Chapter 5. Water Quality

By Gerold Morrison (AMEC-BCI) and Holly Greening (Tampa Bay Estuary Program)

THE WATER QUALITY OF TAMPA BAY and its tributaries is an important ecological and economic issue for the west-central Florida region (Poe and others, 2006; TBEP, 2006). Water quality is a key factor affecting the ecological habitat value provided by the bay and helps to determine the types and numbers of plant and animal species it supports. From an economic perspective, many commercially and recreationally important fish and shellfish species are dependent on the water quality of the bay and its tributaries during some part of their life cycles (Lewis and Estevez, 1988; Wolfe and Drew, 1990; Killam and others, 1992). The economically vital recreation and tourism industries in the region also benefit from good water quality (fig. 5–1).

Seagrass meadows in Tampa Bay — which provide important habitat and food resources for many fish, shellfish (fig. 5–2), bird and mammal species — are directly dependent on good water quality (Dawes and others, 2004). As noted in Chapter 4, because seagrass meadows are so important to the ecology of the bay, managers have adopted numerical goals for the seagrass acreage that should be restored and maintained. Due to the sensitivity of seagrasses to reductions in water clarity, which in Tampa Bay have been associated with nutrient enrichment, much of the bay-wide water-quality management effort has focused on these issues and on the need to maintain water clarity at the levels necessary to reach the adopted seagrass restoration goals (Greening and Janicki, 2006; TBEP, 2006).

Figure 5–1. Terra Ceia Bay and Skyway Bridge from Emerson Point in Lower Tampa Bay. Photo by Holly Greening, Tampa Bay Estuary Program.

Figure 5–2. Bay scallop (*Argopectin irradians*) in seagrass meadow.

In addition to excessive nutrient enrichment and its impacts on seagrasses, other water-quality issues are also important in managing living resources of the bay region. Red tide and other harmful algal blooms (Paerl, 1988; Anderson and Garrison, 1997; Alcock, 2007) cause a variety of environmental impacts and potential human health effects. Elevated levels of mercury in the tissues of fish and other aquatic organisms also have potential impacts on the health of humans and wildlife (USEPA, 1997). Waterborne pathogens, associated primarily with contamination of water by human or livestock fecal material, but also (in some cases) by wildlife and other natural sources, can affect the use of surface waterbodies for recreation and as sources of potable water supplies (World Health Organization, 2003, 2006). As water monitoring technology continues to improve, allowing manmade chemicals to be detected in water samples at concentrations as low as the parts-per-billion (micrograms per liter) or parts-per-trillion (nanogram per liter) level, a number of emerging contaminants have also been identified — including several pharmaceutical and personal care products — whose potential environmental or human health impacts have not yet been thoroughly documented (Kolpin and others, 2002).

Connectivity between the Bay and its Watershed and Airshed

As an estuary, Tampa Bay can be defined very broadly as a “portion of the Earth’s coastal zone where there is interaction of ocean water, freshwater, land and atmosphere” (Day and others, 1989). This definition emphasizes the connectivity that exists between the bay, its watershed (see Chapter 1, fig. 1–3), and its airshed (shown in fig. 5–3). The watershed is the land area that contributes flows of freshwater and waterborne contaminants to the bay, whereas the airshed (Atkeson and others, 2007) is the much larger geographic area that contributes airborne contaminants, such as mercury and various N-containing compounds. To maintain a successful water-quality management program, managers will need to continue recognizing the connectivity that exists between these areas and addressing the sources and loadings of contaminants the bay receives from them.

Figure 5–3. Principal oxidized nitrogen (N) airshed for Tampa Bay. Estimates indicate that more than 35 percent of the atmospheric deposition of N to Tampa Bay originates outside of its watershed. Black oval = Tampa Bay airshed, green area = Tampa Bay watershed. From R. Dennis, Atmospheric Sciences Modeling Division, National Oceanic and Atmospheric Administration and U.S. Environmental Protection Agency.

Eutrophication in Tampa Bay—Past Problems, Recent Successes, and Ongoing Challenges

Like many estuaries throughout the world, one of the primary water-quality challenges facing Tampa Bay is cultural eutrophication — a process whereby human activities in the watershed and airshed lead to increased nutrient influxes to the waterbody, producing levels of over-fertilization that stimulate undesirable blooms of phytoplankton and macroalgae (Cloern, 2001; Bricker and others, 2007). Such blooms harm estuarine ecosystems in several ways. They reduce water clarity and block sunlight, reducing the size, quality, and viability of seagrass meadows and other aquatic habitats. Several bloom-forming phytoplankton species also produce toxins that can negatively affect the structure and function of aquatic food webs (Anderson and others, 2002) and pose health threats to wildlife and humans (World Health Organization, 2003, 2006; Burns, 2008; Havens, 2008). As phyto-plankton and macroalgae die and decompose, dissolved oxygen (DO) is removed from the water column and bottom sediments. Because an adequate supply of DO is essential to the survival of most aquatic organisms, such

Figure 5–4. Fish kill associated with a bloom of the microalgae *Pyrodinium bahamense* and very low dissolved oxygen readings in Old Tampa Bay, 2008. Photo by Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute.

reductions can have substantial impacts on the local fauna. Fish and other highly mobile organisms can often disperse from areas with reduced DO levels, but both they and the less mobile benthic infauna can be physiologically stressed or killed by lengthy exposures to DO values that reach hypoxic (DO < 2.0 milligrams per liter; mg/L) or anoxic (DO = 0 mg/L) levels (Gray and others, 2002; fig. 5–4).

Although phytoplankton and macroalgae require about 20 different nutrients and minerals to survive and reproduce (Reynolds, 2006), the macro-nutrients nitrogen (N) and phosphorus (P) tend to be the most important factors driving the eutrophication process in surface waterbodies (NRC, 2000). In pristine environments the availability of N and/or P is usually low enough to limit algal growth rates. By adding large amounts of biologically available N or P to surface waters, human activities can reduce or eliminate these nutrient limitations and stimulate bloom development.

Manmade sources of N and P that are contributing to eutrophication in the Tampa Bay watershed and elsewhere include urban, residential (fig. 5–5), and agricultural stormwater runoff, municipal sewage discharges, malfunctioning or improperly sited septic systems, and nutrient-enriched industrial discharges (for example, from facilities involved in the manufacture or shipping of fertilizer products) (TBEP, 2006). In addition, the combustion of fossil fuels for transportation, electric power generation, and other human uses generates atmospheric N oxide emissions, and residential and agricultural fertilizer applications and other agricultural activities generate atmospheric ammonia emissions. These N oxide and ammonia emissions can contribute to the nutrient loads received by the bay and by many other surface waterbodies (for example, Paerl, 1997; Poor and others, 2001; Pollman, 2005; TBEP, 2006).

Estuaries and other coastal waterbodies vary a great deal in their susceptibility to eutrophication. The susceptibility depends largely on their flushing characteristics and hydraulic residence times, which are influenced by tidal forces, freshwater inflows, and bathymetry (Bricker and others, 2007). Estuaries in which water and nutrients are rapidly flushed allow insufficient time for algal blooms and other symptoms of eutrophication to develop, and show relatively low susceptibility to nutrient influxes. Those with longer residence times allow more time for nutrients to be taken up by phytoplankton and macroalgae, providing opportunities for undesirable blooms to form and persist (Cloern, 2001). Most parts of Tampa Bay appear to be flushed relatively quickly, particularly during periods when adequate freshwater inflows and favorable winds occur (Goodwin, 1989; Weisberg and Zheng, 2006; Meyers and others, 2007). This makes the bay as a whole less sensitive than it would otherwise be to increasing nutrient influxes.

Despite its relatively rapid flushing characteristics, however, Tampa Bay exhibited symptoms of extreme nutrient enrichment during the late 1970s and early 1980s (Johansson, 1991), a period when it was receiving much larger nutrient loading than it does today (Zarbock and others, 1994; Janicki Environmental Inc., 2008). Those symptoms included large and frequent blooms of phytoplankton and macroalgae (fig. 5–6), reduced water clarity, reductions in the areal extent and ecological quality of seagrass meadows, increased variability in DO concentrations, and increased frequency of stressfully low DO levels. Eutrophication impacts were particularly severe in Hillsborough Bay, the part of Tampa Bay that was receiving the largest contributions from municipal sewage effluent and industrial leaks and spills during that period (Santos and Simon, 1980; Johansson and Squires, 1989; Johansson, 1991; Johansson and Lewis, 1992).

Figure 5–5. Residential lawn fertilization.

Figure 5–6. Macroalgae (*Ulva*) mat in Hillsborough Bay. Photo by Roger Johansson, City of Tampa.

Fortunately, water quality in the bay is much better now than it was in the late 1970s and early 1980s, making Tampa Bay one of the few estuaries in the U.S. that has shown evidence of improving environmental conditions in recent decades (Johansson and Lewis, 1992; Greening and Janicki, 2006; Morrison and others, 2006; Bricker and others, 2007; Duarte and others, 2009). These water-quality improvements have been due, in large part, to upgrades in wastewater-treatment practices at municipal wastewater-treatment plants in the region (fig. 5–7). Since 1980, all wastewater-treatment plants that discharge to the bay or its tributaries have been required by State legislation (the Grizzle-Figg Act; Section 403.086, Florida Statutes) to meet advanced wastewater-treatment standards, a step that has reduced annual nutrient loads from these sources by about 90 percent (Johansson, 1991; Johansson and Lewis, 1992; TBEP, 2006). In addition to these infrastructure upgrades, the bay has also benefited from:

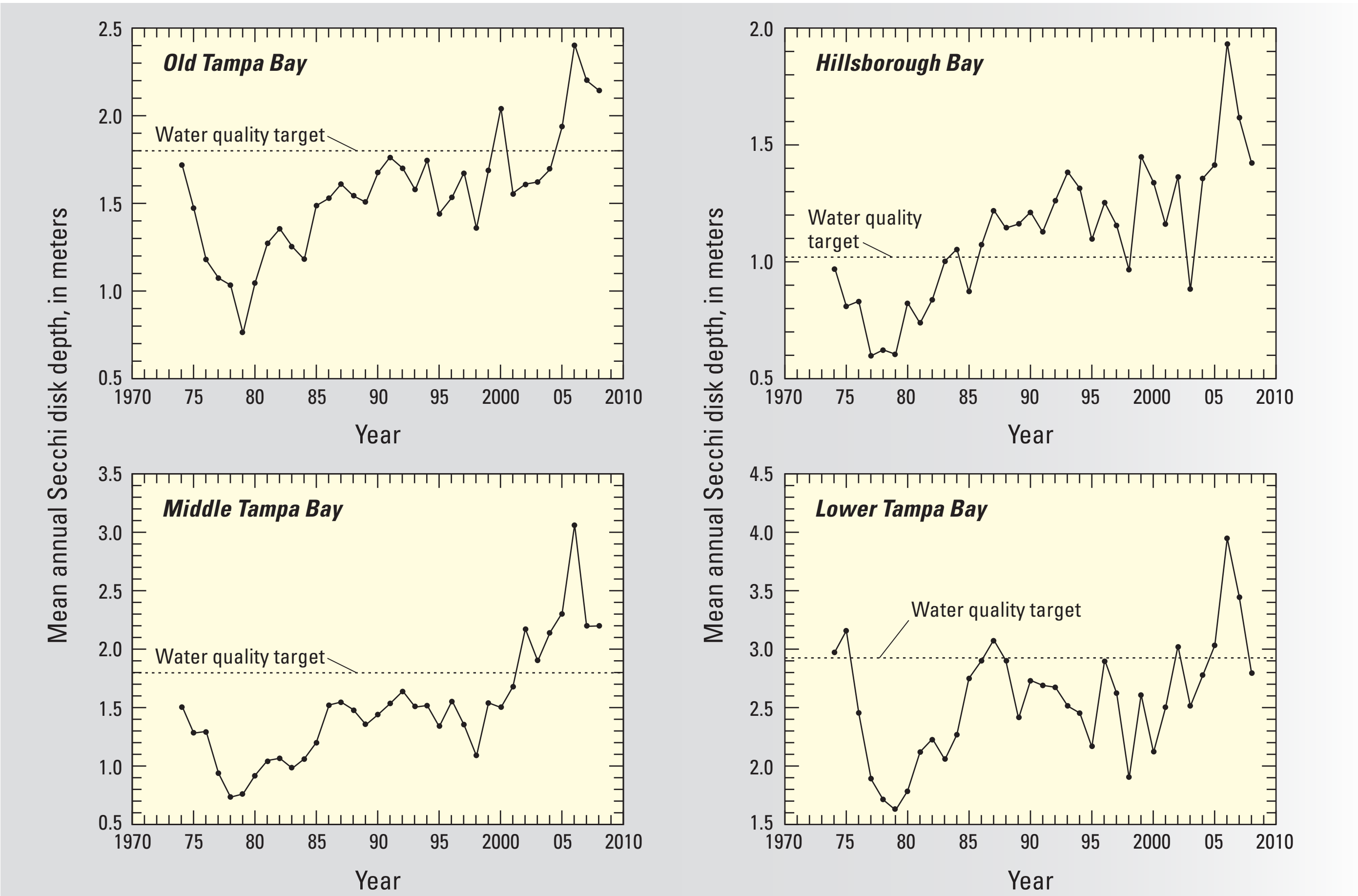
- Reductions in dredge-and-fill activities;
- Reduced discharges from fertilizer manufacturing facilities and port facilities during the shipping of fertilizer products;
- Reduced atmospheric N emissions from electric power generating stations;
- Improvements in urban and industrial stormwater management practices; and
- Improved pollution control by agricultural operations (Greening and Janicki, 2006; TBEP, 2006).

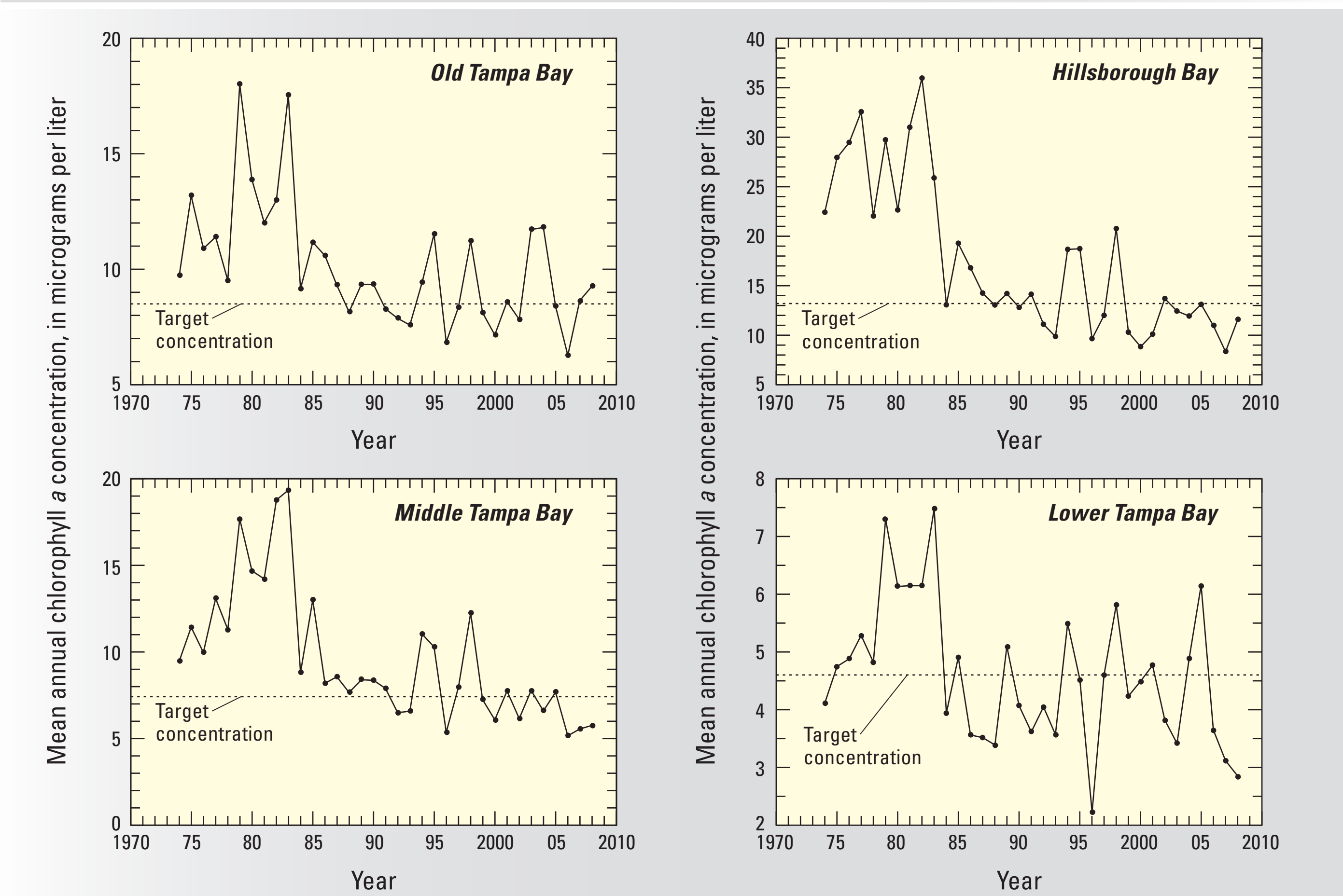
Time-series plots of a number of important water-quality indicators, including water clarity, chlorophyll *a* (an indicator of phytoplankton abundance), and DO concentrations, show the water-quality impacts that occurred during the late 1970s and early 1980s and the improvements that have occurred since that time (figs. 5–8, 5–9, 5–10) Additional information on the eutrophication issue and management of nutrient loadings is given below.

