

Chapter 7. Sediment Contaminants and Benthic Habitat Quality

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TAMPA BAY IS A HIGHLY urbanized estuary (fig. 7–1) that receives discharges from a variety of point and nonpoint sources, including urban stormwater runoff, industrial and municipal wastewater discharges, atmospheric deposition of automotive and industrial emissions, accidental spills, illegal dumping, and runoff and drift of pesticides applied to residential, commercial, and agricultural lands (Frithsen and others, 1995; Carr and others, 1996; Long and others, 1996; McConnell and Brink, 1997; Cabezas and others, 1998; Fernandez, 2002). These discharges have occurred at varying rates for many years and contain a number of chemical compounds, which, if present in sufficiently high concentrations, can be toxic to aquatic organisms. After being discharged into the bay or its tributaries, a part of the incoming contaminant load adheres to particulate matter and is deposited in bottom sediments.



Figure 7–1. Industrial facility located at the Port of Tampa in Hillsborough Bay, in the northeastern subbasin of Tampa Bay. Photo by Nanette O'Hara, Tampa Bay Estuary Program.

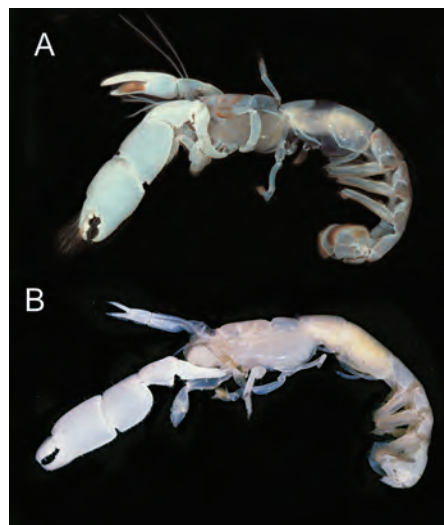


Figure 7-2. Two common species of ghost shrimp from Tampa Bay: A, *Lepidophthalmus louisianensis*; and B, *Sergia trilobata*. Photo by Darryl Felder and Paul Klerks, University of Louisiana, Lafayette.

When present at elevated concentrations, sediment contaminants, such as metals and synthetic organic compounds, can have a number of adverse environmental impacts, including acute or chronic toxicity, sublethal behavioral or mutagenic changes, or changes in the density or taxonomic composition of water-column (demersal) or of bottom-dwelling (benthic) fish or invertebrate fauna (NRC, 1989). Some contaminants — such as DDT, mercury, and polychlorinated biphenyls (PCBs) — also tend to accumulate in biological tissues, giving them the potential to reach elevated concentrations in fish or shellfish tissue, which can cause health risks for those organisms' predators and for human consumers (MacDonald and others, 2004). The presence of potentially toxic levels of chemical contaminants in sediments, therefore, is a significant bay management issue.

Contaminant concentrations found in estuarine sediments are affected by a number of factors. Sediments in the immediate vicinity of contaminant discharge points — such as urban stormwater outfalls, industrial or municipal wastewater discharges, or port areas where potentially toxic cargo is loaded and ships are repainted, repaired, and refueled — often show concentrations of metals and synthetic organic chemicals that are elevated in comparison to natural background levels (Long and others, 1991, 1994). Because many toxicologically active contaminants are associated with the surfaces of sediment particles, the particle size distribution at a site is also important in determining contamination levels. Due to their physical and chemical characteristics, fine-grained sediments — particularly those in the silt-clay or “mud” size range ($< 64 \mu$ diameter) — tend to adsorb contaminants, and estuarine sites containing high densities of sediments in this size range frequently exhibit elevated contaminant concentrations (NRC, 1989; Brooks and Doyle, 1991).

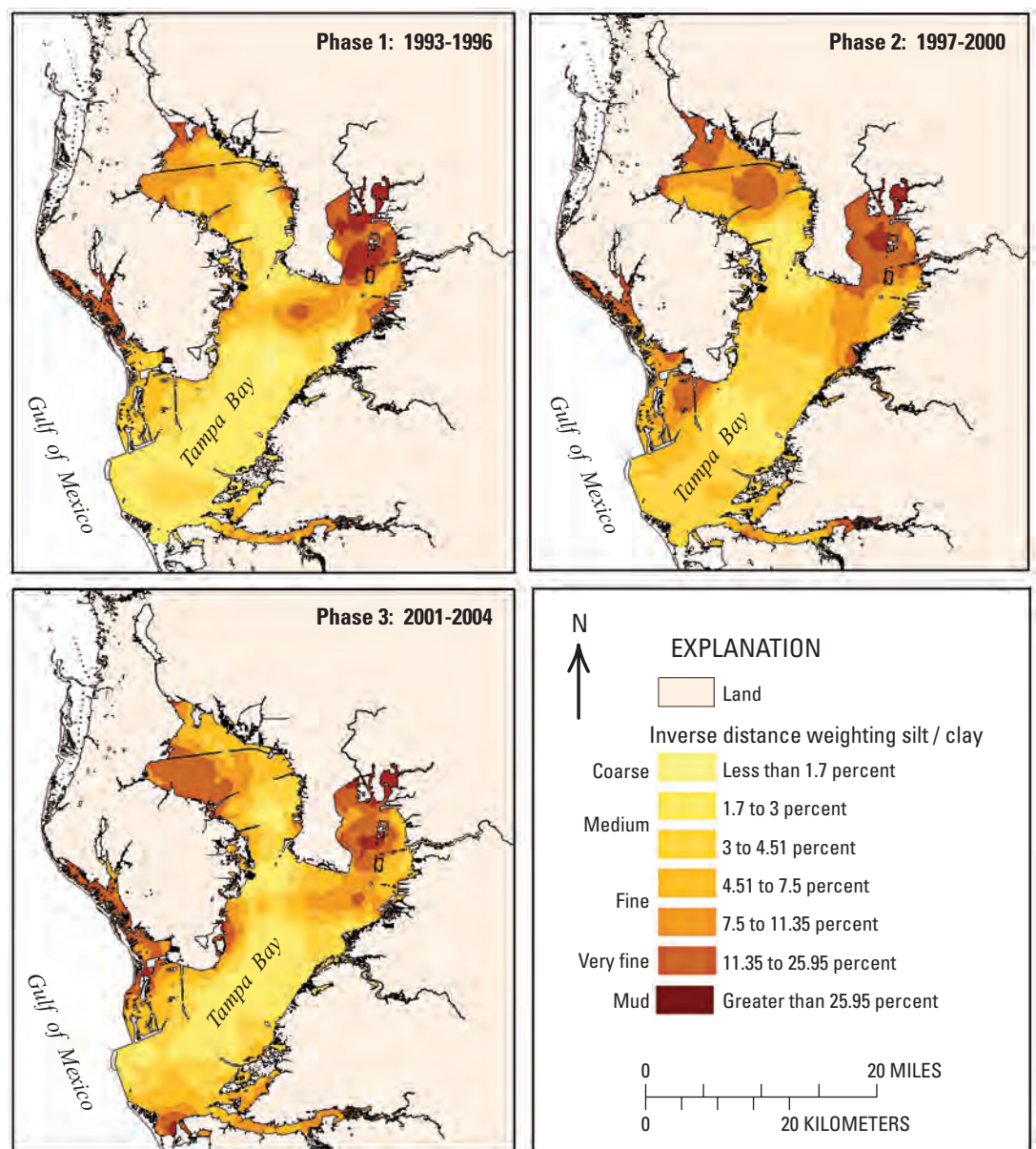
Organisms living on or in the sediments can also affect the spatial distribution of contaminants. Ghost shrimp, for example, occur at high densities in many estuaries and maintain large burrow systems in the sediments, which can extend to a depth of about 1 to 2 m. In Tampa Bay, Klerks and others (2007) found that levels of both cadmium and zinc were significantly higher in burrow walls than in surface sediments (fig. 7-2). They concluded that the presence of ghost shrimp burrows contributes to spatial heterogeneity of sedimentary metal levels, whereas bioturbation of sediments by ghost shrimp results in a significant flux of metals to the sediment surface and may act to decrease heterogeneity of metal levels in sedimentary depth profiles.

Anthropogenic sediment contaminants may also affect seagrasses and their faunal associates. As discussed in box 7-2, levels of metals were examined in sediments and seagrass tissue from 15 different locations throughout Old Tampa Bay, Middle Tampa Bay, and Lower Tampa Bay. Concentrations of nickel, copper, and zinc in seagrass tissues exceeded concentrations in surrounding sediment at all 15 sample locations. Concentrations of lead in seagrasses exceeded sediment concentrations in 13 locations, chromium in 12 locations, and arsenic in 6 locations, implying bioaccumulation of metals at these sites. These results indicate that an in-depth study of the impact of metal contaminants on seagrass growth may be warranted. Concentration of metals in seagrass tissue followed by deterioration of dead seagrass may provide a mechanism for remobilization and transport of contaminant metals throughout

Tampa Bay. Additionally, bioaccumulation of metals in seagrass may provide a mechanism for transport of metal contaminants to higher trophic-level animals feeding on seagrass.

During the 1980s and 1990s, a number of depositional areas containing high concentrations of fine-grained sediments were mapped in Tampa Bay and identified as potential “hot spots” of sediment contamination (Brooks and Doyle, 1991; Schoellhamer, 1991). These include several low-energy areas in Hillsborough Bay and Old Tampa Bay, and localized areas, such as dredged canals and marinas around the bay’s periphery (Johansson and Squires, 1989; Brooks and Doyle, 1991; Schoellhamer, 1991). In general, currents and waves in open areas of Middle Tampa Bay and Lower Tampa Bay appear to have sufficiently high energy to prevent significant accumulations of fine-grained sediments (Brooks and Doyle, 1991). However, the shipping channels in these parts of the bay must be maintenance-dredged on a regular basis to maintain adequate depths for commercial shipping use (TBEP, 2006), and the quality of sedimentary material that accumulates in these channels has not been thoroughly characterized. Information on the distribution of fine-grained sediments in the major segments of Tampa Bay, based on recent benthic sampling, is shown in fig. 7–3.

Figure 7–3. Estimated distribution of sediments in Tampa Bay over three time periods, based on information from benthic monitoring program. Data from Karlen and others (2008).



Data from C. Meyer, PCDEM, November, 2006
Project: EMAP_Spatial_06_Phase3_siltclay.mxd

Box 7–1. Albino Mutation in Red Mangroves

By Kimberly K. Yates (U.S. Geological Survey–St. Petersburg, Florida) and Ed Proffitt (Florida Atlantic University)

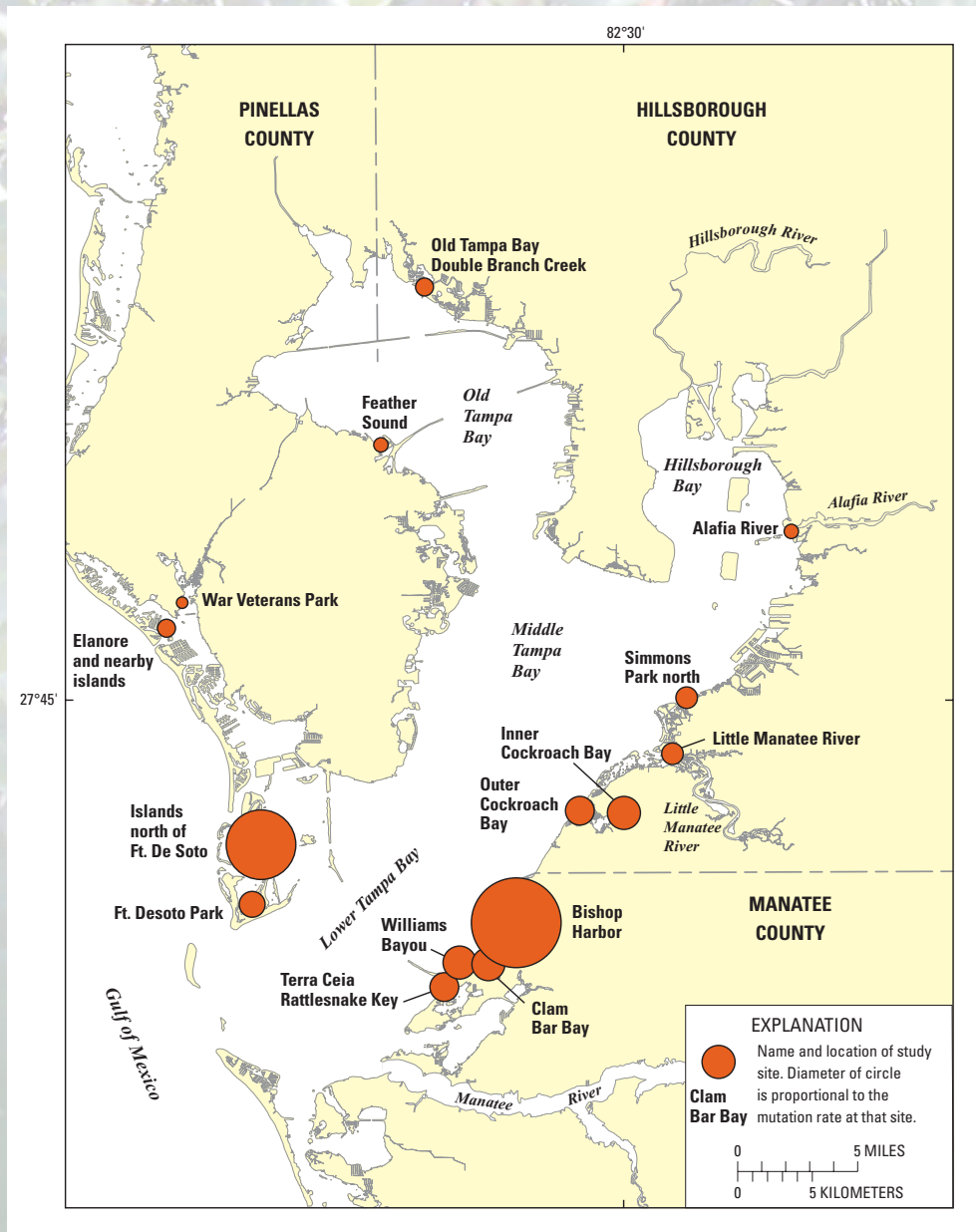
Mutagens are substances that tend to increase the frequency of genetic mutations and may include inorganic and organic chemicals, metals, radioactive substances, ultra-violet light, and high temperatures, among others. Petroleum hydrocarbons resulting from oil spills are typically degraded very quickly in tropical marine waters (Botello and Castro-Gessner, 1980). However, PAHs resulting from degradation of hydrocarbons are easily incorporated into underlying sediments that support the growth of wetland plants and submerged vegetation. PAHs are highly mutagenic and cause recessive mutations in the red mangrove species, *Rhizophora mangle* L., resulting in chlorophyll deficiency, albinism in mangrove propagules, and impaired reproduction (Klekowski and others, 1994). The mutation occurs in the apical meristems of seedlings that become trees. These trees then express the mutation as a 3:1 ratio of normal to albino propagules. This mutation can also be seen in offspring trees whose propagules came from mutated trees. In growing trees that are affected by this mutation later in their life-stage, the mutation may occur in the apical meristem of a single branch. In this case, all of the propagules on secondary stems that grow from that mutated branch will show albinism, but the rest of the tree will appear normal. This mutagenic effect is easily identifiable, making the red mangrove an ideal species for assessing the effects of historic contamination events on mangrove forests (box 7–1, fig. 1).

