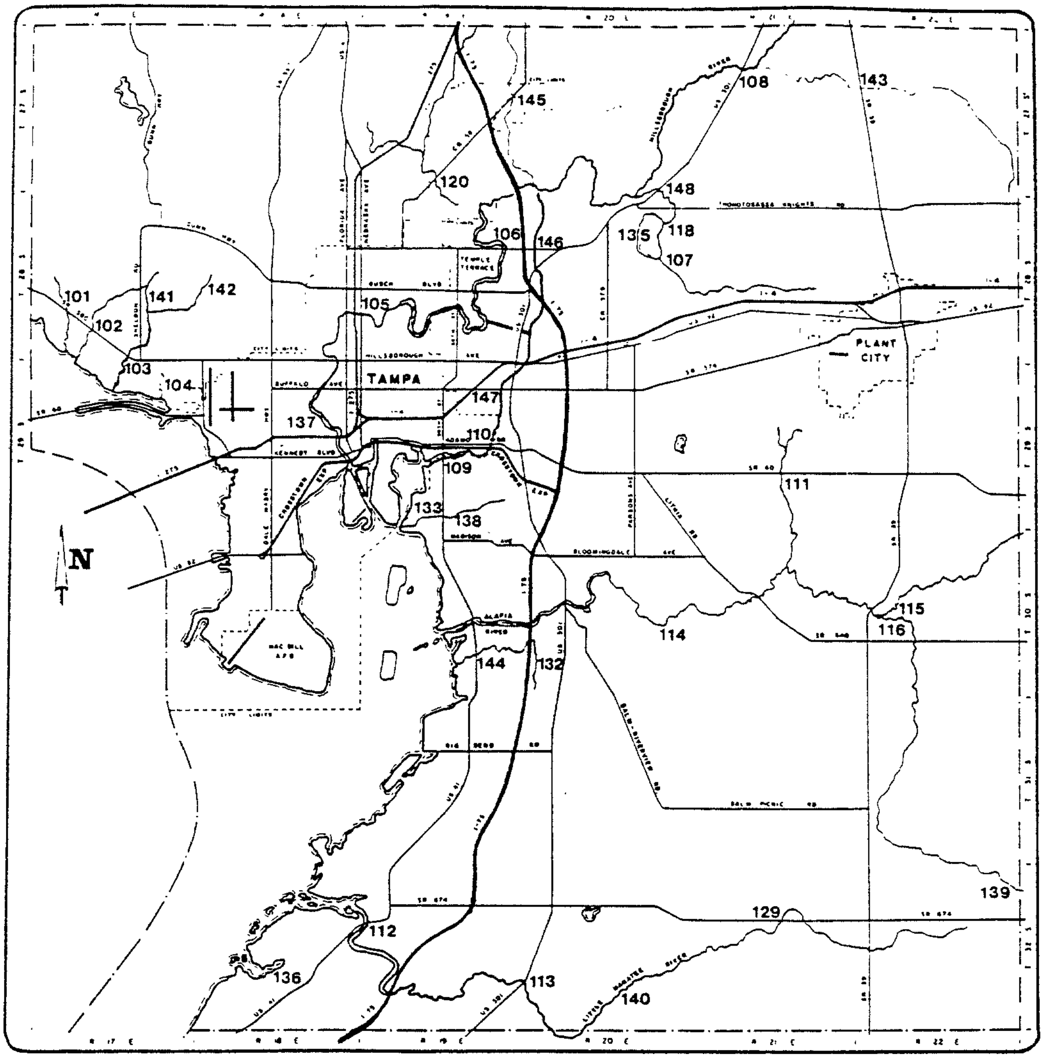

Figure 2. Freshwater monitoring stations, Hillsborough County, Florida.

absorption of radiation by water itself or by substances dissolved in water measured as "color" and the scattering of radiation by suspended matter measured as "turbidity" (Jerlov 1976). A Secchi disk is used to determine the vertical penetration of light.

An indication of the light climate of a body of water can be acquired by measuring turbidity, color, and light penetration. A comparison of these parameters throughout the waters of Tampa Bay can provide information concerning not only the relative degree of water clarity, but also the location of point and nonpoint sources of water pollution.

The waters of Tampa Bay were monitored monthly for turbidity, color and effective light penetration. Water samples were collected from mid-depth and analyzed in the laboratory for the determination of turbidity and color. Effective light penetration was measured in the field utilizing a 20 cm diameter Secchi disc.

Turbidity. Turbidity in water is the optical property of a sample which causes light to be scattered. Turbidity in water may be caused by the presence of suspended matter such as clay, silt, finely divided organic and inorganic matter, plankton or other microscopic organisms (Standard Methods 1989). Excessive turbidity in a body of water decreases the light intensity as it penetrates the water column resulting in a decreased compensation point of photosynthesis with a concomitant reduction of flora

(Estevez 1987). The injurious effect of turbidity may also result in the deposition of sediment on the surface of benthic flora and fauna (Stern and Stickle 1978, Wright 1978).

During 1988, turbidity annual averages ranged from 3 to 10 NTU; in 1989, the annual average turbidity range increased 5 to 13 NTU. In 1988, annual average turbidity of 7 NTU or greater were restricted to small areas in Hillsborough Bay (HB) and Old Tampa Bay (OTB); in 1989, all of Hillsborough Bay and portions of Old Tampa Bay and Middle Tampa Bay (MTB), as well as coastal areas of Lower Tampa Bay (LTB), had annual turbidity averages of 7 NTU or greater.

Figure 3 provides a historical perspective of turbidity in four major subdivisions of Tampa Bay (Lewis and Whitman 1985) by presenting the annual average turbidity values from 1974 through 1985. The graph shows considerable variability from year to year with no clear trend. Usually, Hillsborough Bay has had the highest turbidity values as compared to the three other areas of the Tampa Bay. Hillsborough Bay had high levels of turbidity in 1978 and 1979; concurrently, that area of Tampa Bay was being dredged as part of the Tampa Harbor Deepening Project. Turbidity values in Hillsborough Bay dramatically declined in 1980 to the lowest average measured, only to be followed in 1981 by a dramatic increase to the highest average measured at that time. The cause of the extreme variation in turbidity measurements for these two years is not known; quality control data was good. Meteorological conditions (wind and rain) may be a factor. In 1982 the turbidity average for Hillsborough Bay increased again and then decreased in 1983. Old Tampa Bay and Middle Tampa Bay had declining turbidity averages in 1982 and 1983. Turbidity values fluctuated only slightly in the other major subsections of Tampa Bay during 1984 and 1985, while increased turbidity was measured in Hillsborough Bay. However, 1983 was a year of unusually low turbidity in Hillsborough Bay; therefore, the increase noted during 1984 and 1985 may represent normal fluctuations rather than a trend towards increasing turbidity in Hillsborough Bay.

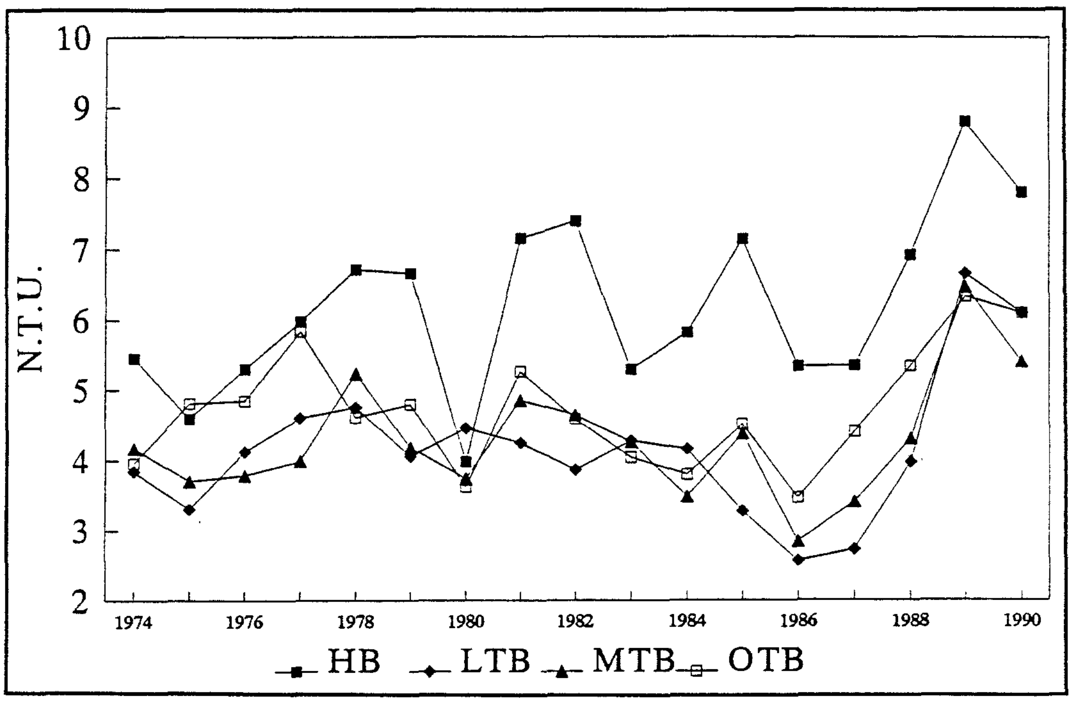

Figure 3. Tampa Bay: turbidity, 1974-1990 annual means by area.

In 1986, turbidity decreased to the lowest level for the period of record in all areas of the bay except for Hillsborough Bay, which also showed a marked reduction in turbidity from 1985. In 1987, the turbidity level in Hillsborough Bay remained low and essentially unchanged from 1986. The three other areas of the bay showed moderate turbidity increases in 1987 relative to 1986.

In 1988 and 1989 turbidity values continued to increase. For Hillsborough Bay and Tampa Bay values were at the highest average level in 1989 for the period of record. Although the 1990 annual averages for turbidity of the major subdivisions of Tampa Bay were lower than 1989, the averages remain higher than those of the mid-1980s.

The variable nature of the turbidity values from year to year is the result of several factors such as dredging (the Tampa Harbor Deepening Project), the occurrence of algae blooms (particularly in Hillsborough Bay), and runoff. Resuspended sediment and particles caused by wind driven waves in the shallower areas and prop wash by the large ships in the deeper portions of Tampa Bay are also contributing factors.

Effective Light Penetration (Secchi). In 1988, among stations deep enough to accurately determine annual average effective light penetration (Secchi), readings ranged from 32" in Hillsborough Bay to 141" in Lower Tampa Bay near the Skyway Bridge. All stations in McKay Bay and Hillsborough Bay had annual averages that were less than 70". Several stations in Old Tampa Bay also had annual averages less than 70". Light penetration generally improved toward the mouth of Tampa Bay averaging greater than 90" at most stations in Middle and Lower Tampa Bay. The best single light penetration measurement taken during 1988 was 240" at station 93 in lower Tampa Bay, near Egmont Key.

In 1989, annual averages for effective light penetration ranged from 38" in Hillsborough Bay to 133" in Lower Tampa Bay near the Skyway Bridge. All stations in McKay Bay and much of Hillsborough Bay averaged less than 50". The western section of Old Tampa Bay, including the Largo Inlet and Cooper's Bayou, also averaged less than 50" in 1989. Light penetration generally improved toward the mouth of Tampa Bay, averaging 110" or more. The best light penetration measurement taken during 1989 was 276" taken at station 23 in Lower Tampa Bay.

Figure 4 depicts trends for effective light penetration in Tampa Bay by comparing the annual averages of four areas of the bay from 1974 through 1989. The graph shows that over the years Hillsborough Bay has consistently had the poorest light penetration. All areas of the bay had decreasing effective light penetration from 1974 to 1979. The effective light penetration trend began improving in 1980 for all areas of the bay. Hillsborough Bay has had improving light penetration for each year since 1980 until 1985 when a slight decrease was measured. Light penetration continued to increase through 1988 in Hillsborough Bay. Lower Tampa Bay has had the greatest rate of improvement for light penetration, with significant increases occurring in 1984, 1985, and 1986. Light penetration values in 1988 were quite similar to 1987 levels in Lower Tampa Bay; however, a decrease was observed in 1989. Light penetration showed little change in Old Tampa Bay during 1988 and decreased slightly in 1989. Middle Tampa Bay showed a large increase for light penetration in 1986; the following years a decline in light penetration has occurred, although the value for Middle Tampa Bay in 1987 was higher than most other years. In 1988 and 1989 the trend indicates a slight decrease in water clarity in all areas of the bay relative to the mid-1980s. Overall effective light penetration has increased during the period of record.

Bacteria

Ambient water samples are collected each month from mid-depth and analyzed for total and fecal coliform. The purpose of bacteriological analyses is to determine suitability of the water for recreational purposes, to monitor the impact of domestic and industrial waste effluent discharges, to help determine stormwater impacts, to identify the sources of bacterial pollutants, and to evaluate the effectiveness of the pollution abatement program.

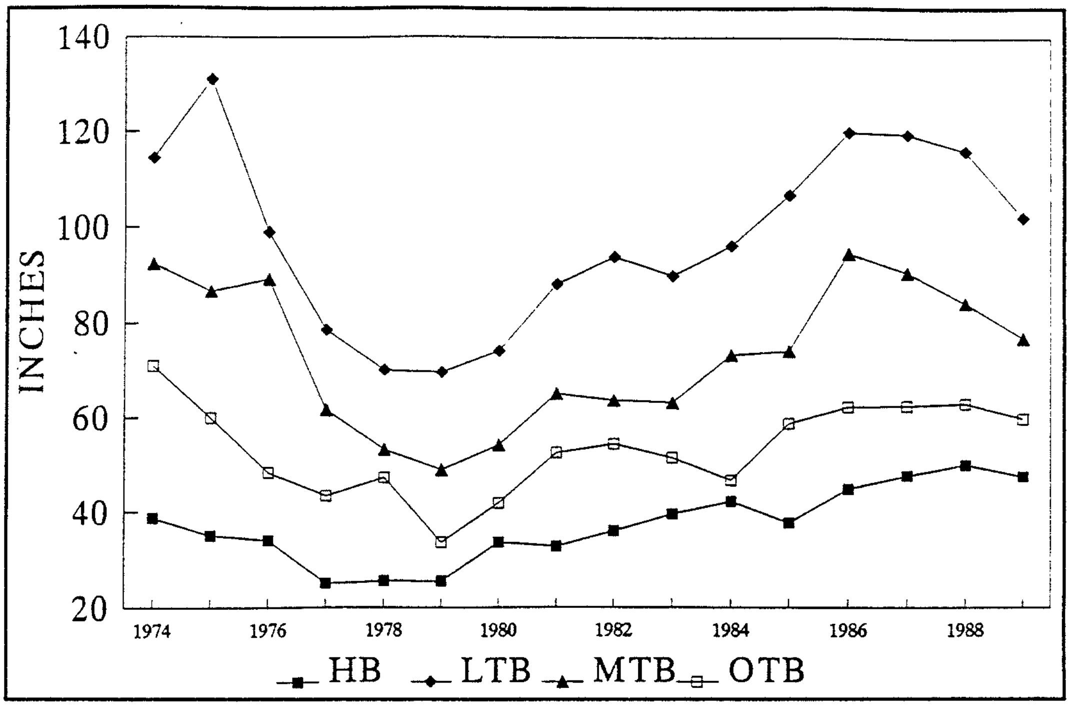

Figure 4. Tampa Bay: Secchi, 1974-1989 annual means by area.

Coliforms are widely distributed in nature and occur not only in human feces but in other media, such as sewage, surface waters, soils, and vegetation. They enter the natural waters directly through fecal discharge or via stormwater runoff, flooding, or inadequate sewage treatment. The presence of high numbers of coliform bacteria is deemed to be indicative of the presence of potentially harmful pathogenic micro-organisms. Coliforms tend to be more persistent in the aquatic environment; their survival rate depends on a variety of environmental factors. As a group, coliforms are considered harmless and often helpful to the digestion and vitamin synthesis in the body. However, some coliforms, such as *Escherichia coli* strains, are known to cause enteritis, while others may combine with fecal streptococcus strains to cause mild genito-urinary tract infections (Standard Methods 1989).

Fecal coliforms are a subgroup of the total coliform group. Their presence in water specifically indicates fecal waste contamination by warmblooded animals. These organisms occur relatively infrequently except in association with fecal pollution. Survival of the fecal coliform group is shorter in environmental waters than for the coliform group as a whole. It follows, then, that high densities of fecal coliforms are indicative of relatively recent fecal pollution. Fecal coliforms generally do not multiply outside the intestines of warmblooded animals. The major species in the fecal coliform group is *Escherichia coli*, and represents the possible presence of enteric pathogens.

The 54 bay stations and 35 tributary stations in the Water Quality Monitoring Network were analyzed for total and fecal coliforms. Selected stations were also routinely analyzed for fecal streptococcus. Water samples were collected from mid-depth and analyzed for total and fecal coliform utilizing the membrane filter method.

The annual averages for total and fecal coliforms were calculated for each station. During 1988, four bay stations had annual averages greater than 100 total coliform colonies per 100 ml of water. These same four stations also had fecal coliform concentrations greater than 100 colonies per 100 ml. The stations with elevated annual averages were #2 (mouth of the Hillsborough River), #74 (mouth of the Alafia River), #60 (north of Courtney Campbell Causeway), and #62 (Courtney Campbell Causeway). In 1989, only two stations, #2 and #74, averaged greater than 100 total coliform colonies and again these same two stations had fecal coliform concentrations greater than 100 colonies per 100 ml.

The trend graph for total coliform bacteria, Figure 5, shows the average total coliform level for the four largest subdivisions of Tampa Bay from 1974 to 1989. During the 16 years presented in the graph the trend has been a dramatic reduction in total coliform. The most significant reduction occurred in Hillsborough Bay in 1980.

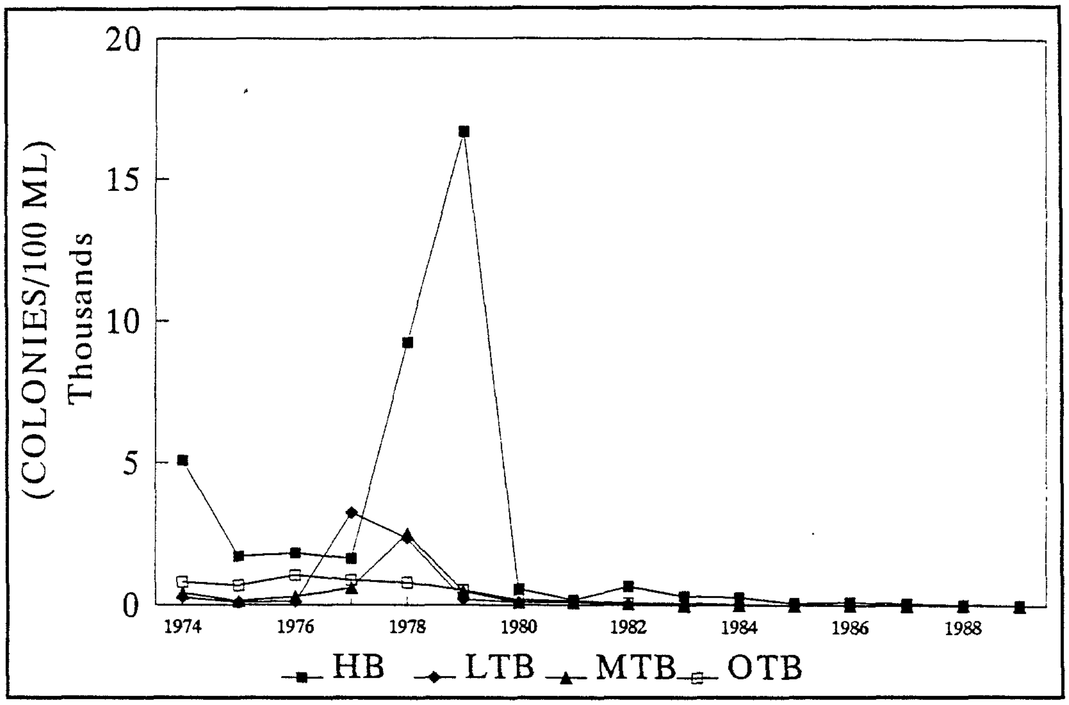

Figure 5. Tampa Bay: total coliform, 1974-1989 annual averages by area.

Chlorophyll

Chlorophyll analysis is useful as an indicator of the general ecology of a body of water because chlorophyll is an indirect measure of the quantity of planktonic algae present. Plankton algae are usually microscopic and often single-celled organisms that live suspended in the water column. The life cycles of these organisms can respond quickly to environmental conditions. In this way, chlorophyll can be used as an indicator of short and long term phenomena such as nutrient enrichment. Although planktonic algae are the basis of the food web, they can also be directly or indirectly responsible for fish kills, odors, discoloration of waters, and the reduction of water clarity.

Chlorophyll is not an absolute indicator of planktonic biomass since some species do not contain chlorophyll and when chloroplasts are present, they vary in number, size and pigment content per cell. Light, nutrients and other factors also influence the quantity of chlorophyll per cell so that their horizontal and vertical distribution in a body of water becomes important. Despite these variables and limitations, chlorophyll determinations are a useful indicator of phytoplankton population (Standard Methods 1989).

The amounts of chlorophyll *a*, *b*, *c* and total chlorophyll have been measured since 1972. Water samples are collected from mid-depth and analyzed using the trichromatic method, (Standard Methods 1989).

In 1988, annual average chlorophyll *a* concentration ranged from 1.8 µg/l near Egmont Key at the mouth of Tampa Bay to 19.1 µg/l in McKay Bay. In 1989, the highest annual average chlorophyll *a* concentration was 28.9 µg/l at station 8 in Hillsborough Bay near the mouth of the Alafia River; the lowest average chlorophyll *a* concentration was 3.1 µg/l inside Egmont Key at the mouth of Tampa Bay.

During 1988 and 1989, chlorophyll *a* concentrations throughout the bay remained at levels significantly lower than the early 1980s. In 1988 and 1989, most of Tampa Bay averaged less than 10 µg/l of chlorophyll *a*. The highest concentrations of chlorophyll *a* were in upper Hillsborough Bay and in the Largo Inlet area of Old Tampa Bay.

Figure 6 depicts yearly trends in chlorophyll *a* concentrations in four areas of Tampa Bay by comparing annual average chlorophyll *a* from 1974 through 1990. The graph shows that during the period of record, Hillsborough Bay has consistently had the highest chlorophyll *a* concentrations. This condition correlates with the numerous algae blooms observed in Hillsborough Bay. Old Tampa Bay and Middle Tampa Bay have not experienced algae bloom problems and the graph shows the chlorophyll *a* levels in these areas were consistently lower than Hillsborough Bay during the 1970s and early 1980s. Since 1982, the chlorophyll *a* concentration in Hillsborough Bay has declined. During recent years, the annual average for chlorophyll *a* in Hillsborough Bay has been much closer to the levels measured in Middle Tampa Bay and Old Tampa Bay. Lower Tampa Bay receives good flushing by the Gulf of Mexico and consequently has the lowest chlorophyll *a* concentration in the bay. Linear regression analysis indicates that chlorophyll *a* concentrations have declined in all areas of the bay, most dramatically in Hillsborough Bay.

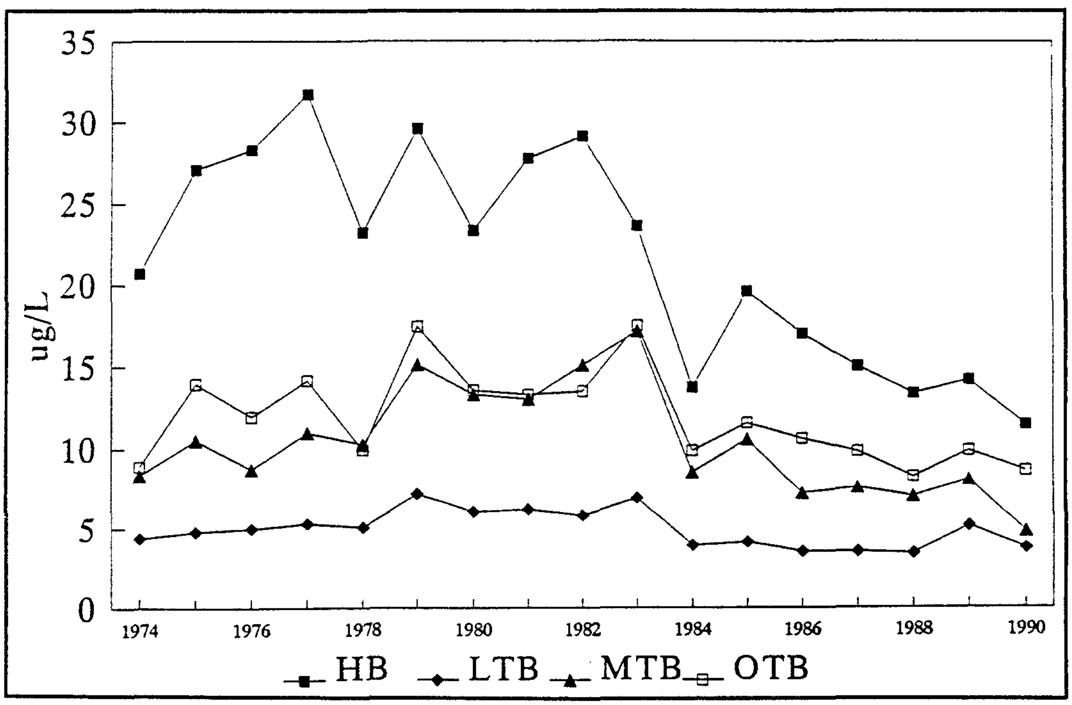

Figure 6. Tampa Bay: chlorophyll a, 1974-1990 annual means.

Yearly averages for chlorophyll *a* tend to relate to rainfall amounts. High rainfall amounts in 1979 and 1983 resulted in correspondingly higher chlorophyll concentrations. Similarly, lower chlorophyll concentrations were recorded in the years with lower rainfall amounts. However, the pattern is not absolute; other variables are involved. Normal rainfall was measured in 1988, while 1989 was a below-average rainfall year. However, chlorophyll *a* concentrations for all areas of the bay were greater in 1989 than 1988. The rainfall amount was well below normal in 1990 and correspondingly a decrease in chlorophyll *a* was measured in each of the four major subdivision of Tampa Bay.

General Water Quality Index

It is generally agreed among the scientific community that the primary water quality issue for Tampa Bay is eutrophication (high nutrient loading) resulting in increased algae growth, reduced light penetration and an unstable, often very low, oxygen environment (Lewis and Estevez 1988). The water quality index was primarily developed with this issue in mind.

It is important to bear in mind that the water quality index used here is not intended to be an absolute measure of conditions in Tampa Bay, but rather, simply as an index for quick assessment of water quality. Each parameter must be considered by itself to actually assess its role and influence in the bay's water quality.

The method used to generate the water quality index utilizes fixed scales for the parameters considered, allowing the comparison of data from one year to the next, hence, the development of a general water quality trend for Tampa Bay and its various subunits (Dunnette 1979). The results are expressed as water quality index (WQI) points. The scale represents discrete values in a relative relationship. The greater the WQI, the better the water quality and conversely, a lower WQI represents poorer conditions.

The index is an aggregate value of several parameters, combined in such a manner that the parameter's relative environmental significance is a factor in the final WQI value. The parameters incorporated into this index are dissolved oxygen, chlorophyll *a*, total coliform, biochemical oxygen demand, total phosphorus, total Kjeldahl nitrogen, and effective light penetration. For each parameter a scale of "good to bad" has been devised and subindex points are assigned. Each subindex value is multiplied by the parameter's relative environmental significance and combined with the other subindex values to produce the final WQI which will be in the range of 1-100 points. A score of 100 points represents the highest water quality possible.

In 1988, annual WQI values ranged from 52.7 points at the mouth of the Alafia River to 92.8 points in Lower Tampa Bay near Egmont Key. The lowest monthly WQI value was 41.3 points at station 74 in the Alafia River near the Highway 41 bridge; the highest monthly WQI value was 96.2 points at station 93 in Lower Tampa Bay.

In 1989, WQI values at most stations were quite similar to the 1988 values. The highest annual water quality index value was 92.5 points at station 93 in Lower Tampa Bay. Again, the lowest annual WQI value, 53.5 points, was at station 74 at Highway 41, near the mouth of the Alafia River. The lowest monthly WQI was 31.0 points at station 74. The highest monthly WQI was 99.5 points at station 93 in Lower Tampa Bay near Egmont Key.

Generally, the higher water quality index values are found in Lower and Middle Tampa Bay. Lowest water quality values are in Hillsborough Bay near the mouth of the Hillsborough and Alafia Rivers, and in the vicinity of the harbor including East Bay, McKay Bay and Seddon Channel. The index also indicated lower water quality at the station in Old Tampa Bay near the Baycrest/Dana Shores area, north of the Courtney Campbell Causeway.

A major attribute of this type of index is that it allows for comparing water quality from year to year. The lack of total Kjeldahl nitrogen data prior to 1980 restricts the time interval to which the water quality index can be applied. The trend graph, Figure 7, shows that since 1981 water quality has improved in each of the four major subdivisions of Tampa Bay.

Although Hillsborough Bay has the poorest water quality in the bay, the WQI derived for this subdivision has consistently increased for the period of record. While WQIs derived for Hillsborough Bay were virtually identical in 1988 and 1989, (71.3 points), over the 9-year period from 1981 through 1989, the WQI for Hillsborough Bay has increased by 16 points. The 1988 and 1989 WQIs for Hillsborough Bay represent a 3 point increase over the 1987 WQI.

The improvement in Hillsborough Bay is reflected by a 9 point increase in the WQI for Middle Tampa Bay. Old Tampa Bay and Lower Tampa Bay have each registered approximately a 7 point increase during the interval.

During 1987, slightly lower WQI values were derived for Lower, Middle, and Old Tampa Bay. The slight decline in water quality might be attributed to the increased rainfall of 1987, or may reflect a fluctuation within the normal range of values for these areas of the bay.

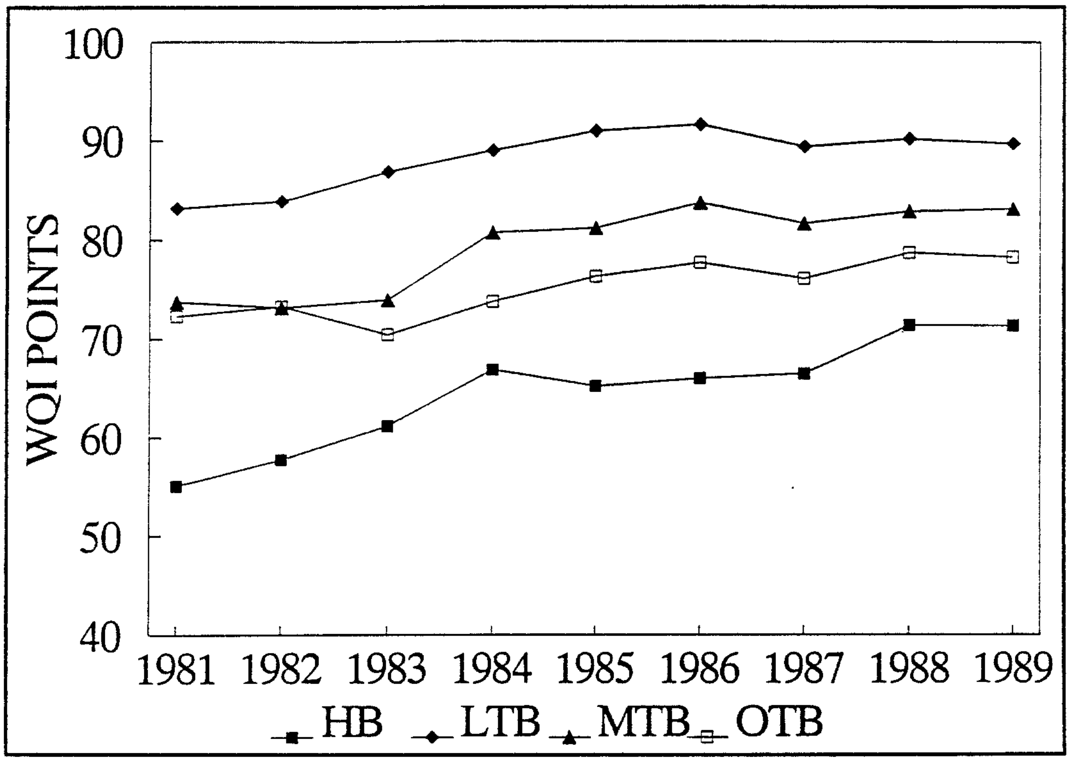

Figure 7. Tampa Bay: general water quality, 1974-1989 annual values by area.

Coastal Streams of Northwest Hillsborough County:
Double Branch Creek, Channel A, Rocky Creek, Sweetwater Creek

These are four small coastal tributaries to Old Tampa Bay. The sampling stations are relatively short distances from the open water of Old Tampa Bay. Hence, all of these stations have brackish water; the data indicates that in recent years, 1988 through 1990, salinity has increased at all stations (Boler 1990).

Double Branch Creek is located in western Hillsborough County, near the county line. Station 101 is monitored from the Hillsborough Avenue bridge. The station is just downstream from the confluence of the two branches. Until recently the headwaters of this stream have drained improved pasture and undeveloped land. This area is currently being developed for residential use. The watershed has a relatively high water table.

Channel A (Station 102) is monitored from the Hillsborough Avenue bridge. This system was channelized in the late 1960s and early 1970s as part of a grand scheme of drainage systems that was planned for Town 'n' Country and northwest Hillsborough County. A flow control structure, maintained and operated by the Southwest Florida Water Management District, serves as a salinity barrier. Station 102 is located about a half mile downstream of the structure and approximately two miles from the open water of Old Tampa Bay. The water tends to have the most consistent salinity range of these four streams. Upstream of the flow control structure, Hillsborough County operates the River Oaks Advanced Waste Treatment (AWT) facility, which discharges 10 million gallons per day of AWT effluent. Channel A will be receiving AWT effluent from two more large regional wastewater treatment facilities that will soon be on line.

Station 103 represents EPC's monitoring of lower Rocky Creek. This station is located at Hillsborough Avenue in the Baycrest area. EPC has another station farther upstream at Waters Avenue. At Station 103, the stream is relatively broad and shallow. It is the least saline of the four streams. This stream also has a flow control structure or salinity barrier located about 1.5 miles upstream of Station 103. The headwaters of Rocky Creek drain the area generally located between Dale Mabry Highway and Gunn Highway. Hillsborough County operates a regional AWT facility—Dale Mabry North—that currently discharges through a series of retention ponds in Carrollwood Village into Brushy Creek and ultimately into Rocky Creek. The County is currently cross-connecting this facility with the River Oaks AWT facility and the effluent will be redirected to Channel A.

Sweetwater Creek, station 104, is monitored at Memorial Boulevard. Downstream of this station is the Dana Shores subdivision. In this area, the creek has been channelized and encased in vertical concrete seawalls. The hardened shoreline ends just upstream of Memorial Boulevard. Sweetwater Creek receives about 0.5 million gallons per day of effluent from the Florida Cities Water Utilities Wastewater Treatment Plant.

Total Coliform

Total coliform, as indicated by annual averages (Figure 8a), has declined in all streams, especially in Sweetwater Creek. From 1977 through 1980, Sweetwater Creek had annual averages greater than 20,000 colonies per 100 ml of water. The maximum annual average of record occurred in 1980, which was 45,000 colonies per 100 ml. The 1990 annual average for Sweetwater Creek is 3050 colonies per 100 ml. Over the period of record, Rocky Creek and Double Branch Creek have exhibited very similar patterns. Despite improving water quality, with respect to bacteria, only Channel A has had annual averages that meet the safe swimming standards.

Effective Light Penetration (Secchi)

All of these streams exhibit a definite trend toward improving water clarity (Figure 8b). As with bacteria, the greatest improvement has occurred in Sweetwater Creek. In 1977, Sweetwater Creek had an annual average of 16" for the Secchi depth; the 1990 annual average was 48".

The fairly low Secchi values for Double Branch Creek are related to tannins, which give the stream its characteristic "black water". The tannins come from the numerous cypress domes in the drainage basin and floodplain of the creek.

Chlorophyll a

Channel A has consistently had the highest chlorophyll *a* values and has exhibited only moderate variability (Figure 8c). Sweetwater Creek has shown high variability;

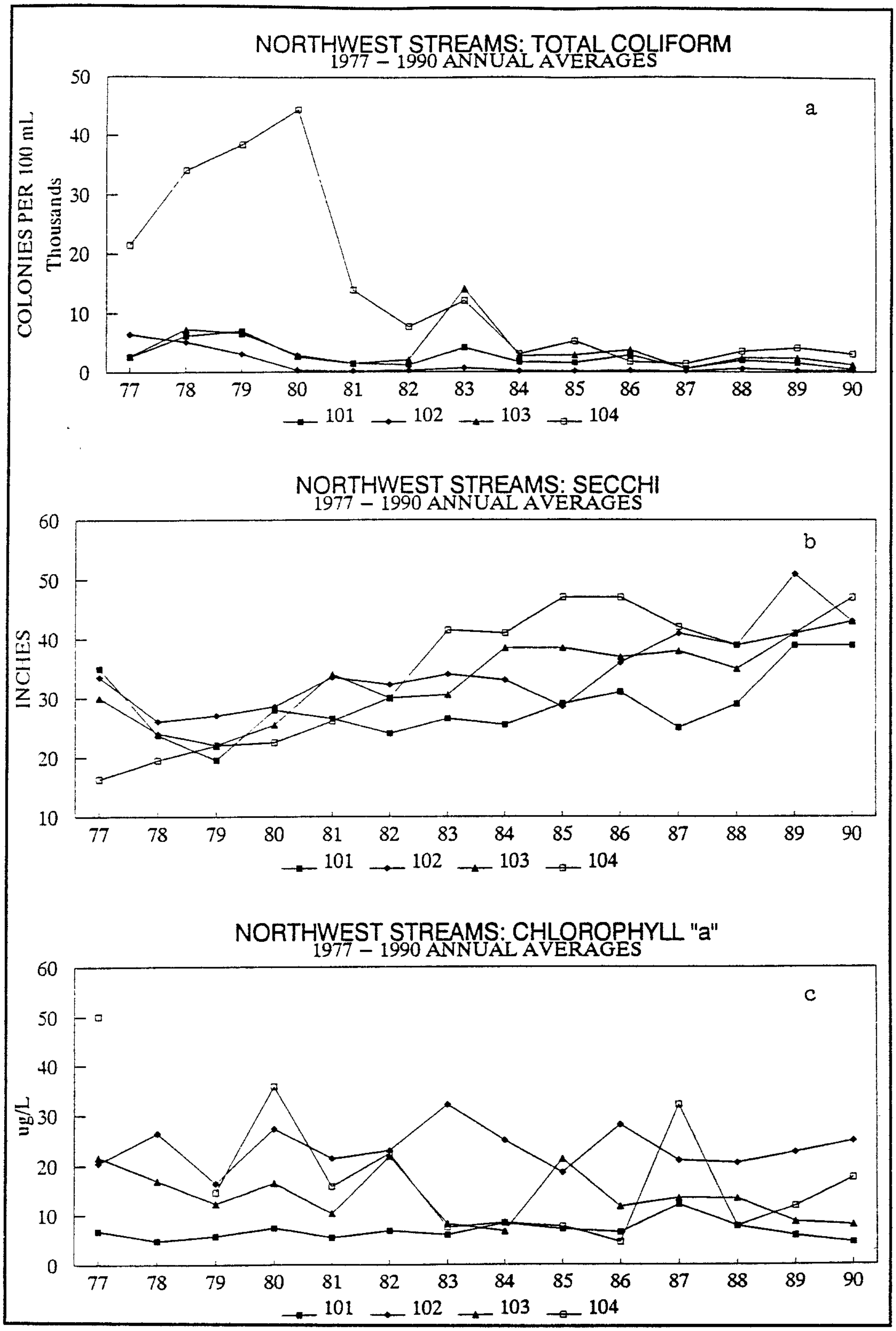

Figure 8. Northwest streams: a) total coliform; b) Secchi; c) chlorophyll a, 1977-1990 annual averages.

in 1987 the annual average was 37 µg/l. Double Branch Creek has had the lowest and most consistent annual average chlorophyll of these streams, probably as a consequence of low light penetration, the dark colored water, and the consistent flow in the creek.

Hillsborough River

The Hillsborough River provides the largest flow of freshwater to Tampa Bay. This is a river with a "split personality", caused by the dam located near Rowlett Park in north Tampa. Originally built to generate electricity, the dam now provides the drinking water reservoir for the City of Tampa. The water quality issues above the dam are quite different from those downstream of the dam. The lower Hillsborough suffers impacts associated with urbanization such as stormwater runoff and reduced flushing caused by the diversion of water for potable uses. The upper Hillsborough River drainage basin is largely undeveloped or used as rangeland.

The annual averages of various paramenters for four stations located on the Hillsborough River are presented graphically in Figures 9-12. Stations below the dam are #137 at Columbus Avenue and #105 at 22nd Street near Rowlett Park; above the dam, the stations are #106 at Fowler Avenue Bridge and #108 at Highway 301 at Hillsborough River State Park. The parameters presented are biochemical oxygen demand, effective light penetration (Secchi), total coliform, and total nitrogen.

Biochemical Oxygen Demand

Generally, the upper Hillsborough has lower biochemical oxygen demand (BOD) than the lower Hillsborough River (Figure 9). Station 106 at Fowler Avenue has lower values and a more consistent level of BOD, without large year-to-year fluctuations that are exhibited at the other stations. High variability is exhibited at the Columbus Avenue station and at station 108 in the upper Hillsborough River near the State Park.

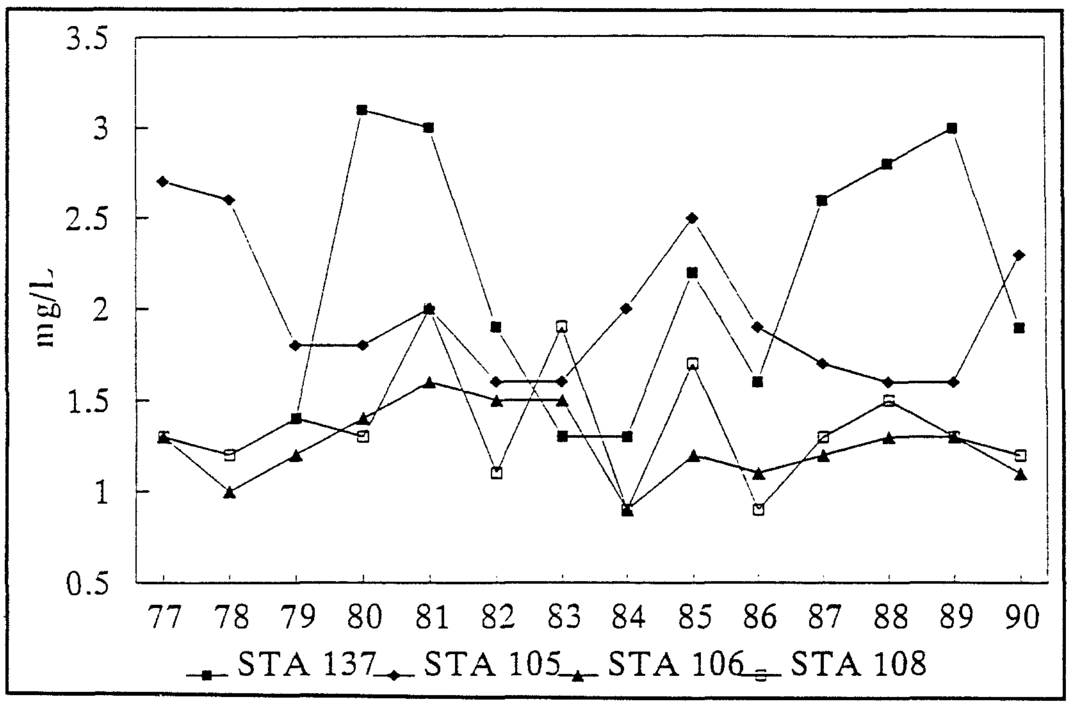

Figure 9. Hillsborough River: biochemical oxygen demand, 1977-1990 averages.

Station 137 showed a steady increase in BOD from 1986 through 1989, then a sudden decrease in 1990. During the same period, at station 105 near Rowlett Park the reciprocal pattern occurred; BOD decreased from 1985 through 1989, then a large increase occurred in 1990. High values at the Columbus Avenue station in 1980 and 1981 reflect problems with sewage collection systems for the City of Tampa.

Effective Light Penetration (Secchi)

Water clarity throughout the Hillsborough River has improved from 1982-83 to present as indicated by increasing annual average Secchi values (Figure 10). Station 105, at Rowlett Park, has consistently had the best Secchi values. As a consequence of the heavy rainfall that occurred in 1979, all stations had reduced annual average Secchi values.

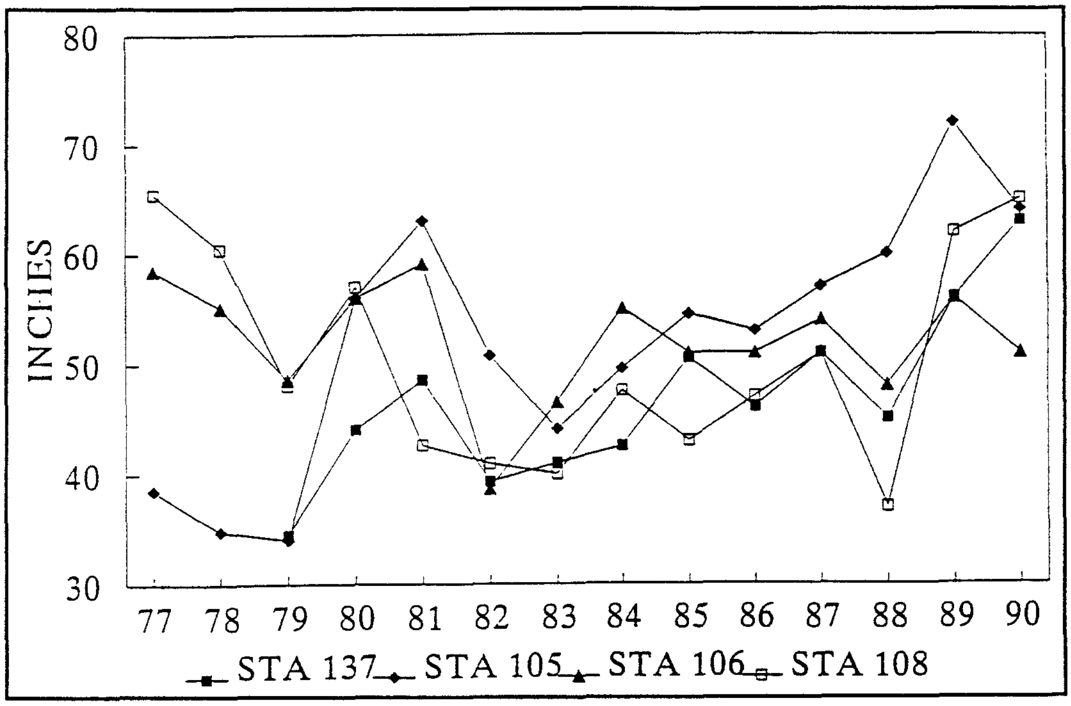

Figure 10. Hillsborough River: Secchi, 1977-1990 annual averages.

Total Coliform

For the period of record, all stations have shown a reduction in total coliform (Figure 11). Since 1980, the Hillsborough River at Fowler Avenue has had annual averages within the safe swimming standards. All stations had low annual averages for total coliform in 1990, most probably the consequence of the low rainfall.

The annual average at Columbus Avenue in 1979 was greater than 230,000 colonies per 100 ml of water. High values in 1981 are attributable to a failed sewage collection lift station which resulted in a large discharge of raw sewage to the Hillsborough River.

In the mid-1970s, a swimming pool was built at Hillsborough River State Park and the swimming area in the river was closed due to the high coliform values measured in the river. The total coliform concentrations at Station 108 have declined and annual averages are often within the state standards for swimming. However, individual total coliform samples, especially those during the summer months, are usually 4-5 times greater than the swimming standard.

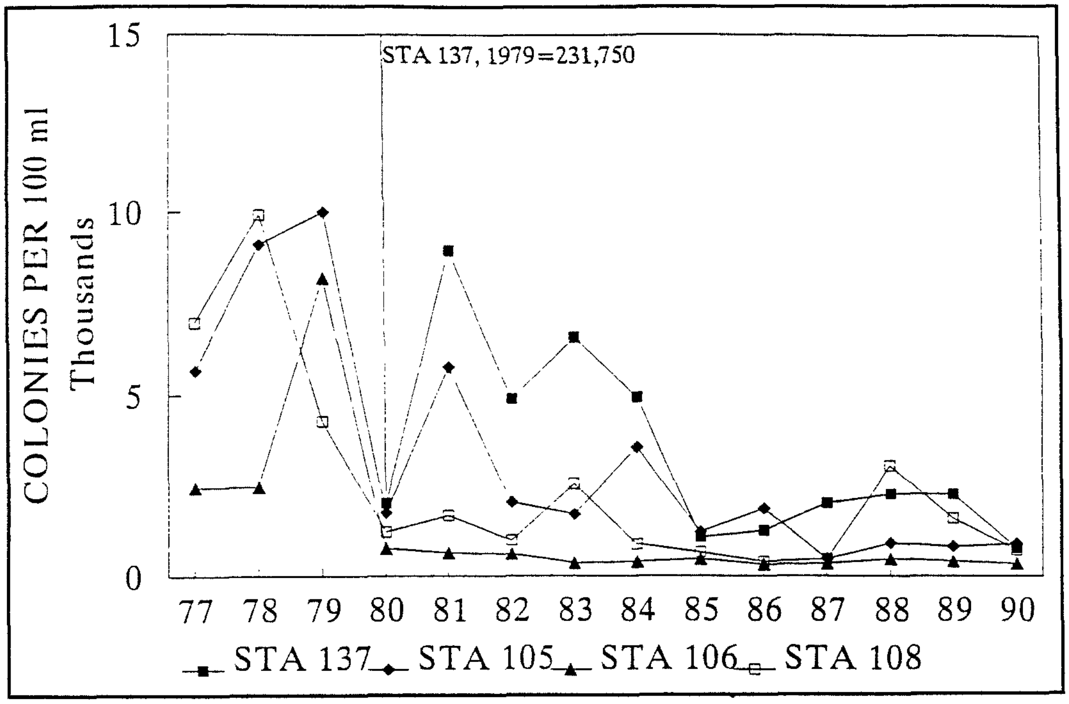

Figure 11. Hillsborough River: total coliform, 1977-1990 annual averages.