Figure 5–7. Aerial view of H.F. Curren wastewater-treatment plant. Photo by Southwest Florida Water Management District.

Figure 5–8. *Opposite page, top* Water clarity as measured by average annual Secchi disk depth, 1974–2008, for Hillsborough Bay, Old Tampa Bay, Middle Tampa Bay and Lower Tampa Bay. Horizontal lines depict Tampa Bay Estuary Program water-quality targets. All points above lines are meeting targets. Data from Environmental Protection Commission of Hillsborough County.

Figure 5–9. *Opposite page, bottom* Chlorophyll *a* annual average concentrations, 1974–2008, for Hillsborough Bay, Old Tampa Bay, Middle Tampa Bay, and Lower Tampa Bay. Horizontal lines depict Tampa Bay Estuary Program target concentrations supporting seagrass growth. All points below lines are meeting targets. Data from Environmental Protection Commission of Hillsborough County.


Mean annual Secchi disk depth, in meters
Old Tampa Bay
Water quality target
Hillsborough Bay
Water quality target
Middle Tampa Bay
Water quality target
Lower Tampa Bay
Water quality target
Year
2.5
2.0
1.5
1.0
0.5
3.5
3.0
4.5
4.0
1970
75
80
85
90
95
2000
05
2010


Mean annual chlorophyll a concentration, in micrograms per liter
Old Tampa Bay
Target concentration
Hillsborough Bay
Target concentration
Middle Tampa Bay
Target concentration
Lower Tampa Bay
Target concentration
Year
20
15
10
5
0
40
35
30
25
8
7
6
4
3
2
1970
75
80
85
90
95
2000
05
2010

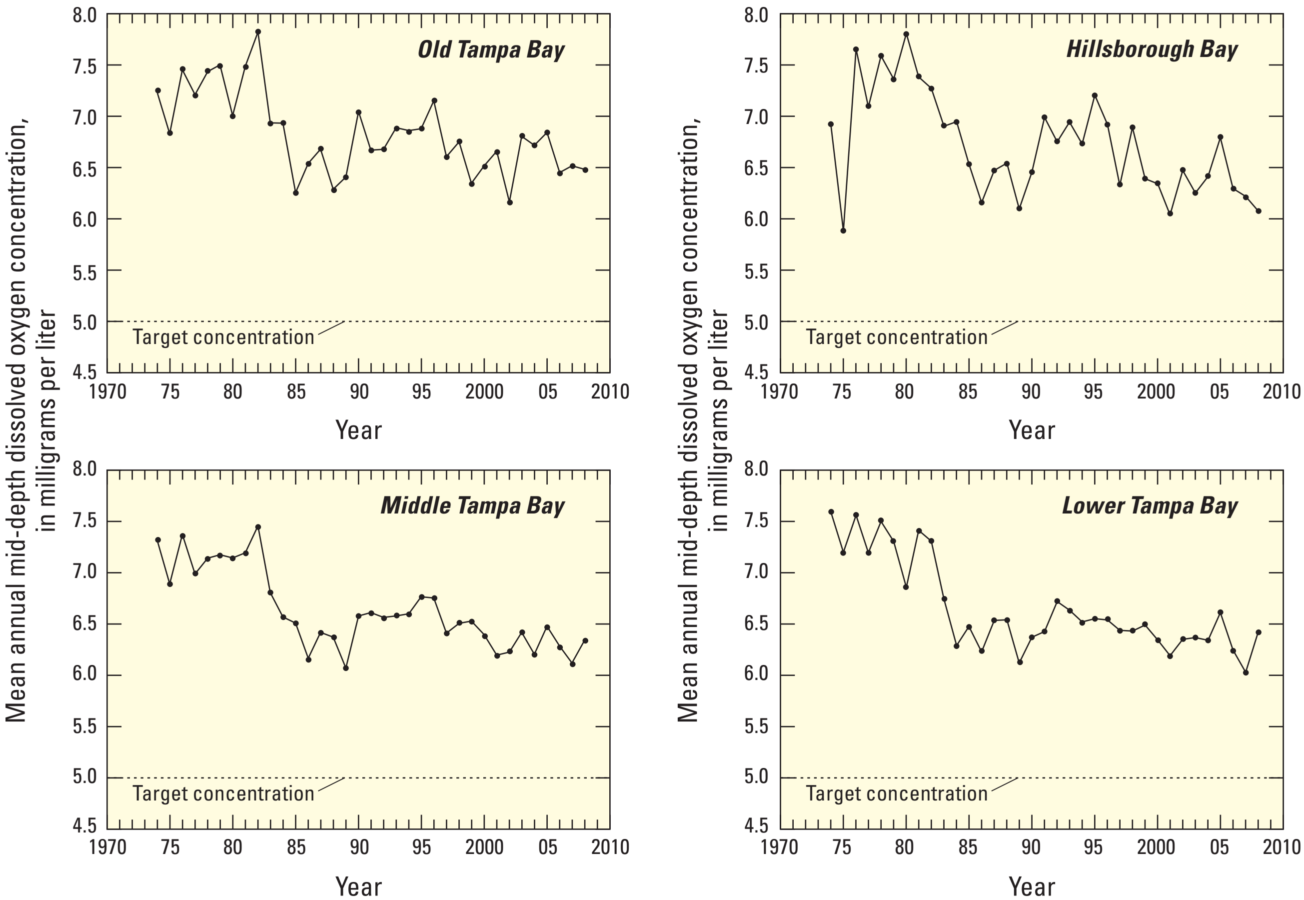


Figure 5–10. Average annual mid-depth dissolved oxygen concentrations, 1974–2008, for Hillsborough Bay, Old Tampa Bay, Middle Tampa Bay and Lower Tampa Bay. Horizontal lines depict State criteria for daily average dissolved oxygen concentrations. Data from Environmental Protection Commission of Hillsborough County.

Despite the dramatic nutrient-related water-quality improvements that have occurred in Tampa Bay since the 1980s, other water-quality issues still remain to be addressed. Within the watershed, the FDEP and the USEPA have identified a large number of freshwater bodies that are not currently meeting State or Federal water-quality standards and, therefore, are designated as "impaired" (fig. 5–11). The bay itself is also designated as impaired due to elevated levels of mercury that are found in several fish species inhabiting its waters. Currently, numerous rivers and all estuarine and marine waterbodies in Florida are listed as impaired for this reason. Portions of the bay and watershed are also classified as impaired due to occasionally elevated levels of fecal indicator bacteria, which prevent those areas from meeting their designated uses as swimming beaches or approved shellfish harvesting areas (FDEP, 2001). Portions are also still classified as impaired because of excessive nutrient enrichment, although all major bay segments have been meeting locally developed N load management and water clarity goals in recent years.

The Tampa Bay estuary and its watershed are not unique in containing a large number of impaired waters. The USEPA estimates that over 40 percent of surface waters in the United States do not currently meet

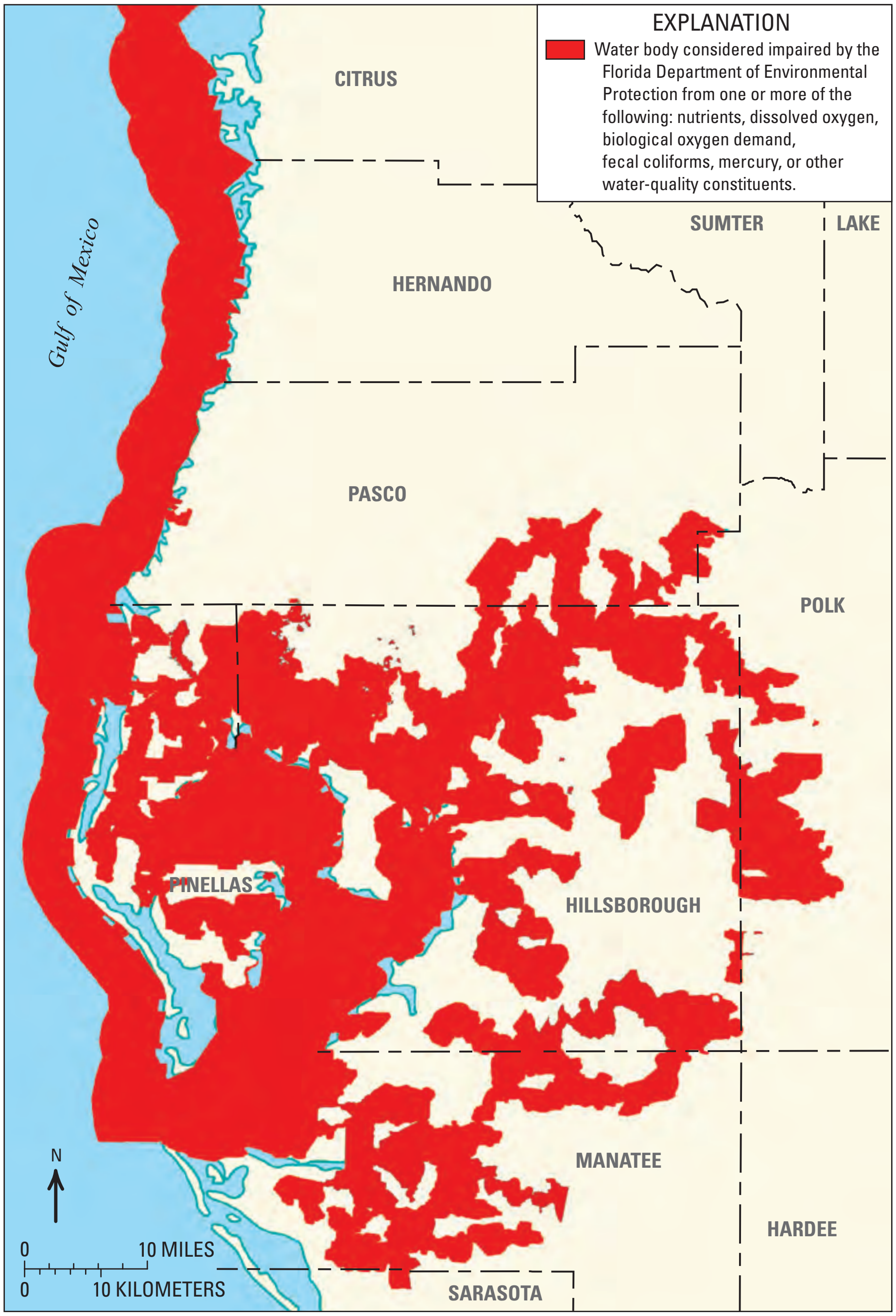


Figure 5–11. Impaired waterbodies within the Tampa Bay watershed, 2009. Red areas are considered impaired for one or more of the following: nutrients, dissolved oxygen, biological oxygen demand, fecal coliforms, mercury or other water-quality constituents. From Florida Department of Environmental Protection.

water-quality standards. On a nationwide basis, this amounts to over 20,000 individual river segments, lakes, and estuaries. These impaired waters include about 300,000 mi of rivers and shorelines and about 5 million acres of lakes, which are impacted primarily by excess nutrients, sediments, and potentially pathogenic microorganisms. The USEPA estimates that a large majority of the U.S. population — 218 million people —lives within 10 mi of an impaired waterbody (*http://www.epa.gov/owow/tmdl/*).

Under the Federal Clean Water Act, States are required to identify the impaired waters within their jurisdiction and to develop total maximum daily loads (TMDLs) that estimate the reductions in pollutant loads that need to be achieved to eliminate those impairments. For a given impaired waterbody, a TMDL estimates the maximum amount of a particular impairment-causing pollutant that the waterbody can receive and still meet water-quality standards. A TMDL also allocates the allowable load among point and nonpoint pollutant sources (NRC, 2008). Both the State of Florida and the USEPA are currently involved in developing TMDLs to address water-quality impairments in Tampa Bay and its contributing watershed.

Water-Quality Monitoring

As figures 5–8 through 5–10 demonstrate, Tampa Bay managers are fortunate to have a water-quality database that extends back to the early 1970s, allowing them to track changes in water quality that have occurred over a period of more than three decades. Much of the long-term monitoring has been done by EPCHC, which began collecting water-quality data in 1974 and currently conducts monthly monitoring of more than 50 stations in Tampa Bay (fig. 5–12) and a comparable number of stations in the Hillsborough County part of the watershed. The EPCHC monitoring network has been a particularly valuable data source for tracking long-term changes in bay water quality, because it includes most major bay segments, extends over the longest time period, and has striven to maintain consistent sampling and analytical methods throughout the period. From 1978 through 2011, the City of Tampa has also maintained an active monitoring program, focused primarily on Hillsborough Bay but extending into other bay segments. In recent decades, Manatee County, Pinellas County, FDEP, and SWFWMD have also carried out monitoring programs, providing valuable water-quality information for the parts of the bay and watershed that fall within their jurisdictions.

A Regional Ambient Monitoring Program, organized by the TBEP in the early 1990s and continued by local governments and agencies through the present, provides a coordinating forum that works to maintain consistency in the sample collection and analytical methods used by the various monitoring organizations. Over time, with support from the National Estuary Program offices in Sarasota Bay and Charlotte Harbor and other local and regional organizations, the Regional Ambient Monitoring Program has expanded, encouraging greater methodological consistency and data comparability among surface-water monitoring programs throughout the west-central and southwest Florida region.

The availability of the consistent, long-term water-quality data provided by the local monitoring programs in the Tampa Bay area has supported bay management efforts in a number of important ways, allowing managers to document the negative impacts of the excessive pollutant loads that the bay

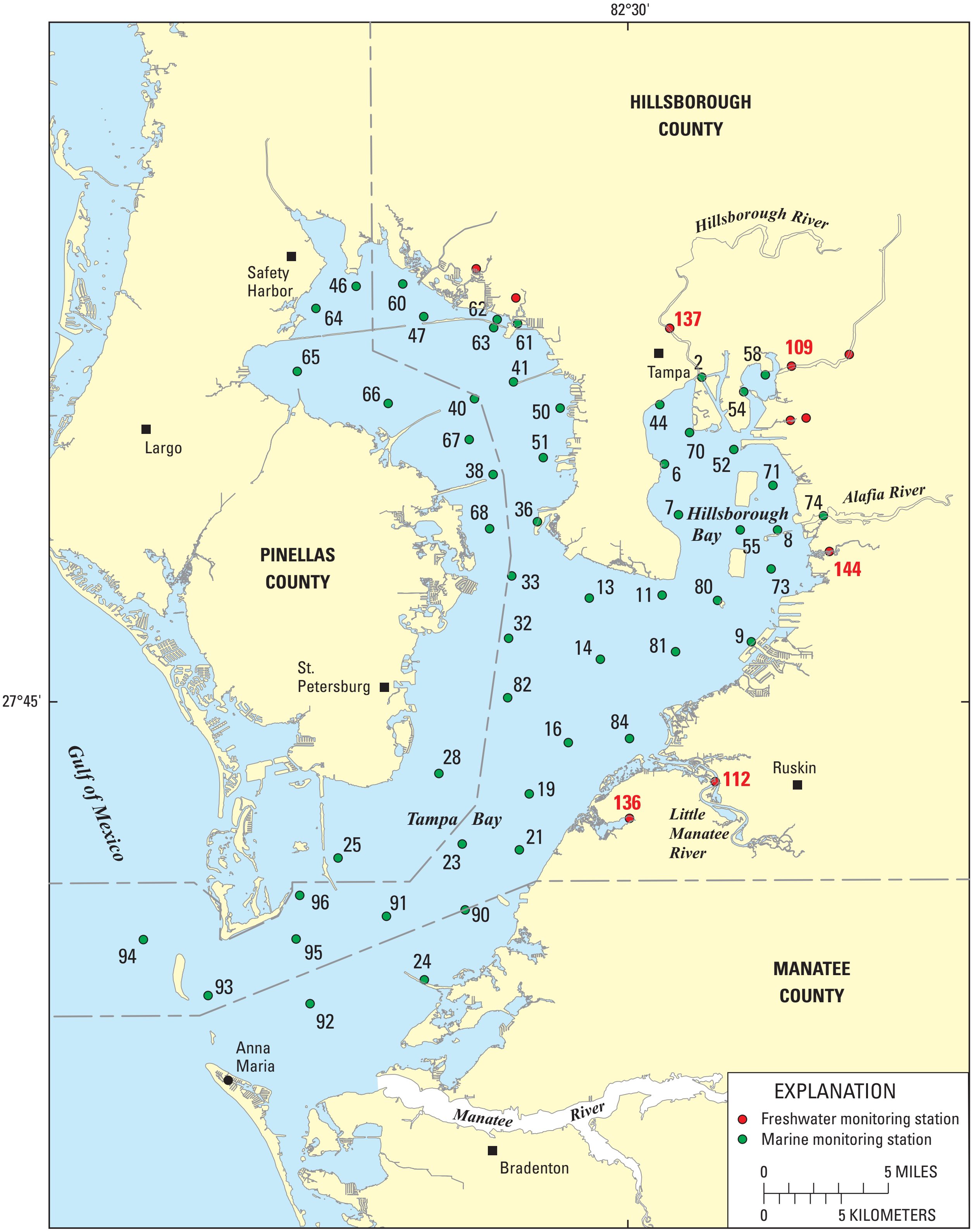


Figure 5–12. Environmental Protection Commission of Hillsborough County water-quality stations in Tampa Bay. From Environmental Protection Commission of Hillsborough County.

