BIVALVED SHELLFISH CONTAMINATION ASSESSMENT

FINAL

Prepared for: Sarasota Bay National Estuary Program
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I. EXECUTIVE SUMMARY

This project primarily addressed contamination levels in the edible tissues of recreationally important bivalve shellfish, *Crassostrea virginica* (oysters) and *Mercenaria* spp. (hard clams). Contaminants examined included pathogenic bacteria, toxic metals, pesticides and petroleum based compounds. Existing data were used to provide background information on the ecological requirements of these species and the ecological and potential human health impacts of selected contaminants. No formal health risk assessment was conducted.

Background

For shellfishing, the bulk of use impairments nationwide are attributed to bacterial and viral contamination, the presence of biotoxins such as that in red tides, followed by toxic compounds (pesticides, PCB's, and then metals). Current classification of waters by FDNR are designed to protect against these hazards, with approximately 3,000 acres classified as conditionally approved. This area is substantially smaller than the Class II waters designated as suitable for shellfish harvesting and propagation by the Florida Administrative Code. While FDNR plans a reclassification for this area, expanded opened waters will require a further commitment to continued sanitary monitoring.

As for the nation, declines in Florida landings of bivalved shellfish have been attributed to overharvesting and continuing expansion of areas affected by non-point source pollution. The bulk of the Florida oyster landings are currently from the northern Gulf coast, while the east coast is the dominant area for clams. These landings are relatively low percentages of the national totals. Locally, the commercial fisheries for oysters and clams collapsed in 1967 and 1971, respectively, and clam fisheries in this area have historically been erratic, possibly eliminated by extensive red tides. Information on recreational effort is minimal, and statistics available lump efforts for lobster, shrimp, scallops, and crabs together with bivalves.

Within the Sarasota Bay study area, 47 percent of the watershed is currently listed as 'developed', i.e., either residential, commercial, institutional, industrial, transportation, or power or sewage treatment plants land uses. Estimates at build-out are that 82 percent will be similarly classified with concomitant increases in non-point source loadings of some parameters.

Aquaculture or other commercial efforts within the study area are deemed unlikely for the following reasons: 1) difficulty of obtaining leases of subtidal State lands, 2) high degree of urban influence on water quality, 3) relatively small area of approved waters, 4) high incidence of red tides in this region of Florida, 5) poor shipping characteristics of dominant local *Mercenaria* species, and 6) lack of subtidal oyster habitat and consequently small sized individuals.

This Project

Following a summary presentation of the ecological requirements of various life stages of oysters and hard clams, the sources to the environment of the contaminants addressed in this study are summarized. Selected health impacts reported for these contaminants, and any regulatory or agency recommended exposures are also presented. Any ecological impacts to the shellfish themselves, where this can has been reported as a function of the tissue rather than exposure concentration, are also listed.

Population Surveys

The field work conducted under this project consisted of two, seasonally based, population surveys of *Mercenaria,* using a timed effort approach to mimic recreational shellfishing. While not as quantitative as raked quadrats, this technique was much less destructive in grassy areas and raking was used to verify the efficacy against smaller clams in particular. The species surveyed is described as *Mercenaria* spp., since the dominant or native *M. campechiensis* can readily hybridize with *M. mercenaria.* Location and condition of oyster bars were noted during the same survey. Following each of the population surveys,' tissues were collected from ten locations each for oysters and clams for contaminant analyses. Stations were geographically distributed to permit comparisons, and only one stations was located within the conditionally approved waters.

There was little seasonal variation apparent in abundance or size of clams. The distribution of *Mercenaria* varied significantly with sediment type, being most numerous in sandy mud. There was a notable absence of clams on the east side of Sarasota Bay, which, although observed in other regions on this coast, is not readily explained. There were also very few clams in the Midnight Pass area, which was potentially attributed to unfavorable salinities and perhaps low current velocities. Clams were most abundant on the western shore of Sarasota Bay (largely in the conditionally approved area), western Anna Maria Sound, and New Pass where between 21 and 35 individuals could be found in a 30 minute effort. There were no significant correlations noted between a data set of quarterly water quality data and clam abundance.

Predacious molluscs were observed primarily in the northern portion of the study area. The less valuable, larger, 'chowder' clam predominates due both to predation, rapid growth rates, and potentially poor recruitment. Smaller sized clams were noted both near the passes, and in shell or coarse substrate, while larger individuals were found in sand. Correlations of clam length with some water quality parameters is likely the result of the distribution of smaller individuals near the passes and the dominant water quality in those regions. Recreation clammers were noted during the surveys but none were harvesting in the approved areas which is difficult to access. The amount of recreational harvest, whether for consumption or for use as fishing bait, is unknown.

While oysters were ubiquitous, they were predominantly found intertidally, in the southern portion of the study area, and at the mouths of tributaries to the Bay. The salinities produced in these regions are undoubtedly beneficial from the standpoint of reducing predation, although numerous predators were noted. The intertidal growth habit which results from high levels of predation produces a small and less commercially valuable individual.

Bacteriological Contaminants

For bacteriological contaminants, fecal coliform standards were exceeded slightly at many stations, but no tissues exceeded the National Shellfish Sanitation Program (NSSP) criteria of 230 per 100 g tissue. The water column and tissues were not highly polluted, nor were there high levels of vibrios. These low counts of coliform bacteria should be tempered by the fact that little rain was experienced during the supposed wet season. In considering the *Vibrio* results, one should note that the warmest season, when vibrios are most numerous, was not sampled. Of the vibrios, the most frequently identified were V. *alginolyticus V. parahaemolyticus* and *A. hydrophila,* with *V. vulnificus* occurring only in the spring and at selected stations in the water column and oyster tissue samples. Results suggest that the major groups of vibrios and aeromonas are a part of the normal ecosystem and not of human fecal origin.

Metal Contaminants

For metal contaminants, there was no significant seasonal variation in tissue concentrations for either species. Again little rainfall was recorded during the wet season and contaminant loads could be abnormally low. Much higher levels of some metals were reported in oysters than in clams, in particular, copper and zinc. This was true even when clams and oysters were collected from nearby stations, and is attributed to species-specific physiological strategies for metal detoxification.

In both clams and oysters, there were significant variation with stations location for all metals with the exception of mercury. This geographic variation in tissue concentrations were particularly evident for copper and zinc in oyster tissue. Oysters near the mouths of Hudson Bayou, Phillippi Creek, and South Creek and clams from near Phillippi Creek and Hudson Bayou (Selby Gardens) contained the highest concentrations of selected metals.

Sarasota Bay oysters either approximate or are lower than national averages for cadmium, copper, mercury and zinc. Arsenic levels are slightly higher and lead concentrations are substantially higher than national averages, although the Sarasota Bay average is still less than 40 percent of the maximum lead concentration recorded for any of the National Status and Trends stations. Cadmium concentrations are approximately one fourth of the national average. In relation to Florida Gulf coast values, Sarasota Bay oysters are lower than average in cadmium and mercury, average for arsenic, slightly above average for copper and zinc, and well above average for lead. The average lead in Sarasota Bay oyster tissues was still less than 40 percent of the maximum observed in the Florida NS&T stations, but the Sarasota Bay Hudson Bayou concentration of 6.9 µg/g exceeded the highest lead value reported (5.4 µg/g) for either Florida or the nation.

No station averages exceeded Food and Drug Administration (FDA) action levels or NSSP recommendations, for those metals which FDA addresses. Almost all of the clam stations, however, and some of the oyster sites exceeded the more restrictive Canadian action levels for lead. Rough assumptions on shellfish consumption rates (5 g/day), various agency recommendations on acceptable daily intake (ADI) rates of toxic metals, and Sarasota Bay tissue concentrations were used to estimate what percent of an acceptable intake would be constituted by shellfish consumption. Shellfish from most sites would not exceed 20 percent of an ADI for all metals except arsenic. Of these metals the most exposure was from zinc in oysters. For arsenic, quantities reported exceeded 100 percent of the ADI. The ADI, however, is based upon inorganic arsenical compounds and the organically bound arsenic found in seafood is considered to be substantially less toxic.

<u>Toxic Organics</u>

For the study as a whole, eight of the 18 pesticides under analysis were found in shellfish and concentrations were generally very low. There were no obvious geographic, or species distributions of the tissue concentrations, trace amounts of pesticides being found in most regions of the study area and in both clams and oysters. Those pesticides observed at stations during the first sampling were generally not detected in tissues during the second sampling. A greater variety of pesticides was detected in the fall samples than from those collected in spring, and the most common were dieldrin, DDE, and BHC isomers. Of the station averages of the pesticides detected, the highest concentrations were usually contained in oysters, but these organisms were also more directly exposed, as stations were preferentially near the mouths of tributaries.

No pesticide exceeded the FDA action levels, nor did any violate the slightly more stringent Canadian Action levels. The oysters collected in the spring from Phillippi Creek, however, did contain DDE in concentrations equal to seven percent of the FDA action level of 5,000 ng/g. *Mercenaria* from Blackburn Bridge also contained approximately 12 percent of the FDA action level for chlordane. One sample of oyster tissue from Hudson Bayou contained five percent of the total DDT (the sum of all DDT, DDE, and DOD) allowed by FDA during the fall sampling. This level was considered high in relation to the 1986-1988 NS&T data. The erratic nature of their presence at the same site indicates that loadings are intermittent and associated with either resuspension of older contaminated sediments or with new applications of approved carbamates or organophosphates.

Only trace amounts of polycyclic aromatic hydrocarbons were detected in *Mercenaria* spp. and *C. virginica* tissues. Compounds indicative of both petroleum (petrogenic) and combustion (pyrogenic) sources were present, but the bulk were considered to be pyrogenic in origin. There was no seasonal, species, or geographic distribution to these results, indicating no chronic source of these compounds to Sarasota Bay.

Recommendations and Research Needs

As shellfish in Sarasota Bay generally do not appear to be grossly polluted, recommendations for bacterial (fecal organisms) and toxic compound control and reduction was based on reducing non-point source (NPS) loadings of particulates. Particular watersheds could obviously benefit from these techniques more than others. More information on the quantity and types of compounds subject to aerial deposition may allow a more effective use of available funds for NPS control.

Development of biologically based sediment criteria would afford the best protection to the bivalved species, but species-specific thresholds must be developed. These thresholds must extend beyond conventional acute and chronic toxicity assessments, and help to define the ecological impacts of these toxic compounds.

As vibrios are apparently endemic to the estuarine environment, control human exposure to these pathogens will continue to focus on education of at-risk individuals, primarily those with blood, liver, or immunological disorders.

The low bacterial counts in shellfish from Sarasota Bay lends support to the Florida Department of Natural Resources proposed reclassification and expansion of approved waters. These counts would undoubtedly increase under conditions of high rainfall, however, and the State shellfish sanitation program will need to support, through continued surveys, any additional areas opened.

