beginning in 1983, *D. antillarum* began to die in large numbers, possibly due to a waterborne pathogenic agent (Lessios, 1988). On reefs where sea urchin density was high, algal biomass was low and dominated by algal turfs and crustose algae; macroalgae were not abundant (Carpenter, 1990a). Following the reduction of long-spined sea urchin density, algal community biomass and macroalgae increased (Carpenter, 1990a). Interestingly, algal community productivity decreased (Carpenter, 1990a), and although the numbers of herbivorous fish increased, algal biomass remained high (Carpenter, 1990b). Since *D. antillarum* not only preys on settled coral larvae, but also grazes on algae that inhibit coral larval settlements, the density of this sea urchin may have additional implications for coral community structure (Sammarco, 1980).

Tropical seagrass distribution, productivity, and biomass may also be affected by herbivores and pathogens. Herbivores that feed directly on tropical and subtropical seagrasses include parrotfish, surgeonfish, sea urchins, sea turtles, and manatees. Although much attention has been paid to the "wasting disease" of the temperate seagrass *Zostera*, pathogens of tropical and subtropical seagrasses have received little attention (Zieman, 1982). Barren areas around coral reefs, or "halos," have been attributed to the grazing action of fish and to the sea urchin *Diadema antillarum* (Ogden et al., 1973). Outbreaks of sea urchin (*Lytchechinus variegatus*) grazing on seagrasses have been reported for the west coast of Florida (Zieman, 1982). The historical impact of the formerly large populations of green sea turtles (*Chelonia mydas*) and manatees (*Trichechus spp.*) and the impact on seagrasses due to their decline are virtually unknown (Hay, 1984). One study in two bays of St. John, in the U.S. Virgin Islands (Williams, 1988b), determined that the green sea turtle population was grazing near the carrying capacity of the *Thalassia* bed (i.e., most of the newly produced leaf mass was being consumed by the turtles). In Florida Bay, beginning in 1987, a pathogenic protozoan was suspected as the cause of the "wasting disease" of *Thalassia testudinum* (Robblee et al., 1991). More than 23,000 ha were reported to be affected, and 4,000 ha were completely lost, with areas in protected basins being affected most severely (Robblee et al., 1991). However, other factors such as elevated water temperature, recent decline in frequency of hurricanes, elevated salinity, and chronic hypoxia of the sediments were also noted as possible causative factors.

Lugo et al. (1981) cited herbivore outbreaks as a nonhuman stressor to mangroves. Odum et al. (1982) reviewed the effect on mangrove trees of herbivores, wood borers, and mangrove pathogens. Animals that grazed on live mangroves in Florida included deer, crabs, and insects, such as beetles, moth and butterfly larvae, grasshoppers, and crickets. Reported mangrove pathogens included species of fungi (Odum et al., 1982). Odum et al. (1982) concluded that although infestation rates were high, the magnitude of infestation was related to long-term fluctuations in salinity (increased salinity resulted in more infestation).

Odum et al. (1982) raised the possibility that human interference with water flow could be involved in the changes in salinity and resultant changes in the wood borer population, and they recommended further study. Normal annual fluctuations in rainfall could also play an important role.

6.4.8 Physical Damage and Changes in the Hydric Regime

Severe physical impacts can crush live coral or cause breakage, fractures, or tissue lesions, which provide areas susceptible to invasion by pathogens (Peters, 1984). The loss of coral cover and the concomitant loss of substrate complexity can result in changes in the fish community, which may change over time as the area is recolonized (Dennis and Bright, 1988). The open space created can also be quickly colonized by benthic algae.

Physical damage and changes in coral community structure have been noted following hurricanes (e.g., Jaap, 1984; Rogers et al., 1982; Rogers et al., 1983); ship groundings (e.g., Tilmant 1987; Dennis and Bright, 1988); trampling (e.g., Liddle, 1991, Povey and Keough, 1991); dredging and channel construction (e.g., Adey et al., 1981); coastal construction (e.g., Rogers, 1982); recreational activities such as snorkeling, skin diving, and scuba diving (e.g., Tilmant, 1987; Rogers, 1988); and deployment of boat anchors (e.g., Jaap,1984; Tilmant, 1987; Rogers, 1988).

Physical disturbances to seagrass beds include the destruction caused by storms and hurricanes, as well as direct losses due to dredge and fill operations and damage due to boat propellers and anchors. Zieman (1975) concluded that seagrass beds may be the least susceptible of the coastal marine communities to severe storm damage. The root and rhizome system of seagrass beds is apparently able to withstand the severe storm waves generated, although detached grass blades may be washed up on shore in great quantities (Zieman, 1975).

Williams (1988b) estimated the damage rate to seagrass beds at 1.8 percent per year in two bays in St. John and determined that seagrass recolonization of scars was minimal after a 7-month period. External physical alterations of the environment may also affect seagrass beds. Zieman (1982) hypothesized that channelization of drainage in the Florida Everglades may have resulted in increased salinity of Florida Bay due to rerouting of freshwater flows away from the bay, resulting in the invasion of the more salinity-tolerant seagrass *T. testudinum*.

Direct physical damage results in immediate loss of mangrove trees and may affect the structure of the mangrove along the perimeter of the affected areas. For example, storms and hurricanes may exert extensive physical damage to mangroves. One hurricane in Florida in 1960 (Hurricane Donna) caused the loss of from 25 to 100 percent of mangrove trees over a 40,000-ha area (Odum et al., 1982).

Alterations in the hydric regime are an additional source of stress particular to the mangrove environment. Because the structure and extent of the mangrove system are strongly dependent on the exchange of fresh seawater, the flushing of fresh water, and the input of fresh sediments, the mangrove will rapidly respond to physical alterations that increase or decrease marine or freshwater influences (Lugo et al., 1981; Cintrón and Schaeffer-Novelli, 1983). These changes may be induced by human activity through the construction of roads or dikes or through upland development, which will alter the pattern of water exchange, and hence salinity, within the mangrove. These changes may also be caused by natural events such as long-term deposition or transport of sediments, which would remove or deposit barriers to water exchange, or through more catastrophic events such as storms and hurricanes. An additional source of stress, not addressed in previous sections, is the effect of standing water on mangroves. Water levels that submerge mangrove aerial roots for long periods during the wet season can cause mangrove mortality (Odum et al., 1982).

Examples of mangrove mortality due to human interference with water exchange in mangrove areas are numerous. Stream channelization and the construction of large drainage networks may impair coastal mangroves (Odum et al., 1982; Jiménez et al., 1985). Coastal road construction, if not properly fitted with drainage culverts, can hinder mangrove water circulation (Zucca, 1982). Diking and flooding, a technique often used to enhance waterfowl refuges, may also be fatal to mangroves (Odum et al., 1982). Human encroachment into mangrove areas has an immediate detrimental effect. Mangrove areas are also lost directly as a result of dredge and fill operations and the clearing of mangroves for urban and residential development (Odum et al., 1982).

Most Significant Factors Affecting Impact Modeling

- Three key resources (mangroves, seagrass beds, coral hard bottoms) constitute the major coastal aquatic habitats of southeast Florida.

- Physical, chemical, and biological processes occurring in the ecosystems should be subdivided into multiple categories to fully understand each of the factors affecting habitats and organisms; thus, multiple conceptual models may be required.

- Issue-specific conceptual models should include perturbation sources, affected ecosystem resources, processes affecting perturbation exposure, and processes affecting resource stress.

- Sediments, nutrients, and toxics contributed by point and nonpoint sources alter water quality by changing light, salinity, and dissolved oxygen.

- Other complex factors, such as temperature, pathogens and predators, and physical damage, will also affect the degree and extent of impacts on the biota in these communities.

- For the communities of southeast Florida, the only way to truly assess the impact of the POTW discharges and nonpoint source water quality alterations on the biota of the area is to sample. Without empirical data, no responsible determination can be made.

7. DATA ANALYSES

The information in this section was based on available data obtained from various Federal, State, and local agencies in the southeast Florida study area and from published reports. Particle deposition was mapped for the POTW outfalls and estimated pollutant loadings for point and nonpoint sources were compared.

7.1 DECAL Model Application

The impacts of wastewater discharges in coastal waters are largely exhibited in changes to sediment composition and in subsequent effects on benthic communities (e.g., coral) in the vicinity of the effluent. In the South Florida study area, six municipal POTWs (Delray Beach, Boca Raton, Broward, Hollywood, Miami-Dade North, and Miami Central) discharge to the ocean through submerged outfall diffusers. Because of buoyancy, discharged sewage effluent rises through the water column and seawater is entrained in the waste plume. Subsequent transport and dilution of the wastefield are controlled by coastal transport and mixing processes.

Transport processes include tidal oscillations, wind-driven currents, and large-scale circulation patterns. Tidal motion plays a significant role in distributing the sewage effluent over the tidal excursion zone. Tidal motions in coastal waters typically follow elliptical paths, with the major axes paralleling the shoreline. Oscillation periods range from 12 to 25 hours depending on the relative strength of the major tidal components. For typical tidal velocities (on the order of 6 cm/sec), the major axis of the tidal-excursion ellipse is several kilometers in size. Nontidal flows are composed of wind-driven (or pressure-driven) currents and large-scale mean circulation. The wind-driven currents may exhibit significant variation, often reversing direction in cycles of 4 to 8 days depending on the passage of weather systems. Off the coast of South Florida, nontidal flows are primarily influenced by the Florida Current.

The DECAL (deposition calculation) model provides a simple computerized tool for predicting particle deposition and accumulation of organic material in sediments near municipal ocean outfalls. The model was formulated on the basis of coastal transport, particle transport, and organic carbon cycles. The model includes the effects of coagulation and settling of effluent particles and natural material. The DECAL model was originally developed for the Ocean Data Evaluation System (ODES), which resides on EPA's National Computer Center mainframe computer at Research Triangle Park, North Carolina.

The input parameters for DECAL include the effluent discharge flow rate, the effluent solids concentration, the outfall diffuser location and geometry, the density structure and depth of the water column, the phytoplankton productivity rate, and a simplified description of ocean currents. Three modeling coefficients are required for computing particle deposition and organic accumulation in surface sediments: (1) a second-order coagulation/settling rate coefficient,

(2) a decomposition rate coefficient for suspended organic material, and (3) an interfacial removal rate coefficient for sedimented organic material.

DECAL calculations for accumulations of organic materials in sediments can be used for predicting environmental impacts from the six ocean outfalls in the South Florida study area. In particular, the model was used to determine the spatial impact of organic accumulation of waste from these outfalls on coral reef resources in the study area.