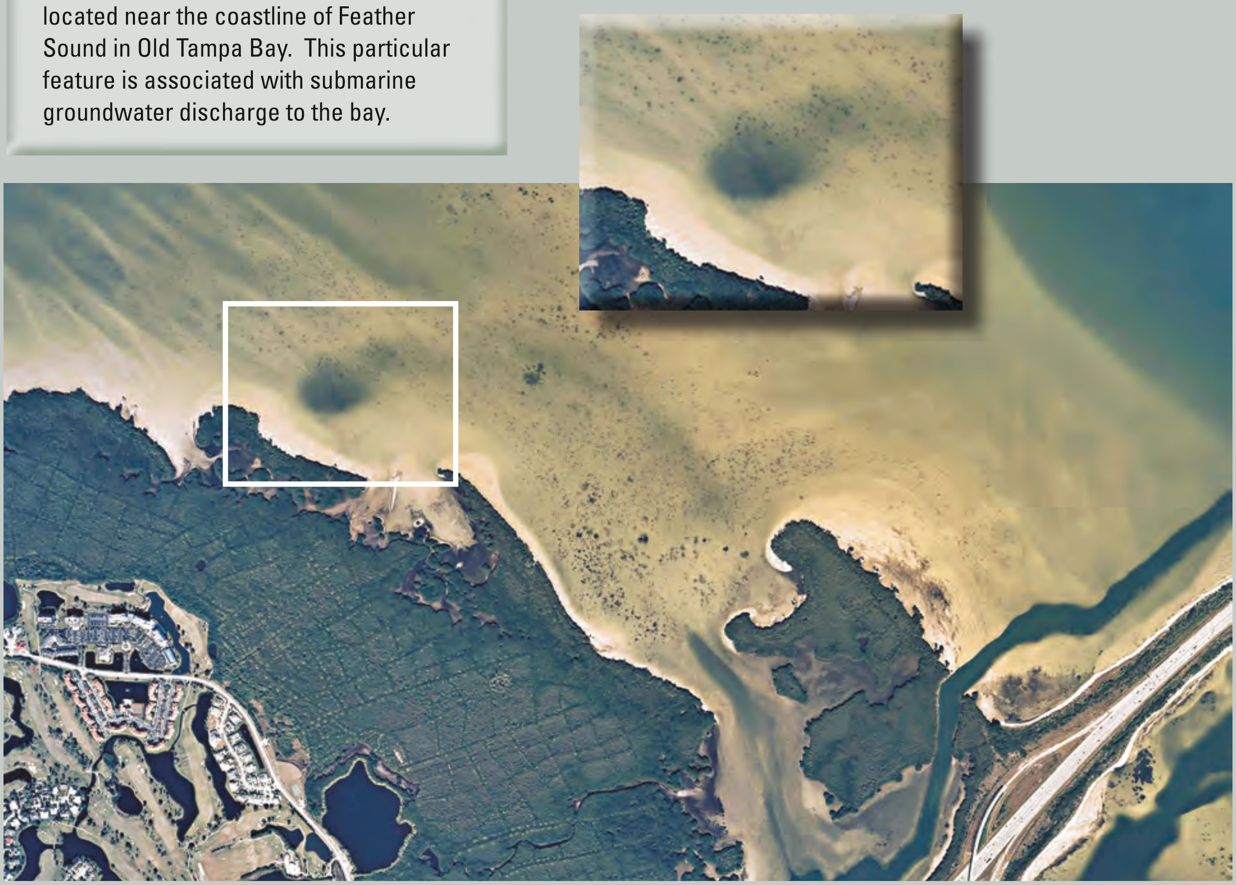
Box 5–1. Coastal Groundwater Exchange in Tampa Bay

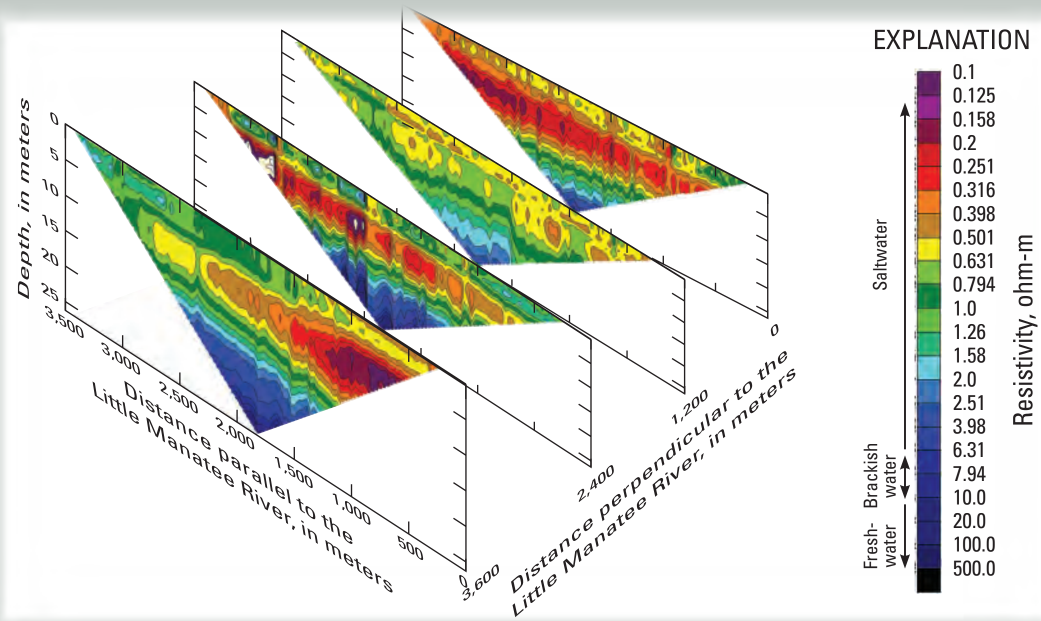
By Kimberly K. Yates (U.S. Geological Survey–St. Petersburg, Florida) and
Peter W. Swarzenski (U.S. Geological Survey–Santa Cruz, California)

Developing an accurate water budget for the Tampa Bay region is a critical component for monitoring the quantity and quality of freshwater available for human consumption, and to ensure a healthy estuarine ecosystem today. An accurate water budget is also needed to manage Tampa Bay wisely into the future under expected environmental stressors, such as sea-level change and continued urbanization. Surface-water runoff from principal rivers and creeks into the bay can be quantified using routine streamgauging techniques. However, the coastal rivers of Florida also contain an additional hydrologic component — base flow (Swarzenski and Yates, 2005). The underlying geology of the Tampa Bay area is characterized by karstic limestone topography and porous sediment that provides conduits for significant groundwater flow toward Tampa Bay (see Chapter 4). This persistent flow of coastal groundwater plays an important role in the transport of nutrients and some contaminants to the bay. The quantity and quality of this submarine groundwater discharge has until recently been overlooked in water and constituent budgets for the bay.

The USGS combined data on the structural geology of Tampa Bay with a variety of geochemical and modeling techniques to measure the quantity and quality of submarine groundwater discharge to Tampa Bay. Seismic profile data (see Chapter 4) were used to identify geologic features, such as sinkholes and collapse features that may act as conduits for submarine ground-water flow (box 5–1, fig. 1). A technique called marine continuous resistivity profiling was used to identify whether or not specific geologic features were associated

Box 5–1, Figure 1. A sinkhole feature located near the coastline of Feather Sound in Old Tampa Bay. This particular feature is associated with submarine groundwater discharge to the bay.





Box 5–1, Figure 2. Three dimensional resistivity profile taken near the Little Manatee River located on the eastern shoreline of Middle Tampa Bay. Axes indicate position of the transect relative to the river mouth. Dark blue colors indicate fresh and brackish water, green and red colors indicate saltier water. Depth (in meters) is the depth below the bay floor.

with freshwater masses located beneath the bay floor. Continuous resistivity mapping is performed by towing a series of current-producing and potential electrodes behind a boat. These sensors send an electrical pulse into the seafloor that bounces back to the sensors, indicating whether the water or sediment below the seafloor is fresh or salty. Box 5–1, Figure 2 shows a three-dimensional resistivity profile taken near the Little Manatee River in Tampa Bay, which depicts a freshened water lens that extends out into the bay and beneath the bay floor (Swarzenski and Yates, 2005; Swarzenski and others, 2007a; Kroeger and others, 2007). Salinity was measured in pore water from sediment cores (see Chapter 4) that correspond to locations with freshened water masses to confirm their presence (Swarzenski and Baskaran, 2007). Groundwater samples were also taken from about 70 locations throughout the bay area for groundwater-salinity and nutrient analyses (Kroeger and others, 2007).

Three different methods were used to quantify submarine groundwater discharge into the bay:

(1) Measurement of naturally occurring radionuclides, including radium and radon (Swarzenski and others, 2007a)

(2) Calculation of a watershed water budget (Kroeger and others, 2007); and

(3) Numerical modeling (Cliff Hearn, ETI contractor, personal comm., 2005).

Submarine groundwater discharge rates calculated from the distribution of radium-223, 224, 226, and 228 ranged from 1.6 to 10.3 m^3 d^{-1} per meter of shoreline length depending on the sampling season. Based on the watershed water-budget method, the rate of submarine groundwater discharge to the bay is estimated at 2.9 m^3 d^{-1} per meter of shoreline. Estimates of these discharge rates based on continuous radon measurements were 5.6 m^3 d^{-1} per meter of shoreline. These radon-based measurements indicate that flow of brackish and saline groundwater to the bay also represents a significant component of submarine groundwater discharge. Results indicate that the ratio of freshwater submarine groundwater discharge flux to streamflow into the bay is about 20 to 50 percent. Based on these estimated discharge rates and measurement of nutrient concentrations in groundwater samples, nutrient loads (N as TDN, DIN, or $NO^{2+}NO^{3}$ and phosphate as PO_4^{3-}) to the bay due to freshwater submarine groundwater discharge was estimated at 40 to 100 percent of the stream-discharge loads. These results indicate that the transport of groundwater and nutrients to the bay via submarine groundwater discharge is significant compared to river and stream loads.

Box 5–2. Bay Region Atmospheric Chemistry Experiment (BRACE) Study

By Holly Greening (Tampa Bay Estuary Program); Noreen Poor, (University of South Florida); and Tom Atkeson (Florida Department of Environmental Protection)

The Bay Region Atmospheric Chemistry Experiment (BRACE) study was developed in response to the persistent increasing trend in N oxide emissions in Florida. It assessed potential effects of these emissions on the air quality and ecological health of the Tampa Bay area to:

- Improve estimates of N deposition to the bay;
- Apportion atmospheric N between local, regional, and remote sources;
- Assess the impact of utility controls on N deposition; and
- Provide a technical basis for developing more effective community control strategies to reduce N deposition.

Box 5–2, Figure 1. The Bay Region Atmospheric Chemistry Experiment (BRACE) data-collection station, located at the east end of the Gandy Bridge. Photo by Noreen Poor.

In response to an initial estimate that direct deposition of atmospheric N contributed about 30 percent to the total N load to Tampa Bay, the TBEP began monitoring rainfall and ambient air concentrations of N at an urban bayside location in 1996. Flux calculations from observational data supported the initial loading estimate, and raised questions about contributions from indirect atmospheric N deposition and the sources of N to the airshed. Model predictions describe this region as centered over peninsular Florida, roughly elliptical, and roughly three times the size of the bay region (see fig. 5–3).

The BRACE study began in 2000 and included both long-term and short-term intensive measurement campaigns, as well as concurrent special studies. BRACE planners sought experimental designs that balanced project resources between measurements that would support mesoscale modeling, offered direct evidence of source contributions and N deposition rates, took advantage of new technologies, and explored new theoretical constructs. BRACE participants included managers, scientists, engineers, and technicians from the Argonne National Laboratory, EPCHC, FDEP, NOAA, Pinellas County Department of Environmental Management, TBEP, Texas Tech University, USEPA, University of Maryland, University of Miami, University of Michigan, University of South Florida (USF), and URG. The project was supported by the FDEP, Tampa Electric Company, and in-kind contributions from BRACE participants.

Within the framework established by the project goals, BRACE researchers improved N deposition estimates by expanding the air pollutant monitoring network (box 5–2, figs. 1 and 2), by deploying state-of-the-art sensors and monitors, and by analyzing and interpreting meteorological and air pollutant concentration data with sophisticated atmospheric chemistry and physics models. Coupled with the meteorological and emissions data, BRACE measurements enabled researchers to reconstruct a four-dimensional image of N emissions, dispersion, transport, and transformation; to analyze the role in N processing and transport of the land-sea breeze and regional wind convergence zones; to identify deficiencies in N emissions inventories; and to calculate total N deposition rates over the Tampa Bay watershed, including the direct total N deposition rate to Tampa Bay. The N species of interest were NO, NO_2, HNO_3, HNO_2, NO_z (that is, NO_y-NO_x), NH_3, NH_4^+, and organic amines. NO, NO_2, HNO_3, HNO_2, PAN and other organic nitrates, $NO_3^\bullet$, and N_2O_5 comprise NO_y.

The pollutants of interest, the models, and the modeling objectives dictated the temporal and spatial scales of the observations. Measurements on shorter time scales, for example, allowed better resolution of regional air pollution plumes and improved agreement with equilibrium and kinetic assumptions inherent in many model algorithms. New technologies made possible near real-time monitoring of solar radiation, actinic flux, wind speed and direction, temperature, relative and specific humidity, and concentrations of nitrogen oxide and nitrogen dioxide, nitric acid, total oxidized nitrogen species (NO_y), nitrate, ammonia, ozone, carbon monoxide, sulfur dioxide, mercury, organic carbon (OC), black carbon (EC), volatile organic compounds (VOCs), metals, and aerosol mass and number.

The measurements provided a better understanding of:

- The magnitude and composition of gaseous and aerosol N species;
- Nitrogen-deposition velocities and fluxes, both to the watershed and directly to the bay surface;
- Source emissions and the contributions of those emissions to regional air quality; and
- The limitations on instrument and model performance. Results from the BRACE study were summarized by Atkeson and others (2007).

Box 5–2, Figure 2. Meteorological data (wind speed and direction, air temperature) and physical data from Tampa Bay (current speed and direction, water temperature) are collected at several stations within Tampa Bay. Photo by Mark Luther.

received during the mid to late 1970s, and to track the improvements in bay water quality that have occurred since that time. It has also played a critical role in the development of the water-quality targets that are being used to guide the seagrass restoration program, as discussed in Chapter 4. Similarly, long-term monitoring data from bay tributaries have provided important information on water-quality patterns and trends in those river systems and the pollutant loads they deliver to the bay.