While harvest pressure for human consumption appears low within the study area, any enhancement in this resource may generate additional interest and pressure. There is an unknown amount of harvesting for use as finfish bait. Overall, there is a lack of information on population dynamics, including recruitment, predation pressures, and harvesting pressure, which should be quantified to manage the resource and protect from overharvest,

The NPS controls for particulate removal would also act to improve the detention of stormwaters and increase the dry season base flow. This restoration of altered flows would be very beneficial to oyster populations in the southern portion of the study area. Additional support for flow restoration could be found from a determination of the paleoenvironment in the study area by the use of morphological characteristics of current and indian midden oyster shell.

As NPS controls improve, and FDNR expands the approved shellfish harvesting area, a formal health risk assessment will become more pertinent. At this time in addition, a wet season tissue sampling will become essential to quantify what could be worst case tissue contaminants and the suite of analytical compounds could be further expanded to include selected PCB isomers. More event oriented sampling could provide additional information on maximum loads received by the estuary.

Resource enhancements at this time can include both clam seeding and cultch placement to increase the populations, but should be coupled with small scale investigation to determine either optimum locations or rates of success. These enhancements will be difficult to evaluate economically, as for the most part they will not likely result in any direct increase in recreation potential. There are, however, valid ecological inducements for these enhancements, in order to 1) increase biomass and productivity of the estuary, 2) in the case of oyster reefs, provide additional habitat for invertebrate fauna and juvenile fish, 3) provide increased shoreline stability, and 4) reduce sediment resuspension through increased oyster reef area and subsequent wave damping.

II. INTRODUCTION

Project Summary

The contamination assessment of bivalve shellfish in Sarasota Bay, Florida (Figure 1), is one part of the overall study plan for the National Estuary Program which is addressing the bay. The project focused on two recreationally important bivalve shellfish, *Crassostrea virginica* (oysters) and *Mercenaria* spp. (hard clams) and primarily addressed contamination levels in the edible tissues of these organisms, including pathogenic bacteria, pesticides and petroleum based compounds, and toxic metals. Geographic comparisons allowed the evaluation of various regions of the Bay within a local state and regional context. Literature retrievals were used to provide background information on the ecological requirements of these species and the ecological and potential human health impacts of selected contaminants. Based on the possible origins and fates of the studied contaminants, recommendations were offered for the protection and enhancement of this fishery. The entire project was conducted under an Environmental Protection Agency approved Quality Assurance Plan.

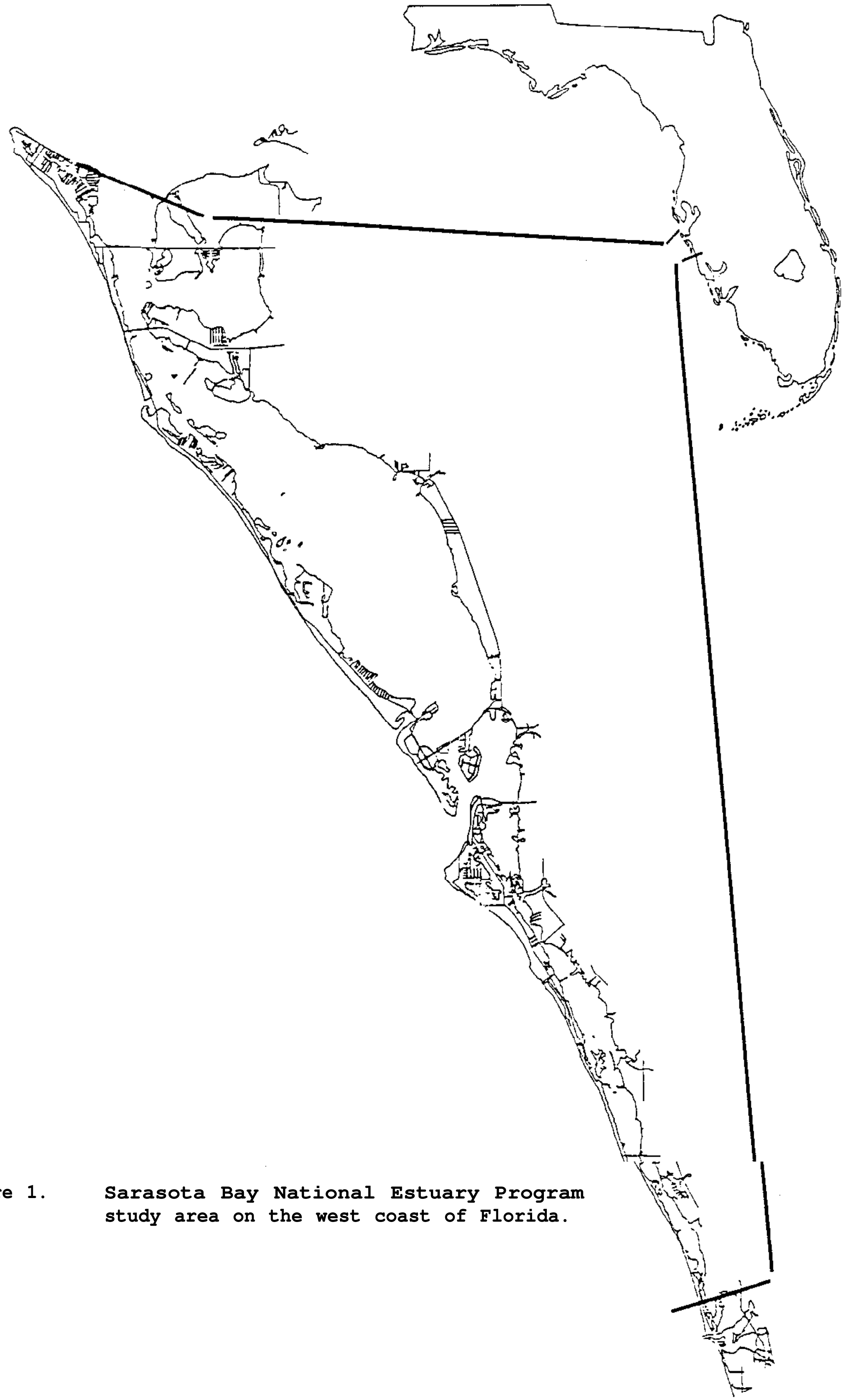

Figure 1. Sarasota Bay National Estuary Program study area on the west coast of Florida.

III. BACKGROUND

III.A. Classification of Shellfish Harvesting Waters

The National Shellfish Sanitation Conference (NSSC), which issues guidelines for shellfish harvesting, treatment and interstate commerce, was initiated to prevent the transmission of disease by edible shellfish. The program (NSSP) is a cooperative project of the U.S. Food and Drug Administration, various state agencies tasked with these responsibilities, and the shellfishing industry. Within Florida, the state role is filled by the Florida Department of Natural Resources (FDNR). Through the cooperation of both state and federal governments, diseases transmitted by contaminated shellfish have been greatly reduced.

Under the NSSP guidelines, regions are classified under state authority into one of the five following categories described by Broutman and Leonard (1988).

- Approved — Waters may be harvested for the direct marketing of shellfish at all times.

Harvest Limited

- Conditionally Approved — Waters do not meet the criteria for approved waters at all times, but may be harvested when criteria are met.
- Restricted — Shellfish may be harvested from restricted waters if subject to a suitable relaying or purification process.
- Conditionally Restricted — Shellfish may be harvested and subsequently purified when criteria are met.
- Prohibited — Harvest cannot occur at any time.

Currently in the state of Florida, over 2.2 million acres are recognized as potential shellfish areas. Only 22 percent of this is approved for harvesting and 5 percent is conditionally approved, with the remainder either prohibited or unclassified.

Classification requires that states perform sanitary surveys, sampling waters a specified number of times per year for bacteriological parameters, evaluating actual or potential pollution sources, and examining the hydrologic and meteorologic conditions that may affect pollutant transport. Of the above categories, the conditionally approved and restricted categories require the most state resources to maintain and lack of resources often dictate that areas remain unclassified. Florida, in 1990, allocated $0.29 per classified acre to the state shellfish program which is identified as being below both the median and the average of $0.47 for all coastal continental states (NOAA, 1991).

As a result, sanitary surveys in Florida have been completed for only 50 percent of the 2.2 million acres of shellfish waters by 1988 (Broutman and Leonard, 1988). This situation is reflected within the NEP study area where waters outside the barrier islands, south of New Pass, and north of the Sarasota/Manatee County line are listed as unclassified. Increasing attention to this problem is evidenced by FDNR's efforts and a soon to be released updated and expanded classification.

Sources of fecal coliforms which can cause waters to be classified as harvest-limited include sewage treatment plants, failed septic systems, industrial wastes as from seafood or paper and pulp processing facilities, boating activities, agricultural runoff from grazing lands or lands fertilized with manure, and fecal material from wildlife, including bird rookeries (Broutman and Leonard, 1988).

Stormwater runoff is perceived as one of the major water quality problems within the Bay (FDER, 1988) and is reported to contain bacterial levels comparable to secondarily treated effluent (Galvin, 1987). Conditionally approved waters are generally those affected by non-point source pollution during periods of high rainfall. It is significant to note that, within the Gulf of Mexico states and between 1971 and 1985, only 774 acres of the approximately 5.7 million acres of shellfish waters have been upgraded in classification due to sewage treatment plant construction. Gulf-wide during this period, classification upgrades affected 16,000 acres while downgrades were applied to 770,000 acres, primarily to increasing recognition of non-point source impacts (Broutman and Leonard, 1988).

Between 1985 and 1989, an increasing percentage of harvest-limited waters were designated as affected by urban runoff, from 23 percent to 38 percent. Increases were also observed in the percentages affected both by septic and boating influences (NOAA, 1991). Most recent classification changes appear to be the result of changes in management policy (increased sampling, political decisions, or default to unclassified in the absence of monitoring) rather than attributable to changes in water quality (NOAA, 1991).

There are additional management approaches to control marine biotoxins, which in this region are produced by the 'red tides' or blooms of the dinoflagellate, *Gymnodinium breve.* The region between Tampa Bay and Charlotte Harbor has historically received the greatest number of outbreaks of this organism. During these blooms, all waters are closed to harvest as shellfish can concentrate lethal amount of toxins and analyses of the water column and shellfish tissues are required prior to reopening. Also of concern are the incidence of disease from *Vibrio* spp., which appear to be indigenous to the marine environment. There is a lack of agreement on the management options for this bacterial disease, but some feel that minimization of time and temperature during shipment of shellfish is beneficial. There is no disagreement that at-risk populations for this infection should be educated as to the severity of effects.

Classification for harvest, however, predominantly relies heavily on bacterial indicator species which do not always fully reflect the health risks of shellfish consumption. The correlation of viral and vibrio species with fecal coliforms, for instance is reported to be poor (Blake and Rodrick, 1983) and a national shellfish indicator study is underway to develop both direct detection methods for enteric viruses and for their indicators (LUMCON, 1991).

Pollutant contamination is ranked behind both bacterial and viral contamination and biotoxins (ciguatera, paralytic and neurotoxic shellfish poisoning) in order of importance (NAS, 1991). Similarly, while classification does include a shoreline survey to identify actual or potential pollution sources, including industrial or agricultural waste waters and pesticides, the survey does not typically include an actual analysis of contaminant concentrations in tissues (FDA, 1989). A recent census of state fish and shellfish advisory programs (Cunningham et al., 1989) which examined non-bacterial closures identified pesticides and polychlorinated biphenyls (PCB), followed by metals, as the three primary causes for the issuance of consumption advisories. Florida, however, although analyzing between 26 and 50 waterbodies at this time for tissue contamination, was collecting no samples in coastal waters.