To make accurate predictions, the DECAL model requires ocean current velocity data in the vicinity of the simulated outfall. NOAA, under its SEFLOE II study, recently began collecting ocean current data near most of the outfalls in the South Florida study area. It was anticipated that data from the SEFLOE II study would be used for the DECAL modeling exercise; however, only data from one current meter (near the Broward outfall in the vicinity of Hillsboro Inlet) were available in time for this analysis. Data from this meter covered the period November 25, 1991, to December 20, 1991, or a duration of about 25 days. Analysis of these data showed that the current flowed to the north 49.6 percent of the time with a mean velocity of 22.3 cm/s and to the south 31.2 percent of the time with a mean velocity of 26.2 cm/s. Lee and McGuire (1972) analyzed 9 months of current data off Boca Raton and reported that the current flowed to the north 62 percent of the time with a mean velocity of 23.2 cm/s and to the south 31 percent of the time with a mean velocity of 20.1 cm/s. The east-west series indicated currents to the east 3.5 percent and to the west 3.5 percent of the time with equivalent mean velocities of 8.2 cm/s. There was very little difference between the literature values of Lee and McGuire (1972) and the 1991 SEFLOE II data. Since the length of record used by Lee and McGuire (1972) was much longer than that of the SEFLOE II data, it was decided that their historical data would be used as input for the DECAL model. The DECAL model was applied separately at all six POTWs in the study area using the above north-south current velocities. The cross-shore currents in the east-west direction were considered negligible and were not included in the DECAL model runs.

Short-term currents are attributed to a semi-diurnal tidal component in both the alongshore (6° north) and cross-shore (96° north) directions. Normally, tidal current amplitudes are assigned based on calculations of cumulative variance for periods of less than 1 day. Amplitudes of 35 cm/sec were assigned to both the alongshore and cross-shore directions based on analysis of the 25-day 1991 SEFLOE II data from the current meter at Broward. The computed tidal amplitudes correlated well with NOAA historical data at the entrance to Miami Harbor, which indicate an M2 tidal amplitude of about 36 cm/sec. (The M2 tidal constituent is the primary semi-diurnal frequency associated with the moon's gravitational force.) Phase shifts of tidal velocities in the longshore and cross-shore directions are taken as 45° and 0°, respectively, and are considered to represent an average condition of spreading by tidal motion.

Input parameter values for the DECAL model for each of the six POTWs are listed in Table 7-1. The effluent solids concentration (SS_w), effluent discharge rate (Q_w), effluent BOD concentration (BOD_w), and toxic concentration (TOX_w) were determined from discharge monitoring data for the period July 1990 to July 1991 (see Appendix B). Values for the three modeling coefficients required by decal are:

B = 2.0×10^{-6} L/mg/day
KD = 0.1/day
KS = 5.0×10^{-4} /day

Table 7-1. DECAL Input Data

Parameter	Delray	Boca Raton	Broward	Hollywood	Miami-Dade North	Miami Central
Qw (m^3/sec)	0.635	0.517	2.734	1.604	3.795	5.464
SSw (mg/L)	11.5	4.7	6.6	17.2	19.4	14.8
BODw (mg/L)	10.0	8.0	12.0	10.3	13.1	16.8
TOXw (mg/L)	2.4	0.9	1.0	0.1	0.6	2.9
YAXIS (deg)	0.0	0.0	0.0	0.0	0.0	0.0
XL (km)	6.0	6.0	6.0	6.0	6.0	6.0
YL (km)	16.0	16.0	16.0	16.0	16.0	16.0
X_0, Y_0 (km)	3.0, 4.0	3.0, 4.0	3.0, 4.0	3.0, 4.0	3.0, 4.0	3.0, 4.0
DL (m)	115	115	115	115	115	115
THETA (deg)	0	0	0	0	0	0
TDEPTH (m)	29.3	27.4	45.7	27.4	32.8	33.5
DEPTH (m)	29.2	27.3	45.6	27.3	32.7	33.4
PTOT (gC/m^2/day)	1.6	1.6	1.6	1.6	1.6	1.6
SL (mg/L)	0.0	0.0	0.0	0.0	0.0	0.0
PAXIS (deg)	006	006	006	006	006	006

where:

Q_w = discharge flow rate (m^3/sec)
SS_w = effluent solids concentration (mg/L)
BOD_w = BOD concentration in effluent (mg/L)
TOX_w = toxic parameter concentration in effluent (mg/L)
YAXIS = Y-axis rotation (clockwise from north) in degrees
XL = study area length along X-axis (km)
YL = study area length along Y-axis (km)
X_0, Y_0 = location of diffuser with respect to the origin (km)
DL = diffuser length (m)
THETA = orientation of diffuser (degrees)
TDEPTH = total water column depth (m)
DEPTH = water depth below pycnocline (m)
PTOT = phytoplankton productivity (gC/m^2/day)
SL = in situ production of BOD (mg/L)
PAXIS = orientation of principal axis of ocean currents (azimuth from north in degrees)

where B is the second-order coagulation/settling rate coefficient, KD is the first-order organic material decomposition rate coefficient, and KS is the first-order sediment decomposition rate coefficient.

Results of the DECAL model runs are shown in Figures 7-1 and 7-2. Steady-state distributions for organic accumulation in sediments were calculated, and contours ranging from 0.2 to 8.0 g/m^2 are presented. The two Miami outfalls show the largest accumulation of organic material in the sediments, as is expected because of their large discharge and high suspended sediment concentration. The Boca Raton outfall shows the smallest organic accumulation. Coral reefs in the study area were digitized, and the DECAL results were superimposed over them to determine the locations where coral resources were most susceptible to impacts from the POTW outfalls.

The coral reef locations were provided by Dade County and Broward County and are not available for the entire coastline of the study area. Based on the available data and DECAL model results, the coral resources (i.e., the shaded areas in Figures 7-1 and 7-2) are impacted to some degree by organic waste deposition from all the outfalls with the possible exception of Delray Beach.

7.2 Comparison of Relative Contributions to Water Quality from POTWs and NPS Loadings

Table 7-2 compares the estimated pollutant loadings and concentrations for point (POTW outfall) and nonpoint sources of pollution. Table 7-2 combines the data presented previously in Tables 4-5 and 5-3. Because the reporting criteria for the POTWs differ from the surface water parameters typically monitored, comparisons were made for BOD, total phosphorus, and total nitrogen only.

7.2.1 Comparative Relative Contributions

Comparison of the relative contributions (pounds of pollutant per million gallons of discharge) shows that the POTWs' contribution is greater than that of the nonpoint component.

7.2.2 Comparative Absolute Contributions

When comparing the absolute contributions of the listed parameters, it can be seen that the nonpoint source contribution is greater. For only six canal discharges, the BOD, total phosphorus, and total nitrogen loadings are 82,915 pounds per day, 6,632 pounds per day, and 63,313 pounds per day, respectively. There are eight additional primary canals that discharge to coastal waters within the study area for which monitoring data are not available. The drainage basins for these canals exhibit development patterns similar to those associated with the canals for which there are data. A comparison of the canal discharges to a total loading of 38,439 pounds per day, 7,279 pounds per day, and 49,197 pounds per day (BOD, phosphorus, and nitrogen, respectively) for all six POTWs within the study area demonstrates that the nonpoint source component has a greater absolute contribution to water quality impacts when considering the cumulative contribution of all the drainage canals in the study area.

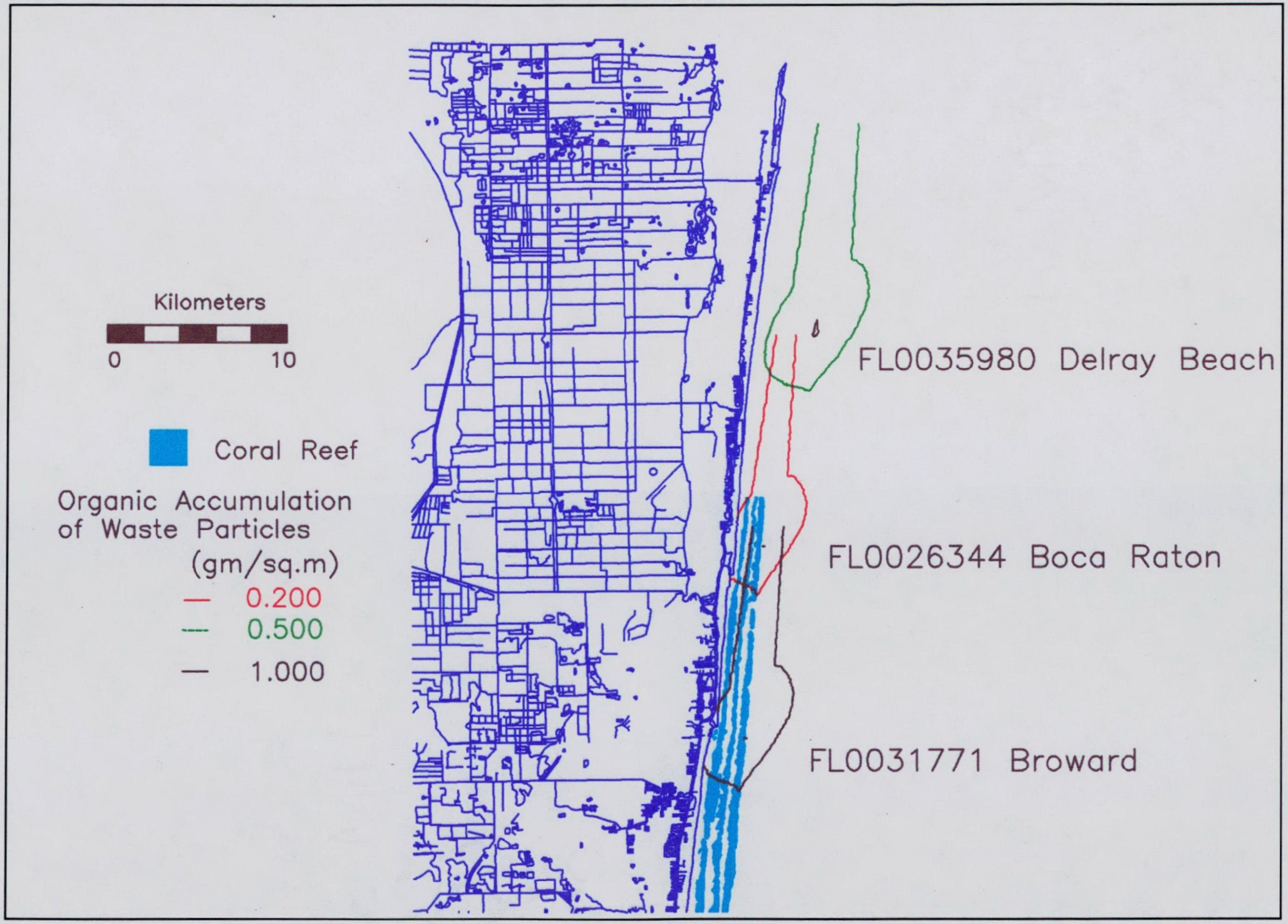


Figure 7-1. DECAL model results for Delray Beach, Boca Raton, and Broward POTWs.