The USGS (Proffitt and Travis, 2005) compared the frequency of trees exhibiting albino propagules in four historically contaminated sites and 11 uncontaminated sites located throughout Tampa Bay. The four contaminated sites had a known history of

contamination by either oil spills or spills and discharges from phosphate plants, and included islands north of Fort Desoto, Eleanore Island area, Simmons Park, and Bishop Harbor. Out of 16,989 counted trees, 97 showed the albino mutation. Highest mutation rates were located near the mouth of Tampa Bay, and lower rates were



Box 7–1, Figure 1. Red mangrove tree (*Rhizophora mangle* L.), showing the mutagenic effect of propagule albinism resulting from contamination by polycyclic aromatic hydrocarbons. Albinism manifests itself in Red Mangrove trees as discolored (off-white to reddish) propagules. Photo by Ed Proffitt, Florida Atlantic University.



Box 7-1, Figure 2. Location of study sites in Tampa Bay, Florida. The diameter of circles is proportional to the mutation rate at that site. Image by Ed Proffitt, Florida Atlantic University.

located in Old Tampa Bay, Middle Tampa Bay, and Hillsborough Bay (box 7-1, fig. 2). Mutations were significantly greater in contaminated, as opposed to uncontaminated, sites. There was, however, no difference in stand reproduction effort or mean rank tree size between contaminated and uncontaminated sites. This baseline dataset can be used to assess before and after effects for future oil spill events as a metric to gauge future pollution abatement efforts, and to assist resource managers in the development of wetland restoration projects that require mangrove transplants or a source of mangrove propagules.

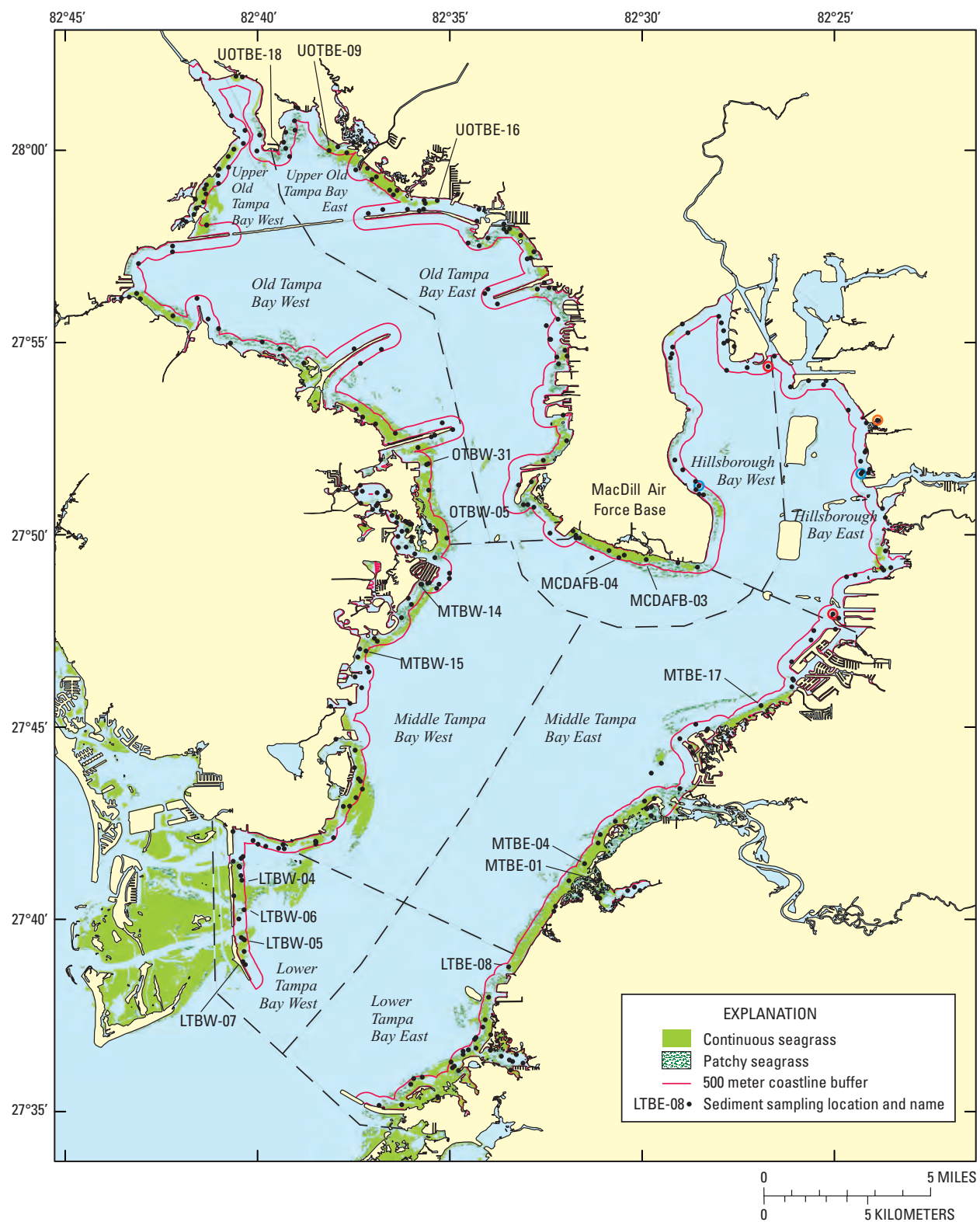
Box 7–2. Bioaccumulation of Select Metals in Seagrass Tissues

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As noted in Chapter 4, improvements in water quality and implementation of a N management strategy have resulted in the re-growth of seagrasses in Tampa Bay between 1982 and 2008. However, seagrass growth remains limited in some areas within Tampa Bay where N load targets are met and light availability is sufficient. These locations coincide with areas of increased concentrations of metal contaminants. Previous investigations (Nicolaidou and Nott, 1998; Campanella and others, 2001; Fourqurean and Cai, 2001; Macinnis-Ng and Ralph, 2002; Amado and others, 2004; Macinnis-Ng and Ralph, 2004a,b; Whelan and others, 2005) indicate that seagrasses may be influenced by the toxicity of sediment contaminants and the degree to which seagrasses translocate and accumulate contaminants in their vascular tissues. A relation between contaminant uptake and standing biomass could provide an important link between the concentration of contaminants in sediments and in seagrass tissue. Researchers at USGS performed a preliminary investigation on the relation between metal concentrations in sediments and in seagrass tissue from samples collected in Tampa Bay.

Brooks and Doyle (1991) observed that concentrations of metal contaminants in sediments are highest in Old Tampa Bay and west-central Hillsborough Bay. Subsequently, Zarbock and others (1996) concluded that the most contaminated sediments in Tampa Bay were located in upper and middle Hillsborough Bay, parts of Old Tampa Bay, Boca Ciega Bay, and western Middle Tampa Bay. Grabe (1999) found that about 1 percent of sediments in Tampa Bay were subnominal and had a high probability of being toxic, but evidence that metals directly affect seagrasses does not exist.

Seagrass samples and their surrounding sediments were collected from 15 different locations throughout Old Tampa Bay, Middle Tampa Bay, and Lower Tampa Bay (box 7–2, fig. 1). Seagrass samples were placed in plastic zip-lock bags, and stored at 4 °C. The seagrasses were identified to genus level then thoroughly soaked and rinsed with deionized water. Macroscopic epiphytes were removed from seagrass leaves using a PVC scraper. Samples were subsequently transferred to clean zip-lock bags and shipped in coolers maintained at 4 °C to the National Water Quality Laboratory in Denver, Colorado. Upon arrival at the laboratory, samples were rinsed with ASTM Type I water before digestion. Samples were subjected to microwave-assisted acid digestion EPA method 3052 (USEPA, 1996).



Box 7-2, Figure 1. Map showing location of sampling sites in Tampa Bay.

Extracts were analyzed by cold vapor atomic absorption spectrophotometry (mercury only), inductively coupled plasma-mass spectrometry, or inductively coupled plasma-atomic emission spectrometry (Hoffman, 1996).

Sediment samples were collected from the top 4 to 6 cm of bottom sediments with a Petite Ponar and were prepared for analysis by sieving through a 2-mm nylon sieve to remove large pieces of shell, detritus, and marine seagrass. Samples were placed in labeled bottles and stored in a -24 °C freezer. Prior to chemical analysis, the samples were air dried at room temperature, ground in a mortar and pestle, and dried at 103 °C. A 0.5-gram sample was digested using aqua regia and analyzed by inductively coupled plasma/mass spectroscopy (ICP/MS) Perkin Elmer SCIEX ELAN 6100. International certified reference materials USGS GXR-1, GXR-2, GXR-4, and GXR-6 were analyzed at the beginning and end of each batch of samples. Internal control standards were analyzed every 10 samples and a duplicate was run for every 10 samples. In addition to the internal quality control, 5 samples were analyzed in duplicate.

Results of these analyses (box 7–2, table 1) indicate that concentrations of nickel, copper, and zinc in seagrass tissues exceeded concentrations in surrounding sediment at all 15 sample locations. Concentrations of lead in seagrasses exceeded sediment concentrations in 13 locations, chromium in 12 locations, and arsenic in 6 locations, indicating bioaccumulation of metals at these sites. A total of 80 percent of the seagrass samples and 33 percent

Box 7–2, Table 1. Analysis of six trace elements of seagrass tissue and corresponding sediments, Tampa Bay, Florida, July, 2003.

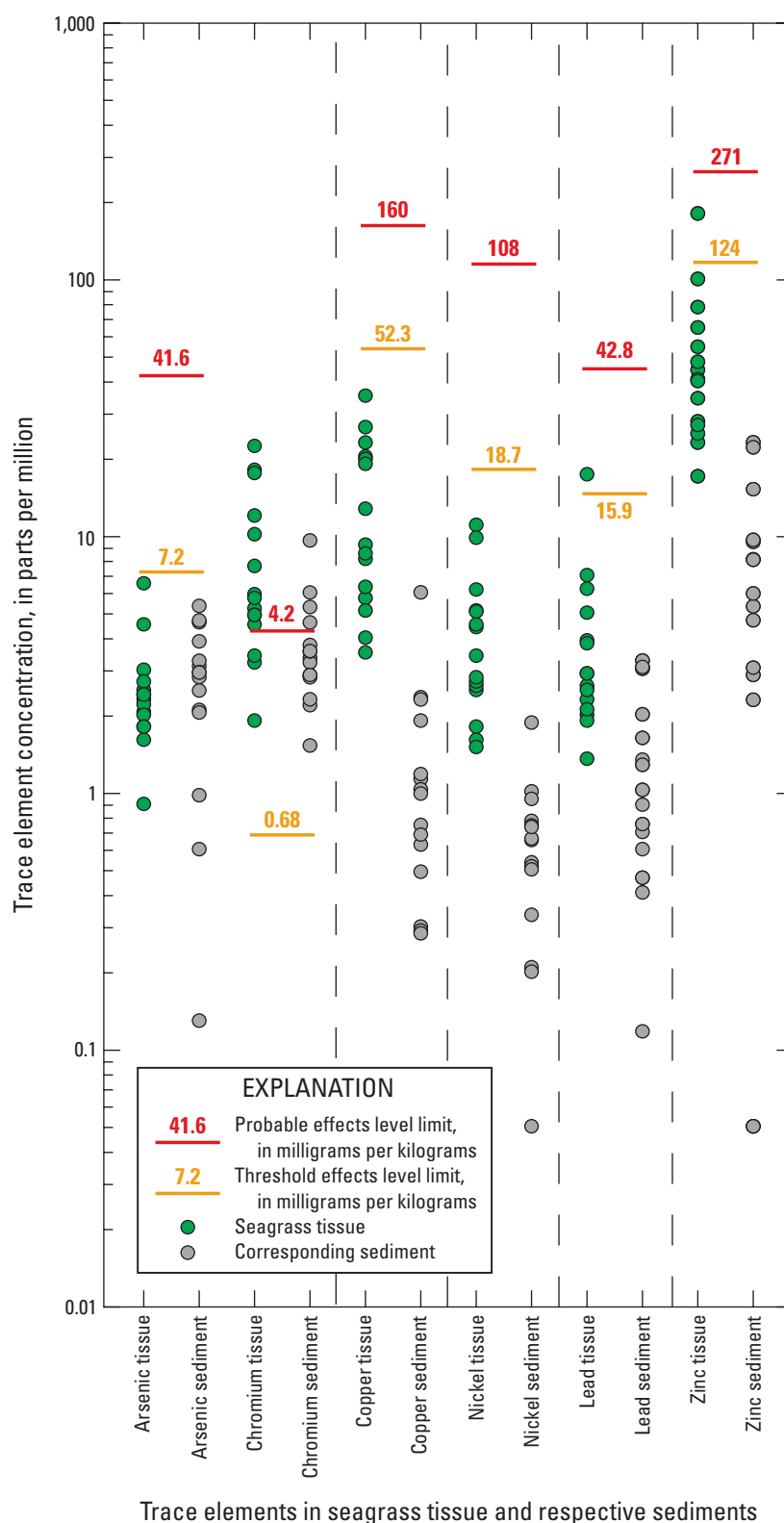
[Trace elements in parts per million; sediment trace element concentrations below detection limits are reported as 0.05 ppm]