Classification of Sarasota Bay shellfish waters was originally performed by the Florida State Board of Health in the early 1970's (Mr. Don Hesselman, personal communication). Current classification of the Sarasota Bay study area by FDNR is illustrated in Figure 2. There are no approved waters within the study area. There is a small conditionally approved area (3,000 acres) on the west side of Sarasota Bay, near the southern end of Longboat Key, and south of the Manatee-Sarasota county line. These conditionally approved waters are opened or closed primarily based on the presence of red tide organisms and toxin.

The remainder of Sarasota Bay in Manatee and Sarasota Counties, some 20,000 acres, is prohibited. Palma Sola Bay was conditionally approved in the past, but in 1981 was temporarily closed and has remained so until this year when it was reclassified as prohibited. The study area south of the Ringling Bridge in Sarasota, and north of the Cortez Bridge are unclassified. The study area ends at a conditionally approved area in lower Tampa Bay.

Proposed revisions to this classification, on the basis of sanitary surveys by FDNR during the preceding year, will classify additional areas. Subject to the results of public hearings, prohibited waters will expand south almost to Phillippi Creek and north along the eastern shore of the Bay to include waters off of Tidy Island and in Palma Sola Bay. A narrow fringe of prohibited waters will also be established along the western edge of the Bay from Bishops Point to the Cortez Road Bridge. Conditionally approved waters will also expand as previously unclassified waters are classified, and will extend northward up the center of the Bay

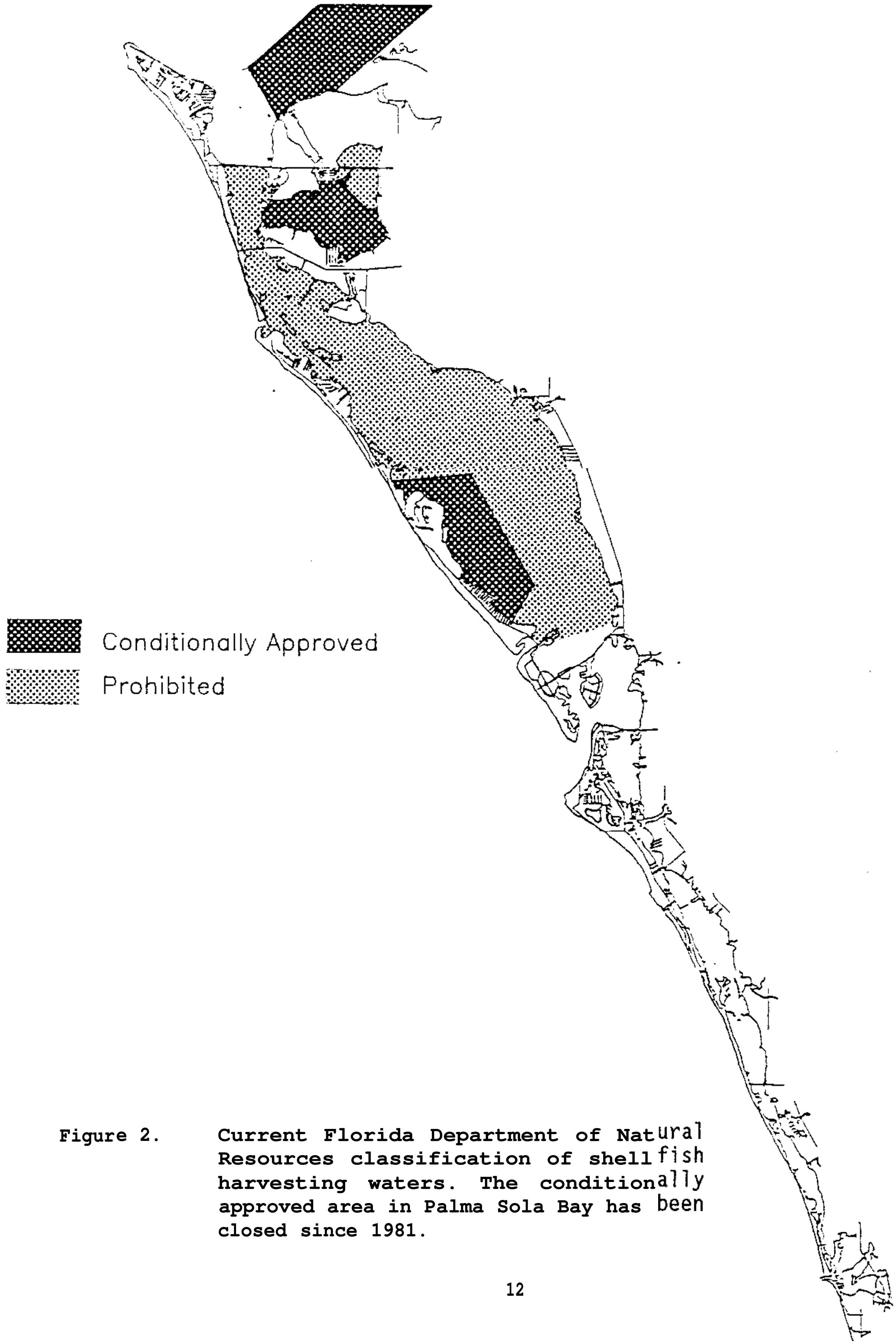

Figure 2. Current Florida Department of Natural Resources classification of shellfish harvesting waters. The conditionally approved area in Palma Sola Bay has been closed since 1981.

from the present location to near Sister and Jewfish Keys. There will also be an additional segment identified in mid-Bay, roughly between Stephens Point and City Island which will be identified as conditionally restricted, from which shellfish may be harvested if subsequently subjected to a defined depuration or cleansing process. Like the conditionally approved areas, this conditionally restricted category is closed as necessary based on rainfall amounts and/or the presence of red tide.

The Florida Department of Environmental Regulation, while not responsible for shellfish sanitation, does maintain separate water quality standards for those areas defined as Class II waters (Florida Administrative Code, Chapter 17-3). These are areas whose designated use includes shellfish harvesting and propagation. The water quality standards, because of the bioaccumulation potentials, are some of the most stringent applied to surface waters in the state. Within the study area, Class II waters (Figure 3) include Palma Sola Bay, Sarasota Bay from the Cortez Bridge south the Manatee County line, and the western portion of Sarasota Bay (west of the Intracoastal Waterway) between the Manatee County line and the Ringling Bridge. The remainder of the study area is classified as Class III, or suitable for the propagation and management of fish and wildlife.

III.B. Landings

NOAA (1991) has identified declines in landings of shellfish on a nationwide basis, with overharvest remaining a significant cause for the reduction in natural or 'wild' stocks. Diseases have been especially noted since 1957 and include the parasitic protozoans infestations known as MSX and Dermo *(Dermocystidium marinum)*, and other viruses, to which shellfish resistance may be compromised by other environmental stresses (NOAA, 1991).

Florida's declines in landings of oysters and clams are attributed both to overharvesting and to increasing areas affected by the pollution associated with coastal development (NOAA, 1991). MSX or Dermo instances have not been conclusively reported for the southwest Florida coast and only tentatively identified in the Jacksonville area. Hard clams are apparently less susceptible to these pathogens than are oysters. While parasitic loads are high in local oysters, these parasites do not appear to be fatal ones, although they may lower the overall health of the animals (Dr. Norm Blake, personal communication).

In 1985, 60 percent of the U.S. oyster harvest was from states in the Gulf of Mexico region, which is also identified as one of the fastest growing areas of the country and thus subject to increasing urban development and non-point source discharges. By 1989, oyster landings were sharply reduced, both nationally (from 26 to 12 million pounds) and in Florida (from 4.4 to 1.5 million pounds). During this same period, Florida landings declined from approximately 10 percent of the national

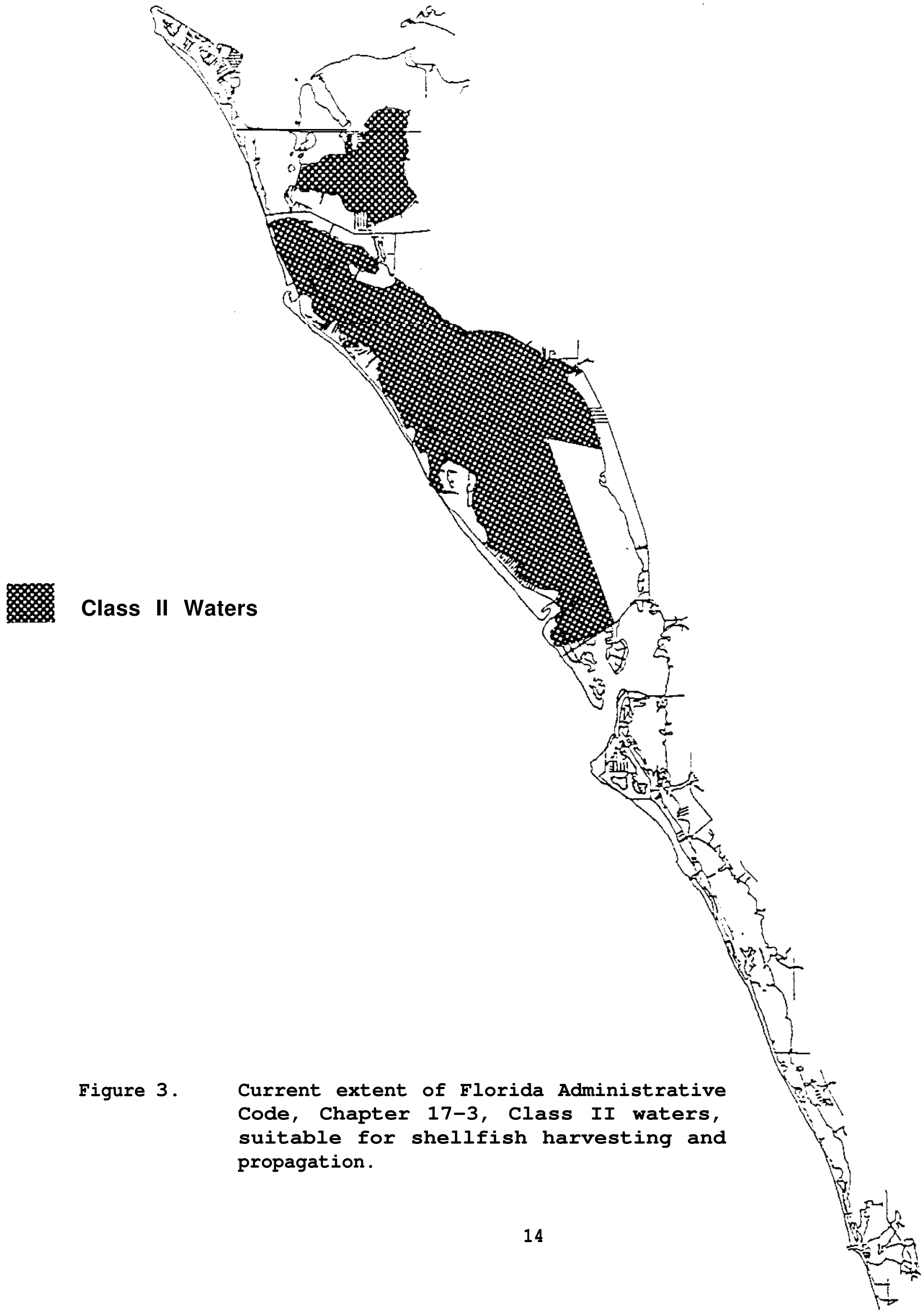

Figure 3. Current extent of Florida Administrative Code, Chapter 17-3, Class II waters, suitable for shellfish harvesting and propagation.

total to near 5 percent (NOAA, 1991) and reflected the impacts of several major storms in the Gulf Coast.