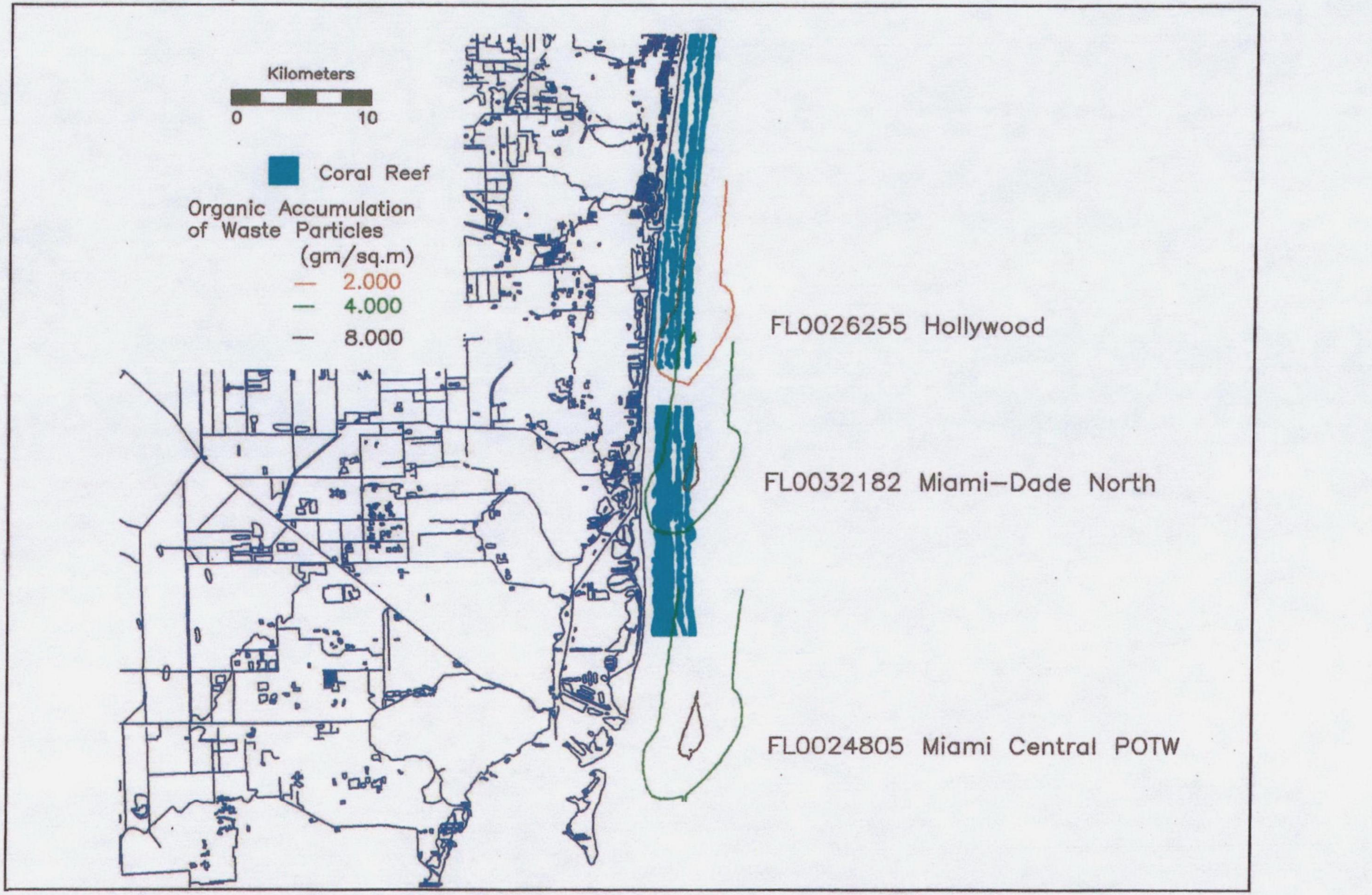


Figure 7-2. DECAL model results for Hollywood, Miami-Dade North, and Miami Central POTWs.

Table 7-2. Comparison of Point and Nonpoint Source Pollutant Loadings

Site	Flow (mgd)	BOD (lb/day)	BOD (lb/10^6 gal)	Total P (lb/day)	Total P (lb/10^6 gal)	Total N (lb/day)	Total N (lb/10^6 gal)
Delray Beach (PS)	14.5	1,167	80.5	307[a]	21.2[a]	2,039[a]	140.6[a]
Broward County (PS)	62.4	6,201	99.4	1,323[a]	21.2[a]	8,773[a]	140.6[a]
Boca Raton (PS)	11.8	793	67.2	250[a]	21.2[a]	1,659[a]	140.6[a]
Holly-wood (PS)	36.6	3,126	85.4	776[a]	21.2[a]	5,146[a]	140.6[a]
Miami-Dade (north) (PS)	86.6	9,576	110.5	1,512	17.5	7,936	91.6
Miami-Dade (central) (PS)	124.7	17,576	140.9	3,111	24.9	23,644	189.6
1 (NPS)	0.064	0.97	15.2	0.04	0.62	0.8	12.5
2 (NPS)	2,761	30,802	11.2	2,228	0.82	2,383	8.5
3 (NPS)	1,603	2,970	1.9	543	0.34	2,038	1.3
4 (NPS)	924	11,272	12.2	267	0.29	9,907	10.7
5 (NPS)	2,562	28,659	11.2	2,785	1.09	21,020	8.2
6 (NPS)	750	9,211	12.3	749	1.5	6,764	9.0

PS = point source; NPS = nonpoint source

[a] There were no data for these facilities for total P and total N; the values were derived by averaging the concentrations for the two Miami-Dade facilities.

8. DISCUSSION

Several factors need to be evaluated when considering the contributions of point and nonpoint pollution in the study area, including the lack of consistent, good-quality monitoring data for nutrients, toxics, and sediment loading, and additional research needs.

8.1 Limitations of This Study

The point sources examined here were very specific: the six publicly owned sewage treatment facilities discharging to coastal waters. There are other point sources of pollution within the study area that were not within the scope of this task. Some of these are listed in Table 8-1. In addition, there may be nutrient and other pollutant inputs in groundwaters discharging to the nearshore area (Lapointe et al., 1990). However, the POTW ocean outfalls have the potential to have the most impact on the coral ecosystem and other benthic communities because of their proximity to the habitat.

Because the nonpoint pollutants were measured in the inshore environment, it is difficult to know what percentage, if any, actually are transported to the reef areas in quantities significant enough to have an impact. Assuming a worst case—all nonpoint source pollutants are transported offshore—their contribution would be greater than that of the point sources. However, this is probably not the case. Heavy metals (while not evaluated here, a significant component of urban runoff) most likely bind to the sediment and fall out of suspension in the canals or close to shore. The hydrodynamics of the intracoastal system prevent many of the pollutants from migrating offshore; however, in the absence of a comprehensive monitoring and modeling program, this is difficult to evaluate.

Most important, the lack of comprehensive data on nonpoint source pollution in the area makes it difficult to effectively evaluate the contribution of such pollution to offshore impacts. Because the nonpoint sources of pollution can be controlled through both structural measures, such as stormwater treatment facilities, and nonstructural controls, such as watershed management planning, landscaping ordinances, and public education programs, their contribution to water quality problems in the study area can be effectively controlled.

For the point sources examined in this study, monitoring for nitrogen and phosphorus is not required for all of the POTWs; therefore, nutrient loadings cannot be determined. Without data on nutrient loadings, the impacts of the point source discharges on the sensitive biological communities cannot be fully ascertained.

Furthermore, no critical analyses of the dynamics of the POTW discharges have been conducted since the SEFLOE I study in 1987-88. The SEFLOE II study is currently being conducted, but data from the study are not yet available. Current data from the Broward County outfall could not be used. The SEFLOE studies have not been fully successful in determining the impacts of

Table 8-1. Other Private and Public Wastewater Treatment Facilities in Southeast Florida (City of Broward)

NPDES No.	Facility Name	RWDLAT	RWDLONG
	PEMBROKE PINES CS		
FL0027367	PLANTATION SOUTH STP	260606	801404
FL0027928	PLANTATION NORTH	260836	801404
	CORAL SPRINGS WEST	260832	800602
	CORAL SPRINGS EAST	260832	800602
	MIRAMAR COLL SYS		
	B.C.U.P. #3		
	DEERFIELD BCH COLL SYS		
	OAKLAND PARK CS	261130	800750
FL0037559	SUNRISE STP NO. 1		
	POMPANO BCH WW COLLN SYS		
FL0021342	EXECUTIVE AIRPORT WWTP	260712	800812
	RIVERLAND ROAD WWTP		
	SIXTH STREET SLUDGE	260800	801000
FL0020524	CORAL RIDGE WWTP	261030	800640
FL0020362	PORT EVERGLADES STP	260712	800812
	MARGATE STP		
	GULFSTREAM STP		
	BONAVENTURE CS		
	FERNCREST STP		
FL0033499	DAVIE STP		
	WESTON CS		
FL0029211	COOPER CITY STP		
	LAUDERHILL WEST WWTP	260832	800602
	LAUDERHILL LAKES CS	260908	801307
	MODERN PCF		
FL0020320	NORTH LAUDERDALE STP	261000	800500
	DANIA CS		
	PALMDALE CS		
FL0037575	SUNRISE STP NO. 3		
FL0037079	SUNRISE STP 2	255617	800703
	WEST TAMARAC STP		
	CORAL SPGS ID STP		

the POTW discharges on water quality. The attempts to determine the degree of bacterial die-off from exposure to salt water were largely inconclusive. To date, there are no studies addressing the impact of residual chlorine, contained in the effluent, on the marine environment off southeast Florida.

The SEFLOE I study examined plume direction for only 2 days. Oceanographic studies have shown that current reversals occur on the order of every 7 to 10 days along the southeast Florida coast. These current reversals, also known as spin-off eddies, are the primary force affecting the coastal waters in this area. They advect Florida Current water onto the shelf and move coastal waters eastward off the shelf. The residence time of the waters located on the "coastal strip" (the area between the west edge of the Florida Current and the shoreline) is approximately 1 week. To fully understand what biological communities are being impacted and the extent of the impact, comprehensive current data would need to be collected and then modeled.

8.2 Other Issues

Recently, an unusually high number of algal blooms have been reported by scientists on the southeast coast of Florida at Palm Beach, Delray Beach, Boca Raton, Jupiter, and Juno Beach (Stephen M. Blair, Dade County Department of Environmental Resource Management, personal communication). These blooms contain an unidentified green alga, *Codium* sp. Little is known about the ecology of this species. Most algal blooms occur as the result of short-term advantageous changes in the environment (e.g., temperature variations or nutrient enrichment). The environmental factors needed by *Codium* sp. to increase in biomass are present in these localized coastal areas and may lead to serious problems for the coral community. These blooms have occurred only in the last 3 to 4 years, with the size and area of the blooms increasing. A cyanobacterium, *Microcoleus lyngbyaceus*, has appeared in blooms in the Indian River, Florida Keys, and coastal waters of Dade County.

Minor changes in terrestrial runoff, increased sewer outfalls, upwelling, increased nutrients, and alterations in the currents or tidal pattern can change coastal water quality and promote algal blooms. The Palm Beach coastal area is also a suitable habitat for *Codium* sp. to flourish, possibly because upwelling of the Florida Current transports nutrients and deep-water *Codium* sp. populations up to the surface waters, increasing the potential for blooms to occur. Although at this time a definite cause is unknown, scientific studies need to be undertaken to determine the cause(s) of the blooms, the impact of *Codium* sp. and cyanobacteria on the coral communities, and the direction of ocean currents off southeast Florida.