Total Nitrogen

Stations 137, 105,and 106, exhibit very similar annual averages values for total nitrogen (Figure 12). These values are typical of most Florida streams. However, the annual averages for station 108, near the Hillsborough River State Park, are appreciably higher; these values are higher than 80% of Florida streams. The annual averages for this station exhibit a remarkable sawtooth pattern—up one year, down the next year—which corresponds closely with the pattern of annual averages for BOD at the same location.

Lake Thonotosassa

The largest lake in Hillsborough County, Lake Thonotosassa has a surface area of 819 acres. It is one of the few large lakes in the County with public access; accordingly it receives a great deal of use, especially for fishing and water skiing. Lake Thonotosassa has been designated as a Surface Water Improvement and Management (SWIM) priority water body. Plans are in place to implement improvement strategies and projects.

Almost all of Lake Thonotosassa water enters by way of Baker Creek, which provides drainage from the south (North Brandon, Seffner, Mango) and receives Pemberton Creek which drains the area to the east of the lake (Plant City, Knights). There are two major point sources of pollution—effluent from Plant City STP and Treasure Isle, Inc. (Florida Snoman), a seafood processing and packing facility. Non-point source pollution includes agricultural runoff from poultry and dairy farms as well as pastures and orange groves.

Lake Thonotosassa is eutrophic (Cowell et al. 1975, Carlson 1977, Dawes et al. 1987, Hand et al. 1988) and is notorious for the massive fish kill that occurred in 1969. The lake still suffers periodic fish kills as a result of a more or less permanent blue-green algae bloom.

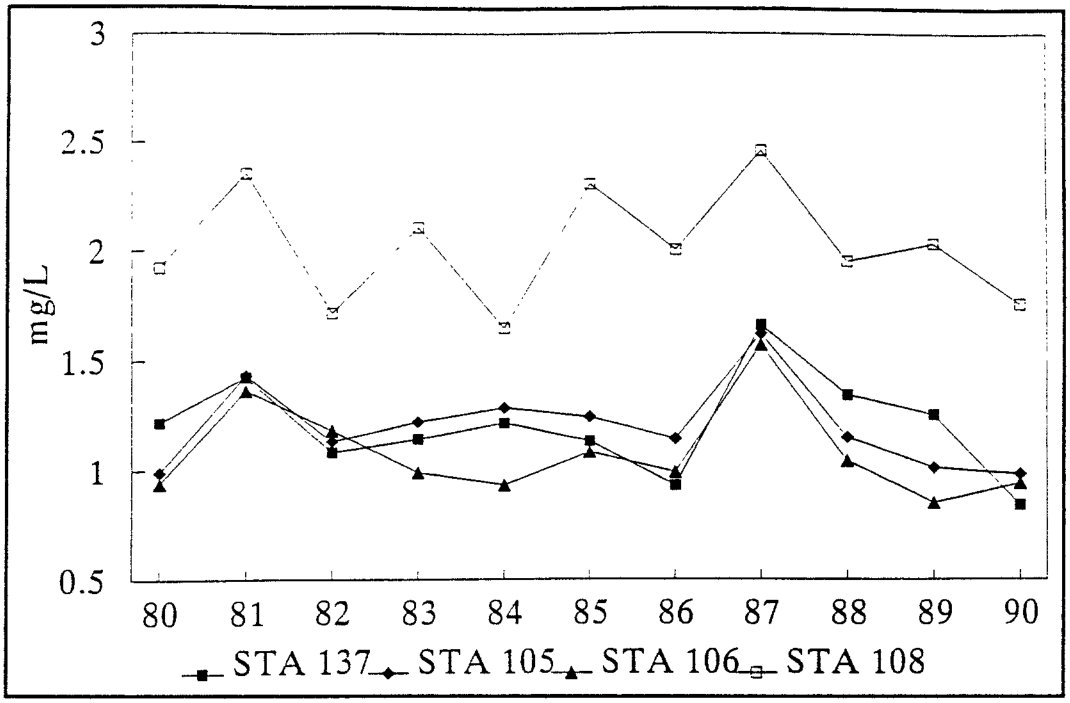

Figure 12. Hillsborough River: total nitrogen, 1980-1990 annual averages.

Nutrients

Nitrogen has generally averaged about 2.0 mg/l during the period of record (Figure 13a). The higher annual averages correlate with the activities of the two major point source contributors. Phosphorus has averaged about 0.5 mg/l during most of the record. In recent years the annual average for total phosphorus in the lake has been increasing. Water quality data from Baker Creek, just upstream from the lake, has also shown a definite increase in total phosphorus. These values are in the 80th percentile for all Florida lakes (Friedeman and Hand 1989).

Chlorophyll a

The high level of nutrients in Lake Thonotosassa supports an abundant population of phytoplankton. The annual averages indicate that chlorophyll *a* is increasing in the lake (Figure 13b). Floating blue-green algal masses were often observed during the 1990 sampling activities.

In 1986 the annual average chlorophyll *a* for the center of the Lake was 76.1 µg/l. At the mouth of Flint Creek, just upstream of the control structure, where water is exported from the lake, the 1986 annual average was 90.6 µg/l. Chlorophyll *a* values greater than 67 µg/l represent the 90th percentile for Florida lakes (Friedeman and Hand 1989).

There are a variety of ways to generate a trophic state index; they are based on the ratio or assessment of nitrogen, phosphorus, Secchi and chlorophyll levels (Carlson 1977; Cowell et al. 1975, Dawes et al. 1987, Hand et al. 1988). Regardless of the actual index used, Lake Thonotosassa is identified as eutrophic.

Dissolved Oxygen

Values are exactly as one would expect of a hypereutrophic lake—very high dissolved oxygen (DO) at the surface, and very low dissolved oxygen at the bottom (Figure 13c). Surface oxygen values represent a supersaturated condition and annual averages have consistently been greater than 10 mg/l, with individual readings as high

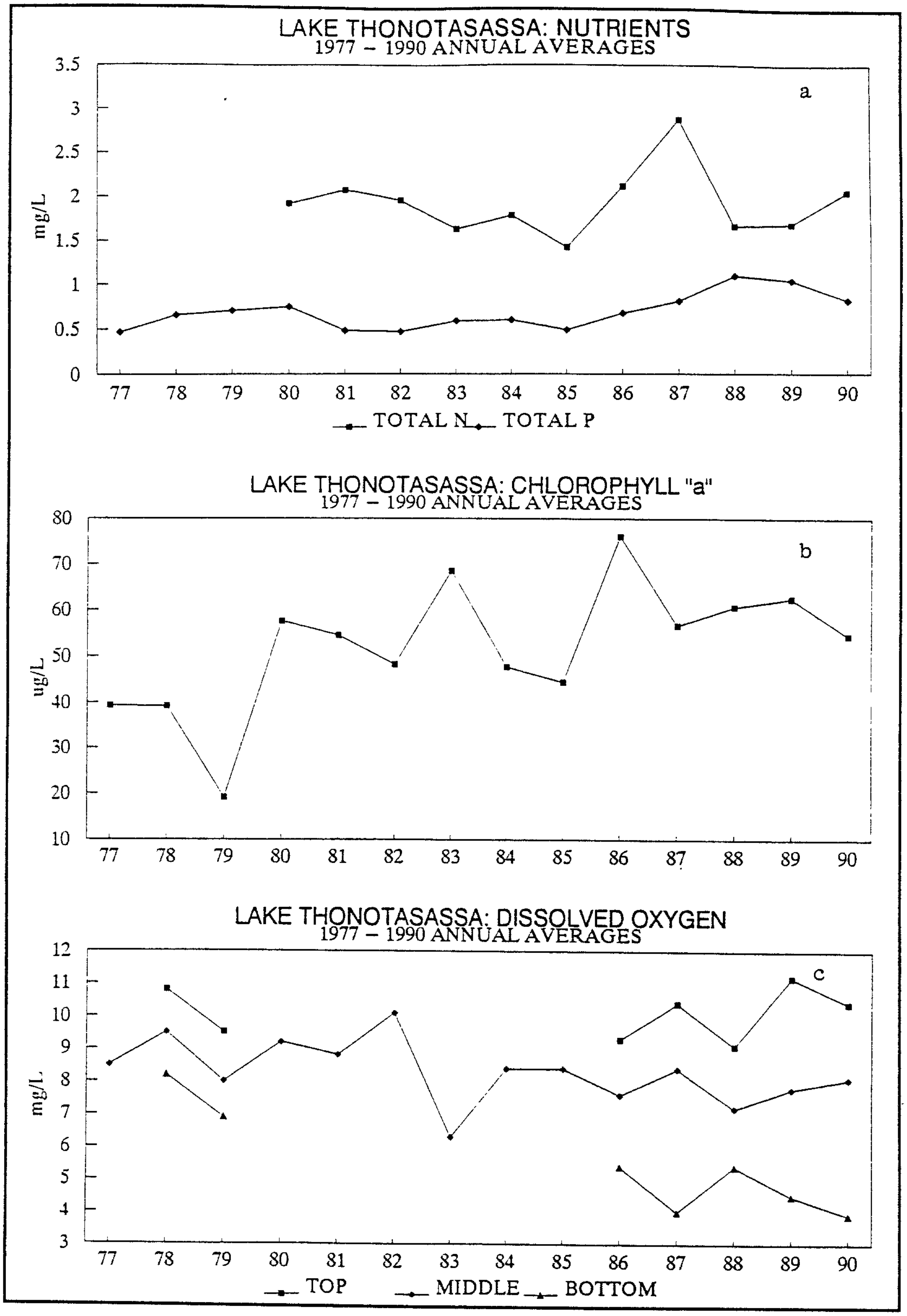

Figure 13. Lake Thonotosassa: a) nutrients; b) chlorophyll a; c) dissolved oxygen, 1977-1990 annual averages.

as 14 mg/l. During the summer months dissolved oxygen is usually less than 1.0 mg/l at the lower depths, usually 10.5′ to 11.0′ below the surface (2.0′ to 2.5′ above bottom); an anoxic zone exists for much of the summer. The sediments in the central part of the lake are made up of a dark, highly organic muck, 2′ to 3′ thick. This layer undoubtedly exerts a sizable oxygen demand.

In recent years, fish kills in the lake have occurred primarily in the juvenile shad population, hence the ratio of biomass to individuals is relatively low. These fish are plankton feeders and form large schools; they are especially sensitive to low dissolved oxygen. The high respiration rate of the blue-green algae population, coupled with a low rate of photosynthesis that can be caused by several days of overcast skies with low light, is usually sufficient to depress oxygen levels and cause fish kills.

Palm River

The Palm River was one of the four natural rivers in Hillsborough County. In the 1960s, a grand scheme for flood control—the consequence of Hurricane Donna—was begun. It included a massive channelization of the Palm River and Six-Mile Creek; the system now bears little resemblance to a small coastal river. Less than a half mile north of Highway 60, a flow control structure/salinity barrier has been constructed. The aquatic system upstream of this structure is now called the Tampa Bypass Canal. The Tampa Bypass Canal continues northerly and intersects with the Hillsborough River near the point where Interstate 75 crosses the river. Here, another control structure has been built to divert water from the Hillsborough River when deemed necessary. Designed and contracted by the Army Corp of Engineers, it was turned over to the Southwest Florida Water Management District to be maintained and operated; the system has been seldom used.

Currently, we monitor at two stations in the brackish water and at two stations upstream of the control structure in the Tampa Bypass Canal. The data for selected parameters at station 110 is presented. Station 110 is located in the Palm River at Highway 60, just downstream of the flow control structure. Water depth is nominally 15′ depending on tidal conditions; little net flushing occurs.

Chlorophyll a

The annual averages for chlorophyll *a* are relatively high, typically about 30 µg/l, as indicated in Figure 14. In 1990, the annual average was 33.2 µg/l.

Since 1986, annual averages of chlorophyll *a* have fluctuated widely, reflecting an extremely dynamic phytoplankton community. In 1986, the annual average was 55.4 µg/l; in 1988 the annual average was 48.6 µg/l. In these years, significant populations of *Schizothrix calcicola* were identified.

In 1989, an apparent "crash" of the phytoplankton population occurred, as evidenced by an annual average for chlorophyll *a* of 5.7 µg/l. The most identified planktonic species was *Euglena elasticans*.

Effective Light Penetration (Secchi)

Annual averages for Secchi indicate a trend toward improving water clarity (Figure 14). In the late 1970s, the annual averages for the Secchi depth at this station in Palm River was about 24"; the 1990 value was 44.0".

Dissolved Oxygen

Dissolved oxygen levels are generally disparate vertically in the water column, as is often the case in poorly flushed systems with large or dynamic phytoplankton populations (Steidinger 1985). During the period of record, dissolved oxygen at mid-depth in the Palm River has failed to meet state standards for Class III waters (Figure 15).

Surface oxygen is highly variable. Typically, measurements are made in mid-afternoon at this station. Individual DO readings often exceed 10 mg/l.

Almost invariably, an anoxic condition exists in the lower part of the water column. As indicated by the graph, the mid-depth also shows low dissolved oxygen. It is common to measure bottom DO as less than 1.0 mg/l.

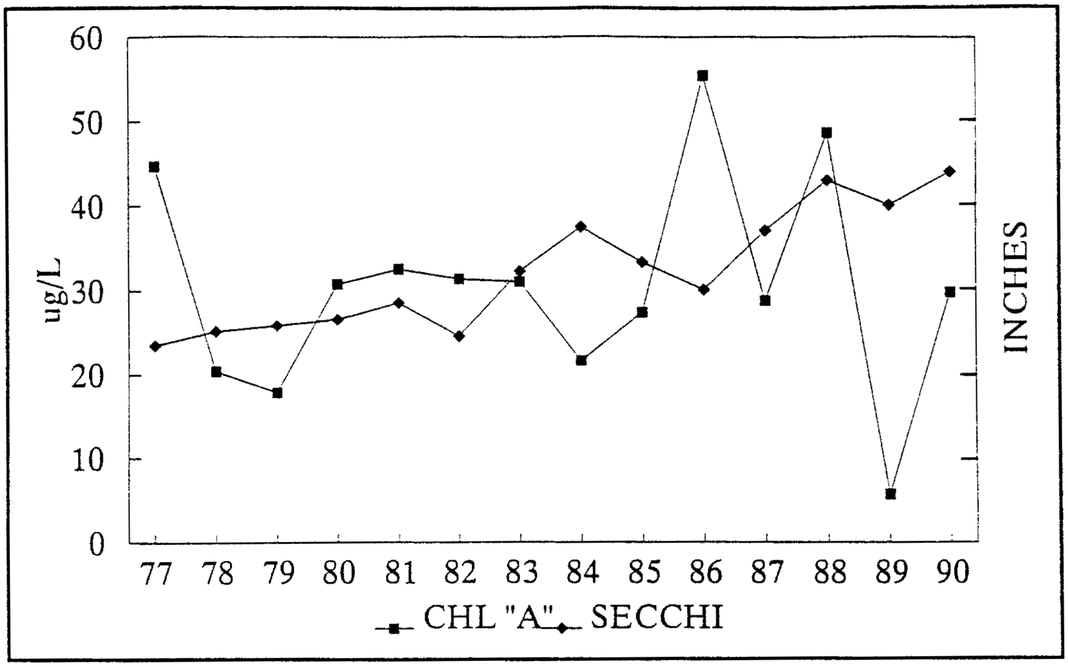

Figure 14. Palm River: chlorophyll a and Secchi, 1977-1990 annual averages.

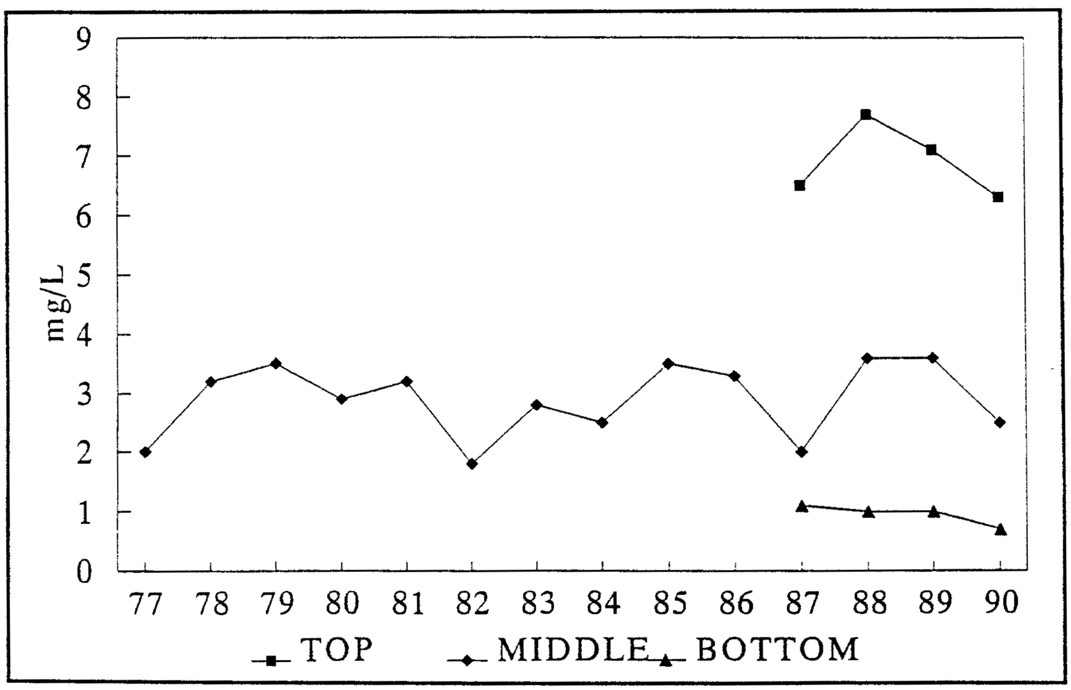

Figure 15. Palm River: dissolved oxygen, 1977-1990 annual averages.

Delaney Creek

Delaney Creek has been the "red-headed step-child" of Tampa Bay's tributaries. In its lower reaches, the creek continues to receive permitted industrial effluents and industrial stormwater runoff. The mouth of Delaney Creek receives a hot water discharge from TECO's Gannon facility. In addition, phosphate fertilizer loading terminals are located at and near the mouth of Delaney Creek. These facilities have been identified as a source of intense nutrient loading to Tampa Bay. Historically, a secondary lead smelting facility was located at Highway 41, adjacent to Delaney Creek. Operation of this facility impacted the creek with battery acid wastes and metals—lead, cadmium, and arsenic.

Slightly upstream from Highway 41, Delaney Creek continues to be the receiving water for the permitted industrial discharge from Nitram, Inc., a nitrogen fertilizer manufacturer. Over the years, the permitted discharge from this source has been reduced by two orders of magnitude. However, Nitram, Inc. continues to be permitted to discharge total nitrogen at a concentration of 7.0 mg/l and a rate of 50 lbs/day as monthly averages.

In the upper reaches, the creek has been channelized for drainage and flood control. Historically, the land has been used for rangeland and dairy farms. In addition, Black Gold, a commercial operation which stockpiled and composted cow manure for packaging and sale as fertilizer, was located adjacent to the creek. Poor management procedures and heavy rain events often resulted in this material being washed into the creek. This facility is no longer in operation.

Currently, the upper reaches of the Creek are under intense development; land use will be residential and for a regional shopping mall. In fact, the creek has again been rerouted and will be incorporated into the stormwater management system of the mall.

Nitrogen

In 1980, EPC began using a Technicon Autoanalyzer for nutrient analysis. Prior to 1980, analysis for nitrogen was done by wet chemistry methods; these methods allow for a large degree of analytical error and quality assurance is poor. Nitrogen data prior to 1980 are of dubious accuracy and are probably low by as much as one order of magnitude.

The 1980 value, which quality assurance data indicate are quite accurate, was 113.0 mg/l (Figure 16a). The 1990 annual average was 2.3 mg/l. This dramatic change is primarily attributed to improved pollution control practices at Nitram, Inc. The reduction in concentration also reflects a large reduction in loading rate. However, to meet the concentration limits of their current DER industrial permit, Nitram, Inc. dilutes its industrial effluent with freshwater. The elevated annual average in 1987 is attributed to a single event; a storage tank holding ammonium nitrate ruptured and an undetermined amount of product entered Delaney Creek.

Turbidity

The annual average for turbidity showed a sharp increase in 1983, up to 34 NTU; while values have declined since then, they remain elevated (Figure 16b). The 1990 annual average was 22 NTU. Historically, elevated turbidity was attributed to runoff from Black Cow and the dairy operations. More recently, turbidity has occurred as a consequence of land development, including the building of I-75.

Total Coliform

During the period of record, high coliform levels, both total and fecal, have been recorded in Delaney Creek. The 1976 annual average for total coliform was 299,000 colonies per 100 ml; the 1990 annual average for total coliform was 5,570 colonies per 100 ml (Figure 16c). While greatly reduced from earlier years, the more recent annual averages still represent high coliform pollution. The creek still drains a dairy pasture and is regularly visited by the livestock.

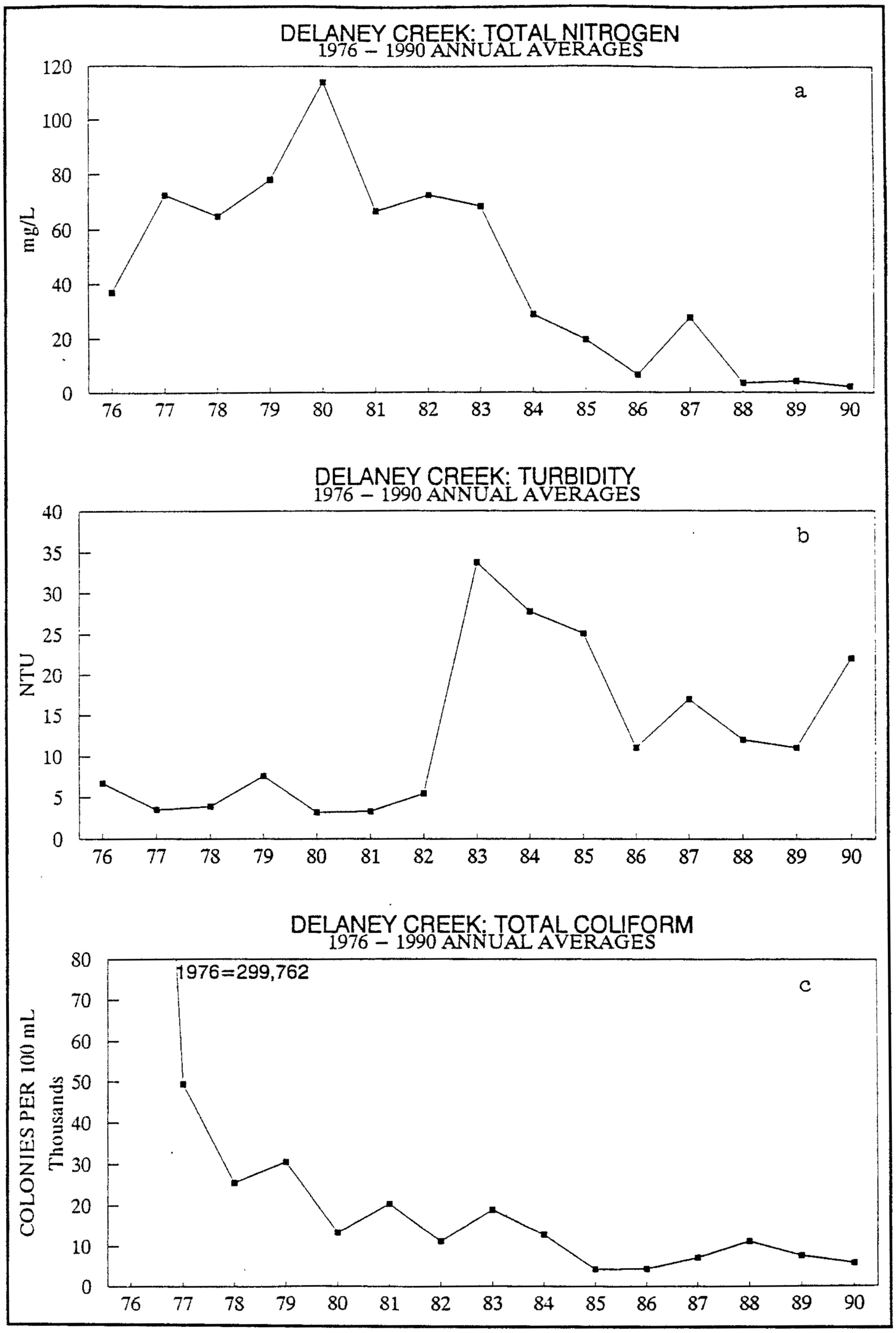

Figure 16. Delaney Creek: a) total nitrogen; b) turbidity; c) total coliform, 1976-1990 annual averages.

Alafia River

The headwaters of the Alafia River have suffered impacts from the phosphate industry since the early 1900s. The Alafia River has long been regarded as the most polluted of the major tributaries to Tampa Bay and though this distinction is probably true, it is more "stinging" than the river deserves.

The Alafia drains approximately 430 square miles in Hillsborough and Polk Counties. In eastern Hillsborough County, the Alafia River splits into two streams of relatively equal size, the North Prong and the South Prong. In the drainage basin of the South Prong, only mining and beneficiation occurs; in the basin of the North Prong, chemical/fertilizer manufacturing and processing occur. The Alafia has several smaller branches throughout its course, including Turkey Creek which has had poor water quality through the period of record (Boler 1990).

Data for the following stations on the Alafia River are presented in this paper: station 74 at Highway 41; station 114 at Bell Shoals Road; station 115 in the North Prong; and station 116 in the South Prong just upstream of their confluence. In addition to the stations listed here, EPC has a station on Turkey Creek at Highway 60 and a station well upstream on the South Prong, just inside the Hillsborough County line.

Phosphorus

Throughout the period of record, the North Prong has had appreciably more total phosphorus than the South Prong, in the range of five to six times greater (Figure 17a). In both 1988 and 1989, the annual average for total phosphorus in the North Prong was greater than 5 mg/l. The elevated values for total phosphorus measured at Bell Shoals station mirrors the influence of the North Prong's contribution to the system. At Highway 41, the concentrations for total phosphorus are approximately the same as those measured in the South Prong. If the concentration of total phosphorus in the North Prong could be reduced to a level similar to the South Prong, the net loading of total phosphorus to Hillsborough Bay would be reduced by more than 0.5 million pounds per year.

Biochemical Oxygen Demand

The freshwater stations have annual average BOD of 1.0 to 1.5 mg/l (Figure 17b). These values are typical of Florida streams. BOD has apparently declined in the mid-reaches of the river through late 1970s. Although the lower Alafia River has shown a slight increase in BOD since 1980, the annual average BOD has been about 3.0 mg/l at Highway 41. This value is typical for Florida estuarine water.

Total Coliform

Beginning about 1980 and continuing to present, a large decline in total coliform has been observed (Figure 17c). Annual averages for all stations range from 1000 to 1500 colonies per 100 ml. Prior to 1980, annual averages for total coliform were generally about 5000 colonies per 100 ml. In 1978 and 1979, the annual averages exceeded 10,000 colonies per 100 ml at some stations.

Little Manatee River

The Little Manatee River is the smallest of the four major tributaries to Tampa Bay and the least impacted. There are two small residential communities in the drainage basin but land use is primarily agricultural. The Department of Environmental Regulation has classified the Little Manatee River as an Outstanding Florida Water (OFW) from Highway 674 downstream to Tampa Bay. This designation affords the highest environmental protection to the river and prohibits activities in the drainage basin that would degrade water quality.

EPC'S Surface Water Quality Monitoring program has four stations in the Little Manatee River. The stations are #112 at Highway 41, #113 at Highway 301, #140 at

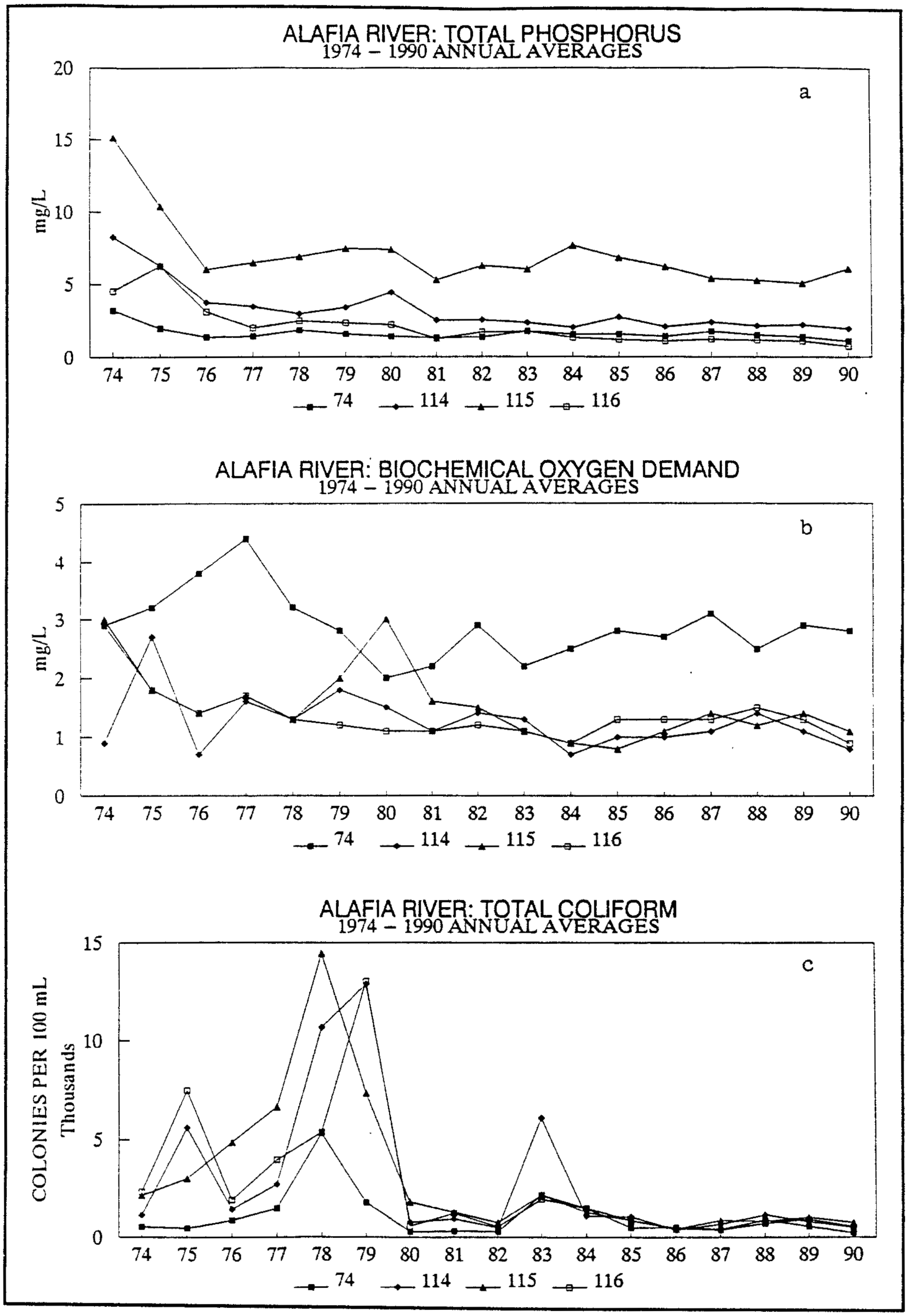

Figure 17. Alafia River: a) total phosphorus; b) biochemical oxygen demand; c) total coliform, 1974-1990 annual averages.

State Road 579, and #129 at Highway 674. No data are presented here; this tributary is discussed in detail elsewhere in this volume (Flannery et al., Peebles et al., Vargo et al.).

CONCLUSIONS

The data for water clarity parameters in Tampa Bay indicate a general north to south trend toward decreasing color, decreasing turbidity and increasing light penetration. The combined effect of these parameters indicate a general north to south trend toward increased water clarity and more favorable light climate. Over the entire period of record, water clarity has improved; however, in the past three years, the data indicates a moderate decline in water clarity. It is unclear if these data represent the normal variability of water clarity or if they represent the beginning of a trend of declining water clarity. Although the range of data for the tributaries is more variable, water clarity has improved in the tributaries.

During the 1970s, the high bacterial contamination that existed in Tampa Bay was largely attributed to inadequate or poorly operated sewage treatment plants discharging (point source pollution) into the area's coastal creeks. When these sewage treatment plants were taken off line or upgraded to AWT quality, a noticeable improvement in water quality, especially with respect to bacterial contamination, was observed. The reduced coliform bacteria population measured in the bay appears to have become established as the ambient condition.

The tributaries have shown a large reduction in bacteria during the period of record, but for the most part, the bacterial water quality has not changed much in the past several years. The tributaries are primarily affected by stormwater runoff resulting from rainfall (nonpoint source pollution), which is very difficult to predict or control, resulting in bacterial loading as well as other types of pollution.

Dissolved oxygen in many of the tributaries continues to fail to meet state standards. This is especially true at the lower levels of the water column in systems that are nutrient enriched or receive limited flushing.

During the period of record, chlorophyll *a* concentrations have declined and this has resulted in observable improvement in water quality. The greatest reduction in chlorophyll *a* occurred in Hillsborough Bay subsequent to the City of Tampa upgrading the Hookers Point sewage treatment plant to AWT standards.

The water quality in the tributaries remains a function of land use and the effectiveness of systems for pollution control and stormwater management. During the period of record of the Surface Water Quality Monitoring program, measurements confirm improved water quality in many of the major surface water systems including Tampa Bay. During the same period, regulatory efforts have been most focused primarily at point sources of pollution. Ambient surface water quality can be improved by upgrading treatment or eliminating point sources of pollution. Efforts to abate pollution caused by stormwater have been initiated, but remain limited in scope and application. With greater implementation and sophistication of stormwater management, additional improvement in water quality will probably be observed.

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STABLE CARBON AND OXYGEN ISOTOPE VARIATIONS IN WATERS OF THE TAMPA BAY ESTUARY

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done 5/7/99

ABSTRACT

The relationships between salinity and the $^{18}O/^{16}O$ ($\delta\ ^{18}O$ in ppt versus SMOW) in the water and the $^{13}C/^{12}C$ ($\delta\ ^{13}C$ in ppt versus PDB) in the dissolved inorganic carbon are being studied. For surface water $\delta\ ^{18}O$ is linearly related to salinity. At 0 ppt salinity the extrapolated $\delta\ ^{18}O$ is -2.2, reflecting the composition of precipitation in the Tampa Bay area. At high salinities of about 35 ppt, δ is about +1.6 ppt, reflecting the composition of open Gulf of Mexico surface waters. $\delta\ ^{13}C$ and salinity also show a linear relationship. At 0 ppt salinity, the extrapolated $\delta\ ^{13}C$ value is about -10 ppt which is somewhat higher than would be expected for dissolved inorganic carbon (DIC) resulting from the weathering of limestone (δ~0) with organic derived CO_2 [δ~-26]. This may be explained by isotopic exchange with atmospheric CO_2. At salinities approaching 35 ppt, $\delta\ ^{13}C$ values are about -2 ppt, again reflecting that of open Gulf waters and/or isotopic equilibration with atmospheric CO_2. These parameters are being used to get estimates for the exchange of water types in Tampa Bay and a better understanding of the recharge of deep aquifers in the Tampa Bay region.

INTRODUCTION

Over the past forty years the stable isotope compositions of carbon and oxygen in naturally occurring materials have been used to gain a better understanding of natural processes such as photosynthesis, petroleum formation, and the hydrological cycle. Relatively few stable isotope studies have been conducted in Florida and especially in the Tampa Bay area. This preliminary report on the $\delta\ ^{13}C$ of dissolved inorganic carbon (DIC) and $\delta\ ^{18}O$ of water as a function of salinity in the Tampa Bay estuary is intended to be the basis of more detailed subsequent studies designed to better understand the various sources and anthropogenic effects of carbon on Tampa Bay waters and information on the pathway of water during transport from the surface to underground aquifers. ($\delta\ ^{13}C$ and $\delta\ ^{18}O$ values in this paper are given in terms of the per mil deviations from the conventional PDB and SMOW standards, respectively.)

A simple representation of the processes which control carbon isotope compositions is given in Figure 1. It is based on the classical weathering reaction,

$$CaCO_3 + CO_2 + H_2O \dashrightarrow 2HCO_3^- + Ca^{++}.$$

$CaCO_3$ with a $\delta\ ^{13}C \approx 0$ reacts with organic derived CO_2 with δ~-26 (average composition of C_3 land plants) to produce HCO_3^- with a δ of ~-13 (one carbon from each reactant). The CO_2 may exchange with atmospheric CO_2 with a δ~-8 and shift the bicarbonate towards its equilibrium value of zero or it may exchange with the organic derived CO_2 and shift toward -18, the equilibrium value when gaseous CO_2 is -26. Generally the shift is towards the former rather than the latter. It should be noted that in this simple model $\delta\ ^{13}C$ of HCO_3^- may have a range of 18 ppt.

The $\delta\ ^{18}O$ composition of fresh water is controlled by the fractionation of oxygen isotopes during evaporation and condensation. This is nicely explained by the Raleigh distillation model given in Figure 2. At 25°C the $\delta\ ^{18}O$ of water vapor is about -9 ppt. The first condensate in equilibrium with the vapor has a $\delta = 0$; if this condensate is removed before re-evaporation and/or isotope exchange can occur, the remaining vapor becomes depleted in ^{18}O. Subsequent condensate is also somewhat depleted in ^{18}O. The net result of these processes is that precipitation is increasingly depleted in ^{18}O in going polewards, with arctic and antarctic precipation becoming as light as -40 ppt. Florida precipation is close to the tropical source of water vapor and shows only a slight depletion with a composition of about -3 ppt vs SMOW.

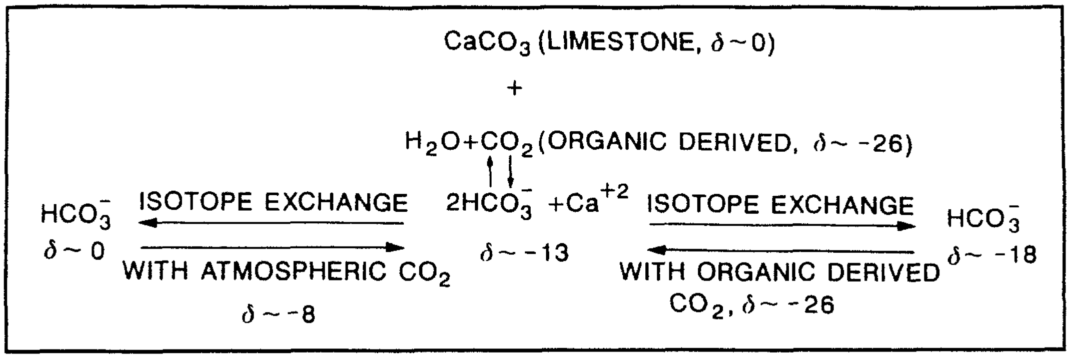

Figure 1. Model to explain variations in $\delta\ ^{13}C$ of dissolved inorganic carbon.

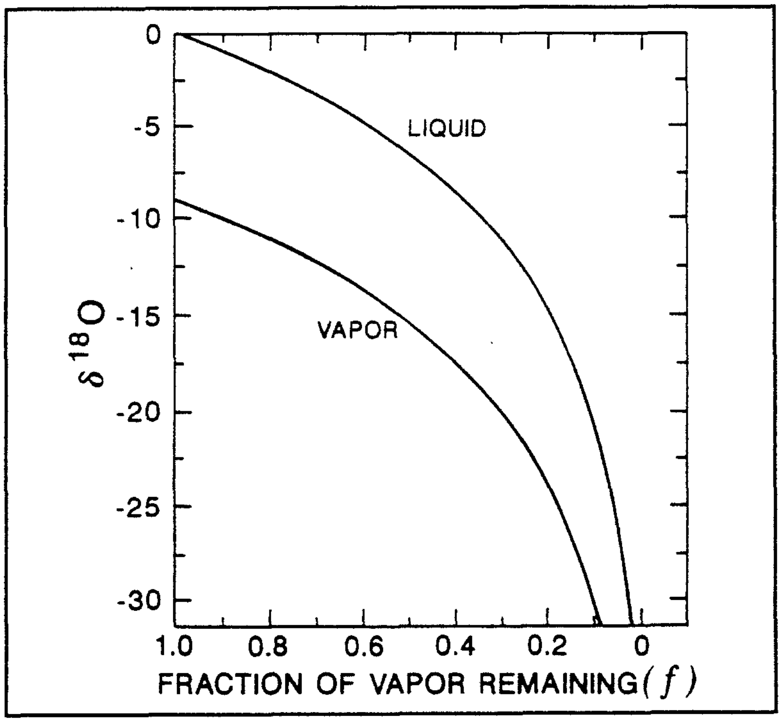

Figure 2. The Raleigh model for fractionation of oxygen isotopes during condensation of gaseous water at 25°C.

METHODS

Sampling Locations, Collection and Storage

Samples were collected in 500 ml glass screw cap jars. The jars were completely filled with the water sample to eliminate the possibility of exchange with atmospheric CO_2 in the air in the jar. Sampling locations are given in Table 1 and Figure 3. All numbered locations were taken from shore on March 19, 1990 and were immediately refrigerated and kept in the dark until their analysis over the next month.

Analytical Procedure

Salinities were determined by titrating water samples with silver nitrate using the modified Grasshoff (1983) method. Endpoints were detected by the change of precipitate color from white to light pink using sodium fluorescein indicator. Briefly, one ml of sample was titrated with a $AgNO_3 \approx [0.05\ M]$ solution which had been standardized against a NaCl solution with a salinity of 35.00 ppt. $\delta\ ^{13}C$ values were

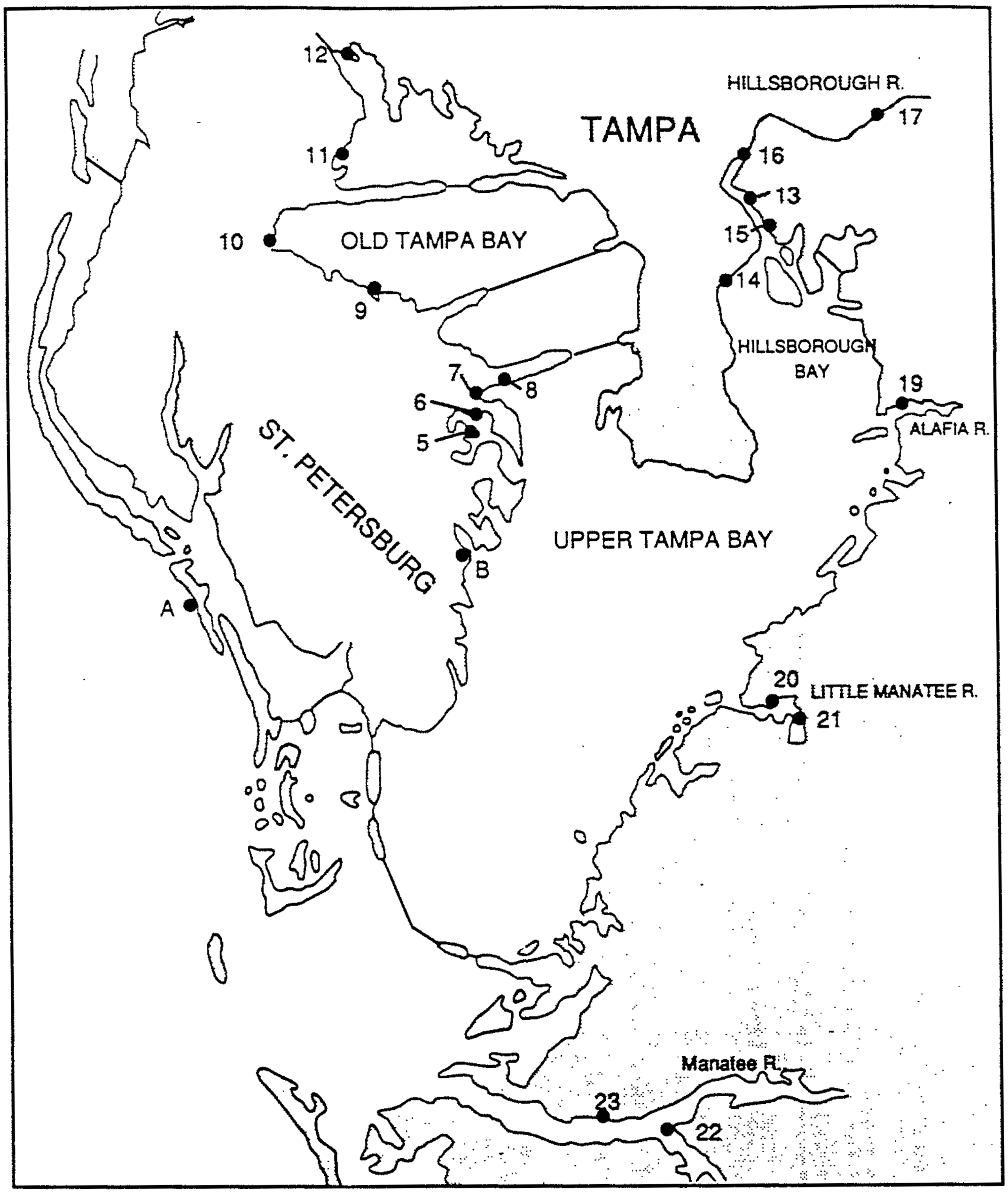

Figure 3. Sampling locations around Tampa Bay.

obtained for dissolved inorganic carbon by reacting 60 ml water samples with about 5 ml 85% H_3PO_4 in a partially evacuated side armed reaction vessel. The evolved CO_2 was collected at liquid N_2 temperature after passing through an isopropanol LN_2 slush bath (T=-80°C) to remove water vapor. $\delta\ ^{18}O$ was determined by equilibrating a 25 ml aliquot of cylinder CO_2 with 25 ml of the water sample in 60 ml plastic syringes for one hour on a commercial paint shaker. The molar ratio of gas to water was 1 to 1200. The equilibrated CO_2 was collected and processed as above for carbon and measured relative to a laboratory SMOW standard.

In order to get an estimate of the rate of exchange of atmospheric CO_2 with dissolved inorganic carbon (DIC), the following experiment was set up on the roof of the Marine Science building at the University of South Florida in St. Petersburg, located at about the middle of the western coastline of Tampa Bay; this location should have approximately the same temperatures and wind velocities as the open bay.

11.4 liters of distilled water, poisoned with sodium azide to inhibit biological activity and having a HCO_3^- concentration of 2.33 mM, about that of sea water, was added to a round pan (34 cm X 13 cm) and placed under a platform to stop direct sunlight from hitting the water. The platform was placed about 10 cm above the surface of the water to allow the wind to freely flow across the water. 60 ml of the water were withdrawn periodically and analyzed.

Table 1. Sample locations in the Tampa Bay area.

SAMPLE NO.	DESCRIPTIVE LOCATIONS
5	North St. Petersburg, near Riviera Bay; Turner Creek near 4th St. N/79th Ave.
6	North St. Petersburg, near Riviera Bay; Weedon Park
7	Near Gandy Bridge; Snug Harbor, north of Riviera Bay
8	Near Gandy Bridge, north of Snug Harbor
9	Bayou Canal; near Roosevelt Blvd., St. Pete/Clearwater Airport and 49th Street
10	Allen Creek; near U.S. 19, north of Roosevelt Blvd.
11	Alligator Lake/Shore Boulevard, Safety Harbor
12	Safety Harbor, near Oldsmar/S.R. 580
13	Hillsborough River, near I-75
14	Hillsborough Bay, near Bayshore Boulevard
15	Hillsborough River, close to Crosstown Expressway
16	Hillsborough River, near Buffalo Avenue
17	Hillsborough River, 40th Street/Sligh Avenue
18	Alafia River, Riverview; 5 miles from Tampa Bay
19	Alafia River, close to I-75; one mile from Tampa Bay
20	Little Manatee River, close to U.S. 41; 2.5 miles from Tampa Bay
21	Little Manatee River, near I-75; 6 miles from Tampa Bay
22	Manatee River, 8 miles from Tampa Bay
23	Manatee River, 5.5 miles from Tampa Bay
B-2	Big Bayou, collected on February 6, 1991
B	Big Bayou, collected on December 6, 1990
A	Gulf of Mexico/Treasure Island, collected on December 9, 1990
A-2	Gulf of Mexico/Treasure Island, collected on February 7, 1991

RESULTS AND DISCUSSION

Because a linear relationship was expected between the two isotope parameters and salinity, demonstrating a simple mixing between two end members, each isotope parameter was plotted relative to salinity. Figure 4 shows the data for $\delta^{13}C$ and salinity. Linear regression analysis gives the equation $y = -10.41 + 0.240\,X$ ($r = 0.812$). At salinity = 0, the intercept value for $\delta^{13}C$ is -10.4 ppt. This is heavier than the value that would be obtained by the weathering reaction given above for organic derived CO_2 reacting with marine carbonate. This additional shift may be due either to exchange with atmospheric CO_2 or interaction with carbonate minerals and groundwater. Generally, samples with values falling below the regression line have probably exchanged with organic derived CO_2, those falling above with atmospheric CO_2. Extrapolation of the trend to 35 ppt salinity gives a δ value of about -2 ppt, a value close to that expected for equilibration with atmospheric CO_2.

Figure 5 shows the relationship between $\delta^{18}O$ and salinity. The zero intercept of this linear regression ($y = -2.220 + 0.108X$, $r = 0.917$) is -2.20 ppt. For thirteen samples of rainwater collected between November 6, 1990 and February 14, 1991 on the roof of the Marine Science building at USF in St. Petersburg, $\delta^{18}O$ values ranged from -0.38 to -5.50 with a mean value of -2.11, which almost exactly matches the extrapolated value given above. The value calculated from the linear regression equation for a salinity of 35 ppt is +1.58. This value is about 1 ppt higher than expected and is presumably related to the loss of oxygen-16 rich water vapor from surface water during evaporation.