Sediment site	Arsenic			Chromium			Copper			Nickel			Lead			Zinc		
	Tis-sue	Sedi-ment	Ratio	Tis-sue	Sedi-ment	Ratio	Tis-sue	Sedi-ment	Ratio	Tis-sue	Sedi-ment	Ratio	Tis-sue	Sedi-ment	Ratio	Tis-sue	Sedi-ment	Ratio
LTBE08	2.5	2.1	1.2	12.0	2.3	5.2	19.0	0.29	65.7	6.2	0.5	12.0	2.1	0.41	5.2	179.3	2.3	78.4
LTBW04	2.4	0.1	18.6	1.9	3.5	0.5	23.0	0.99	23.3	1.8	0.7	2.7	1.9	0.89	2.1	39.9	3.1	13.1
LTBW06	2.1	2.8	0.7	3.4	1.5	2.2	19.9	0.28	70.4	5.1	0.05	101.0	1.4	0.12	11.5	25.0	0.05	499.0
LTBW-05	4.5	3.1	1.5	4.5	3.2	1.4	4.0	0.3	13.3	4.5	0.2	22.5	2.5	0.7	3.6	26.9	9.6	2.8
MCDAFB-03	0.9	0.6	1.5	4.9	2.8	1.8	8.5	6	1.4	2.8	0.5	5.6	3.8	0.6	6.3	23.0	0.05	460.0
MTBE01	3.0	3.2	0.9	10.1	5.3	1.9	35.0	2.30	15.2	9.8	0.9	10.4	6.2	1.34	4.6	99.8	8.1	12.4
MTBE04	2.3	4.6	0.5	7.6	3.5	2.1	5.1	0.68	7.5	2.7	0.5	5.1	2.6	1.28	2.0	17.0	5.9	2.9
MTBE17	1.8	2.9	0.6	5.7	2.9	2.0	6.3	0.75	8.4	2.6	0.2	12.5	2.9	0.46	6.3	40.6	22.0	1.8
MTBW14	2.2	2.9	0.8	5.2	2.2	2.4	8.1	0.49	16.5	2.5	0.3	7.5	2.3	3.03	0.8	47.4	23.0	2.1
MTBW15	2.7	4.7	0.6	17.5	3.7	4.7	3.5	1.90	1.8	1.5	0.7	2.3	2.9	3.07	0.9	44.1	15.1	2.9
OTBW05	2.0	5.3	0.4	4.9	2.9	1.7	26.4	1.03	25.7	3.4	0.7	4.6	3.9	0.75	5.2	54.3	4.7	11.6
OTBW31	1.6	3.9	0.4	3.2	3.3	1.0	12.7	1.18	10.8	1.6	0.7	2.2	2.0	1.02	2.0	77.4	2.9	27.1
UOTBE09	2.4	2.0	1.2	18.0	6.0	3.0	20.3	1.13	18.0	5.1	1.0	5.1	7.0	2.01	3.5	64.5	9.4	6.8
UOTBE16	6.5	1.0	6.7	5.9	9.6	0.6	9.2	2.34	3.9	11.0	1.9	5.9	17.3	3.25	5.3	34.2	8.0	4.3
UOTBE18	1.8	2.5	0.7	22.3	4.6	4.9	5.7	0.63	9.1	4.4	0.8	5.7	5.0	1.63	3.1	27.8	5.3	5.3

of the sediment samples exceeded PEL limits for chromium. Highest concentrations of chromium were found in samples from upper Old Tampa Bay and Middle Tampa Bay. Only one seagrass sample from upper Old Tampa Bay and one sample from Lower Tampa Bay east exceeded TEL limits for lead and zinc, respectively (box 7-2, fig. 2). Highest concentrations of lead were found in samples from upper Old Tampa Bay, whereas highest concentrations of zinc were found in Lower Tampa Bay. A Pierson's correlation analysis of metal concentrations in tissues versus sediments for the six metals listed in table 1 indicates a significant correlation between seagrass and sediment concentrations only for lead and nickel. However, results of this analysis may be affected by age of seagrass tissue, which was unaccounted for in this preliminary investigation.

These results indicate that a more in-depth study on the impact of metal contaminants to seagrass growth is warranted. Concentration of metals in seagrass tissue, and the subsequent deterioration of dead seagrass, provide a mechanism for remobilization and transport of contaminant metals throughout Tampa Bay. Additionally, bioaccumulation of metals in seagrass provides a mechanism for direct transport of metal contaminants to higher trophic level animals feeding on seagrass.

Box 7-2, Figure 2. Concentrations of selected trace metals in seagrass tissues and their corresponding sediments from 15 different sites throughout Tampa Bay (see box 7-2, table 1).



Contaminant Concentrations and Distribution

Since the mid-1980s, a combination of short-term studies and long-term monitoring programs has assessed the levels and distribution of contaminants in bay sediments (Brooks and Doyle, 1991; Long and others, 1991, 1994; Carr and others, 1996; McCain and others, 1996; Grabe and Barron, 2002; Grabe and others, 2002; Karlen and others, 2008). A brief synopsis of those studies is provided in table 7–1. Several studies have used the “sediment-quality triad” approach, which provides simultaneously collected information on sediment chemistry, sediment toxicity, and benthic faunal composition to develop a weight-of-evidence characterization of sediment-quality conditions at each monitoring site (Long and Chapman, 1985; Zarbock and others, 1996). Due to resource limitations in other studies, only two components of the triad — usually sediment chemistry and benthic taxonomy data — were collected (table 7–1) (Zarbock and others, 1996).

Long and Greening (1999) compiled the sediment-quality information that had been provided by these studies through 1997, and compared the contaminant concentrations observed in Tampa Bay against nationally developed sediment-quality guidelines to characterize the locations and sizes of contaminated sites, estimate the potential for negative biological effects within those sites, and compare sediment-quality conditions in the bay with conditions in highly urbanized estuaries in other geographic areas in Florida and the United States. Based on these analyses, Long and Greening (1999) identified several “hot spots” (fig. 7–4) where concentrations of one or more toxicants exceeded national sediment-quality guidelines. Although the bay did not appear to be as heavily contaminated as some other urban estuaries,

Table 7–1. Chronology of Tampa Bay sediment assessment and management activities.

[NOAA, National Oceanic and Atmospheric Administration; USEPA, U.S. Environmental Protection Agency; TBEP, Tampa Bay Estuary Program. From Long and Greening, 1999; and H. Greening, unpublished]

Year	Activity
1986	University of South Florida sediment quality surveys (Brooks and Doyle, 1991)
1990	NOAA Mussel Watch data summary; fish bioeffects monitoring initiated by NOAA
1991	NOAA sediment quality and NOAA/USEPA oyster bioeffects surveys initiated
1993	FDEP initiates development of State sediment quality guidelines; annual benthic surveys initiated by TBEP partners
1994	FDEP publishes State sediment quality guidelines (MacDonald, 1994); results of NOAA sediment quality surveys published (Long and others, 1994)
1995	TBEP sources and loadings report published (Frithsen and others, 1995) TBEP triad report (Zarbock and others, 1996), risk assessment report (McConnell and others, 1996), and source inventory reports published (McConnell and others, 1996); NOAA fish bioeffects report (McCain and others, 1996), and Carr and others (1996) sediment quality report published
1999	Joint NOAA/TBEP summary report on sediment quality published (Long and Greening, 1999)
2004–2005	Tampa Bay benthic index developed (Malloy and others, 2007)
2007	Priority areas identified for development of sediment quality action plans
2008	Work begun on initial action plans

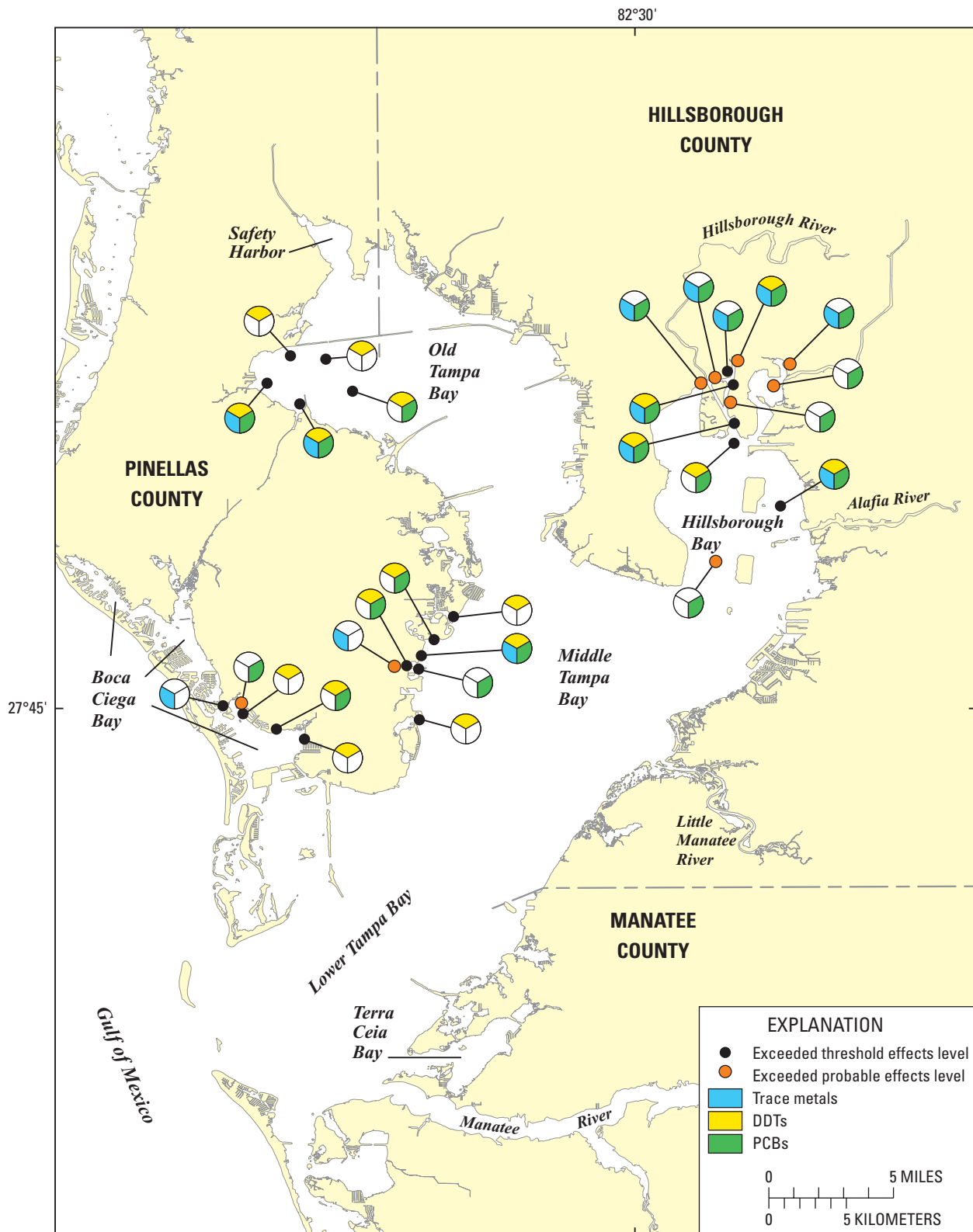


Figure 7-4. Sampling sites designated as “hot spots” when chemical concentrations exceeded at least one threshold effects level or probable effects level guideline value. From Long and Greening (1999); data from Zarbock and others (1996).

a number of contaminant hot spots were found immediately offshore from the City of Tampa in Hillsborough Bay and McKay Bay, near the City of Clearwater in Old Tampa Bay, and near the City of St. Petersburg in Middle Tampa Bay and Boca Ciega Bay (fig. 7–4). One type of toxicity test conducted by Long and others (1994), whereby sea urchin egg fertilization rates were measured following exposure to sediment pore water from a number of sites in Tampa Bay, indicated that moderate levels of toxicity may be widespread in the bay. Other tests carried out at the same time did not corroborate this result, however, and Tampa Bay sediments are not generally considered to be severely contaminated (Long and Greening, 1999).