The commercial clam fishery in Florida has historically been very erratic, with major fisheries collapsing suddenly (Arnold, unpublished manuscript). A red tide outbreak is thought responsible for the 1947 decline in the Ten Thousand Islands (Godcharles and Jaap, 1973; Steidinger et al., 1973) while another effort in Charlotte Harbor was also short lived (Woodburn, 1962). More recently, clam landings have, like oysters, also exhibited declines in Florida between 1985 and 1989 (0.2 to 0.02 million pounds) while national harvests, dominated by landings on the middle Atlantic Coast decreased only slightly (from 144 to 121 million pounds). Most of the commercial Florida clam harvest is reported to come from the east coast (Arnold, unpublished manuscript; NOAA, 1991).

Landing records for Sarasota and Manatee Counties for the period 1951 to 1985 (FDNR, 1986) indicate that the commercial oyster fishery collapsed in the area in 1967 while no further commercial clamming occurred after 1971. Peak years for both fisheries took place in 1964 and 1965, and may be related to a short-lived clam fishery in the Charlotte Harbor area (Woodburn, 1962) and cultch placement in leased areas of Tampa Bay (Finucane and Campbell, 1968). A hurricane in 1966 may have contributed to the decline of the oyster industry, although the last direct hit was recorded in 1953 (Reynolds, 1976). Discharge to the Bay from the City of Sarasota's sewage treatment plant began in the 1950's and the closure of harvestable areas coupled with increased harvesting pressure on the remaining regions undoubtedly produced the local decline of these industries.

Recreational harvesting is largely unquantified, although thought to be important both in the study area and nationally (Stanley and Dewitt, 1983). A lack of shoreline access in the Sarasota Bay study area undoubtedly restricts the activity (Stevely et al., in press). Quantitative data on this activity were not available until 1985, the first year in which U.S. Fish and Wildlife Service separated recreational shellfishing statistics from total saltwater fishing. Shellfishing for this survey was defined as both molluscan and non-molluscan (i.e., oysters, clams, mussels, scallops, crabs, lobsters, and shrimp), but does not distinguish between species or assess the amount of the catch.

Florida leads the nation in the number of individuals shellfishing (over 450,000), each of whom participated just slightly less than eight days per year (NOAA and NMFS, 1991). Nationally, the demographic profile also indicates that these individuals are composed of more males and Caucasians, have higher household incomes, almost half are between the ages of 25 and 45, and that fewer are over age 65 than the general population (NOAA and NMFS, 1991).

III.C. Landuse and Access

The dominant land use of the contributing watersheds for this region of Florida is identified as agriculture (Broutman and Leonard, 1988). The non-point source project Sarasota Bay (CDM, 1992) also identifies more current assessments specific to the study area which indicate that 47 percent of watershed lands within Sarasota and Manatee Counties are in residential, commercial, institutional, industrial, transportation, or power or sewage treatment plants land uses. The remainder is divided between waterbodies and wetlands, forested uplands, woodlands, open areas and agricultural activities. Build-out conditions identified in this same report indicate that the residential and industrial categories will then comprise 82 percent of the counties area. Loadings of selected metals (lead and zinc) are projected to roughly increase between 10 percent and 40 percent and other urban contaminants such as petroleum products would also be expected to increase.

Other sources of loading traditionally considered in estuarine investigations include marinas and other areas of boat concentrations. The coastline of the study area is considerably hardened and numerous residential canals have light to moderate numbers of boats. Figure 4 illustrates the denser accumulations of boats by displaying commercial marina operations.

The highly developed nature of the coastal regions has also resulted in a limited amount of recreational shore access to Bay waters. Public access points to the Bay, not including Gulf sites, are illustrated in Figure 5 and are concentrated in comparatively few areas. Increasing access

III.D. Aquaculture

Aquaculture activities are a relatively limited importance in this region of Florida. There are no active shellfishing production leases of submerged lands on the southwest coast (Mr. John Stevely, personal communication) and State policy apparently discourages their issuance, with unresolved issues such as appropriate reimbursement amounts, and whether or not leases of public lands for exclusive private use are actually in the public interest. Existing leases are generally of long standing, particularly those in the northern Gulf Coast of Florida. On the other hand, the Department of Labor has supported through direct and relatively recent funding, training in the methods of clam aquaculture.

Following resolution of the above concerns, it also appears that new leases would be more likely in the northern Gulf Coast where developmental pressures are substantially less and water quality correspondingly better. The likelihood of obtaining leases in the Sarasota Bay area, with the urban influence and comparatively small areas of conditionally approved and conditionally restricted waters appears small.

Marinas and Yacht Clubs

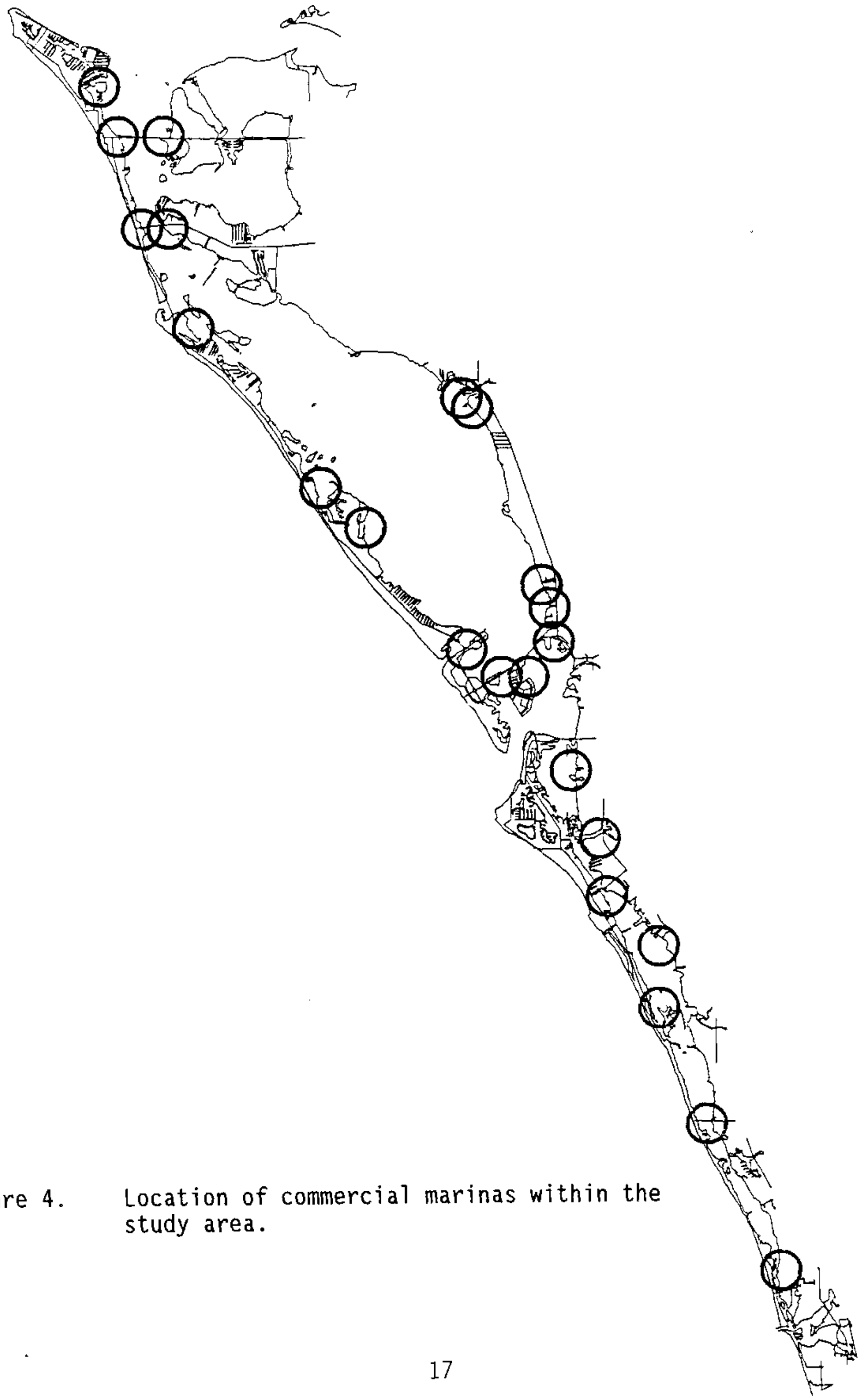

Figure 4. Location of commercial marinas within the study area.