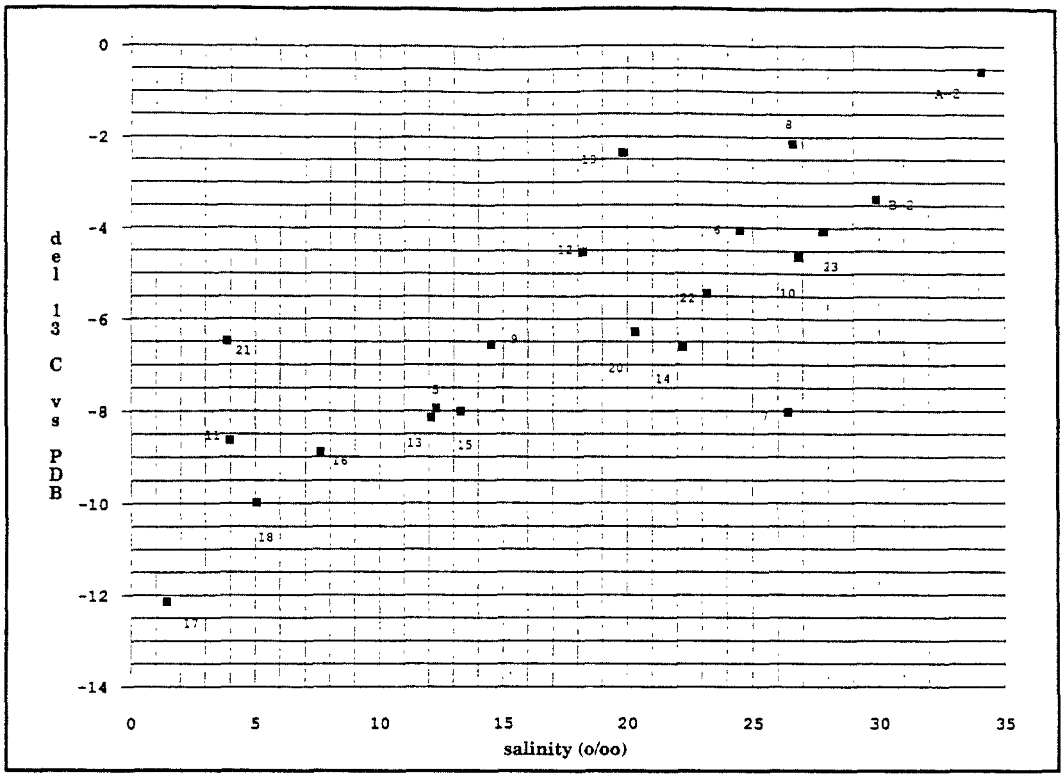

Figure 4. $\delta\ ^{13}C_{PDB}$ for dissolved inorganic carbon versus salinity in ppt. Numbers indicate sampling locations on Figure 3 and Table 1.

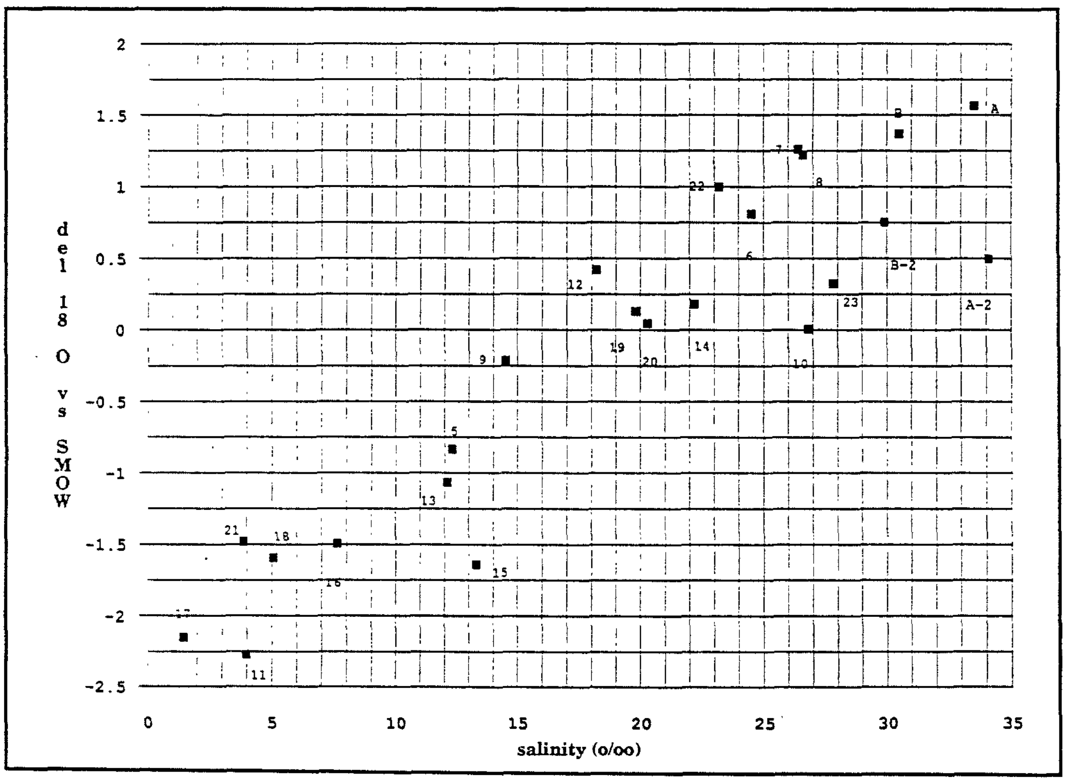

Figure 5. $\delta\ ^{18}O_{SMOW}$ versus salinity in ppt. Numbers indicate sampling locations on Figure 3 and Table 1.

The water supplies of the cities of St. Petersburg, Safety Harbor, Clearwater, Orlando and Tampa are all derived from the Floridan aquifer. $\delta\ ^{18}O$ values are -1.7, -2.0, -2.6, -2.0 and -2.8, respectively, with a mean value of -2.2 which again corresponds to the mean value for rainfall during the year. However, a range of about 5 ppt in the $\delta\ ^{18}O$ of rainfall has already been seen to date. Continued monitoring of the amounts and isotopic composition of rainfall during the coming year should help determine the reasons for these observations. Also needing confirmation is an inverse relationship between the amount and $\delta\ ^{18}O$ of rainfall.

The data given in Figure 6 show a marked change in ^{13}C with time in the direction for isotopic equilibration with atmospheric CO_2. For a freshwater mass with a δ of -13, the data suggest that it will take only a day or two to change to -10, the intercept for the trend shown in Figure 4. Thus, it appears that isotopic exchange is as important as water mass mixing in changing the isotope signature of waters being brought into the bay by runoff.

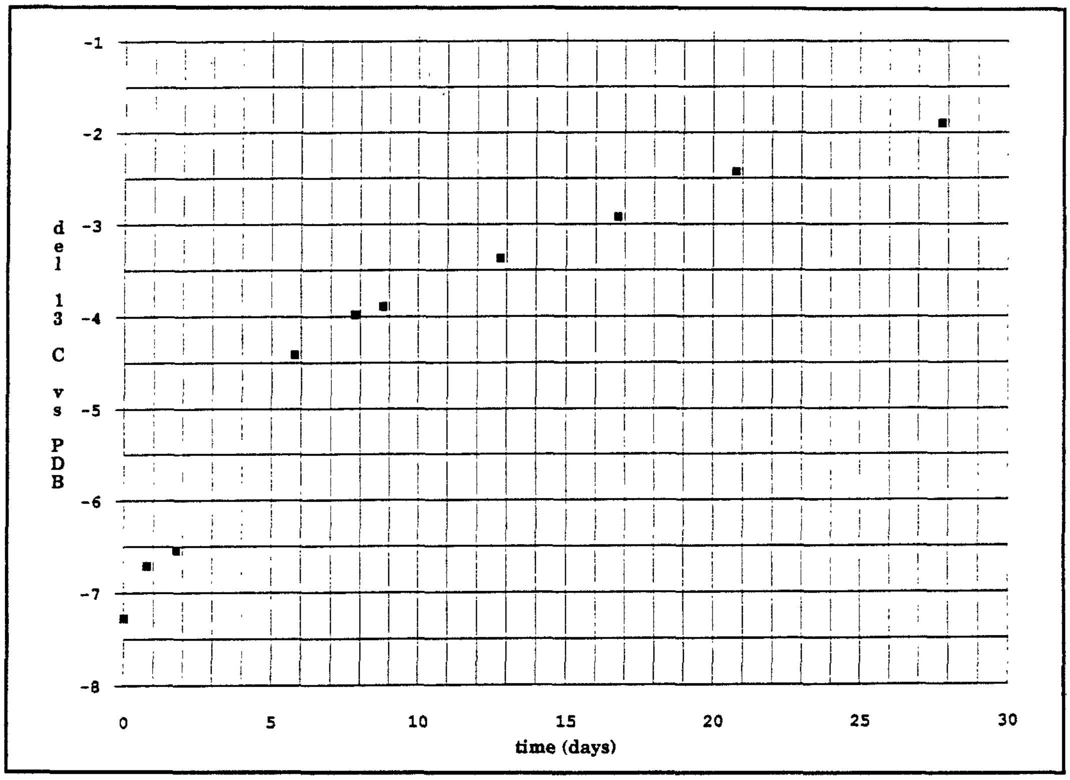

Figure 6. $\delta\ ^{13}C_{PDB}$ versus time in days for exchange experiment.

Each new study generally provides several leads that could be followed subsequently. This one has been no exception. Questions remain about the $\delta\ ^{13}C$ of biogenic CO_2 and its rate of exchange with DIC, the exact source and the $\delta\ ^{13}C$ of the nonbiogenic CO_2, the temporal changes of ^{18}O in rainfall, and the apparent inverse relationship between the amount of rainfall and its $\delta\ ^{18}O$. This study has shown once again that stable isotopes can provide a new understanding and provoke new questions about the biogeochemical environment around us.

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DISCHARGE MEASUREMENT TECHNIQUES FOR NUTRIENT LOAD COMPUTATIONS IN TIDALLY DOMINATED RIVERS

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Y. E. Stoker

ABSTRACT

Methods to determine seasonal and annual nonpoint source nutrient loads by measurements at the mouth of the tidally dominated Alafia River are currently being developed by the U.S. Geological Survey in cooperation with the Surface Water Improvement and Management department of the Southwest Florida Water Management District. Methods are being developed to 1) measure flood and ebb tidal discharge continuously, and 2) determine average constituent concentrations during selected flood and ebb tidal events in order to compute both upstream and downstream constituent loads. Tidal discharge measurement techniques (item 1) is the subject of this paper. The need for accurate flow measurements is a requirement for load computations and estimates. Tributary tidal influence prevents the use of standard discharge rating techniques and requires special measuring techniques and instrumentation.

Discharge is defined as the product of cross-sectional area and average water velocity in the cross section. Techniques to determine continuous discharge in the tidal Alafia River include the development of stage-area and index velocity-mean velocity relations and the measurement of an index velocity at a point in the stream using an electromagnetic flow meter. Unit (15-minute) index velocity and stage data, along with respective relations, provide a means for computing continuous discharge for load computations.

The success of the Alafia River experiment is attributed to the reconnaissance efforts that provided information for locating and operating an index velocity point. Index velocity proved to have high correlation with measured mean velocities.

INTRODUCTION

Nutrient enrichment in estuaries and coastal areas generally is accepted as a potential problem that contributes to the dedgradation of water quality in these areas. Excessive nutrients can result in blooms of phytoplankton and macroalgae, which can restrict the growth of submerged plants such as seagrasses through shading effects. About 80% of the seagrasses in Tampa Bay have been lost between about 1940 and 1981 (Lewis et al. 1985). These losses were attributed to progressive eutrophication and reduction in light penetration due to increased concentrations of micro- and macroalgae.

Water quality in Tampa Bay has been monitored for nearly 20 years by the Hillsborough County Environmental Protection Commission (Boler 1988). A water quality index has been formulated by the Environmental Protection Commission to allow comparison of water quality in the bay with previous years. The index is computed from concentrations of dissolved oxygen, chlorophyll *a*, total coliform bacteria, total phosphorus, total organic nitrogen plus ammonia, biochemical oxygen demand, and effective light penetration. The index has shown that water quality in the bay is poorest in Hillsborough Bay, with the lowest index values often found near the mouth of the Alafia River (Boler 1989).

The Tampa Bay Regional Planning Council (1990) indicated that eutrophication in Tampa Bay caused by nutrient overenrichment is a priority issue that needs to be addressed. In response to this need, the Surface Water Improvement and Management (SWIM) department of the Southwest Florida Water Management District is coordinating several studies that will provide information needed to determine the nutrient budget of Tampa Bay. Components of the budget are nutrient imports to the bay from point sources, tributaries, direct runoff from coastal areas, groundwater contributions, direct rainfall, release from the sediments and exports from the bay by way of losses to the Gulf of Mexico, adsorption to the sediments, removal by fisheries, and losses to ground water.

Flow and nutrient concentration data are available at stations on many of the major tributaries to Tampa Bay. These gaging stations, however, are upstream of the tidal part of the rivers. Total flows and nutrient loads to the bay can be extrapolated from data at upstream sites, but this technique is not precise (Richards 1989). In the Tampa Bay basin, the areas surrounding the tidal parts of inflowing rivers often are highly urbanized or industrialized. Additional nutrient loads to this part of the river would be unaccounted for using extrapolation techniques to compute total loads. Accurate load estimates for a tributary require accurate flow data (Richards 1989), and in tidal rivers, this component of load computations is the most difficult to measure. The U.S. Geological Survey, in cooperation with SWIM, is developing techniques that will be used to measure nutrient loads near the mouth of the Alafia River, a tributary to Hillsborough Bay. Once defined, these techniques could be used at other tributaries to measure total nutrient loads. Methods are being developed to 1) measure flood and ebb tidal discharge continuously, and 2) determine average constituent concentration during selected flood and ebb tidal events in order to compute constituent loads. The purpose of this paper is to describe discharge measurement techniques used for the Alafia River at Gibsonton, Florida (Fig. 1), and to give information on methods of developing procedures for continuous discharge computations.

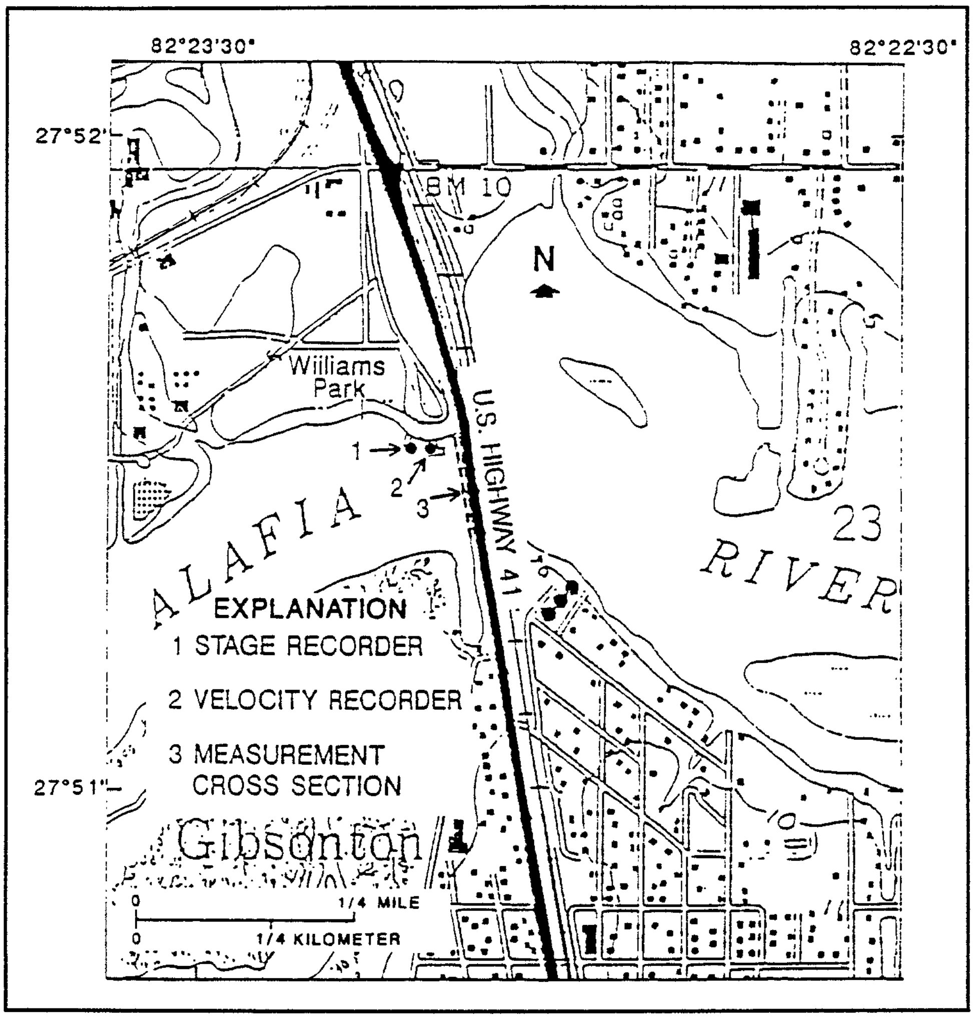

Figure 1. Alafia River study site, location of instruments, and cross section.

DISCHARGE MEASUREMENTS AND RATINGS

Discharge is defined as the product of cross-sectional area and average water velocity in the cross section. The vertical axis Price AA current meter is the standard meter used by the U.S. Geological Survey to measure velocity (Fig. 2). The principle of operation is based on the proportionality between the velocity of the water and the resulting angular velocity of the meter rotor. By placing the meter at a point in a stream and counting the number of revolutions of the rotor during a measured interval of time, the velocity of the water at that point can be determined. To determine water velocity from revolution rate of the meter, it is necessary to apply a rating or equation.

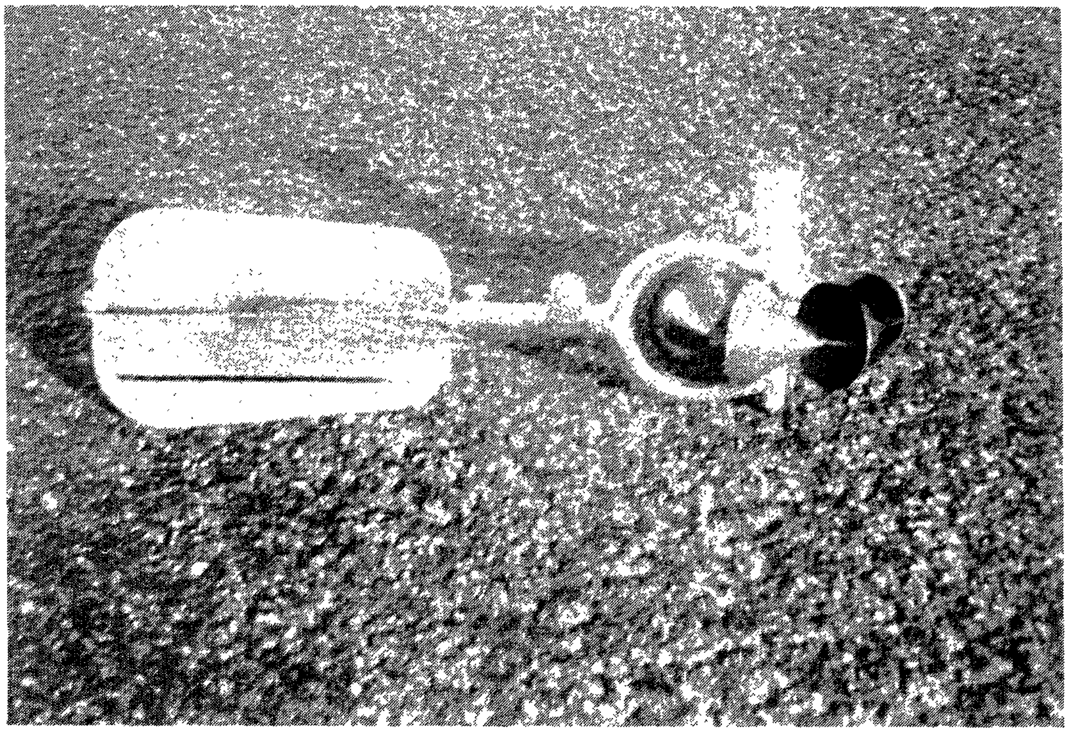

Figure 2. Price current meter.

To measure discharge at a cross section, stream depths, subsection widths, and velocities are measured at 20 to 25 or more points in the cross section. In the event that flow is not normal to the section, horizontal angle coefficients are observed and recorded for width corrections. The discharge is then computed by summing the products of the partial areas of the stream cross section and their respective average velocities. The equation

$$Q=\sum(a\ v)$$

represents the computation where Q is total discharge, a is an individual partial cross-section area, and v is the corresponding mean velocity of the flow normal to the partial area (Buchanan and Somers 1969).

The conversion of a record of stage to record of discharge is made by the use of a stage-discharge relation (rating). The physical element or combination of elements in the stream channel that controls the relation between stage and discharge is known as a control. The control may be simple or compound; natural, such as a sandbar, rock outcrop, or channel constriction; or it may be manmade, such as a weir, flume, or dam. A simple discharge rating (Fig. 3) that relates stage to discharge can be developed for many streams by observing or recording the stage of the pool immediately upstream from the control and periodically measuring the total discharge. Measurements need to be made over the range of stage expected to occur. Measurements are then plotted on rectangular or log-log graph paper to define a rating. The control is the stabilizing factor in the relation between stage and discharge. If control conditions remain stable, the rating will be stable and rating definition is simplified. Figure 4 illustrates the stage-control relation and the effect of a shifting sandbar control that is subject to change with time.

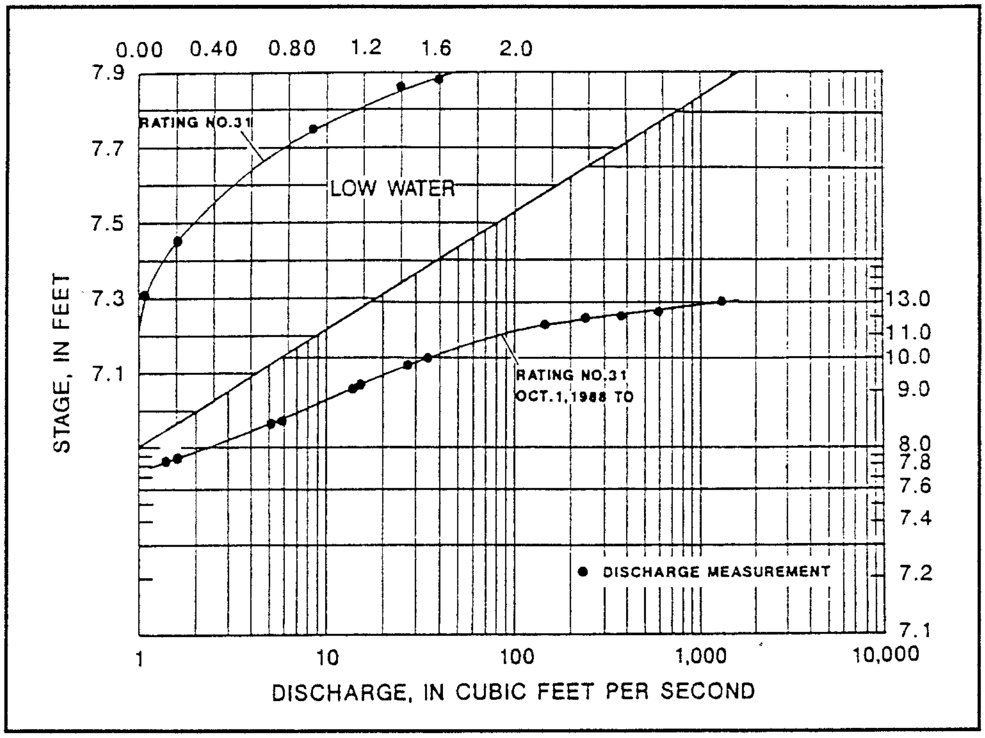

Figure 3. Simple stage-discharge rating.

A shifting control may be caused by scour or filling, lodging of debris, growing or dying vegetation, or backwater from a tributary, as well as other changing conditions. Minor shifting control conditions are adjusted for routinely by defining the shift from the base rating with additional discharge measurements. A new rating may be defined by discharge measurements, or mathematical means may be used for applying temporary shift adjustments.

At any given discharge, the effect on the stage at the gaging station that is attributable to the operative control element(s) is known as backwater. As long as the control elements are unvarying, the backwater for a given discharge is unvarying, and the discharge is a function of the stage only; the slope of the water surface for steady flow at that stage also is unvarying. If some of the control elements are variable for any given discharge, the stage at the station and the slope are also variable.

TIDAL DISCHARGE MEASUREMENTS AND RATINGS

At gaging stations on tide-affected streams, the tide is considered a variable control element. Determination of continuous discharge in tidal rivers, therefore, requires an accurate definition of both stage and velocity as continuous functions of time. Cross-sectional area is a well-defined function of tidal stage that can be determined by fathometric and planimetric methods. With this relation, continuously recorded tidal stage can be transformed into a continuous history of cross-sectional area fluctuations. In many field situations, a similar relation can be established between average water velocity in the cross section and a representative index velocity at some fixed location. With this relation, continuously recorded index velocity can be transformed into a continuous history of cross-sectional average water velocity fluctuations. The accuracy of continuous discharge computed in this way is dependent on the accuracy of recorded field measurements and the accuracy of stage-area and index velocity-average velocity relations.

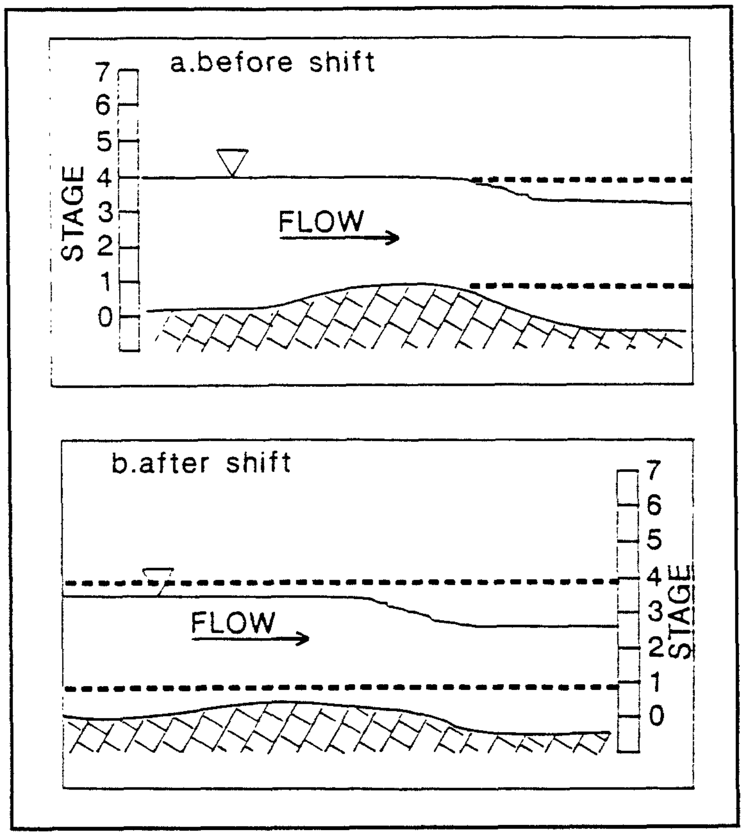

Figure 4. Effect of a shifting sandbar control on river stage.

One of the difficulties in making velocity observations in tidal streams is that the flow can reverse direction during a measurement. During periods of high freshwater runoff, stratification may occur so that near-surface velocities remain outgoing during flood tides while the bottom velocities are incoming. Meters used for tidal measurements need to be capable of indicating direction as well as velocity. The Price meter does not have this capability but can be equipped with an electronic compass for indicating direction. An acoustic or electromagnetic meter with integral compass also may be used.

Because flow is constantly changing in a tidal stream, measurements must be made in minimal time to minimize rate-of-change errors. Normal horizontal and vertical velocity distribution is demonstrated in Figures 5 and 6. Vertical distribution of velocity on a tidal stream may appear similar to the diagrams in Figure 7 at unique times in a tidal cycle.

Reconnaissance on the Alafia River included making visual inspections of likely sites, making preliminary cross-section profiles, and observing vertical velocity distribution to determine a probable location for the initial index-velocity measuring point. Generally, a satisfactory rating can be obtained for tidal streams when an index velocity point in or near the measurement cross section can be found that correlates well with the measured mean velocity. Figure 8 shows the vertical velocity distribution at three subsections in the Alafia River cross-section, and Figure 9 shows a comparison of those three verticals with the recorded index velocity for a 26-hour synoptic water quality sampling period in September 1990.

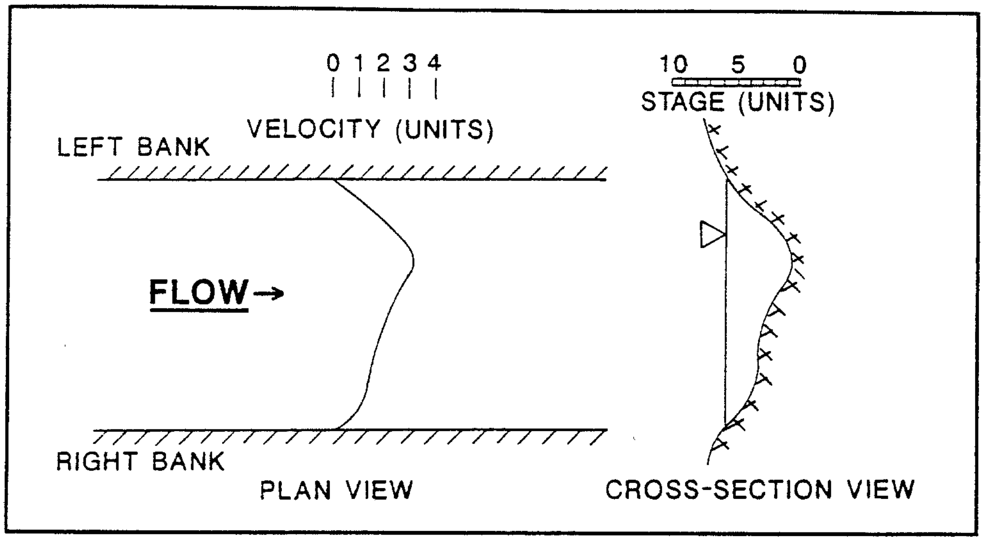

Figure 5. Normal horizontal velocity distribution in a stream.

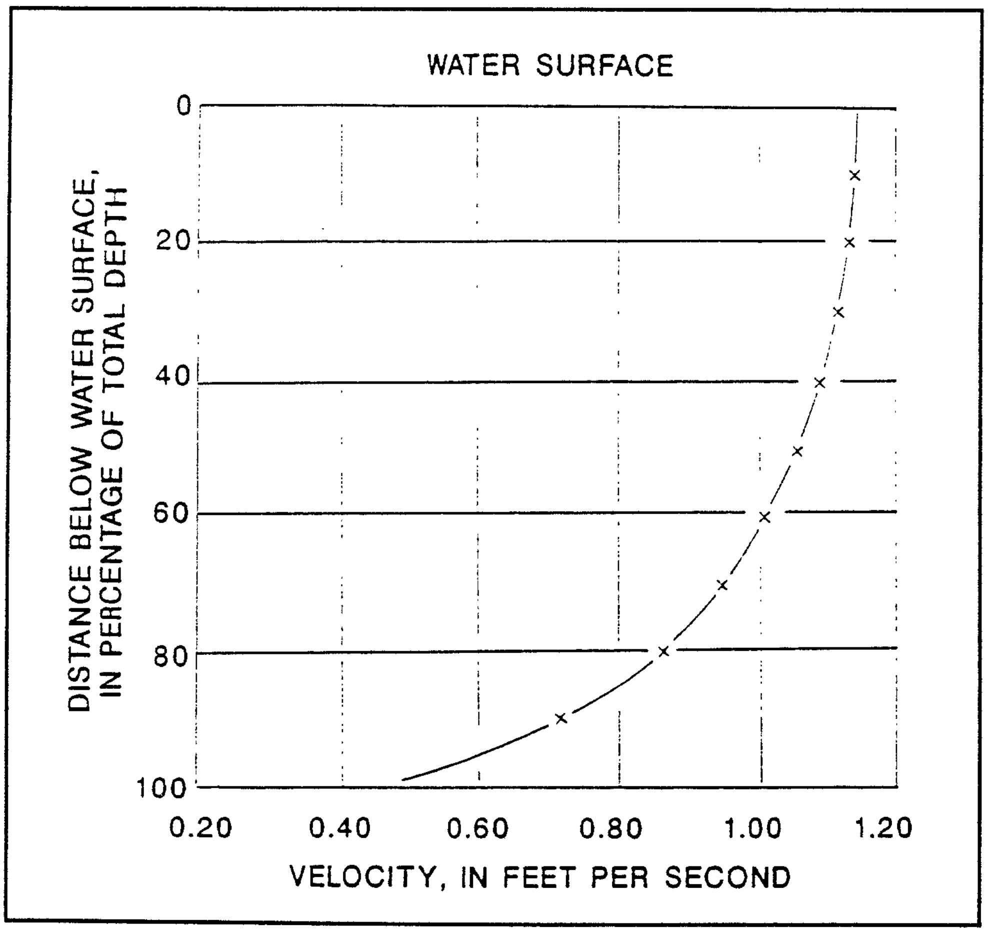

Figure 6. Normal vertical velocity distribution in a stream.

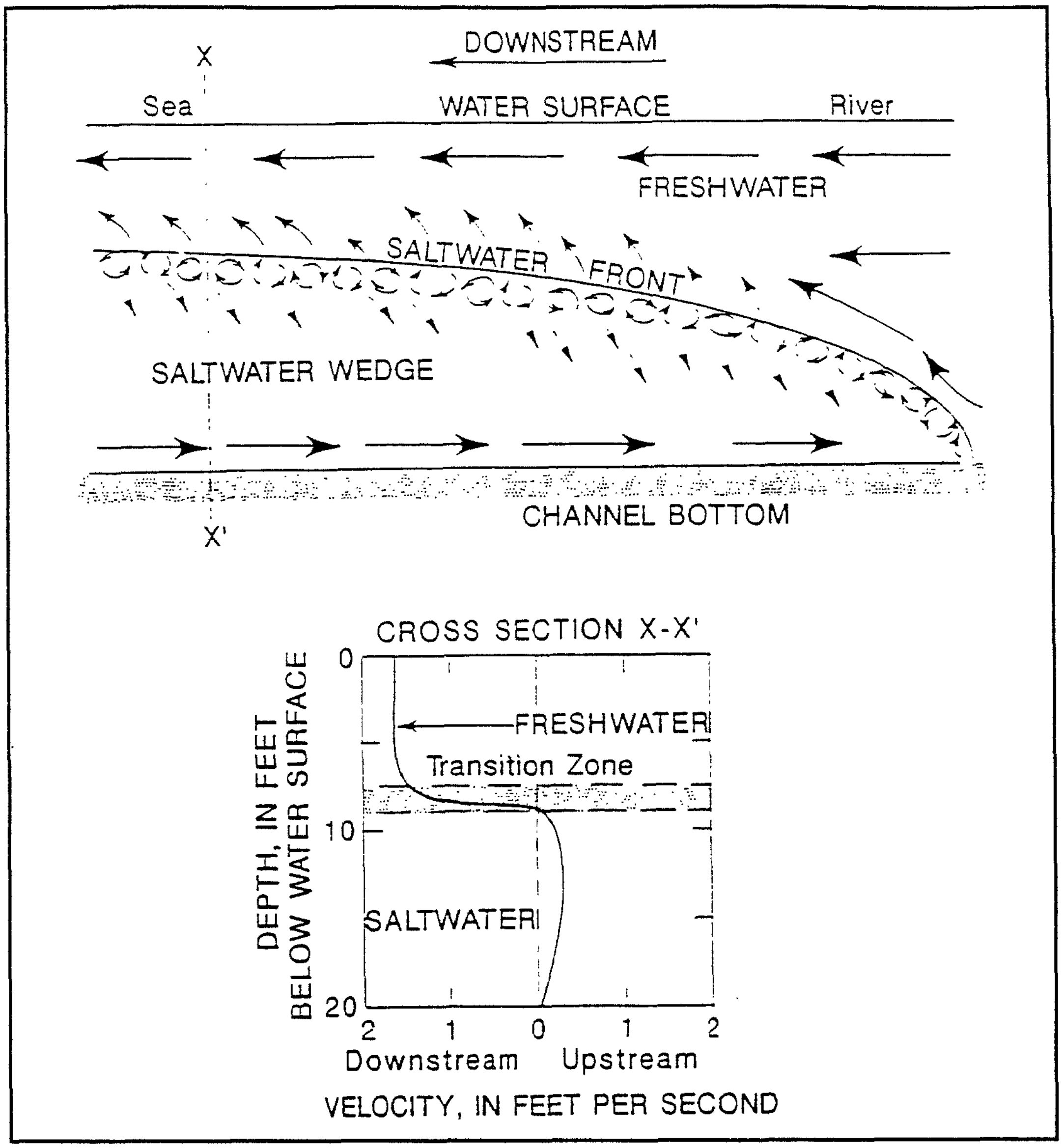

Figure 7. Velocity profiles showing variability of vertical distribution possible for some sites (modified from Giese et al. 1985, fig. 1.6).

A summary of discharge measurement data used to develop the index point-mean velocity and stage-area relations (Fig. 10) is given in Table 1. These relations are used to calculate discharge from index velocity and stage data. Discharge is calculated by entering the index velocity-mean velocity relation with the time-weighted index velocity to obtain the mean velocity, then entering the stage-area relation with the time-weighted stage to obtain the area. The area is then multiplied by the mean velocity to obtain the total discharge for the given time interval. Possible seasonal variations and physical changes require additional discharge measurements to verify relation stability.

A bidirectional electromagnetic flow meter (EFM) at the selected index point (Fig. 1) measures index velocity at 15-minute intervals. The velocity data are stored by an electronic data logger (EDL). Stage data are recorded by an analog-to-digital recorder (ADR) at the same time interval. Eastern Standard Time is used year round to facilitate using the data base for modeling.

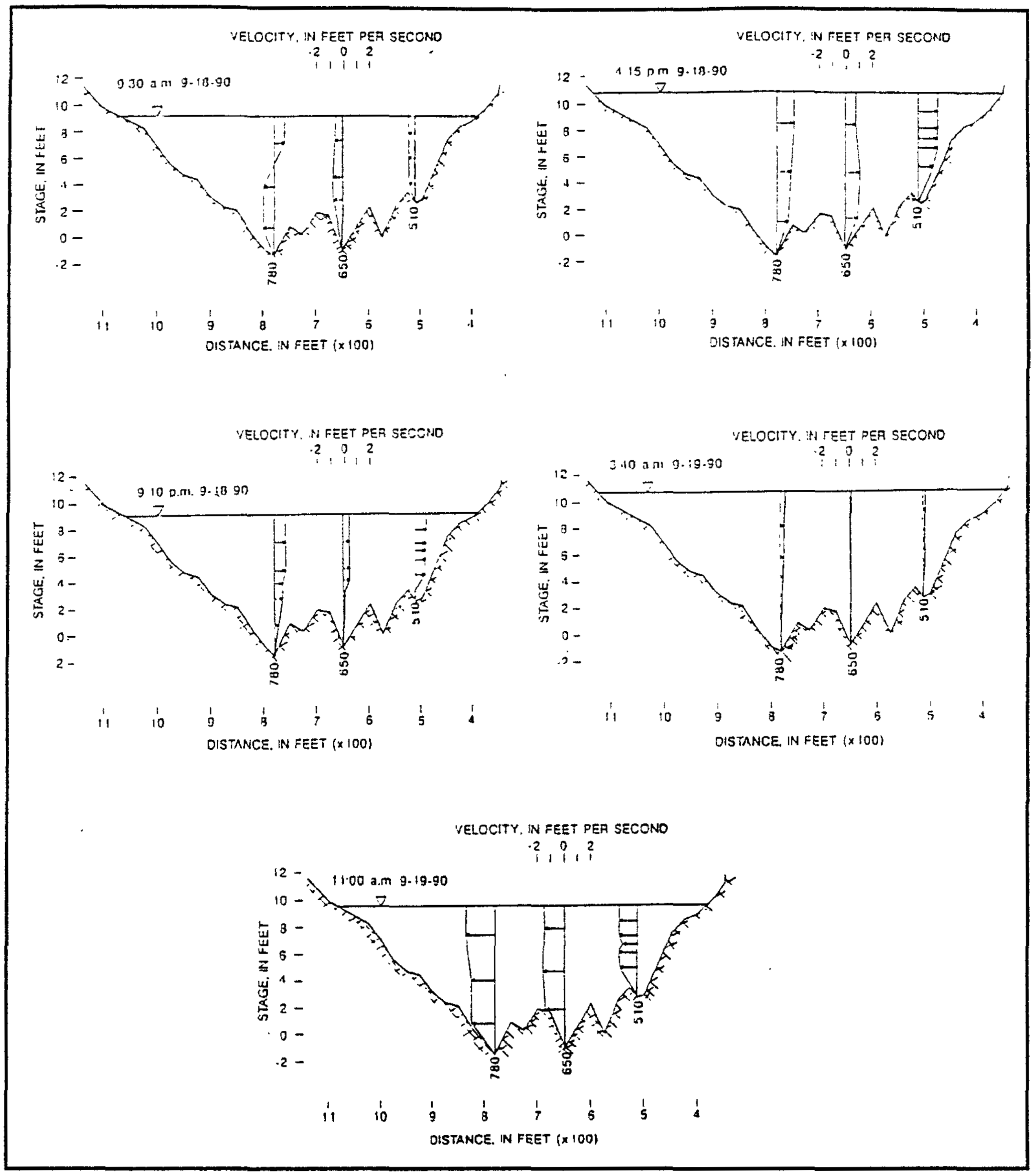

Figure 8. Selected velocity profiles at three subsections of the Alafia River measurement cross section.

An ultrasonic velocity meter (UVM) will be installed at a subsection of the bridge cross section as part of this experiment. In this application, the UVM provides an index velocity in a selected path across part of the channel instead of only at a point as provided by the EFM. The UVM measures the velocity of flowing water by means of an ultrasonic signal that moves faster downstream than upstream. Meters of this type are useful in determining discharge at streamflow sites where the relation between discharge and stage varies with time because of variable backwater conditions.

The UVM is a nonmechanical, nonintrusive device that is capable of measuring lower velocities than can be measured with a current meter. It can provide a continuous and reliable record of water velocities over a wide range of conditions, but several constraints apply:

1. Accuracy is reduced and performance is degraded if the acoustic path is not a straight line. The path can be bent by reflection if it is too close to a stream

boundary or by refraction if it passes through density gradients resulting from variations in water temperature, suspended sediment concentrations, or salinity. Reflection from stream boundaries can cause signal cancellation if boundaries are too close to the signal path.

2. Signal strength is attenuated by particles or bubbles that absorb, spread, or scatter sound. The concentration of particles or bubbles that can be tolerated is a function of the path length and frequency of the acoustic signal.
3. Changes in streamline orientation can affect system accuracy if the variability is random.
4. Errors relating to signal resolution are much larger for a single detection scheme than for multiple detection schemes (Laenen 1985).

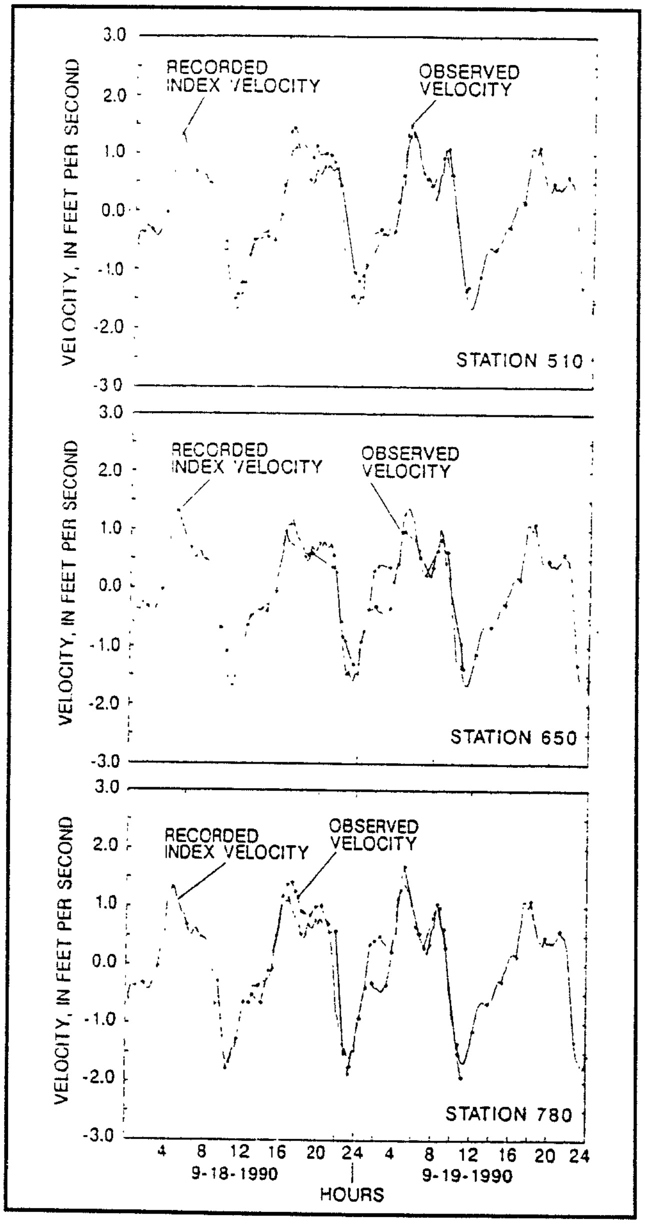

Figure 9. Mean velocities and recorded index velocities at three subsections.

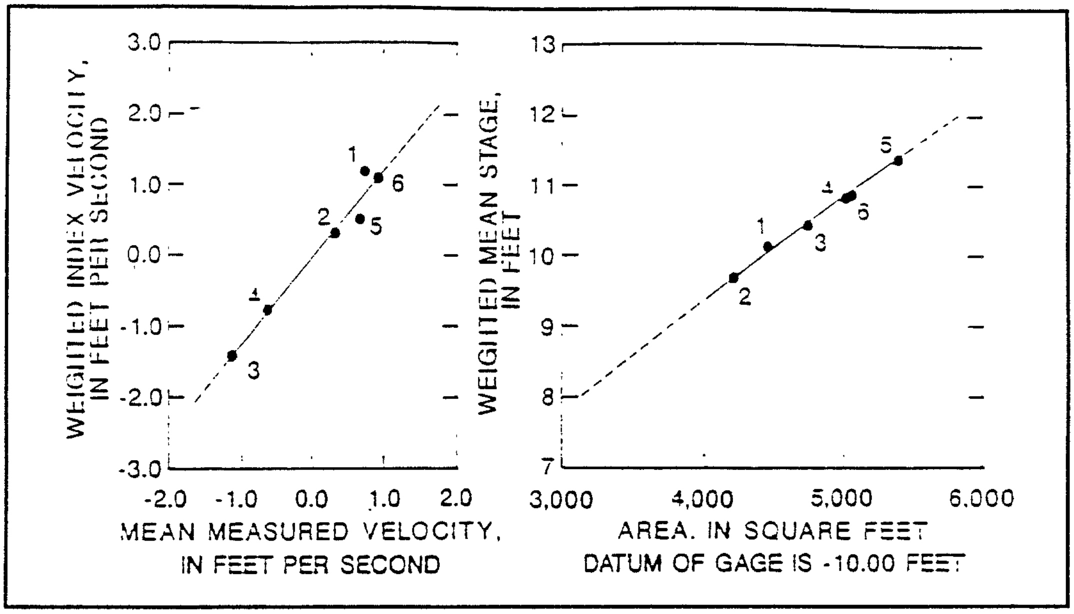

Figure 10. Index velocity-mean velocity and stage-area relations for the Alafia River.

Table 1. Discharge measurement summary data for the Alafia River at Gibsonton.

MEASUREMENT NUMBER	DATE	WEIGHTED MEAN STAGE (ft)	AREA (ft^2)	WEIGHTED INDEX VELOCITY (ft/s)	MEAN MEASURED VELOCITY (ft/s)	DISCHARGE (ft^3/s)
1	8/28/90	10.13	4,450	1.17	0.74	3,270
2	8/28/90	9.67	4,200	.30	.33	1,370
3	10/2/90	10.44	4,740	-1.44	-1.13	-5,370
4	10/2/90	10.84	5,020	-.79	-.63	-3,160
5	10/2/90	11.39	5,390	.48	.66	3,540
6	10/2/90	10.84	5,060	1.07	.92	4,640

The UVM, although more costly and complicated than the EFM, generally requires less maintenance for continuous operation. Figure 11 shows typical UVM components; Figure 12 illustrates the line velocity path of an acoustic pulse; Figure 13 shows voltage representation of transmit and receive pulses.

Specific conductance and temperature will be monitored at points near the bottom and near the surface in the subsection where the UVM is located. These properties will be useful in analyzing the effects of stratification on the operation of the UVM and will provide data needed in verification of the location of the freshwater-saltwater interface.

SUMMARY

Accurate flow measurements are needed to make load computations and estimates. When standard discharge rating techniques can be applied, the computations are simplified. However, tributary tidal influence in streams such as the Alafia River presents unique and difficult measuring conditions that require special measuring techniques and instrumentation.

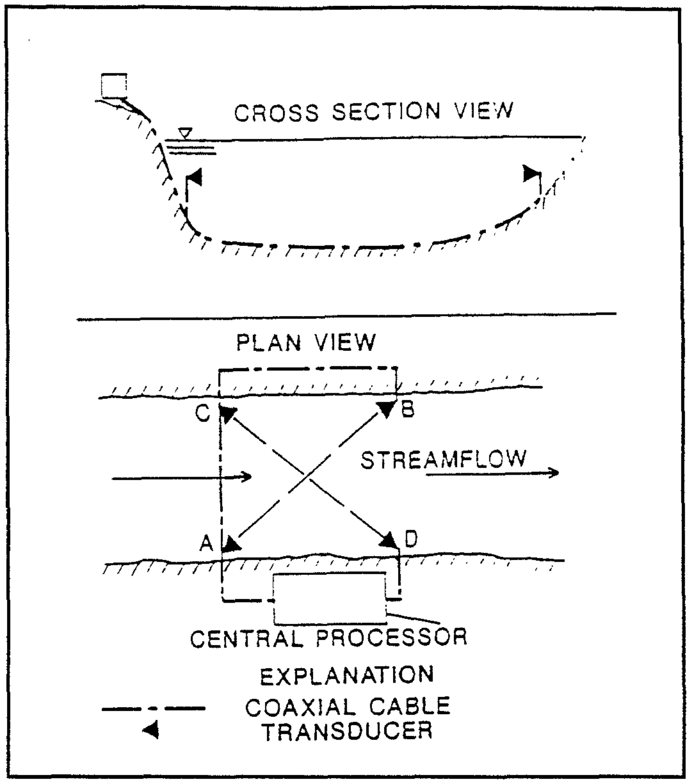

Figure 11. Typical location plan for single level cross path UVM components.