Estimating Pollutant Loads and Bay Responses

In addition to monitoring data, managers also rely on several other types of information to guide their efforts to protect and restore water quality. Estimates of the quantities of different pollutants being discharged to the bay and its tributaries are developed and updated every few years by the TBEP, and likely magnitudes of key pollutant discharges and the locations where they enter the bay are determined (for example, Pribble and others, 2001; Poe and others, 2005; Janicki Environmental, 2008). Computer models (for example, Janicki and Wade, 1996; Morrison and others, 1997; Wang and others, 1999) have been used to summarize managers' current knowledge, assumptions, and hypotheses regarding bay responses to those pollutants and to predict how water-quality conditions will respond to changes in pollutant loadings.

The methods used to estimate pollutant loadings are technically challenging, and the results contain considerable uncertainty that must be taken into account when considering potential management actions. Estimation methods have improved over time, thanks to technological refinements that increase the sensitivity and accuracy of monitoring instruments, and due to improvements in managers' understanding of the transport and fate of different types of pollutants in the aquatic environment. However, certain categories of sources, such as dry atmospheric deposition and submarine groundwater discharge, remain difficult to measure and are continuing sources of uncertainty. Recent USGS research (Kroeger and others, 2007; Swarzenski and others, 2007a; box 5–1) has made important advances in the estimation of submarine groundwater discharges to the bay. Similarly, the Bay Region Atmospheric Chemistry Experiment (BRACE) study, a collaborative multiagency research effort (Atkeson and others, 2007; box 5–2), along with a recent NOAA atmospheric modeling study (Dennis and Arnold, 2007), have provided updated estimates of dry and wet atmospheric deposition of N and other water-quality constituents to the bay and its watershed.

The ability of computer models to simulate and forecast water-quality responses to changing pollutant loadings also tends to improve over time, as advances in computer technology encourage the development and use of more detailed, realistic, and computationally demanding algorithms. The ongoing management effort should also encourage improvements in the realism of the water-quality models that are applied to the bay, as managers and modelers go through an iterative process of developing and testing model predictions against the water-quality changes that are observed over time.

Adaptive Management

The Tampa Bay water-quality management program is a collaborative, flexible, multidisciplinary effort that has evolved in response to changes in technology, data availability, and scientific understanding. In order to address the inherent uncertainties and complexities of bay responses to changing pollutant loads and other environmental conditions, the program has adopted an "adaptive management" (Holling, 1978; Lee, 1993) approach. Although adaptive management has been defined in a number of ways, its goal is to improve managers' understanding of — and ability to achieve — a set of well-defined ecological objectives, using a combination of carefully designed management interventions and evaluations of monitoring data. Environmental responses to the interventions are monitored, and the resulting information is used to update and refine management actions (for example, NRC, 2000; Greening and Elfring, 2002; Gregory and others, 2006). The adaptive nutrient management strategy used in Tampa Bay (fig. 5–13) is based on this approach, incorporating periodic evaluations of water-quality and seagrass-management goals as well as annual evaluations of water-quality monitoring data and the redirection of management actions on an as-needed basis.

Current and Anticipated Water-Quality Management Issues

Monitoring and improving water quality remain top priorities for resource managers in Tampa Bay, particularly in the face of increased stress to water resources as the bay area population continues to grow. Nutrient input and cycling, eutrophication, and its effects on phytoplankton productivity and seagrass growth are critical issues that scientists and resource managers are addressing in the context of current environmental conditions and the need for continued management of water quality in the future.

Nutrient Inputs and Eutrophication

At the watershed scale, management of estuarine eutrophication generally focuses on the control of both N and P inputs (Cloern, 2001; Howarth and others, 2002; Paerl, 2008, 2009). In the temperate-zone estuary systems where most eutrophication studies have been carried out, P is usually the nutrient of greatest concern in the freshwater tributaries and tidal fresh zones (areas where salinities are typically less than 0.5 ppt), because excessive P loads to these areas often stimulate blooms of undesirable phytoplankton (Likens, 1972; Schindler, 1975; Reynolds, 2006), particularly cyanobacteria (Chapman and Schelske, 1997; Chorus and Bartram, 1999; Paerl, 2008). In contrast, N is generally the nutrient of greatest concern in the brackish and marine areas, where salinities typically vary from 0.5 to about 35 ppt. This is because N is the nutrient that more frequently limits phytoplankton productivity in these higher-salinity nearshore locations (NRC, 2000; Cloern, 2001). Although this generalization is not true for all Florida estuaries or coastal waters — several of which are known to be P limited (Myers and Iverson, 1981; Fourqurean and others, 1992) or co-limited by N and P (Mortazavi and others, 2000) — it does appear to be true of Tampa Bay (Johansson, 1991; Vargo and others, 1991; Wang and others, 1999; Johansson, 2005).

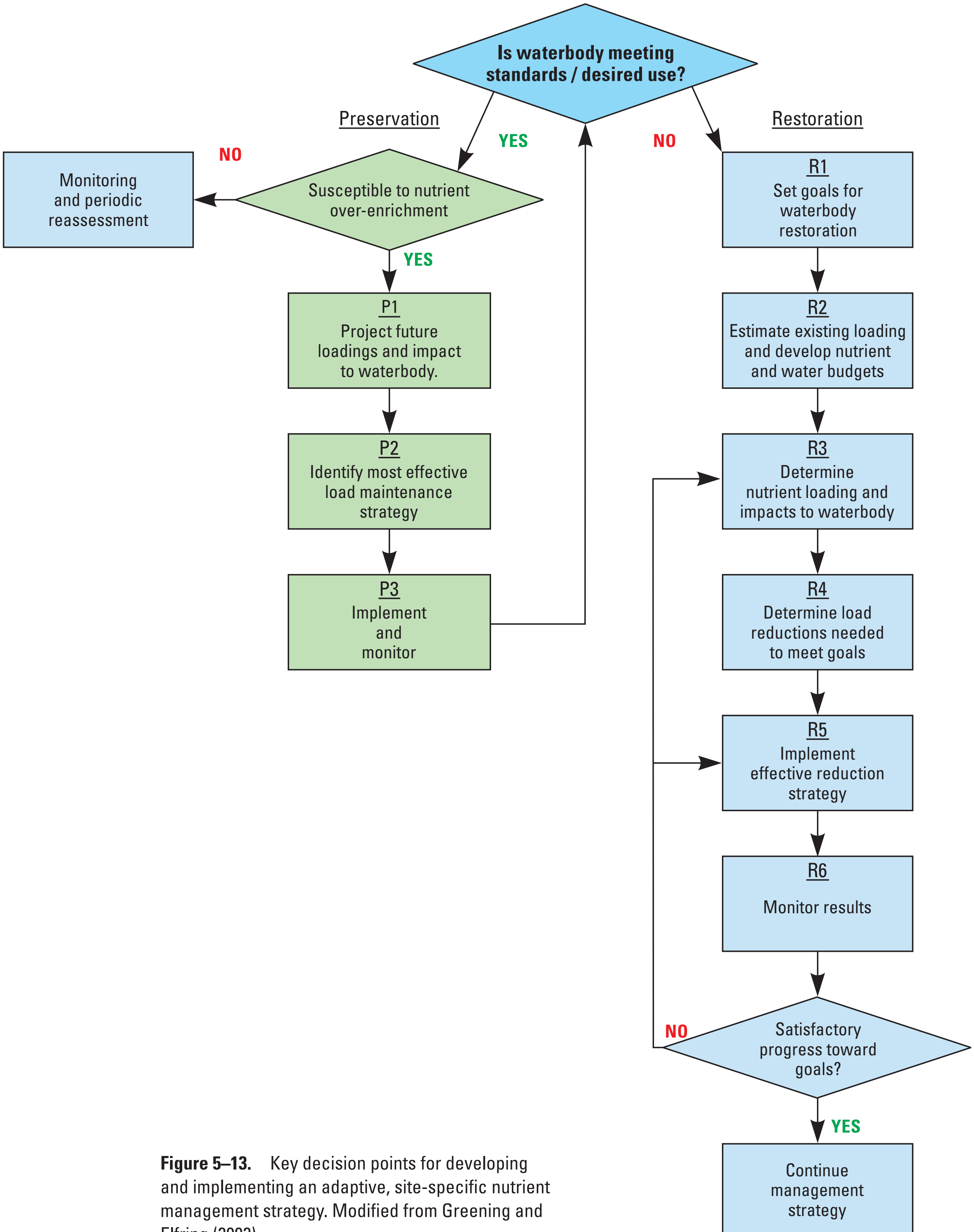


Figure 5–13. Key decision points for developing and implementing an adaptive, site-specific nutrient management strategy. Modified from Greening and Elfring (2002).

Because of the geological characteristics of its watershed, Tampa Bay is somewhat unusual with respect to the P loadings it receives. Each of the four largest rivers that discharge to the bay — the Hillsborough, Alafia, Little Manatee, and Manatee Rivers — drains a part of the central Florida phosphate district, which is located in parts of Hillsborough, Manatee, and Polk Counties (fig. 5–14). The phosphate district also includes the areas of Manatee, Polk, and DeSoto Counties that lie in the adjacent Charlotte Harbor watershed. As its name implies, the district contains large deposits of a phosphate-rich geological matrix — the Bone Valley Member of the Peace River Formation (Spechler and Kroening, 2007) — which is a mix of clay, quartz sand, dolomite, and phosphate ore that is mined and processed to produce commercial fertilizer and livestock feed-supplement products (McClellan and Eades, 1997).

The central Florida phosphate district, along with a smaller mining area located in northern Florida, produces the largest annual tonnage of phosphate ore (>30 million tons in 1990) of any U.S. State and accounts for about 30 percent of total world production (McClellan and Eades, 1997). The rivers and streams that drain the phosphate district contain unusually high concentrations of P, in comparison to surface waters in other parts of Florida and the United States, due to a combination of natural leaching and discharges from phosphate mining and processing operations (Odum, 1953).

Because of the very large P loads that Tampa Bay receives from its watershed, concentrations of soluble-reactive P — the water-quality indicator that is commonly used to estimate levels of the inorganic form of P that is directly taken up by phytoplankton — usually exceed phytoplankton requirements in most parts of the bay and in the tidal reaches of its major tributaries (Johansson, 1991; Vargo and others, 1991; Wang and others, 1999). The Charlotte Harbor estuary and its Peace River Basin, which also drains a part of the mining district, exhibit similar elevated soluble-reactive P levels (McPherson and Miller, 1994).

In the freshwater tributaries of the Tampa Bay and Charlotte Harbor watersheds, these elevated soluble-reactive P levels contribute to a number of environmental and economic impacts. Currently, more than 80 freshwater lakes and stream segments in the Tampa Bay watershed, and more than 50 lakes and stream segments in the Peace River watershed of Charlotte Harbor, are classified as water-quality impaired by FDEP and USEPA due to elevated concentrations of nutrients and chlorophyll, extreme diurnal fluctuations in DO concentrations, and other symptoms of excessive eutrophication (FDEP, 2003, 2005, 2006, 2009). These impairments affect the quality of life of area residents and impact the environmental services provided by the impaired waterbodies. Development of TMDLs for the impaired waterbodies, and implementation of water-quality improvement and management programs to bring them into compliance with State and Federal standards, are costly processes. The tidal freshwater reaches of several rivers in the region, which are similar to freshwater lakes in some of their hydrologic and water-quality characteristics, also show evidence of excessive eutrophication and may require the development and implementation of nutrient TMDLs.

At a more conjectural level, recent research suggests that in some years the P and colored dissolved organic material discharged from the Tampa Bay and Charlotte Harbor watersheds may contribute to the development of blooms of N-fixing cyanobacteria (*Trichodesmium* spp.) in nearshore waters of the Gulf of Mexico (Walsh and others, 2003, 2006). Under certain

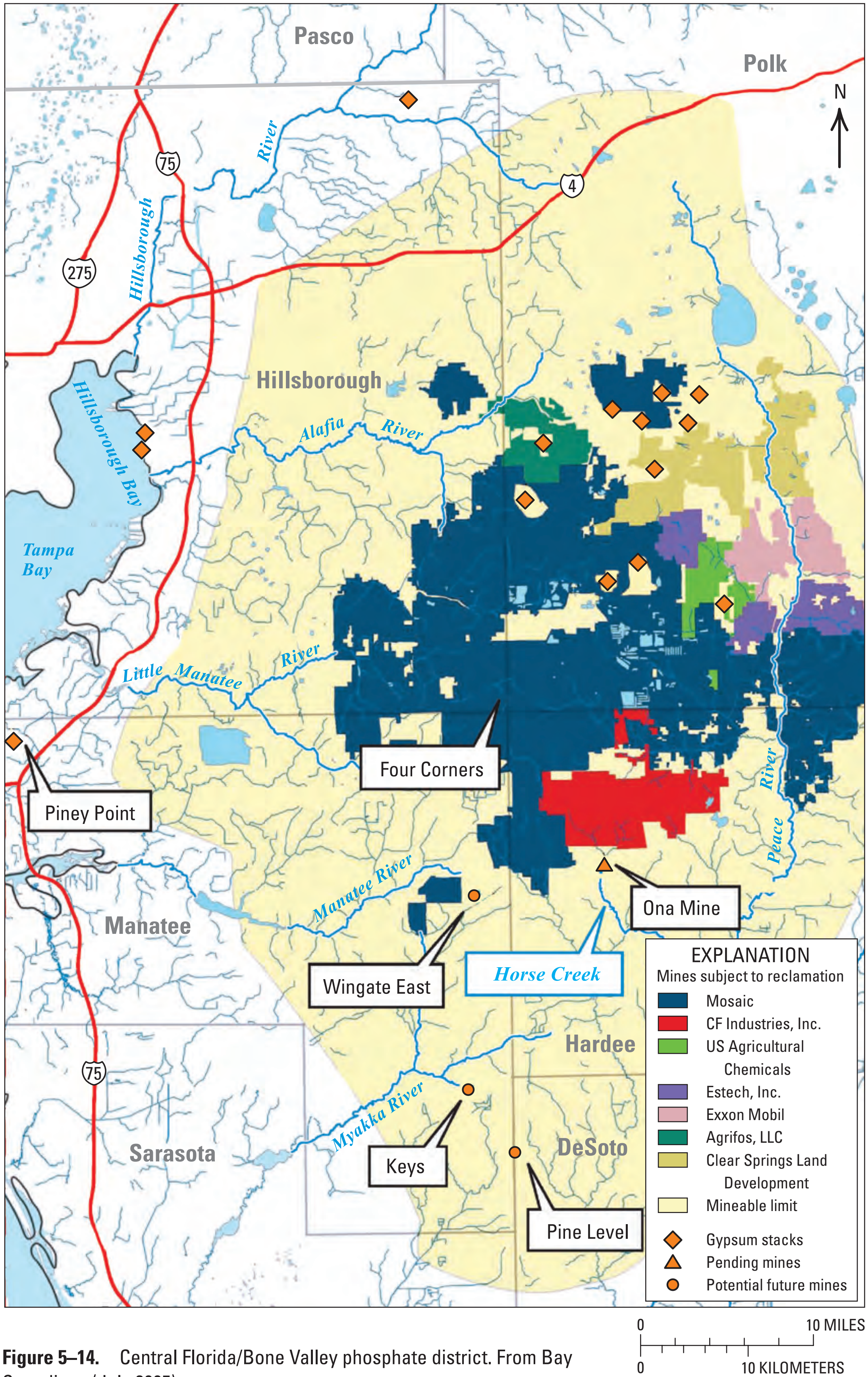


Figure 5–14. Central Florida/Bone Valley phosphate district. From Bay Soundings (July 2005).

conditions the *Trichodesmium* blooms may, in turn, provide nutrients that help support blooms of red tide (*Karenia brevis*) in the nearshore area located between the mouths of Tampa Bay and Charlotte Harbor (Walsh and others, 2001, 2003, 2006). As noted below, however, relationships between land-based nutrient discharges and red tide blooms are not well understood and are an area of active research and debate (summarized by Alcock, 2007).

Factors Affecting Phytoplankton Productivity in Tampa Bay

Within Tampa Bay itself, phytoplankton productivity is influenced by a number of environmental factors. Bioassay experiments conducted in Hillsborough Bay (Johansson, 1991, 2005) indicate that P is not a limiting nutrient in that part of the bay and that the availability of N is more likely to limit phytoplankton growth there. Short-term bioassays conducted by Vargo and others (1991) in the Little Manatee and Alafia rivers and nearby parts of Tampa Bay produced varying responses to N and P additions. The results indicated that, at times, inorganic N and P were both present at elevated (nonlimiting) concentrations in those areas. Vargo and others (1991) concluded that phytoplankton populations in Tampa Bay can be considered "nutrient sufficient to borderline N limited" for short-term photosynthesis requirements.