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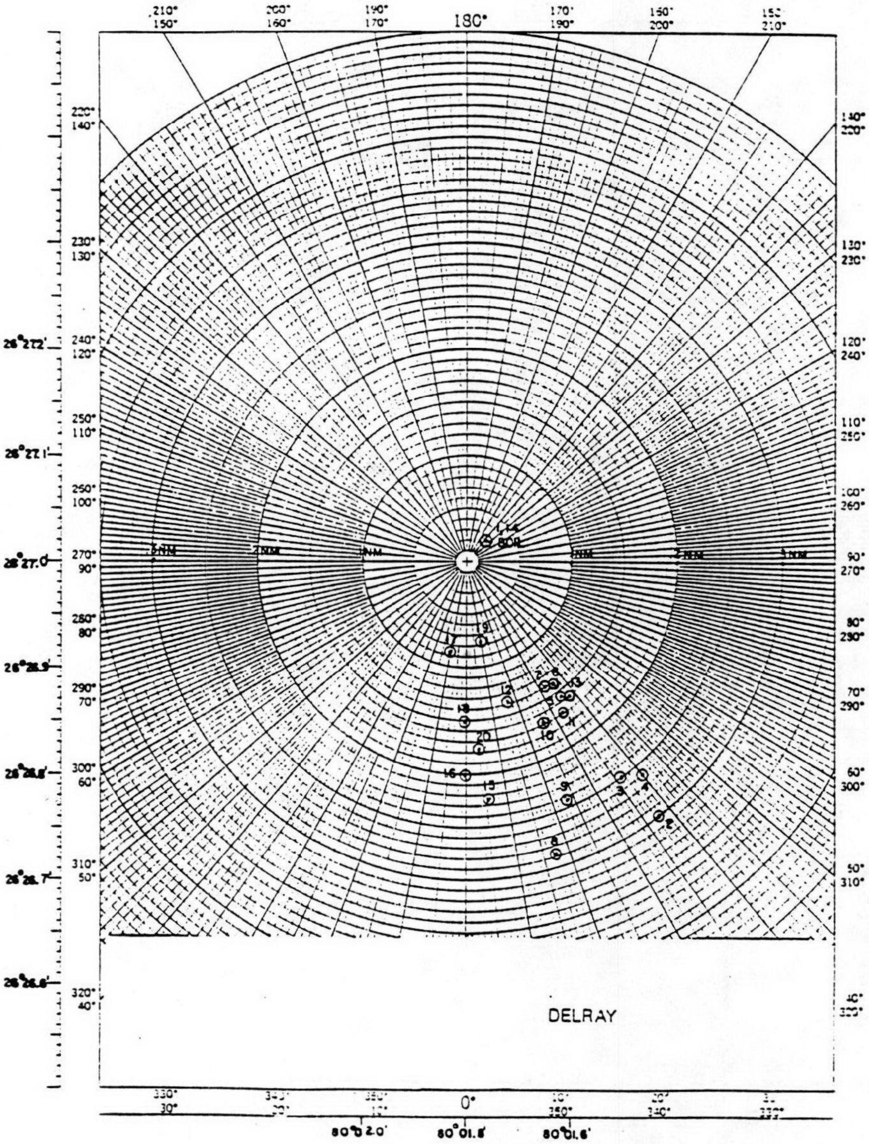
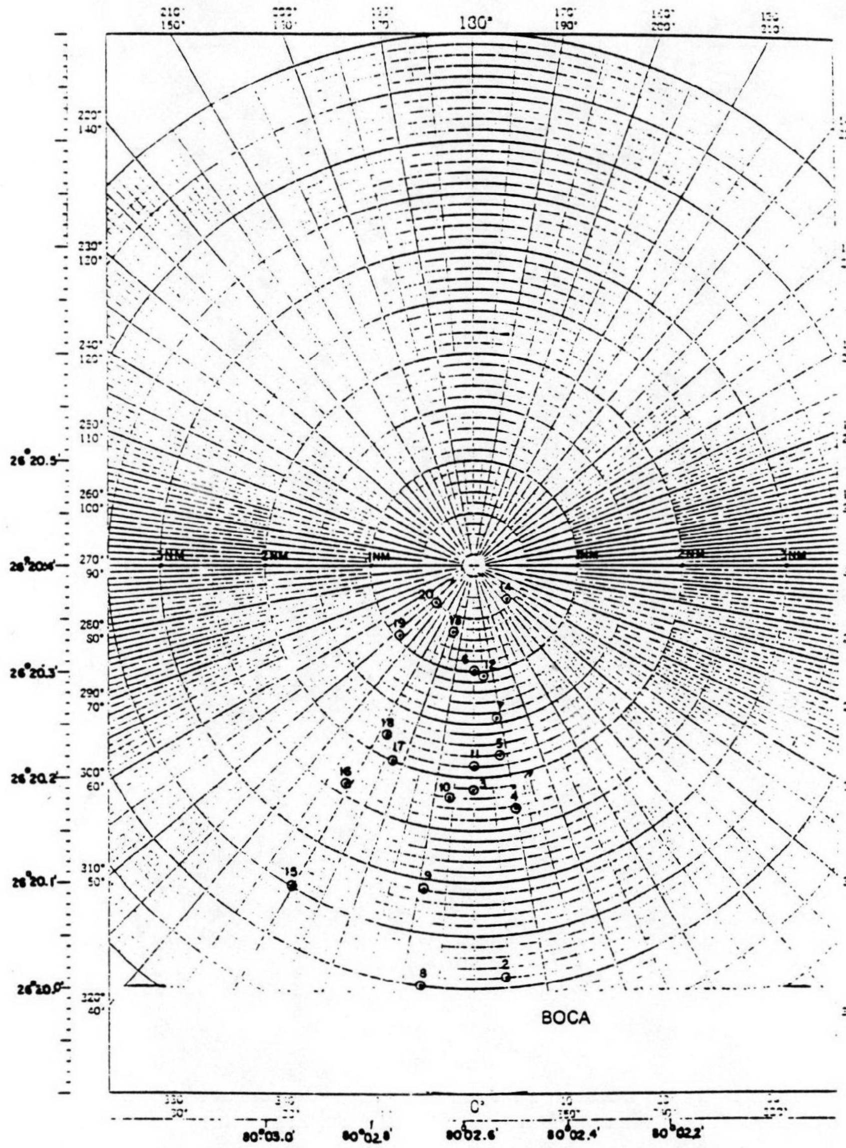
Appendix A