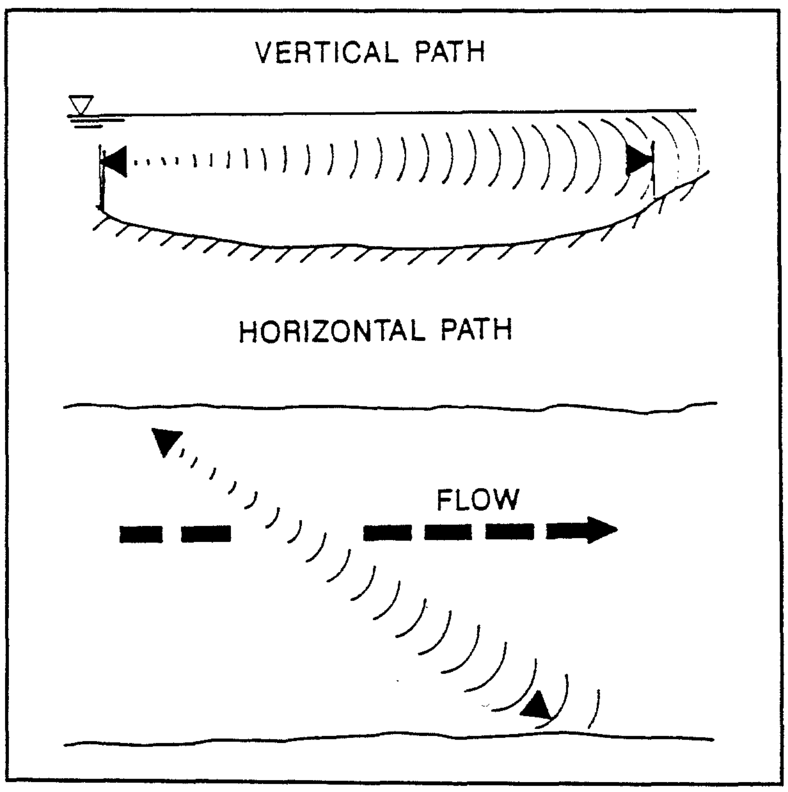

Figure 12. Acoustic pulse line-velocity path (modified from Laenen 1985).

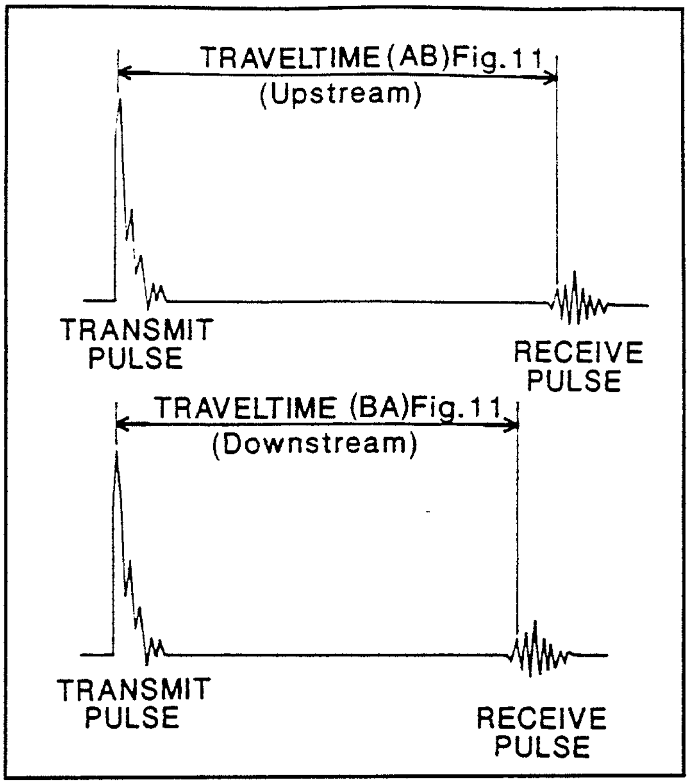

Figure 13. Voltage representation of UVM transmit and receive pulses at upstream and downstream transducers (from Laenen 1985).

In the Alafia River, index velocities are measured at a point in the river using an electromagnetic flow meter, and these velocities are used in conjunction with stage data and index velocity-mean velocity and stage-area relations to compute discharge. The success of this measurement technique is attributed to the reconnaissance efforts that provided information for locating and operating an index velocity point. The index point proved to have high correlation with measured mean velocities. Unit (15-minute) velocity and stage data, along with respective ratings, provide a means for computing continuous discharge for load computations. The method provided here may be used for other tidally influenced tributaries, but reconnaissance and discharge measurements will be required for each site to establish viability.

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LONG-TERM TRENDS OF NITROGEN LOADING, WATER QUALITY AND BIOLOGICAL INDICATORS IN HILLSBOROUGH BAY, FLORIDA

J. O. R. Johansson done 5/7/99

INTRODUCTION

At the time of the first BASIS meeting in 1982, water quality conditions in Hillsborough Bay were very degraded and the bay had been in this state for many decades. Few improvements had been seen by 1982, despite that three years earlier one major point source of nutrients to the bay—the City of Tampa's wastewater plant at Hookers Point—had converted its process from primary to state of the art advanced wastewater treatment. The cost of this conversion was close to $100 million.

Within a year after the Hookers Point conversion in 1979, total coliform counts in Hillsborough Bay were reduced dramatically. However, it was a surprise and of concern to some of those working with the bay that no apparent improvement of other important water quality parameters and biological indicators, such as dissolved oxygen and chlorophyll, had occurred several years after the conversion. Further, no reduction of the dominant late summer to early winter blue-green phytoplankton community had been noted. It was not until 1984 that chlorophyll concentrations were reduced substantially, coincidental with a large decrease in planktonic blue-green algae. Dissolved oxygen in bottom waters of central Hillsborough Bay during the summer also improved at this time. Another concurrent sign of the recovery of Hillsborough Bay and areas close to Hillsborough Bay was the limited return of seagrasses to the shallow tidal and subtidal bars at the perimeter of the bay (Avery, this volume; Johansson and Lewis, in press; Lewis et al., this volume).

It is now evident that substantial improvements of several water quality parameters and biological indicators have occurred in Hillsborough Bay and other subsections of Tampa Bay since the mid-1980s. Although it is impossible to definitely demonstrate the cause of the recovery, a general understanding of the Tampa Bay ecosystem, coupled with comparative information from other estuaries, strongly suggests that the documented improvements have resulted from a large reduction of nitrogen loading from external sources. Nitrogen loading has not only been reduced from domestic wastewater sources, but also from industrial sources—specifically from the large and economically important central Florida fertilizer industry.

This report addresses general background information on Hillsborough Bay water quality, compares long-term nitrogen loading trends to water quality indicators, and recommends management options for improving bay conditions. First, the study area and its water quality history during the last several decades are described. The condition of Hillsborough Bay in the late 1960s was reported by the Federal Water Pollution Control Administration (FWPCA 1969). This was the first, and is to date, the most comprehensive eutrophication management study conducted in Tampa Bay; results and recommendations from the study are referenced extensively in this report. Second, long-term trends of nitrogen loading to Hillsborough Bay are discussed and compared to long-term trends of water quality and phytoplankton parameters. Specifically, nitrogen loading from external sources and ambient chlorophyll concentrations measured by the FWPCA are compared to the current Hillsborough Bay loadings and chlorophyll concentrations. In addition to the FWPCA (1969) report, sources of information used are from the City of Tampa Bay Study Group (COT), the Hillsborough County Environmental Protection Commission (HCEPC), the U.S. Geological Survey (USGS), the Fish and Wildlife Service (FWS), the Tampa Port Authority (TPA), the Southwest Florida Water Management District (SWFWMD), the Tampa Bay Regional Planning Council (TBRPC), and the Florida Department of Environmental Regulation (FDER). Finally, the report addresses water quality management needs, specifically the need to develop tools to help protect and enhance the natural resources found in Tampa Bay. A simple eutrophication model which

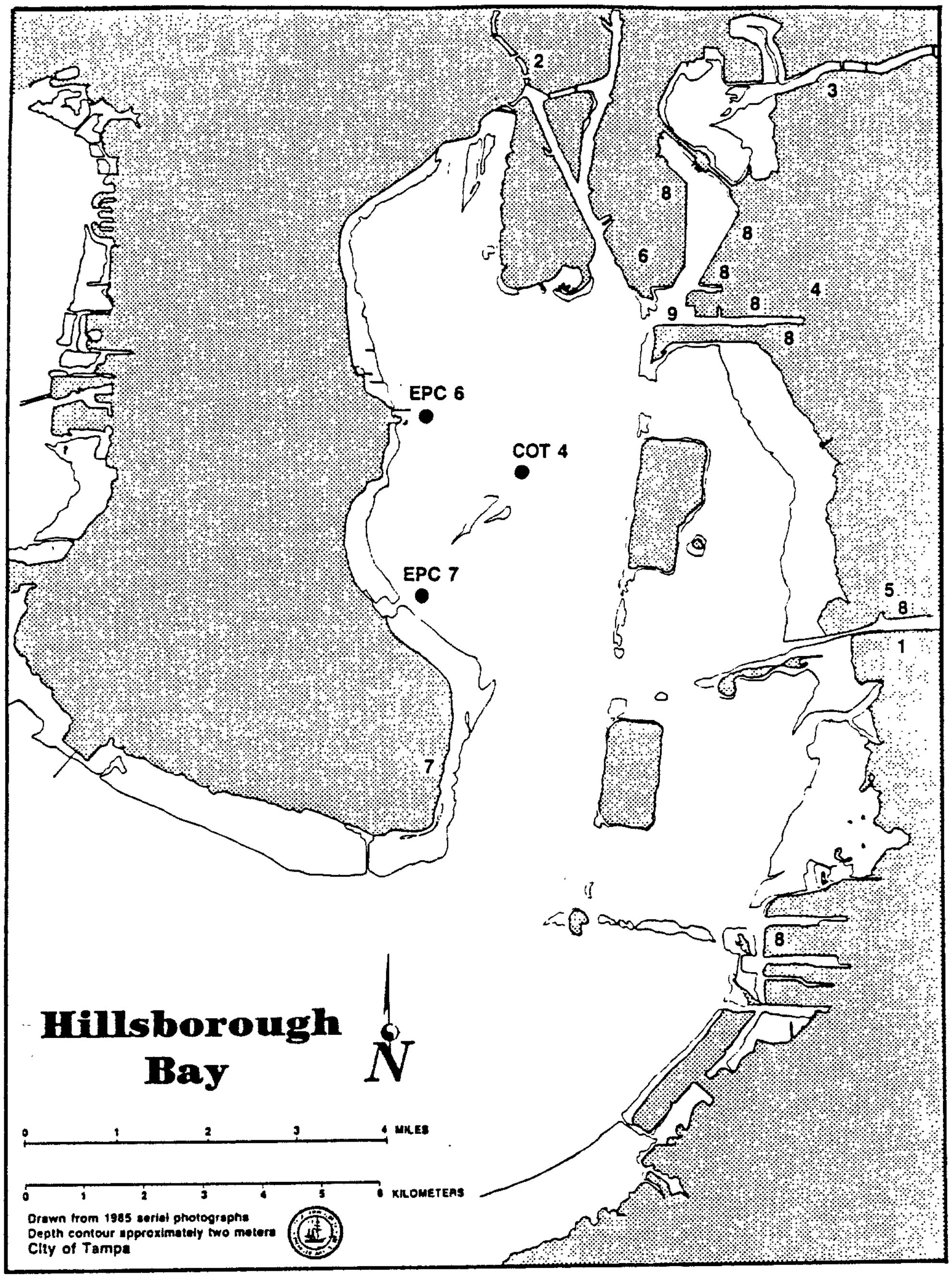

Figure 1. Hillsborough Bay, Florida.

1. Alafia River
2. Hillsborough River and downtown Tampa
3. Palm River/Tampa Bypass Canal
4. Nitram, Inc. and Delaney Creek
5. Cargill Fertilizer, Inc.
6. Hookers Point Wastewater Plant
7. MacDill AFB Wastewater Plant
8. Fertilizer shiploading terminals
9. Sutton Channel in East Bay

may link external nitrogen loading to valuable resources such as seagrasses and other benthic communities is discussed.

THE STUDY AREA

Tampa Bay is a shallow subtropical estuary located on the Gulf of Mexico coast of central Florida. It is one of the largest estuaries in the southeastern United States with an open water surface area of 1030 km^2 and with a watershed area of 5700 km^2. Population growth of the Tampa Bay region is one of the fastest in the United States. Approximately 1.7 million people now live within 80 km of the bay (TBRPC 1989). Hillsborough Bay is the eastern uppermost section of the Tampa Bay system with an open water area of 105 km2 (Figure 1). Although this is only 10% of the Tampa Bay surface area, three of Tampa Bay's five major rivers empty into Hillsborough Bay and the bay receives approximately 60% of the total surface freshwater flow to Tampa Bay (Goodwin 1987). Generally, Hillsborough Bay is the head of the Tampa Bay estuarine system, the area where freshwater and seawater mix, and it should not be surprising by this fact alone, that Hillsborough Bay historically had the "poorest" water quality of the Tampa Bay subsections. However, human impacts in the drainage basin have certainly been the major cause of eutrophication and degraded water quality.

The City of Tampa, which borders Hillsborough Bay to the north and the west, discharges stormwater to Hillsborough River and to the bay directly. Tampa also operates the largest domestic wastewater treatment plant in the area and currently discharges approximately 60 MGD of highly treated wastewater into the upper portion of the bay. Agricultural lands, extensive phosphate mines and many large fertilizer processing plants are located within the drainage area of the Alafia River to the east. In addition, several fertilizer facilities, such as processing plants, storage facilities and shiploading terminals are located near the eastern shores of the bay. The bay serves as a major shipping port of fertilizer products. In 1988 some 20 million short tons of fertilizer material were shipped from Hillsborough Bay (TPA, unpublished), which comprises approximately 80% of the United States production of phosphatic fertilizer and 50% of the production of nitrogen-containing fertilizer (Bureau of the Census 1988).

PAST WATER QUALITY CONDITIONS

Almost 40 years ago, Tampa Bay was described as grossly polluted due to discharges of poorly treated domestic wastewater and industrial waste from phosphate mines, citrus canneries and other industrial sources (Galtsoff 1954). However, water quality problems specific to Hillsborough Bay may have been present as early as 1928 when occasional noxious odors along the western shore of Hillsborough Bay were noted (FWPCA 1969). The odor was associated with abundant growth of macroalgae and poor water quality caused by discharges of untreated domestic and industrial waste. Odors continued sporadically until the City of Tampa completed the Hookers Point primary wastewater treatment plant in 1951. With the start-up of the primary plant the odors apparently subsided temporarily. However, odors from large amounts of macroalgae decomposing on beaches and next to seawalls became severe during the early 1960s; as a result, citizens living close to Hillsborough Bay complained to the authorities about the noxious odors. Also during this period, the water in the bay was not suitable for body contact due to high bacteria content. Signs were posted at several popular beaches to warn swimmers of the contaminated water.

The Federal Water Pollution Control Administration Study

In 1965, the City of Tampa and the Florida State Board of Health requested technical assistance from the FWPCA to help identify the causes of the poor water quality and the noxious odors in Hillsborough Bay and to recommend solutions to the problems. FWPCA conducted an extensive study of Hillsborough Bay water quality problems in 1967 and 1968 (FWPCA 1969). The FWPCA study confirmed the earlier observations of highly deteriorated conditions in Hillsborough Bay. Indicators of the poor water quality included high turbidity, high fecal coliform counts, anoxic bottom waters and large amounts of drift macroalgae in the shallows.

The study identified point source discharges with high nutrient and organic content as the underlying cause of the odor and degenerated water quality in Hillsborough Bay. Specific sources identified were the Alafia River and the fertilizer industry in its basin, the City of Tampa's wastewater plant at Hookers Point, the U.S. Phosphoric Products Company (now Cargill Fertilizer, Inc.) and the Nitram Chemical Company (now Nitram, Inc.). After an extensive monitoring program of these sources, it was estimated that the Alafia River and the chemical companies supplied 63% of the total nitrogen and 94% of the total phosphorus entering the bay. The Hookers Point facility discharged 32% of the total nitrogen and less than 5% of total phosphorus. However, this facility was the major contributor of organic material to the bay, supplying more than 85% of all carbonaceous material from point sources. Further, the study established that nitrogen was the growth-limiting nutrient in Hillsborough Bay and that any reduction in available nitrogen could be expected to produce a corresponding reduction in plant growth. Table 1 lists total nitrogen loadings from all sources identified and measured by FWPCA (1969) and one additional source, losses from shiploading terminals, which are discussed below.

Table 1. Total nitrogen loadings from major external sources to Hillsborough Bay during two time periods, 1967-68 and 1987-90.

SOURCES	TN (METRIC TONS/YR) 1967-68	1987-90	PERCENT CHANGE
Alafia River + fertilizer industry	1170	430	-63
Hillsborough River	160	280	75
MacDill AFB Wastewater Plant	20	0	-100
Nitram, Inc. + Delaney Creek	870	60	-93
Palm River/Tampa Bypass Canal	50	390	680
Hookers Point Wastewater Plant	1210	240	-80
Cargill Fertilizer, Inc.	350	20	-94
Fertilizer shiploading terminals	140	770	450
TOTAL	3970	2190	-45

The study recommended several short-term measures to improve conditions in Hillsborough Bay. In addition, an extensive long-term water quality management plan was outlined to restore and protect the bay. This plan recommended a 90% reduction of nitrogen loading to the bay from the sources studied to reduce phytoplankton and macroalgae growth. Further, to improve the dissolved oxygen conditions in the bay, it was recommended that the Hookers Point facility remove 90% or more of the carbonaceous material discharged. The study concluded that the degraded conditions in Hillsborough Bay developed over many years and, likewise, efforts to restore the bay would take time. However, it was also suggested that the implementation of the management plan, combined with the natural self-purification process of the estuary, may eventually restore Hillsborough Bay to a healthy aquatic environment.

City of Tampa Wastewater Discharges

The FWPCA (1969) management plan recommended that the City of Tampa upgrade its Hookers Point wastewater treatment plant from primary to secondary treatment, to greatly reduce the discharges of nitrogen and organic material to Hillsborough Bay. The City of Tampa initiated plans to construct an upgraded facility in 1970. The decision to upgrade was based on the recommendations of the FWPCA report and by public outcry in response to the poor conditions in Hillsborough Bay (Tampa Tribune 1969). However, in 1972, before plans for the secondary upgrade had been completed, the Florida legislature passed a law which required all domestic wastewater dischargers to tidal waters of west central Florida to provide advanced wastewater treatment (AWT). AWT was defined as 5 mg/l BOD_5, 5 mg/l total suspended solids, 3 mg/l total nitrogen and 1 mg/l total phosphorus. Directed by this law, the City of Tampa upgraded the Hookers Point facility from primary to AWT with a 60 MGD capacity. Nitrogen removal has been maintained at AWT levels since 1979. The AWT phosphorus requirement has been waived by the state because of evidence indicating that nitrogen is the limiting nutrient for algal growth in Hillsborough Bay and other sections of Tampa Bay (FWPCA 1969, COT 1983, FDER 1983).

Loadings of total nitrogen and carbonaceous material to Hillsborough Bay from the Hookers Point facility were reduced substantially when the plant converted from primary to advanced treatment. Current nitrogen loadings have been reduced by 80% compared to the loadings measured by the FWPCA study (Table 1), despite the fact that the discharge flow from the plant has nearly doubled since 1967-68.

Fertilizer Industry and Alafia River Impacts

The second major source of nutrients identified by the FWPCA (1969) management plan was the Alafia River and the fertilizer industry. The report specifically referenced three sources—Nitram, Cargill, and the Alafia River and the fertilizer companies located in its upper basin.

During the late 1800s, rich phosphate deposits were discovered east of Hillsborough Bay and in 1908 the first large vessels were used for transport of phosphate rock from the Tampa Bay area (Tiffany and Wilkinson 1989). Prior to the 1960s, fertilizer production and export consisted mostly of phosphate rock. Later, the production and export of processed fertilizer containing nitrogen became increasingly important. In 1990, approximately 10 million short tons of ammonium-phosphate product were exported from the Port of Tampa in Hillsborough Bay (TPA, unpublished). This large nitrogen processing industry in Tampa Bay is unique among estuaries.

The FWPCA (1969) study estimated that 63% of the point and nonpoint nitrogen sources entering the bay was derived from the combined contribution of Nitram, Cargill, and the Alafia River and the fertilizer companies located in its upper basin. Current nitrogen loading from the two large nitrogen processing industries, Nitram and Cargill, has been reduced by more than 90% since 1967-68 (Table 1), apparently through improved regulation of the industry and better production practices (Estevez and Upchurch 1985). Similarly, nitrogen loadings from the Alafia River have also been reduced, but to a smaller extent. However, it is unclear what fraction of the Alafia River loadings are contributed by the fertilizer industry. Other sources, such as runoff from agricultural lands and urban areas, must also be important nitrogen sources to the river. The Alafia River, which supplied slightly less nitrogen to Hillsborough Bay than the Hookers Point facility in 1967-68, is now the largest of the nitrogen sources originally identified and measured by FWPCA (1969).

In addition, the Hillsborough River and the Palm River/Tampa Bypass Canal (TBC) system show loading increases between 1967-68 and 1987-90 (Table 1), which may be a result of increased development in the basins. However, a major portion of the increase shown for the TBC system was caused by an unusually large discharge in

September 1988 following a large rain event. Excluding this event would result in an average nitrogen loading from this system of approximately 160 metric tons/year.

Although nitrogen loading from the fertilizer industry sources identified and measured by FWPCA (1969) has been reduced, it should be recognized that the study did not account for nonpoint losses of nitrogen-containing fertilizer from storage facilities and shiploading terminals located near or next to Hillsborough Bay. This loss is mainly in the form of surface runoff and dust. There are no quantitative measurements of these losses. The potential loading to the bay has been estimated from the amount of ammonium-phosphate product shipped from the Port of Tampa (TPA, unpublished). Abu-Hilal (1985) lists estimates of fertilizer product losses, ranging from 0.05% to 1%, from transportation and shiploading at the Port of Aquaba in the Gulf of Jordan. The most conservative estimate, 0.05% loss, has been used in this report to calculate nitrogen losses from the Hillsborough Bay storage facilities and shiploading terminals.

Port of Tampa shipping statistics suggest that nitrogen lost from storage facilities and shiploading terminals was a minor component of nitrogen loading to Hillsborough Bay during the period of the FWPCA study (Table 1). However, based on the amounts currently shipped, the loss of material may now be one of the largest sources of external nitrogen to Hillsborough Bay. In addition, water column measurements of dissolved ammonia in the port area of Hillsborough Bay (Figure 2) often show high concentrations of nitrogen near these facilities in comparison to central Hillsborough Bay locations (COT, unpublished). The concentration peaks in the port area generally occur during, or immediately following, rain events. These field measurements suggest that losses from the loading facilities may be a relatively large nitrogen source to Hillsborough Bay. Furthermore, future loadings from these facilities may become increasingly important, because the amount of nitrogen containing fertilizer shipped from the Port of Tampa is increasing rapidly (TPA, unpublished; Figure 3).

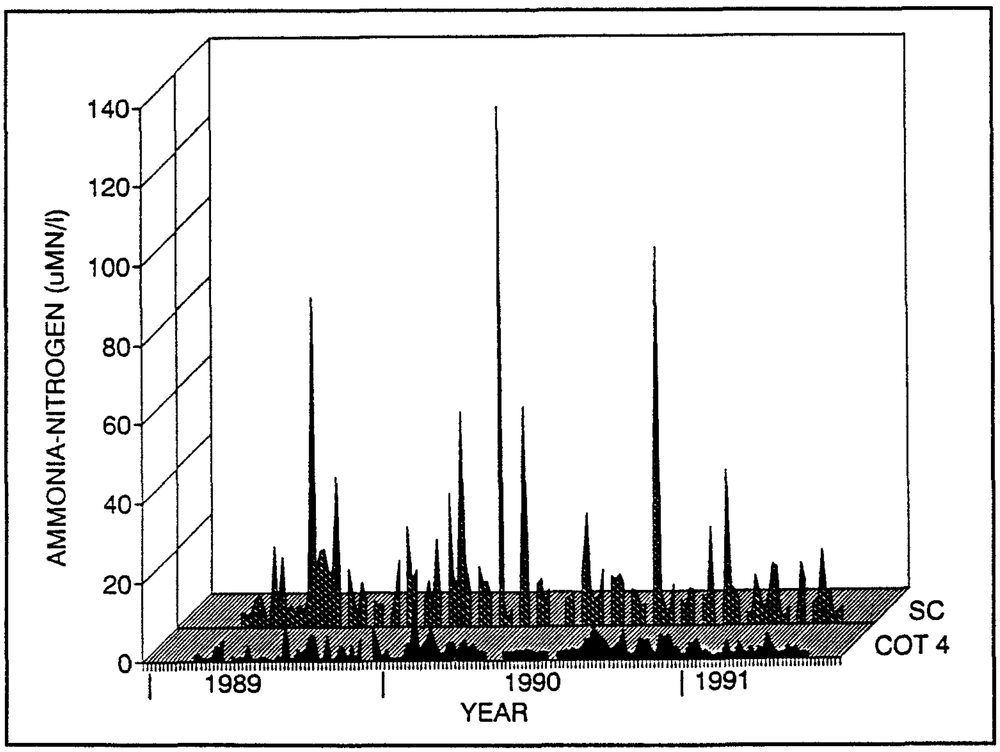

Figure 2. Surface ammonia-nitrogen concentrations in Sutton Channel (SC) and central Hillsborough Bay (COT 4), 1989-1991.

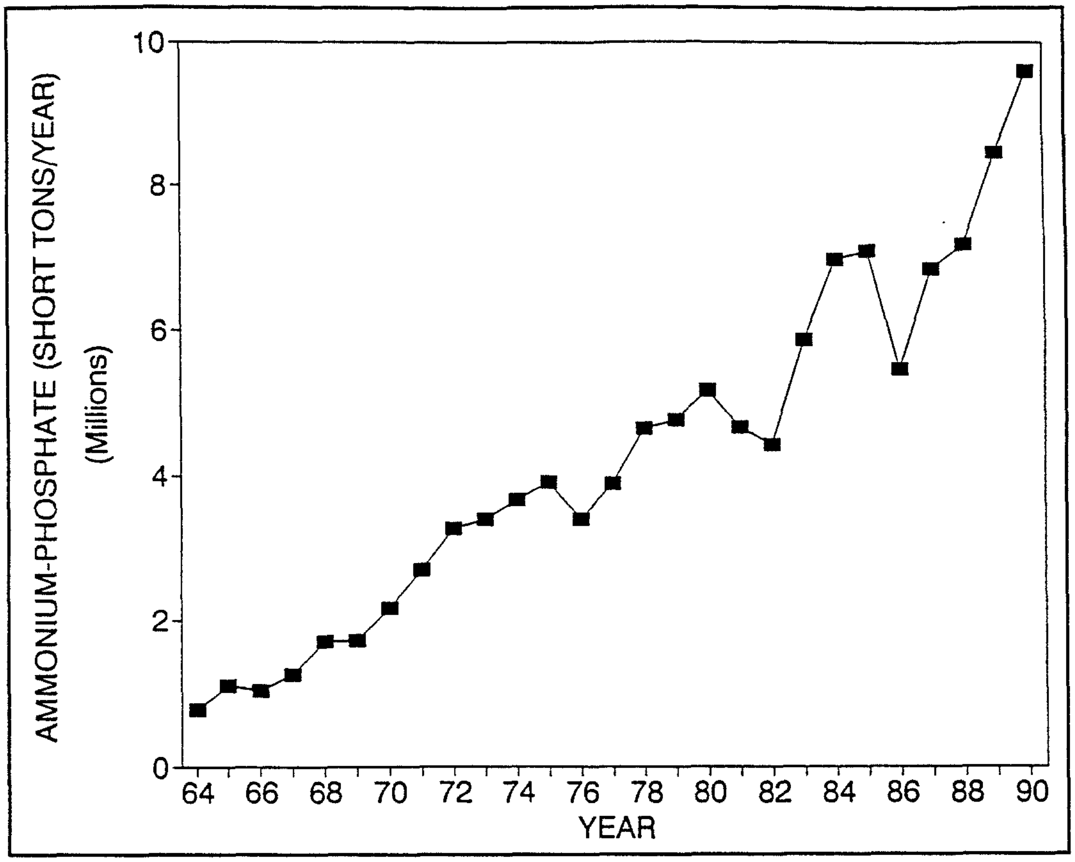

Figure 3. Outbound ammonium-phosphate fertilizer from the Port of Tampa, Hillsborough Bay, 1964-1990.

LONG-TERM TREND OF EXTERNAL NITROGEN LOADING

The comparison of total nitrogen loading measured by FWPCA (1969) for the period 1967-68 and current loadings for the period 1987-90 indicate that there has been a 64% reduction in loading from the sources identified in the FWPCA study. If the estimated losses from the shiploading terminals are included in this comparison, then the reduction is decreased to 45%. The comparison clearly establishes that there has been a substantial reduction of nitrogen loading from these external sources to Hillsborough Bay during the last two decades. However, the nitrogen loading record must be examined in greater detail to determine, as accurately as possible, when the large changes occurred. This detail is needed to establish a meaningful relationship between loading and bay conditions. Annual nitrogen loading rates have been calculated for the largest sources identified in the FWPCA report (the Alafia River, the fertilizer industry and Hookers Point) and the estimated loading from shiploading terminals (Figure 4). Of course, these are not the only external sources of nitrogen to Hillsborough Bay. Atmospheric loadings, most stormwater loadings, and loadings from groundwater, septic waste and several tributaries have not been included in the calculations. However, the included sources may be the most important ones and also the sources that have shown the greatest change over the period of study. Kelly and Harwell (1990) suggest that it may be more relevant to determine the response of a system from a relative change over time than from an absolute understanding of the system.

Information describing the sources used for loading calculations and the procedures used to deal with missing data are detailed in the Appendix. It should be emphasized that loadings for the fertilizer industry are very approximate for the period 1969-80. Figure 4 indicates that nitrogen loading remained relatively high and stable until 1979, when a substantial reduction began. Both the upgrade of Hookers Point and changes in fertilizer industry practices apparently caused the large nitrogen reduction. Loadings have remained relatively low since 1981.

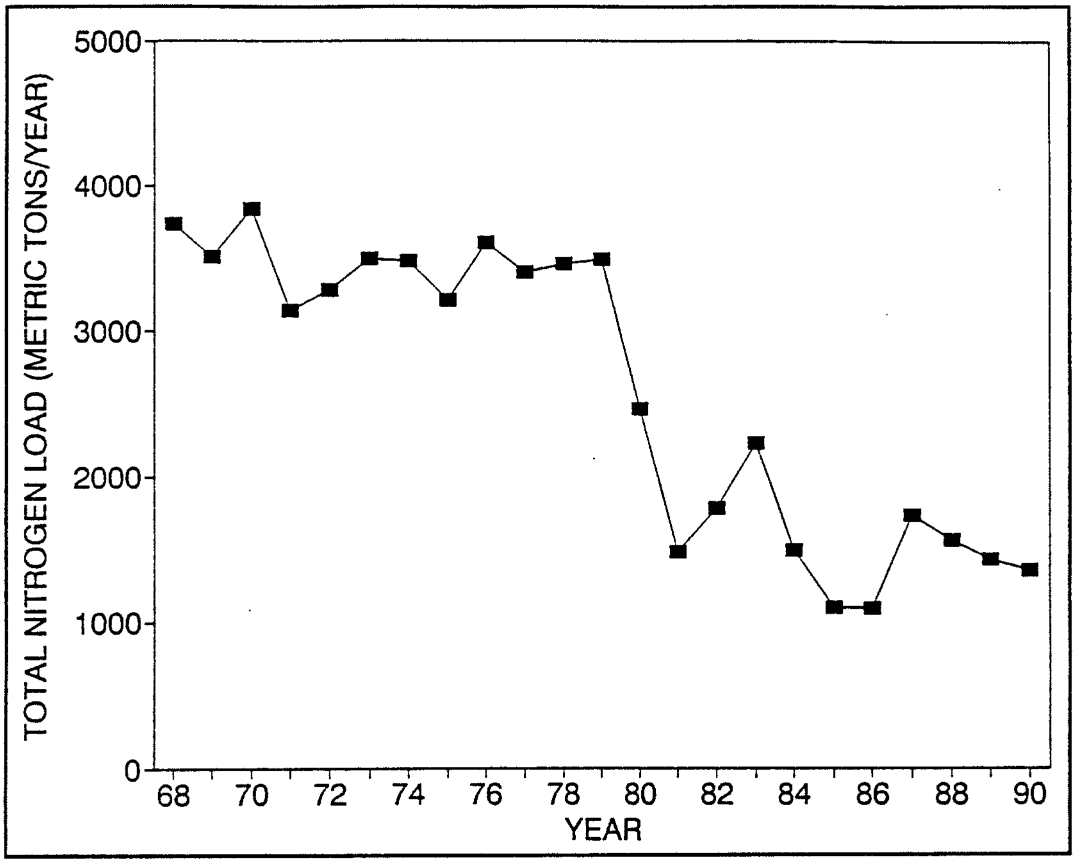

Figure 4. Total nitrogen loading to Hillsborough Bay from major external sources, 1968-1990.

Phytoplankton Biomass and Blue-Green Algae Abundance

With the recent reduction of nitrogen loading to Hillsborough Bay, reductions in phytoplankton biomass and changes in phytoplankton composition could be expected. Several reports discussing nitrogen loading to Tampa Bay have suggested that a direct relationship exists between nitrogen availability and growth of phytoplankton and macroalgae (FWPCA 1969; Spaulding et al. 1989; Johansson and Lewis, in press). Further, this relationship has been demonstrated, particularly for phytoplankton, in many estuaries world wide (Boynton et al. 1982). Chlorophyll *a* concentration is an estimate of phytoplankton biomass, but it is also an important indicator of estuarine eutrophication and has been linked to seagrass survival (Cambridge et al. 1986, Pearce 1991). The long-term Tampa Bay record shows a substantial chlorophyll reduction, particularly in Hillsborough Bay, since the mid-1980s (Figure 5). Sources of the long-term chlorophyll record are identified in the Appendix. Hillsborough Bay annual average concentrations have decreased from approximately 30 µg/l during the period prior to 1984 to the current level of less than 15 µg/l. The chlorophyll decrease is similar to the reductions in nitrogen loading discussed above. Although the large loading reduction occurred just prior to 1980, ambient chlorophyll concentrations did not decrease substantially before 1984. Three years may have been necessary for the bay's internal processes to equilibrate to the new level of nitrogen loading after several decades of excessive loadings from Hookers Point and the fertilizer industry. However, with smaller reductions of nitrogen loading anticipated from future management actions, the time lag may be shorter. Recent information has shown that several estuaries retain nitrogen poorly and may export most of the nitrogen they receive (Nixon 1987; Nowicki and Oviatt 1990; Boynton et al., in press).

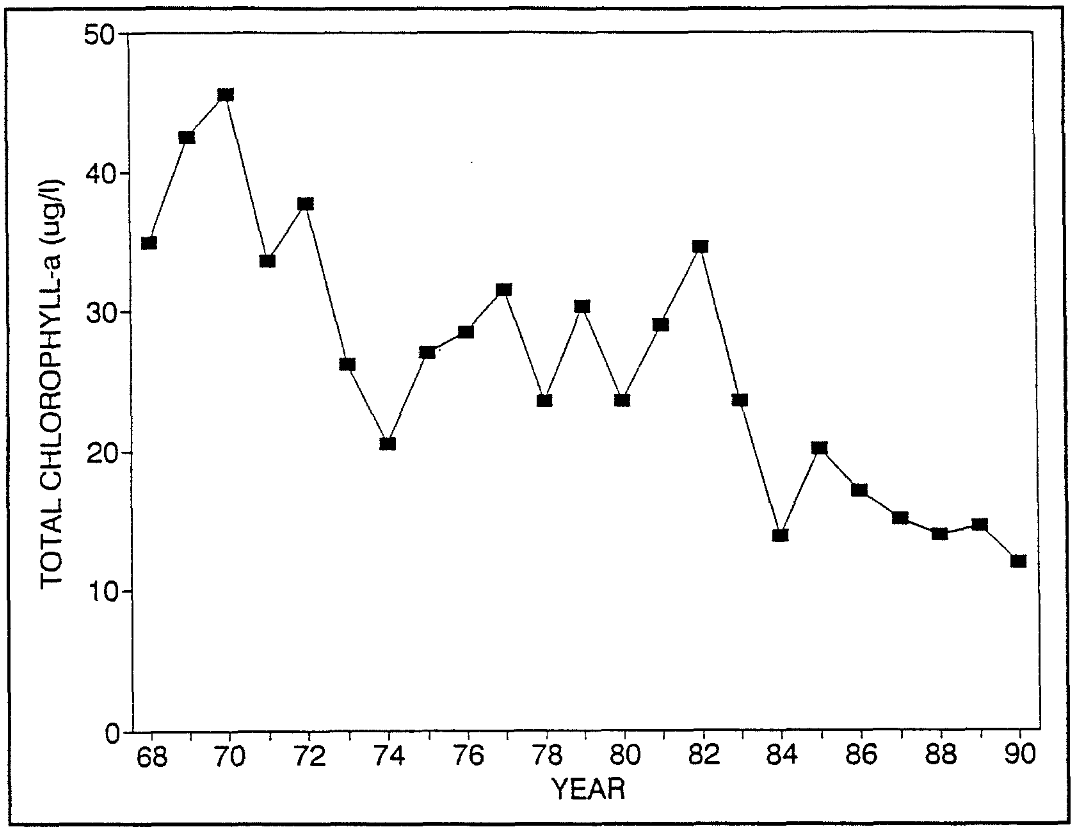

Figure 5. Total chlorophyll a concentrations in Hillsborough Bay, 1968-1990.

The substantial decrease of chlorophyll in Hillsborough Bay correlates with a large reduction of a planktonic filamentous blue-green alga (*Schizothrix calcicola sensu* Drouet), which prior to 1984 dominated the phytoplankton population from late summer to early winter (Johansson et al. 1985; COT, unpublished; Figure 6). This alga has been present in much reduced concentrations since 1984. There is no information to support that this blue-green alga is able to fix atmospheric nitrogen. Nutrient bioassay experiments with natural phytoplankton communities dominated by *S. calcicola*, from both Hillsborough Bay and Old Tampa Bay, clearly demonstrated the community to be strongly nitrogen limited (COT 1983). Therefore, growth of this blue-green alga is apparently limited by available water column nitrogen, and it is not surprising that its biomass has been reduced as nitrogen loadings have decreased. Many blue-green algae are considered nuisance species and are often indicators of poor water quality (Pearl 1987). The reduction of blue-green algae biomass agrees with other indicators suggesting decreased eutrophication in Hillsborough Bay.

Further, the decrease in blue-green algae biomass during late summer to early winter may be a key factor of the recent recovery of Hillsborough Bay. For example, the blue-green reduction should have caused less organic matter (blue-green algae filaments) to settle to the bottom, and sediment oxygen demand should have been lowered (see below). Similarly, there should have been less water column respiration. These changes should have improved water column and bottom oxygen conditions

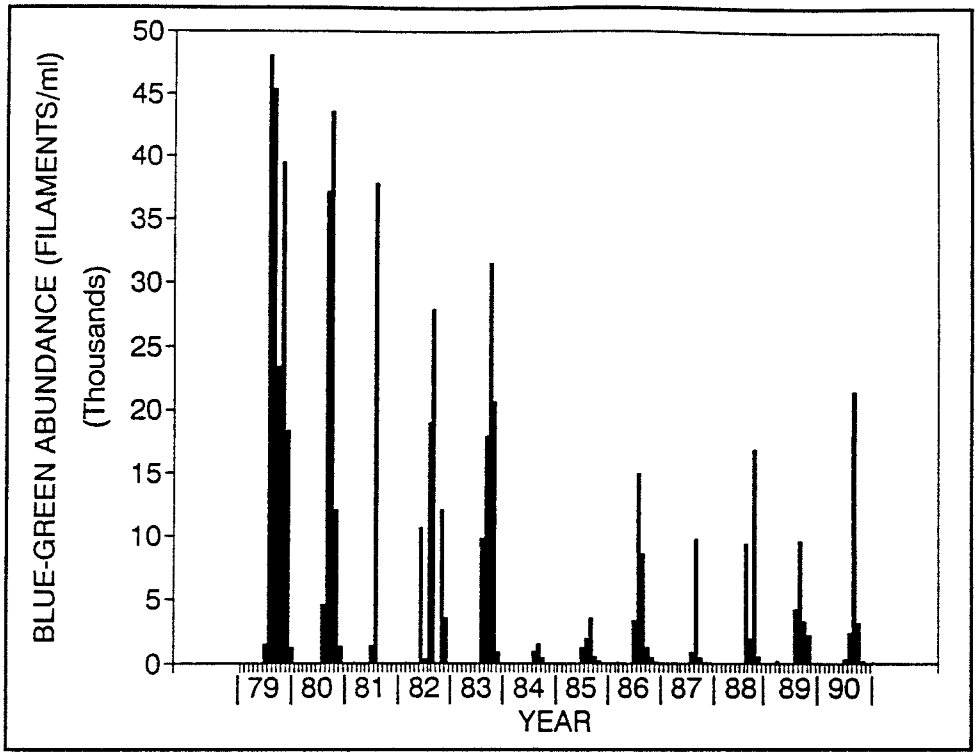

Figure 6. Monthly concentrations of a filamentous blue-green alga (Schizothrix calcicola sensu Drouet) in surface waters of central Hillsborough Bay (COT 4), 1979-1990.

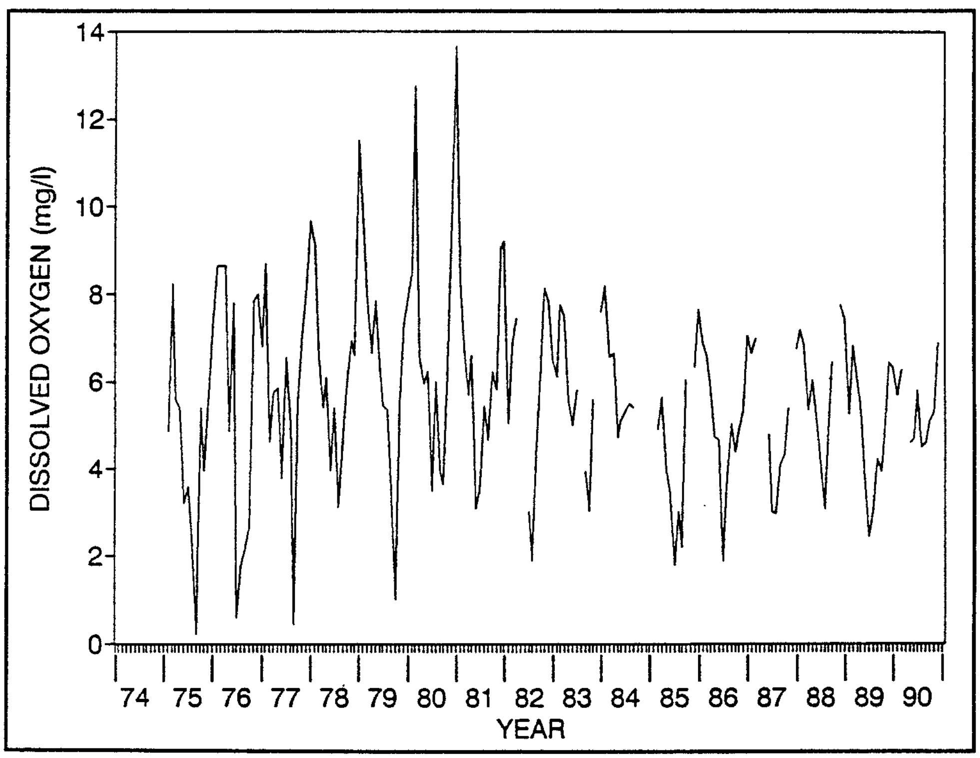

Figure 7. Monthly averages of near-bottom dissolved oxygen concentrations in Hillsborough Bay (EPC 6 and 7), 1974-1990.

during the critical summer period. Evidence that these changes have occurred concurrent with the reduction in blue-green algae abundance is apparent in the long-term near-bottom oxygen record for two stations located in the muddy section of Hillsborough Bay (HCEPC, unpublished; Figure 7). The annual amplitude of the near-bottom dissolved oxygen curve has narrowed considerably compared to the period of high blue-green algae abundance. In addition, hypoxia events have also become less frequent.

Phytoplankton biomass is an important factor limiting water column light penetration in phytoplankton-dominated estuaries such as Hillsborough Bay. Further, the dense concentrations of blue-green algae found in the bay prior to 1984 may have additionally suppressed light penetration. Kirk (1977) found that blue-green algae reduce light to a greater degree than other phytoplankton types. These findings may lend support to the improved Secchi depth readings in Hillsborough Bay after the blue-green algae biomass was reduced (Boler 1990; HCEPC, unpublished; Figure 8). Sufficient water column light penetration is essential for the survival of submerged seagrasses.

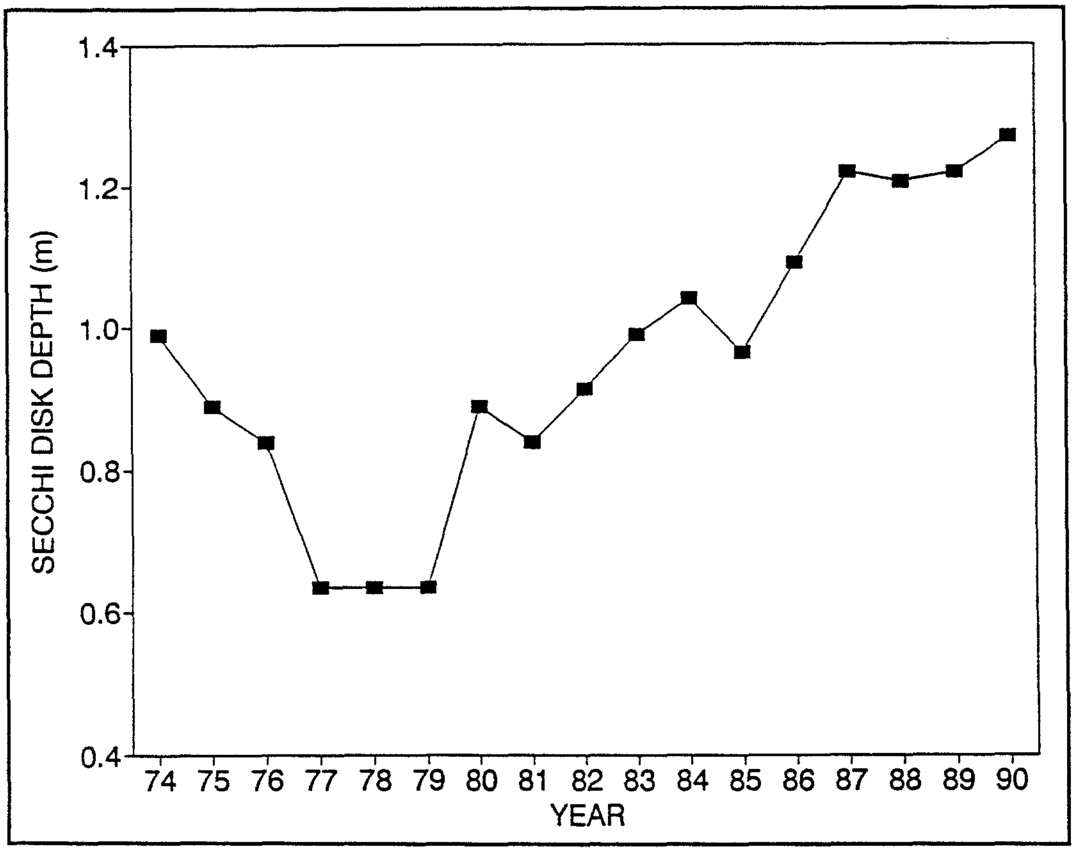

Figure 8. Secchi disk depth in Hillsborough Bay, 1974-1990.

Seagrass Recovery

Hillsborough Bay, and Tampa Bay as a whole, have had serious losses of seagrass. Historic records show that the areal coverage of Tampa Bay seagrasses has decreased dramatically during the last 100 years. In 1982, approximately 20% of the originally estimated seagrass coverage still remained (Lewis et al. 1985). Most seagrasses in Hillsborough Bay were lost between 1950 and 1980. However, modest seagrass recolonization was observed in Hillsborough Bay and other sections of Tampa Bay in 1984 and 1985 following the decrease in chlorophyll concentrations. Seagrass cover continues to increase in these areas. More detailed discussions of this issue are given by Avery (this volume), Johansson and Lewis (in press), and Lewis et al. (this volume).

Rainfall and Chlorophyll Concentration

Nutrient loading from runoff caused by rainfall has been suggested as an important driving force influencing Hillsborough Bay water quality (Lewis and Estevez 1988; Lewis et al., this volume). It is suggested that lower than average rainfall in the Tampa Bay area during the last three to five years may be responsible for the improvements in water quality and the recolonization of seagrasses in Hillsborough Bay and other areas of Tampa Bay. It is often difficult to separate ecosystem responses caused by natural variability from responses caused by management actions (National Research Council 1990). However, a general evaluation of the importance of rainfall on long-term water quality conditions in Hillsborough Bay is attempted here. Figure 9 shows the 1975-90 annual rainfall record at Tampa International Airport (TIA) (NOAA 1991), and at 27 locations in the Hillsborough and Alafia River basins that are monitored by SWFWMD (SWFWMD, unpublished). During the period 1975-90, annual rainfall at TIA has been consistently lower than the average rainfall over the basins. The result of this comparison suggests that many rainfall measuring sites must be used to estimate rainfall affecting Hillsborough Bay. Average rainfall during the recent period of low chlorophyll levels (1984-90) was approximately 6% lower than the long-term record (1975-90). It is unlikely that this small reduction in rainfall could have exclusively caused the substantial reduction in chlorophyll.

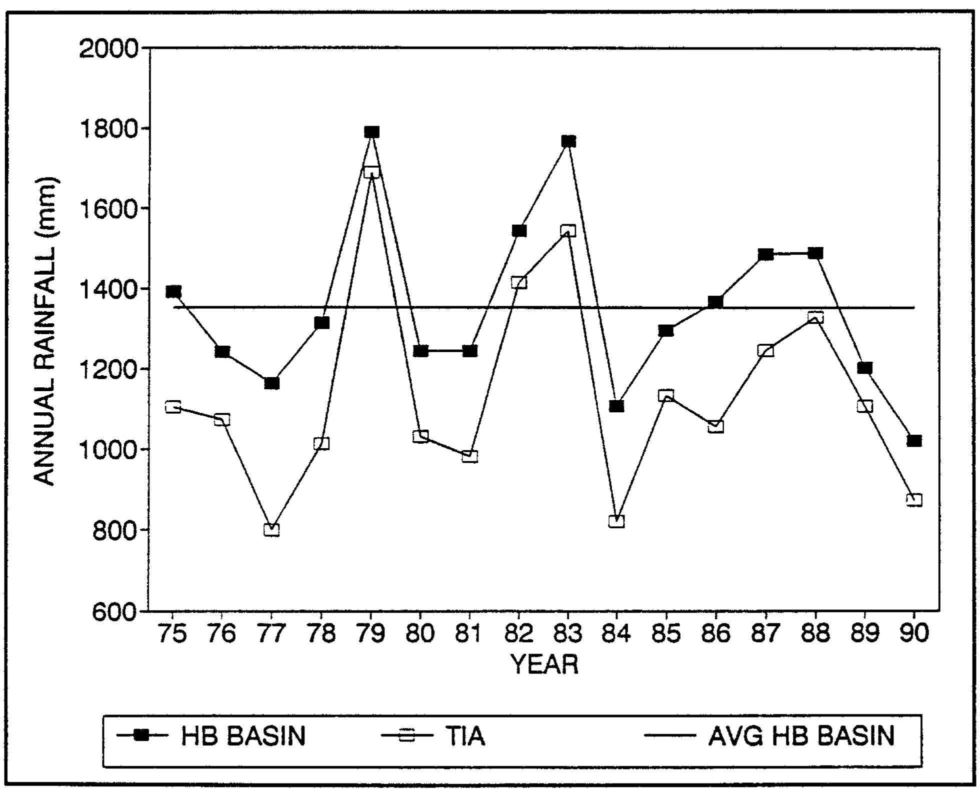

Figure 9. Annual rainfall at Tampa International Airport and at 27 rainfall measuring sites in the Hillsborough Bay basin, 1975-1990.