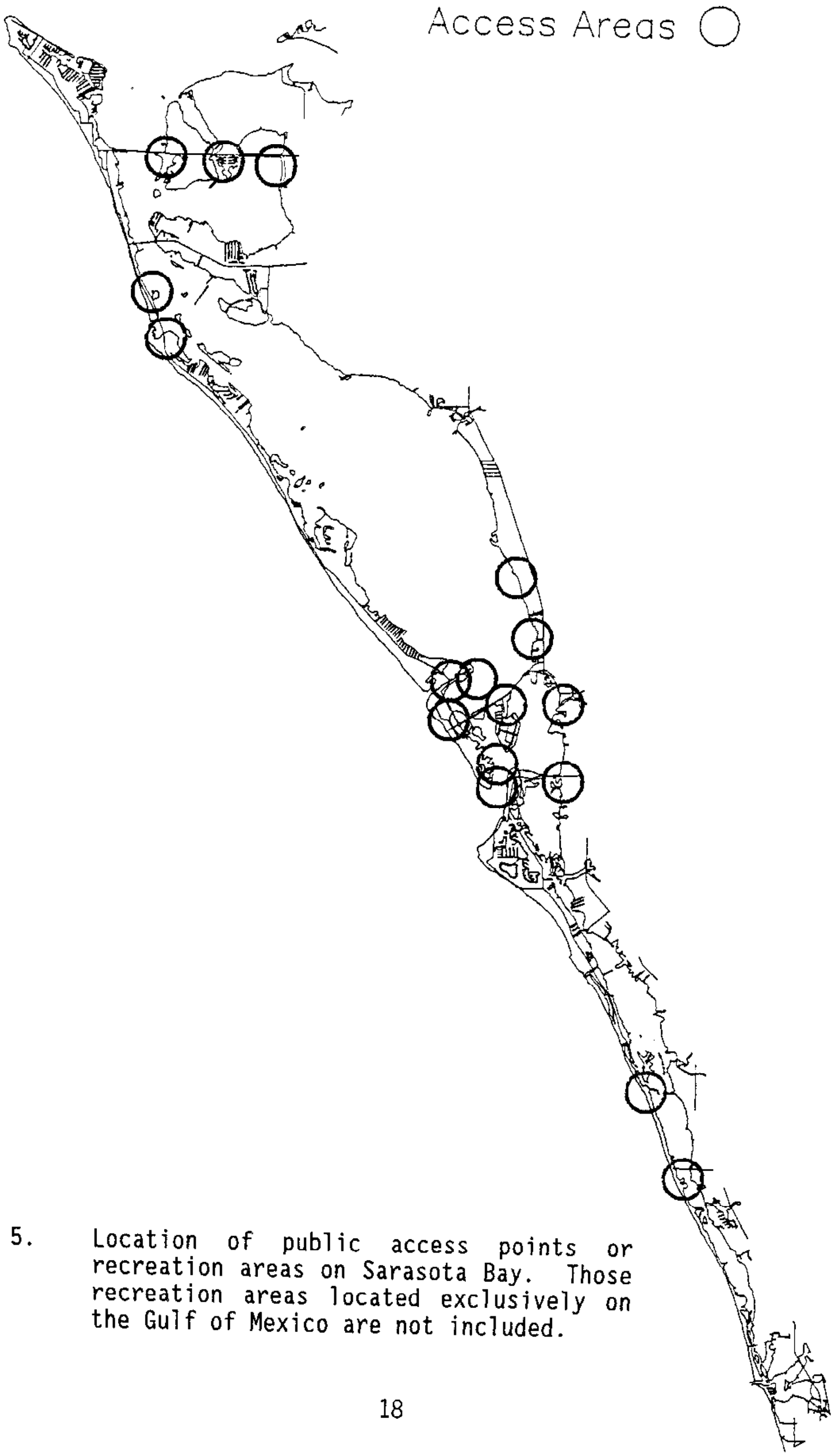

Figure 5. Location of public access points or recreation areas on Sarasota Bay. Those recreation areas located exclusively on the Gulf of Mexico are not included.

In addition, the compatibility of the industry, with the staked and marked lease plots of bottom and/or water column, with the high recreational traffic in Sarasota Bay is questionable. Those areas classified for conditional or restricted harvest are some of the most heavily traveled.

Restricted waters opens the possibility of clam culture followed by relaying or depuration of clams in waters of sufficient bacteriological quality. The approved waters for depuration are limited in size, leases would certainly be required, and the obstacles to aquaculture are identical for depuration. Technology enhanced depuration methods such as ozonation or irradiation may increase the feasibility of aquaculture in restricted waters, but ozonation is not approved for food products due to concern over residual oxidation byproducts and irradiation is not widely available and has been received with mixed opinions by the general public.

The conditional classification on the waters in Sarasota Bay also indicates that there would certainly be periods during which harvest is not allowed. With a summertime rainy season in this area, that would not generally affect the fall, winter and spring oyster harvests, but year-round clam harvests would certainly be curtailed at times. Other considerations are that this region of Florida's coast receives the bulk of the red tide outbreaks reported and these outbreaks typically occur during the fall and winter. Since 1978, 13 of the 14 shellfish bed closures in Manatee County and 18 of the 20 in Sarasota County due to red tide have occurred during oyster harvesting months.

Clam harvesting on a commercial scale in the Sarasota Bay region would also be affected by the physiological and growth characteristics of *Mercenaria.* While the dominant clam of commerce, *M. mercenaria,* forms hybrids with *M. campechiensis* (Dillon and Manzi, 1989), it is generally thought to be beyond the southern limit of its range in this region, and *M. campechiensis* predominates. *M. campechiensis* is much less tolerant of holding out of water than is *M. mercenaria* and successful shipping of live shell stock is more difficult. Clams in this region also grow beyond a commercially useful size (approximately 75 mm) within a 2-3 year period and their commercial value declines once past the 25 to 40 mm (approximately 1" to 1.5") 'littleneck' stage which may occur within 18 months. While this rapid growth would be optimum for aquaculture, it also means that the opportunistic harvesting of wild stocks depends on finding these organisms within a much narrower time window. Harvesting pressure will also reduce the population lifespans, which may be significant from the standpoint of limiting reproductive potential (Arnold, unpublished manuscript).

Oysters in the Sarasota Bay area do not grow in the vast subtidal assemblages which are commercially harvested in Appalachicola Bay, but occur as a narrow fringing community associated with mangrove roots and concentrated at creek mouths. Few are of legally harvestable size at this time (greater than 3 inches in length). A number of factors, from substrate type, current velocity, or salinity regime may be responsible for these differences. The creeks around which oysters concentrate in

Sarasota Bay, while providing reduced salinity regimes in which oysters are protected from predation, also deliver those bacterial contaminants in non-point source runoff which force the closure of beds.

These combination of factors make commercial aquaculture or harvest of wild populations of clams or oysters unlikely in the foreseeable future in Sarasota Bay. Recreational harvesting will undoubtedly continue, however, and confirms the validity of the recreational emphasis of this project.

III.E. Red Tides

The Florida red tide is produced by blooms of a dinoflagellate, *Gymnodinium breve,* which produces potent neurotoxins (Steidinger, 1983, 1990). These blooms represent a natural process, and are apparently unlinked to any pollution or human disruption of the marine environment. The blooms typically begin in the Gulf of Mexico 40-80 miles offshore and move slowly southeast toward the Tampa Bay area with the prevailing ocean currents. As the bloom progresses, the density of red tide organisms increases to several million cells in each liter of seawater, and the affected area expands to many square miles. The associated toxins cause extensive fish kills and create severe respiratory irritation to humans along the shore (Baden, 1983; Steidinger and Haddad, 1981; Pierce, 1986).

Ecological impacts of G. *breve* on bivalves are relatively unknown. Acute effects include mass mortality of invertebrates including bivalves (Tiffany and Heyl, 1978; Roberts et al., 1979) during intensive blooms, but this is not a common occurrence. Steidinger et al. (1973) relate the collapse of one of the most productive clam fisheries in the Ten Thousand Islands (Schroeder, 1924) to a major outbreak in 1946 and 1947. C. *virginica* valve closure is reported unaffected by G. *breve* (Sievers, 1969). *Mercenaria* and *C. virginica,* however, both respond with valve closure to other toxic dinoflagellates (Ray and Aldrich, 1976; Shumway and Cucci, 1987). In other geographic regions, *M. mercenaria* is reported to alter feeding behavior in the presence of the organism responsible for paralytic shellfish poisoning (not G. *breve)* and scallops will reportedly cease feeding and starve in the presence of other toxic phytoplankton species (Bricelj et al., 1990).

Even red tide populations well below the fish kill level pose a serious problem for public health through bivalve shellfish contamination, which occurs as the shellfish concentrate the dinoflagellates in their normal filter-feeding process (Cummings and Stevens, 1970; Music et al., 1973). While moderate red tides do not usually appear to harm the shellfish, toxin levels can be harmful to humans when contaminated shellfish are consumed. The illness produced is termed neurotoxic shellfish poisoning (NSP) (Baden, 1973) and symptoms include nausea from exposure to beach spray, or ciguatera-like symptoms of abdominal pain, vomiting, diarrhea, and neurological impairment from consumption of bivalves. The illness has no specific treatment and is not usually fatal (NAS, 1991).

Control of this public health hazard is effected by the closure of all approved or conditionally approved shellfish harvesting areas. Before reopening, the waters must not only be free of G. *breve* for a specified time period, but shellfish meats themselves must produce no results in a mouse bioassay. The impact on commercial shellfishing and fishing industries during these periods is substantial.

Because of the severe economic and public health effects of red tides, much consideration has been given to controlling the blooms. Control may be feasible within confined areas such as fish hatchery and aquaculture ponds, and research is underway to assess various methods such as aeration, chlorination, or ozonation. Control in the broad expanses of the Gulf of Mexico, however, or even within the Bay is impractical in the near future, even assuming that the role the red tide may play in the ecology of the coastal and estuarine regions is perfectly understood.

Although red tides have been reported to cause mortality in shellfish (Tiffany and Heyl, 1978), the most common effect on clams and oysters in Sarasota Bay has been G. *breve* toxin accumulation, leading to closure of shellfish beds for public safety. This region of Florida's coast receives the bulk of the red tide outbreaks, and shellfish beds have been closed 34 times in Sarasota Bay (Sarasota and Manatee Counties) over the past 13 years (1978 through 1991), for a total of 1,750 days. The duration of closures ranged from 18 days to 156 days, with an overall average for the 12 year period of 52 days per year (FDNR, 1991). On a longer term record, between 1900 and 1980, red tides have been reported in 24 of the 80 years (30 percent) and more recently, from 1941 to 1980 in 18 of the 40 years (45 percent) (Estevez et al., 1984). Globally, the incidence and geographic distribution of toxic phytoplankton blooms appears to be increasing (Taylor, 1989) although this could be argued as merely an improvement in monitoring and reporting. Whether human impacts have contributed to this apparent trend is unclear.

Because red tides do not occur every year, the number and duration of closures vary from year to year. On the average for the last thirteen years, however, shellfish beds were closed due to red tide 37 percent of the time. In addition, red tides typically occur during the fall and winter. Since 1978, 13 of the 14 shellfish bed closures in Manatee County and 18 of the 20 in Sarasota County due to red tide have occurred during oyster harvesting months. This high percentage and seasonal timing of closures could present a serious problem for shellfishing interests if the Bay's shellfish resources were to be developed for the commercial market.

III.F. Ecological Requirements - Hard Clams

Mercenaria (hard clam or quahog) is a bivalve shallow water mollusk and is found from the Gulf of St. Lawrence, south to Florida, Cuba, and Mexico. Some species have been transplanted and form stable populations in California and England (Abbott, 1974). *M. campechiensis* is the southern species of the genus and is found from New Jersey south, while *M. mercenaria* occupies the northern portion of the range to as far south as the Gulf of Mexico. A subspecies, *M. mercenaria texana* is also identified in the northern Gulf of Mexico (Abbott, 1974).

The two species are very similar, ranges of the two overlap in regions, and reciprocal hybrids occur. Physiological differences exist between the two species, however, and include differences in growth rates and metabolism, even following acclimation. Most notably, *M. campechiensis* is reported to gape much more quickly during handling and transport from intolerance to refrigeration, thus lessening its commercial utility and value (Menzel and Sims, 1962; Otwell et al., 1986). Hybridization has attempted to reduce this trait and improve the marketability of live shell-stock in the southern ranges (Menzel, 1962; Menzel and Sims, 1962).