Identification of Contaminants of Concern

The sediment chemistry analyses summarized by Long and others (1991, 1994) and Long and Greening (1999) indicated that mixtures of a number of different toxicants were present in most of the highly and moderately contaminated areas of the bay, and that these compounds were likely acting together to induce toxicity and other adverse biological effects. The presence of elevated contaminant levels in tissues of oysters and fish, and the detection of significant toxicity levels in laboratory bioassays (using bulk sediments and sediment pore water), indicated that toxicants were bio-available in the sediments (fig. 7–5). Observations of acute and sublethal toxicity in bioassays (Long and others, 1994), and histological responses in the tissues of field-collected fish (McCain and others, 1996), provided additional indications that the contaminants were capable of impacting local biological resources (Long and Greening, 1999).

To identify the specific contaminants that were most likely to produce adverse biological effects, ecological and human health risk assessments were conducted (McConnell and others, 1996). The ecological risk assessment evaluated potential impacts from direct and/or food web exposure for several benthic organisms (representative deposit feeders, filter feeders, omnivores, and carnivores), and for aquatic and terrestrial piscivores (McConnell and others, 1996). The human health risk assessment evaluated potential impacts to recreational fishermen consuming fish containing elevated levels of contaminants with a known potential for bioaccumulation (McConnell and others, 1996). Through these analyses, toxicants in four major groups were identified as COC for Tampa Bay, due to their presence at elevated concentrations in sediments in one or more areas of the bay, their presence at sites whose sediments showed toxicity to laboratory test organisms, and the potential biological risks they posed at the concentrations observed in the bay (Long and others, 1994; McConnell and others, 1996; Long and Greening, 1999). The COC identified through this approach included:

- Metals (cadmium, chromium, copper, lead, mercury, and zinc);
- Pesticides (chlordane, dieldrin, DDTs, endosulfan, and mirex);
- PCBs; and
- Polycyclic aromatic hydrocarbons (PAHs).

The potential ecological and human health risks posed by these contaminants are summarized in table 7–2.

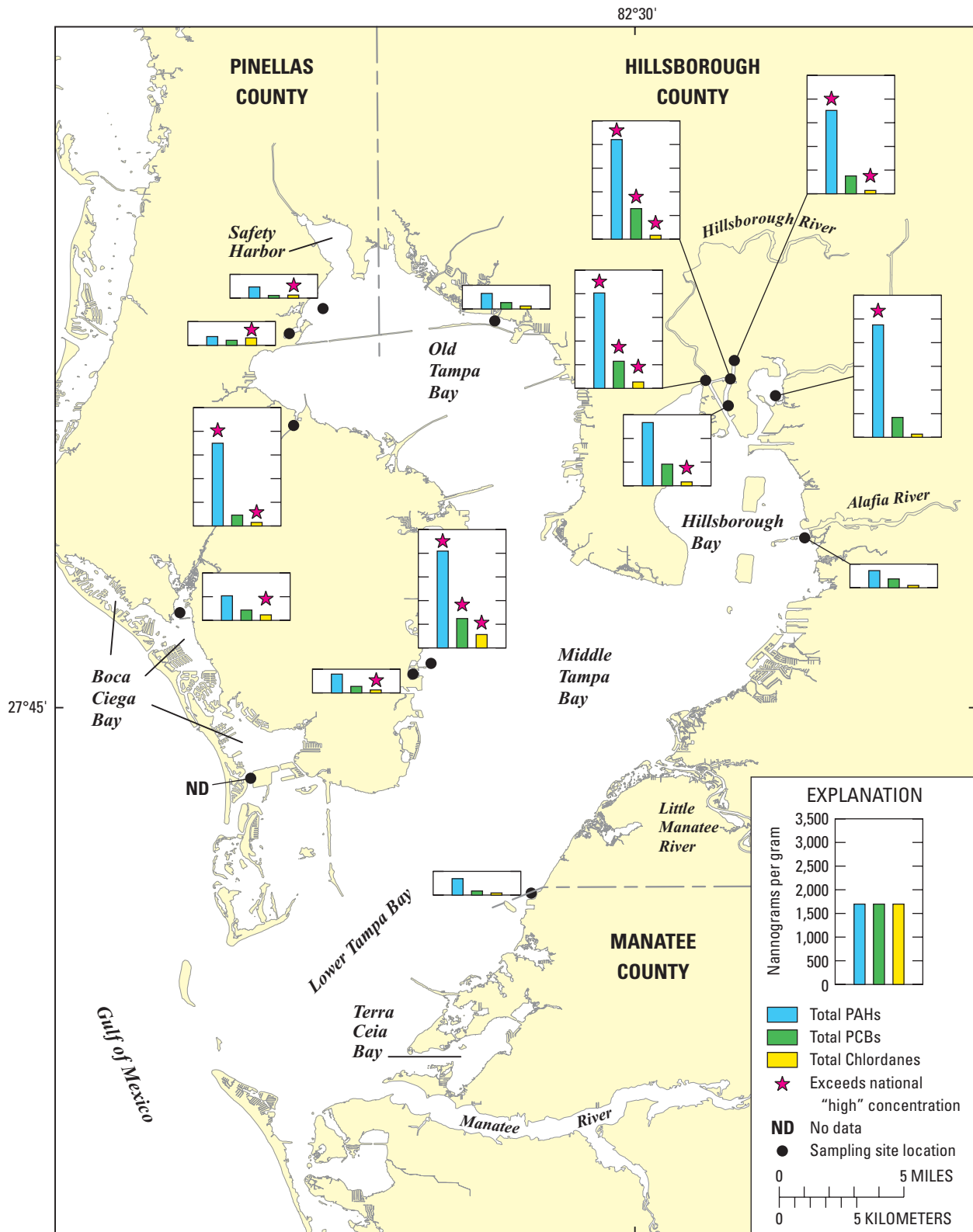


Figure 7-5. Concentrations of most organic toxins were highest among resident oysters collected in northern Hillsborough Bay and lowest in oysters from Old Tampa Bay and Lower Tampa Bay. From Long and Greening (1999); data from U.S. Environmental Protection Agency.

Table 7–2. Potential biological and human health effects of Tampa Bay contaminants of concerns.

[PAHs, polycyclic aromatic hydrocarbons; PCBs, polychlorinated biphenyl. From Long and Greening, 1999]

Contaminants of concern	Biological effects
Cadmium	Acute toxicity
Chromium	Acute toxicity; mammalian carcinogen
Copper	Acute toxicity
Lead (banned in U.S. gasoline)	Acute toxicity; chronic effects; human health hazard
Mercury	Acute toxicity, bioaccumulation in biota, behavioral toxin, growth and development effects
Zinc	Acute toxicity
Chlordane (banned)	Acute toxicity
Dieldrin (banned)	Liver damage, immune suppression, decreased fertility, carcinogen/mutagen
DDT (banned)	Animal and human carcinogen, impairment of bird reproduction, bioaccumulate
Endosulfan	Acute toxicity, bioaccumulate
Mirex (banned)	Acute toxicity, growth/development reduction, reduced bird reproduction
PAHs	Acute toxicity, carcinogenic/mutagenic
PCBs (banned)	Acute toxicity, carcinogenic, bioaccumulate

Risk-Based Assessment of Contaminant Concentrations

Although contaminant concentrations are measured in many sediment monitoring programs, interpretation of the resulting data is often difficult (Wenning and Ingersoll, 2002). Contaminated sites usually contain a mixture of chemicals, and the biological availability of potentially toxic compounds in estuarine sediments — either when alone or present in mixtures — is poorly understood. To counter these problems, a weight-of-evidence approach, initially recommended by Long and Morgan (1991), has been widely used to develop sediment-quality guidelines for evaluating the potential ecological and human health risks that may be associated with the COC concentrations observed at a given monitoring site. In Florida, this approach has been used by the FDEP to develop sediment-quality guidelines for the State's coastal waters (MacDonald and others, 1996). These guidelines have been adopted by the TBEP and its partners for assessment of sediment quality in Tampa Bay (MacDonald and others, 2004).

Long and Morgan (1991) developed their weight-of-evidence approach as an informal tool for analyzing coastal sediment chemistry data collected through a national coastal monitoring network — NOAA's National Status and Trends Program. They compiled a database containing information generated by the program and other sediment-quality surveys, using three analytical approaches:

- Equilibrium partitioning, which assumes that the distribution of contaminants between different compartments of the sediment matrix, for example sediment solids versus interstitial pore water, is predictable based on the physical and chemical

properties of the contaminants. This approach has been supported by the results of sediment toxicity tests, which indicate that positive correlations exist between the biological effects observed and the concentrations of nonpolar organic contaminants measured in the interstitial water.

- Spiked-sediment toxicity bioassays, a laboratory-based method whereby clean sediments are spiked with known concentrations of contaminants (individually or in mixtures) to determine causal relationships between chemical concentrations and biological responses, such as mortality, reductions in growth or reproduction, or other physiological changes. This method has been used to provide toxicity information for various types of sediments, usually involving single contaminants or relatively simple mixtures of contaminants.
- Effects-based approaches, whereby simultaneously collected sediment chemistry and biological data are combined to identify ranges of concentrations at which individual contaminants cause observable biological effects. The effects may include toxicity to benthic or water column species (measured using laboratory bioassays) or changes in the abundance, diversity, or taxonomic composition of the benthic invertebrate fauna (measured using field sampling methods).

MacDonald and others (1996) constructed an expanded version of the Long and Morgan (1991) database, enlarging it with data from a number of additional studies performed in the coastal waters of Florida and other regions of the southeastern United States. For each contaminant, the observations in the expanded database were divided into two sets — an “effects dataset” (EDS), and a “no-effects dataset” (NEDS) — whereby toxicity effects were either observed or not observed, respectively.

The range of sediment contaminant concentrations that are not likely to be associated with adverse biological effects on aquatic organisms, designated the “minimal effects” range, was defined using a two-step process. First, a threshold effects level (TEL) was calculated to estimate the upper limit of the range of contaminant concentrations at which adverse biological effects were observed. For a given contaminant, the TEL was calculated as the geometric mean of two values, the 15th percentile concentration in the EDS (denoted EDS-L) and the median value in the NEDS (denoted NEDS-M):

$$TEL = \sqrt{(EDS - L) * (NEDS - M)}$$

When present at concentrations of less than the TEL value, sediment-associated contaminants are not considered to represent significant hazards to aquatic organisms (MacDonald, 1994; Long and MacDonald, 1998).

For each contaminant, a probable effects level (PEL) was also calculated to estimate the lower limit of the range of contaminant concentrations at which adverse biological effects are likely to be observed. For a given contaminant, the PEL was calculated as the geometric mean of the median concentration in the EDS (denoted EDS-M) and the 85th percentile concentration in the NEDS (NEDS-H):

$$PEL = \sqrt{(EDS - M) * (NEDS - H)}$$

When present at concentrations greater than the PEL value, sediment-associated contaminants are considered likely to produce adverse biological impacts (MacDonald, 1994).

At concentrations between the TEL and PEL values, adverse biological effects are possible, but it is difficult to predict the occurrence or severity of those effects. When contaminant concentrations fall within this range, further site-specific investigations are typically needed to determine if sediment-associated contaminants at the site represent significant hazards to aquatic organisms. Such investigations could focus initially on identifying the probable sources of the contaminant, possibly through the use of the FDEP metals interpretive tool (Schropp and others, 1990), and then on determining the toxicity of in-situ sediments, possibly using toxicity bioassessment techniques.

MacDonald and others (1996) emphasized that sediment-quality guidelines, which were developed using these methods, address acute toxicity only and do not address the potential for bioaccumulation of persistent toxic chemicals or their potential adverse effects on higher trophic levels of the food web. Areas of the bay where these TELs and PELs have been exceeded for one or more contaminants are summarized in figure 7–6.

Contaminants of Concern Sources and Estimated Inputs

The spatial distribution of sites containing elevated sediment contaminant levels and the mixtures of toxicants found at those sites suggest that the COC present in Tampa Bay have originated from a combination of localized point sources and diffuse nonpoint sources (Long and Greening, 1999). Frithsen and others (1995) and McConnell and Brink (1997) used sediment contaminant data compiled from previous monitoring studies and regulatory agency permitting files to identify potential sources and estimate annual influxes of these chemicals to different parts of the bay (fig. 7–7). Because of limitations in the available point source discharge data, and uncertainties associated with calculations of nonpoint source loadings, influx values generated by these studies should be viewed as initial order-of-magnitude estimates (Frithsen and others, 1995). As an additional caveat, the atmospheric deposition estimates provided in these studies address only the quantities of the various toxicants that are deposited directly on the bay surface. Many of the Tampa Bay COC can be transported considerable distances via the atmosphere and then deposited on land. Some of this deposited material can then be mobilized during rain events and carried to surface-waterbodies in stormwater runoff. As a result, in terms of identifying original contaminant sources, the relative importance of atmospheric deposition is probably underestimated because in many of these initial estimates the atmospherically deposited contaminants are not recognized as such in the stormwater runoff. With these cautions in mind, Frithsen and others (1995) provided the following COC loading estimates on a bay-wide basis.

Cadmium — About 7,700 lbs of cadmium was estimated to enter the bay each year as a result of atmospheric deposition (46 percent), permitted domestic and industrial point sources (32 percent), and urban stormwater runoff (21 percent). No estimates could be provided for agricultural runoff. The largest percentage of the estimated load was discharged to Hillsborough Bay (39 percent) and Old Tampa Bay (23 percent).