The relationship between rainfall over the Hillsborough Bay drainage area and ambient chlorophyll concentrations for the period 1975-90 is plotted in Figure 10. The lack of a relationship is encouraging because it implies that management actions taken during the last decade or two, aimed to decrease point-source nitrogen loading to the bay, have apparently been effective in reducing eutrophication.

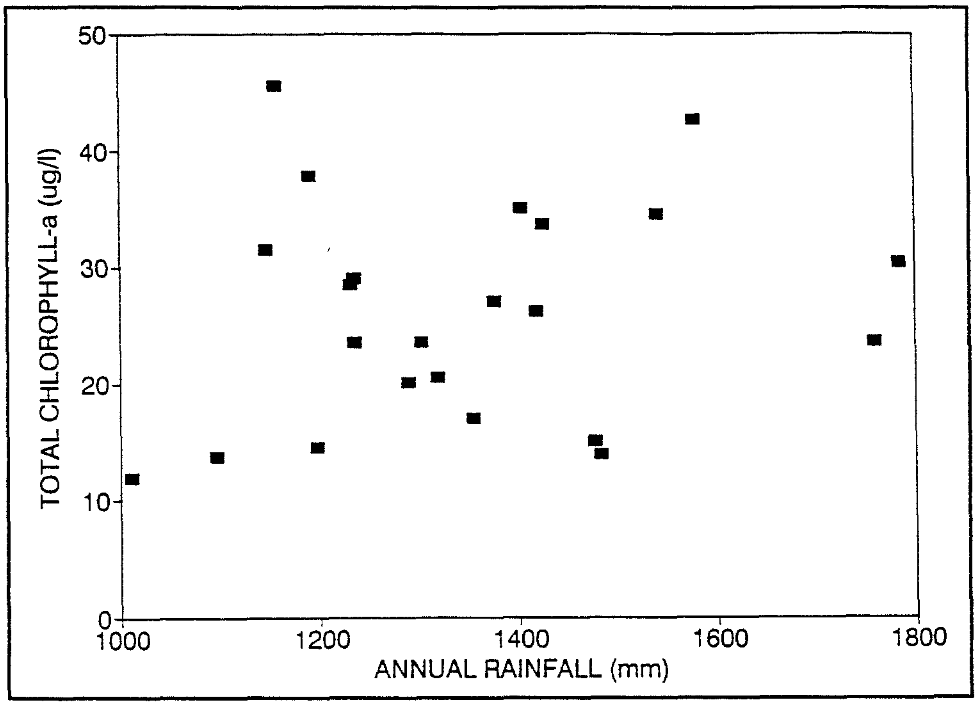

Figure 10. Relationship between annual rainfall over the Hillsborough Bay basin and Hillsborough Bay total chlorophyll a concentrations, 1968-1990.

NITROGEN LOADING AND CHLOROPHYLL CONCENTRATION

The long-term trend of nitrogen loading to Hillsborough Bay from the largest sources identified by the FWPCA in 1969 (the Alafia River, the fertilizer industry, and Hookers Point) and the estimated loading from the shiploading terminals have been plotted against the long-term Hillsborough Bay chlorophyll record (Figure 11). The linear relationship shown is statistically significant ($P>0.01$), but the regression coefficient ($R^2=0.49$) is weak. It is interesting to note that if a three-year time lag (see above) is assumed between nitrogen loading and the response in chlorophyll, then a much stronger regression coefficient is found ($R^2=0.76$) (Figure 12). In either case, the relationship suggests that over the period analyzed, each 150 metric tons/year reduction in nitrogen loading from major external sources has corresponded to a 1 µg/l reduction of ambient chlorophyll. This relationship describes a simple eutrophication model. With additional work to refine the relationship, it could be used by Tampa Bay managers to evaluate eutrophication abatement strategies.

The model must be used with care for predictions of future Tampa Bay chlorophyll concentrations, pending a better understanding of the chlorophyll/nitrogen loading relationship. It can not be assumed that the long-term relationship will remain linear as future nitrogen loading reductions are implemented. Several natural estuarine eutrophication control processes may become increasingly important with additional nitrogen reductions. FWPCA (1969) referred to these controls as processes of self-purification and suggested that these, combined with management actions, may eventually restore Hillsborough Bay to a healthy aquatic environment. Consequently, predictions of future chlorophyll concentrations from

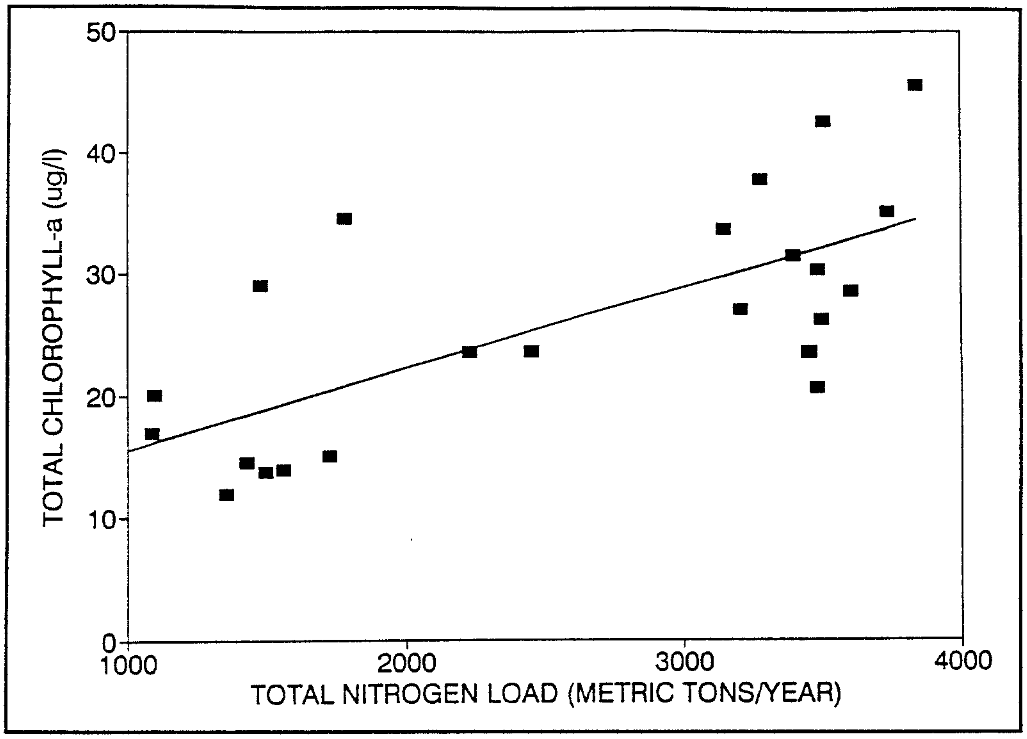

Figure 11. Relationship between total nitrogen loading to Hillsborough Bay from major external sources and Hillsborough Bay total chlorophyll a concentrations, 1968-1990.

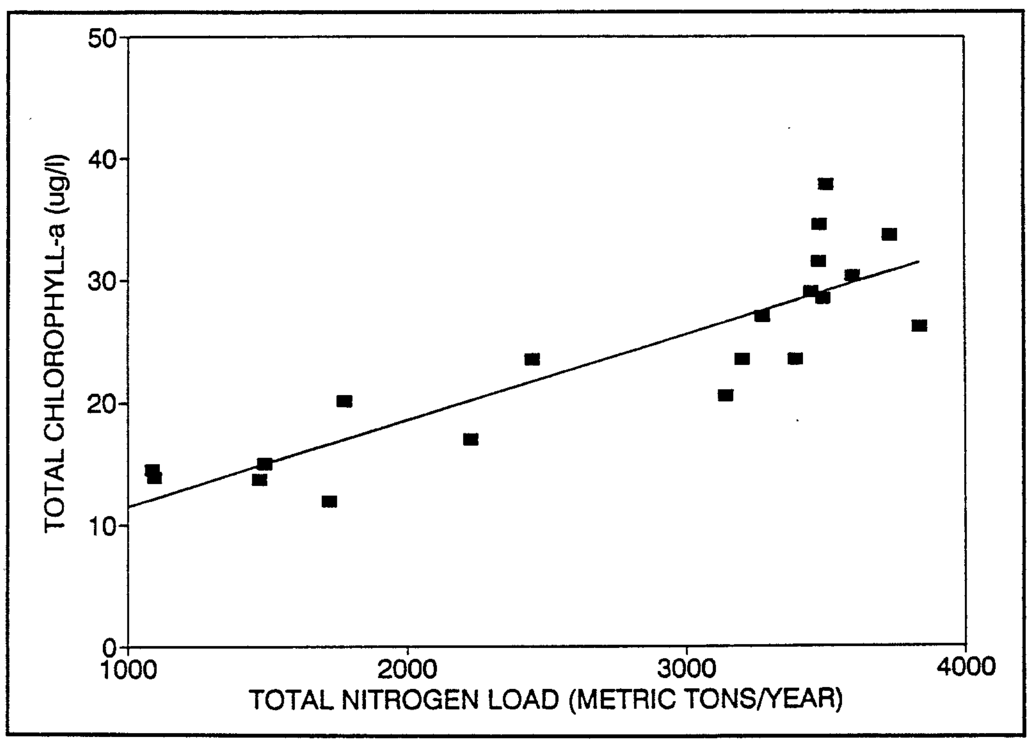

Figure 12. Relationship between total nitrogen loading to Hillsborough Bay from major external sources (1968-1987) and Hillsborough Bay total chlorophyll a concentrations (1971-1990). A three-year time lag has been applied to the chlorophyll concentrations in relation to the nitrogen loadings.

the long-term relationship without accounting for the natural eutrophication control processes may result in overestimated chlorophyll levels. Therefore, it is important to identify and evaluate the natural processes which could have large impacts on the development and the ultimate results of eutrophication management strategies. Three of these processes are addressed below. These may be some of the most important ones; however, there may be other important processes not addressed here.

First, hypoxia of bottom waters during the summer, specifically in Hillsborough Bay, which has the largest area of muddy bottom of the Tampa Bay system (Johansson and Squires 1987; USGS, unpublished), has been an annual phenomenon (Santos and Simon 1980a and 1980b). Although large areas in the deeper sections of the bay still experience hypoxia, it is now evident that both the area covered by and the duration of poor summer oxygen conditions have been reduced (see above and Figure 7). The improvement may be related to the substantial reduction of blue-green algae which occurred in 1984 (see above and Figure 6).

With the recent increase of oxic conditions in time and space it can be assumed that denitrification in the surface sediments has become increasingly important (Kemp et al. 1990; Koop et al. 1990). The loss of nitrogen by denitrification is an important natural eutrophication control process in estuaries, and may account for as much as half of the terrestrial input (Seitzinger 1988). Denitrification losses must therefore be included in Tampa Bay eutrophication strategies to accurately project results of management actions and to avoid costly overprotection. Field measurements and/or extensive comparisons with other estuaries should be conducted to estimate current and future denitrification rates.

Second, the potential of benthic filter feeding communities to reduce phytoplankton biomass and chlorophyll concentrations may increase from present levels as bottom oxygen conditions improve and areas of suitable habitat increase in Tampa Bay. Benthic macroinvertebrate filter feeding has been shown to act as a natural eutrophication control in many shallow estuaries (Officer et al. 1982, Hily 1991). Extremely clear water and low chlorophyll concentrations have been noted concurrent with dense concentrations of a solitary tunicate that has occasionally been present during the winter since 1987 in shallow areas of Hillsborough Bay (Pinson 1991). Also, survival of the benthic filter feeding amphipod *Ampelisca* appears to have improved recently with better bottom oxygen conditions (COT, unpublished). Santos and Simon (1980a and 1980b) attributed large scale die-offs of *Ampelisca* and other benthic animals to annual recurring hypoxia events of bottom waters in Hillsborough Bay during the late 1970s. A comprehensive benthic inventory study and a long-term monitoring program should be initiated. Further, the benthic habitats found in the large muddy areas in Hillsborough Bay should receive protection from damage caused by activities such as shrimp trawling and bait purse-seining. Protection of these habitats would also reduce sediment resuspension (USGS, unpublished).

Third, seagrasses may become important storage of nutrients, and as such, act as natural eutrophication control processes. It has been shown in Chesapeake Bay that seagrasses assimilate nitrogen primarily during spring and summer, and thus effectively reduce the amount of nitrogen available to the phytoplankton during the most active phytoplankton growing period (Kemp et al. 1984). Seagrass mortality and nitrogen release usually occurs during late fall and early winter when the nitrogen demand by the phytoplankton is relatively small. Kemp et al. (1983) estimated that the submerged vascular plant community that existed in Chesapeake Bay in 1960, before large vegetation losses occurred, could have acted as a seasonal sink for 7% of the nitrogen input from external sources. Therefore, the restoration of Tampa Bay seagrass meadows may not only increase physical habitat, particle trapping and food resources, but may also accelerate the recovery from eutrophication.

It is important to estimate and include the natural control processes in predictions of future chlorophyll concentrations. Consequently, before these processes are better understood, use of the nitrogen loading/chlorophyll model is limited to conservative estimates of future chlorophyll concentrations in Hillsborough Bay. With these shortcomings in mind, two examples are given below to illustrate the potential use of the model as a management tool.

First, the loss of nitrogen from the fertilizer loading terminals in Hillsborough Bay has been estimated at approximately 770 metric tons/yr (see above). If this estimate is accurate and the loss was eliminated, then the ambient chlorophyll concentrations in Hillsborough Bay should be reduced from the current 15 μg/l to a conservatively estimated concentration of near 10 μg/l. Actions are now underway to reduce this source and a study is planned to evaluate its current impact on water quality.

Second, if current external loadings to Hillsborough Bay from the sources discussed in Figure 4 were reduced by more than 1000 metric tons/year, a level of 5 μg/l or less of chlorophyll would be reached. This chlorophyll concentration has been suggested as the required level for Tampa Bay seagrass survival and propagation by TBRPC (1989). Improved handling of fertilizer at the shiploading terminals will probably account for a significant fraction of the needed reductions. However, the 1000 metric tons/yr reduction may be a difficult goal to reach in the near-term. The large external nitrogen sources to Hillsborough Bay that were identified several decades ago have already been reduced substantially. Several remaining potentially large sources, all inadequately evaluated, include losses from shiploading terminals, stormwater runoff from urban and agricultural lands, and loadings from the atmosphere, groundwater and septic waste. Assuming that the shiploading terminal losses will be corrected, then it may become difficult to implement additional large scale nitrogen reductions. However, it is encouraging that the projection above does not include the potential increase in future nitrogen losses from natural eutrophication control processes and may, as a result, be overly pessimistic.

These examples have illustrated the potential of the nitrogen loading/chlorophyll model to predict future Tampa Bay conditions and responses to management actions. Additional work to refine the relationship include improved loading estimates and increased knowledge of natural control processes. Further, the relationship needs to be linked to important natural resources, such as seagrasses and soft bottom communities, to help protect these and other valuable resources. A model based on the nitrogen loading/chlorophyll relationship is currently used in an Australian coastal embayment to manage and protect seagrasses (Pearce 1991). In Chesapeake Bay, a similar model is proposed to maintain adequate bottom oxygen conditions and protect benthic communities (Boynton et al., in press).

CONCLUSION

Several important water quality and biological indicators have recently improved in Hillsborough Bay in an apparent response to a substantial reduction in nitrogen loading to the bay from many important external sources. The reduction may have been as large as 50% and is related to actions aimed at reducing nutrient loadings to Tampa Bay, specifically from wastewater plants and the fertilizer industry. There is little evidence that the recent lower than average rainfall has been a significant factor of the improvement. Seagrasses have apparently responded to the improved water quality by colonizing shallow areas in Hillsborough Bay and other sections of Tampa Bay. The return of seagrass meadows is in an important sign of bay recovery.

In this report, a first attempt has been made to relate the long-term nitrogen loading to Hillsborough Bay from external sources to water quality conditions in the bay. A simple eutrophication model based on the nitrogen loading/chlorophyll relationship is presented. However, much work is still needed to refine the relationship. Specifically, the major external nitrogen sources and the natural eutrophication control processes should be better quantified. Atmospheric loadings, most stormwater loadings, and loadings from groundwater, septic waste and several tributaries have not been included in the current calculations due to lack of adequate information. However, the sources which have been addressed are some of the most important ones and also the ones which probably have shown the greatest change over the period of study. It is important that the eutrophication model is based on a long-term record spanning several decades to allow for separation of natural and manmade effects, and also to reduce the potential for costly overprotective actions. The model should be used in combination with a comprehensive monitoring program of loading sources, water quality, biological indicators and important natural control processes. Further, the nitrogen loading/chlorophyll model should be linked to valuable natural resources, such as seagrasses and soft bottom communities, to help protect these and other important resources in Tampa Bay.

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APPENDIX

The source of all nitrogen loadings for the period 1967-68 (Table 1) and the year 1968 (Figure 5), except shiploading losses, are from FWPCA (1969). Shiploading losses have been estimated as 0.05% of the amount of ammonium-phosphate product shipped from the Port of Tampa (TPA, unpublished).

Loadings from rivers and creeks for the period 1969-90 have been calculated from the sources listed below and at the locations described by Johansson and Lewis (in press). However, the Alafia River lacks concentration information for the period 1969-1973; therefore, concentrations for 1969-73 were calculated from 1969-73 flows and a regression relationship between flows and concentrations for the period 1973-79. Further, Delaney Creek lacks flow information for the period 1969-84 and concentration measurements for the period 1969-80. Therefore, 1980-84 loadings were calculated from a regression relationship between Delaney Creek and Bullfrog Creek flows for the period 1985-89. Delaney Creek loadings for the period 1969-79 were estimated to be equal to 1968 Nitram loadings.

River and creek flows are from USGS (1976, 1977, 1978, 1979, 1980a, 1980b, 1982, 1983, 1985, 1986a, 1986b, 1987, 1988, 1989, 1990), USGS (unpublished) and SWFWMD (unpublished). Concentrations are from Wilkins (1980, 1981, 1982), Cardinale and Boler (1984), Boler (1986, 1988, 1990) and HCEPC (unpublished). Additional Alafia River concentrations are from USGS flow references.

Cargill Fertilizer, Inc. supplied point-source loadings for the period 1981-90. Loadings for the period 1969-80 were estimated to be equal to 1968 loadings.

Hookers Point Wastewater Treatment Plant loadings for the period 1975-90 are from plant operation reports. Loadings for the period 1969-74 have been estimated from a linear relationship between 1968 and 1975 measurements.

McDill AFB Wastewater Treatment Plant has had no direct discharge to the bay during the last decade.

Chlorophyll concentrations are from FWPCA (1969), Saloman and Taylor (1972), Saloman (1973, 1974), Collins and Finucane (1974), Saloman and Collins (1974), Shaw and Wilkins (1975, 1976, 1977, 1978), Wilkins (1979, 1980, 1981, 1982), Cardinale and Boler (1984), Boler (1986, 1988, 1990), and HCEPC (unpublished).

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STATUS OF NATURALLY OCCURRING AND INTRODUCED *HALODULE WRIGHTII* IN HILLSBOROUGH BAY

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done 5/7/99

ABSTRACT

In 1986, the City of Tampa Bay Study Group (BSG) started a seagrass program to complement other ongoing research assessing the environmental status of Hillsborough Bay. The initial survey in 1986 located 137 patches of naturally occurring Halodule wrightii (Ascherson). The areal coverage of these patches was nearly 2000 m^2. A second survey in 1989 found about 400 patches with areal coverage totaling 5000 m^2. In 1986, eight patches were selected as study sites. Total areal coverage of the study sites increased from 150 m^2 in 1986 to 1175 m^2 in 1990. In 1987, the BSG transplanted H. wrightii to eight locations in Hillsborough Bay. About 10.7 m^2 of donor material were transplanted as sod blocks to seven areas and 2.3 m^2 of material were used to plant bare root units in the eighth area. In 1990, three years after planting, transplant areal coverage was estimated at 825 m^2. Many intertidal and shallow subtidal areas in Hillsborough Bay are being recolonized by H. wrightii, apparently in response to recent improvements in water quality. As a result of increased water clarity, sufficient light may be available to allow seagrass recolonization to occur in shallow areas of Hillsborough Bay. If Hillsborough Bay water quality continues to improve, H. wrightii should recolonize most areas which historically had seagrass coverage.

INTRODUCTION

Seagrass meadows are an important constituent of the Tampa Bay ecosystem. They are among the most productive environments on earth (McRoy and McMillan 1977) and provide food, shelter and a nursery for many fish and invertebrates (Phillips 1960, Livingston 1984). Seagrass can significantly contribute to the organic carbon budget (Kemp et al. 1984) and are used indirectly in detrital food chains (Thayer et al. 1975). Established seagrass beds can improve water quality by reducing water velocity through the leaf canopy, which may enhance particulate settling rates (Ginsburg and Lowenstam 1958), and by decreasing the potential for erosion and resuspension of benthic sediments (Kemp et al. 1984).

Environmental degradation related to urbanization around Tampa Bay has resulted in an 81% reduction of historical seagrass coverage in the past century (Lewis et al. 1985). Lewis and Phillips (1980) reported a 79.6% loss of seagrass coverage in Hillsborough County from 1876 to 1980, with significant losses occurring in the past thirty years.

In 1960, Phillips (1962) surveyed seagrass coverage in Tampa Bay. In western Hillsborough Bay, between Gadsden Point and Ballast Point, extremely sparse coverage of *Diplanthera wrightii* (*Halodule wrightii*) was noted and *Ruppia maritima* was the only species observed north of Ballast Point. In eastern Hillsborough Bay, *R. maritima* was found near Delaney Creek, with sparse coverage observed south of the Alafia River. Phillips found *H. wrightii*, *R. maritima* and *Syringodium filiforme* in the Big Bend area, which is just north of Apollo Beach. Two decades later, Lewis et al. (1985) reported Hillsborough Bay supported only ephemeral *R. maritima* beds.

Hillsborough Bay has been severely impacted by urbanization. Dredge and fill operations have increased turbidity and removed substrate which may provide a suitable habitat for seagrass (Lewis 1977). Also, nutrient inputs from municipal and industrial sources caused increases in phytoplankton and macroalgae biomass, which reduce light available to seagrass beds. In the late 1960s, the Federal Water Pollution Control Administration (1969) implicated the City of Tampa Hookers Point Wastewater Treatment Plant as a major point source polluter in Hillsborough Bay.

In the past two decades, measures have been taken to alleviate impacts of municipal and industrial discharge to Hillsborough Bay. The City of Tampa upgraded the Hookers Point facility to advanced wastewater treatment in 1979, which resulted in the removal of over 90% of BOD, suspended solids, and nitrogen. Also, nutrient loadings to freshwater systems which drain into eastern Hillsborough Bay have been reduced due to increased regulation of the fertilizer industry (Estevez and Upchurch 1985; Johansson and Lewis, in press).

Reduction in nutrient loadings to Hillsborough Bay has resulted in substantial improvement in water quality since the mid-1980s. Water quality information collected by the City of Tampa Bay Study Group (City of Tampa 1988) and the Hillsborough County Environmental Protection Commission (Boler 1990) show a decrease of phytoplankton biomass concurrent with greater water clarity. Also, the Tampa Bay Regional Planning Council (1989) reviewed information on chlorophyll *a* and noted that chlorophyll *a* concentrations, after a period of elevation from 1969-1983, have returned to pre-1969 levels.

Aerial photographs from 1983-1986 provided evidence of minor seagrass renewal in Hillsborough Bay. This evidence prompted the City of Tampa Bay Study Group (BSG) to initiate a program to investigate the response of seagrass to improvements in water quality. Documentation of seagrass coverage in Hillsborough Bay began in April 1986 with a thorough groundtruthing effort in which each patch of *H. wrightii* was measured for areal coverage and mapped. A second groundtruthing effort was completed in October 1989. In addition, study sites have been established to follow seasonal trends of *H. wrightii*. In June 1986, the BSG and Mangrove Systems, Inc. (MSI) groundtruthed an area of eastern Middle Tampa Bay for seagrass. The groundtruth data was used to produce maps documenting re-establishment of *H. wrightii* between 1983 and 1988 using aerial photographs.

In 1987, the BSG, in cooperation with the Florida Department of Natural Resources and National Marine Fisheries Service Tampa Bay Experimental Seagrass Planting Effort, transplanted *H. wrightii* into Hillsborough Bay (City of Tampa 1991). This effort was designed to locate areas of Hillsborough Bay suitable for seagrass transplanting, to establish a source of vegetative material, and to determine if artificially introduced seagrass could generate functional seagrass communities.

STUDY SITES

Bay Study Group — Mangrove Systems, Inc.

In June 1986, the BSG and MSI sampled six transects, oriented perpendicular to the shoreline, in the intertidal-subtidal zone in eastern Middle Tampa Bay (Mangrove Systems, Inc. 1986, City of Tampa 1988). Two transects were previously examined by Lewis and Phillips (1980). The species composition and percent cover of seagrass and drift algae were estimated at 80 m intervals along each transect. Data from the groundtruth effort and color aerial photographs (scale 1"=500′) were used to compare seagrass coverage in eastern Middle Tampa Bay between 1983 and 1986. Subsequently, the City of Tampa contracted MSI to estimate seagrass coverage in an area delineated by a 3 x 1 km rectangle located approximately 350m from shore utilizing vertical aerial photographs from 1983-1986 and 1988 (Figure 1). This area, with little macroalgae cover and only one seagrass species, *H. wrightii*, was selected to minimize interpretive errors.

Natural Seagrass

The City of Tampa (1988) selected eight patches of *H. wrightii* in Hillsborough Bay, representing a variety of geographic locations and substrate characteristics, for detailed study (Figure 2). A square grid (Figure 3) was set up to encompass each *H. wrightii* patch and the boundaries of each patch were measured in relation to the grid at 0.5 m intervals.

Areal coverage, short shoot density, blade length, salinity, surface water temperature, and depth were recorded in spring, summer, and fall surveys. In addition, subjective observations on epiphytic cover and condition of the seagrass were noted. The boundary of a patch was plotted on paper and the areal coverage calculated using a planimeter.

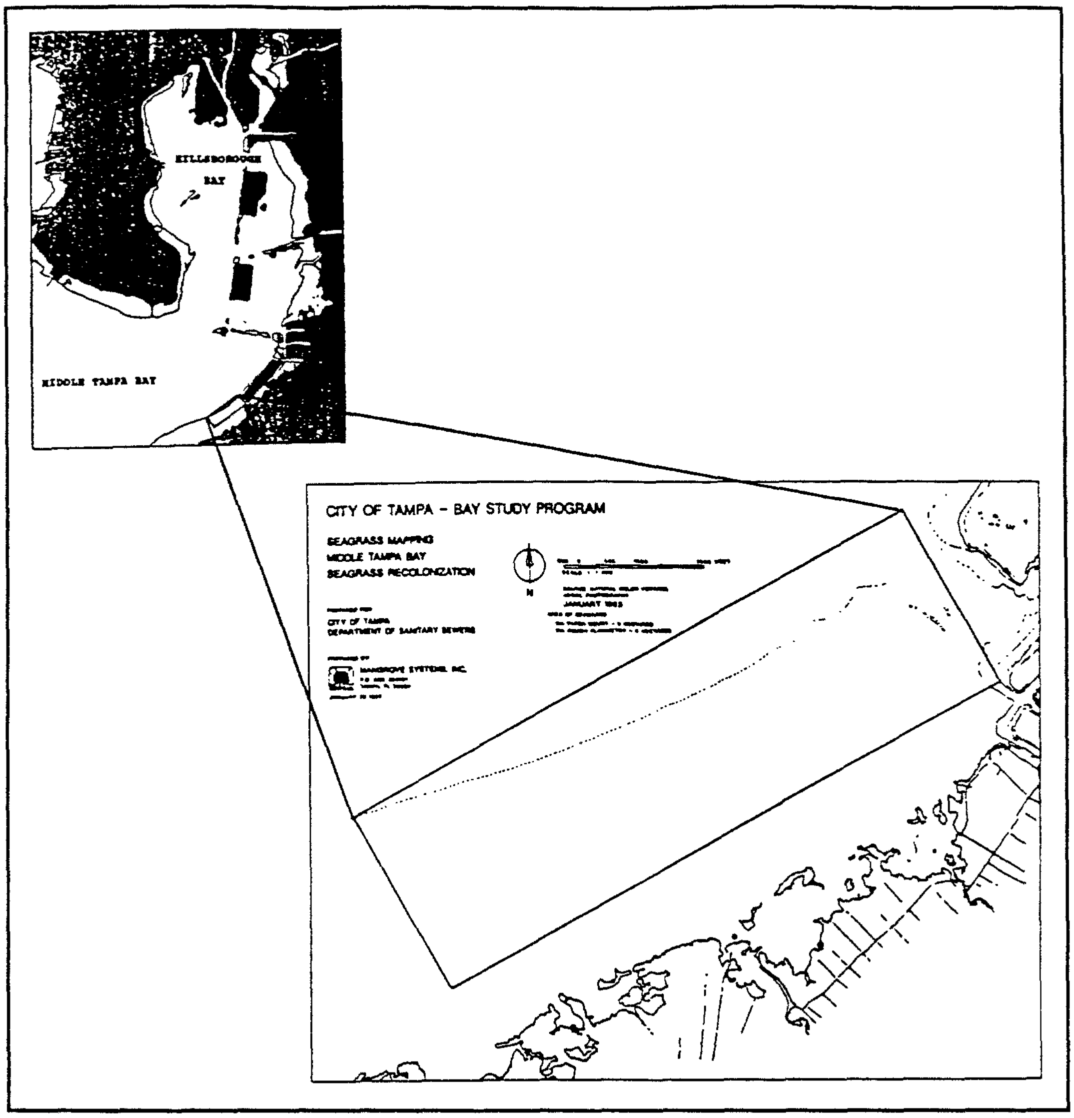

Figure 1. Halodule wrightii study site in eastern Middle Tampa Bay.

Transplanted Seagrass

Halodule wrightii sod blocks and bare root units were used in the 1987 transplanting effort. Source material was obtained from beds to be impacted from the widening of the Courtney Campbell Causeway traversing Old Tampa Bay. Approximately 350 sod blocks were planted in seven areas around Hillsborough Bay (Figure 4). An average sod block measured 14 x 23 x 15 cm and contained 170 short shoots and 23 apical meristems. In the eighth area (Figure 4), 861 bare root units were planted in a 10 x 20 m plot using methods described by Fonseca et al. (1987). An average bare root unit contained an average of 15 short shoots and 3 apical meristems.

Each transplant area was visited in the spring, summer and fall. Areal coverage, short shoot density, blade length, epiphytic cover, salinity, surface water temperature, and depth were recorded. The major and minor axis of each sod block was measured and the areal coverage estimated using the formula for an ellipse. In the first four surveys, 10% of the bare root units were randomly selected for an estimate of areal coverage. Areal coverage of each selected unit was estimated using the formula for an ellipse. Total areal coverage was estimated by multiplying the average bare root unit areal coverage times the number of surviving units. However, after the fourth survey, the bare root units began to coalesce and the areal coverage was determined using the grid method as in the natural seagrass study sites.

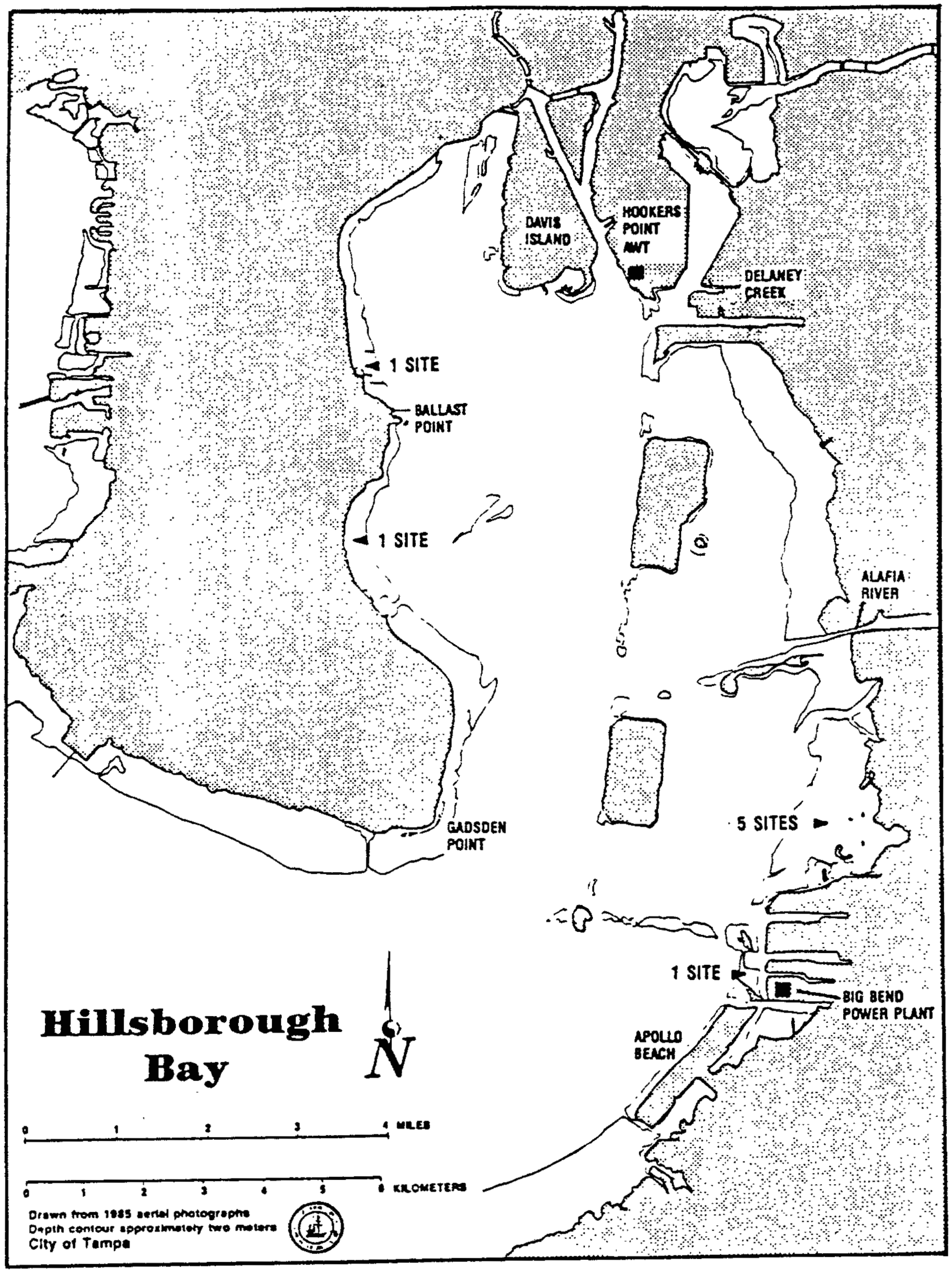

Figure 2. Halodule wrightii study sites in Hillsborough Bay.

Aerial Surveys

Seagrass surveys of Hillsborough Bay and adjacent areas have been conducted using on-site groundtruthing and aerial photography. Monthly low altitude surveys by helicopter and annual high altitude surveys by fixed-wing aircraft are used for oblique and vertical photography, respectively.

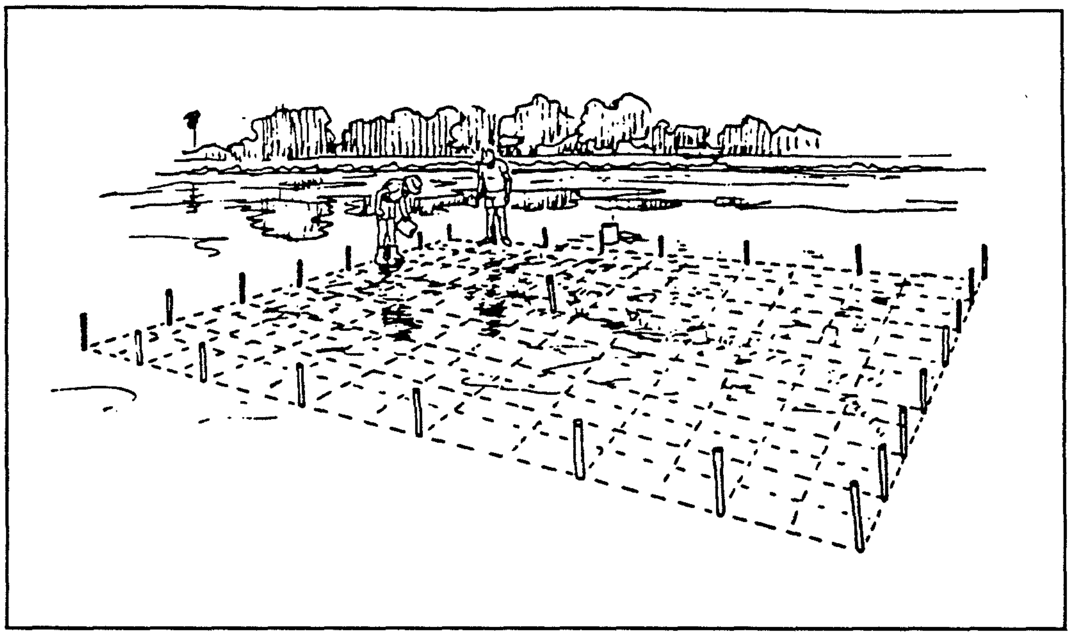

Figure 3. Grid system of a Halodule wrightii study site with PVC pipes at 1 m intervals. Dashed lines represent measuring tape placement for measurement of patch dimensions.

RESULTS AND DISCUSSION

Middle Tampa Bay Survey

In the six transects sampled in eastern Middle Tampa Bay, *Halodule wrightii* was the dominant species observed (Mangrove Systems, Inc. 1986). *Thalassia testudinum* was the next most frequently observed species and areas of *Ruppia maritima* were observed in the shallow inshore areas at each transect. MSI concluded there was greater seagrass coverage in the southern transects in 1986 compared to 1983. Further, inshore stations at each transect had greater seagrass coverage in 1986.

Recent seagrass recolonization has been documented in the 3 km^2 study area mapped by MSI. MSI reported no *H. wrightii* in the study area in the 1983 or 1984 aerial photographs. The 1985 photograph indicated seagrass renewal in the study area and areal coverage was determined at 6.4 ha. Subsequent photographs from 1986 and 1988 showed about 13 ha and 60 ha of *H. wrightii*, respectively (Figure 5).

Hillsborough Bay 1986 and 1990 Surveys

The BSG has completed two thorough groundtruth efforts to document *Halodule wrightii* coverage in Hillsborough Bay. The initial effort in 1986 located 137 patches of *H. wrightii* with total areal coverage of nearly 2000 m^2. In the second survey conducted three years later (October 1989), the BSG located 394 patches of *H. wrightii*, an increase of 190%, and the total areal coverage had increased by 140% to 4700 m^2. After reviewing photographs from monthly Hillsborough Bay overflights and evaluating study site information, the BSG estimated *H. wrightii* coverage to be 8000 m^2 in October 1990.

Seagrass recolonization patterns may reflect spatial differences in Hillsborough Bay water quality. Recolonization of shallow tidal flats in western Hillsborough Bay has occurred at the same elevation as seagrass coverage observed in 1986 and there has been no indication of recolonization in deeper areas. In contrast, recolonization in southeastern Hillsborough Bay has occurred at the same elevation as pre-existing beds and in areas 25-50 cm deeper than seagrass coverage observed in 1986. Recolonization trends in southeastern Hillsborough Bay may reflect better water quality in this portion of the bay.

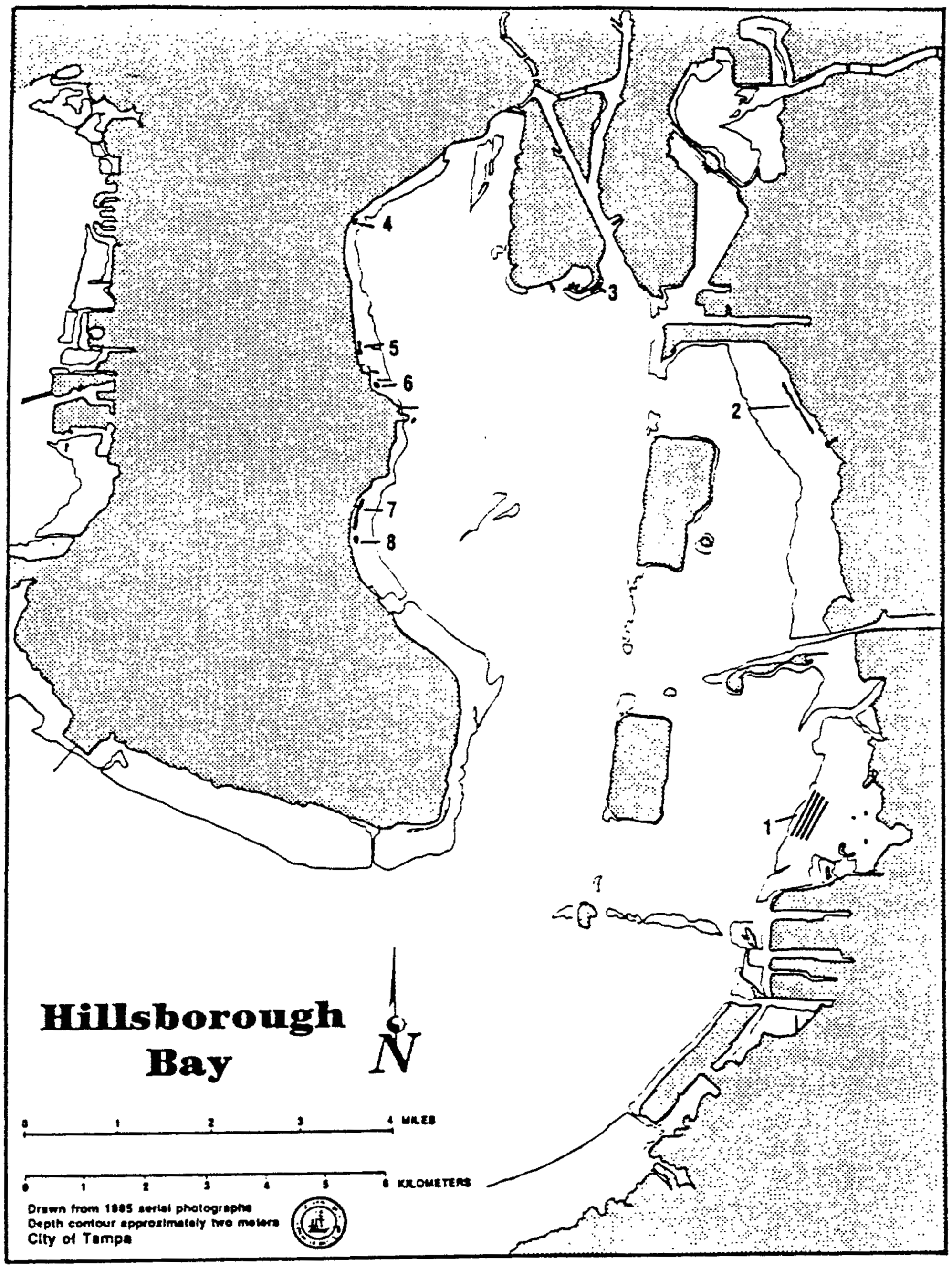

Figure 4. Halodule wrightii test planting sites.

Hillsborough Bay Study Sites

Total areal coverage for the study sites of natural *Halodule wrightii* was 147 m^2 in 1986 and had increased to 1178 m^2 in 1990 (Figure 6). Total areal coverage calculated in 1990 has included coalition of coverage at three study sites with adjacent patches of *H. wrightii* not previously included in the calculations.

The study sites were originally selected to estimate *H. wrightii* growth rates in Hillsborough Bay. However, the information was also used to estimate 1989 *H. wrightii* areal coverage in Hillsborough Bay for comparison with the areal coverage determined from the 1989 groundtruthing effort. Estimates of areal coverage using study site information were 40% higher than areal coverage determined by the groundtruth effort. Therefore, extrapolations made from study site information cannot be used to accurately estimate total *H. wrightii* coverage. Frequent surveys using aerial photography and on-site groundtruthing are imperative for accurate determination of seagrass coverage.

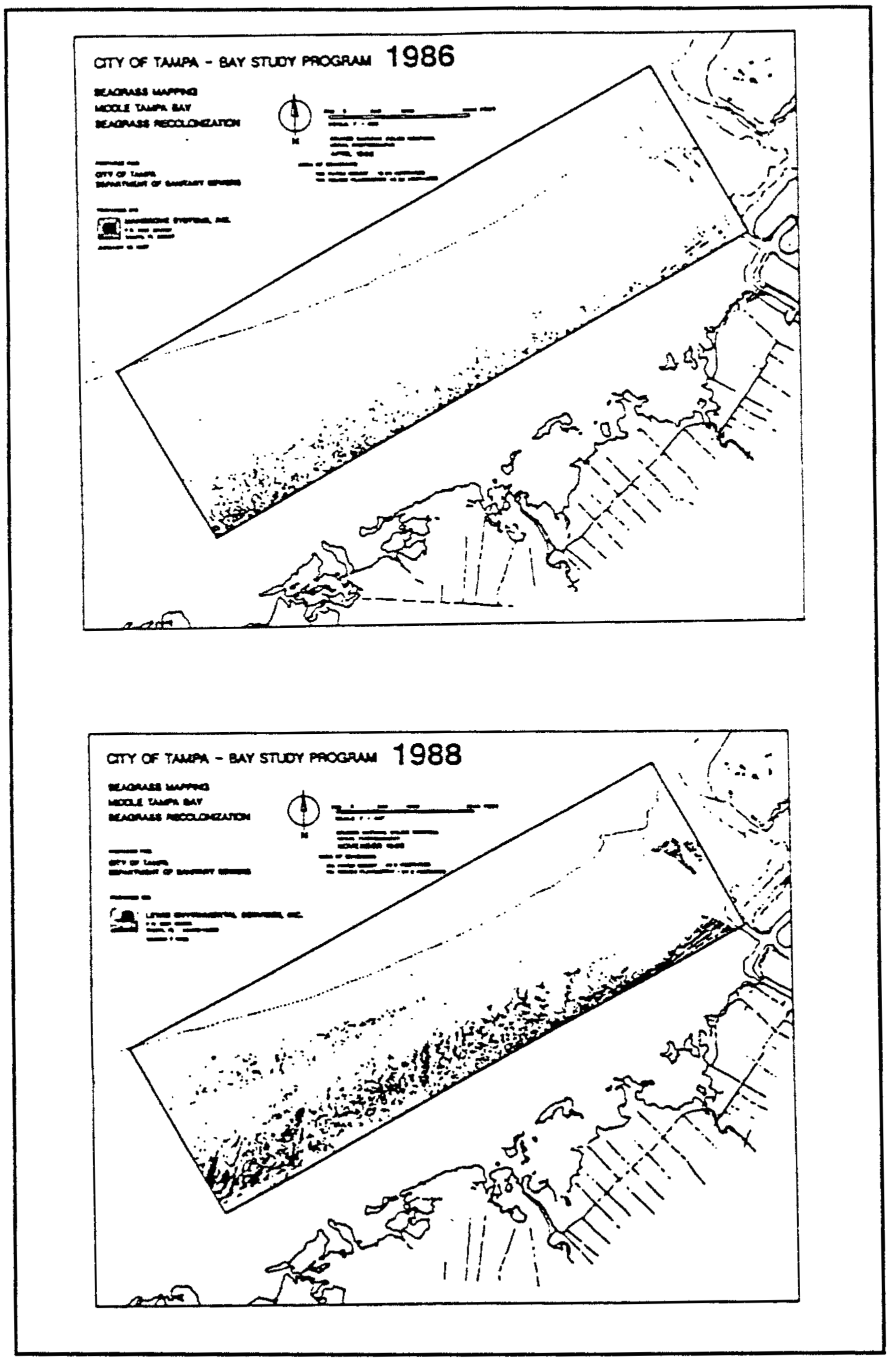

Figure 5. Halodule wrightii coverage in eastern Middle Tampa Bay in 1986 and 1988.

Natural Recolonization

Halodule wrightii coverage appears to be increasing rapidly in several areas of Hillsborough Bay. It is uncertain how revegetation of areas barren of seagrass is initiated as there has been no documentation of *H. wrightii* seed production in

Hillsborough Bay. However, bioturbation of seagrass, probably by the stingray, *Dasyatis sabina*, is a frequent occurrence and is one method to initiate rhizome dispersal. Also, *H. wrightii* may be uprooted by natural events (erosion by waves or tides) or by anthropogenic impacts (ship-generated waves, propeller cuts, dredge and fill). The polychaete, *Diopatra cupraea*, has been observed to add seagrass fragments to its habitat. These fragments may subsequently root and provide vegetative material for colonization.