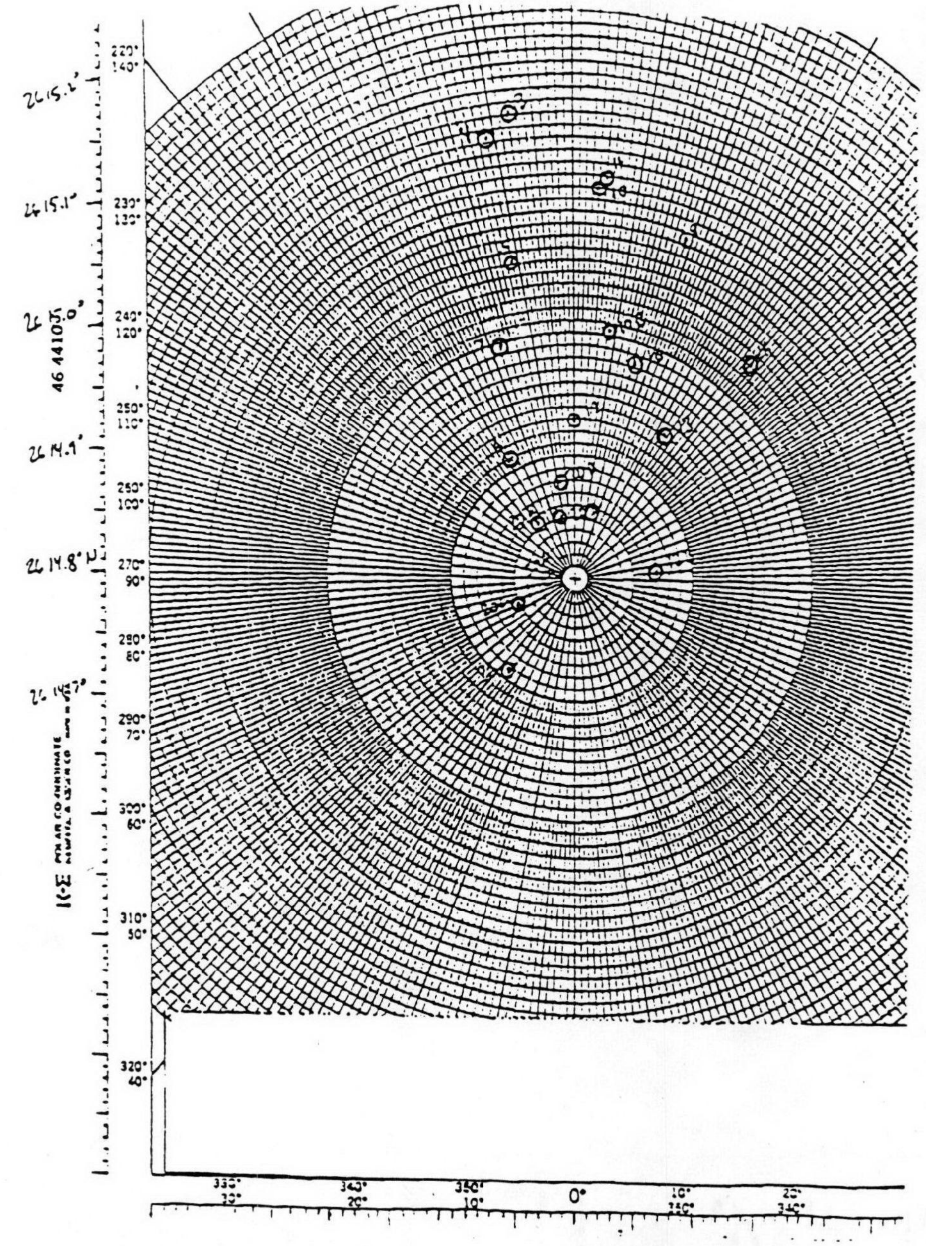
SEFLOE SAMPLING STATION POSITIONS



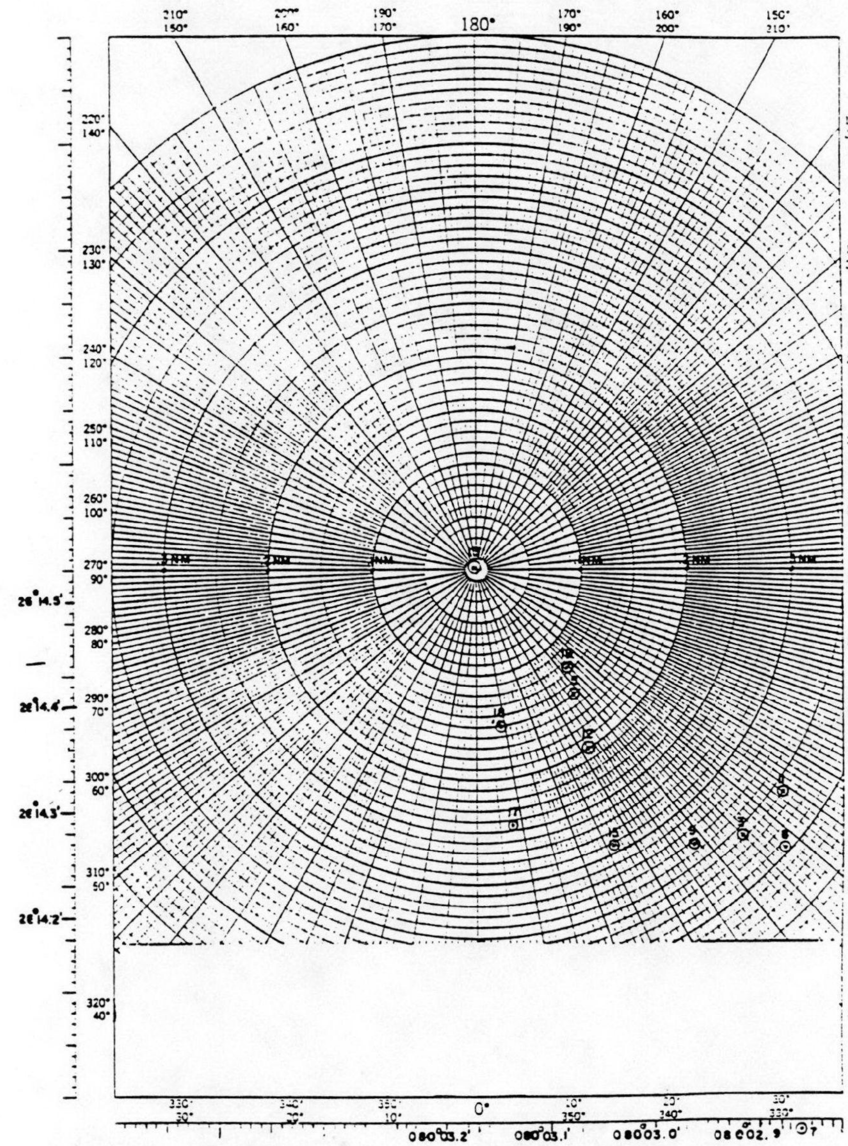
Sampling station positions (01/13/88) for Delray Beach SEFLOE study.



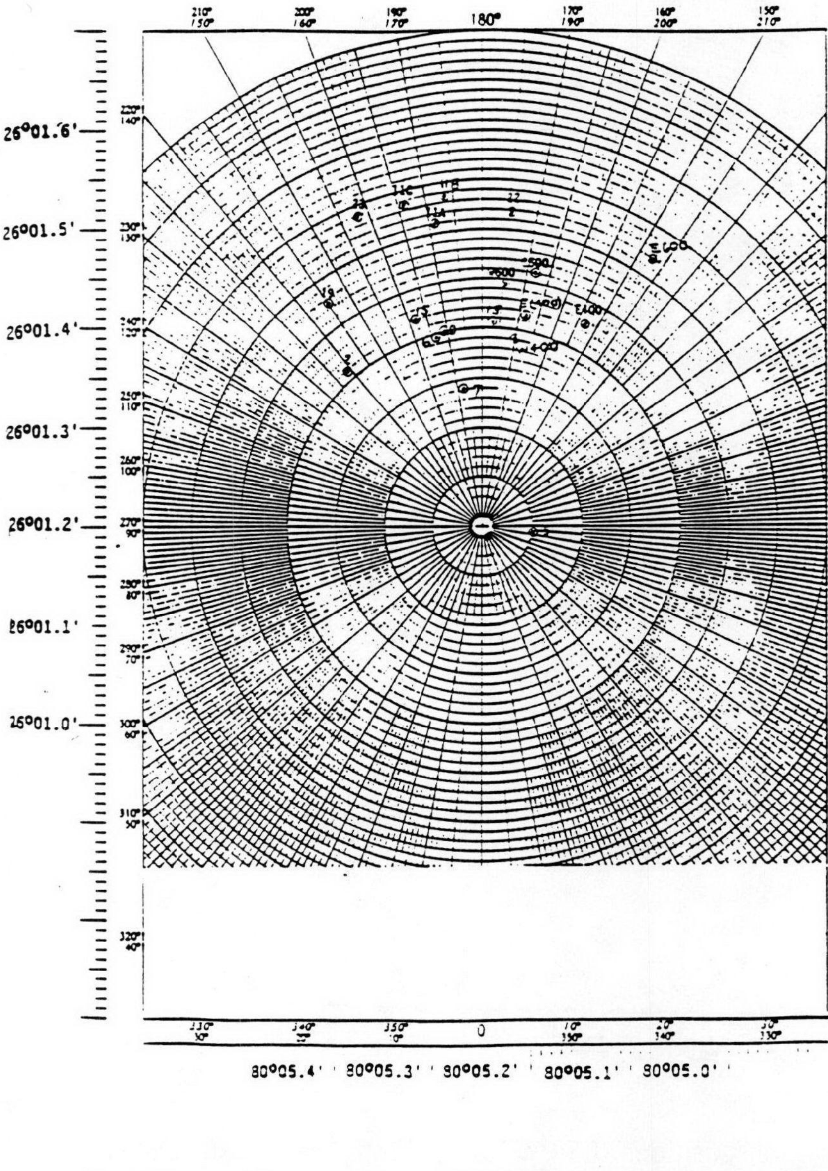
Sampling station positions (01/12/88) for Boca Raton SEFLOE study.



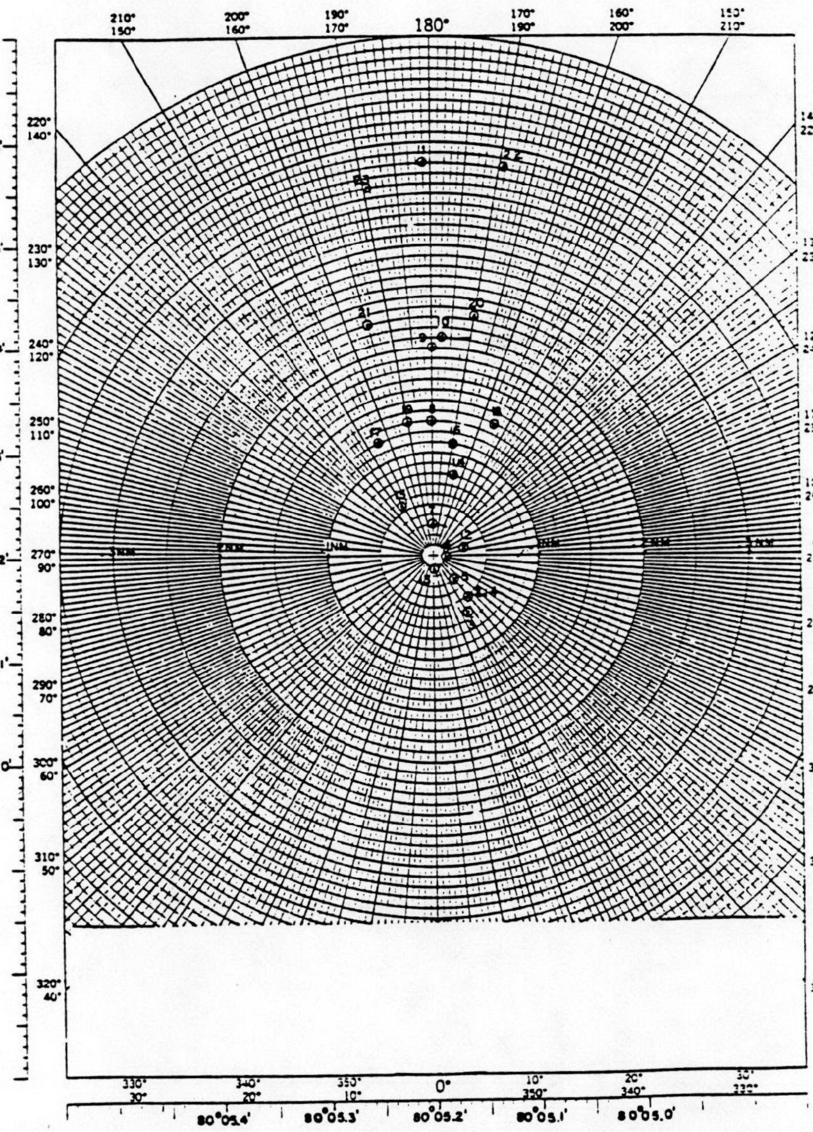
Sampling station positions (10/08/87) for Broward County SEFLOE study.



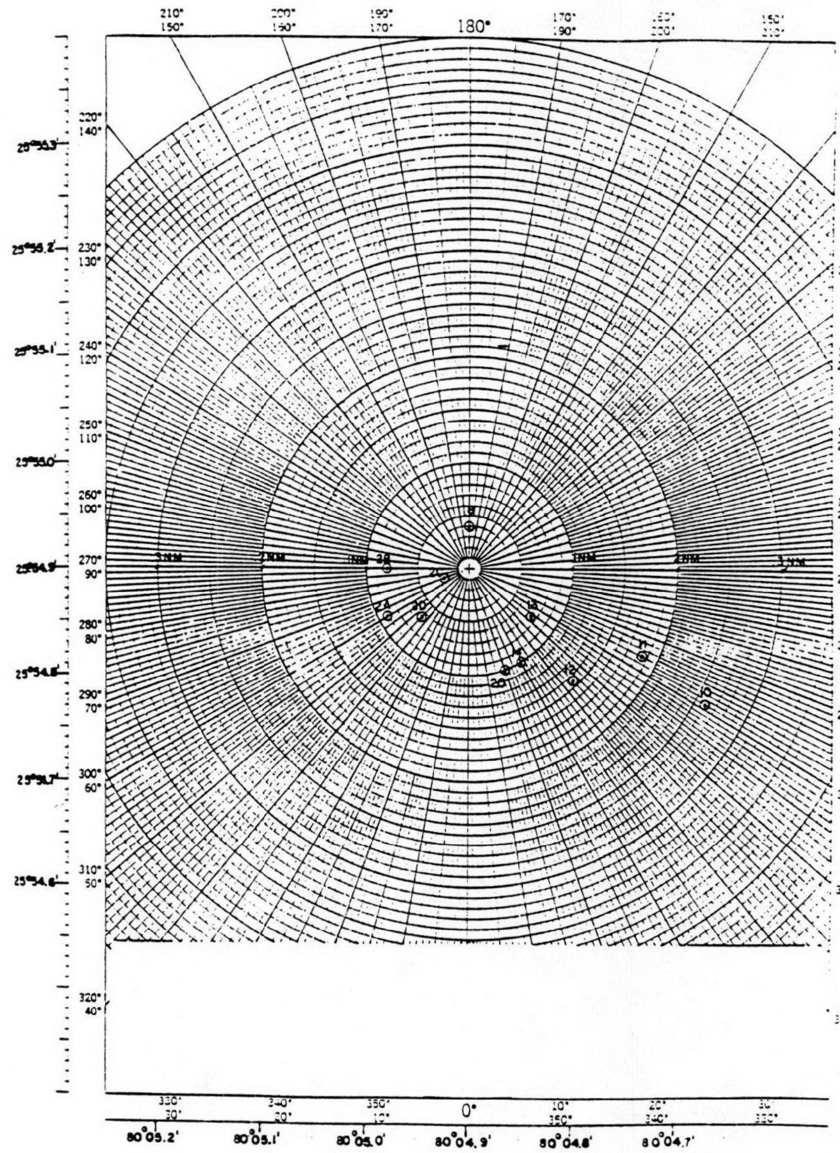
Sampling station positions (01/11/88) for Broward County SEFLOE study.