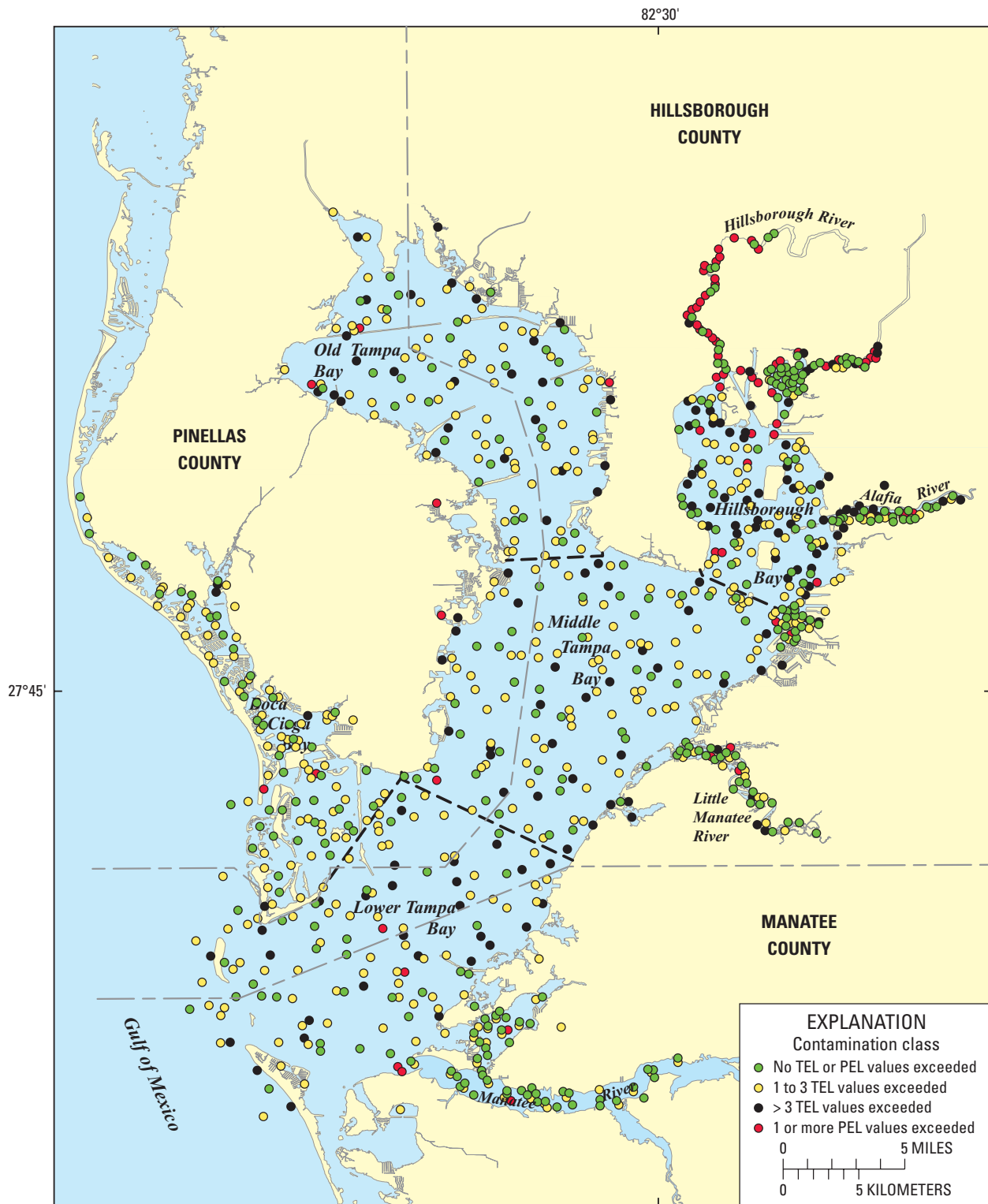


Figure 7-6. Threshold effects level and probable effects level exceedances in Tampa Bay sediments. From Janicki Environmental, Inc. (2002).

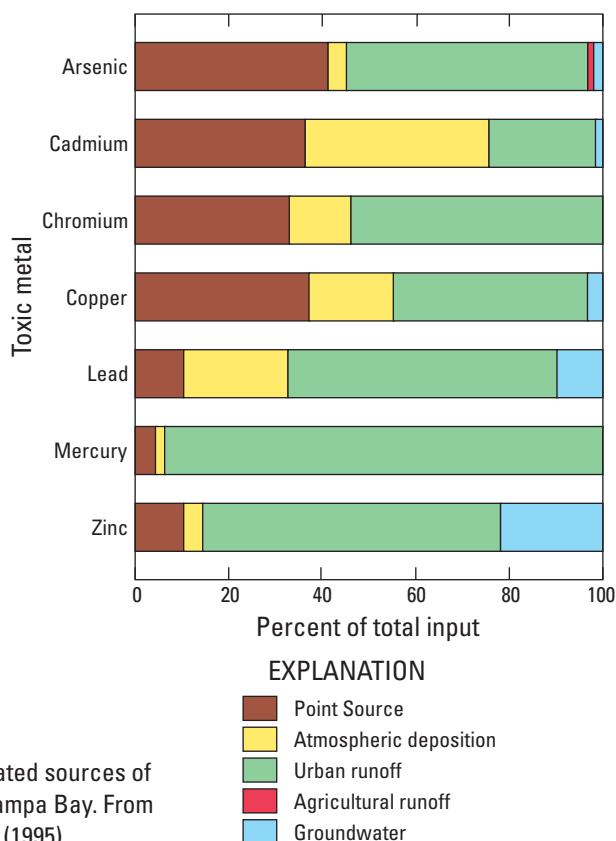


Figure 7-7. Estimated sources of metal loadings to Tampa Bay. From Frithsen and others (1995).

Chromium — Estimated annual load was 32,200 lbs, primarily via urban runoff (57 percent), permitted domestic/industrial wastewater facilities (29 percent), and atmospheric deposition (13 percent). Loads from agricultural runoff could not be estimated. The largest amounts of chromium were discharged to Hillsborough Bay (44 percent) and Old Tampa Bay (25 percent), but relatively large amounts were also discharged to Middle Tampa Bay (11 percent), Boca Ciega Bay (9 percent), and Manatee River (8 percent).

Copper — The annual load of copper was estimated at 27,500 lbs, and entered primarily via urban stormwater runoff (43 percent), permitted domestic wastewater discharges (35 percent), and atmospheric deposition (18 percent). Portions from agricultural runoff could not be estimated. The majority was discharged to Hillsborough Bay (39 percent) and Old Tampa Bay (23 percent).

Lead — An estimated 109,533 lbs annual load of lead entered the Bay via urban stormwater runoff (60 percent), atmospheric deposition (12 percent), domestic and industrial wastewater discharges (11 percent), and groundwater (9 percent). Portions from agricultural runoff could not be estimated. The majority was discharged to Hillsborough Bay (38 percent), Old Tampa Bay (20 percent), Middle Tampa Bay (12 percent), Boca Ciega Bay (10 percent), and Manatee River (7 percent).

Mercury — Due to its importance as a water-quality management issue, a more detailed discussion of mercury is provided in Chapter 5. Frithsen and others (1995) estimated an annual Hg load to Tampa Bay of about 1,300 lbs, primarily via urban stormwater runoff (95 percent).

As noted in Chapter 5, however, the global mercury cycle includes a large atmospheric component, and the bulk of the mercury entering Tampa Bay via stormwater runoff is thought to be deposited on the watershed as a result of atmospheric wetfall or dryfall.

Zinc — The estimated annual load of zinc was about 332,000 lbs, entering primarily via urban stormwater (66 percent) and groundwater (20 percent). Bay segments with a highly urbanized watershed (Hillsborough Bay, Old Tampa Bay, Middle Tampa Bay, and Boca Ciega Bay) received the largest estimated loads.

Chlordane — Current loads of this highly restricted chlorinated pesticide presumably reflect its earlier, more widespread use as an agricultural pesticide and urban termiticide. Estimated annual loads were about 2,100 lbs, primarily from agricultural (78 percent) and urban (21 percent) stormwater runoff.

DDT — As with chlordane, loads of this currently banned pesticide presumably reflect its widespread use in earlier decades. Current loads were estimated at about 3,400 lbs/yr primarily via agricultural runoff (95 percent). No estimate could be determined for point sources or groundwater.

Dieldrin — Estimated annual loads were about 1,600 lbs/yr, almost all (99 percent) via agricultural runoff.

Endosulfan — Estimated annual loads were about 1,000 lbs/yr, almost all (95 percent) via agricultural runoff.

PCBs — Atmospheric deposition was estimated to generate loads of about 23 lbs/yr, but was the only source category for which estimates could be calculated because of lack of data.

PAHs — As with PCBs, estimates of annual loads of PAHs (23 lbs/yr) could be calculated for atmospheric sources only, because of lack of data.

The sources and locations estimated by Frithsen and others (1995) are generally consistent with the spatial distribution of sediment contaminant hot spots that have been found in the bay, which tend to be located in low-energy areas, near heavily urbanized stormwater drainage systems and industrial and municipal wastewater discharges.

McConnell and Brink (1997) developed more detailed estimates of COC sources and loadings for hydrologic subbasins that discharge to Hillsborough Bay and Boca Ciega Bay, two areas that had been identified as containing contaminant hot spots in a number of earlier studies (Long and others, 1991, 1994). Priority subbasins draining to Hillsborough Bay included:

Delaney Creek subbasin for point sources of chromium, copper, and nickel, and stormwater sources of metals and pesticides;

- Two lower Hillsborough River subbasins for stormwater sources of metals, PAHs, and pesticides;
- Ybor Channel subbasin for point sources of copper and nickel and stormwater sources of metals; and
- Tampa Bypass Canal subbasin for stormwater sources of metals and pesticides.

Priority subbasins discharging COC to Boca Ciega Bay included Long Bayou, Joe's Creek, Cross Bayou Canal, Lake Seminole, and an unnamed subbasin that discharged directly to the bay (McConnell and Brink, 1997). COC for Boca Ciega Bay were limited to PAHs and PCBs, whose primary sources were identified as stormwater runoff and atmospheric deposition (McConnell and Brink, 1997).

The Tampa Bay Sediment Quality Management Strategy

To provide a consistent bay-wide management framework for identifying and addressing the sources and effects of sediment contamination, and linking the management effort to the protection of the bay's living resources, the TBEP and its partners have adopted two major resource-based goals:

- To provide sediment-quality conditions in Tampa Bay that protect or, where necessary, restore the native benthic faunal community; and
- To maintain and, where necessary, restore sediment-quality conditions to levels adequate to ensure that fish and other aquatic organisms are safe for consumption by humans and wildlife.

In a more abbreviated form, these goals have been restated (TBEP, 2006) in the following way:

- To reduce the amount of toxic chemicals in contaminated bay sediments and protect relatively clean areas of the bay from contamination.

To meet these goals, the sediment-quality guidelines developed by MacDonald and others (1996) for Florida coastal waters, along with other biological, physical, and water-quality information, have been used to characterize sediment-quality and benthic-habitat conditions in Tampa Bay, and to classify sites according to their need for protection or restoration.

Ideally, such a characterization would be based on the widely used “sediment-quality triad” approach, which combines information collected from simultaneous sediment chemistry, sediment toxicity, and benthic community surveys (Long and Chapman, 1985). Due to resource limitations, however, few sediment toxicity studies have been performed in the bay area (Zarbock and others, 1996). As a result, the Tampa Bay sediment characterization process has relied primarily on two components of the triad — sediment chemistry and benthic community structure data — with an implicit assumption that a given level of chemical contamination will produce levels of toxicity in Tampa Bay that are comparable to toxicity levels that have been documented at those contaminant levels in other geographic areas.

In an initial application of this approach, definitions of “healthy” and “degraded” benthic sites were developed for the bay based on known sediment contamination levels and bottom-water DO concentrations (Wade and others, 2005). “Healthy” benthic sites were defined as those exhibiting mid-day bottom DO concentrations of >4.5 mg/L, with no exceedances of TEL or PEL contaminant levels. “Degraded” conditions were defined as either having mid-day bottom DO concentrations of <2.5 mg/L or having a PEL exceedance for one or more contaminant(s). DO was included as an additional variable because low DO concentrations have well-documented negative impacts on the benthic community (Gray and others, 2002) and are often associated with anthropogenic pollutant loadings or bathymetric alterations in Tampa Bay (Santos and Simon, 1980; Johansson and Squires, 1989).

This classification method was applied to 578 benthic sites that were sampled under the Tampa Bay Benthic Monitoring Program from 1993 through 2000 (Wade and others, 2005). Among these sites, 52 percent were designated as “healthy,” 13 percent as “degraded,” and the remaining 35 percent either fell in an intermediate sediment-quality range or received no designation due to missing data.

With respect to the physical attributes of salinity and sediment grain size, sites classified as healthy or degraded were located across the observed range of salinity conditions. Sites classified as degraded were also located across the range of grain size (percent silt-clay) conditions, but sites classified as healthy were located primarily in sandy (large grain size) sediments (Wade and others, 2005 (fig. 7–8).