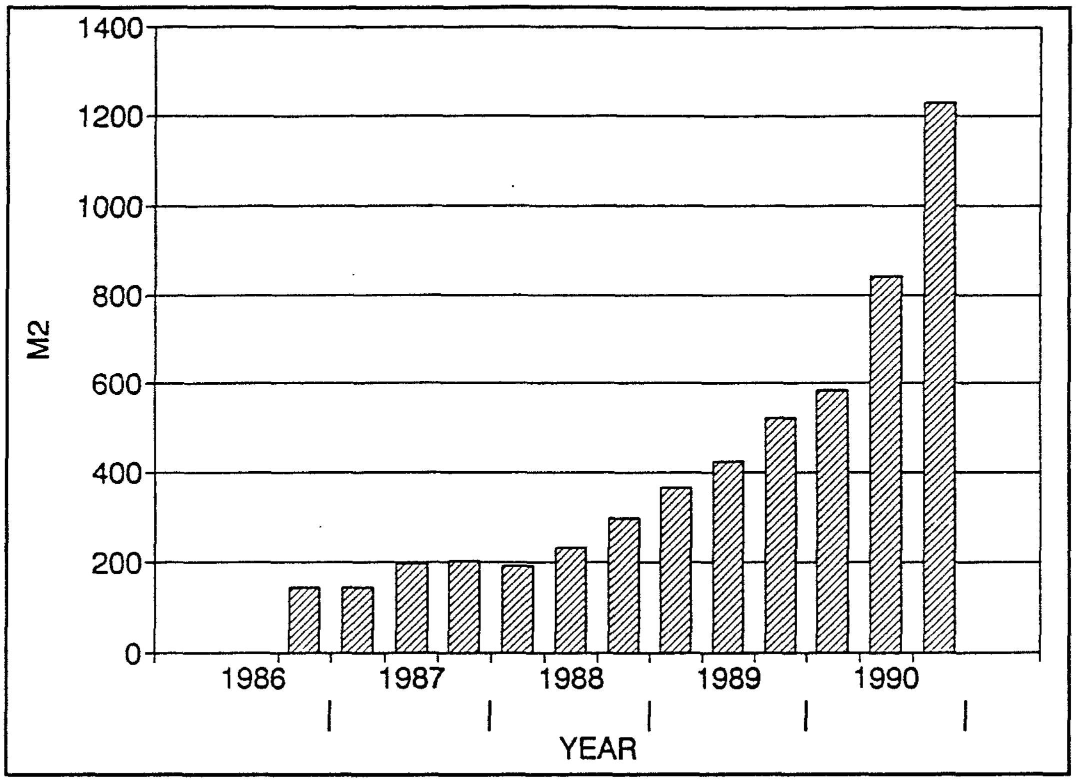

Figure 6. Total areal coverage for the Halodule wrightii study sites in Hillsborough Bay.

Transplants

In October 1990, three years after the initial planting of *Halodule wrightii* sod blocks and bare root units, transplants persisted in five of eight test areas. The transplanted material has expanded from an initial coverage of 13 m^2 in 1987 to over 800 m^2 in 1990 (Figure 7). Of the 350 sod blocks initially planted, about 40 have persisted. Areal coverage of the sod block planting in 1990 was about 525 m^2. An estimated 35% of the 861 bare root units survived nearly one year after planting. However, subsequent coalition of the bare root units made additional survival estimates impossible. Areal coverage of the bare root planting in 1990 was about 290 m^2.

In southeastern Hillsborough Bay, one *H. wrightii* sod block transect was planted at a depth 75 cm deeper than a similar transect 250 m closer to shore. Generally, *H. wrightii* short shoots in shallower areas reach maximum density by the summer and begin to decrease in the fall. However, short shoot densities in the deeper transect do not follow a seasonal pattern, with the exception of winter senescence. Also, areal expansion in the deeper transect is erratic and does not follow the trend of regular seasonal growth seen in the shallow transplants. The lack of seasonality in short shoot density and areal expansion may indicate that water quality is not yet sufficient to allow recolonization to occur in deeper areas which historically had seagrass coverage.

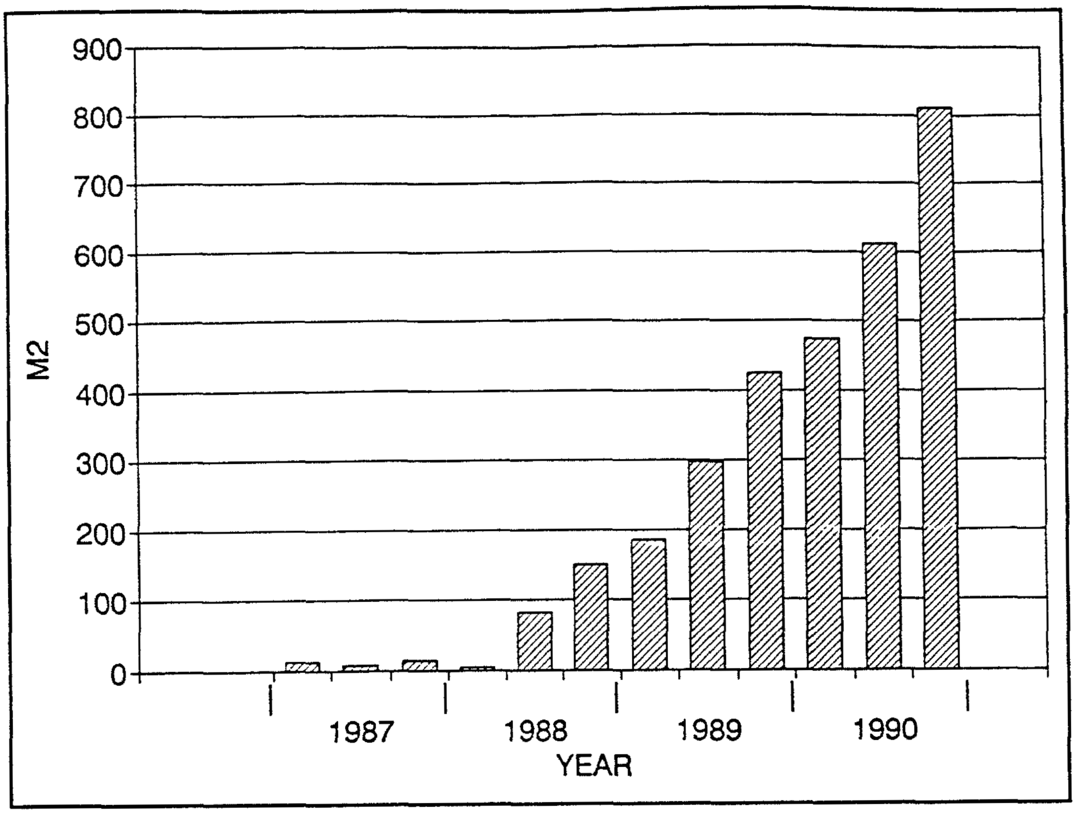

Figure 7. Areal coverage of transplanted Halodule wrightii in Hillsborough Bay.

Results from the transplant effort have been encouraging. Many sites suitable for seagrass transplanting have been identified and vegetative material has been provided to areas where little or no natural *H. wrightii* existed in 1987. Both methods have produced seagrass patches which have similar short shoot densities (5000/m^2) and maximum average blade length (30-35 cm) as the donor site. Meyer et al. (1990) concluded that *H. wrightii* transplanted into Tampa Bay in 1987 attained many functional characteristics of the natural reference beds within three years.

Recent Impacts on Seagrasses

Light availability for seagrass may be attenuated through shading by epiphytic or unattached algae, or by decreased water clarity (Cambridge et al. 1986). Thick mats of macroalgae, mostly *Ulva lactuca*, apparently killed naturally occurring and transplanted *H. wrightii* in two areas of Hillsborough Bay in late 1987 and early 1988. Also, an increase in phytoplankton biomass can reduce water clarity and apparently is the limiting factor allowing recolonization to occur in deeper areas which historically had seagrass. The appearance of macroalgae mats and occasional phytoplankton blooms may reflect the nutrient loading still occurring in Hillsborough Bay (Johansson and Lewis, in press).

Resuspension of sediments into the water column may exacerbate light attenuation in the water column. Tides, wind, ship channel maintainence dredging, ship traffic, commercial seafood harvest, and stormwater runoff promote resuspension and reduce light needed for seagrass persistence and growth.

Waves generated by maritime traffic may affect seagrass distribution. Two transplant areas may be considered high energy zones and about 20% of the sod blocks planted in these areas were probabaly lost to erosion.

Lack of awareness by commercial and recreational boaters may cause additional losses of seagrass. Propeller cuts in natural and transplanted *Halodule wrightii* beds in Hillsborough Bay have been seen. Boaters may be unaware of where seagrasses occur or they may not consider the impacts boats may have on seagrass beds.

Future Considerations

As Hillsborough Bay water quality continues to improve, *Halodule wrightii* should continue to colonize most shallow areas which historically had seagrass meadows. Therefore, large-scale *H. wrightii* transplant efforts in Hillsborough Bay may not be necessary. However, transplanting of *Syringodium filiforme* and *Thalassia testudinum*, the species next in colonizing succession, should be attempted in selected areas of Hillsborough Bay. Limited transplantings of these species would determine their ability to grow in Hillsborough Bay, and introduced material may provide a vegetative source for recolonization.

Activities which impede seagrass recolonization must be evaluated and arbitrated in favor of a healthier estuary. Continued identification and reduction of nutrient loadings to Tampa Bay is imperative. Also, activities which promote resuspension of sediments into the water column need to be controlled and minimized. Finally, commercial and recreational boaters must be educated on the importance of seagrass meadows, the effect that boating activities may have on seagrass, and efforts to revitalize and protect seagrass as a resource.

CONCLUSION

After several decades of declining seagrass coverage in Hillsborough Bay, limited *Halodule wrightii* recolonization is occurring concurrently with improving water quality. Monitoring changes in the seagrass community may be a useful tool to determine the health of Tampa Bay.

A standardized seagrass monitoring program should be established immediately in all sections of Tampa Bay. Annual photographic surveys from aircraft would provide documentation of the extent of Tampa Bay seagrass meadows. Further, transects should be established throughout Tampa Bay to follow the response of seagrass to changes in water quality.

Transplant efforts have been successful. Areas of Hillsborough Bay which are now able to support seagrass have been identified and a source of vegetative material has been established in several locations to enhance recolonization. Also, transplanted *H. wrightii* in Hillsborough Bay apparently provides a habitat similar to natural reference beds.

Large-scale *Halodule wrightii* transplant efforts may not be necessary due to the ability of the seagrass to rapidly cover a suitable site. Vegetative material established by small scale plantings in areas lacking seagrass may provide sufficient source material to promote recolonization. However, small-scale test plantings of *Syringodium filiforme* and *Thalassia testudinum* into selected areas of Hillsborough Bay should be considered in the future.

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RECENT AREAL EXPANSION OF SEAGRASS MEADOWS IN TAMPA BAY, FLORIDA: REAL BAY IMPROVEMENT OR DROUGHT-INDUCED?

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K. D. Haddad
J. O. R. Johansson

INTRODUCTION

Lewis et al. (1985) summarized the information available on seagrass meadows in Tampa Bay at the first Bay Area Scientific Information Symposium in 1982. They reported that their best estimate of seagrass meadow coverage at that time (c. 1981) for the estuary was 5,750 ha (14,203 acres). Comparing this coverage with aerial photography dating back to 1938 and maps back to 1848, they estimated historical seagrass cover to have been 30,970 ha (76,496 acres). These data are compared with recent efforts by other researchers to determine similar trends in seagrass cover. The c. 1981 mapping data have been re-examined for this analysis and updated through 1988 to determine if the seagrass recovery process observed and reported by Johansson and Lewis (in press) has occurred baywide.

METHODS

Using the Florida Department of Natural Resources Marine Resources Geographic Information System (MRGIS), 1988 true color vertical aerial photography at 1:24,000 scale (Southwest Florida Water Management District [SWFWMD], Surface Water Improvement and Management [SWIM] Program) was computer digitized and interpreted. This effort was similar to previous efforts for c. 1950 and 1982 photography (Haddad 1989). The digitized data are presented as baywide maps (Figures 1-3).

RESULTS

Table 1 summarizes the seagrass areal cover for the three time periods (1950, 1982, 1988) by portion of the bay. All areas except Old Tampa Bay showed increases between 1982 and 1988. This may be an interpreting error within the SWFWMD - SWIM seagrass maps. This is being reinterpreted to check the potential error.

We feel comfortable, however, in stating that there has been a minimum of 919 ha (2,271 acres) of new seagrass meadows added to the Tampa Bay System during the 1982-1988 period. Our preliminary review of the 1990 SWIM photography indicates that this increase may be continuing. The Middle Tampa Bay area offshore of Wolf Branch that has been monitored by Johansson and Lewis (in press) is shown in Figure 4. An additional increase beyond that apparent in the 1988 photography is evident.

Table 1. Seagrass coverage in Tampa Bay: 1950, 1982, 1988 (hectares).

	1950	1982	1988
Old Tampa Bay	4,393	2,405	2,119
Hillsborough Bay	1,110	0	25
Middle Tampa Bay	3,844	1,636	2,287
Lower Tampa Bay	2,471	2,030	2,272
Terra Ceia Bay	297	304	399
Manatee River	51	53	99
Boca Ciega Bay	4,282	2,335	2,482
TOTAL	16,448	8,764	9,683

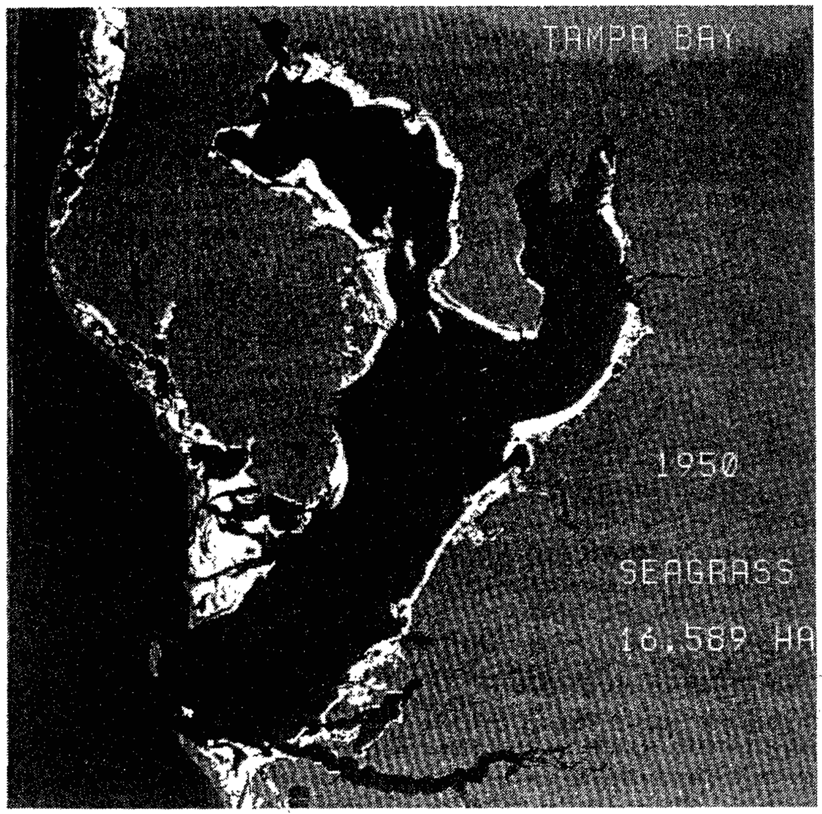

Figure 1. Seagrass coverage in Tampa Bay, 1950.

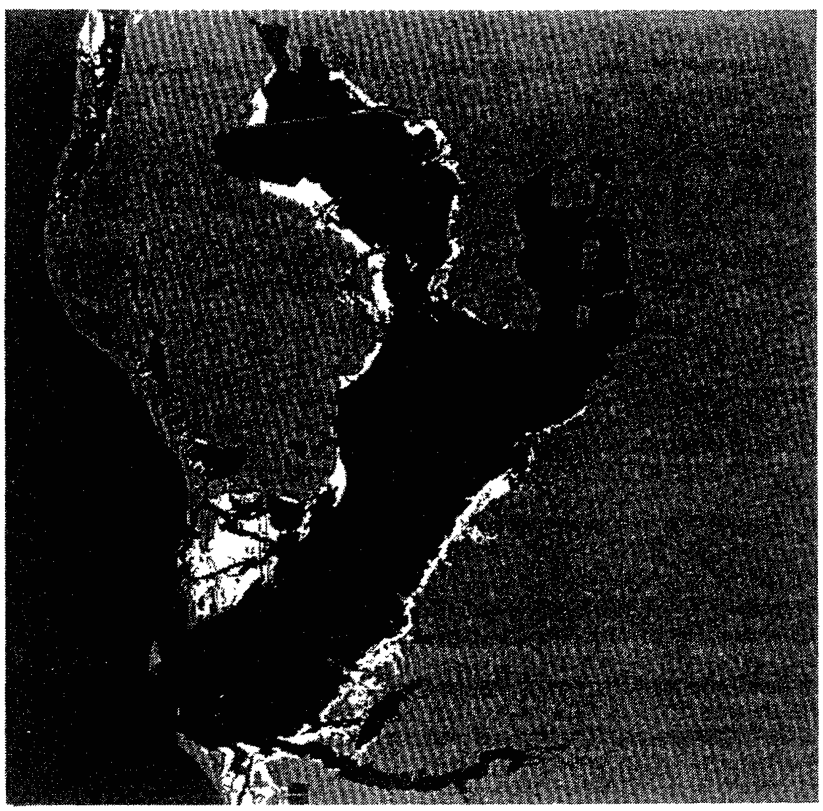

Figure 2. Seagrass coverage in Tampa Bay, 1982.

Figure 3. Seagrass coverage in Tampa Bay, 1988.

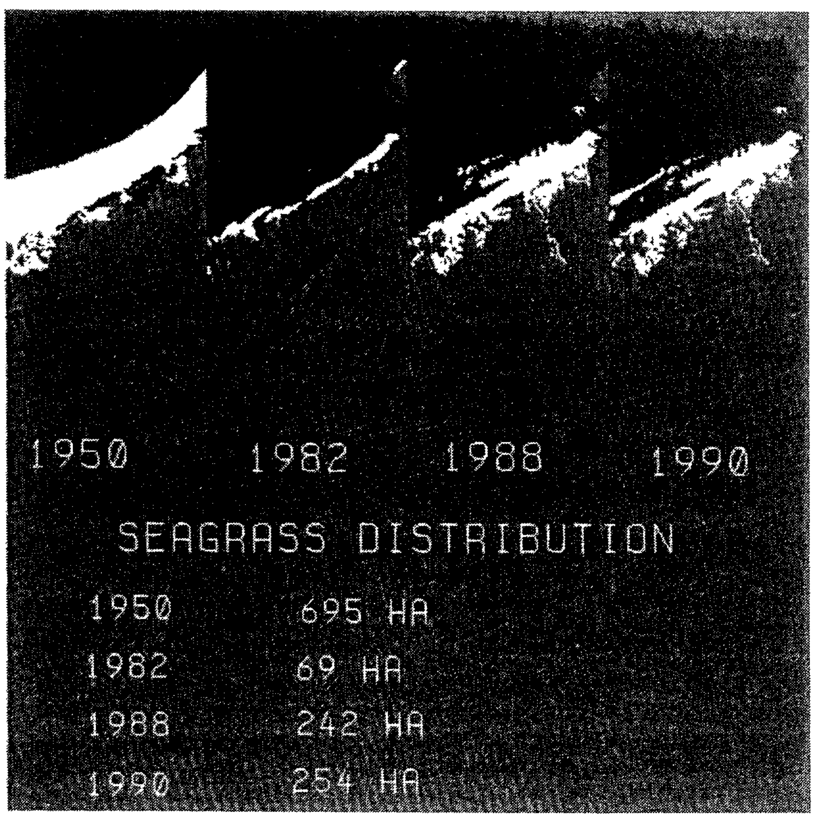

Figure 4. Seagrass recovery in Middle Tampa Bay, offshore of Wolf Branch.

DISCUSSION

Johansson and Lewis (in press) have documented improvements in water quality apparently associated with concurrent improvements to sewage treatment, particularly at the City of Tampa's Hookers Point Sewage Treatment Plant. These improvements are at least partially responsible for the significant decreases in chlorophyll *a* and increases in water clarity (Johansson, this volume) documented in Hillsborough Bay and Middle Tampa Bay.

At the same time however, a multi-year drought has significantly reduced watershed drainage into the bay. Kenworthy et al. (1988) report that tannin-stained "brownish-yellow colored water" produced as a result of a hurricane close to Hobe Sound significantly attenuated photosynthetically active radiation reaching seagrass in the sound. This effect lasted for several months. This possible impact, as well as additional nutrient inputs from a saturated watershed, have the potential to reverse the water quality improvements to date.

We recommend, at a minimum, continued detailed seagrass mapping every two years, preferably every year, in order to accurately document seagrass areal cover trends during normal rainfall years.

A comparison of our results with previous estimates of seagrass coverage in Tampa Bay is presented in Table 2.

Table 2. Tampa Bay seagrass trends (hectares).

YEAR	HECTARES	YEAR	HECTARES	PERCENT CHANGE	NO. OF YEARS	REFERENCE
1876	30,970	1982	5,750	-81	106	Lewis et al. 1985
1957	24,031	1982	12,648	-47	25	TBRPC 1986
1950	16,325	1982	9,120	-44	32	Kunneke and Palik 1984
1940	11,405	1983	5,566	-51	43	Palmer and McClelland 1988
1950	16,448	1982	8,764	-47	32	this paper

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COMPARISON OF BENTHIC MACROINFAUNA AMONG NATURAL AND PLANTED *SPARTINA ALTERNIFLORA* (GRAMINEAE) AND *HALODULE WRIGHTII* (POTAMOGETONACEA) FROM TAMPA BAY

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INTRODUCTION

Planting of intertidal (saltmarsh) and subtidal (seagrass) vegetation has become popular in recent years largely due to the mitigation value for offsetting developmental impacts. Such plantings have been implemented with the intention of creating or restoring marine and estuarine habitat that are of considerable importance or "value" to the shallow water ecosystems. Implicit in this approach is the concept that creation of such habitat results in increased habitat diversity and enhanced production of marine resources, such as commercial and recreational fisheries, as well as the noncommercial fauna such as invertebrates and wading birds.

The long-term success of "created" habitat is very poorly documented, and recent investigations of mitigated habitat by the Florida Department of Environmental Regulation have indicated that the success of such projects is dismally low. Scientific data quantifying the habitat value of restoration sites are severely lacking.

A marine habitat research and restoration project was initiated by Mote Marine Laboratory under contract to the Florida Department of Natural Resources in July of 1986. The project was designed to obtain measurable physical, chemical and biological data of known importance to proper ecological functioning, and compare these parameters between planted and natural habitats. The objectives of the study were to determine experimentally if planted saltmarsh and seagrass communities assume the same ecological functions as their natural counterparts. Project components included water quality, sediment chemistry, and plant, macrofaunal, meiofaunal, and fisheries monitoring. This paper presents the results of the macrofaunal monitoring.

Benthic epifaunal and infaunal invertebrates are important components of the aquatic food chain both as adults and as larval plankton. They are abundant in fisheries nursery areas typically associated with aquatic vegetation such as seagrasses and marshes. The decline in estuarine aquatic vegetation and marsh areas has been indicated as a factor in the decline in abundance of west Florida fisheries stocks. Benthic infauna were therefore chosen as a monitoring component to serve as one of the indicators of successful establishment of marsh and seagrass plantings.

Shoal grass

STUDY SITES

Three sites were selected for habitat restoration and subsequent monitoring, two sites in Pinellas County and one site in Manatee County. In Pinellas County, an unvegetated portion of Lassing Park (St. Petersburg) was planted with *Halodule wrightii* in June 1987, and *Spartina alterniflora* was planted at Pinellas Point in June-July 1986, and replanted in the spring of 1987. In Manatee County, an area south of Port Manatee, known as the Hendry or Redfish Creek site, was planted with *Spartina alterniflora* in June of 1987.

METHODS AND MATERIALS

Benthic fauna are known to show distinct faunal stratification between upper and lower marsh zones caused by differing periods of immersion (Subrahmanyam et al. 1976, Subrahmanyam and Coultas 1980). Therefore, two locations were sampled at the Pinellas Point planted and control sites: 1) upper intertidal one-half of the planted area; and 2) the lower intertidal one-half of the planted area. The natural control site was also sampled in this manner. Three stations were sampled at the Pinellas Point site: a bare "control" (PBC), a vegetated "control" (PVC), and planted *Spartina* (PS). Two stations were sampled at the Hendry Delta site, one a bare "control" site and a second planted with *Spartina*. Three stations were sampled at Lassing Park: a bare "control", a *Halodule* vegetated "control", and a *Halodule* planted plot. Preplanting sampling events were conducted at each site to establish baseline conditions.

Infaunal samples were collected using a 7.6 cm (3 in) diameter x 15 cm length, PVC core providing a surface area of 45.6 cm^2. Twelve randomly located replicate samples were collected from each area for each sampling event. Each sample was washed through a 0.5 mm mesh box sieve to remove silt and fine sand. Sieved samples were preserved with a 10% formalin solution buffered with seawater and containing rose bengal stain to facilitate sorting. Ten of the twelve replicate infaunal samples collected at each site were subsequently analyzed for benthic fauna. All organisms collected within each sample were identified and enumerated to the lowest practical taxonomic level (genus and species for most organisms).

Estimates of infaunal community parameters were calculated from the enumerated and identified fauna. Data were also analyzed using community analysis techniques that estimate the levels of similarity (or difference) between faunal assemblages. Tabulated and calculated parameters included: faunal density (number of organisms/m^2); species richness (number of taxa for each station); species diversity, Shannon's H' index using log base 10 (Shannon and Weaver 1963); equitability or evenness of distribution, Pielou's index (Pielou 1966); and faunal similarity, utilizing the Bray-Curtis Index (Bray and Curtis 1957, Field and McFarlane 1968). Hierarchical cluster analysis was conducted utilizing the Bray-Curtis Index and group average sorting (BioStat II Software, Pimentel and Smith 1986).

RESULTS

Successional trends for each site are presented below. Due to space limitations, details of species abundances and composition are not presented.

Pinellas Point *Spartina alterniflora* Monitoring

Table 1 lists the community parameters for each station by sampling date. A total of 221 taxa were collected from the three Pinellas Point stations over the course of the monitoring program. The lower vegetated control (PVC-Lower) station exhibited the greatest number of taxa (158) over all sampling events, followed by the upper vegetated control station (PVC-Upper, 135 taxa) and the lower planted station (PS-Lower, 103 taxa). The upper planted station (PS-Upper) and the bare control site had the lowest cumulative numbers of taxa (99 and 96 taxa, respectively).

Species richness ranged from a low of 13 taxa for the lower marsh planted site (PS-Lower) in August 1986 to a high of 55 taxa for the January 1988 sampling of the lower vegetated control station (PVC-Lower). The trends in species numbers for each station for all sampling events are illustrated in Figure 1. With the exception of one sampling event (March 1989), the number of species found within the vegetated control station (both upper and lower) was greater than the number of species found within the planted site. There were two occasions when the bare control station (PBC) exhibited a greater number of taxa than either of the vegetated controls. Conversely, the species numbers exhibited very little difference for comparisons between the planted site and bare control. The bare control station had a greater number of taxa than either the upper or lower planted stations for eight of the thirteen sampling events. Prior to April 1988, the number of taxa collected at the vegetated control sites was distinctly greater than the number collected at the planted site. For the July and November 1988 collections, the number of taxa recovered from the control site dropped to approximately the number found at the planted site. By March 1989, the number of taxa were very similar among the upper planted, lower control and bare control stations, while the lower planted and the upper control were similar to one another, but not to the other stations.

Figure 2 illustrates the differences in species richness among the three plots, for species lumped according to major taxonomic groups. The numbers of arthropod species (principally crustaceans) showed the greatest differences among stations, for August 1986 through April 1988. After April 1988 the plots exhibited a convergence in species numbers, then a slight divergence for the last sampling event. Molluscs showed the least differentiation among plots and lacked consistent patterns. There were large differences in the number of annelid species among plots at the beginning of the study, but species numbers became more similar during later sampling events.

Table 1. Benthic faunal parameters for each Pinellas Point sampling station and date.

SAMPLING DATE		NUMBER TAXA	NUMBER INDIVIDUALS	INDIVIDUALS PER M^2	SHANNON	PIELOU
Bare Control Station						
1986	AUG	15	159	3487	2.09	.77
	SEP	15	403	8838	1.37	.51
	NOV	23	1352	29649	1.37	.44
1987	JAN	25	1183	25943	1.61	.50
	MAR	22	612	13421	1.94	.63
	APR	26	897	19671	2.02	.62
	MAY	31	1223	26820	1.70	.49
	JUL	21	484	10614	1.65	.54
	SEP	23	395	8662	1.79	.57
1988	JAN	34	834	18289	1.99	.56
	APR	31	563	12346	1.72	.50
	JUL	34	796	17456	2.29	.65
	NOV	30	1472	32281	1.75	.51
1989	MAR	21	357	7829	1.46	.48
Planted Lower Marsh Station						
1986	AUG	13	290	6360	1.60	.62
	SEP	18	407	8925	1.52	.53
1987	JAN	18	604	13246	1.76	.61
	MAR	20	471	10329	2.01	.67
	APR	26	496	10877	1.98	.61
	MAY	29	971	21294	1.39	.41
	JUL	21	193	4232	2.30	.76
	SEP	20	262	5746	1.95	.65
1988	JAN	31	507	11118	2.14	.62
	APR	33	182	3991	2.86	.82
	JUL	22	199	4364	1.66	.54
	NOV	27	676	14825	1.91	.58
1988	MAR	34	333	7303	1.94	.55
Planted Upper Marsh Station						
1986	AUG	18	179	3925	1.82	.63
	SEP	15	211	4627	1.71	.63
1987	JAN	17	327	7171	1.86	.66
	MAR	20	364	7982	2.00	.67
	APR	20	413	9057	2.22	.74
	MAY	26	665	14583	1.64	.50
	JUL	22	274	6009	2.42	.78
	SEP	16	160	3509	2.12	.77
1988	JAN	28	491	10768	1.95	.59
	APR	21	176	3860	2.26	.74
	JUL	20	221	4846	1.71	.57
	NOV	29	622	13640	2.21	.66
1989	MAR	23	295	6469	1.04	.33

(continued)

Table 1 continued.

SAMPLING DATE		NUMBER TAXA	NUMBER INDIVIDUALS	INDIVIDUALS PER M^2	SHANNON	PIELOU
Vegetated Control Station - Lower Marsh						
1986	AUG	35	249	5461	2.49	.70
	SEP	26	300	6579	2.25	.69
	NOV	40	672	14737	2.25	.61
1987	JAN	45	1237	27127	2.39	.63
	MAR	41	1244	27281	2.22	.60
	APR	41	1298	28465	2.11	.57
	MAY	50	2312	50702	2.14	.55
	JUL	46	2218	48640	1.95	.51
	SEP	32	1058	23202	2.17	.62
1988	JAN	55	2297	50373	1.75	.44
	APR	50	3695	81031	1.59	.41
	JUL	24	1690	37061	1.06	.33
	NOV	30	1139	24978	1.87	.55
1989	MAR	25	1120	24561	1.02	.32
Vegetated Control Station - Upper Marsh						
1986	AUG	28	531	11645	1.49	.45
	SEP	31	238	5219	2.52	.73
	NOV	28	566	12412	2.03	.61
1987	JAN	38	871	19101	2.18	.60
	MAR	33	1599	35066	1.84	.53
	APR	35	1576	34561	1.97	.55
	MAY	45	1924	42193	2.11	.56
	JUL	39	1000	21930	2.34	.64
	SEP	32	858	18816	1.77	.51
1988	JAN	49	2580	56579	1.60	.41
	APR	40	2732	59912	1.56	.42
	JUL	32	2526	55395	1.63	.47
	NOV	26	1219	26732	1.20	.37
1989	MAR	35	1198	26272	1.28	.36

Faunal density ranged from a low of 3,487 organisms/m^2 at the bare control station (PBC, August 1986) to a high of 81,031 organisms/m^2 at the lower vegetated control station (PVC-Lower, April 1988). Temporal trends in faunal density for each station are illustrated in Figure 1. With only a few exceptions, the vegetated control exhibited the greatest faunal densities. The planted site exhibited the lowest faunal densities for all sampling events with the exception of August 1986, when the bare control had the lowest faunal density. There were no long-term increases in faunal density evident at the planted station during the study period.

Trends in faunal density for the major taxonomic groups (Polychaeta, Mollusca and Arthropoda) are shown in Figure 3. No trends were discernable with regard to site. During spring 1987, mollusc and arthropod densities increased substantially at the vegetative control station, while only slight increases in the densities of these groups were observed at the bare or planted stations during the same period. Similarly, from September 1987 to November 1988, a pronounced increase in the numbers of annelids occurred only within the vegetated control site and not at the bare or planted site.

The vegetated control site exhibited the greatest numbers of taxa. Many taxa listed for this site were "rare" or the result of a single individual occurring during a single sampling episode. There were 72 taxa found within the vegetated control site that were not recovered from either the planted or bare control sites. On the other hand, there were only eight taxa unique to the bare control site, and 24 taxa found only at the planted site. The remaining 104 taxa occurred at all sites but in differing densities.

The oligochaete species complex *Tubificidae* spp., consisting of at least five species, exhibited the greatest numbers of individuals for all Pinellas Point plots. The oligochaete complex, *Enchytraeidae* spp., was the third most abundant taxon, represented by at least two distinct species. At the species level the most abundant taxon was the polychaete *Laeonereis culveri*, followed by *Hargeria rapax* (tanaid) and *Capitella capitata* (polychaete). These five taxa accounted for over 70% of all individuals collected from Pinellas Point.

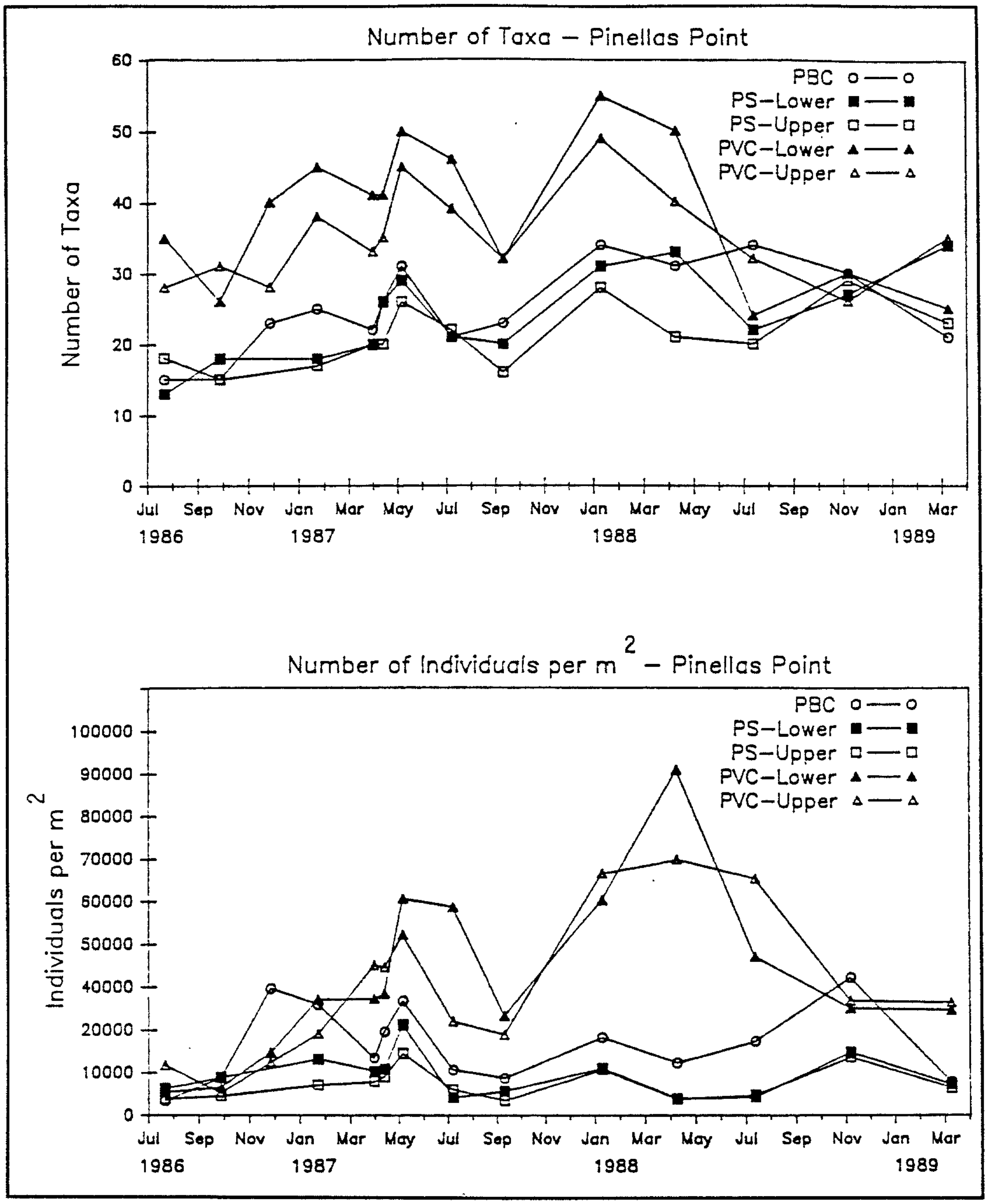

Figure 1. Number of taxa and individuals per m^2 for each Pinellas Point sampling station and date.

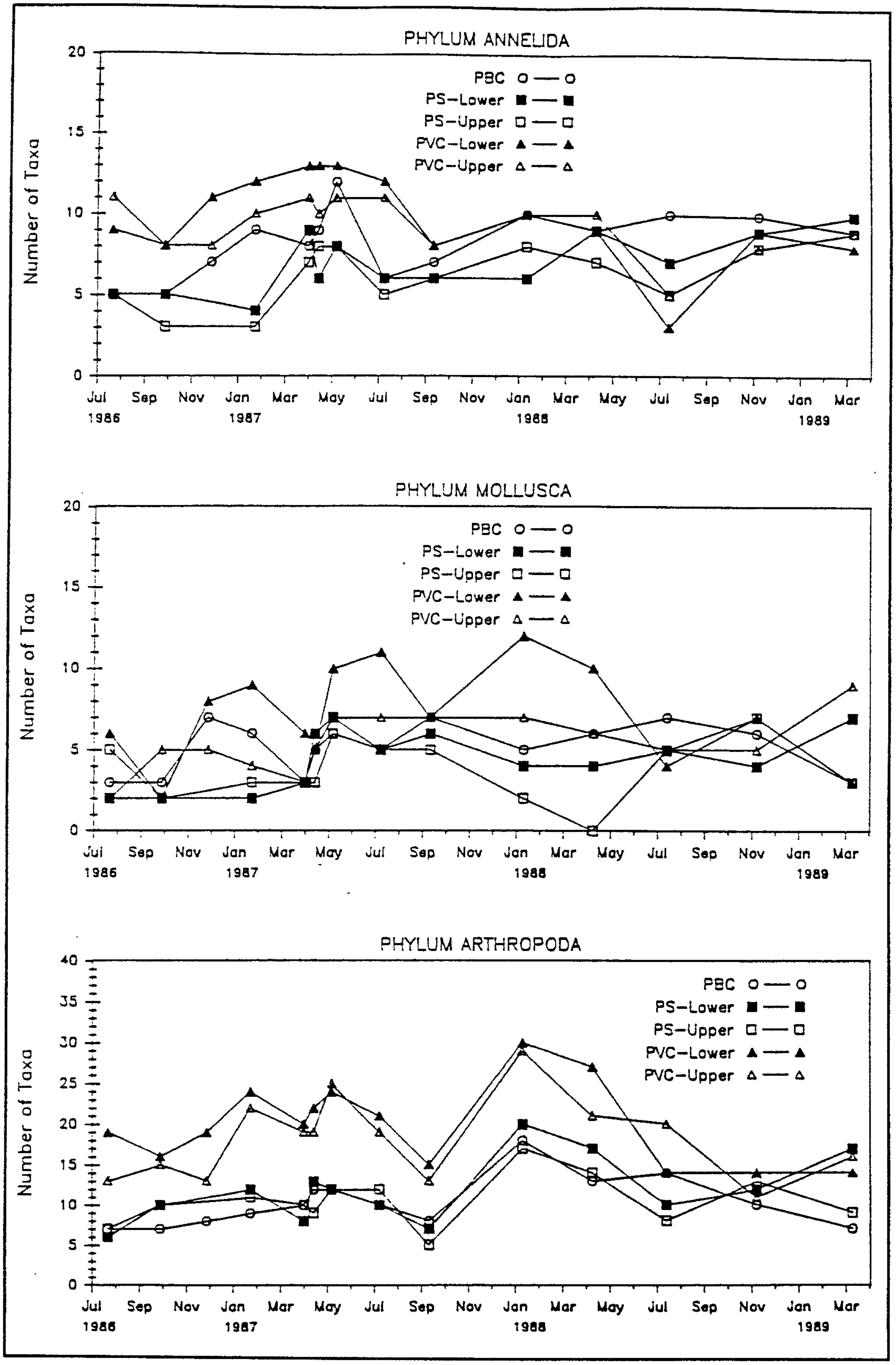

Figure 2. Number of taxa in major faunal groups for each Pinellas Point benthic sampling station and date.

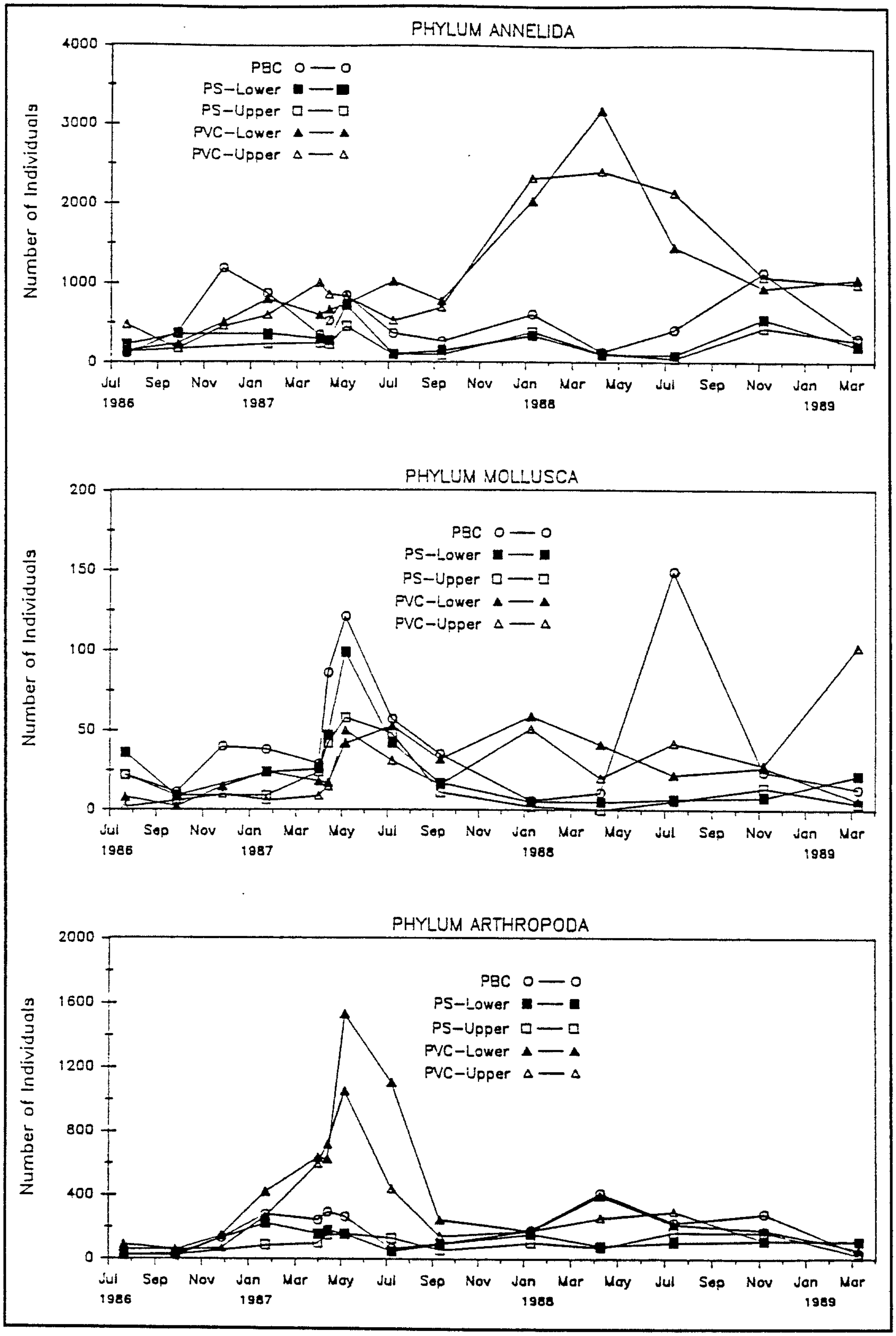

Figure 3. Number of individuals per m^2 in major faunal groups for each Pinellas Point benthic sampling station and date.

Tubificidae spp. were extremely abundant at the vegetated control stations. They were less abundant at the planted site, and even less so at the bare control site. *Enchytraeidae* spp. were found almost exclusively at the vegetated control site. Species that were found predominantly at the vegetated control site included *Hargeria rapax*, *Uca* sp. (decapod, juvenile specimens), *Hemiptera* sp., *Assiminea succinea*, *Gammarus mucronatus*, *Cyathura polita*, *Athenaria* sp., *Balanus* spp., and *Guekensia demissa*. *Laeonereis culveri* was more than twice as abundant at the bare control site than at the other two sites, where its abundance was similar. Additional species which were most abundant at the bare control site were *Almyracuma* sp. A, the bivalves *Parastarte triquetra* and *Tagelus plebius*, and the polychaetes *Aricidea philbinae*, *Eteone heteropoda* and *Leitoscoloplos robustus*. Species exhibiting higher numbers at the bare site were usually common at the planted site. *Capitella capitata* was less abundant at the planted site than at either the bare or vegetated control sites. Other organisms exhibiting a similar pattern of distribution included all species of *Corophium* and Cytherideidae, *Actinaria* sp., *Leitoscoloplos robustus* and *L. foliosus*.

Faunal Similarity

Faunal similarity analysis was performed on data for all station sampling event combinations (2,278 comparisons). All stations showed strong seasonal differences in faunal similarity based on the low to very low similarity indices. The planted site (PSL and PSU) exhibited the strongest similarities to the bare control (PBC) from August 1986 through May 1987, after which time the faunal similarity between these sites decreased. The planted site did not show a marked difference in fauna between upper and lower marsh stations.

The planted site showed very little similarity to the vegetated control (PVL and PVU) for the duration of the study. Of the 728 comparisons between the planted and the vegetated control sites there were only two combinations (0.27%) with a high similarity (PSL and PSU November 1988 vs. PVU November 1986). There were 36 pairs (4.9%) of moderate similarity between the two sites. Thirty of the moderate combinations occurred during September 1986, November 1986 and January 1987. The remaining combinations (690) between the planted and vegetative control sites had low to very low similarity index values. The bare control site illustrated the same pattern of no similarity to the vegetated control site. Overall, the planted site exhibited the greatest level of similarity to itself, between upper and lower marsh comparisons and for temporal comparisons. Of the 325 planted site versus planted site comparisons, 12 were highly similar (3.7%), 45 moderately similar (13.8%), and the remaining 268 comparisons low or very low similarity (82.5%). The bare control and vegetated control also exhibited the highest levels of similarity when compared to themselves for temporal events or for upper and lower marsh comparisons.

Cluster analysis (Figure 4) illustrates the low level of faunal similarity between sampling sites. The highest similarity connections were for within site comparisons over time, or between upper and lower marsh within the same site. The closest between-station connections were for the vegetated control stations (PVC upper and lower), while the planted stations and bare control exhibited affinities only at lower levels of similarity. For 1986 through the first half of 1988, the greatest clustering affinities exhibited by the planted site were with itself (between sampling dates and between upper and lower marsh) and with the bare site. From November 1988 to March 1989, the planted site began to cluster with the vegetated control site.

Hendry/Redfish Creek *Spartina alterniflora* Monitoring

Ten sampling events were conducted at the Hendry/Redfish Creek revegetation site from June 1987 through March 1989. The June 1987 event represents the pre-planting baseline; all other events were post-planting.

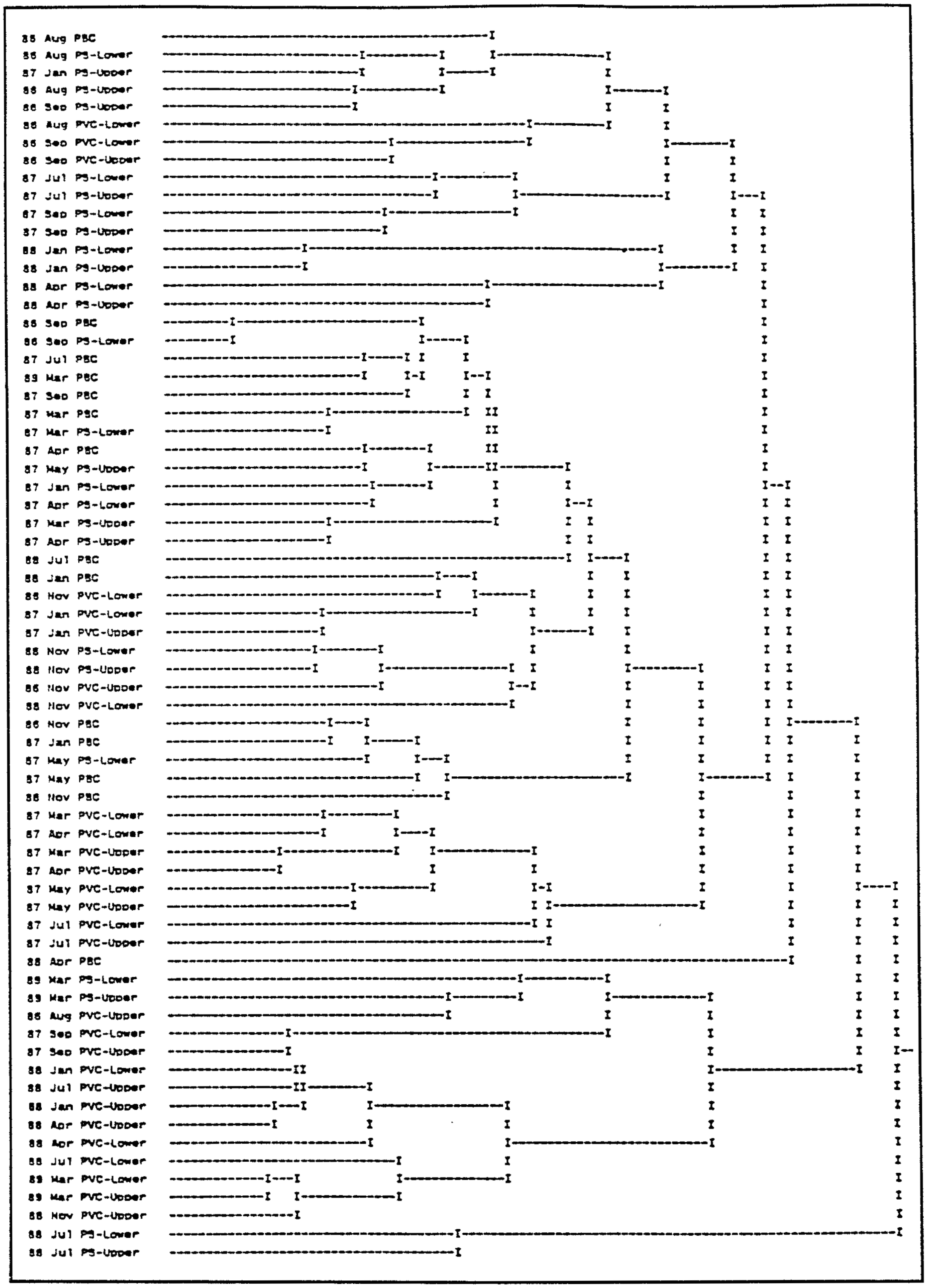

Figure 4. Cluster analysis dendrogram for Pinellas Point benthic sampling stations and dates.