While some morphological distinctions, relative shell weight, smooth areas of the valve, color of the interior, etc., may be used to distinguish between species, the degree of hybridization, and the fact that a population of *M. mercenaria* were apparently released into Sarasota Bay (Ingle, 1949) make biochemical methods and the determination of isozymes the only reliable method of determining between species. While *M. campechiensis* may be assumed to be the dominant species in Sarasota Bay, this is by no means certain and the organism studied under this project will be referred to *Mercenaria* spp.

Contained between the two valves of the hard clam is the animal itself, composed of a mantle cavity (a flap of tissue which contains the gills), an exhalent and inhalant siphon (for food and water), a visceral mass (containing the vital organs, a mucus secreting gland, anus, excretory pore, reproductive opening, and chemical sensory organ), a muscular foot for burrowing, and adductor muscles attached to the valves which allow for opening and closing of the shell.

Feeding is accomplished by filtering phytoplankton out of the water column and specific size preferences are exhibited. An incurrent siphon draws in water to the gills for both respiratory and digestive purposes. Food particles are gathered and collected in mucus and passed on to the digestive system. Depending on the turbidity of the water, organic particulates other than phytoplankton can enter the digestive system.

The hard clam is a consecutive hermaphrodite, initially male until early spawning, when some become female. Both sexes are necessary for reproduction. Spawning varies with temperature and therefore geographical location, but Gulf Coast populations exhibit two seasonal peaks in both spring (December through April) and, secondarily, in fall (Dalton and Menzel, 1983). For *M. mercenaria,* spawning individuals may release more than 20 percent of their annual production (Ansell and Lander, 1967). Males may spawn during the first year of life (less than 15 mm shell length) (Loosanoff, 1937), but females are generally greater than 30 mm (Belding, 1931). Large *M. mercenaria* produce higher numbers of gametes proportional to body tissue weight than do smaller individuals (Peterson, 1983; Pline, 1984). In general, recruitment rates are poorly understood, but in some regions are thought to be less than 1 clam/m^2/year (Haskin, 1955, Haven et al., 1973).

After planktonic egg and veliger stages lasting up to 12 days, the pediveliger larvae crawl across the substrate to a suitable location and then mechanically anchor themselves by a byssal thread to the substrate (Carriker, 1956). Shell substrates with some detritus are optimal, with mud the least colonized (Carriker, 1961; Keck et al., 1974). At approximately 10 mm in length, the juvenile then begins to burrow, and adult hard clams bury from 1 to 2 cm in the sediment (Stevely et al., in press; Pratt and Campbell, 1956), using the siphon tube to reach the overlying water.

Growth rates vary tremendously, both within and between geographic regions, but in general is controlled by temperature, ceasing below 10°C and above 30°C (Pratt and Campbell, 1956; Ansell, 1968). Growth is also affected by sediment type, being higher in sand substrate and reduced with increasing percentages of fine particles (Pratt, 1953; Rhoads and Panella, 1970), and is probably related to the decreased feeding rates and efficiency and the energy expenditure of feeding in a turbid environment (Loosanoff, 1962). An increased density of clams is also associated with lower growth rates (Eldridge and Eversole, 1982) presumably from food competition. Clams within grassbeds are reported to grow more rapidly (Peterson et al., 1984; Arnold et al., 1991), perhaps due to alteration of current regimes (Fonseca et al., 1972) or selective predation (Arnold, 1984).

Rapid growth rates are noted for both *M. campechiensis* and *M. mercenaria* in Florida, with slightly higher rates for native southern clams. *M. campechiensis* in Boca Ciega Bay may reach 77 mm by the end of two years (Saloman and Taylor, 1969). Growth curves for a number of sites on the west coast of Florida, in general, predict a shell height of 35 to 70 mm by the end of the second year, with growth at most locations slowing substantially beyond five years of age. Mature individuals are predicted to range from 85 to 100 mm shell height with those in Boca Ciega Bay reaching 170 mm (Jones et al., 1990). As a result of this growth pattern, it is not unusual to see populations dominated by larger individuals, particularly in areas not subject to intense harvesting pressure (Haven et al., 1973; Hobbs et al., 1975).

Both exhibit rapid growth in spring and fall, and slow growth in winter, but *M. mercenaria* grows most slowly during the summer, while *M. campechiensis* continues relatively rapid growth (Menzel, 1961, 1962). The seasonal variation in growth rates is reflected in the growth increments visible in a radially sectioned shell, with lighter opaque areas indicating rapid growth, and the darker regions slower rates of shell deposition (Barker, 1964).

A brief summary is provided below of important environmental factors effecting hard clam survival, growth and reproduction. The synopsis is primarily drawn from three sources: Habitat Suitability Index

(Mulholland, 1984), species profiles (Eversole, 1987), and the Habitat Requirements for Chesapeake Bay Living Resources (Roegner and Mann, 1991) unless referenced otherwise. All studies where the less common *M. campechiensis* was used are noted, otherwise the hard clam involved in the study was *M. mercenaria.*

pH. Recruitment of *Mercenaria mercenaria* requires pH values greater or equal to 7.0 (Calabrese, 1972). Survival of embryos is reduced at 9.0. Normal growth occurs at 6.75-8.50. Values below 7.0 limit recruitment. Mortality experiments on adults suggested a minimum pH of 7.0. Reproduction was reduced above pH of 9.0. Values of pH may decrease below 7.0 in tide pools and estuaries with poor circulation, heavy siltation, pollution, and hydrogen sulfide production. Acceptable pH levels reported for clams in Chesapeake Bay were 7.0 to 8.75 for eggs and 7.5 to 8.5 for larvae.

Dissolved Oxygen. Morrison (1971) found normal growth of shelled veligers at 4.2 mg/l of dissolved oxygen (DO). Growth ceased at 2.4 mg/l and less. Embryos developed normally at 0.5 mg/l but suffered 100 percent mortality at 0.2 mg/l. All life stages can tolerate nearly anoxic conditions for long periods (Stanley and Dewitt, 1983). Optimum dissolved oxygen levels for the Chesapeake Bay clams were reported as greater than 5.0 mg/l.

Water Currents. Density of clams was greatest in Chincoteague Bay, MD, in currents of 30 to 50 cm/s (Wells, 1957). Faster growth was reported in areas with substantial flow (Kerswill, 1949). The growth was attributed to increased food supply. Sediment type can roughly be correlated with water current (Thorson, 1955).

Salinity. Eggs developed into veligers only between 20.0 to 32.5 ppt (Stanley and Dewitt, 1983). The optimum for egg development was 26.5 to 27.5 ppt (Davis, 1958). Growth was retarded at 22.5 ppt or lower (Castagna and Chanley, 1973). Development from veliger to seed clam was inhibited below 17.5 to 20 ppt. Minimum nonlethal salinity for adults was 12.5 ppt (Castagna and Chanley, 1973). Adults can withstand low salinities by closing shells. In South Carolina mortality was less than 5 percent when salinity was less than 10 ppt. for three weeks while oyster mortality was 50 percent (Burrell, 1977). Clams survived in laboratory experiments for 114 hours in freshwater (Pearse, 1936). Salinity ranges associated with hard clams in Chesapeake Bay were 20 to 35 ppt for eggs, 17 to 35 ppt for larvae and greater than 15 ppt for adults.

Temperature. Three temperature/salinity combinations yielded optimum growth: 30°C/22.5 ppt; 27.5°C/17.5 and 20 ppt; and 25°C/15 ppt. (Davis and Calabrese, 1964). Temperature tolerance increases with age (Kennedy et al., 1974). The Joint Committee on Aquaculture (1983) reported adult spawning between 22° and 28°C and continued growth between 8" and 28°C. Spawning peaks were observed in Florida (Menzel, 1976) when temperatures approached 22° to 24°C. Growth is slowest in summer when temperatures exceed 30-33°C (Menzel, 1963, 1964).

Substrate. Substrate type is the main factor for hard clam settlement (Thorson, 1955). Higher larval settlement was reported in sand than mud (Keck et al., 1974). Increased organic material in sediment may reduce settling because of increased bacteria, reduced dissolved oxygen and increased hydrogen sulfide. Adult *M. campechiensis* did not show consistent correlation with particle size in Tampa Bay (Sims and Stokes, 1967). Soft sediments were given as the principal factor limiting mollusks on bayfill canals in Tampa Bay (Sykes and Hall, 1970). In Georgia, clams were most abundant in sand with shell, next abundant in sand and least in mud (Walker et al., 1980). Sand with shell was not reported to be a common bottom type in South Carolina but supported 68 percent of the hard clams collected (Anderson et al., 1978). Johnson (1977) showed a correlation between clam size and sediment particle size. Pratt (1953) demonstrated that clams grew faster in sand than mud.

Suspended Solids. Eggs did not develop normally at 3.0 or 4.0 g/l suspended solids (Davis, 1960). Growth of veligers was normal at 0.75 g/l, retarded at 1.0 to 2.0 g/l and negligible at 3.0 to 4.0 g/l. Turbidity may have inhibited growth at one Florida site (Menzel, 1961). Increased gill clearing which expends energy and interferes with feeding (Pratt and Campbell, 1956) or increased pseudofeces expulsion (Johnson, 1977) may inhibit growth. Acceptable total suspended solid levels for hard clams were reported as less than 750 mg/l for eggs, less than 250 mg/l for larvae and less than 44 mg/l for adults.

Vegetation. An association of *M. mercenaria* with *Thalassia testudinum* was reported for Florida west coast estuaries (Schroeder, 1924; Woodburn, 1962; Sims and Stokes, 1967; Taylor and Saloman, 1968, 1970; Godcharles, 1971). Peterson (1982) found higher densities in partly vegetated plots than bare and higher densities in densely than in partly vegetated areas.

Other water quality parameters. Clams can tolerate a wide range of concentrations of ammonia, nitrite, nitrate, phosphates and sulfur compounds. Tolerance to nitrate and orthophosphate were reported so high that survival in secondary treated sewage was possible (Epifanio and Srna, 1975). Limits for ammonia were 110-172 mg/l and 1863-1955 mg/l for nitrite.

Predation. Predation appears to be the most important biotic factor regulating hard clam populations (Virstein, 1977; MacKenzie, 1979), although predation on large clams is low (MacKenzie, 1977). Predators include *Callinectes sapidus* (blue crab) which preys on newly settled and seed clams up to 40 mm (Arnold, 1984) and *Busycon* spp. (whelks) which are capable of opening larger clams. Mortality in the absence of predators is low (Eldridge and Eversole, 1982). Predation was seen to reduce experimental plantings in Florida and Georgia by 100 percent (Menzel and Sims, 1964; Godwin, 1968). Clam sets are reported to below in areas with dense populations of adult clams, presumably from filter-feeding activities (Thorson, 1966, MacKenzie, 1977).

Diseases and parasites. Polychaetes, nemerteans, and boring sponges are reported as parasites for hard clams, but are rarely fatal (Taylor and Saloman, 1972; Porter, 1962; Walker et al., 1980). There are few reported diseases (Sindermann and Rosenfield, 1967), but larval populations may be more susceptible.

III.G. Ecological Requirements - Oysters

The other species selected for the study, the eastern oyster, has qualities greatly different from the hard clam. Oysters are much more of an estuarine organism and are most prevalent in the areas of creeks. Also, unlike clams, oysters tend to be community organisms forming reefs or clumps. Within Sarasota Bay, by far the dominant habitat preferred is intertidal.