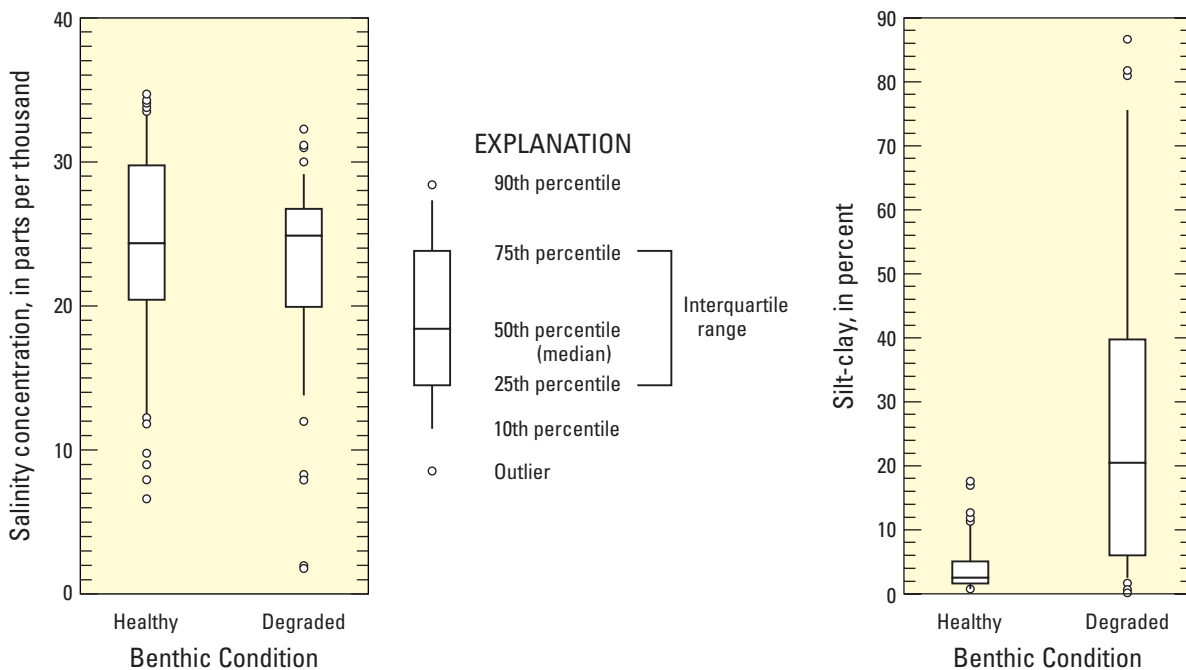


Figure 7–8. Distribution of “healthy” and “degraded” benthic habitats across salinity and sediment grain size in Tampa Bay. From Wade and others (2005) and Malloy and others (2007).

An empirical, statistically based Tampa Bay Benthic Index (TBBI) was then developed based on benthic community characteristics observed at locations classified as “healthy” and “degraded” (Wade and others, 2005; Malloy and others, 2007). Because salinity gradients can confound interpretation of the effects of sediment contaminants on benthic community structure, benthic community metrics, such as species richness and diversity, were corrected for salinity effects by expressing each metric value as the proportion of its expected value based on the salinity level measured at the time the sample was collected. A set of three benthic indicators was selected from an initial list of potential candidates using stepwise discriminant analysis; linear discriminant analysis was used to estimate the coefficients for each of the three variables that would define the TBBI (Wade and others, 2005; Malloy and others, 2007). The selected index was defined as:

$$\text{TBBI} = -0.11407 + 0.32583 \times \text{proportion of expected number of species} \\ -0.15020 \times \text{proportion of total abundance as spionid polychaetes} \\ -0.60943 \times \text{proportion of total abundance as capitellid polychaetes}.$$

The index is scaled from 0 to 100. In order to use the index scores in a resource-management framework, such as for assessing benthic-habitat quality at individual sediment monitoring sites, specific ranges of index values representing “healthy” and “degraded” conditions must be defined. Four potential approaches were evaluated for selecting these ranges:

- A percentile-based approach, whereby index values lower than a given level, for example the 10th percentile, would be designated as “degraded,” and those higher than a given level, for example the 90th percentile, would be designated as “healthy;”
- A regulatory approach, whereby a site whose observed number of species is less than 75 percent of the salinity-based expected value is identified as “degraded;”
- An approach based on the mixture of contaminants observed at a site, which was evaluated using mean PEL quotients following methods described by MacDonald and others (2004); and
- An ecological approach that defines “healthy” and “degraded” conditions by comparison to a reference pool of species, following the methods of Smith and others (2001).

These four approaches identified similar ranges of index values as defining “healthy” and “degraded” sites, providing a strong weight-of-evidence result. Based on this evaluation, index values >87 were defined as “healthy” and values <73 were defined as “degraded.” Among their other attributes, these values maximize the ability of the index to identify truly degraded sites accurately and limit the rate of false positives and negatives to 10 percent (fig. 7–9) (Malloy and others, 2007).

This approach, like the method for characterizing sediment contaminant levels based on TEL and PEL values, puts its greatest emphasis on defining clearly “healthy” and clearly “degraded” benthic-habitat conditions. In the case of the TBBI, sites with index scores ranging from 73 to 86 are simply classified as “intermediate” with respect to benthic-habitat quality. Application of the TBBI for three time periods (1993–1996, 1997–2000, and 2001–2004) is shown in figure 7–10.

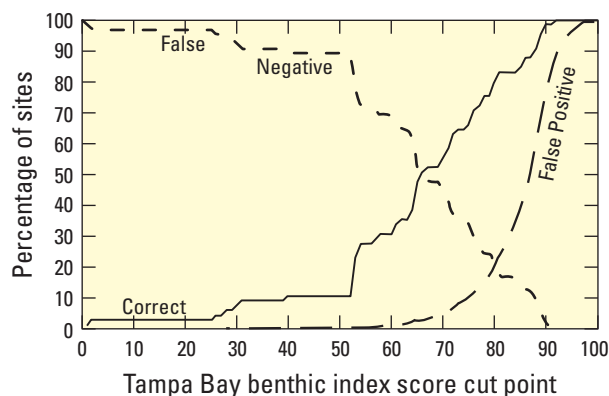


Figure 7–9. Cumulative distribution frequency plot showing the relation between percentage of sites identified correctly as degraded, false negatives and false positives, and the relation between the Tampa Bay Benthic Index score. From Wade and others (2005) and Malloy and others (2007).

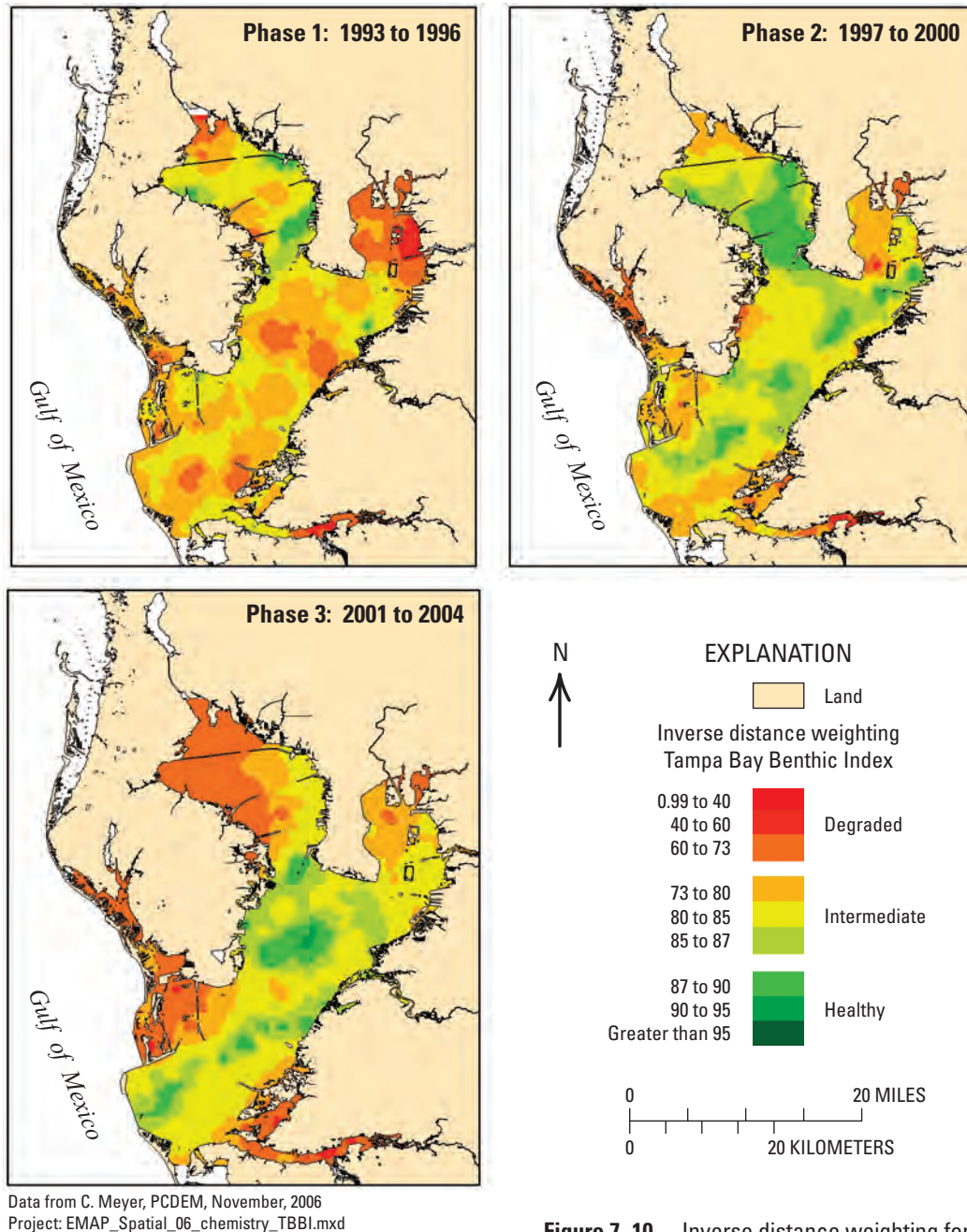


Figure 7-10. Inverse distance weighting for the Tampa Bay Benthic Index for three time periods: 1993–1996, 1997–2000, and 2001–2007. From Karlen and others (2008).

Benthic Diversity and Abundance

Karlen and others (2008) provide a summary of benthic habitat quality and taxonomic diversity data collected through the Tampa Bay benthic monitoring program from 1993 through 2004. The monitoring program design, which is based on sampling conducted during a later summer/early fall “index period” to be consistent with regional monitoring programs conducted by the U.S. Environmental Protection Agency (USEPA) (Engle and others, 1994; Engle and Summers, 1999), is summarized by Courtney and others (1993, 1995). With respect to species richness, the overall median

number of taxa per sample during the period was 35, and the number ranged from 0 to 125. There was a general trend of increasing species richness toward the mouth of the bay, with the highest numbers of taxa being recorded in Lower Tampa Bay and Boca Ciega Bay. The abundance of benthic organisms ranged from 0 to 183,400 organisms per square meter, with a median of 5,950 organisms per square meter (tables 7–3 and 7–4). Among the seven bay segments, Middle Tampa Bay and Old Tampa Bay had the highest median abundances, and the lowest median abundance was in Terra Ceia Bay (table 7–4). The median Shannon Diversity Index was 2.49, and ranged from 0 to 3.94 (tables 7–3 and 7–4). Diversity increased toward the lower bay and was highest in Boca Ciega Bay, Terra Ceia Bay, and Lower Tampa Bay, with no statistical differences among these segments. The lowest median diversity values were in Hillsborough Bay and Manatee River (table 7–4) (Karlen and others, 2008).

Table 7–3. Benthic community summary statistics by year.

[Min, minimum; Max, maximum. From Karlen and others, 2008]

Year	Number of organisms	Number of taxa Median		Number per square meter Median		Shannon Diversity Index Median		Tampa Bay Benthic Index Median	
		Min.	Max.	Min.	Max.	Min.	Max.	Min.	Max.
1993	90	39		7,763		2.64		78.67	
		5	86	250	45,725	0.66	3.52	0.99	93.59
1994	90	32		5,638		2.51		75.65	
		0	74	0	27,825	0.00	3.41	0.00	93.68
1995	134	33		5,475		2.49		82.12	
		0	99	0	183,400	0.00	3.94	0.00	97.83
1996	132	36		7,250		2.42		83.14	
		0	74	0	91,625	0.00	3.66	0.00	96.19
Phase 1 1993–1996	446	35		6,263		2.5		80.96	
		0	99	0	183,400	0.00	3.94	0.00	97.83
1997	123	41		7,175		2.54		82.68	
		0	92	0	49,475	0.00	3.70	0.00	97.24
1998	119	30		3,264		2.55		81.31	
		0	89	0	44,575	0.00	3.57	0.00	96.40
1999	124	36		6,450		2.47		83.41	
		0	120	0	54,175	0.00	3.79	0.00	98.72
2000	86	37		7,663		2.64		84.39	
		2	86	50	43,925	0.69	3.61	13.78	92.93
Phase 2 1997–2000	452	35		6,076		2.53		83.03	
		0	120	0	54,175	0.00	3.79	0.00	98.72
2001	80	31		3,750		2.53		79.77	
		0	88	0	21,675	0.00	3.61	0.00	93.07
2002	83	38		5,850		2.54		81.75	
		0	125	0	97,075	0.00	3.63	0.00	94.83
2003	78	27		4,113		2.40		77.71	
		0	86	0	50,376	0.00	3.58	0.00	96.48
2004	77	36		8,725		2.33		85.55	
		2	101	50	61,125	0.51	3.48	42.95	97.44
Phase 2 2001–2004	318	33		4,938		2.43		80.18	
		0	125	0	97,075	0.00	3.63	0.00	97.44
Cumulative 1993–2004	1,216	35		5,950		2.49		81.41	
		0	125	0	183,400	0.00	3.94	0.00	98.72

Table 7–4. Benthic community summary statistics bay segment.