Using a mechanistic water-quality model, Wang and others (1999) concluded that phytoplankton productivity in the bay during the period 1985 through 1994 was limited primarily by the availability of sunlight (due to light attenuation by phytoplankton, other turbidity sources, and water color) and secondarily by the availability of inorganic N. Wang and others (1999) found that, on average, model-based phytoplankton growth rates were reduced to levels 60 to 80 percent below their potential maximum values because of limited light availability in the four major segments of Tampa Bay, whereas limited N availability caused a smaller reduction (10 to 40 percent below potential maximum growth rates) and P availability caused essentially no reductions. The model predicted that light limitation was more pronounced in the upper bay segments (Hillsborough Bay and Old Tampa Bay) than in Lower Tampa Bay.

Although N appears to be less limiting than light, the model-based simulations predicted that increases in N loadings from external sources would cause chlorophyll *a* concentrations to increase, whereas load reductions would have the opposite effect (Wang and others, 1999). Cases of limitation by physical factors, such as light or temperature, as well as by nutrient availability, apparently occur in a number of aquatic systems (Falkowski and Raven, 2007). Changes in external N loadings to Hillsborough Bay and Old Tampa Bay were predicted to have cascading effects throughout the estuary, due to down-bay transport of nutrients and phytoplankton from those areas to other bay segments (Wang and others, 1999). Similar results were found in TBEP-sponsored studies using empirical (statistically based) modeling approaches (Janicki and Wade, 1996; Greening and Janicki, 2006). The corroborative findings of the mechanistic and empirical models increased managers' confidence in the guidance provided by the two approaches.

Among the many factors that affect phytoplankton productivity in the open waters of the bay, anthropogenic N loading appears to be the primary one that can be effectively controlled by management activities carried out in the watershed (Johansson, 1991; Wang and others, 1999; Greening and

Janicki, 2006; TBEP, 2006). As a result, a principal focus of the Tampa Bay eutrophication management effort has been to cap the annual N loads that enter the bay at levels that appear necessary to achieve the bay-wide water clarity targets and seagrass restoration goals summarized in Chapter 4.

External Nitrogen Sources and Estimated Annual Loadings

The N that is present in an estuary at any given time consists of two components: "new" or exogenous N that has recently been discharged from the watershed or deposited from the airshed, and "recycled" or endogenous N that has been present in the system for some time and has already been cycled through one or more portions of the estuarine food web (Nixon, 1981; Paerl, 1997; NRC, 2000; Seitzinger and others, 2002). For Tampa Bay, the TBEP and its partners have conducted a series of projects to estimate the sources, magnitudes, and pathways of both types of N and their effects on bay water quality.

The assumptions and methods used to estimate annual loads of new N to the estuary have been summarized in a number of technical reports and publications (Pribble and others, 2001; Poe and others, 2005; Greening and Janicki, 2006; Janicki Environmental, 2008):

- The sources of N loads to Tampa Bay are varied (for example, fig. 5–15) and include stormwater runoff, atmospheric deposition, groundwater/springs, and fertilizer losses from port facilities, as well as discharges from traditional point sources such as municipal sewage-treatment plants and industrial facilities;
- Freshwater inputs to the bay via rainfall are estimated using data from a number of National Weather Service and other rainfall-monitoring sites in the watershed. Monthly rainfall records are used to develop estimates of direct wet deposition of N to the bay surface and to estimate N loads in runoff from ungaged parts of the watershed;

Figure 5–15. Nutrient pollution sources include emissions from transportation and point sources, such as stormwater pipes.

- About 57 percent of the watershed is gaged for both flow and water quality, allowing direct estimates of loads (Greening and Janicki, 2006). For the remaining, ungaged areas, loads from stormwater runoff are estimated using statistical (regression-based) methods based on rainfall, land use, soils, and seasonal land-use-specific water-quality concentrations;
- To calculate load estimates for point sources, values for all facilities with direct discharges to surface waters and all land application discharges with an annual average daily flow of $\geq$0.1 Mgal/d are calculated from records that are reported by the facilities to State and Federal regulatory agencies;
- Wet atmospheric deposition of N directly to open waters of Tampa Bay is calculated by multiplying the volume of precipitation onto the bay by N concentration in rainfall. Dry deposition is estimated using a seasonal dry-to-wet deposition ratio derived from 5 years of concurrent wet and dry deposition measurements (Poor and others, 2001);
- Groundwater flows are estimated for each bay segment. Only groundwater inflow that enters the bay directly from the shoreline or bay bottom is considered. Groundwater and septic tank leachate inflows to streams are already accounted for through measured or modeled surface-water flow as nonpoint source loading and, therefore, are not included in groundwater loading estimates. Wet- and dry-season groundwater flow estimates are calculated using a flow net analysis and Darcy's equation, following the methods of Brooks and others (1993). Total N concentration data for surficial, intermediate, and Floridan aquifers are obtained from the SWFWMD ambient groundwater monitoring program.

Estimates of average annual N loads generated using these methods, for a number of different time periods extending from the late 1930s to 2003, are shown in fig. 5–16. The estimated values illustrate the large increase in overall N loads, and in N contributions from point sources, that occurred between the late 1930s and the late 1970s, as well as the >90 percent reduction in annual point source N loads that occurred in the early 1980s. The values also show the estimated increase and subsequent reduction in N loads due to fertilizer product losses at commercial shipping facilities and the increases and subsequent reductions in N loads due to atmospheric deposition that have occurred in recent decades. As a result, although estimated N loads in the late 1970s were dominated by point source discharges, stormwater runoff and atmospheric deposition have been the largest estimated source categories in more recent periods (fig. 5–16). The atmospheric deposition source category depicted in fig. 5–16 estimates only the amount of N that is deposited directly from the atmosphere to the bay surface. From a broader perspective, atmospheric deposition may contribute as much as 50 to 67 percent of the "new" N that is discharged to the bay each year, because much of the N that is measured in stormwater runoff appears to originate as atmospheric deposition on the watershed (TBEP, 2006; Dennis and Arnold, 2007). Recent biogeochemical studies by USGS researchers (Kroeger and others, 2007; Swarzenski and others, 2007b) also suggest that submarine groundwater discharges may be a much larger source of N inputs than was suspected previously, perhaps as large as 40 to 100 percent of the loads carried by rivers and streams (Kroeger and others, 2007; Swarzenski and others, 2007b; box 5–1).

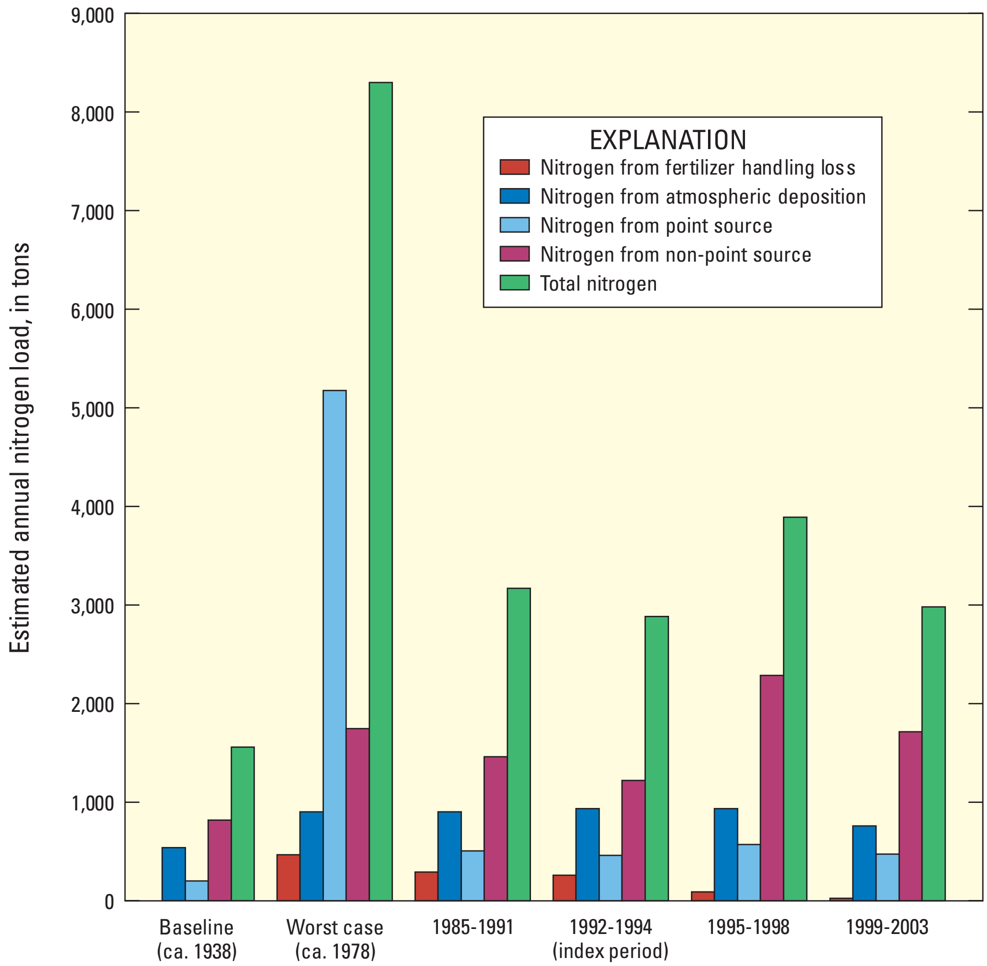


Figure 5–16. Estimated annual nitrogen loads to Tampa Bay during various time periods (1938–2003). Data from Janicki Environmental, Inc. (2008) and Greening and Janicki (2006).

Internal Nutrient Cycling and its Implications for Bay Management

Using a mechanistic water-quality model, Wang and others (1999) simulated annual rates of N and P transport and recycling in the four major segments of the bay for the period 1985 through 1994. For both nutrients the model indicated that, on an annual basis, the magnitude of internal cycling equaled or exceeded external loads during the 10-year simulation period. In the case of P, phytoplankton and sediment release (combined) were estimated to be comparable to annual external loads (Wang and others, 1999). In the case of N, the estimated annual average loss of total N from the bay via advective and dispersive transport (-17,000 tons/yr) was about 4.5 times larger than the estimated annual average load (+3,600 tons/yr) from the watershed. When all the modeled source and loss terms were summed, the result was slightly negative (-1,540 tons/yr), indicating that the total mass of N in the bay decreased over the 1985–1994 period.

For inorganic forms of N, benthic and microbial processes that transform organic N to inorganic N and release it to the water column represented the major source (+65,000 tons/yr). Microbial mineralization of organic N to ammonia N and benthic release of ammonia N contributed the highest estimated annual flux rates. Estimated fluxes due to denitrification were smaller than those due to nitrification by an order of magnitude. This resulted in large predicted ratios of ammonia N to nitrate N (ratios on the order of 4 to 10), which were consistent with available field data (Wang and others, 1999). Recently, Carlson and Yarbro (2006) also measured large ammonia fluxes from sediments in several bay segments. Phytoplankton uptake was the major loss term for inorganic N from the water column (-66,500 tons/yr), whereas advective and dispersive transport were predicted to cause relatively minor losses (-2,500 tons/yr).

Taken together, the available modeling results and bioassay data indicate that relationships between N loading rates and chlorophyll *a* concentrations in the bay are relatively complex (Morrison and others, 1997; Wang and others, 1999). Only a small amount of the phytoplankton standing stock observed at any given time appears to be supported by loadings of "new" N recently delivered from the watershed. A much larger amount is apparently supported by the regeneration of inorganic N within the bay, through microbial and benthic pathways, from pools of recycled organic N. The modeled flux rates and bioassay results also imply that short-term changes in external N loads may not produce immediately detectable changes in chlorophyll *a* concentrations, particularly if the changes are small relative to the size of the internal N pools and occur following a prolonged period of elevated loadings. An example of this situation likely occurred in the bay during the early 1980s. Although the City of Tampa's municipal wastewater-treatment plant at Hookers Point was upgraded to advanced wastewater-treatment standards in 1979, resulting in a marked reduction in annual loadings of "new" N to Hillsborough Bay (Johansson, 1991), reductions in chlorophyll *a* concentrations were not observed until 1983 in Hillsborough Bay and 1984 in other bay segments. As noted by Johansson (1991), the observed lag between the load reductions and chlorophyll *a* response appeared to represent the time period necessary for internal processes within the bay to equilibrate to the new level of N loading following several decades of more elevated anthropogenic loads.

Setting Water-Quality Goals and Nitrogen Loading Goals Based on the Light Requirements of Seagrasses

As noted in Chapter 4, water clarity is a critically important water-quality indicator because it affects the amount of sunlight that penetrates the water column, which in turn affects the growth of aquatic plants, such as phytoplankton, macroalgae and seagrasses (Gallegos, 1994, 2001, 2005; Gallegos and Kenworthy, 1996; Kenworthy and Fonseca, 1996). PAR, which is light in the range of wavelengths from 400 to 700 nanometers (nm), provides the predominant source of energy for these autotrophic organisms. Light striking the water surface is reflected, absorbed, and refracted by suspended particles and dissolved substances in the water column, and by the water itself. As a result, sunlight that penetrates the water surface and enters the top of the water column (incident light, denoted I_O) becomes reduced or attenuated as it travels downward. The amount of light (I_Z) present at any depth z can be described as a function of I_O and z by using the Beer-Lambert exponential decay function:

$$I_Z = I_O\, e^{-kz}$$

where k is the "diffuse downwelling irradiance attenuation coefficient," a measure of water clarity (Day and others, 1989; Gallegos, 1994).

A traditional method for measuring water clarity in the field is through the use of a Secchi disk, a circular disk that is lowered into the water column until it just disappears from sight. The depth at which it disappears is the Secchi depth (Z_S). Although they provide only an approximation (Scheffer, 2004), Secchi depth readings can be used to estimate the attenuation coefficient *k* using an equation of the form:

$$k = c/Z_S$$

where the coefficient *c* usually varies between 1.4 and 1.8 in estuarine waters (Day and others, 1989). Studies in the different segments of Tampa Bay, based on monitoring data provided by the EPCHC and the City of Tampa, have produced estimated *c* values ranging between 1.49 and 1.84 (Dixon, 1999).

In most waterbodies the factors affecting the value of the coefficient *k* typically include attenuation due to water, phytoplankton, colored dissolved organic material, and nonalgal particulate matter. Each of these factors plays an important light-attenuating role in Tampa Bay, although phytoplankton abundance (estimated using measured chlorophyll *a* concentrations) has proven to be the best bay-wide predictor of water clarity and the factor most amenable to watershed management actions (Janicki and Wade, 1996; Greening and Janicki, 2006).

Phytoplankton and macroalgae are adapted to relatively low light levels and are typically able to maintain a positive energy balance (photosynthesis exceeding respiration) at I_Z values as low as 1 to 5 percent of I_O (Day and others, 1989). Seagrasses require substantially more light than phytoplankton and macroalgae, however, with most species requiring on the order of 13 to 37 percent or more of I_O (Dennison and others, 1993; Duarte and others, 2007).

Because of the importance of seagrasses as a biological resource in Tampa Bay, the TBEP and its partners have adopted numerical targets for water clarity levels (expressed as annual mean Secchi depth), chlorophyll *a* concentrations, and N loading rates to help meet seagrass acreage restoration goals that have also been adopted for the bay. As noted in Chapter 4, development and implementation of these targets has followed a multistep process (Greening, 2001; TBEP, 2006):

Step 1: Set specific, quantitative seagrass acreage goals for each bay segment.

In 1996, the local management community adopted a minimum seagrass coverage goal of 38,000 acres, which represents 95 percent of the acreage that was estimated to have been present in the bay in the early 1950s (after subtracting areas that have been rendered nonrestorable by subsequent dredging, filling, and the construction of causeways and other infrastructure; Wade and Janicki, 1993). The early 1950s time period was selected as the baseline for seagrass coverage because it preceded the rapid population increases that have occurred in the watershed in more recent decades, and because aerial photographs from that time period were available for the entire Tampa Bay shoreline and adjacent shallow water (Greening and Janicki, 2006).

Step 2: Determine the light requirements of the target seagrass species (*Thalassia testudinum*) in Tampa Bay.

Field studies carried out in stable *Thalassia* meadows in Lower Tampa Bay indicated that the deep edges of those seagrass beds corresponded to the depth at which 20.5 percent of I_O (the light that penetrates the water surface) reached the bottom on an annual average basis (Dixon, 1999).