Crassostrea virginica, the American oyster, is one of two genera of oysters in the family Ostreidae which occur in waters of the U.S. and is distributed from New Brunswick to the Gulf of Mexico. *C. virginica* is the only species of the genus which appears on the east coast of the U.S., while C. *gigas,* typically quite a bit larger, and *C. columbiensis* and C. *cortziensis* are found on the Pacific coast. The genus is differentiated from other shellfish by the shape of its valves. Instead of both valves being cup-shaped as in the hard clam, the upper valve of the oyster is flat and the lower attached valve is larger and cupped. Although generally elongate, the valves can vary substantially in shape (Abbott, 1974).

Making up the internal body of the oyster is a mantle (containing the gills, mouth, excretory cloaca, and heart), visceral mass containing the reproductive and digestive organs, and a single adductor muscle for valve movement. Like hard clams, oysters have planktonic egg and larval stages, but hard surfaces are required for settlement. This enhances the formation of reefs as the larval spat settle on existing live or dead oyster shell. Reefs tend to develop perpendicular to water current direction,

These reefs are important not only for producing oysters but also affecting water circulation, serving in shoreline stabilization, preventing resuspension of fine sediments and providing habitat for other invertebrates. The most critical environmental factor for oyster survival are salinity fluctuations which reduce predators within the reef (Butler, 1954)

A brief synopsis is provided below of important environmental factors effecting oyster survival, growth and reproduction. Unless stated otherwise, information is drawn primarily from species profiles (Sellers and Stanley, 1984) and Habitat Requirements for Chesapeake Bay Living Resources (Kennedy, 1991).

Temperature. Temperatures above 30-34°C have been shown to impair growth (Davis and Calabrese 1964). The optimum temperatures for adults is 20-30°C. Studies in thermal plumes (Stone and Webster, 1985) found that oysters in the thermal plume had higher growth rates but higher mortality rates. Some regulation of temperature is attained through the communal nature of oysters, as the crowded, vertically oriented growth habits provide mutual shading during periods of low tides in summer.

Salinity. Salinities of greater than 7.5 ppt. are required for spawning (Loosanoff, 1948). Larvae can tolerate salinities from 3.1 to 30.6 ppt (Carricker, 1951). Adults can tolerate from 5 to 30 ppt. with an optimum range of 10 to 28 ppt. (Loosanoff, 1965b). Oysters have been observed to survive 3 ppt. for 30 days. Another study observed oyster mortality in Louisiana after 14 days at 6 ppt. (Anderson and Anderson 1975). The most deleterious salinities are associated with freshwater flooding over a period of weeks (Kennedy, 1991).

pH. Oysters failed to spawn at pH values below 6.0 and above 10.0 (Calabrese and Davis, 1969). Successful recruitment requires pH levels above 6.75 and normal embryonic development occurs at pH 6.75 to 8.75.

Water current. Water above an oyster reef must be renewed 72 times per 24 hours for maximum feeding. Moderate currents are required for oyster survival (Galtsoff, 1964). Excessive currents causing sand transport can damage shells, however, and increased turbidity results in decreased pumping rates (Loosanoff and Tommers, 1948). The tendency of oyster reefs to grow perpendicular to water current flow is indicative of the benefits of current on growth (Bahr and Lanier, 1981).

Suspended Solids. The eastern oyster is capable of withstanding erratic increases of turbidity and most studies of effects of sediment or turbidity on oysters have involved concentrations higher than those found in nature. Nelson found oysters feeding rapidly in water with up to 0.4 g (dry weight/liter) of suspended matter. Oyster eggs and larvae can be killed by suspended sediments (Davis and Hidu, 1969). A concentration of 0.5 grams per liter resulted in 69 percent mortality of eggs and 20 percent of larvae.

Dissolved Oxygen. Oysters have survived for up to 5 days in water with less than 1.0 mg/l presumably through anaerobic metabolism (Sparks et al., 1958). Larvae are able to avoid low dissolved oxygen concentrations by swimming upwards. Periods of days with anoxic waters over shallow oyster beds are probably not deleterious (Kennedy, 1991).

Predators. Numerous oyster predators exist but the most common predator in this area is *Melongena corona* (Florida crown conch) (Sprinkel and Gorzelany, 1985; Stone and Webster, 1985). The boring sponge *Cliona* spp. may also limit the success of subtidal distribution of oysters,

Substrate. The most suitable substrate for larval oyster settlement is oyster shell, as is indicated by the existence of oyster reefs. Oysters can grow on any substrate capable of supporting their weight, however. Oyster reef expansion is somewhat limited to areas with suitable substrate at intertidal levels where oyster predators are limited. This combination of conditions is easily lost through dredging and wetland destruction.

Disease. Two pathogens are capable of causing massive oyster mortality. They are *Dermocystidium marinum* ("dermo") and *Haplosporidium nelsoni* ("MSX"). Occurrence of these diseases has not been monitored in the Sarasota Bay area.

III.H. Contaminants - Ecological and Human Health Impacts

The following sections address in detail the documented ecological and human health impacts of the various contaminants selected for study under this project. The rationale for the selection of these specific contaminants appear later in Section III. Study Design.

III.H.1. Bacteriological Contaminants

Water contact and the consumption of raw shellfish have been identified as one of the major routes for infectious disease transmission to humans in the marine environment (SCAG, 1988) and the pathogens of most concern are those associated with human fecal wastes, such as viruses causing gastroenteritis or Hepatitis A. The correlation between sewage pollution of shellfish waters and outbreaks of disease are well documented (Verber, 1984) and occur since shellfish, which feed on phytoplankton, can also concentrate bacterial or viral organisms. Little is definitively known regarding the effect of human pathogens on the shellfish themselves.

Due to the difficulty and expense of identifying the actual pathogens, however, public health management decisions have traditionally been based on the presence of fecal coliforms as indicators of sewage. Their use is limited by not accurately predicting illness and by their relative susceptibility to chlorination or other sanitation treatments which the viruses do not necessarily share. Attempts are underway to rectify this deficiency with a national shellfish indicator study to develop direct detection methods for enteric viruses and for their indicators (LUMCON, 1991).

Viruses, in particular, have also been reported to persist longer in contaminated shellfish than do fecal coliforms (Canzonier, 1971) and the differing survival rates in the marine environment also compound the difficulty. Some pathogens can persist for a number of days in marine waters or aquatic sediments (Hunt, 1971), but the current bacteriological standards have been generally effective in preventing major outbreaks of disease (FDA, 1989).

Many sources of fecal coliform to the marine environment have been identified, including inadequately treated or raw sewage effluent, improperly functioning septic systems, urban runoff, boating and shipping activities, agricultural runoff, industrial discharges with human waste components, and wildlife. The process of classification of shellfishing waters take these sources into account.

Currently, waters are approved for shellfishing if the fecal coliform median or geometric mean most probable number (MPN) in the water does not exceed 14 per 100 ml of seawater and not more than 10 percent of the samples tested exceed 43 per 100 ml for a five tube dilution test (NSSP, 1991). Fresh and frozen shellfish meats are approved for consumption if the fecal coliform MPN does not exceed 230 per 100 g of tissue and a 35°C aerobic plate count is no more than 50,000,000 per 100 g (NSSP, 1991). Many studies, however, indicate no consistent relationship between water column and shellfish tissue concentrations of coliforms or pathogens (Hunt, 1979).

In addition to fecal pathogens, marine vibrios have also been responsible for a number of shellfish associated illnesses. Some reports (Colwell et al., 1987; Tamplin et al., 1982) have suggested that some potentially pathogenic vibrios are endemic to the estuarine ecosystem and not necessarily of recent human fecal origin and others have detected these organisms in approved shellfishing areas with no fecal contamination and no coliform bacteria (Blake and Rodrick, 1983). The strain of vibrio responsible for cholera has been reported to exist as a free-living organism in the U.S. Gulf Coast (Morris and Black, 1985) although none of the V. cholera-01 detected in shellfish samples were able to produce cholera toxin (Twedt et al., 1981).

Vibrio infections have been most prevalent during summer and fall when waters were warmer (greater than 15-20°C, Baross and Liston, 1970), with roughly 90 percent of the Florida cases reported between April 1 and October 1. In some areas, incidence of V. *vulnificus* in oysters approaches 100 percent during these warmer periods (Tamplin, 1990). While oyster harvesting is restricted during these warmer months, there is no closed season for clamming. Vibrios multiply rapidly following harvest, particularly if holding temperatures are greater than 10°C or transport times extended. While extended refrigeration reduces these densities, it does not completely eliminate the organism from processed shellfish meats (Ruple et al., 1989).

Individuals with specific health problems (liver disease, blood disorders, alcoholism, immunosuppression, malignancy) are apparently more susceptible to these infections (Table 1) and can be subject to a 50 percent mortality rate from *V. vulnificus.* It should be emphasized that these mortality rates are for persons with existing serious health problems and that 85 percent of individuals who were culture-positive for *V. cholerae* non-01 were asymptomatic (Lowry et al., 1989). Collectively, vibrios have been responsible for nearly 14 percent of the bivalve shellfish associated illnesses between 1978 and 1987 for which the causative agent has been identified (NAS, 1991).

For many of the vibrios, the dominant strains do not produce the toxins associated with virulence (Thompson and Vanderzant, 1976). As most fatalities associated with vibrios have involved shipped shellfish (ISSC,

Table 1. Risk factors for *Vibrio* illness associated with seafood consumption.

SPECIES	RISK CATEGORIES
V. cholerae 01	Achlorhydria, subtotal gastrectomy
V. cholerae non-01	Cirrhosis, subtotal gastrectomy, hemochromatosis, immunosuppression
V. hollisae	Not well known
V. mimicus	Not well known
V. paraphaemolyticus	Hemochromatosis, immunosuppression, cirrhosis
V. vulnificus	Hemochromatosis, immunosuppression, cirrhosis, diabetes

1988), measures for prevention of vibrio illness (Table 2) include education of at risk groups, minimization of temperature and time in transport, as well as thorough cooking of shellfish meats. Other control measures include enhanced depuration methods (Rodrick, 1990), or irradiation, although depuration rates of marine viruses, vibrios and other enteric bacteria may differ (Sobsey, 1990).

Vibrio dose response relationships are unknown at this time (NAS, 1991) and while rapid methods for detection of pathogenic species exists, the detection of virulence is a lengthy process. As a result, the utility of field screening or harvest control based solely on vibrio counts is questionable at this time.