[Min, minimum; Max, maximum. From Karlen and others, 2008]

Segment	Number of organisms	Number of taxa Median		Number per square meter Median		Shannon Diversity Index Median		Tampa Bay Benthic Index Median	
		Min.	Max.	Min.	Max.	Min.	Max.	Min.	Max.
Hillsborough Bay	289	25		4,750		2.17		78.62	
		0	66	0	53,825	0.00	3.25	0.00	97.44
Old Tampa	166	35		7,575		2.43		84.86	
		0	69	0	183,400	0.00	3.44	0.00	96.40
Middle Tampa Bay	238	38		7,750		2.53		84.82	
		2	125	50	97,075	0.22	3.79	7.53	96.36
Lower Tampa Bay	182	44		6,100		2.85		83.90	
		2	101	50	54,175	0.68	3.79	24.51	97.83
Mantee River	119	25		5,575		2.29		78.15	
		1	74	300	91,625	0.00	3.51	5.20	95.89
Terra Ceia Bay	68	36		4,025		2.93		77.27	
		1	86	25	17,525	0.00	3.56	26.60	96.19
Boca Ciega Bay	154	42		4,563		2.96		75.33	
		0	120	0	61,125	0.00	3.94	0.00	98.72
Tampa Bay	1,216	35		5,950		2.49		81.41	
		0	125	0	183,400	0.00	3.94	0.00	98.72

About 1,500 benthic taxa were identified from the monitoring program. However, seven taxa represented 25 percent of the overall abundance of benthic fauna. The most abundant species in Tampa Bay was the cephalochordate *Branchiostoma floridae*, which accounted for 5.06 percent of the overall abundance. The mean abundance of *Branchiostoma* was 491 per square meter, with a maximum density of 17,775 per square meter. *Branchiostoma* was found primarily in polyhaline and euhaline salinities, normoxic conditions, and in medium to coarse sediments (fig. 7–11) (Karlen and others, 2008).

The second most abundant species was the cirratulid polychaete, *Monticellina* cf. *dorsobranchialis*, which was the most abundant taxon in Hillsborough Bay and Terra Ceia Bay samples and ranked second in the Manatee River. *Monticellina* cf. *dorsobranchialis* was commonly found in sites with very fine to fine-grained sediments, high mesohaline to euhaline salinities, and a wide range of DO conditions (hypoxic to normoxic) (fig. 7–12) (Karlen and others, 2008).

Other abundant benthic taxa collected during the period included the brachiopod *Glottidia pyramidata*, which was the third most abundant infaunal organism bay-wide, with an average density of 479 per square meter and a maximum density of 94,374 per square meter, mostly as recently settled post-larvae (fig. 7–13). The relative abundance of *Glottidia pyramidata* was highest in Middle Tampa Bay, where it accounted for 14 percent of the benthic abundance. *Glottidia* was found at relatively deeper sites (>2 m) with fine to medium grained sediments, in polyhaline salinities, and at normoxic DO levels.

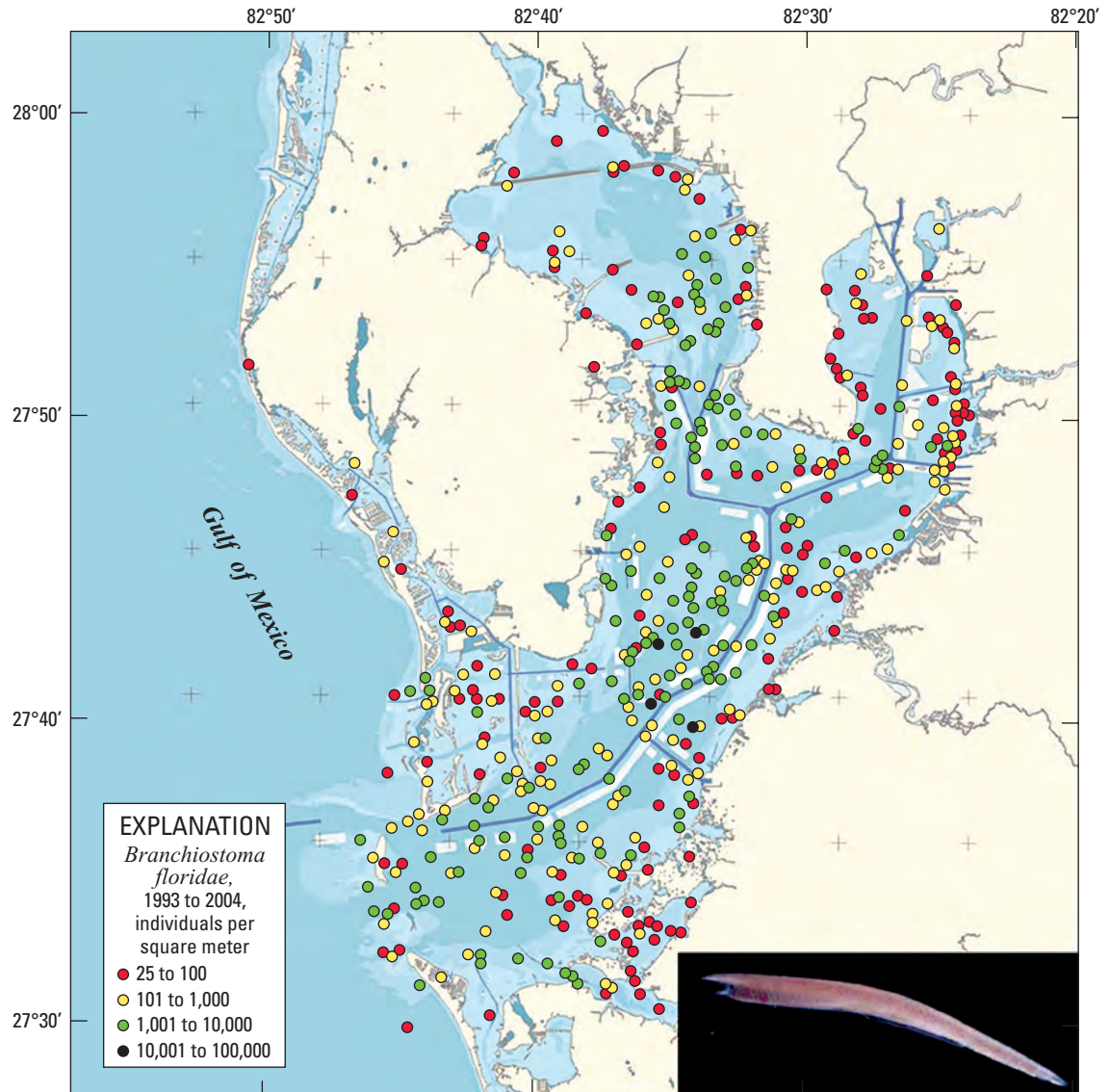


Figure 7–11. Late-summer distribution of *Branchiostoma floridae* in Tampa Bay, 1993–2004. From Karlen and others (2008).

Tubificid oligochaetes (a composite of multiple species composed of immature and/or damaged specimens that could not be identified below the family level) were common across all years and bay segments, ranking fourth overall in relative abundance and comprising 3.30 percent of the organisms collected. The gastropod *Caecum strigosum* was the fifth most abundant infaunal animal bay-wide, and accounted for nearly 3 percent of the bay-wide relative abundance. *Caecum strigosum* was among the most abundant taxa during all years except 2003 and 2004, and was particularly abundant in Middle Tampa Bay, Old Tampa Bay, and Lower Tampa Bay. *Caecum strigosum* was found primarily at deeper sites (>4 m) with coarse sediments (fig. 7–14).

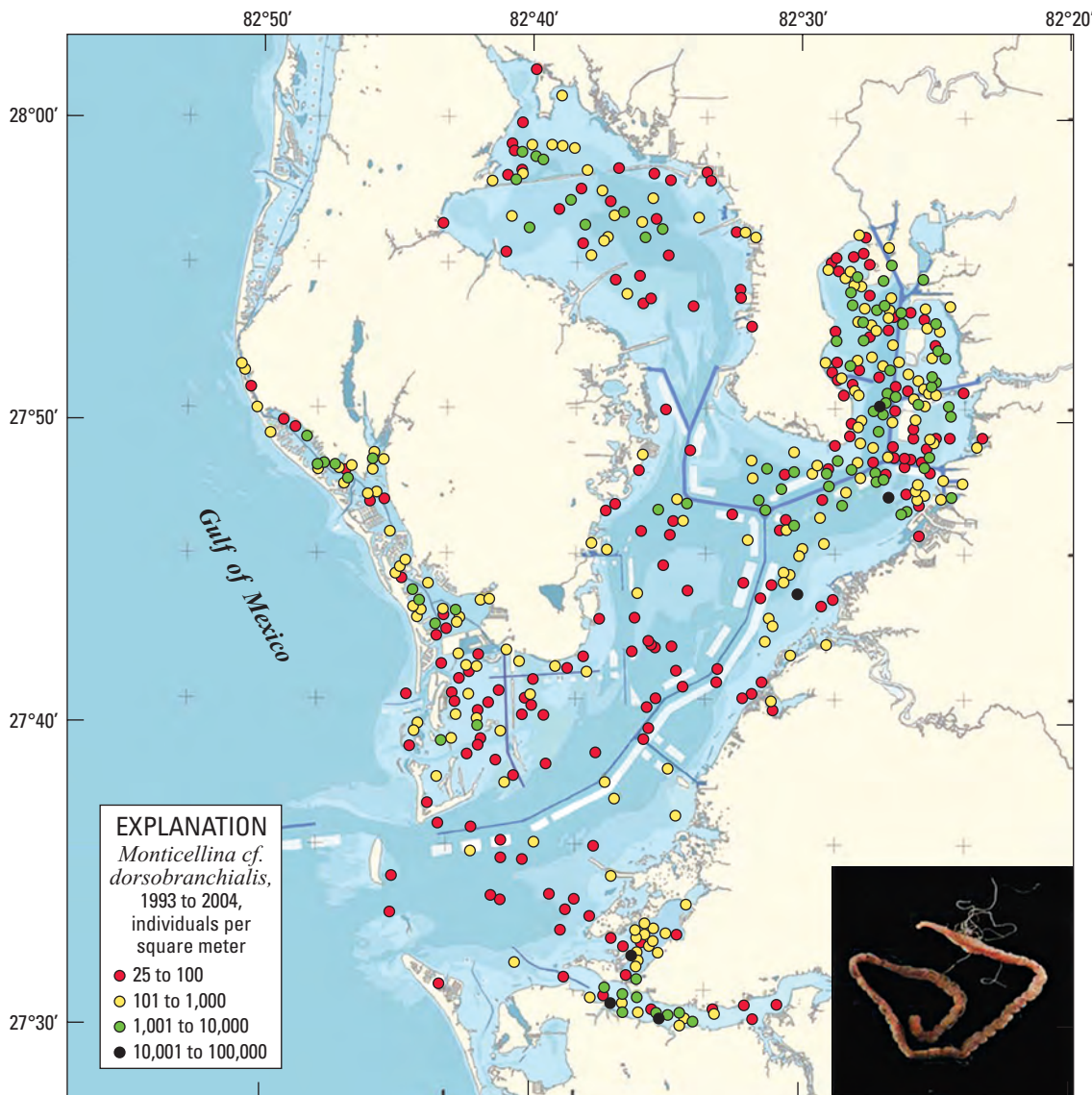


Figure 7-12. Late-summer distribution of *Monticellina cf. dorsobranchialis* in Tampa Bay, 1993–2004. From Karlen and others (2008).

Next Steps and Future Challenges

In 2004, the TBEP and its partners adopted the following sediment-quality protection and restoration strategy to achieve the resource-management goals noted.

Resource Protection: Use the TBBI to define “healthy” and “degraded” benthic-habitat conditions. (This has been done over a series of 4-year time periods, from 1993 through 2004. Scores for these periods are summarized in table 7-5.) The resource protection goal is to maintain the condition of existing “healthy” areas, preventing degradation of sediment quality in those regions of the bay.