Step 3: Determine the water clarity levels necessary to provide adequate light to meet the seagrass acreage goals.

Based on the 20.5 percent light requirement estimated in Step 2, the seagrass acreage restoration goal was restated as a light penetration and water clarity target: to restore seagrass acreage to early 1950s levels in a given bay segment, water clarity in that segment should be restored to a point that allows at least 20.5 percent of I_O to reach the same depths that were reached in the early 1950s. Those depths range from about 1 m in Hillsborough Bay to about 2 m in Lower Tampa Bay (Janicki and Wade, 1996).

Step 4: Determine maximum chlorophyll *a* concentrations that allow water clarity to be maintained at appropriate levels.

Water clarity and light penetration in Tampa Bay are affected by a number of factors, including phytoplankton density (estimated using measured chlorophyll *a* concentrations), colored dissolved organic material (estimated using water color measurements), and nonphytoplankton turbidity. Janicki and Wade (1996) used regression analyses applied to long-term EPCHC monitoring data to develop an empirical model describing water clarity variations in response to these factors in the four largest bay segments. The model that provided the best fit (highest r^2) to the observed water-clarity data took the form:

$$ln\ C_{t,s} = \alpha_{t,s} + \beta_{t,s} * ln\ (Z_{t,s})$$

where $Z_{t,s}$ is the depth to which 20.5 percent of surface irradiance penetrates in month t and bay segment s, $C_{t,s}$ is the average chlorophyll *a* concentration in month t and bay segment s, and $\alpha_{t,s}$ and $\beta_{t,s}$ are regression parameters (Janicki and Wade, 1996; Greening and Janicki, 2006).

Least-squares methods were used to estimate the regression parameters (Janicki and Wade, 1996). Results of the regressions indicated that variation in observed depths to which 20.5 percent of surface irradiance penetrates could be explained by variation in observed chlorophyll *a* concentrations. Monthly specific regression intercept terms were used to avoid any potentially confounding effects of seasonality in independent and dependent variables. The model fit was relatively good ($r^2 = 0.67$). As noted, turbidity and water color were also investigated as possible explanations for the unexplained variation in light penetration; however, no improvement in the model fit was found (Janicki and Wade, 1996; Greening and Janicki, 2006).

The adopted segment-specific annual average chlorophyll *a* targets (ranging from 4.6 to 13.2 µg/L in the four largest bay segments; fig. 5–11) are easily measured and tracked through time and are used as intermediate measures for assessing success in maintaining water-quality requirements necessary to meet the long-term seagrass coverage goal (Greening, 2001; Greening and Janicki, 2006).

Step 5: Determine maximum nutrient loadings that allow chlorophyll *a* concentration targets to be achieved.

A multipronged, weight-of-evidence approach was used to examine relations between annual mean chlorophyll *a* concentrations and external N loads. As noted, monitoring data (Johansson, 1991, 2005) and nutrient bioassays (Vargo and others, 1991) had indicated that N limitation is a factor affecting chlorophyll *a* dynamics in the bay. Empirical observations, such as the dramatic decline in chlorophyll *a* concentrations (fig. 5–11) and improvements in water clarity (fig. 5–10) that followed the large reductions in external N loads that occurred in the early 1980s, indicated it was a meaningful factor.

In an effort to quantify the relationship, mechanistic (Wang and others, 1999) and statistical (Janicki and Wade, 1996) models were developed using external loading estimates provided by the TBEP (Pribble and others, 2001). The researchers examined observed and model-predicted changes in annual average chlorophyll *a* concentrations in response to changing annual N loads during 1986–1994. Across the four major bay segments, the two models predicted similar relations between annual average chlorophyll *a* concentration (expressed in micrograms per liter) and annual estimated N load (expressed in pounds) (Morrison and others, 1997):

Mechanistic model:

$$[\text{Chlorophyll } a] = -61.74 + 5.32 * \log_{10} (\text{estimated N load}); \; r^2 = 0.61$$

Statistical model:

$$[\text{Chlorophyll } a] = -51.67 + 4.49 * \log_{10} (\text{estimated N load}); \; r^2 = 0.66$$

In both cases, the model-estimated N loads to each of the bay segments incorporated estimates of the loads transported from other segments, as well as estimates of loads discharged directly to a segment from the watershed and airshed, to provide adequate simulations of chlorophyll *a* dynamics across the bay as a whole. However, neither model had the capability to predict future changes in internal fluxes of “recycled” N that might occur in response to long-term changes in external loads (Morrison and others, 1997). Therefore, an alternative empirical approach was used to develop N loading targets for the bay (Janicki and Wade, 2006).

Examination of the EPCHC water-quality monitoring data indicated that the water clarity conditions that existed during 1992–1994 allowed an annual average of more than 20.5 percent of subsurface irradiance (I_o) to reach target depths (the estimated depths to which seagrasses grew in the early 1950s) in three of the four largest bay segments (Janicki and Wade, 1996). A N load management strategy based on “holding the line” at the annual loading levels estimated to have occurred during 1992–1994 was, therefore, adopted by the TBEP and its partners (Greening and Janicki, 2006).

Step 6: Implement the nitrogen (N) management strategy and assess its effectiveness.

For consistency with the adaptive management approach, the effectiveness of the adopted N management strategy is assessed annually, by evaluating chlorophyll *a* concentrations and water clarity levels measured by the local monitoring programs in each bay segment during the previous calendar year, and comparing those values to the segment-specific targets (Greening and Janicki, 2006). A decision matrix approach (Janicki and others, 2000; described in box 5–3) is used to determine the level of management response that is appropriate in years when water-quality targets are not met. Changes in seagrass coverage are assessed and reported every 2 years by the SWFWMD.

Box 5–3. Tracking Progress Toward Water-Quality Goals—Application of the Tampa Bay Decision Framework

By Edward Sherwood (Tampa Bay Estuary Program) and Holly Greening (Tampa Bay Estuary Program)

The continued monitoring of water quality and seagrasses in Tampa Bay will allow managers to assess progress toward meeting established goals. An important component of this effort is the routine comparison of mean annual chlorophyll *a* concentrations and light attenuation to desired targets. TBEP has developed a tracking process to determine if water-quality targets are being achieved. The process to track status of chlorophyll *a* concentration and light attenuation involves two steps. The first step utilizes a decision framework to evaluate differences in mean annual ambient conditions from established targets. The second step incorporates results of the decision framework into a decision matrix, leading to possible outcomes dependent upon magnitude and duration of events in excess of the established target (Janicki and others, 2000; Greening and Janicki, 2006).

The recommended management actions resulting from the decision matrix are classified by color (green, yellow and red) into three categories for presentation to the Tampa Bay resource-management community (box 5–3, fig. 1). When outcomes for chlorophyll *a* concentration and light attenuation indicate that both targets are being met (green), no management response is required. When conditions are intermediate (yellow), with the monitoring data indicating relatively small and/or short-lived failures to meet the targets, further examination is needed to determine an appropriate management response. When conditions are problematic (red), with relatively large or longer-term exceedances of one or both targets, stronger management responses are considered for implementation.

Green	"Stay the course;" partners continue with planned projects to implement the CCMP. Data summary and reporting via the Baywide Environmental Monitoring Report and annual assessment and progress reports.
Yellow	TAC and Management Board on caution alert; review monitoring data and loading estimates; attempt to identify causes of target exceedences; TAC report to Management Board on findings and recommended responses needed.
Red	TAC, Management and Policy Boards on alert; review and report by TAC to Management Board on recommended types of responses. Management and Policy Boards take appropriate actions to get the program back on track.

Box 5–3, Figure 1. Management responses to decision matrix outcomes.

Results of the decision matrix from 1974 through 2008 are shown in box 5–3, figure 2 (Sherwood, 2009). The poor water conditions are clearly seen in early years of this time series, followed by marked improvements since 1984.

Since 1996, application of the decision framework has indicated two problematic (“red”) time periods: in 1997 and 1998 in all bay segments (corresponding to high rainfall associated with a strong El Niño event), and in 2003 and 2004 in one bay segment, Old Tampa Bay. Recommendations from the TBEP Technical Advisory Committee (TAC) for management response to the El Niño-associated period were to support immediate actions toward repair of sewer transport and pumping systems and industrial treatment-water holding systems that had failed during high rainfall periods. Actions were taken by municipalities and industrial facilities to address these failed systems. In addition to these immediate actions, the TAC recommendations were to continue monitoring to assess the need for further action following the El Niño event.

Recommendations for action in Old Tampa Bay in response to the decision matrix results in 2003–2004 were quite different than for the bay-wide El Niño-associated event. Following an extensive review of existing data and information, the TAC recommended that an Old Tampa Bay seagrass recovery research program be implemented to examine factors potentially affecting seagrass recovery in that segment of Tampa Bay, followed by development of a recovery and management plan. Initial monitoring results (summarized in Cross, 2007) indicated that some shallow areas in Old Tampa Bay had poorer water quality (and, thus, less light available for seagrasses) than in three other study areas. Epiphytes caused significant light reduction (25 to 32 percent) in all parts of Old Tampa Bay. Transplanted seagrass survival was very low — 0.9 percent after two growing seasons, compared with 21 percent in other areas of Tampa Bay. Additional factors were examined, including high wave energy and loads from submarine groundwater. However, neither of these appeared to be responsible for slower seagrass recovery rates (Griffen and Greening, 2004).

Further evaluations examined additional potential causes of poor water quality and slower seagrass recovery in Old Tampa Bay, including examination of reduced circulation and slower flushing rates (possibly resulting in higher chlorophyll *a* concentrations), local sources of N loading, increased epiphyte loads, high rates of bioturbation (by stingrays and burrowing organisms), and the potential influence of hydrogen sulfide concentrations. Results indicated that the lack of seagrass recovery in Feather Sound was probably due to multiple factors, and that a multipronged management strategy would be required. Ongoing efforts include plans to reduce runoff from adjacent land uses and restoration of fringing mangroves to promote sheet flow through the mangrove system (Cross, 2007).

Historic Results

Year	Old Tampa Bay	Hillsborough Bay	Middle Tampa Bay	Lower Tampa Bay
1975	Red	Red	Red	Green
1976	Red	Red	Red	Yellow
1977	Red	Red	Red	Red
1978	Red	Red	Red	Yellow
1979	Red	Red	Red	Red
1980	Red	Red	Red	Red
1981	Red	Red	Red	Red
1982	Red	Red	Red	Red
1983	Red	Yellow	Red	Red
1984	Red	Green	Red	Yellow
1985	Red	Red	Red	Yellow
1986	Red	Yellow	Red	Green
1987	Red	Yellow	Red	Green
1988	Yellow	Green	Yellow	Green
1989	Red	Yellow	Red	Yellow
1990	Red	Green	Red	Yellow
1991	Green	Yellow	Yellow	Yellow
1992	Yellow	Green	Yellow	Yellow
1993	Yellow	Green	Yellow	Yellow
1994	Yellow	Yellow	Red	Red
1995	Red	Yellow	Red	Yellow
1996	Yellow	Green	Yellow	Green
1997	Yellow	Green	Red	Yellow
1998	Red	Red	Red	Red
1999	Yellow	Green	Yellow	Yellow
2000	Green	Green	Yellow	Yellow
2001	Yellow	Green	Yellow	Yellow
2002	Yellow	Green	Green	Green
2003	Red	Yellow	Green	Yellow
2004	Red	Green	Green	Yellow
2005	Green	Green	Yellow	Yellow
2006	Green	Green	Green	Green
2007	Green	Green	Green	Green
2008	Yellow	Green	Green	Yellow
2009	Yellow	Yellow	Green	Green

Box 5–3, Figure 2. Decision matrix outcomes for 1975–2008. From Sherwood (2009).

Box 5–4. Tampa Bay Nitrogen Management Consortium (TBNMC)—A Collaborative Approach to Meet Water-Quality Targets and Support Seagrass Recovery in Tampa Bay

By Holly Greening (Tampa Bay Estuary Program)

A landmark agreement between more than 40 area government and private industry representatives to limit N pollution in Tampa Bay was finalized in September 2009. The agreement spells out how much N can enter Tampa Bay through stormwater, air pollution, treated wastewater, and industrial discharges through 2012. The limits will maintain N loadings to the bay at existing levels; additional N associated with growth must be offset through additional pollution controls.

In 1996, the TBEP local government and agency partners adopted numeric management targets to restore and protect seagrass beds and restore environmental conditions in Tampa Bay. These resource-based targets include the goal of restoring seagrass acreage to the extent observed in 1950, and numeric targets for water clarity, chlorophyll *a* concentrations, and the total N loads necessary to meet and maintain water-quality targets that support seagrass recovery (detailed in Chapter 5). A multipronged management strategy, implemented by the TBNMC was initiated in 1996 to meet these targets.

In 1998, FDEP proposed and USEPA approved a TMDL for N for Tampa Bay required by Section 303(d) of the Federal Clean Water Act. The TMDL total N loads were based on the resource-based management targets (water clarity, chlorophyll *a* concentrations and the total N loads observed to meet these targets) developed by the TBEP partners to support the environmental recovery of Tampa Bay.

Since 1998, FDEP chlorophyll *a* targets have been met in all four major bay segments, with the exception of 1 year in Lower Tampa Bay and 3 years in Old Tampa Bay (box 5–4, fig 1). Seagrass acreage has increased by more than 4,800 acres bay-wide over this same period, and more than 6,000 acres since the mid 1980s (fig. 4–29).

In December 2007, the public and private participants in the TBNMC (box 5–4, table 1) committed to develop a process to allocate N loads among all sources, to support continued attainment of bay management targets and to be consistent with the required TMDL. The Consortium participants developed N load allocations that equitably distribute the burden of N management across the sectors and sources of N loading within the basin, as well as the total maximum loading of N to each major bay segment. Through this consensus-based process, Consortium participants defined limits to the amount of N they

Year	Old Tampa Bay	Hillsborough Bay	Middle Tampa Bay	Lower Tampa Bay
1975	No	No	No	Yes
1976	No	No	No	Yes
1977	No	No	No	No
1978	No	No	No	Yes
1979	No	No	No	No
1980	No	No	No	No
1981	No	No	No	No
1982	No	No	No	No
1983	No	No	No	No
1984	Yes	Yes	No	Yes
1985	No	No	No	Yes
1986	No	No	Yes	Yes
1987	No	Yes	No	Yes
1988	Yes	Yes	Yes	Yes
1989	No	Yes	Yes	Yes
1990	No	Yes	Yes	Yes
1991	Yes	Yes	Yes	Yes
1992	Yes	Yes	Yes	Yes
1993	Yes	Yes	Yes	Yes
1994	No	No	No	No
1995	No	No	No	Yes
1996	Yes	Yes	Yes	Yes
1997	Yes	Yes	Yes	Yes
1998	No	No	No	No
1999	Yes	Yes	Yes	Yes
2000	Yes	Yes	Yes	Yes
2001	Yes	Yes	Yes	Yes
2002	Yes	Yes	Yes	Yes
2003	No	Yes	Yes	Yes
2004	No	Yes	Yes	Yes
2005	Yes	Yes	Yes	No
2006	Yes	Yes	Yes	Yes
2007	Yes	Yes	Yes	Yes
2008	Yes	Yes	Yes	Yes
2009	No	Yes	Yes	Yes

Box 5–4, Figure 1. Compliance with Florida Department of Environmental Protection approved annual average chlorophyll *a* thresholds for each major bay segment, 1974–2008. The thresholds are: Hillsborough Bay, 15.0 µg/L; Old Tampa Bay, 9.3 µg/L; Middle Tampa Bay, 8.5 µg/L; and Lower Tampa Bay, 5.1 µg/L. Green indicates compliance with these thresholds; red indicates noncompliance. From Tampa Nitrogen Management Consortium.

are permitted to discharge. For example, communities that hold permits to discharge more treated wastewater than they currently are must “hold the line” at current levels — unless they can prove they have lowered N pollution elsewhere in their communities. Participating private sector partners must meet the same restrictions.

Box 5–4, Table 1. Parcipants of the Tampa Bay Nitrogen Management Consortium.