Table 2 lists the community parameters for each station by sampling date. A total of 87 taxa were collected at the Hendry site. Forty-nine taxa were collected from the bare control plot, while 72 taxa were collected from the planted plot. Species counts ranged from a low of only one taxon collected from the bare plot in July 1988 to a high of 31 taxa at the planted plot in August of 1987. For most sampling dates, the planted site exhibited greater numbers of taxa than the bare site, with an average of 16.8 taxa (SD=7.9) and 8.5 taxa (SD=5.1) for the planted and bare sites, respectively. Seasonal variation in species numbers for both plots was pronounced. Species numbers at the bare site ranged from one to 18 taxa, while the planted site had a minimum of nine and a maximum of 31 taxa. Figure 5 illustrates the variation in species numbers through time.

Table 2. Benthic faunal parameters for each Hendry Site sampling station and date.

SAMPLING DATE		NUMBER TAXA	NUMBER INDIVIDUALS	INDIVIDUALS PER M^2	SHANNON	PIELOU
Bare Control Station						
1987	JUN	18	52	1140	2.43	.84
	JUL	9	26	570	1.72	.78
	AUG	13	57	1250	1.73	.68
	OCT	13	31	680	2.33	.91
	DEC	6	8	175	1.67	.93
1988	FEB	8	11	241	1.97	.95
	APR	3	3	66	1.10	1.00
	JUL	1	2	44	.00	N/A
	NOV	8	22	482	1.52	.73
1989	MAR	6	11	241	1.64	.92
Planted Station						
1987	JUN	14	83	1820	1.86	.70
	JUL	19	129	2829	2.31	.79
	AUG	31	532	11666	1.58	.46
	OCT	30	406	8903	2.22	.65
	DEC	18	68	1491	2.23	.77
1988	FEB	9	19	417	1.66	.76
	APR	13	45	987	2.15	.84
	JUL	9	53	1162	1.54	.70
	NOV	12	258	5657	.98	.39
1989	MAR	13	322	7061	.95	.37

Of the 87 taxa found at this site, 34 were common to both bare and planted plots; 38 were found only within the planted plot, and 15 were found only within the bare plot. Three species (the polychaetes *Capitella capitata* and *Laeonereis culveri*, and the fiddler crab, *Uca* sp.) accounted for 70% of the organisms collected at both plots. Taxa common to both plots, such as the dominant polychaetes *Laeonereis culveri* and *Capitella capitata* and the fiddler crab *Uca* sp. (juvenile), were usually much more abundant within the planted plot. Two of the most abundant species found only within the planted plot—*Spirorbis spirillum* (Polychaeta) and *Balanus improvisus* (Crustacea)—are epiphytic and were associated with *Spartina* stalks. Other taxa exclusive to the planted plot (such as the tanaid *Hargeria rapax* and the isopod *Cyathura polita*) were not as abundant but occurred more frequently. All of the taxa unique to the bare site were rare occurrences of one or two individuals.

Taxa are lumped by phyla for bare and planted plots in Figure 6. For both the annelids and the molluscs there was a large drop in species numbers from August 1987 to June 1988. Variation in the number of annelid and mollusc taxa between planted and bare site decreased from 1988 through to 1989. The arthropods exhibited a sharp decline in numbers of taxa from August 1987 to June 1988. The planted site, however, exhibited more stability in the number of arthropod taxa.

Faunal densities ranged from 44 organisms/m^2 at the bare station (July 1988) to 11,666 organisms/m^2 at the planted station (August 1987). The planted station exhibited greater densities than the bare station for all sampling events including the pre-planting baseline collections. Figure 5 illustrates the variation in faunal densities through time for both plots. There was strong seasonal variability in faunal densities at the planted plot, and to a lesser degree at the bare plot. The planted site exhibited much greater variation than did the bare site. By late 1988 (November) and March of 1989 faunal densities had returned to 1987 levels for the planted plot, but remained very low at the bare site.

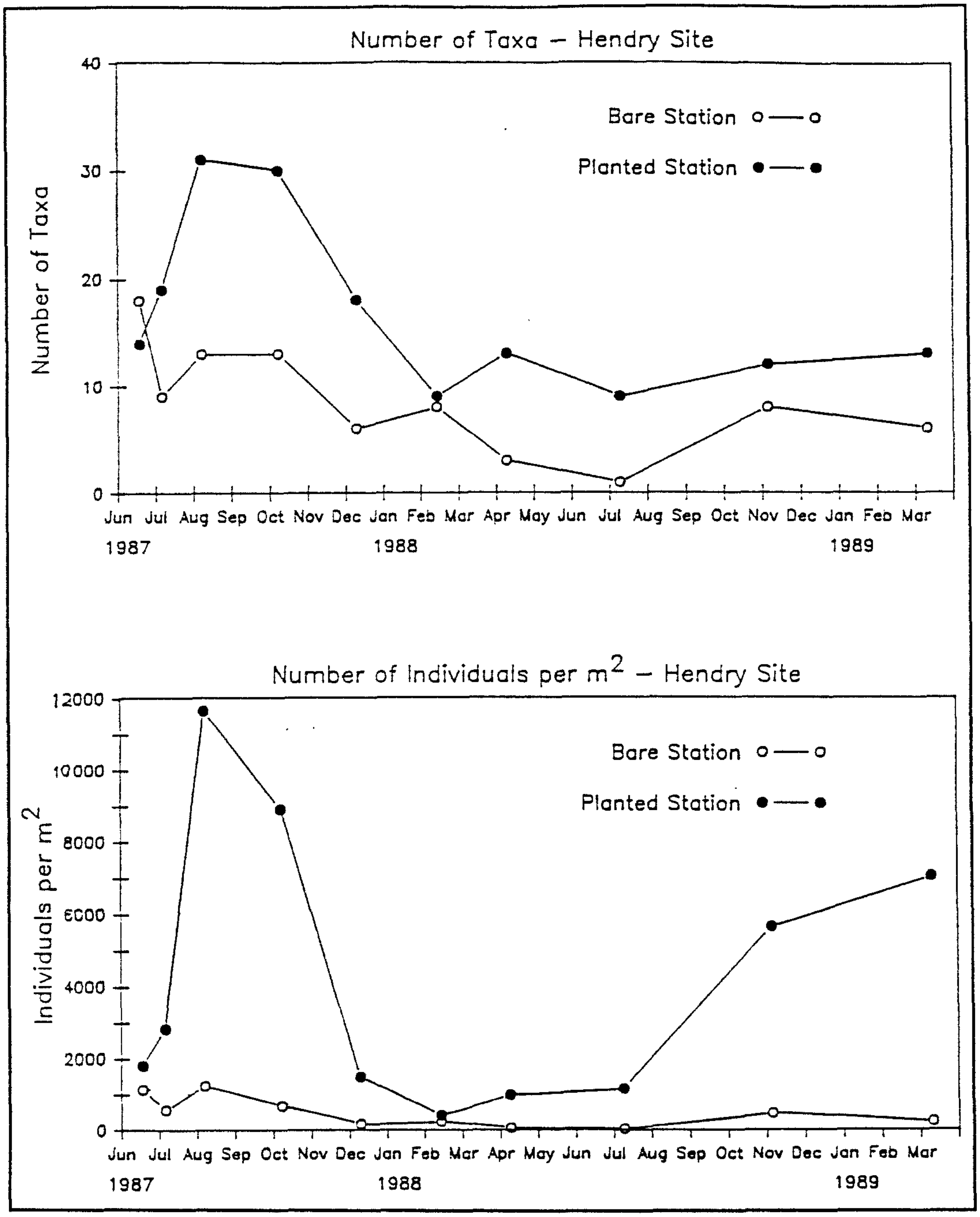

Figure 5. Number of taxa and individuals per m^2 for each Hendry Site benthic sampling station and date.

Figure 7 illustrates trends in faunal densities by phyla. All three phyla exhibited large decrease in numbers between 1987 and 1988. The annelids and crustaceans returned to previous levels by 1989 while the molluscs remained at very low levels. The arthropods (primarily crustaceans) recovered most rapidly from the decline in total numbers.

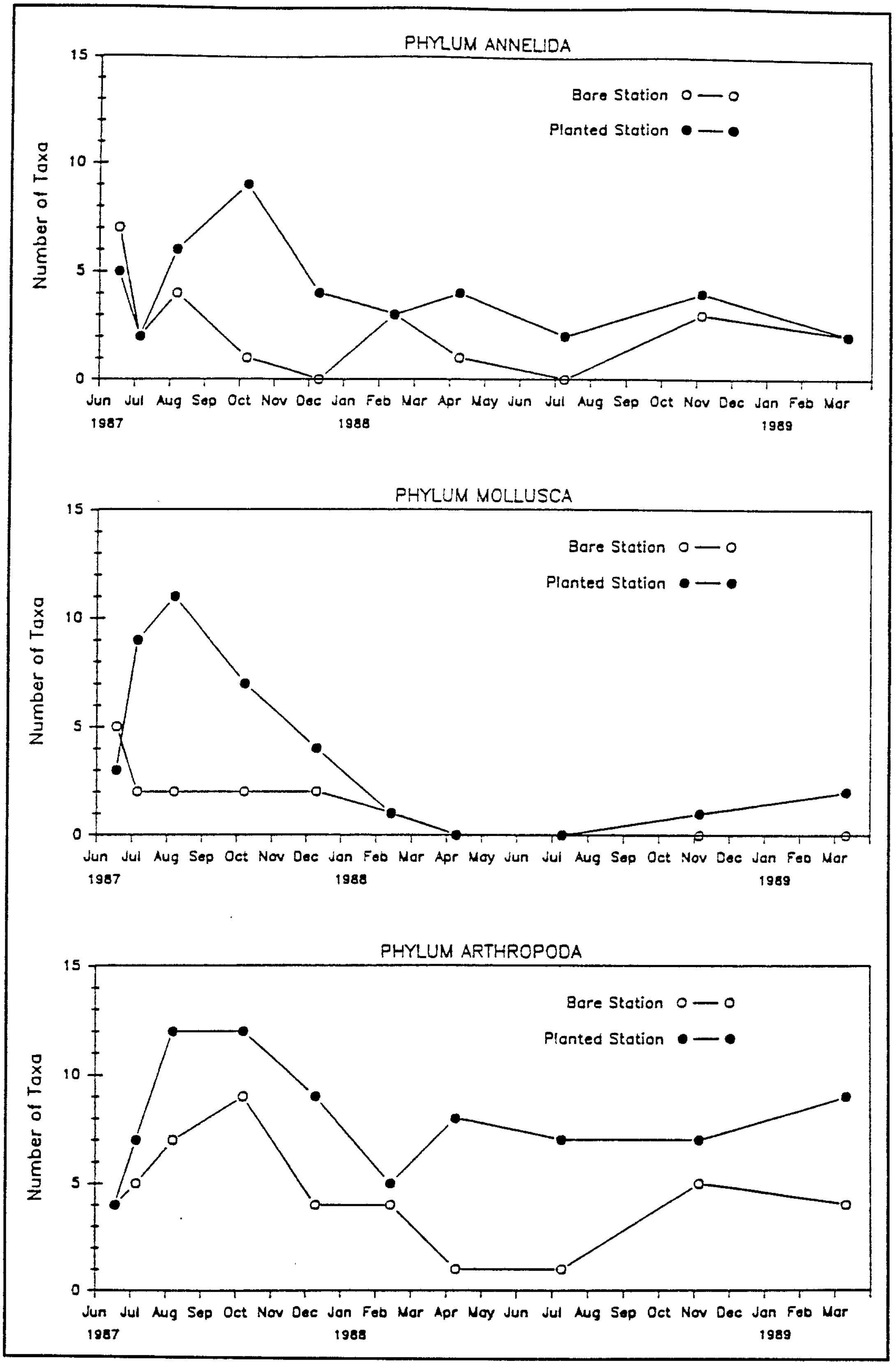

Figure 6. Number of taxa in major faunal groups for each Hendry Site benthic sampling station and date.

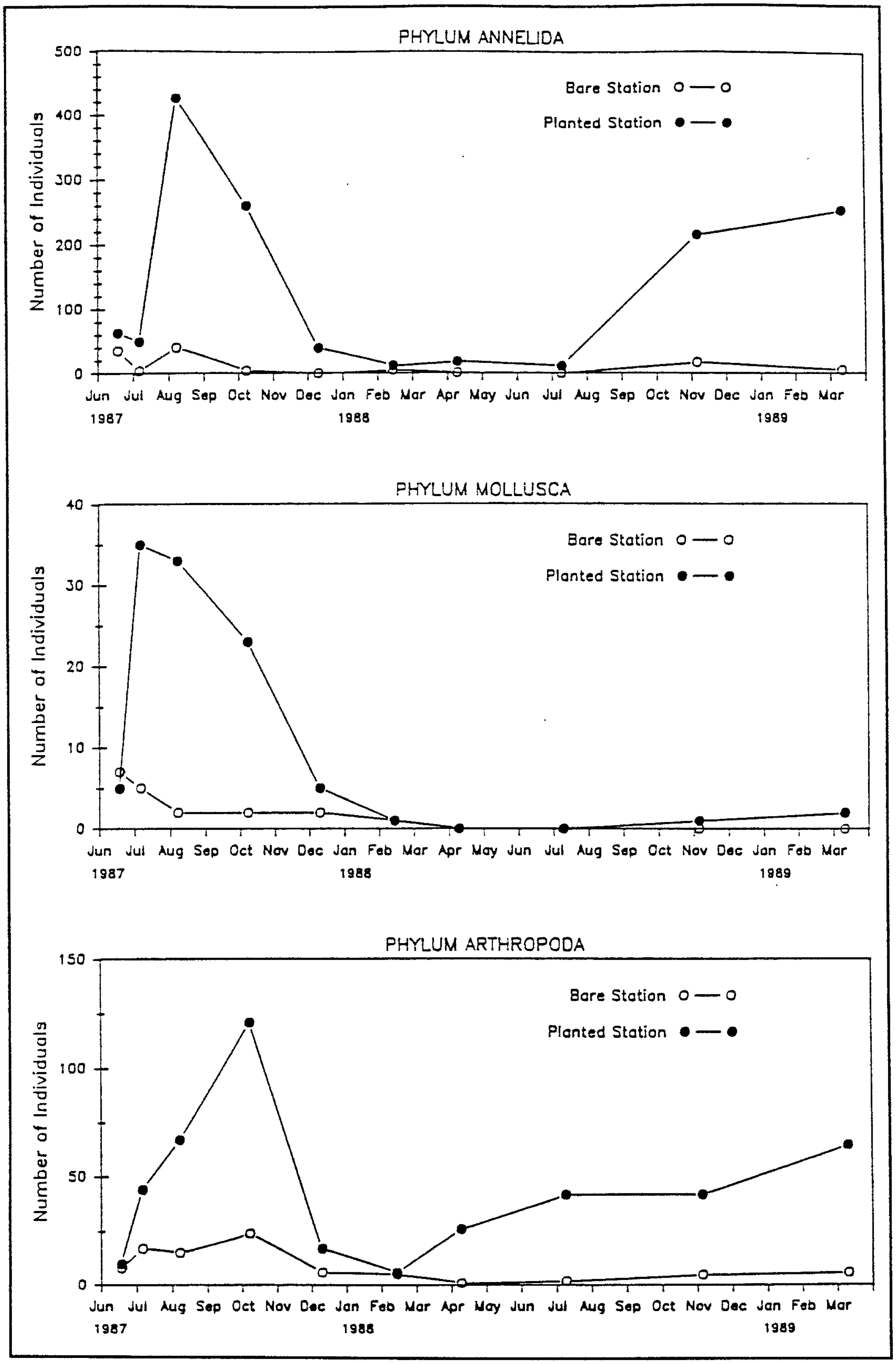

Figure 7. Number of individuals per m^2 in major faunal groups for each Hendry Site benthic sampling station and date.

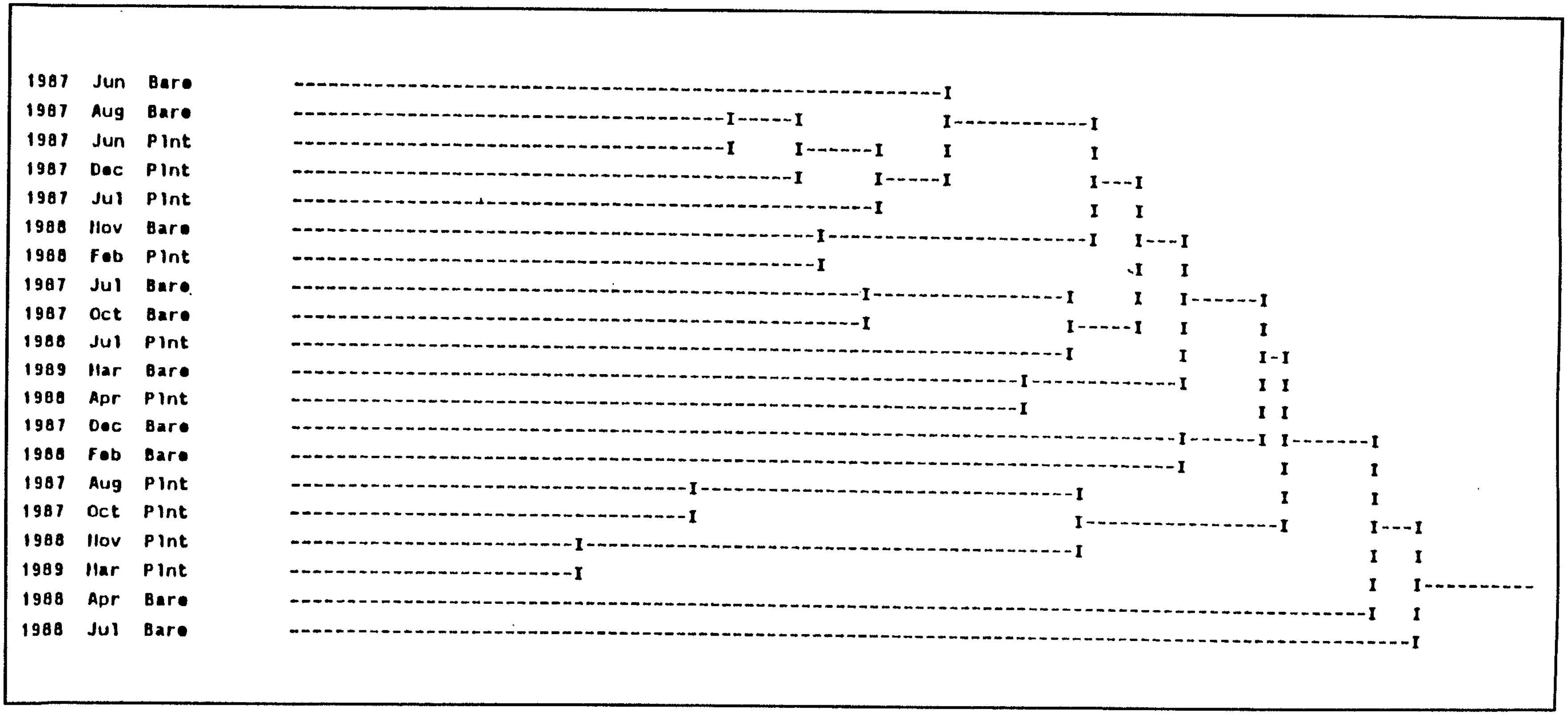

Figure 8. Cluster analysis dendrogram for Hendry Site benthic sampling stations and dates.

Faunal similarity analysis for all station/date combinations revealed that for 190 station pairs there was only one combination of high similarity, five combinations of moderate similarity, 31 low similarity combinations, and 153 very low similarity combinations. The one high similarity value was for the planted station between November 1988 and March 1989. For the moderate similarity combinations three were for bare/planted comparisons, and two for planted/planted comparisons. There were no high or moderate similarities between bare versus bare comparisons. Cluster analysis (Figure 8) illustrates the lack of distinct station (or treatment) separation for the duration of the monitoring.

Shoal grass

Lassing Park *Halodule wrightii* Monitoring

A total of 309 taxa were recovered from Lassing Park. The number of taxa ranged from 46 taxa at the vegetated control station (October 1987) to 106 taxa from the plug planted station (July 1988) (Table 3). The vegetated control station exhibited the greatest number of taxa for eight of the ten sampling events. There was an average of 73 taxa (SD=15.7) at the vegetated control station for all sampling events. The *Halodule* plug planted site exhibited the next overall highest number of taxa with an average of 65.2 taxa (SD=16.7) collected per sampling event. An average of 60.0 taxa (SD=7.8) were collected from the bare control.

A total of 216 taxa was collected from the vegetated control site during the study. Seventy-three taxa (33.8% of vegetated control total) were found exclusively at this site. A total of 196 taxa was collected from the plug station, with 30 unique taxa. The bare station exhibited 173 total taxa, and also had 30 taxa not found at either of the other sites.

The number of infaunal taxa collected from each station throughout the study are illustrated in Figure 9. The number of taxa collected at the planted site closely paralleled the number collected at the bare control.

Figure 10 displays temporal changes in the distribution of species lumped by phyla. All three sites exhibited similar trends in species richness of annelids throughout the study. Molluscs also showed similar trends in species richness for the three sites, although the control station had greater numbers of taxa (except for one sampling event) from February 1988 to March 1989. The arthropods showed the most distinct differences in numbers of taxa between sites. The control site contained consistently greater numbers of arthropod taxa than the bare and planted sites. With one exception (July 1988), the bare and planted sites had similar numbers of arthropods.

Many of the species common to all sites exhibited a preference for either the control or bare/planted sites. No major differences in species composition were observed between the bare and planted station. The gastropod *Diastoma varium* (which feeds on epiphytic growth present on *Halodule*) was much more abundant at the control site than the bare or planted sites. Other taxa that were most abundant at the control station were *Cymadusa compta* (amphipod), and the polychaetes *Aricidea philbinae*, *Prionospio heterobranchia*, *Kinbergonuphis simoni*, *Aricidea taylori*, and *Polydora ligni*. Also common were *Mitrella lunata* (gastropod), *Erichsonella attenuata* (isopod), and *Neopanope texana* (decapod).

Species which occurred more commonly at the bare/plug stations were *Axiothella mucosa* (polychaete), *Ampelisca holmesi* (amphipod), *Acanthohaustorius* sp. A (amphipod), *Mysella planulata* (bivalve), *Paraonis fulgens* (polychaete), *Oxyurostylis smithi* (cumacean), *Brania wellfleetensis* (polychaete), and numerous other taxa at lower densities.

Faunal densities ranged from 10,439 organisms/m^2 at the plug station (December 1987) to 100,833 organisms/m^2 at the control station (June 1987). The control station exhibited the greatest faunal densities for all sampling periods and showed an average density of 52,219 organisms/m^2 (SD=25,991). The bare station had the lowest densities for seven of ten sampling events and an overall average density of 20,289 organisms/m^2 (SD=10,108). Faunal densities at the plug station were similar to those of the bare station, with an average of 23,136 organisms/m^2 (SD=10,212). Pronounced seasonal variations in numbers of organisms were evident. The lowest densities occurred in October and December of 1987 and February of 1988 (Figure 9). The bare control and plug planted site showed similar patterns in faunal density. The vegetated control site showed different trends in faunal density from the bare and plug sites for all sampling except November 1988.

Table 3. Benthic faunal parameters for each Lassing Park sampling station and date.

SAMPLING DATE		NUMBER TAXA	NUMBER INDIVIDUALS	INDIVIDUALS PER M^2	SHANNON	PIELOU
Bare Control Station						
1987	JUN	71	743	16294	3.20	.75
	JUL	66	1047	22961	3.02	.72
	AUG	65	1002	21974	2.69	.65
	OCT	59	578	12675	3.24	.80
	DEC	51	479	10504	3.28	.83
1988	FEB	48	506	11096	3.12	.81
	APR	70	1087	23838	2.96	.70
	JUL	55	828	18158	2.87	.72
	NOV	58	2077	45548	2.08	.51
1989	MAR	57	905	19846	2.67	.66
Vegetated Control Station						
1987	JUN	94	4598	100833	2.81	.62
	JUL	77	3765	82566	1.48	.34
	AUG	65	2232	48947	1.15	.28
	OCT	46	1695	37171	1.13	.30
	DEC	63	1156	25351	2.46	.59
1988	FEB	71	829	18180	2.80	.66
	APR	83	2370	51974	2.63	.59
	JUL	71	2552	55965	2.45	.58
	NOV	67	1480	32456	2.71	.65
1989	MAR	100	3135	68750	3.04	.66
Planted (plug) Station						
1987	JUN	71	895	19627	3.29	.77
	JUL	72	1112	24386	2.98	.70
	AUG	61	882	19342	3.01	.73
	OCT	63	783	17171	3.34	.81
	DEC	53	476	10439	3.36	.85
1988	FEB	49	583	12785	3.11	.80
	APR	71	1428	31316	3.02	.71
	JUL	106	1835	40241	3.48	.75
	NOV	54	1733	38004	1.94	.49
1989	MAR	52	823	18048	2.57	.65

Figure 11 illustrates the temporal trends in faunal densities of organisms arranged by phyla. The numbers of polychaetes were greatest at the control station for seven of ten events (70%). The plug station had greater numbers of polychaetes than the bare site for seven of ten events. The number of molluscs were similar for all sampling events between the bare and plug sites. Molluscs were very abundant at the control site from July 1987 to February 1988, at which time the number dropped sharply to levels similar to the bare station. The number of arthropods was similar between sites, except for June 1987, November 1988 and March 1989.

Of the 435 station/date comparisons of the faunal similarity analysis, there were five high similarity pairs (1.1%), 20 moderate similarity pairs (4.6%), 124 low similarity pairs (28.5%), and 286 pairs (65.7%) of very low similarity. There were no high or moderate similarities (and only one low similarity) between comparisons of the vegetated control site with either the bare or plug sites. In contrast, comparisons between the bare and plug sites exhibited numerous high and moderate faunal similarities, including comparisons between sampling dates.

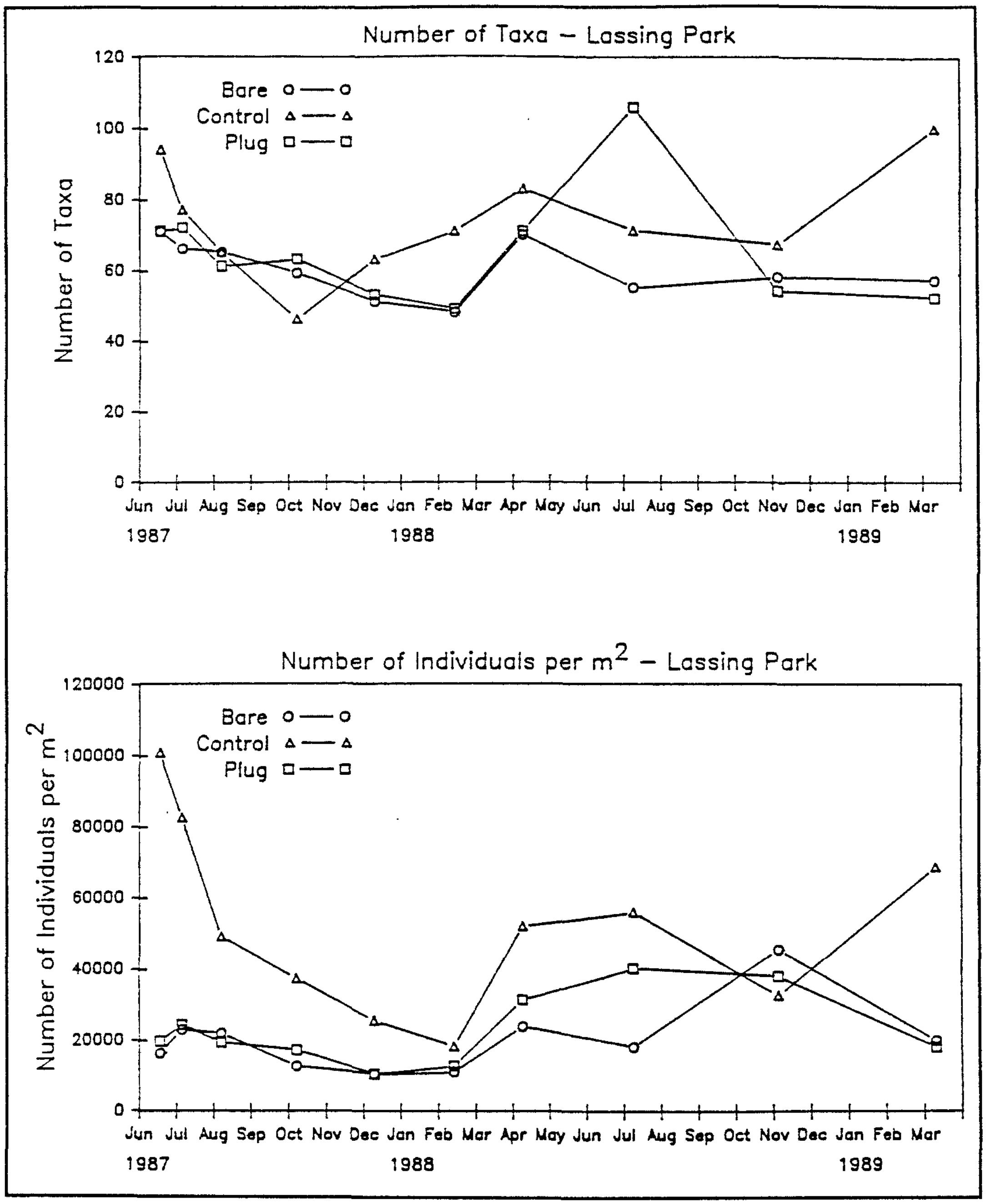

Figure 9. Number of taxa and individuals per m^2 for each Lassing Park benthic sampling station and date.

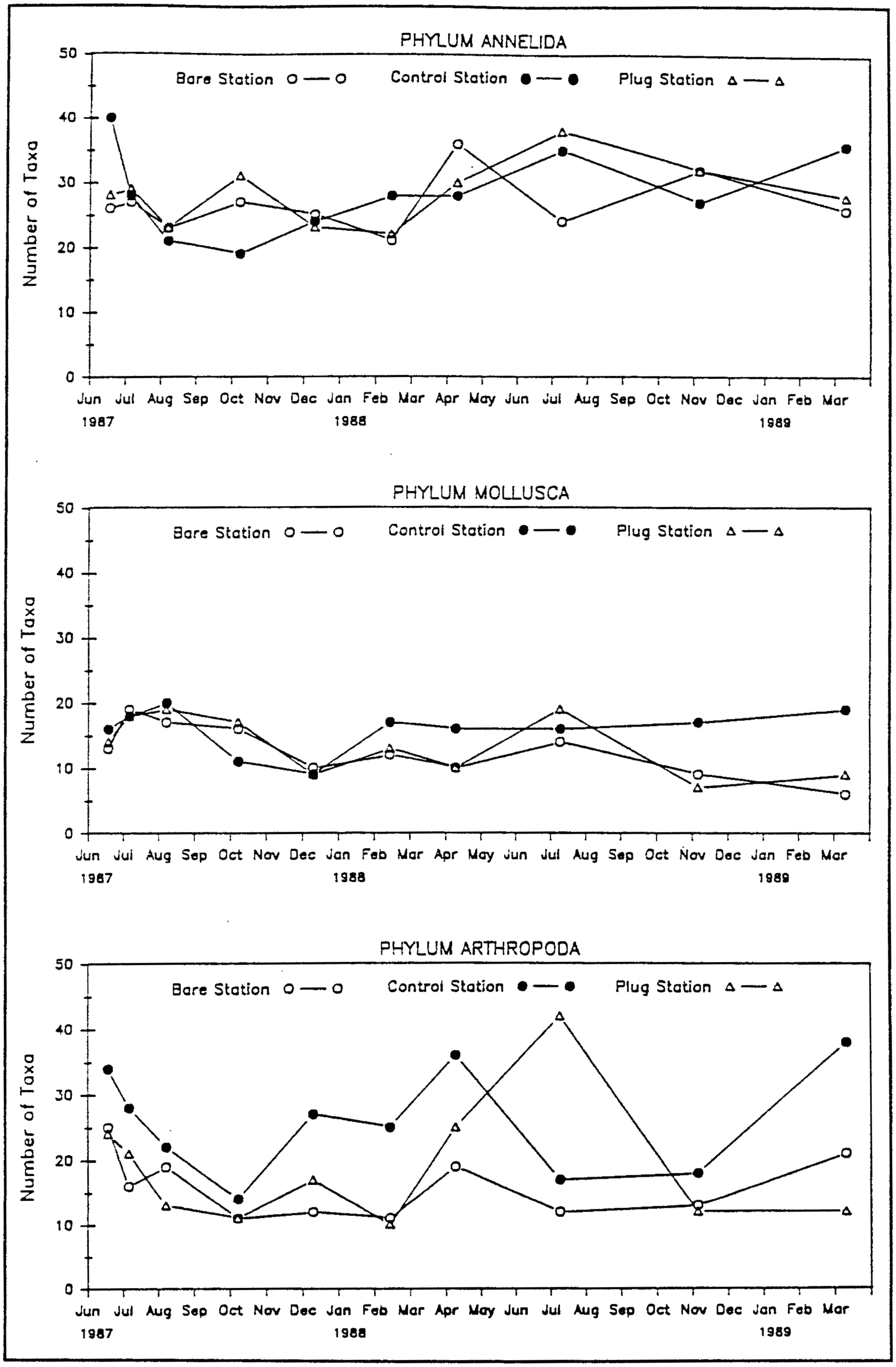

Figure 10. Number of taxa in major faunal groups for each Lassing Park benthic sampling station and date.

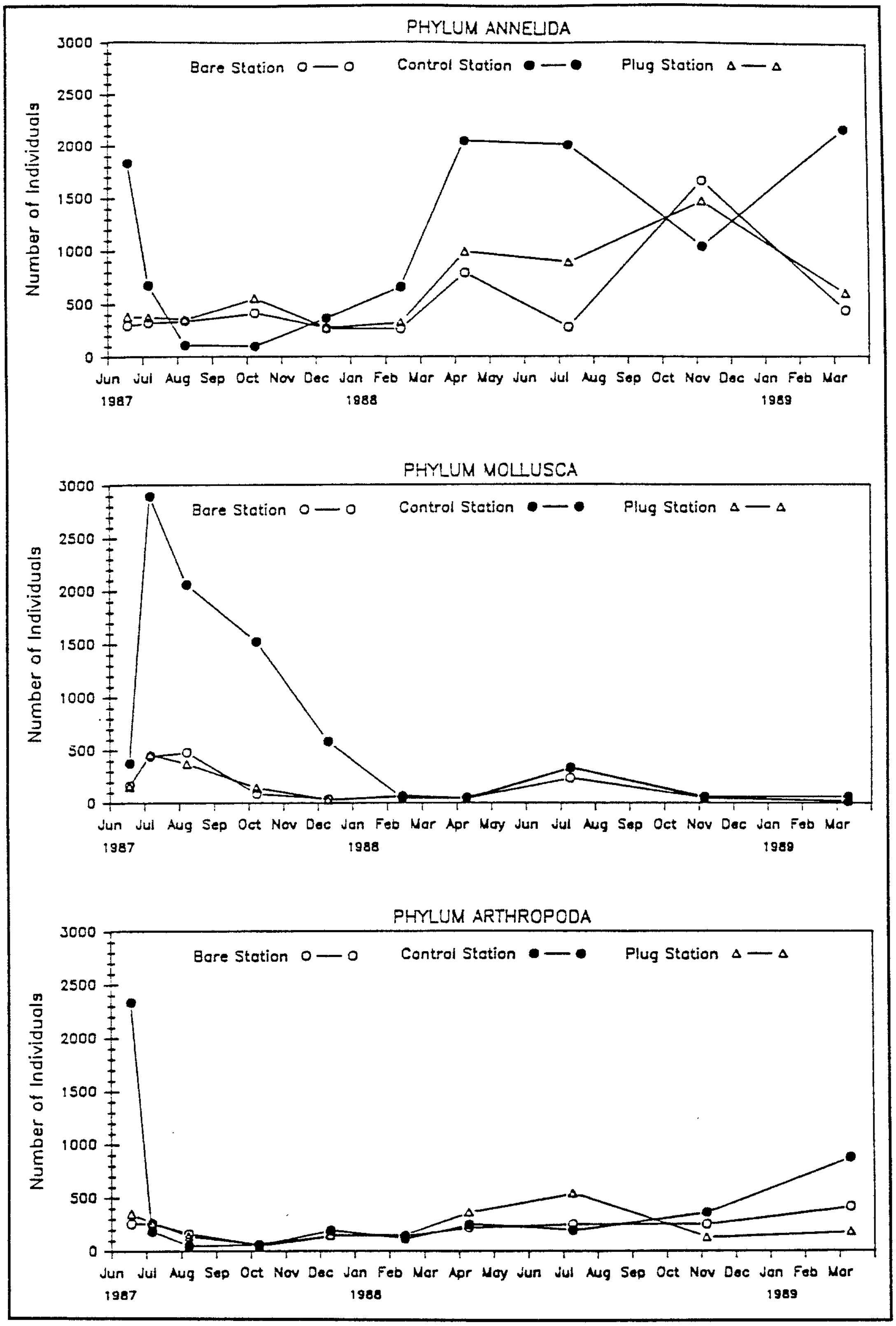

Figure 11. Number of individuals per m^2 in major faunal groups for each Lassing Park benthic sampling station and date.

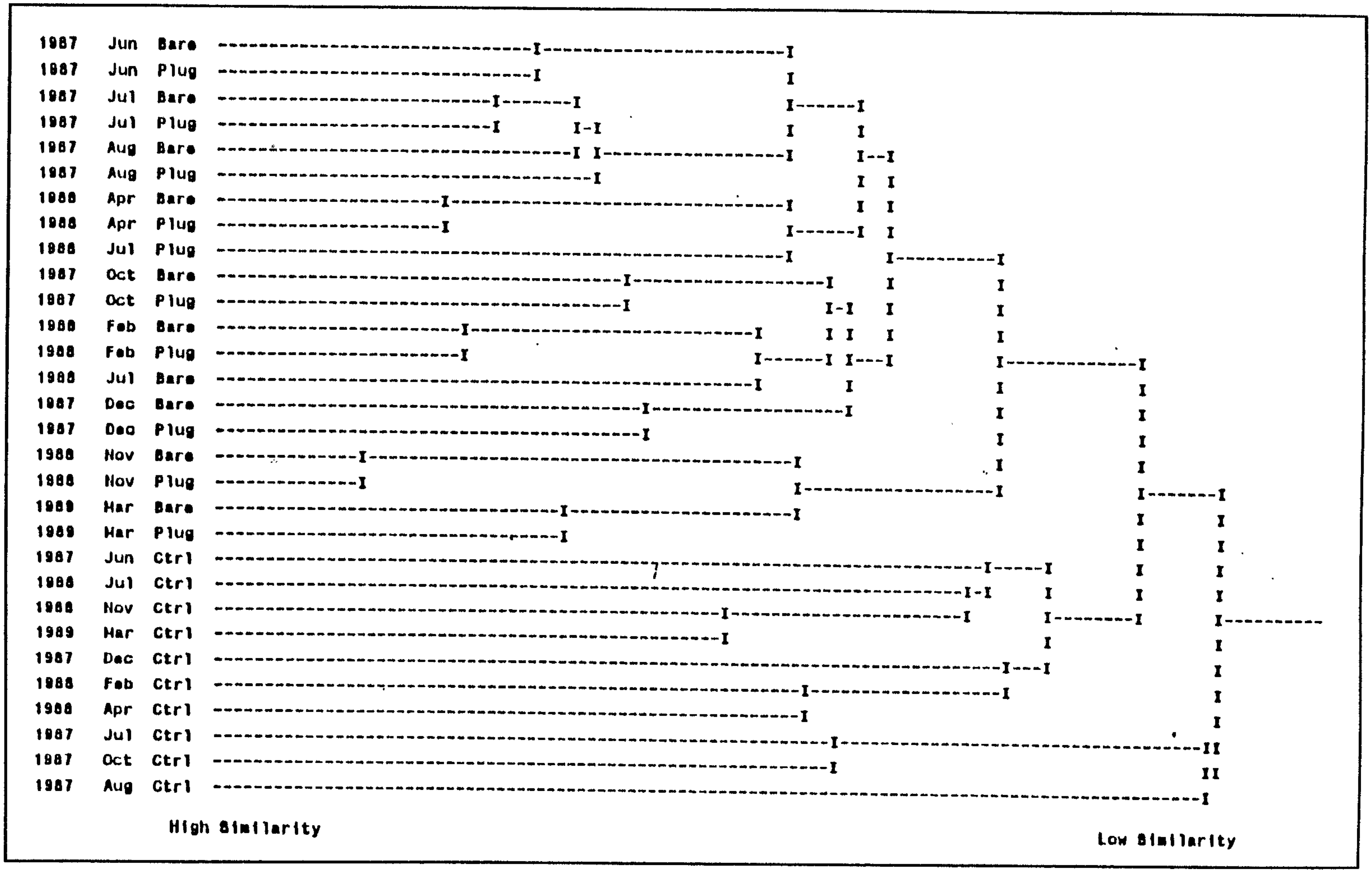

Figure 12. Cluster analysis dendrogram for Lassing Park benthic sampling stations and dates.

Figure 12 illustrates site groupings based on cluster analysis. The cluster diagram illustrates the dissimilarity between the control site and the other two sites. All of the control samples are grouped together at the bottom of the figure. The highest levels of similarity were found between the bare and plug stations for common sampling dates. The sites were less similar when compared between sampling dates.

DISCUSSION

By July 1988 it was apparent that the two saltmarsh plantings—Pinellas Point and Hendry/Redfish Creek—had taken root and were increasing in area of cover and density. In this respect the plantings will probably be successful in the short term, barring any catastrophic defoliations. While it also appears that the *Halodule* plantings of Lassing Park have successfully rooted, it was not immediately obvious that they would become successfully established, due to winter die off and only one year of follow-up monitoring data.

Pinellas Point

This site received two plantings; the first, in August 1986, exhibited a large die-off and was therefore replanted in April 1987. Rapid proliferation subsequently occurred at this site, with the plot nearing total coverage by July 1988. Distinct species preferences for the vegetated control were observed. The oligochaete complex, *Enchytraeidae* spp., consisted largely of a species living within the inner layers of *Spartina* stalks. Most other taxa exhibiting a preference for the control site appeared to be species exhibiting a clinging or crawling mode of foraging.

Similarity measurements indicate that the control marsh remained quite distinct from the bare control and the planted plot. This observation is also borne out by comparisons of the faunal parameters of the three sites. The vegetated control plot had far more species, greater diversity values, and two to four times more biomass (data not presented) than the other sites. Most of the organisms at the bare and planted sites were burrowing forms able to penetrate the root-free, soft sediments. Many of the organisms inhabiting the vegetated control plot are able to utilize the *Spartina* blades for food or attachment, or are well suited to penetrate and live among the sediment-root complex. The density of the *Spartina* at the vegetated control plot made it difficult to observe surface sediment features and fauna. However, there were large numbers of *Littorina irrorata*, an epifaunal grazing gastropod associated with *Spartina*, observed at the control plot. Very few *Littorina* were observed within the planted plot, indicating that perhaps a minimal density of *Spartina* is necessary for colonization by *Littorina*.

If the *Spartina* at this location continues to grow and spread, it is anticipated that the planted site will become more similar to the vegetated control. The close proximity of a naturally established marsh should enable the recruitment of a similar fauna. Presumably, the infaunal community structure is regulated by both surface coverage and subsurface sediment structure. In this case, the planted marsh may take several additional years to form a complete subsurface root structure that will physically and chemically alter the sediment characteristics.

Hendry/Redfish Creek

The sediments of the Hendry/Redfish Creek site are the result of a impoundment spill and are not a natural grain size distribution. The sediments exhibited a low nutrient level and a hard, smooth, compacted surface of fine sand generally of uniform size. Surface sediments had a low porosity with little bioturbation. A thick clay layer approximately 25 cm below the surface further complicates the sediment structure. Thus, the probability of this type of sediment supporting long-term *Spartina* growth and propagation was unknown. As of November 1990, the plantings were growing and propagating.

The Hendry/Redfish Creek revegetation site was the least productive of all the study sites in terms of benthic macrofauna. This site contained the fewest species, lowest faunal densities, and fewest epifaunal features of any site. Prior to planting, the only evidence of benthic infauna was the presence of *Uca* burrows visible at low tide. This plot was planted in an area with no nearby natural saltmarshes although there were extensive mangrove fringes. The lack of a suitable vegetated control plot prevented comparison of faunal composition and community parameters between planted and natural saltmarsh. Very little faunal similarity was displayed between the bare and planted plots at the Hendry site. This may be due, in part, to the low numbers of organisms collected during the study. Both plots were almost totally devoid of benthic fauna from February through July 1988.

The experimental plots at the Hendry site were established in the upper intertidal zone where they were subjected to extended aerial exposure. Unintentionally, the bare plot was situated very slightly higher in the intertidal zone than the planted plot. This resulted in the bare plot being uncovered sooner and remaining dry slightly longer than the planted plot. The three dominant taxa of the bare plot (*Capitella capitata, Laeonereis culveri*, and *Uca* sp.) were found in greater abundance at the planted site. This faunal distribution may have largely been governed by tidal exposure.

After one year of monitoring, indications were that the planted site was becoming more diverse than the bare control and supporting greater numbers of individuals. The recruitment rate of a marsh faunal community is expected to be much slower at this site than at the Pinellas Point location, due to lack of a nearby natural marsh and the foreign nature of the substratum. As the *Spartina* grows and spreads, it can be expected to alter the surface sediments by accumulating a detrital organic layer, and aerating or loosening the compacted sediments by penetration of roots and rhizomes.

Lassing Park

The benthic fauna of Lassing Park (the only seagrass site) was less definitive in trends than the saltmarsh sites. The site exhibited the greatest number of species, greatest faunal densities, and highest diversity values of all of the study sites. All three plots supported large numbers of taxa throughout the study. There were pronounced differences between the vegetated control and the experimental plots (bare and planted). There was not any definite differentiation between the bare and planted plots, indicating that any *Halodule* growth that occurred was not sufficient to alter the macrofaunal community composition.

There were no apparent differences in species composition between the bare and planted plots, but the vegetated control site contained many species not found at the other two sites. The gastropod *Diastoma varium*, which feeds on epiphytes growing on *Halodule* blades, was very abundant due to the dense cover of seagrasses and was therefore dominant at the control station. Fauna which were more abundant at the plug/bare plots were burrowing forms. The species more abundant at the control site, mostly deposit and filter feeders, were found associated with the sediment surface.

Faunal similarity analysis revealed no similarity between the vegetated control site and either the bare or plug sites. The bare and plug sites, in contrast, exhibited numerous high and moderate faunal similarities between them, including comparisons between sampling dates. The fewer high similarity pairs between sampling dates is due to seasonal variations in faunal composition and abundance.

The differences in relative density of *Halodule* in the planted and vegetated control sites was quite pronounced throughout the study. It is likely that significant alteration of the fauna from a bare substratum community to a vegetated substratum community will require much greater *Halodule* growth over the course of several years. These findings indicate that the benthic faunal community of the Lassing Park planted plot remained more closely associated with the bare control community for the duration of the study. All plots, however, supported a rich and diverse fauna. As the *Halodule* plugs grow and spread, and the root structure develops, the faunal composition of the planted area should become similar to the control area. It is apparent that this will take longer than the time during which this study was conducted.

ACKNOWLEDGEMENTS

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ASPECTS OF SEAGRASS RESTORATION IN TAMPA BAY

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ABSTRACT

This study, supported by the Florida Department of Natural Resources, the National Marine Fisheries Service, the University of South Florida and NOAA's Coastal Ocean Program, was conducted to determine if artificially propagated seagrass meadows provide habitat functions similar to those they are intended to replace as mitigation. We monitored the rate of seagrass growth, and shrimp, fish and crab numerical abundance and species composition in transplanted *Halodule wrightii* and *Syringodium filiforme* beds in Tampa Bay over a 2.8-year period. We also examined the recovery of *H. wrightii* and *S. filiforme* donor beds, which included excavations of up to 0.25 m^2, over a one year period. Transplanted shoot density of both *H. wrightii* and *S. filiforme* was predicted to reach an asymptote after 3.38 years. Shrimp and fish abundance in the transplanted *H. wrightii* beds became equal to natural beds after 1.2 years while crab abundance was still not equivalent after 2.8 years. *Syringodium filiforme* transplants followed similar trends. Faunal species composition in transplanted beds was still not similar to their natural counterparts after 2.8 years. *Halodule wrightii* and *S. filiforme* donor bed excavations recovered to equal or greater shoot densities within one year as compared to controls. Improvements in water quality in Tampa Bay, and especially Hillsborough Bay, as evidenced by concomitant studies, indicate that large-scale restoration of seagrass beds could provide a significant fishery resource enhancement with rapid recovery of the beds used for donor stock. The positive results of this study, however, do not support the substitution of created seagrass beds for existing beds as mitigation.

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EVALUATION OF LONG-TERM STUDIES OF THE BENTHIC COMMUNITY IN THE VICINITY OF BIG BEND, TAMPA BAY

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INTRODUCTION

Analysis of benthic infauna is a key element of many environmental monitoring programs. Due to the relative immobility of the constituent organisms and their varied sensitivity to physiological stresses, the benthic community is generally considered to be the most important faunal system in assessing environmental stress (Dills and Rogers 1972). In addition, the relative longevity of benthic organisms makes them valuable indicators of past and present water quality (Mackenthun 1966, McKee 1966, Cairns and Dickson 1971). As a result, countless investigations have been performed worldwide to monitor changes in benthic infaunal communities due to natural and anthropogenic influences.