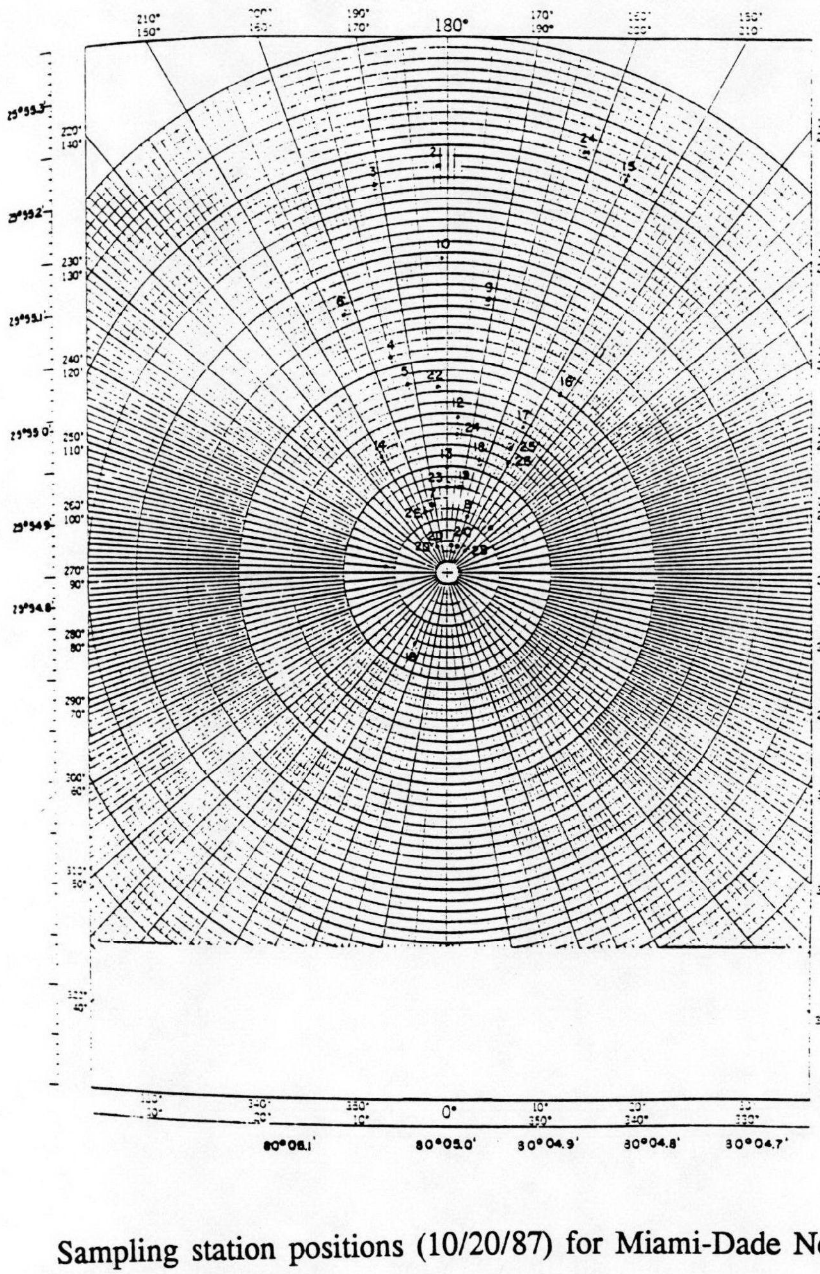
Sampling station positions (06/06/88) for Hollywood SEFLOE study.



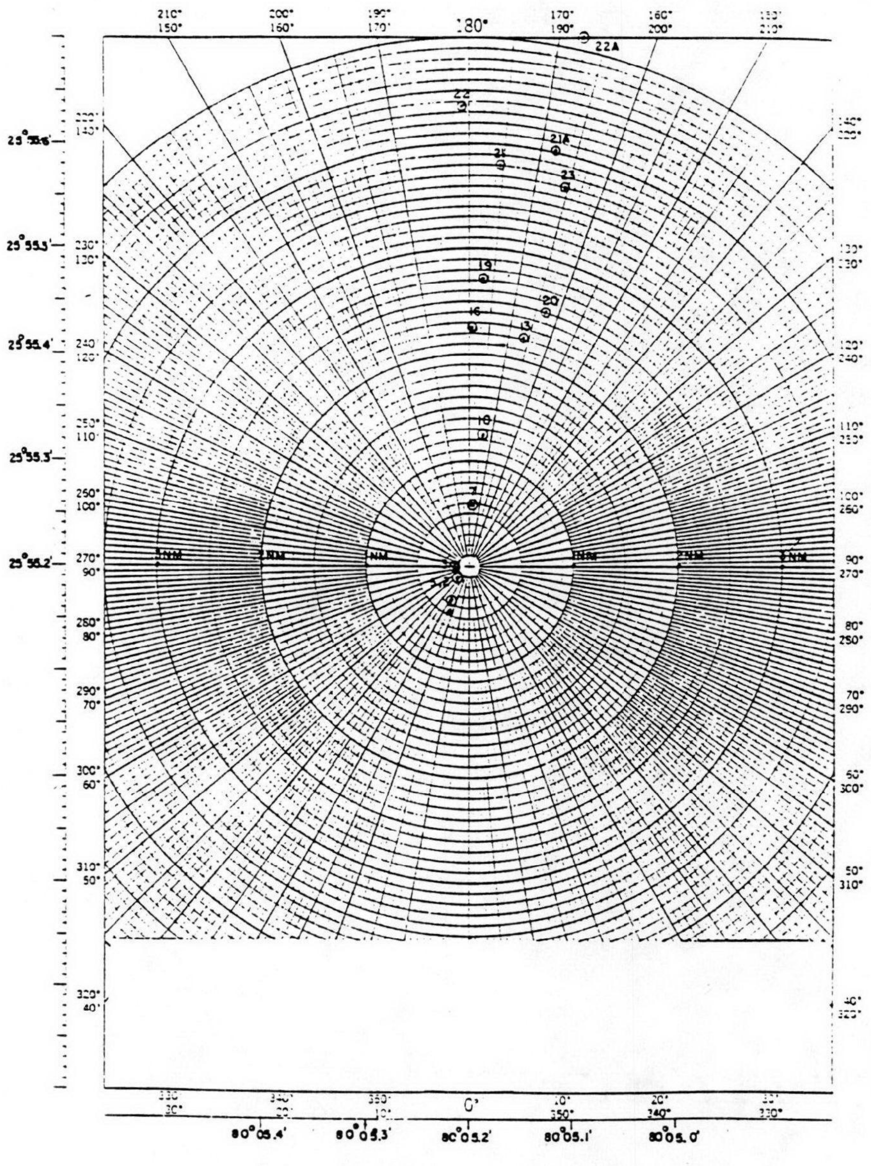
Sampling station positions (06/07/88) for Hollywood SEFLOE study.



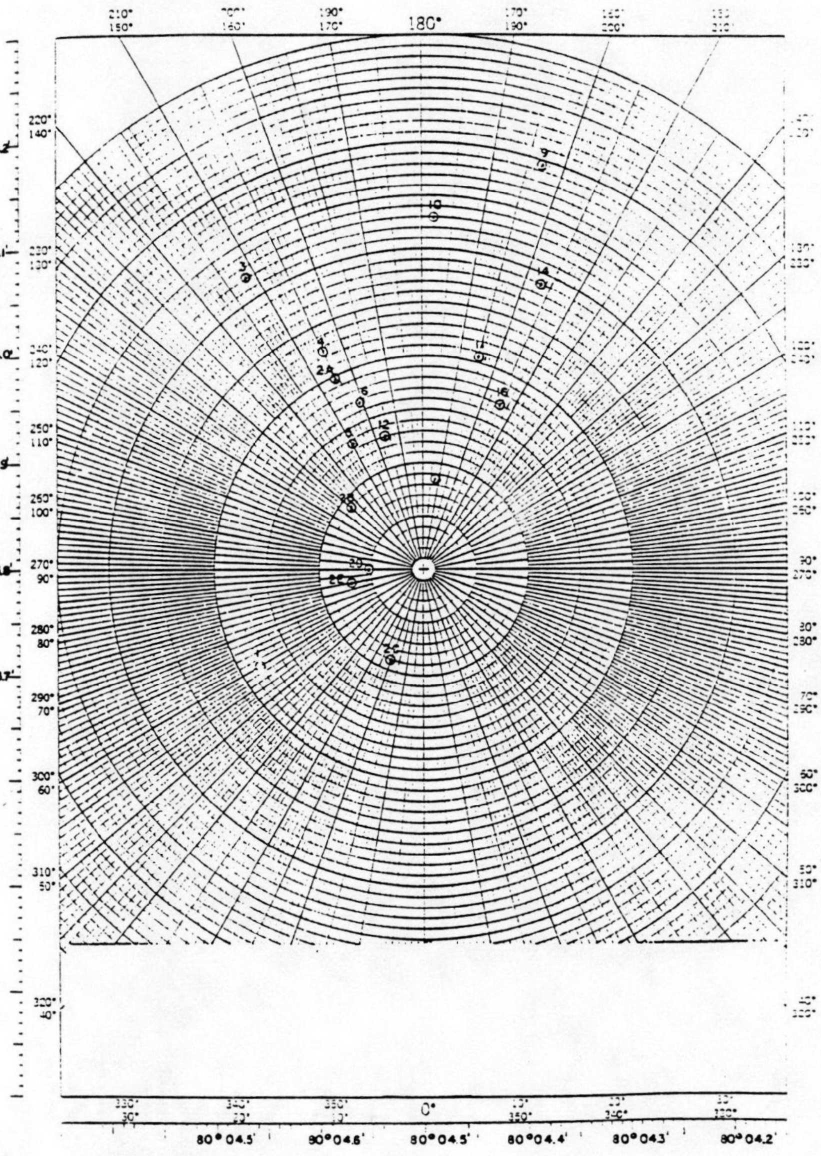
Sampling station positions (10/18/87) for Miami-Dade North District SEFLOE study.