Alafia Preserve (Mulberry), LLC	Hillsborough County
CF Industries	Janicki Environmental, Inc. (technical support)
City of Bradenton	Kerry I & F Contracting
City of Clearwater	Kinder Morgan Bulk Terminals, Inc.
City of Gulfport	LDC Donaldson Knoll Investments, LLC
City of Lakeland	MacDill Air Force Base
City of Largo	Manatee County
City of Mulberry	Mosaic Company
City of Oldsmar	Pasco County
City of Palmetto	Pinellas County
City of Plant City	Polk County
City of Safety Harbor	Southwest Florida Water Management District
City of St. Petersburg	Tampa Bay Estuary Program (coordinator and facilitator)
City of Tampa	Tampa Bay Regional Planning Council
CSX Transportation	Tampa Bay Water
Eagle Ridge (Mulberry), LLC	Tampa Electric Company
Eastern Associated Terminals	Tampa Port Authority
Environmental Protection Commission of Hillsborough County	Trademark Nitrogen
Florida Department of Agriculture and Consumer Services	Tropicana Products
Florida Department of Environmental Protection	U.S. Environmental Protection Agency
Florida Department of Transportation	Yara North America

In September 2009, the Consortium participants finalized and approved their technical process, and proposed total N allocations to all 189 point and nonpoint sources within the Tampa Bay watershed (TBNMC, 2009). In December 2009, FDEP provided their concurrence with the technical basis and allocations. The TBNMC’s collaborative approach to meeting water-quality targets is unique in the country in that public and private N dischargers worked together to define the technical process and N load limits for each of the sources within the watershed. FDEP and USEPA participated in the Consortium and provided concurrence at each major step.

Tampa Bay Nitrogen Management Consortium (TBNMC)

Successful long-term achievement of the N loading strategy and chlorophyll *a* targets could be prevented as a result of ongoing population growth in the watershed, which could potentially lead to increased N discharges to the bay as a result of increased wastewater and stormwater loads. To maintain external N loads at or below the estimated 1992–1994 levels, it appears that an average annual reduction of about 17 tons/yr of total N will be necessary to offset expected load increases generated as a result of population growth (Greening and Janicki, 2006).

To meet the hold-the-line N loading management target, a consortium of local governments, phosphate companies, electric utilities, agricultural interests and regulatory agencies (the TBNMC) was established by the TBEP in 1998 (Greening, 2001; box 5–4). Together, TBNMC members developed a Nitrogen Management Action Plan and made voluntary commitments to implement projects that will help to achieve the adopted N management goals in each bay segment.

Load reductions achieved by the TBNMC and other TBEP partners have generally been calculated on a 5-year basis (as 85 tons per 5-year period) rather than annually, because of the long-term nature of many of the N reduction projects. To ensure that each partner was using similar N load reduction assumptions for similar projects, the TBNMC developed guidelines for calculating N load reduction credits (Zarbock and Janicki, 1997) which are used in developing action plans and estimating project-specific load reductions.

The types of nutrient reduction activities included in the Nitrogen Management Action Plan range from traditional projects, such as stormwater upgrades, industrial retrofits, and agricultural best management practices, to actions not primarily associated with nutrient reduction, such as land acquisition and habitat restoration projects. During 1995–2007, more than 250 projects were submitted by local governments, agencies, and industries for inclusion in the plan (TBNMC, 2008). During the plan's initial 5-year period (1995–1999), completed projects produced an estimated load reduction of 175 tons of total N, exceeding the 5-year reduction target by 90 tons. The estimated load reductions during this period were evenly divided between public and private sector projects (Greening and DeGrove, 2001). Nitrogen-reduction projects for 2000–2004 totaled 228 tons, and for 2005–2009 a reduction of 192 tons of total N are expected (TBNMC 2008; table 5–1).

Table 5–1. Total nitrogen load reductions, 1995–2009.

Drainage basin	5-year reduction target (tons)	1995–1999	2000–2004	2005–2009
Alafia River	11.10	11.15	44.09	28.34
Coastal Hillsborough Bay	11.50	97.25	81.23	83.44
Hillsborough River	8.55	11.98	40.94	10.17
Coastal Middle Tampa Bay	5.45	5.07	1.61	2.32
Little Manatee River	10.80	11.66	3.85	1.16
Old Tampa Bay	11.80	33.32	8.88	43.93
Coastal Lower Tampa Bay	9.70	0.96	1.88	1.71
Manatee River/Terra Ceia Bay	11.60	2.90	42.68	8.34
Boca Ciega Bay	4.60	0.88	2.98	12.30
Total reductions	85.10 (target)	175.17	228.14	191.71

Toxins and Harmful Algal Blooms

Although nutrients and eutrophication have been the primary water-quality issues facing resource managers in the Tampa Bay area in recent decades, management attention has also focused on toxic chemicals, some of which are associated with anthropogenic discharges and others of which are produced as a result of harmful algal blooms. This section addresses known and potential toxins that are found in the water column or in animals associated with the water column. Toxic contaminants associated with bottom sediments, and the benthic biota that is present in and on the sediments, are discussed in Chapter 7.

Mercury in Fish Tissue

Fish consumption advisories are currently in place for many Florida waters, including Tampa Bay, because of the elevated mercury levels that are often detected in certain species. FDEP and USEPA plan to develop a statewide TMDL addressing this water-quality impairment in coming years.

The global mercury cycle involves emissions from land and water surfaces to the atmosphere, transport in the atmosphere on a global scale, possible chemical conversion in the atmosphere, and return to land and water by various depositional processes. Part of the inorganic mercury that is emitted to the atmosphere becomes oxidized (to Hg^{++}) and then methylated, a reaction that is believed to involve a methylcobalamine compound (an analogue of vitamin B12) that is produced by bacteria. This reaction takes place primarily in aquatic ecosystems. The intestinal bacteria of various animal species, including fish, are also able to convert ionic mercury into methylmercuric compounds (CH_3Hg^+), although to a much lower degree. Methylmercury is accumulated by fish and marine mammals and reaches its highest concentrations in large predatory species at the top of the aquatic food web. Consumption of these organisms is the primary route through which it enters the human diet (USGS, 1995; USEPA, 1997, 2007).

From a human health perspective, mercury is a toxic metal that persists in the body and impairs the functioning of the brain and central nervous system. Its effects are particularly harmful to developing nervous system tissues, so consumption advisories are directed particularly toward infants, small children, and women of child-bearing age who are or may become pregnant. A 2006 summary of mercury-related fish consumption advisories for the Tampa Bay region which was developed by EPCHC and TBEP is shown in figure 5–17.

Fish located in
Hillsborough County coastal waters

HIGH RISK – – Do not eat
King Mackerel, Shark, Blackfin Tuna, Cobia, Little Tunny

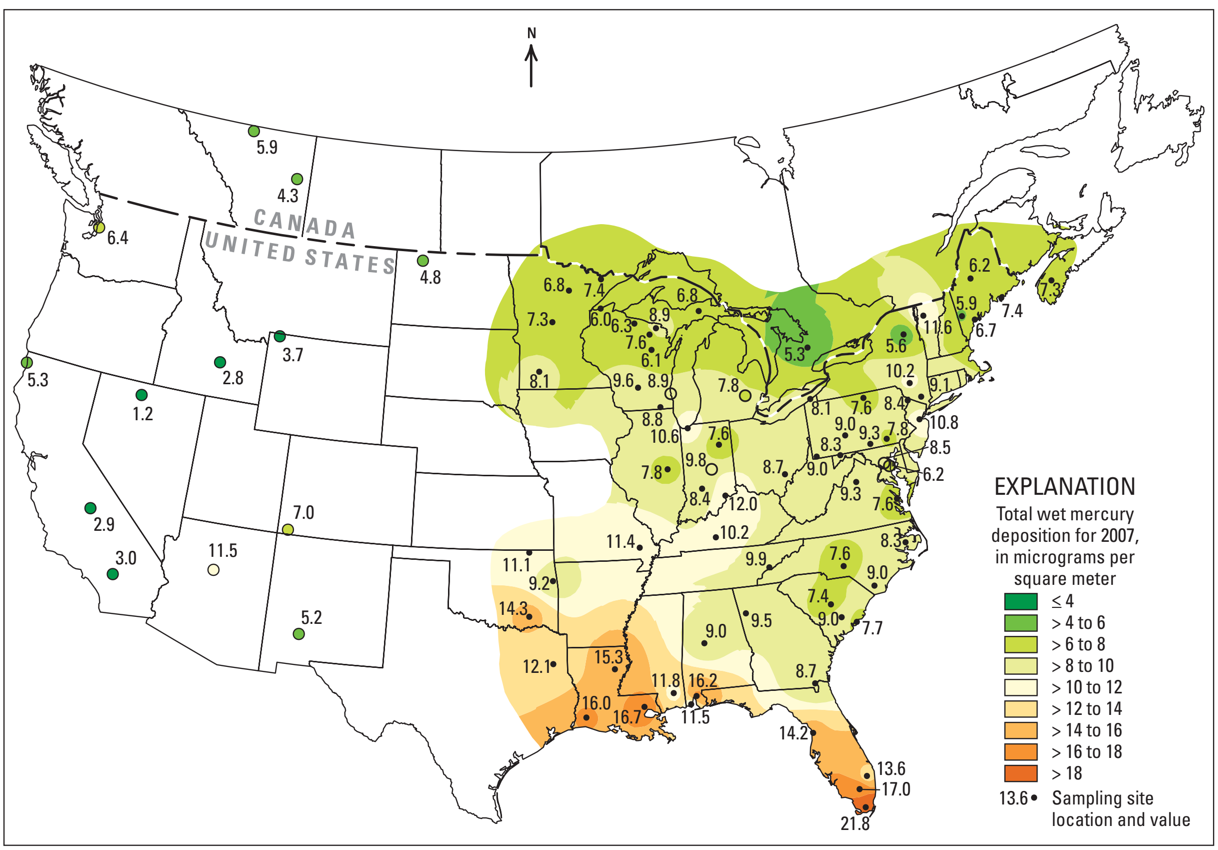
MODERATE RISK – – One per month
Almaco Jack, Atlantic Stingray, Black Grouper, Bluefish, Bonefish, Crevalle Jack, Gafftopsail Catfish, Gag Grouper, Greater Amberjack, Great Barracuda, Gulf Flounder, Ladyfish, Lane Snapper, Mutton Snapper, Pinfish, Red Drum, Red Grouper, Sand Seatrout, Scamp, Sheepshead, Silver Perch, Skipjack Tuna, Snook, Snowy Grouper, Spanish Mackerel, Spotted Seatrout, Wahoo, While Grunt, Yellow-edge Grouper, Yellowfin Tuna

MODERATE TO LOW RISK – – One per week
Atlantic Croaker, Atlantic Spadefish, Atlantic Thread Herring, Atlantic Weakfish, Black Drum, Bluntnose Stingray, Dolphin, Fantail Mullet, Florida Pompano, Gray Snapper, Hardhead Catfish, Hogfish, Lookdown, Pigfish, Red Snapper, Southern Flounder, Spot, Striped Mullet, Striped Mojarra, Tarpon, Tripletail, Vermilion Snapper, White Mullet, Yellowtail Snapper

Figure 5–17. Fish consumption advisories due to mercury contamination for Tampa Bay. From Environmental Protection Commission of Hillsborough County and Tampa Bay Estuary Program.

Sampling surveys across the United States have shown widespread mercury contamination in fish collected from wetlands, lakes, and streams. As of 2006, mercury-related fish consumption advisories had been issued for waterbodies in 48 U.S. States, the exceptions being Alaska and Wyoming (USEPA, 2007). Mercury contamination is also a global environmental issue, and mercury that is emitted to the atmosphere can travel thousands of miles before it is deposited back to the Earth in rainfall or in dry gaseous forms (USGS, 1995; USEPA, 1997). Natural sources of mercury, such as volcanic eruptions and emissions from the ocean, have been estimated to contribute about a third of current worldwide mercury air emissions, whereas anthropogenic emissions account for the remaining two-thirds. These estimates, however, are highly uncertain. Much of the mercury circulating in the modern environment is thought to have been released years ago, when mercury was commonly used in many industrial, commercial, and residential products and processes.

Within the United States, the highest mercury deposition rates are predicted to occur in the southern Great Lakes and Ohio River valley, the Northeast and scattered areas in the South (fig. 5–18), with the most elevated deposition occurring "in the Miami and Tampa areas" (USEPA, 1997). The location of sources, the chemical species of mercury emitted, and climate and meteorology are key factors in mercury deposition, with humid locations often experiencing higher deposition rates than arid locations (USEPA, 1997).



Data from National Atmospheric Deposition Program / Mercury Deposition Network

Figure 5–18. Mercury wet deposition in the United States, 2007. From Mercury Deposition Network.

Harmful Algal Blooms

Several types of harmful algal blooms have been documented in Tampa Bay and its watershed, most frequently involving Florida red tide, other toxin-producing dinoflagellates (primarily in estuarine and marine waters), and cyanobacteria (in eutrophic freshwater bodies).

Florida Red Tide

Generically, a “red tide” is a type of harmful algal bloom that occurs when a population of toxic planktonic algae grows to a sufficiently large size and density to cause visible discoloration of the water (fig. 5–19). Severe blooms can cause a number of undesirable environmental, economic, and human health effects in coastal areas. Although more than 40 species of toxic microalgae live in the Gulf of Mexico, the type that causes red tides in the Tampa Bay area and elsewhere along the west-central and southwest Florida coast is the dinoflagellate, *Karenia brevis*, the Florida red tide organism (Walsh and others, 2006) (fig. 5–20). Most *Karenia* blooms last three to five months and may affect hundreds of square miles of coastal waters and bays (Tester and Steidinger, 1997). Occasionally, however, as happened during 2005–2006, a bloom will continue for as long as 18 months and will affect thousands of square miles.

The environmental impacts of large blooms include mass die-offs of fish, birds, and marine mammals (Fleming and others, 2005). Such blooms impact the economies of affected coastal areas. The blooms reduce levels of recreation and tourism due to aesthetic and odiferous impacts of accompanying fish kills, and respiratory discomfort is experienced by people sensitive to the brevetoxin that is released into the water and air by the red tide organism (Hoagland and others, 2002). In addition to their environmental impacts, brevetoxins can also accumulate in filter-feeding shellfish, such as oysters, clams, and coquinas, sometimes reaching levels capable of causing neurotoxic shellfish poisoning when eaten by humans. Neurotoxic shellfish

Figure 5–20. Red tide organism *Karenia brevis*. Photo by Fish and Wildlife Research Institute.

Figure 5–19. Red tide (*Karenia. brevis*) bloom along Florida’s west coast. Photo by Fish and Wildlife Research Institute.

poisoning appears to be a temporary illness whose symptoms usually appear within a few hours of eating contaminated shellfish and include nausea and diarrhea, dizziness, muscular aches, and tingling and numbness in the tongue, lips, throat, and extremities (Kirkpatrick and others, 2004).

The Florida red tide organism was identified in 1947, but anecdotal reports of red tides in the Gulf of Mexico were provided by Spanish explorers during the 16th century. Small to moderate blooms occur in the gulf in most years, usually in late summer or early fall and most commonly off the coast between Clearwater and Sanibel Island. They can occur anywhere in the gulf, however, and occasionally develop along the southeastern Atlantic coast as far north as North Carolina (Tester and Steidinger, 1997).

Florida red tides were once believed to originate in nearshore areas, because discolored water, fish kills, and respiratory irritation were often observed first around passes and barrier islands. However, more recent information has shown that they begin 10 to 50 mi offshore, on the West Florida Shelf, where resting stages of *Karenia* are believed to persist in the water column or sediments (Tester and Steidinger, 1997; Walsh and others, 2006).

Blooms usually develop in four stages (Alcock, 2007; Tester and Steidinger, 1997). The initiation stage occurs when a *Karenia* population is introduced into an area that provides an appropriate combination of environmental conditions to support it. The second stage is growth, during which the population steadily increases. Within a few weeks, cell concentrations may reach sufficiently high levels to kill fish. The third stage is maintenance, during which the bloom may remain offshore or be moved inshore by wind and currents. If the bloom moves inshore, the increased nutrient levels often encountered in bays and estuaries may allow additional cell growth, and physical factors, such as currents, may concentrate the cells in particular locations. Such blooms can persist in coastal areas for days, weeks, or even months. The fourth stage is dissipation/termination, whereby winds and currents disperse the cells or move the bloom to a different area.

Atmospheric deposition of dust from the Saharan Desert onto the waters of the West Florida Shelf may also play an important role in bloom initiation and growth. Recent hypotheses that have been advanced to explain the high frequency of *Karenia* blooms in this part of the Gulf of Mexico have noted that Saharan dust contains iron, which is deposited over the west coast of Florida during storm events (Walsh and Steidinger, 2001). These dust deposition events frequently precede blooms of *Trichodesmium*, a N-fixing cyanobacteria, which in turn precede blooms of *Karenia*. *Trichodesmium* blooms often occupy thousands of square kilometers of the gulf and can be very dense (Walsh and Steidinger, 2001). The regenerated nutrients from these blooms, particularly regenerated N, could help fuel large red tides, as could decomposing fish from fish kills (Walsh and others, 2006). Because *Trichodesmium* requires abundant P as well as iron to carry out N fixation, the relative importance of different P sources (from ocean upwelling events, benthic releases on the West Florida Shelf, or discharges from land via rivers and estuaries) in the initiation, growth, or maintenance of Florida red tide blooms is also a question of considerable recent research interest (Walsh and others, 2006; Alcock, 2007).