Studies of the benthic faunal communities of Tampa Bay have been summarized by Taylor (1973), Simon (1974), and Simon and Mahadevan (1985). General conclusions regarding these communities include the following observations (Lewis and Estevez 1988):

- The estuary supports "an extremely abundant and diverse assemblage of bottom organisms, except in Hillsborough Bay..." (Taylor 1973);
- Seasonal fluctuations in the abundance and diversity of these organisms are pronounced;
- Opportunistic and "pollution indicator" species are abundant;
- Sediment type appears to be a controlling factor in determining infaunal distribution in the bay; and
- A general increase in species richness and decrease in total population abundance are evident on a north-to-south gradient in the bay.

While information is available on the spatial distribution of macrobenthic communities from Tampa Bay, including patterns associated with environmental gradients, little long-term data exist with which to evaluate the normal dynamics of these communities over extended time periods. Knowledge of the nature and causes of natural fluctuations is critical in assessing the effects of man's activities on estuarine communities. From 1976 to 1986, a series of studies was conducted on the benthic infaunal communities of the Big Bend region of Tampa Bay. Collectively, these studies provide the opportunity to investigate the long-term changes in faunal composition of a shallow subtidal benthic community within the Tampa Bay estuary.

The purpose of this evaluation is to: 1) describe the biotic and abiotic conditions within the benthic community of the Big Bend area; 2) identify within-year and between-year fluctuations in benthic community parameters; 3) identify temporal patterns in the population dynamics of the dominant species and determine the degree to which they shape overall community structure; and 4) compare these findings with those from other benthic communities within Tampa Bay.

MATERIALS AND METHODS — DATA ANALYSIS

Four separate studies (Table 1) were conducted to evaluate changes in benthic community structure due to various operational phases of Tampa Electric Company's Big Bend Steam Electric Generating Station (Mahadevan et al. 1977, 1980; Mahadevan and Culter 1985; Leverone and Mahadevan, 1986). Collectively, these studies provide the opportunity to analyze long-term changes in benthic faunal composition of the Big Bend area of Tampa Bay.

Table 1. Benthic studies conducted at Big Bend, Tampa Bay, between 1976 and 1986.

STUDY	DURATION	STATIONS	FREQUENCY
1	Feb 1976 - Feb 1977	22	Quarterly (every six weeks)
2	May 1979 - Dec 1979	11	Every six weeks
3	Aug 1981 - Feb 1984	5	Bimonthly
4	Apr 1984 - Feb 1986	15	Bimonthly

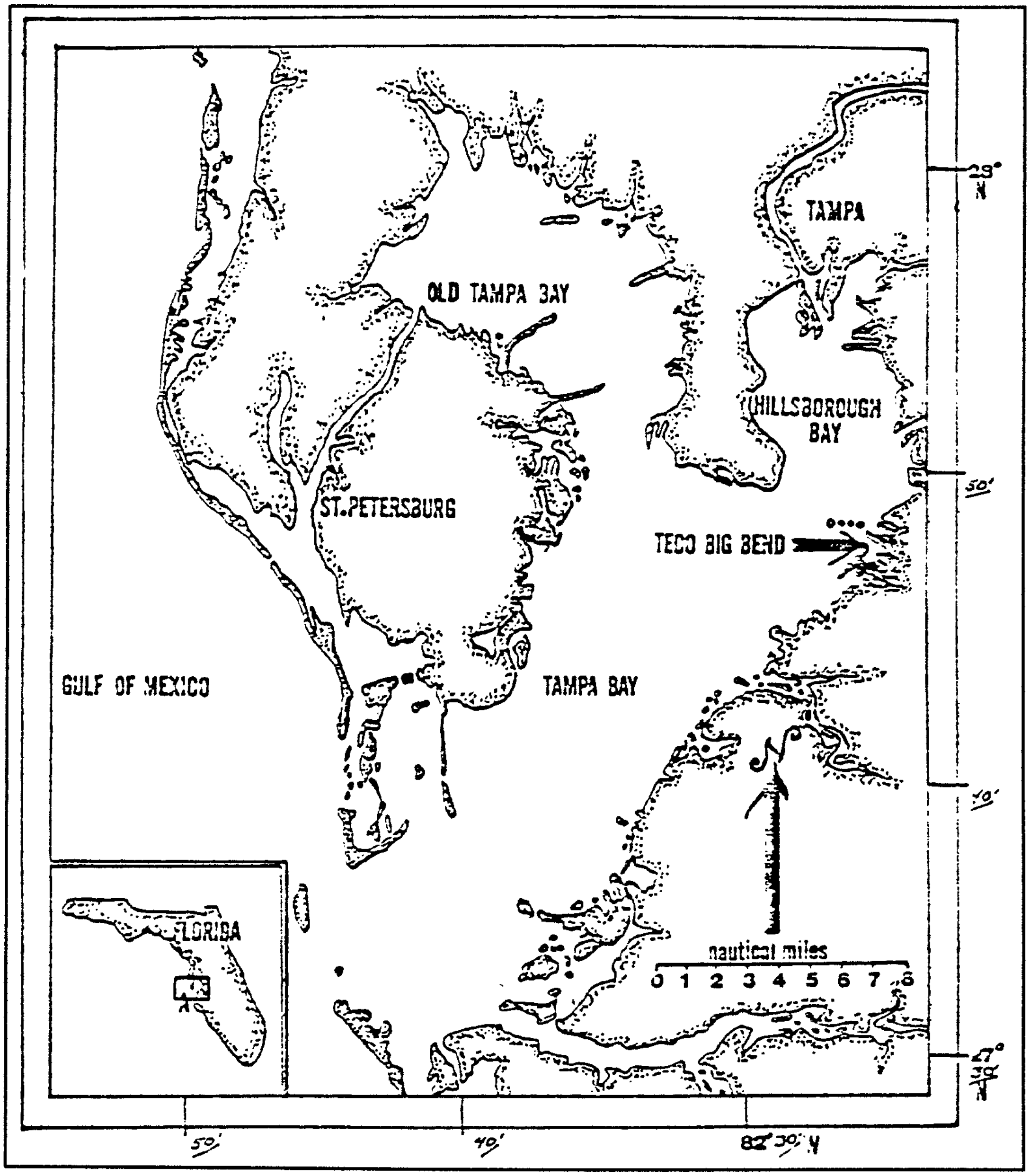

Figure 1. Map of Tampa Bay showing the location of the Big Bend area.

Description of Study Area

Big Bend (Figure 1) is located on the eastern shore of Tampa Bay (latitude 27°47′30"N, longitude 82°23′27"W), just above the line of demarcation between Middle Tampa Bay and Hillsborough Bay (Lewis and Whitman 1985). Benthic habitats in the area are devoid of vegetation. Sandy and muddy bottoms are prevalent with sandy-type habitats dominating nearshore areas (Mahadevan et al. 1977).

Station Selection

From a total of 22 stations, five were initially selected based on their high frequency of sampling over the entire ten year period (Figure 2). Preliminary analyses were conducted to determine the degree of faunal similarity between these stations for the same time period. An association matrix was generated for all station-date combinations and the Bray-Curtis distance quantitative coefficient employed to generate similarity matrices (Bray and Curtis 1957). Using the group-average clustering method, a high degree of similarity was found between stations 5 and 6 as well as between stations 11 and 12, while station 8 was highly dissimilar to these four stations. Station 8 was closest in proximity to the point of discharge from the power plant, and was shown to undergo slight changes in faunal composition as a result of the thermal plume during the colder months of the year (Leverone and Mahadevan 1986). As a result, station 8 was dropped from the long-term analysis. Stations 5 and 6 were located on a thermal transect while stations 11 and 12 were located on a control transect. These four stations were shown to be essentially unaffected by the thermal effluent from the power plant (Mahadevan et al. 1977). These stations were divided into two spatially distinct groupings (designated north and south stations) and each grouping analyzed separately for temporal trends in faunal composition.

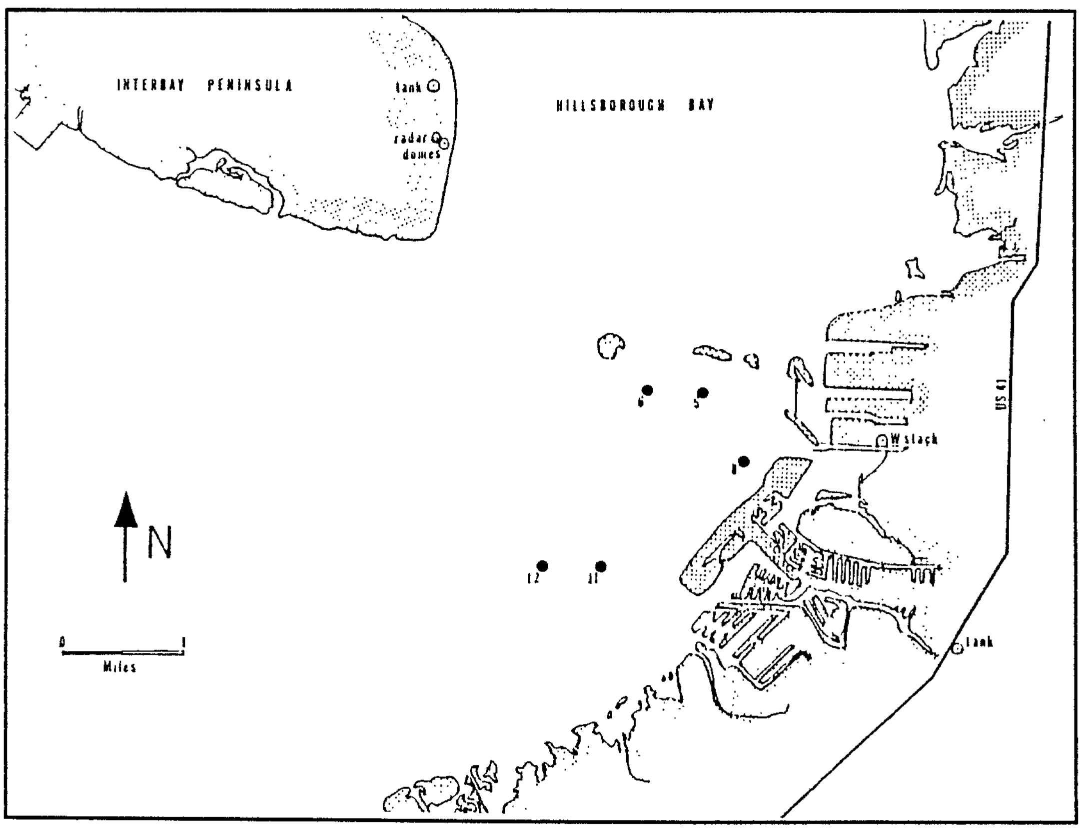

Figure 2. Station locations.

Sampling Frequency

Stations 5 and 11 were sampled every six weeks from February 1976 through February 1977, while stations 6 and 12 were sampled quarterly. During 1979, samples were collected approximately once every six weeks commencing in May and ending in December. Two separate collections were made in August 1979 to correspond with various power plant operations. Beginning in August 1981, all stations were sampled bimonthly until the studies were concluded in February 1986.

Sampling Procedures

Surface and bottom measurements were obtained for temperature, dissolved oxygen, salinity and turbidity. Temperature and dissolved oxygen measurements were made with a YSI Model 57 DO meter. Salinity measurements were made with a Beckman RS2-3 Inductive SCT meter. Surface and bottom water samples were collected with a 5-liter Nisken sampler for laboratory analysis of turbidity.

Faunal samples were collected with a petite Ponar grab, which samples an area of 0.0225 m^2 and penetrates approximately 10 cm in muddy substrates (slightly less in sandy substrates). Six replicates were collected and analyzed from each station. An additional sample was collected for sediment granulometry and percent organic content.

Faunal samples were washed and sieved on a 0.5 mm mesh sieve. A 10% $MgCl_2$ solution was used as a narcotizing agent. Samples were fixed in a 10% formalin/rose bengal solution and transferred to 70% isopropyl alcohol after 48 hours for final preservation. Organisms were separated from the sediments, identified and counted.

Data Analysis

Data were compiled from individual technical reports and collated. Species identifications were checked for synonymies and name changes before data entry. Nematodes and copepods were excluded from analysis. The following faunal analyses were performed:

- Species composition
- Abundance and percent composition of dominant species
- Faunal density (number of individuals/m^2)
- Species richness (number of species/station)
- Species diversity (H', Shannon and Weaver 1963)
- Equitability (J', Pielou 1966)

The coefficient of variation was employed as a measure of the persistence of the dominant populations over time (Sanders 1978). Together with the percent occurrence for each dominant population, a two-dimensional plot showing the relationship among dominant taxa was generated.

A comparison was made between this study and other quantitative benthic studies conducted in Tampa Bay to determine if a north-south gradient in species richness and faunal abundance exists within the Tampa Bay system.

RESULTS

Abiotic Parameters

Mean bottom temperature and salinity for all stations for each sampling date are presented in Figure 3. No appreciable difference in either temperature or salinity was observed among stations for any given sampling event. Temperature ranged from 14°C (February 1977) to 32°C (August 1981) and followed a typical seasonal pattern. Salinity ranged from 15 ppt (September 1979) to 32 ppt (February 1985), with values ranging between 25-28 ppt throughout most of the study.

Sediment mean grain size and percent silt/clay are presented as yearly means for all stations in Figure 4. Mean grain size was approximately 3 phi units and remained consistent from year to year. Yearly silt/clay values were highly variable and showed no long-term pattern. Sediments from the Big Bend area can be characterized as fine to medium sands with a moderate silt/clay content.

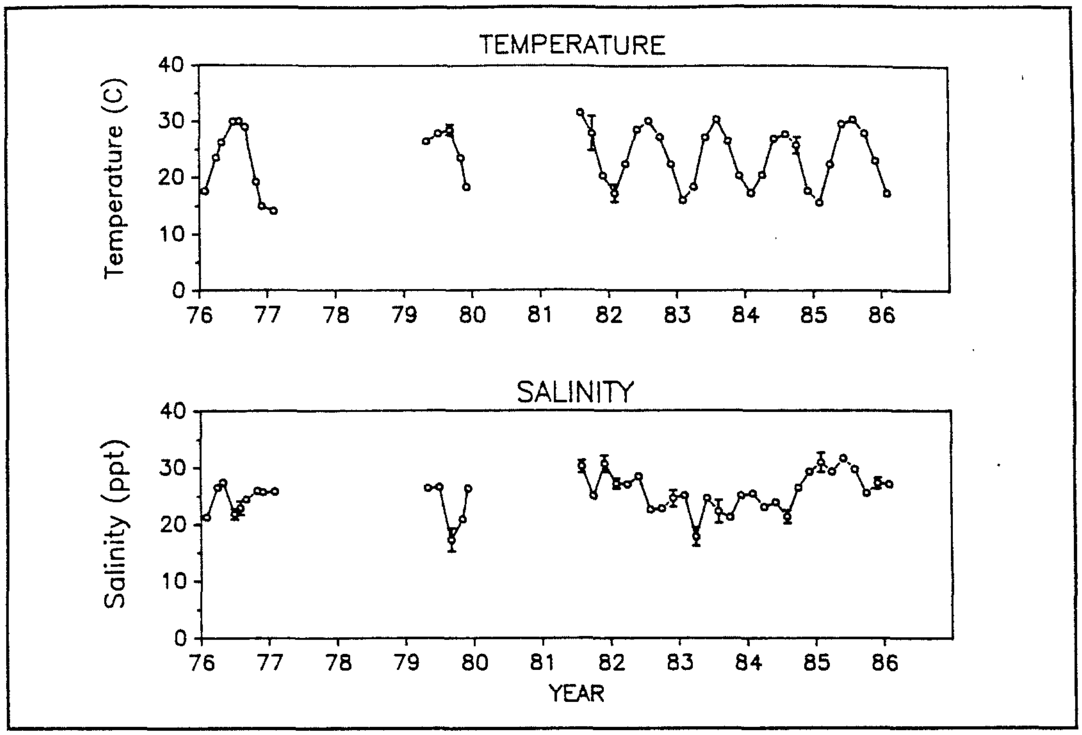

Figure 3. Mean bottom temperature (°C) and salinity (ppt) for all stations for each sampling period, 1976-1986.

Biotic Parameters

Figure 5 shows mean number of taxa and mean number of individuals (all stations-years combined) for each sampling month. This information addresses within-year or "seasonal" variability in faunal composition. The mean number of taxa was highest during winter and spring and lowest during late summer and fall. The mean number of individuals was highest during spring and early summer and lowest during late summer and fall.

The average number of taxa for north and south stations for each sampling date are presented in Figure 6. The number of taxa was lowest during the summer and fall of 1979, 1982 and 1983. Seasonal fluctuations in number of taxa were more pronounced at the north stations, where fewer taxa were present during the summer and fall of 1979, 1982 and 1983 than at the south stations. Since 1981, there has been a steady increase in the number of taxa at both the north and south stations in the Big Bend area.

The average number of individuals for north and south stations for each sampling date are presented in Figure 7. The average number of individuals was lowest at both sites during the fall 1979 followed by fall 1983 and fall 1982. Unusually high numbers of individuals were present in the spring and early summer 1983, followed immediately by a large decline in late summer and fall.

Average values for Pielou's index, a measure of the evenness with which individuals are distributed among species, are shown for north and south stations in Figure 8. Two large drops in the Pielou's index occurred at the north stations during the summers of 1982 and 1983, indicating a faunal dominance by one or a few organisms. Other years showed similar but less drastic reductions in Pielou's index during the summer and fall.

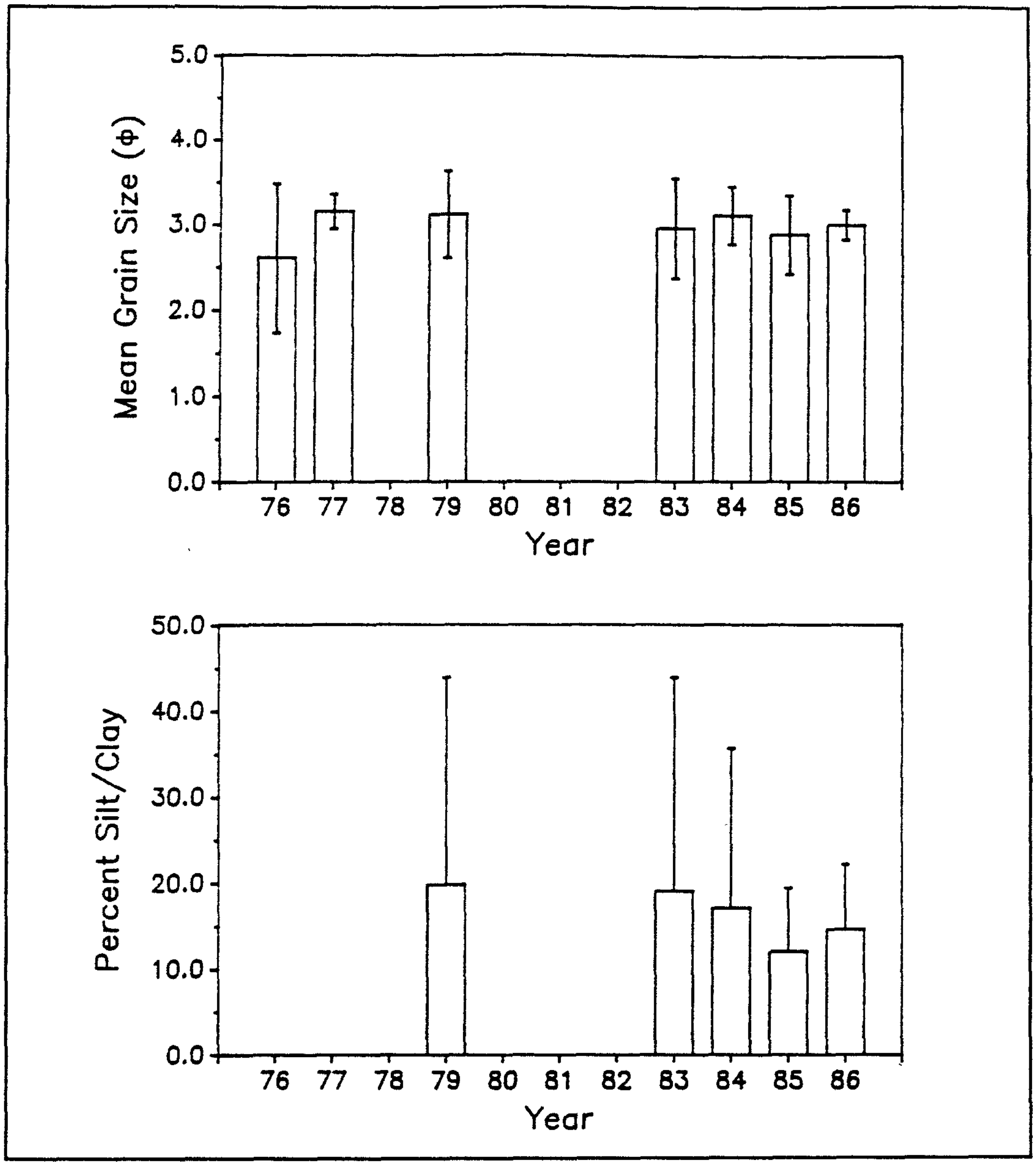

Figure 4. Sediment mean grain size and percent silt/clay for all stations for each year, 1976-1986.

Figure 9 summarizes the mean number of taxa and mean number of individuals (all stations-dates combined) for each year of the study. Mean number of taxa was lowest in 1979. Since 1979, there has been a steady increase in the mean number of taxa at the study site. The mean number of individuals was also lowest in 1979 and highest in 1983. No trends in the mean number of individuals were evident during the course of the study.

Dominant Taxa Analysis

Figure 10 is a plot of the coefficient of variation (CV) against the number of occurrences for each of the 25 dominant taxa from the study. Both terms are a measure of the "persistence" of a species over time, with the CV reflecting the variation in species population numbers. This figure presents a spatial portrayal of the different reproductive strategies employed by the various dominant species. Most of the dominant taxa exhibited a moderate coefficient of variation and a moderate to high number of occurrences. Several species have been highlighted to show the various extremes in reproductive strategy employed by the infauna of the Big Bend area. These reproductive strategies and the corresponding population dynamics of the highlighted species are described below.

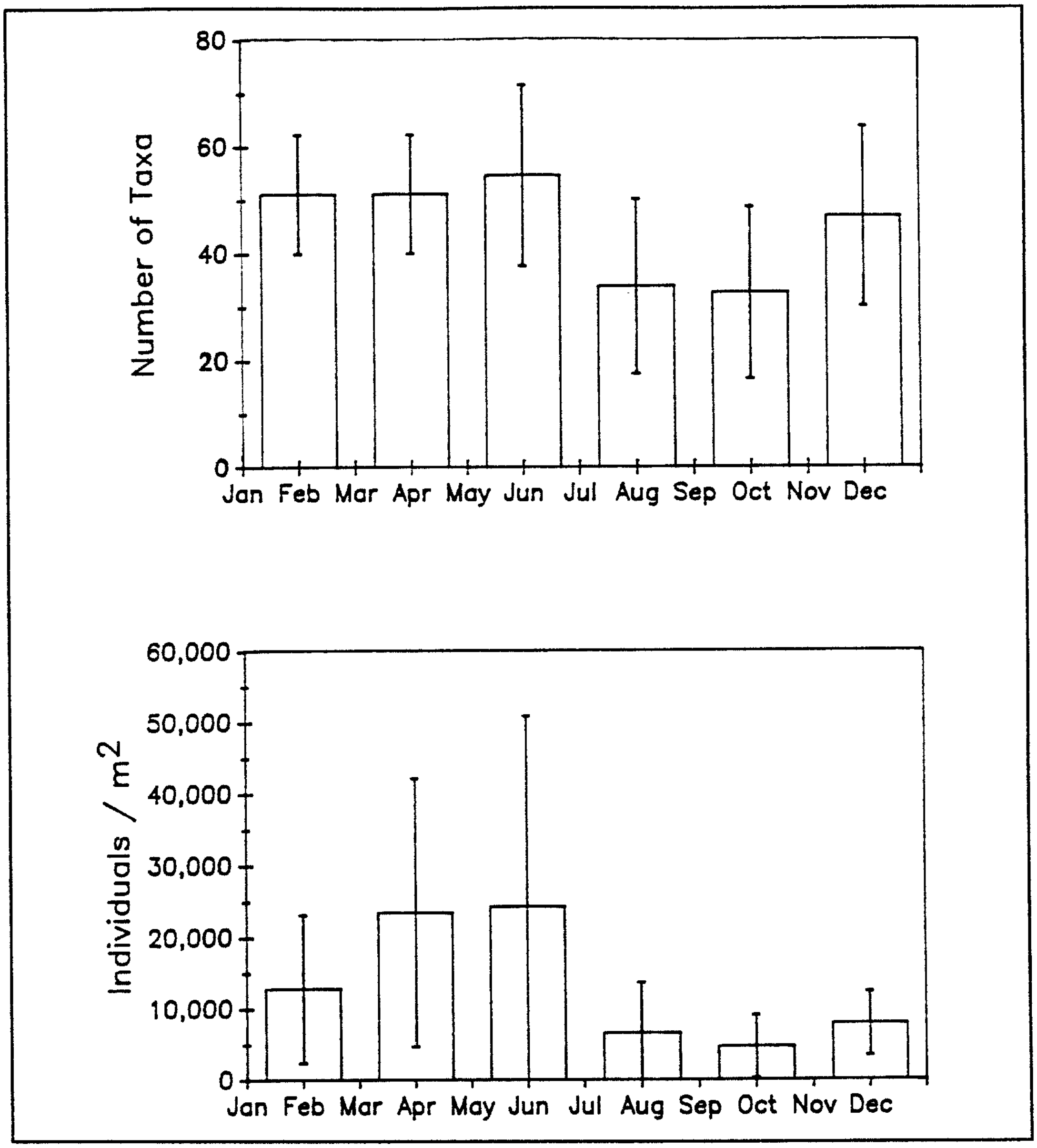

Figure 5. Mean number of taxa and mean number of individuals (all stations-years combined) for each sampling month, February-December.

Cumella sp. A (cumacean crustacean) exemplifies those species with a low number of occurrences and a high CV. These species are frequently absent from the study area, but occasionally experience large "population explosions" and greatly affect overall community structure. *Paraprionospio pinnata* (polychaete), on the other extreme, exhibited a high number of occurrences and a low CV, indicative of persistent species with a regular reproductive cycle. The majority of dominant species fell into a middle grouping characterized by moderately high numbers of occurrences and a low to moderate CV.

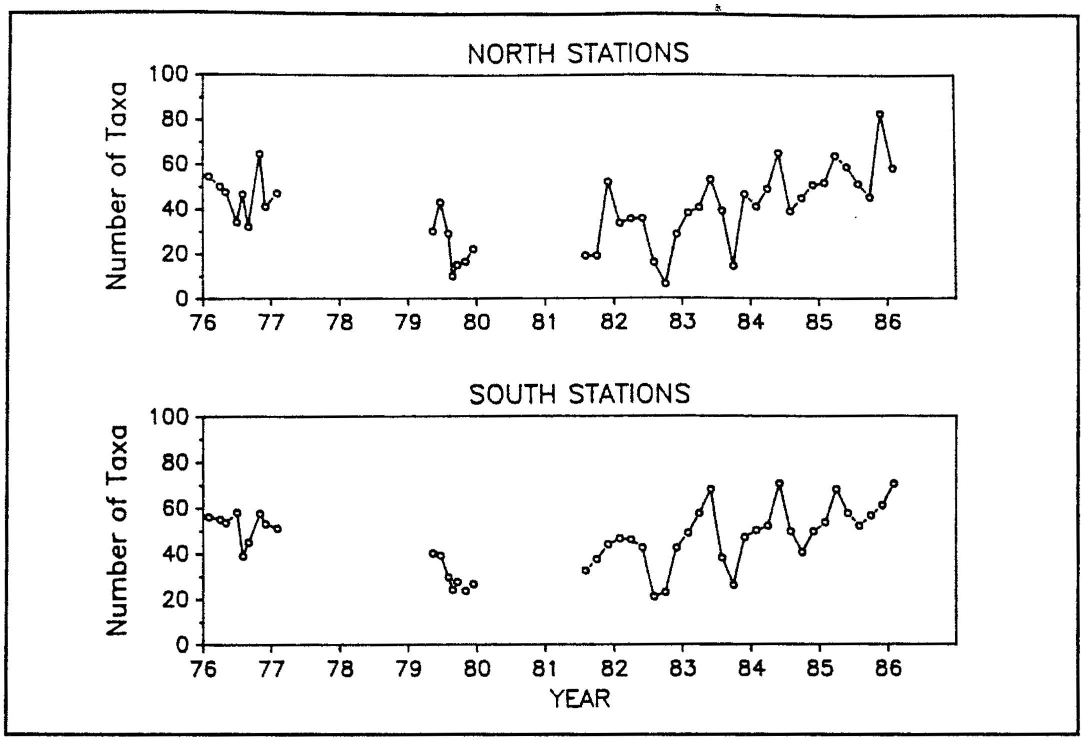

Figure 6. Average number of taxa for north and south stations for each sampling period, 1976-1986.

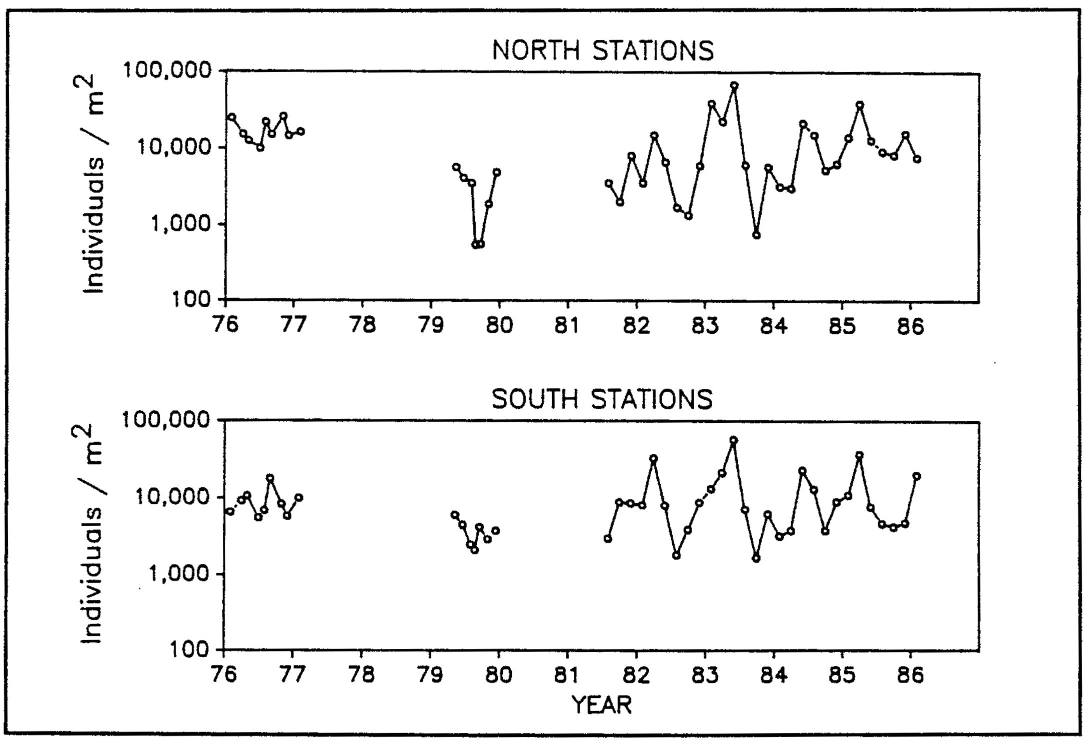

Figure 7. Average number of individuals for north and south stations for each sampling period, 1976-1986.

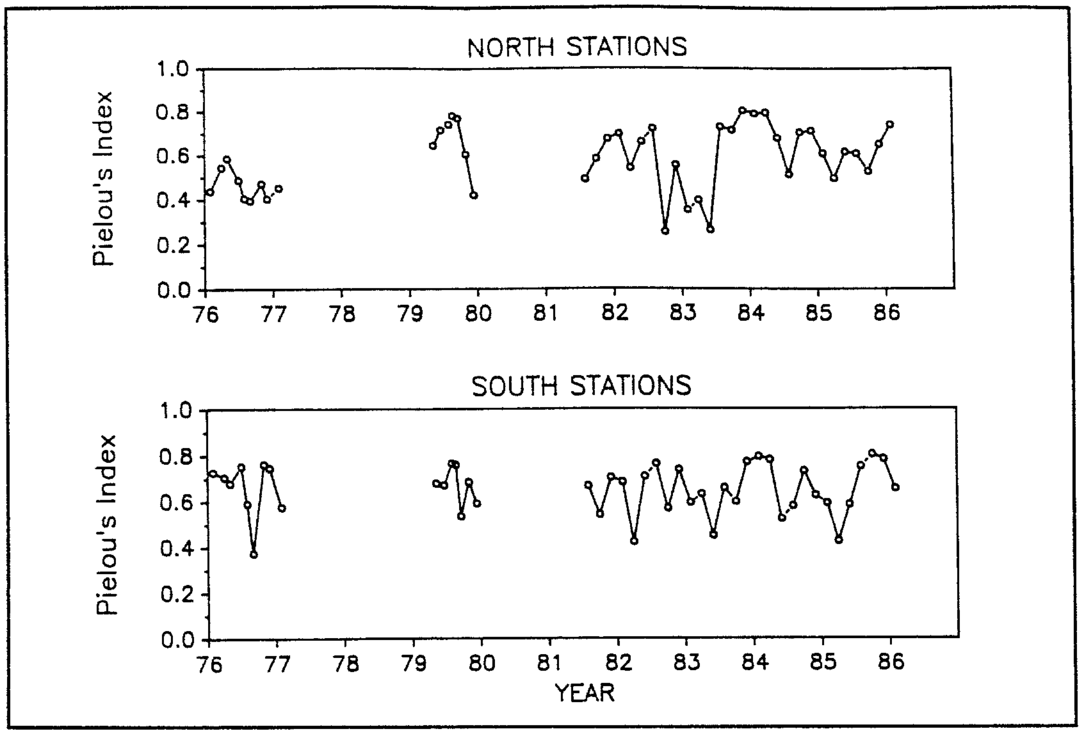

Figure 8. Average values for Pielou's index for north and south stations for each sampling period, 1976-1986.

Temporal trends in population dynamics of the five highlighted species are shown in Figures 11-15. *Cumella* sp. A (Figure 11) exhibited a large population increase during the first half of 1983, after which the species essentially disappeared from the study area. *Mulinea lateralis* (bivalve mollusc) also underwent a huge increase in numbers during the same period (Figure 12), but remained an integral part of the infauna during the rest of the study. Large population increases of the bivalve, *Tellina versicolor*, on the other hand, occurred during 1985 and 1986 (Figure 13). Finally, *Pinnixa pearsi* (decapod crustacean) and *Paraprionospio pinnata*, which represent fauna with a low CV and high number of occurrences, exhibited regular annual patterns in reproduction (Figures 14 and 15).

Comparisons of average species richness and faunal densities reported for subtidal benthic studies from Tampa Bay are presented in Table 2. Locations in Hillsborough Bay typically exhibited the lowest species richness, while regions of Lower Tampa Bay supported the highest number of species. Reported faunal densities varied throughout the bay system, but were generally within the range of 10-20,000 individuals/m^2. One notable exception to this pattern was the presence of 73,000 individuals/m^2 in Hillsborough Bay between 1975-1978.

DISCUSSION

The collected studies from Big Bend, Tampa Bay provide the opportunity to investigate both seasonal and long-term patterns in benthic community structure. The simultaneous collection of water column and sediment physical data allow for the potential to relate changes in faunal composition to physicochemical factors or possible environmental perturbations.

Most estuarine soft-bottom benthic communities appear to exhibit substantial seasonal variability in faunal composition (Boesch et al. 1976). The faunal community from Big Bend was no exception, experiencing distinct seasonal cycles in abundance and number of taxa throughout the study. Each year, faunal density and diversity were highest from late winter to early summer, followed by sharp reductions from late summer through fall. Similar seasonal patterns have been reported for other estuaries, including Apalachicola Bay (Mahoney and Livingston 1982) and Chesapeake Bay (Boesch et al. 1976). These seasonal cycles are typically driven by complex interactions between physical forces, such as temperature and salinity, and the more subtle biological factors which affect reproduction, recruitment and survival. The persistent regularity in the seasonality of the Big Bend benthic fauna probably reflects stable community dynamics associated with healthy, unperturbed environments. Additionally, the number of taxa found at Big Bend increased steadily from 1981 to 1986, after a low in 1979. While it is difficult to assess the nature or cause of this increase, it should be noted that increases in the number of taxa at Big Bend coincides with improvements in water quality in Hillsborough Bay (HCPEC 1987, City of Tampa 1988), located just to the north of Big Bend.

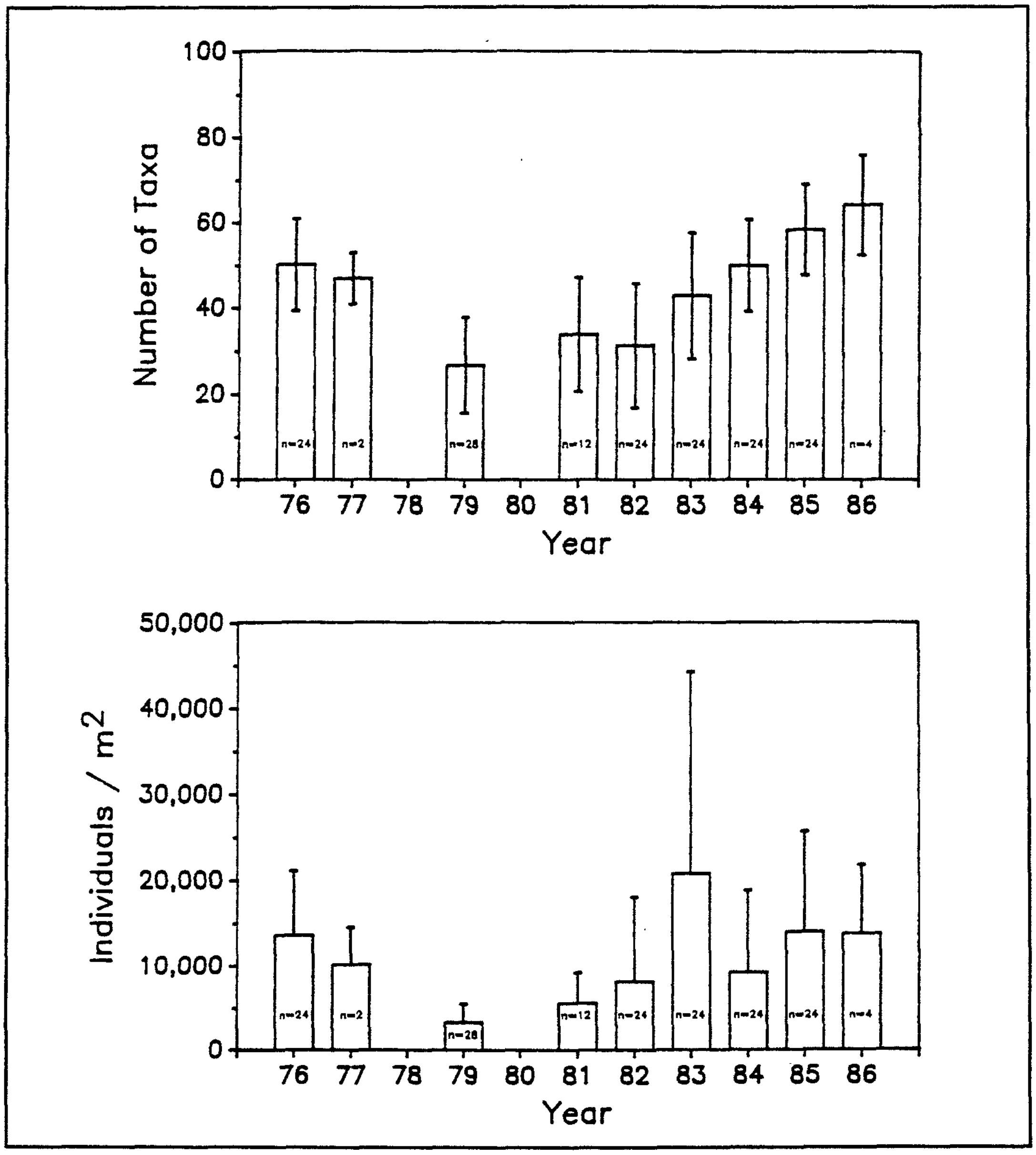

Figure 9. Mean number of taxa and mean number of individuals (all station-dates combined) for each year of the study, 1976-1986.

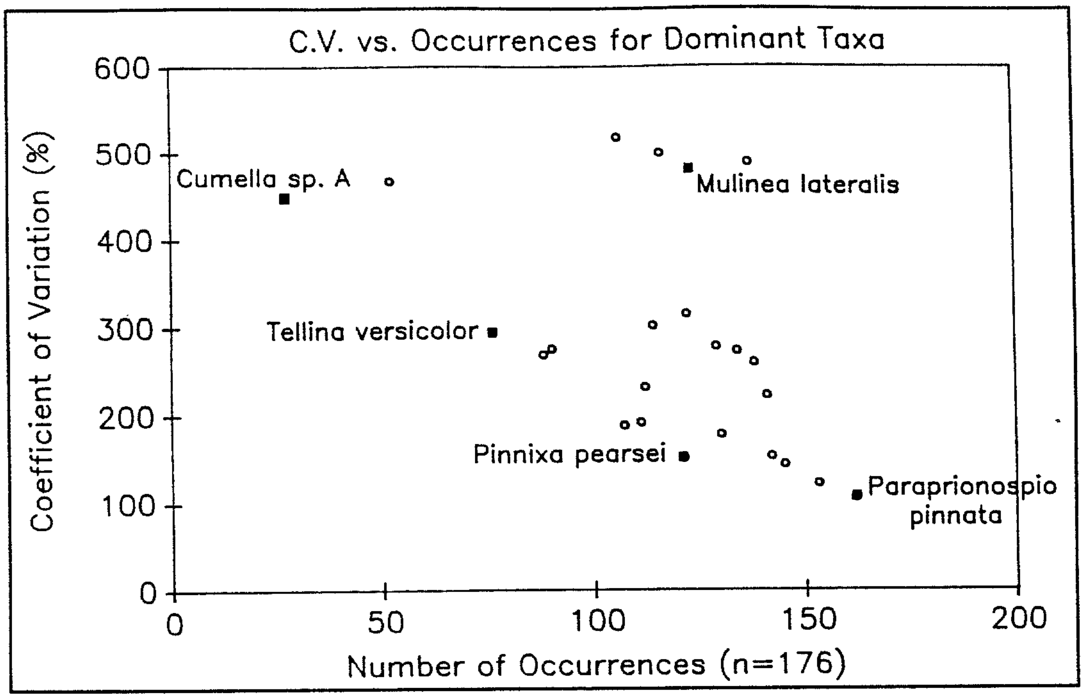

Figure 10. Coefficient of variation (%) versus number of occurrences for the 25 dominant taxa from the study area; ■ - highlighted species; O - remaining taxa.

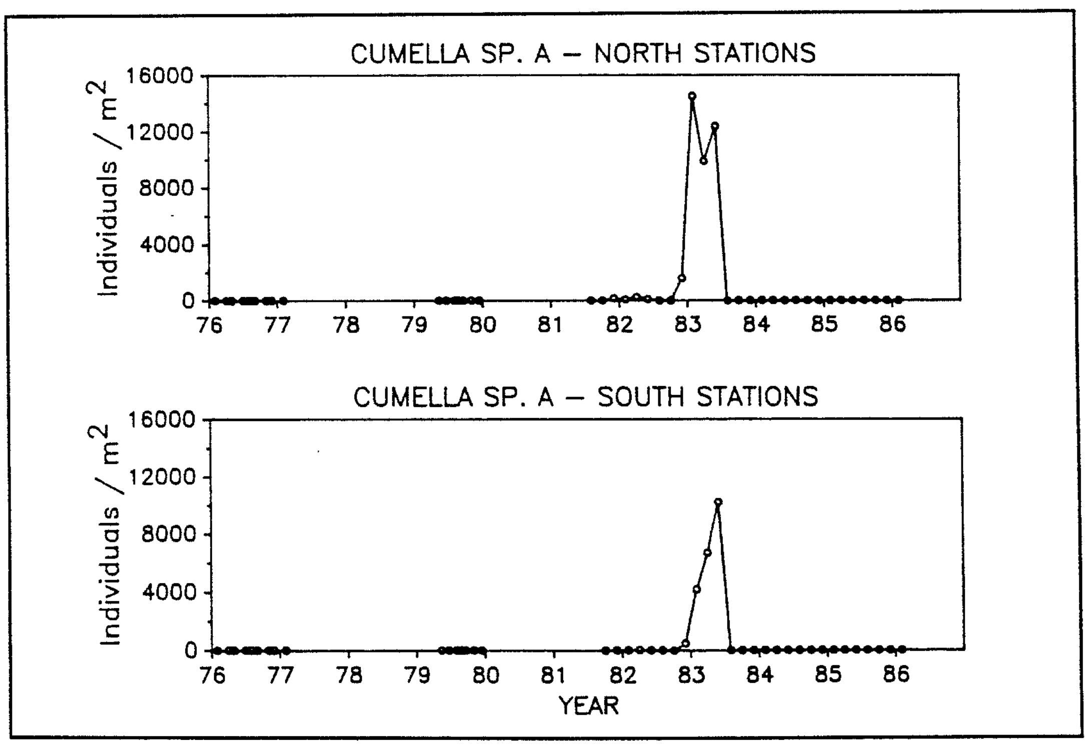

Figure 11. Average abundance of Cumella sp. A for north and south stations for each sampling period, 1976-1986.

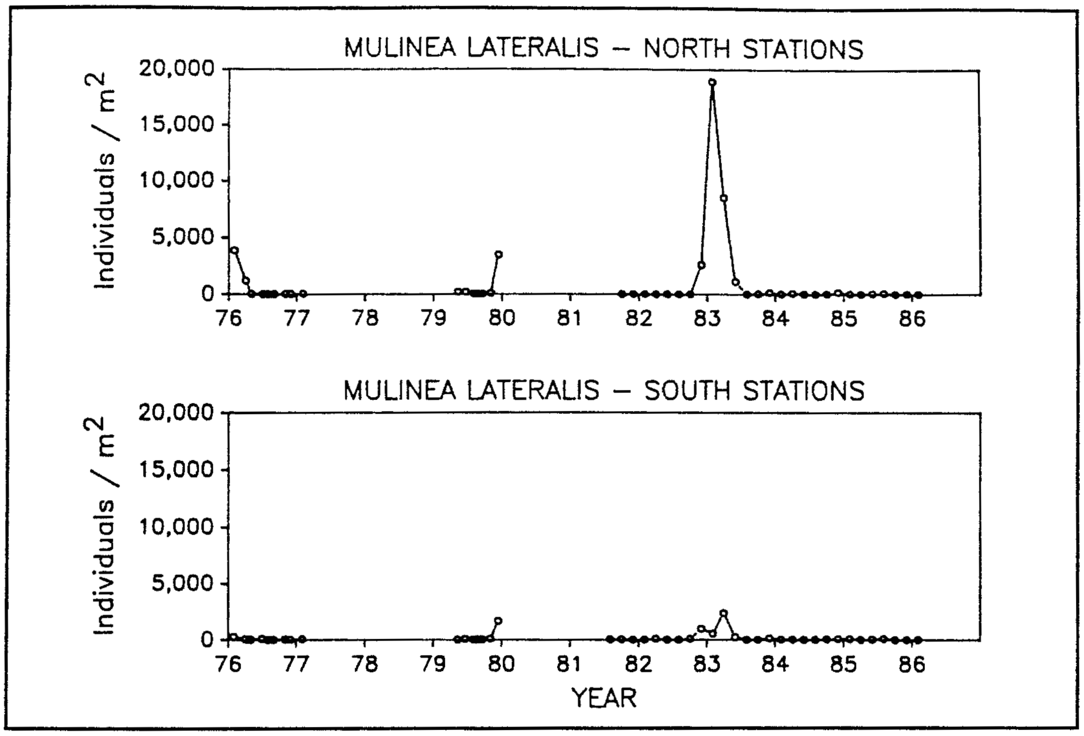

Figure 12. Average abundance of Mulinea lateralis for north and south stations for each sampling period, 1976-1986.

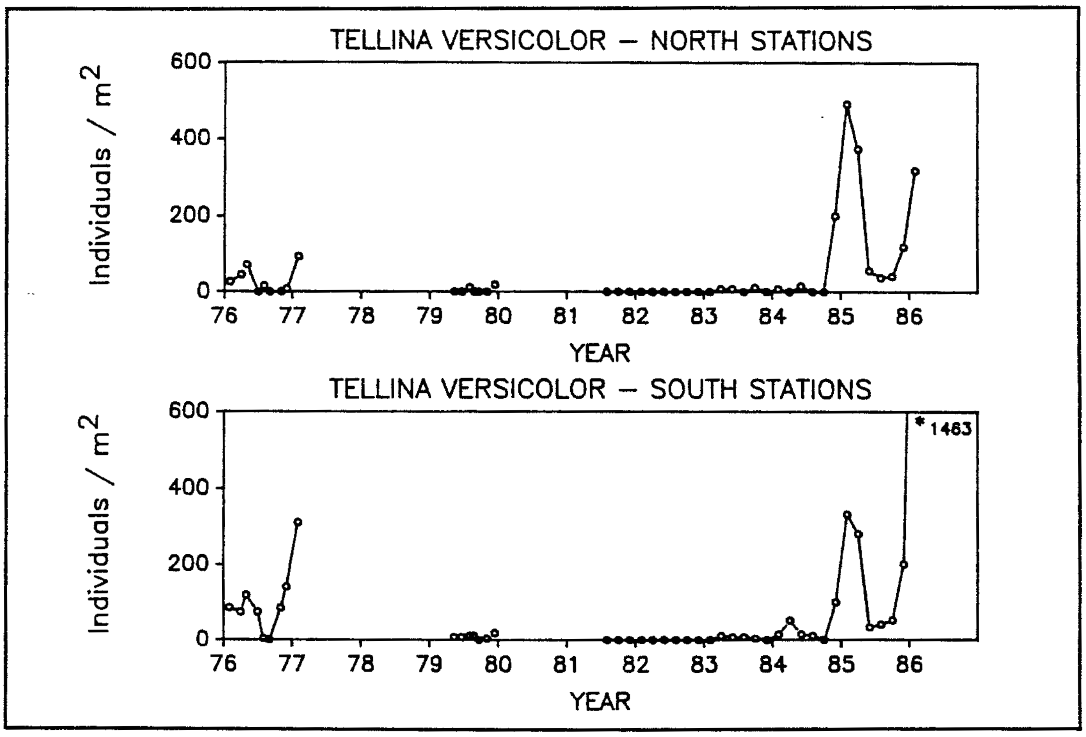

Figure 13. Average abundance of Tellina versicolor for north and south stations for each sampling period, 1976-1986.