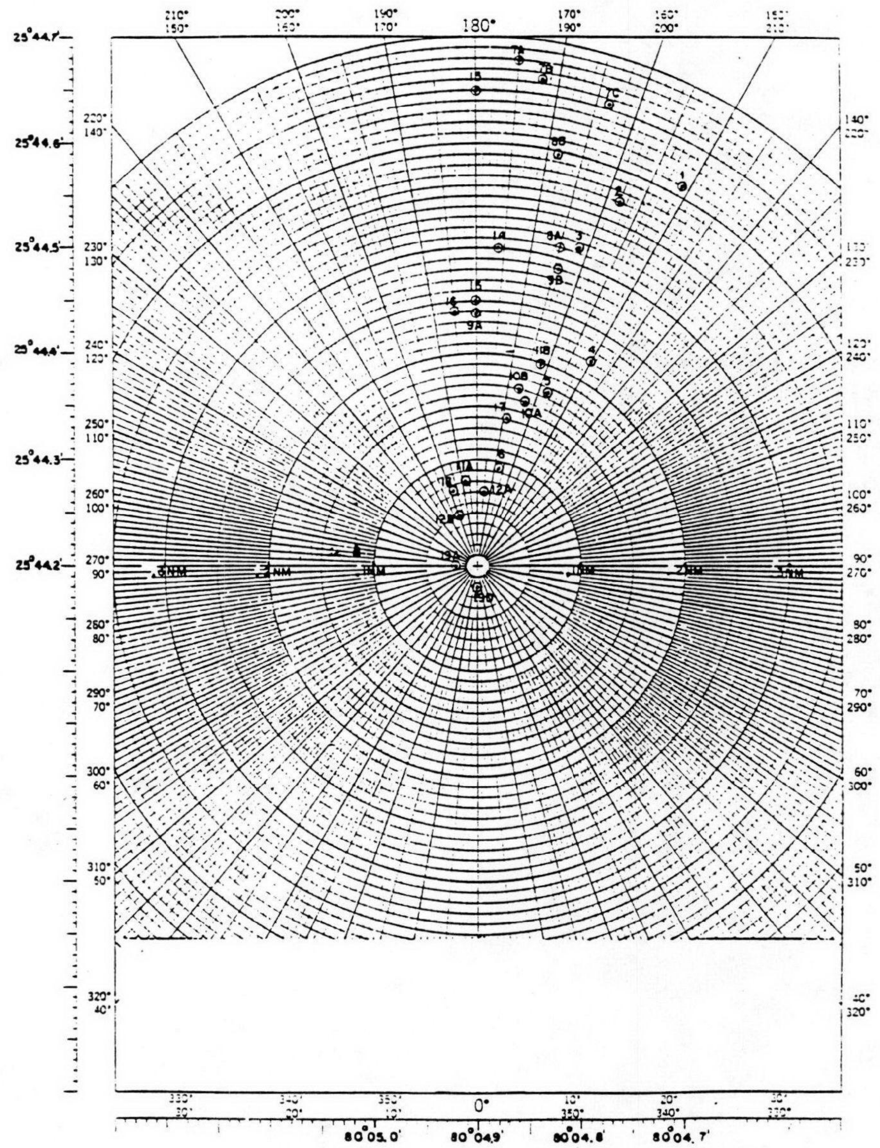
Sampling station positions (10/20/87) for Miami-Dade North District SEFLOE study.



Sampling station positions (06/10/88) for Miami-Dade North District SEFLOE study.



Sampling station positions (10/27/87) for Miami-Dade Central District SEFLOE study.



Sampling station positions (01/24/88) for Miami-Dade Central District SEFLOE study.

Appendix B

NPDES PERMIT LIMITS AND MONITORING DATA

NPDES PERMIT LIMITS AND MONITORING DATA FOR DELRAY BEACH POTW

FL0035980	FLOW		BOD 5-DAY		BOD 5-DAY LOADING		TSS		TSS LOADING		FECAL COLIFORM		pH	TOTAL RESIDUAL CHLORINE
	MGD		MG/L		LBS/D		MG/L		LBS/D		#/100 ML		S.U.	MG/L
	AVG	HIGH	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	RANGE	DAILY MAX
PERMIT LIMITS	REPORT		30.0	45 0	6005 0	9007.0	30 0	45 0	6005 0	9007.0	200.0		6 0 – 8 5	REPORT
MONITORING DATA														
07/90	13.28	15.02	7.0	9.0	799 0	1030.0	7 0	10.0	799 0	1230 0	73.0	245 0	6 7 – 7 2	1 60
08/90	14.13	14 76	7.0	8 0	842 0	975 0	7.0	8 0	779 0	941 0	8 0	182 0	6 7 – 7 2	1 50
09/90	13 73	13.93	7 0	9 0	832 0	991 0	7.0	8 0	774 0	931 0	81 0	312 0	6.8 – 7.1	1 30
10/90	14.17	14.87	8 0	10 0	972 0	1192 0	7 0	11.0	791 0	1374 0	79 0	189 0	6 8 – 7 2	1 30
11/90	14 40	14.65	9 0	10.0	1019 0	1233 0	6 0	8 0	732 0	967.0	79 0	248 0	6 8 – 7 3	3 40
12/90	14.46	15.30	11.0	13 0	1287 0	1513.0	11.0	16.0	1300 0	1953 0	93 0	198 0	6 5 – 7 5	1 40
01/91	16 18	18.17	11 0	14 0	1490 0	1920 0	17 0	30 0	2250 0	4018 0	104 0	1840 0	6 6 – 7 2	2 80
02/91	16.13	16 48	2 0	27.0	N/A	N/A	21.0	26 0	2869 0	3436 0	113 0	188 0	6 6 – 7 2	5 00
03/91	15.73	16 05	21.0	24 0	2697 0	3062.0	18 0	22 0	2413 0	2966 0	121.0	187 0	6 5 – 7.6	2 80
04/91	15.12	15 80	22.0	26 0	2734.0	3192.0	20 0	23.0	2541 0	2789 0	129 0	206 0	6 2 – 7 2	2 70
05/91	14 00	15.26	11.5	16 5	1355 0	1919 0	13.0	20 0	1522 0	2303 0	139 0	540 0	6.3 – 7 1	2 40
06/91	14.49	16.00	6.1	7.1	681.0	688.0	7.0	10.0	800 0	1113 0	141.0	168 0	6 6 – 6 9	1 20
07/91	13.03	13.54	8 0	10 0	848 0	1092.0	8 0	10 0	848 0	1011 0	145 0	161 0	6 3 – 7.2	4 15
YEARLY AVERAGE	14 5		10 0		1196 6		11 5		1416 8		100 4			2 4

NPDES PERMIT LIMITS AND MONITORING DATA FOR BOCA RATON POTW

FL0028344	FLOW		BOD 5-DAY		BOD 5-DAY LOADING		TSS		TSS LOADING		FECAL COLIFORM		pH	TOTAL RESIDUAL CHLORINE	
	MGD		MG/L		LB/D		MG/L		LB/D		#/100 ML		S U.	MG/L	
	AVG	HIGH	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	RANGE	MONTHLY AVG	WEEKLY AVG
PERMIT LIMITS	REPORT		30 0	45 0			30 0	45 0			200 0		6.0 – 8 5	REPORT	REPORT
MONITORING DATA															
07/90	11 420	12 098	4 6	6 7	437.4	674 8	4 0	4 0	380 4	402 9	3 0	384 0	6 3 – 8 2	1 0	1 0
08/90	11 367	11 455	6 0	7.6	567 9	724 9	4.0	5 0	378 6	476 9	3 0	2 0	6 3 – 7 0	1 0	1 0
09/90	10 975	10 979	7 0	9.1	639 7	831 9	5 0	6 0	456 9	548 5	3 0	5 0	6 4 – 7 0	1 0	1 0
10/90	11 598	11 761	5 4	6 4	521 5	626 6	5 0	7 0	462 9	685 5	3 0	1 0	6 2 – 7 2	0 9	0 9
11/90	11 375	11.405	5.9	6 7	558 6	636 3	5 0	5 0	473 6	474 8	3 0	6 0	6 2 – 6 9	1 0	1 0
12/90	11 718	12.191	9.0	11.0	878 2	1118 7	5 0	5 0	487 9	507 6	3 0	3 0	6 3 – 7 0	0 6	0 6
01/91	12 389	12 683	15 0	20 0	1547 4	2113 9	5 0	6 0	515 6	634 2	3 0	4 0	6 1 – 7 1	0 8	0 8
02/91	12 519	12 698	10 2	10 7	1063 3	1131 4	6 0	7 0	625 5	740 2	2 0	6 0	6 3 – 7 0	0 8	0 8
03/91	12 301	12 672	12 0	18 4	1229 2	1972 2	4 0	5 0	409 7	535 9	2 0	38 0	6 0 – 7 1	0 8	0 8
04/91	12 500	12 736	5 7	7 0	593 3	742 4	5 0	7 0	520 4	742 4	2 0	1 0	6 0 – 7 2	0 9	0 9
05/91	11 554	12 103	8 0	11.0	769 7	1106 6	5 0	7 0	481 1	705 5	1 0	2.0	6 4 – 6 7	0 9	0 9
06/91	12 082	12.596	7.0	9.0	704 2	944 0	4 0	5 0	402 4	524 4	1.0	1 0	6 3 – 6 9	0 8	0 8
07/91	11 848	12 351	8.1	11.3	799 1	1162.2	4 0	7 0	394 6	719 9	1 0	2040 0	6 3 – 6 3	0 8	0 8
YEARLY AVERAGE	11.8		8 0		793 1		4 7		462 3		2 3			0 9	

NPDES PERMIT LIMITS AND MONITORING DATA FOR BROWARD COUNTY POTW

FL0031771	FLOW		BOD 5-DAY		BOD 5-DAY LOADING		TSS		TSS LOADING		FECAL COLIFORM		TOTAL PHOS	TOTAL NITROGEN	pH	TOTAL RESIDUAL CHLORINE
	MGD		MG/L		LBS/D		MG/L		LBS/D		#/100 ML		MG/L	MG/L	S U.	MG/L
	AVG	HIGH	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	1/MONTH	1/MONTH	RANGE	DAILY MAX
PERMIT LIMITS	REPORT		25 0	40 0	REPORT	REPORT	30 0	45 0	REPORT	REPORT	200 0	REPORT	REPORT	REPORT	6.0 – 8 5	REPORT
MONITORING DATA																
07/90	61.47	63 98	24 0	27 0	12364 0	13085 0	6 0	9 0	3177 0	4237 0	5 0	80 0	N/A	N/A	6.8 – 7 5	N/A
08/90	61 90	64 10	22.0	29 0	11160 0	15647 0	10 0	13 0	4824 0	6409 0	5 0	10 0	N/A	N/A	6 9 – 7.7	N/A
09/90	56 81	61 68	26.0	29 0	12597.0	13160 0	9 0	11 0	4293 0	5284 0	5 0	10.0	N/A	N/A	6 8 – 7 4	N/A
10/90	64 63	69 45	28 0	32 0	15075 0	17604 0	7 0	10 0	4015 0	5190 0	5 0	400 0	N/A	N/A	6 7 – 7 4	N/A
11/90	56 63	60 04	8 0	17 0	3839.0	8361 0	7 0	9.0	3181 0	4294 0	5 0	10 0	N/A	N/A	6 8 – 7 4	N/A
12/90	57 31	60 13	3 0	4 0	1547 0	1855 0	7 0	7 0	3220 0	3633 0	4 0	80 0	N/A	N/A	6 7 – 7 7	N/A
01/91	63 97	68 76	6 0	10 0	3435 0	5623 0	8 0	10 0	4453 0	5523 0	4 0	90 0	N/A	N/A	6 9 – 7 8	N/A
02/91	64.54	68 34	5 0	7 0	2566 0	4102 0	7 0	8 0	3939 0	4163 0	5 0	2000 0	N/A	N/A	7 1 – 7 9	N/A
03/91	64 06	67 94	6 0	10 0	2911 0	5008 0	5 0	6 0	2624 0	3651 0	5 0	60 0	N/A	N/A	6 9 – 7 6	N/A
04/91	62 36	69 14	5 0	8 0	2344.0	3392 0	7.0	7 0	3464 0	4406 0	4 0	20 0	N/A	N/A	6 9 – 7 7	N/A
05/91	59 22	65 43	4 0	5 0	2100 0	2512 0	4 0	5 0	2160 0	2585 0	4 0	30 0	N/A	N/A	7 0 – 7 7	N/A
06/91	71 38	79 66	8 0	12 0	4543 0	6967 0	5 0	7 0	2982 0	4928 0	4 0	10 0	N/A	N/A	6 9 – 7 6	N/A
07/91	64 25	66 88	11 0	15 0	6088 0	7776 0	4 0	5 0	2176 0	2535 0	4 0	10 0	N/A	N/A	7.2 – 7 7	N/A
YEARLY AVERAGE	62 4		12 0		6200 7		6 6		3446 8		4 5					