Because of the economic and environmental impacts caused by major red tide blooms, coastal residents and policymakers have asked researchers and resource managers to determine whether the blooms are becoming more frequent or more severe over time, and whether anthropogenic factors, such as cultural eutrophication of bays and other coastal waters, contribute to bloom severity or persistence (Hu and others, 2006; Walsh and others, 2006; Alcock, 2007; Brand and Compton, 2007; Stumpf and others, 2008; Vargo and others, 2008). Given the complexity of the *Karenia* life cycle, and the large number of factors that affect the initiation, persistence, and severity of blooms, these questions have proven very difficult to answer. Alcock (2007) has provided an overview, based on a review of recent research results, which is summarized in box 5–5.

Other Harmful Algal Blooms

On a worldwide basis, the man-induced eutrophication that has occurred in recent decades has coincided with a higher frequency and severity of primarily dinoflagellate harmful algal blooms in estuarine and marine waters, and of cyanobacteria (also known as blue-green algae) harmful algal blooms in nutrient-enriched freshwater bodies (Anderson and others, 2002; Reynolds, 2006).

Box 5–5. Frequently Asked Questions about Florida Red Tide

Excerpt from Alcock (2007)

Are Florida red tides getting worse?

“Possibly. Harmful algal blooms appear to be getting worse throughout the world. Some of the forcing factors believed to play a role in the worldwide trend are increased nutrient enrichment resulting from population growth and land use practices and increased water temperatures due to global climate change. Although the general trend appears to be worsening, trends for specific harmful algal blooms can embody more uncertainty. This is particularly true for offshore blooms, such as *Karenia brevis*, the organism that causes Florida red tides. Southwest Florida has endured red tide blooms on a near-annual basis over the past two decades, and the 2005 bloom was one of the most severe on record. However, Florida red tide blooms of similar intensity and duration have been confirmed as far back as 1948–1949, and anecdotal evidence suggests that severe blooms have scourged the region for hundreds, if not thousands of years. There is broad consensus that Florida red tides have been especially active in recent years, but putting this decade into historical perspective is extremely difficult due to a lack of data suitable for determining historical trends” (Alcock, 2007).

Can coastal pollution exacerbate Florida red tides?

“Probably. The recipe for Florida red tides is complex. The relative importance of different ingredients - nutrient sources and other environmental factors — varies over the different stages of a bloom and it is possible that the specific recipe responsible for red tides varies from bloom to bloom. Terrestrial nutrient fluxes are one of many ingredients that can contribute to a red tide bloom, and coastal pollution exacerbates these fluxes. Most scientists agree that red tide blooms initiate offshore before being transported inshore by wind and ocean currents. They believe coastal runoff is unlikely to affect the early stages of a bloom, but when a bloom moves inshore, they acknowledge that runoff can play a role in intensifying or prolonging a bloom. Assessing the relative importance of terrestrial nutrient sources, including coastal pollution, remains a top research priority” (Alcock, 2007).

In the Gulf of Mexico and nearby regions, harmful algal blooms are associated with a number of human health impacts, including ciguatera poisoning, paralytic shellfish poisoning, and neurotoxic shellfish poisoning. Ciguatera poisoning is caused by ingestion of ciguatoxins — produced by the dinoflagellate *Gambierdiscus toxicus* — which can reach toxic concentrations in tropical reef fish and their predators. Consumption of affected barracuda, grouper, sea bass, snapper, and a number of other marine fish that commonly live between latitudes 35°N and 35°S causes the disease in humans. Ciguatera poisoning is rarely fatal. Common symptoms include nausea, vomiting, diarrhea, cramps, excessive sweating, headache, muscle aches, burning sensations in the extremities, weakness, itching, lowering of blood pressure, and dizziness.

Paralytic shellfish poisoning (PSP) is associated with several species of dinoflagellates that produce at least 12 known toxins, of which the saxitoxins are best understood. PSP toxins are typically found in bivalve mollusks (mussels, clams, oysters) and have also been detected in crabs and snails associated with coral reefs. In humans, symptoms of PSP begin from 15 minutes to 10 hours after eating contaminated shellfish. Symptoms are generally mild and begin with numbness or tingling of the face, arms, and legs. This is followed by headache, dizziness, nausea, and muscular incoordination. In cases of severe poisoning, muscle paralysis and respiratory failure occur, and in these cases death may occur in 2 to 25 hours (*http://www.whoi.edu/science/B/redtide/illness/psp.html*). Although PSP is most prevalent in the relatively low-temperature coastal waters of the Pacific States and New England, the syndrome has also been reported in Central America.

A total of 28 cases of saxitoxin poisoning occurred in the eastern United States during 2002–2004, associated with the consumption of puffer fish from the Indian River Lagoon in eastern Florida (Landsberg and others, 2006). The dinoflagellate *Pyrodinium bahamense* was identified as the likely source of the toxin (Landsberg and others, 2006). Large blooms of *Pyrodinium* have occurred in Tampa Bay in recent years (Badylak and others, 2007). Fish kills have occurred in conjunction with these blooms (for example, fig. 5–4), but appear to have been related to reduced DO levels rather than to algal toxicity.

Pharmaceutical and Personal Care Products (PPCPs), and other Emerging Contaminants

Recent improvements in laboratory analytical methods now allow researchers to detect the presence of manmade chemicals — such as prescription drugs, over-the-counter pharmaceuticals, and personal care products (PPCPs) such as soaps, shampoos, and fragrance compounds — at very low concentrations in surface and groundwater samples. These chemicals, which are also referred to as emerging contaminants, come from a variety of sources, such as sewage-treatment plant effluent, runoff from agricultural land uses, and discharges from septic systems. Conventional sewage-treatment practices vary greatly in their abilities to eliminate them from wastewater streams. A recent report from the European Union (Ternes, 2006) notes that several studies in Europe and North America have documented the occurrence of pharmaceuticals and estrogens in wastewater effluents and ambient waters (Daughton and Ternes, 1999; Kolpin and others, 2002). In general, the concentrations of PPCPs in wastewater-treatment plant effluents were quite low, ranging from the ng L^{-1} to the low µg L^{-1} range (the parts-per-trillion to low parts-per-billion range). In surface waters, the concentrations generally ranged between 10 and 500 ng L^{-1}. In groundwater and drinking water, PPCP residues were detected up to the µg L^{-1} level. What is unclear is whether, or in which cases, these residues may pose risks for aquatic ecosystems or humans.

Pathogen-Related Water-Quality Impairments

Pursuant to the Federal Clean Water Act, the FDEP has adopted water-quality criteria for fecal coliform bacteria to protect human health in cases where waterborne pathogens could potentially be present in waters that are used for recreation, shellfish harvesting, or potable water supply. These criteria are frequently exceeded in a number of waterbodies in the Tampa Bay watershed and in parts of the bay itself (fig. 5–11). As a result, these waterbodies and parts of the bay have been designated as impaired, and FDEP and USEPA are developing TMDLs to address the impairments.

From a water-quality management perspective, however, interpretation of the existing criteria is complicated by the fact that, in most cases, the fecal coliform bacteria that are detected in surface-water samples are not human pathogens. Instead, they are indicator organisms that are used to estimate the likelihood that water has come into contact with fecal material (from humans or other warm-blooded vertebrates) and may, therefore, contain disease-causing organisms that pose a risk to public health. A considerable amount of research has shown that, as a group, fecal coliform bacteria are not particularly accurate indicators of the presence of these human pathogens in surface waters, particularly in subtropical and tropical areas (for example, Rose and others, 2001; McLaughlin and Rose, 2006).

Because the shortcomings of the existing fecal coliform indicators are well known within the public health and water-quality management communities, intensive work is being carried out by a number of organizations to develop cost-effective indicators that provide more accurate estimates of human health risks. Currently, however, no single indicator or analytical method has been found that provides greater accuracy in estimating risk, reasonable cost, and feasibility for day-to-day use by laboratory personnel in local monitoring programs. Until cost-effective alternatives are found and adopted, local management programs will need to continue using the existing indicators and other available information as effectively as possible to minimize human health risks associated with waterborne pathogens.

In the Hillsborough County part of the Tampa Bay watershed, FDEP, TBEP and local partners are using a basin management action plan approach in an effort to address bacteriological water-quality impairments in this way. The approach seeks to identify the subgroup of impaired waterbodies that pose actual human health risks, and to take steps to identify and correct the causes of those impairments as quickly and cost effectively as possible (Morrison and others, 2010).

Anticipated Future Challenges from Ongoing Population Growth

As noted, the human population in the three counties adjacent to Tampa Bay has more than quadrupled, from less than 500,000 in 1950 to more than 2 million in 2000 (see Chapter 1, fig. 1–5). The population in these three counties reached 2.4 million in 2006 and is projected to reach 2.8 million by 2015, 3.2 million by 2030, and more than 4 million by 2050. Without careful planning and appropriate action, this population increase could have serious implications for the Tampa Bay water-quality management effort and its ability to continue meeting the water-quality targets that have been set to protect and restore the living resources of the bay.

The TBEP partnership has estimated that projected levels of population growth could potentially cause annual N loadings to the bay to increase by about 17 tons/yr, due to increasing wastewater discharges, stormwater runoff, and atmospheric deposition from additional vehicular traffic and electric power generation (TBEP, 2006). In order to meet the hold-the-line total N management target, management actions must be taken to achieve N load reductions equivalent to the anticipated increases. As recently as 2008 (the most recent year for which data were available at the time this was written) these reductions were continuing to occur, and chlorophyll *a* concentrations in the bay continued to meet targets. However, it is not clear how far into the future this level of success can be maintained.

Much of the new N entering the bay each year arrives as a result of stormwater runoff (fig. 5–21). Increasing urbanization — with its associated construction of rooftops, parking lots, roadways and other impervious surfaces — can potentially affect these loading sources. Although the impacts of increasing imperviousness can be ameliorated to some extent by environmentally sensitive development practices and rigorous stormwater management requirements (Schueler and Holland, 2000; NRC, 2008), it is clear that the large and increasing amounts of imperviousness in the watershed will represent an ongoing challenge for water-quality managers.

In addition to stormwater runoff, the TBEP (2006) has noted a number of other sources of total N and toxic contaminants for which future load reductions will need to be achieved. These include industrial and municipal point sources, which contribute roughly 30 percent of the total bay loadings of arsenic, cadmium, chromium, and copper, as well as low levels of other contaminants. Watershed residents sometimes contribute to this problem by improperly disposing of toxic cleaners or solvents in central sewer or stormwater management systems.

Septic systems, which are estimated to serve about 20 percent of the population in the Tampa Bay watershed, are also an issue. High densities of mostly older septic tanks can contribute to degraded water quality (nutrients and pathogens) in creeks where circulation is limited and the water table is near the ground surface. Allen's Creek in Pinellas County, several creeks in Hillsborough County, and some tributaries in the Tampa area that discharge to McKay Bay are among those thought to be most affected (Rose and others, 2001).

Springs that feed into bay rivers and smaller tributaries can be impacted by septic tank leachate, especially in areas with highly porous soils. Preliminary estimates indicate that groundwater and springs contribute 5

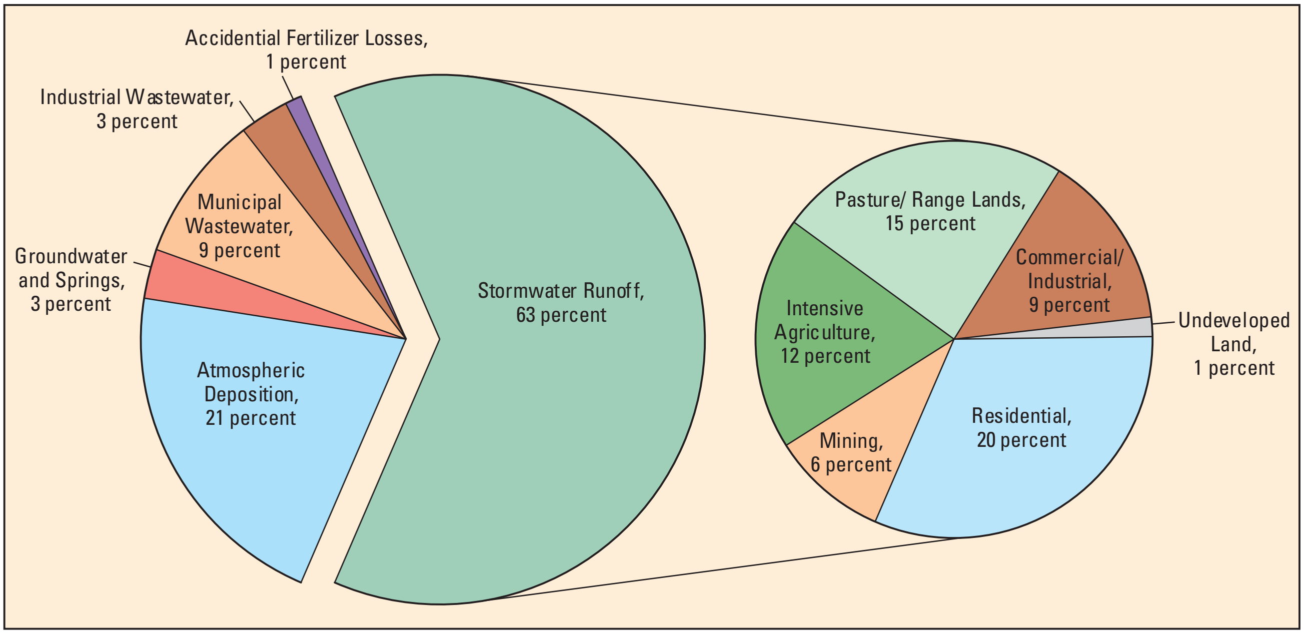


Figure 5–21. Sources of total nitrogen to Tampa Bay, 2003–2007 (Janicki Environmental Inc. (2008).

percent or more of the total N loadings to the bay (Janicki Environmental Inc., 2008). Nitrogen (particularly nitrate) concentrations in several springs in the area are increasing in response to factors, such as septic system discharges and historical agricultural fertilizer use (Jones and Upchurch, 1993).

Disposal of sewage sludge and septage poses an additional management issue, particularly in the Hillsborough and Manatee River Basins, because of the number of permitted disposal sites. Different agencies regulate sewage sludge and septage disposal sites, and it is sometimes difficult to determine how much material is being spread and how it is handled. Additionally, some of the sludge disposed of in the Tampa Bay watershed is brought in from outside the region.

Approximately 2 percent of the total N loadings to the bay is attributed to fertilizer products which are lost while in storage or during ship loading at local ports. These “fugitive emissions” have declined substantially since 1991, however, as a result of efforts to improve storage and handling practices at port facilities (TBEP, 2006).

In addition to these sources, the effort to keep pace with continuing population growth will also strain the municipal wastewater collection and treatment systems operated by local governments in the watershed. A 1994 survey conducted by the TBEP estimated the amount of money that is expended each year to manage and monitor bay quality and administer environmental programs (TBNEP, 1995). That study, based on fiscal year 1994–1995 budgets, indicated that more than $250 million was spent annually by Federal, State and local agencies and governments on the restoration and management of Tampa Bay. By far, the largest percentage of that overall amount — 68 percent, or roughly $170 million — was attributed to construction, maintenance, and administration of wastewater collection, treatment, and reuse systems.

Despite these expenditures, however, occasional leaks and spills from municipal wastewater collection and treatment systems remain a fact of life in many communities. That issue was highlighted during the summer of 1995, when the City of St. Petersburg was forced to discharge about 15 Mgal of untreated sewage into Boca Ciega Bay to minimize sewage backup into homes. Excessive rainfall had infiltrated the wastewater collection system and caused the overflows. Similar events have taken place in the City of Tampa in recent years, and other communities around the bay and nationwide experience similar problems during periods of heavy rainfall. To keep the bay safe for swimming and shellfish harvesting in the future, local communities will need to grapple with infrastructure improvements that will ensure that the significant investments made to upgrade sewage-treatment facilities are not diminished by chronic failures in collection networks (TBEP, 2006). Continuation of the collaborative watershed management effort, involving each of these source categories and residents of the region who contribute to them, will clearly be needed for ongoing maintenance and restoration of water quality and the bay's living resource.

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