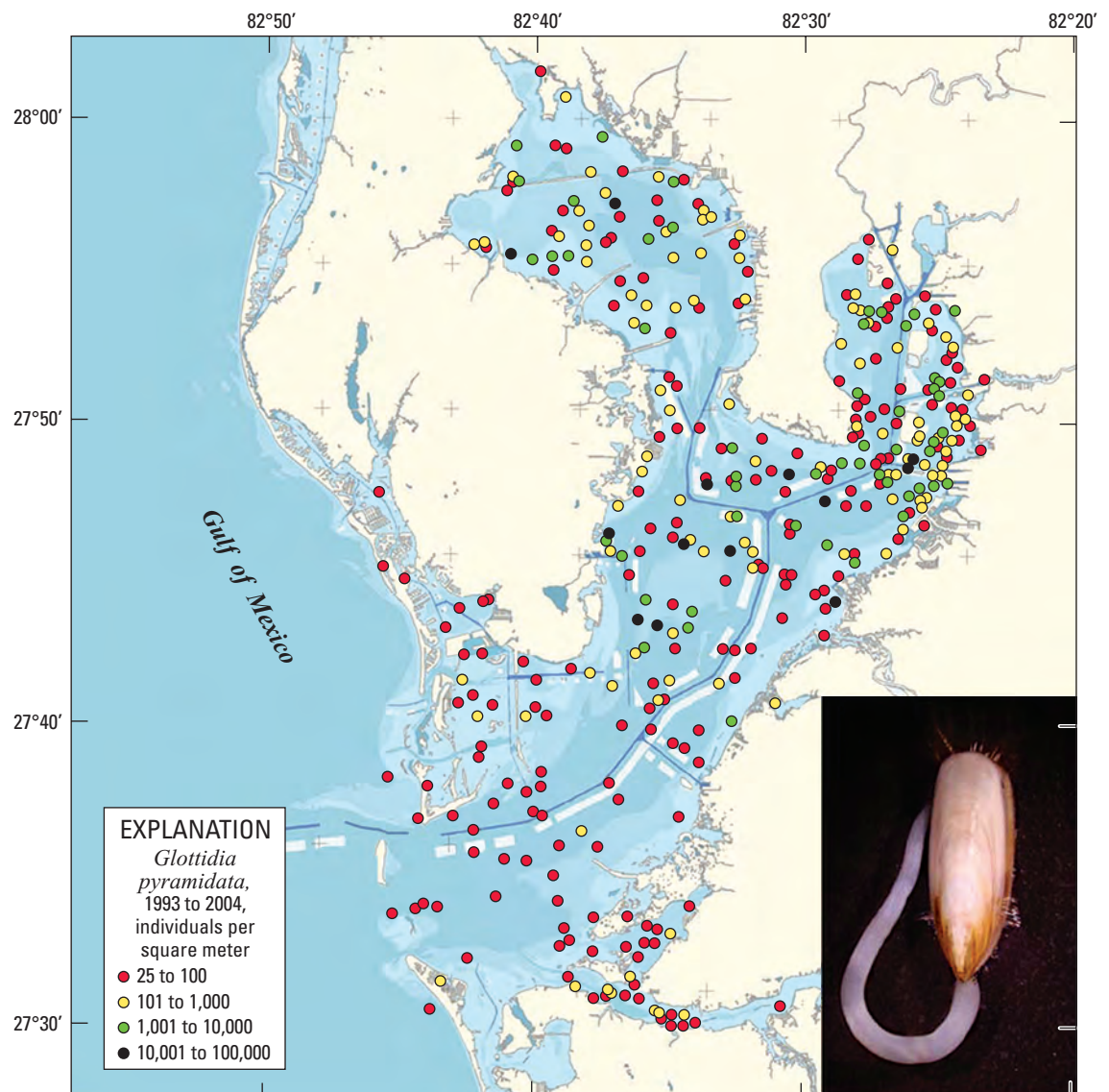


Figure 7-13. Late-summer distribution of *Glottidia pyramidata* in Tampa Bay, 1993–2004. From Karlen and others (2008).

Resource Restoration: develop and implement site-specific restoration plans for sites exhibiting “degraded” sediment chemistry and benthic-habitat conditions. Nine priority areas, shown in figure 7–15, have been identified for restoration plan development. These areas include:

1. Palm River/McKay Bay
2. Ybor Channel
3. West Davis Islands
4. East Bay
5. Largo Inlet
6. Westshore
7. Bayboro Harbor/Port of St. Petersburg
8. Apollo Beach/Big Bend, and
9. Upper Old Tampa Bay (north of the Courtney Campbell Causeway).

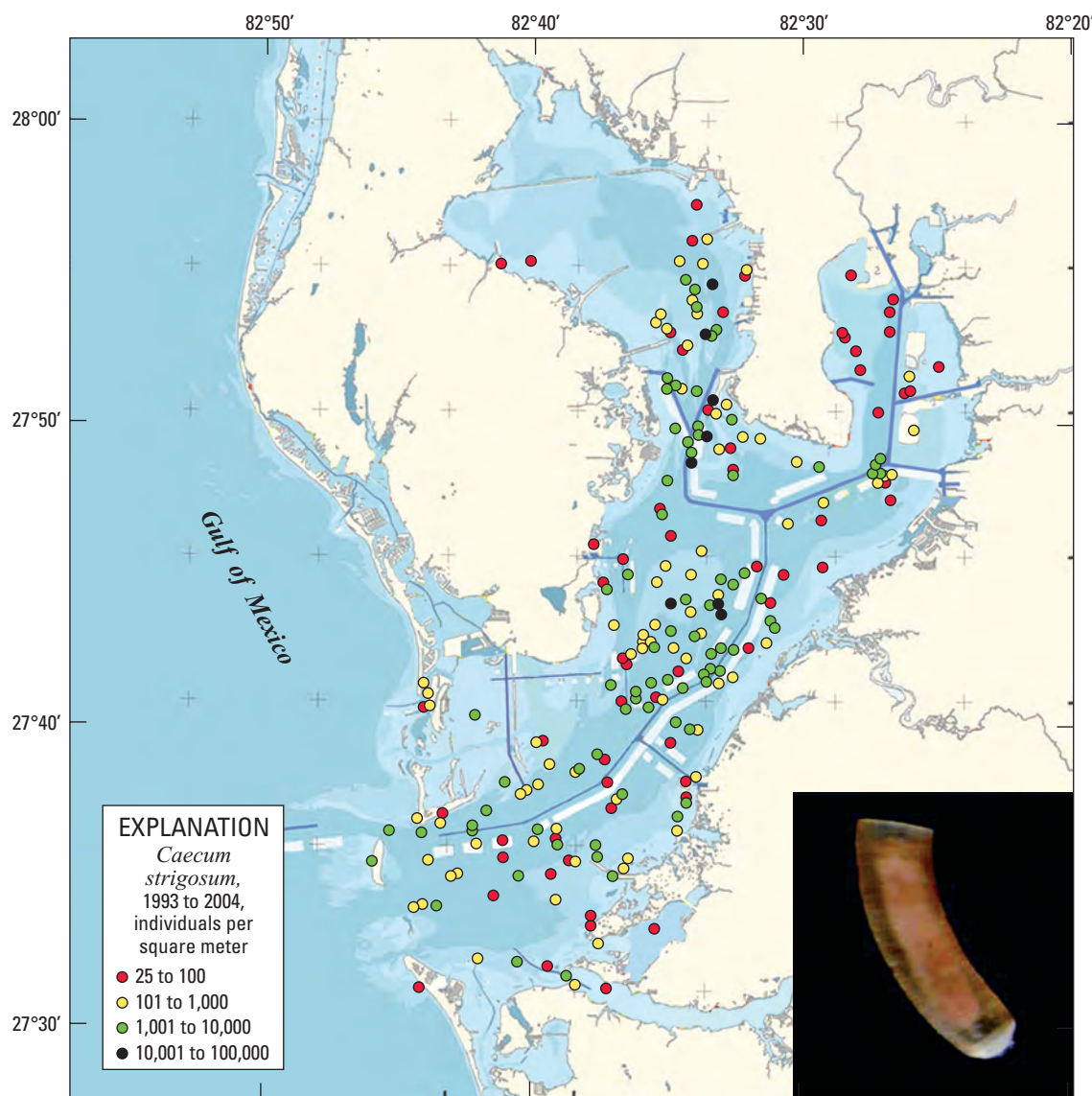


Figure 7-14. Late-summer distribution of *Caecum strigosum* in Tampa Bay, 1993–2004. From Karlen and others (2008).

Due to funding limitations, it is anticipated that restoration plans for these areas will be developed in phases, over a multiyear period. They are expected to include recommendations for stormwater improvements and other source control and pollution abatement projects, and may address sediment remediation activities, such as capping of contaminated sediments with clean fill. An evaluation of alternative methods of achieving sediment-quality targets (Hackett and others, 2003) summarizes existing typical sediment contamination control methods, and describes alternative management options to achieve sediment-quality targets, including estimates of costs associated with these methods.

Table 7-5. Summary of Tampa Bay Benthic Index benthic habitat quality classifications of sites monitored during 1993–2004.

[TBBI, Tampa Bay Benthic Index; <, less than; >, greater than. Data from Karlen and others, 2008]

	Number of sites	Percent				
		Undefined ¹	Empty ²	Degraded ³	Intermediate ⁴	Healthy ⁵
1993–1996	446	1.12	2.02	30.04	44.84	23.09
1997–2000	453	1.10	0.88	21.85	49.23	28.04
2001–2004	318	1.57	1.57	29.56	41.51	27.36
Hillsborough Bay	290	1.72	4.83	33.10	41.72	20.34
Old Tampa Bay	166	0.00	0.60	16.87	46.99	35.54
Middle Tampa Bay	238	1.26	0.00	14.71	49.58	35.71
Lower Tampa Bay	182	0.55	0.00	18.68	46.70	34.62
Manatee River	119	4.20	0.00	38.66	46.22	15.13
Terra Ceia Bay	68	1.47	0.00	38.24	44.12	17.65
Boca Ciega Bay	154	0.00	1.95	40.26	44.16	13.64
Tampa Bay (total)	1,217	1.23	1.48	26.87	45.60	26.05

¹TBBI < 0 (due to artifact in index calculation algorithm).²TBBI = 0 (no organisms present in sample).³TBBI=0–73.⁴TBBI = 73–87.⁵TBBI > 87.

In 2004, a sediment-quality working group, convened by the TBEP, recommended that the following additional tasks be evaluated for inclusion in the management strategy:

- Determine “achievability” during the development of restoration targets, and identify and remove “nonrestorable” areas, such as major port facilities from the target restoration acreage;
- More precisely identify current sources of contaminants and the areal extent of already degraded benthic habitats (for example, through additional targeted sampling and subbasin mapping); and
- Assess year-to-year variability in the extent and severity of contaminated sites, and verify whether degraded sites are consistently degraded, or whether temporal variations or trends in contamination levels exist.

With respect to reducing future contaminant loadings to already contaminated areas and protecting currently uncontaminated areas from future contamination, the available information indicates that management of diffuse sources, such as stormwater runoff and atmospheric deposition, will be important challenges (Frithsen and others, 1995; McConnell and others, 1996; McConnell and Brink, 1997). Permitted point sources are also thought to be substantial sources of some COC, however, and discharges from those sources could be examined to identify opportunities for contaminant load reductions (McConnell and Brink, 1997). As detailed sediment-quality action plans are developed for individual areas on the priority list, contaminant contributions from permitted sources can be assessed for each area, and the feasibility of reducing contaminant loads from those sources can be evaluated.

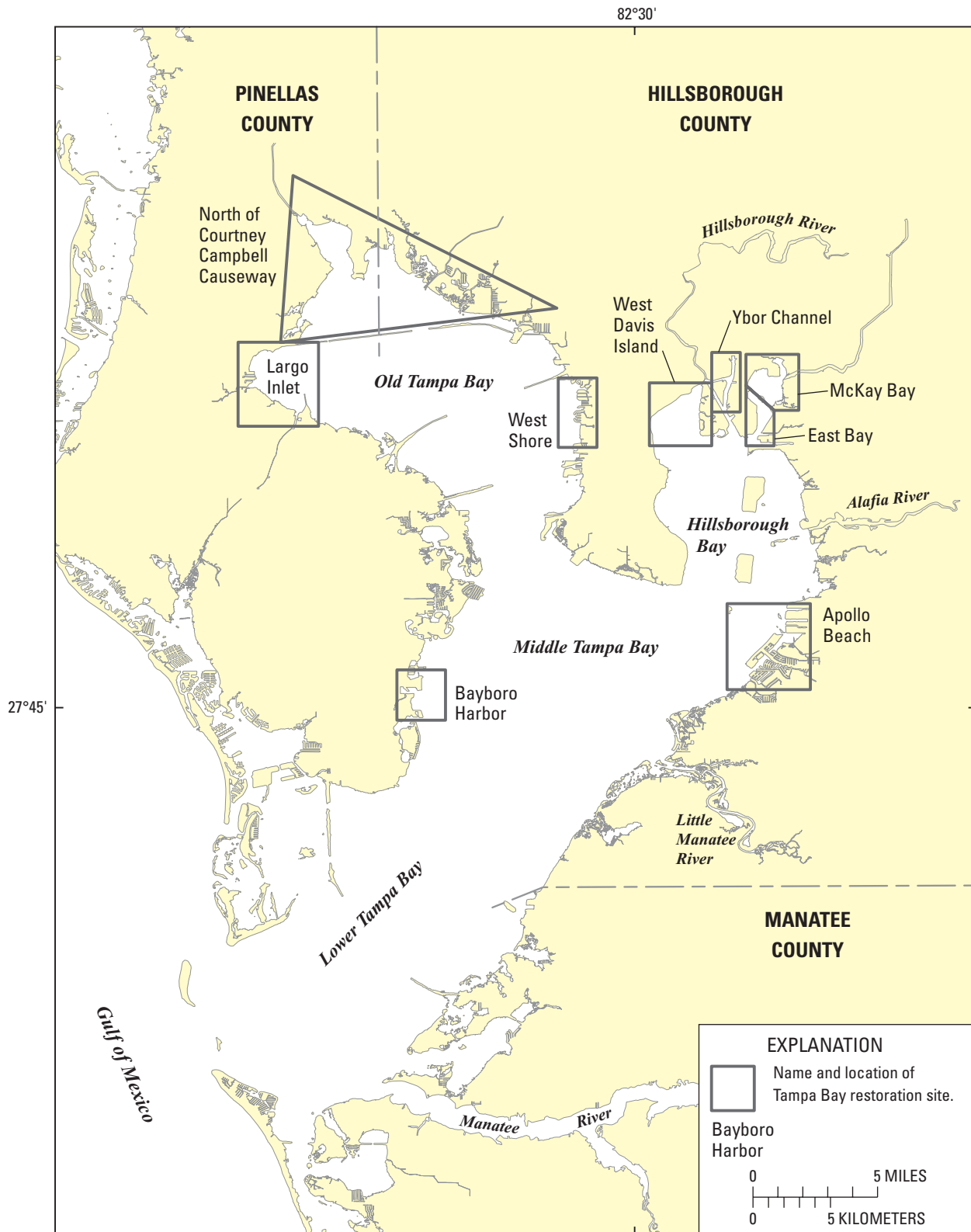


Figure 7–15. Nine priority areas in Tampa Bay that have been identified for restoration plan development. From Tampa Bay Estuary Program Sediment Quality Working Group.

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