NPDES PERMIT LIMITS AND MONITORING DATA FOR HOLLYWOOD POTW

FL0026255	FLOW		BOD 5-DAY		BOD 5-DAY LOADING		TSS		TSS LOADING		FECAL COLIFORM		TOTAL PHOS	TOTAL NITROGEN	pH	TOTAL RESIDUAL CHLORINE
	MGD		MG/L		LBS/D		MG/L		LBS/D		#/100 ML		MG/L	MG/L	S U	MG/L
	AVG	HIGH	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	1/MONTH	1/MONTH	RANGE	DAILY MAX
PERMIT LIMITS	REPORT		30.0	45.0	REPORT	REPORT	30 0	45 0	REPORT	REPORT	200 0		REPORT	REPORT	6 0 – 8 5	REPORT
MONITORING DATA																
07/90	35 3	38 3	15 0	16 0	4277 0	4453 0	34 0	43 0	8965 0	12056 0	5 0	11 0	N/A	N/A	6 0 – 7 5	N/A
08/90	35 7	39 1	8.0	10 0	2274 0	3154 0	17 0	26 0	5136 0	8012 0	4 0	9 0	N/A	N/A	6 0 – 7 0	N/A
09/90	35 0	37 0	7 0	7 0	1951 0	2048 0	13 0	13 0	3894 0	3707 0	2 0	3 0	N/A	N/A	6 0 – 7 2	N/A
10/90	35 9	39 4	8.0	10 0	2539.0	2950 0	15 0	19 0	4545 0	6338 0	7 0	10 0	N/A	N/A	6 0 – 7 6	N/A
11/90	34.3	35 7	14.0	21 0	4073 0	5985 0	28 0	47 0	8050 0	13453 0	13 0	24 0	N/A	N/A	5 3 – 7 2	N/A
12/90	33 5	34 4	18 0	20.0	5165 0	5729 0	15 0	19 0	4238 0	5267 0	2 0	4 0	N/A	N/A	5 9 – 7 3	N/A
01/91	38.1	41 1	12 0	13 0	3962 0	3987 0	13 0	16 0	4236 0	5507 0	5 0	21 0	N/A	N/A	5 9 – 6 5	N/A
02/91	39 1	43 0	11 0	11 0	3485 0	4095 0	15 0	17 0	4882 0	5327 0	6 0	18 0	N/A	N/A	6 1 – 6 9	N/A
03/91	33 5	35 1	10 0	12 0	2752.0	3131 0	13 0	15 0	3570 0	4400 0	5 0	12 0	N/A	N/A	6 2 – 6 7	N/A
04/91	35 6	39 4	10 0	13 0	3129 0	3714 0	11 0	13 0	3336 0	4422 0	19 0	28 0	N/A	N/A	6 1 – 6 7	N/A
05/91	36 2	40 3	7 0	7 0	1973 0	2103 0	10 0	12 0	3111 0	3423 0	11 0	23.0	N/A	N/A	5 3 – 7 5	N/A
06/91	42 9	46 1	10 0	14.0	3505 0	4039 0	27 0	44 0	9496 0	15398 0	6 0	15 0	N/A	N/A	5 9 – 7 5	N/A
07/91	40 2	44.5	4 0	9 0	1548 0	2981 0	13 0	15 0	4300 0	4797 0	3 0	4 0	N/A	N/A	4 8 – 7 3	N/A
YEARLY AVERAGE	36 6		10 3		3125 6		17 2		5272 2		6 9					

NPDES PERMIT LIMITS AND MONITORING DATA FOR MIAMI-DADE NORTH POTW

FL0032182	FLOW		BOD 5-DAY		BOD 5-DAY LOADING		TSS		TSS LOADING		FECAL COLIFORM		TOTAL PHOS	TOTAL NITROGEN	pH	TOTAL RESIDUAL CHLORINE	
	MGD		MG/L		LBS/D		MG/L		LBS/D		#/100 ML		MG/L	MG/L	S.U.	. MG/L	
	AVG	HIGH	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	1/MONTH	1/MONTH	RANGE	MONTHLY AVG	WEEKLY AVG
PERMIT LIMITS	REPORT		30 0	45 0	REPORT	REPORT	30 0	45 0	REPORT	REPORT	200 0		REPORT	REPORT	6 0 – 8 5	REPORT	
MONITORING DATA																	
07/90	81 42	82 25	9 0	11.0	6353 0	7245 0	14 0	17 0	9267.0	11151 0	17 0	34 0	N/A	N/A	6 1 – 6 4	0 54	0 56
08/90	93 23	100.40	11 0	12.0	6886.0	9230 0	21 0	29 0	15703 0	19050 0	23 0	40 0	N/A	N/A	6 0 – 6 6	0 52	0 53
09/90	92 40	95 07	12 0	12 0	6891 0	9904 0	22 0	25 0	16858 0	20132 0	7 0	11 0	N/A	N/A	6 0 – 6 6	0 55	0 57
10/90	96 46	106.30	12 0	14 0	9684 0	12031 0	17 0	20 0	13476 0	17992 0	35 0	83 0	N/A	N/A	6 1 – 6 6	0 56	0 62
11/90	87 50	89 14	12 0	14 0	8675 0	10080 0	19 0	25.0	14152 0	18863 0	11 0	23 0	N/A	N/A	6 2 – 6 7	0 59	0 64
12/90	80 67	84 94	11 0	12 0	7351 0	7968.0	21 0	27 0	13768 0	17539 0	29 0	93 0	N/A	N/A	6 0 – 6 6	0 55	0 57
01/91	81.78	84 60	13 0	14 0	9208 0	9898 0	19 0	23 0	13185 0	15218 0	26 0	48 0	N/A	N/A	6 0 – 6 5	0 57	0 61
02/91	83 98	87 47	21 0	24 0	14753 0	16726 0	28 0	34 0	19592 0	23352 0	38 0	59 0	N/A	N/A	6 0 – 6 6	0 54	0 55
03/91	81 38	85 08	25 0	31 0	16968.0	20498 0	28 0	31.0	19205 0	21028 0	14 0	63 0	N/A	N/A	6 2 – 6 7	0 59	0 64
04/91	86 03	87 46	10 0	10 0	6820 0	7408.0	16 0	17 0	11437.0	12078 0	7 0	10 0	N/A	N/A	6 2 – 6 7	0 65	0 70
05/91	83 73	91 27	11.0	14 0	7555 0	10660 0	12 0	15 0	8775 0	11219 0	6 0	18 0	N/A	N/A	6 2 – 6 6	0 63	0 74
06/91	88 60	94 37	12 0	15 0	N/A	N/A	16 0	24 0	N/A	N/A	19 0	66 0	1 57	9 63	6 3 – 7 0	0 57	0 62
07/91	88 50	91 58	11 0	14 0	N/A	N/A	19 0	26 0	N/A	N/A	15 0	20 0	2 65	12 40	6 1 – 6 7	0 94	1 00
YEARLY AVERAGE	86 6		13 1		9578 5		19 4		14138 0		19 2		2 1			0 6	

NPDES PERMIT LIMITS AND MONITORING DATA FOR MIAMI CENTRAL POTW

FL0024805	FLOW		BOD 5-DAY		BOD 5-DAY LOADING		TSS		TSS LOADING		FECAL COLIFORM		TOTAL PHOS	TOTAL NITROGEN	pH	TOTAL RESIDUAL CHLORINE
	MGD		MG/L		LBS/D		MG/L		LBS/D		#/100 ML		MG/L	MG/L	S.U.	MG/L
	AVG	HIGH	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	MONTHLY AVG	WEEKLY AVG	1/MONTH	1/MONTH	RANGE	DAILY MAX
PERMIT LIMITS	REPORT		30 0	45 0	REPORT	REPORT	30 0	45 0	REPORT	REPORT	200 0	REPORT	REPORT	REPORT	6 0 – 8 5	REPORT
MONITORING DATA	-															
07/90	[illegible] 3	136 6	18 0	23.0	18614.0	23703.0	10 0	12 0	10369.0	18570.0	3 0	400 0	N/A	N/A	6 1 – 6 9	3 1
08/90	134 3	142 9	17 0	18 0	19306.0	21717.0	8 0	8 0	9260 0	15692.0	2 0	74000.0	N/A	N/A	6 0 – 7 0	2 0
09/90	131 0	137 6	20 0	22 0	21514.0	24046.0	11 0	14 0	12409.0	21284.0	2 0	860 0	N/A	N/A	6 1 – 7.1	2 0
10/90	131.0	142 3	15 0	15 0	16109.0	17066.0	10 0	11 0	11234.0	17414.0	2 0	120.0	N/A	N/A	6 3 – 7 0	2 0
11/90	124 6	135 0	13 0	16 0	13640.0	17387.0	11 0	12 0	11708.0	19963.0	2 0	240[illegible]	N/A	N/A	6 1 – 7 0	3 1
12/90	113 2	119 1	17 0	20 0	16271.0	19613.0	18 0	28 0	17382.0	58314.0	1 0	21.0	N/A	N/A	6 1 – 7 1	3 5
01/91	122 9	129 3	18.0	20 0	N/A	N/A	22 0	29 0	N/A	N/A	2 0	25.0	4 27	24 60	6 0 – 7 1	3 5
02/91	124 8	130.3	17 0	21 0	N/A	N/A	21 0	27 0	N/A	N/A	2 0	10 0	3 67	21 90	6 2 – 6 8	2 5
03/91	122 5	127 4	16 0	17 0	N/A	N/A	15 0	16 0	N/A	N/A	2 0	13 0	3 04	27 30	6 1 – 6 9	3 5
04/91	117 1	119.4	17 0	18 0	N/A	N/A	22 0	28 0	N/A	N/A	2 0	61 0	2 74	22 40	6 3 – 7 0	2 4
05/91	111 5	119 4	18 0	23 0	N/A	N/A	19 0	23 0	N/A	N/A	3 0	1280000.0	2 39	25 70	6 3 – 7 1	3 5
06/91	130 8	141 8	18 0	23 0	N/A	N/A	13 0	16 0	N/A	N/A	4 0	530 0	1 98	18 30	6 1 – 7 1	3 5
07/91	130 7	138.8	14 0	16 0	N/A	N/A	12 0	20 0	N/A	N/A	2 0	200 0	2 98	19 20	6.1 – 7 1	3 5
YEARLY AVERAGE	124 7		16 8		17575 7		14 8		12065.3		2 2		3 0	22 8		2 9

Appendix C

CONTACT LIST FOR SOUTH FLORIDA WATER QUALITY STUDY

CONTACT LIST FOR SOUTH FLORIDA WATER QUALITY STUDY

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