There are ten species of vibrios that have been isolated from shellfish or estuaries and cause gastrointestinal and wound infections (Table 3) and all have been associated with oysters in particular. The most well known of the vibrios, *V. cholerae* is receiving widespread public attention since cholera in some South American countries has reached epidemic proportions. Since 1979, over 50 cases of oyster-associated cholera and other vibrio-related illnesses have occurred in Florida. Nationally, *V. cholerae* non-01 was responsible for slightly over 5 percent, or 120 cases, of the pathogenic bivalved shellfish illness of known etiology between 1978 and 1987, followed by *V. vulnificus,* and *V. parahaemolyticus* (NAS, 1991). Table 4 lists the various reservoirs for these vibrios and emphasizes that these organisms are ubiquitous in the marine environment. From an ecological rather than human health standpoint, selected species of vibrio *(V. alginolyticus)* are known to be pathogenic to the larval stages of bivalves (Tettlebach et al., 1984) and to cause substantial mortalities at oyster hatcheries (Leibovitz, 1979).

At least three species of aeromonas *(A. hydrophila, A. sobria,* and *A. cavial)* are also suspected as seafood associated pathogens responsible for gastroenteritis, and are known to cause serious infections or septicemia from contact with untreated surface waters (Burke et al., 1983). These organisms, together with vibrios, are members of the genus *Vibrionaceae.* Of the three species, *A. hydrophila* occurs widely and is recovered from such diverse sources as swimming pools, fresh and salt waters, and estuarine fish and shellfish, and appears to be a normal component of estuarine bacterial populations (Ward and Hackney, 1991). Densities in both water and shellfish increase during the warmer months. The organism can infect all human organ systems, but is most common in the gastrointestinal tract and the bloodstream. Similar to the disease incidence noted for vibrios, individuals with compromised immune systems or liver disease are more at risk.

As do the vibrios, these organisms also appear to be naturally occurring aquatic bacteria, and some strains appear to be virulent, with others producing no symptoms. Between 1978 and 1987, 0.3 percent of the total pathogenic shellfish illnesses for which the etiology was identified were attributable to these organisms. Approximately 70 percent of all shellfish illnesses reported by the FDA's Northeast Technical Support

Table 2. Measures for the prevention of *Vibrio* illness associated with seafood consumption.

SPECIES	Fecal Coliform Standard	Adequate Cooking	Depuration	GMP's*	Educating Risk Groups
V. cholerae 01	±	+	±	±	+
V. cholerae non-01	-	+	±	-	+
V. hollisae	?	+	?	?	+
V. mimicus	?	+	?	?	+
V. paraphaemolyticus	±	+	±	±	+
V. vulnificus	-	+	±	±	+

*Good Manufacturing Practices (GMP)

Table 3. Vibrio species associated with human disease.

SPECIES	DISEASE	TRANSMISSION
V. alginolyticus	Septicemia, wound infection	Raw oysters, clams, seawater
V. cholerae 01	Gastrointestinal, wound infection	Boiled crabs, shrimp, seawater, turtle, raw oysters
V. cholerae non-01	Gastrointestinal; Septicemia	Crab, shrimp, turtle, seawater, raw oysters
V. damsela	Septicemia	Raw oysters, seawater
V. fluvialis	Gastrointestinal	Raw oysters, seawater
V. furnissii non-01	Gastrointestinal	Raw oysters, (cooked seafood), seawater
V. hollisae	Gastrointestinal	Raw oysters, seawater
V. mimicus	Gastrointestinal	Raw oysters, boiled crayfish, seawater
V. metschnikovii	Septicemia	Raw oysters, seawater
V. parahaemolyticus	Septicemia, wound infection, gastrointestinal	Cooked crabs, shrimp, seawater, lobster, raw oysters
V. *vulnificus*	Septicemia, wound infection, gastrointestinal	Raw oysters, seawater

Table 4. Reservoirs for the various species of *Vibrios*.

SPECIES	RESERVOIR
V. cholerae 01	Man, freshwater, sea water, oysters, fish
V. cholerae non-01	Man, birds, sea water, oysters, clams, fish
V. fluvialis	Gulf coastal water, oysters
V. furnissii	Gulf coastal water, oysters
V. hollisae	Coastal waters, oysters, clams
V. mimicus	Coastal waters, oysters, clams
V. paraphaemolyticus	Coastal waters, oysters, clams, crabs, shrimp, finfish
V. vulnificus	Coastal waters, oysters, clams, finfish, shrimp

Unit, however, are of unknown etiology, and so numbers of actual cases by organism are thought to be much higher than reported.

III.H.2. Metal Contaminants

III.H.2.a. Metals - Sources and Human Health Impacts

While previous Food and Drug Administration (FDA) publications presented seafood tissue standards which addressed mercury (7.0 µg/g), lead (0.5 µg/g), and cadmium (1.0 µg/g) (FDA, 1982), current rules set formal tolerance levels for PCB's alone (2 mg/g), with action levels established for ten classes of organic compounds and mercury of which only seven apply to shellfish (NAS, 1991). These limits appear in Table 5. In addition, FDA tolerances or action levels are applied to production and trade in seafood. In commerce, any one retailer may have products from many different geographic regions, and so the action levels may not be effective in protecting recreational fishermen who may consume shellfish from a restricted area over a long period of time.

The National Shellfish Sanitation Program has, with the participation of the FDA and state agencies, established (although never formally adopted) guidelines or alert levels for several metals in both oysters, hard and soft shell clams. These values were established, however, not upon toxicity data, but on a survey of average metal concentrations from 1975 (NAS, 1991).

One difficulty in controlling human exposure through limits on shellfish tissue concentrations is that average seafood consumption rates may vary widely with population and geographic region. Seafood consumption (including both fish and shellfish) has been reported to range between 6 and 100 g/day (Landolt et al., 1987). Maximally exposed individuals in the northeast have been estimated to consume 165 g/day of which 16 g/day was soft-shelled clams, while "typical" consumers average slightly over 3 g/day of mixed (fish and shellfish) seafood (Metcalf & Eddy, 1988). Rhode Island residents consume an average of 1.2 g/day of quahogs, up to a maxima of 15.0 g/day, while national average consumption rates for fish and shellfish combined is between 15 and 19 g/day (Kipp, 1990; NAS, 1991). National average consumption rates for oysters and clams from commercial landings data in 1987 averaged 0.37 g and 0.68 g/day, respectively, and included both domestic and imported organisms (NAS, 1991).

For metals, recent work (NAS, 1991) identifies arsenic, cadmium, lead, mercury, and selenium as those metals with potentially adverse human health effects from seafood consumption, based on persistence, bioconcentration potential, human toxicity, concentrations observed in seafood, and gastric absorption potential. Copper, iron, manganese and zinc are viewed to have a lesser potential for toxicity and there is consequently less information on toxic responses.

Arsenic

Arsenic compounds are quite prevalent in the environment, and are typically present as organic compounds. Anthropogenic inputs to the environment exceed inputs from natural weathering by a factor of three. Increases in this element in the environment typically occur as the result of smelting operations, fossil fuel combustion, ceramics and glass industries, herbicide application, and other industrial uses. As of 1988, over 80 percent of the arsenic manufactured was used in agricultural products such as insecticides, fungicides, wood preservatives, plant growth stimulants, dyes, and veterinary medicines (Eisler, 1988).

Health impacts of arsenic include mutagenic, teratogenic and carcinogenic effects (Nagymajtenyi et al., 1985) and correlations between cancer, bronchitis, and pneumonia have been established with airborne arsenic compounds (NRCC, 1978). Other chronic and acute effects include skin cancer, gangrene of hands and feet, hearing loss, liver kidney and heart damage, and neurological abnormalities (Pershagen and Vahter, 1979).

Absorption of arsenic following ingestion is between 70 and 90 percent (NAS, 1991). The organic forms of arsenic present in fish have not been reported to be toxic (NAS, 1991; ATSDR, 1989) and are certainly less toxic than the inorganic trivalent arsenic compounds. As a result, the toxicological impacts of consuming arsenic in seafood are relatively unknown.

Cadmium

Cadmium is a relatively recent environmental contaminant (NAS, 1991). Natural levels of cadmium are enhanced via municipal effluents and industrial discharges from pigments, plastics, and alloy processes (USEPA, 1980b). Uses of the element include battery manufacture, dental amalgams, pigments in paints, galvanizing and electroplating. Cadmium is also noted in runoff from agricultural areas using phosphatic fertilizers (Phillips and Russo, 1978).

Cigarette smoke and food intake constitute the major non-occupational exposure route for this metal (Klaassen, 1986; McKenzie et al., 1986). Uptake in humans via the gastrointestinal tract is approximately 5 percent, while higher for pulmonary exposure (Metcalf & Eddy, 1988). Even in populations with high reported dietary intakes of cadmium from marine foods, smoking was reported to be the major factor determining blood cadmium levels (Hansen, 1990).

Human health impacts from long-term ingestion of cadmium include elevated protein levels in urine accompanied by renal dysfunction, kidney stone formation, and mineral metabolism disturbances resulting in osteoporotic conditions. At higher concentrations and exposures of cadmium, teratogenicity, and reduced fertility have been observed in rats (USEPA, 1984c). Exposure through inhalation has classified cadmium as a probable human carcinogen, but there is no corresponding data for effects of ingestion (Metcalf & Eddy, 1988).

Copper

Elevated levels of copper in the environment are frequently the result of corrosion byproducts, sewage treatment plant discharges, and other industrial uses. Algacides and fungicides frequently contain copper compounds and copper based paints have long been the mainstay of antifouling efforts in the marine environment. Similar to other metals, copper solubility is controlled by concentrations of complexing agents, pH, temperature, and other variables and absorbs onto clays, sediments, and organic particles. Many of the complexes appear to be non-toxic (USEPA, 1980c).

Copper is an essential element for both humans, plants, and crustaceans alike, and, compared to elements like mercury and cadmium, is considered fairly low in toxicity. Symptoms of acute copper toxicity include gastric disturbances, vomiting, jaundice and coma. Insufficient data exist to classify this element for carcinogenicity, if any, and no human teratogenicity has been reported from oral exposure.

Lead

There are numerous industrial uses of lead due to its corrosion and radiation resistance. It is commonly used in electronic equipment, batteries, pigments and plastics, as a gasoline additive, in electroplating processes, and in construction materials. Spent ammunition evidently contribute to elevated lead levels in feeding waterfowl. The prevalence of uses results in relatively high levels of "background exposure" to this element in food and water consumption, particularly from gasoline combustion. There is some evidence of increasing environmental concentrations of lead within the last 200 years (Shukla and Leland, 1973). High background levels in human blood have also been observed in extremely non-industrial areas and appear to be unrelated to dietary intakes. This would appear to indicate that atmospheric deposition plays a role in the human exposure to this metal (Hansen, 1990).

Adult human absorption of lead from the gastrointestinal tract is estimated at between 5 and 15 percent (USEPA, 1984e; Goyer, 1986). Increasing absorption with decreasing age place the young at particular risk. Anemic responses from increased blood cell fragility, neurological effects, learning disabilities, and histological renal changes have all been established. The half life in bone is roughly 20 years, in blood, approximately a month (NAS, 1991). Based on some animal evidence, lead is classified as a possible human carcinogen.

Mercury

This element is used in lighting appliances, in photography, paint manufacture, fungicides, and dental amalgams. Industrial discharges and mining are responsible for much of the environmental mercury levels. Much of the distribution occurs through the atmosphere, followed by removal through precipitation (Metcalf & Eddy, 1988). The pulp and paper industry, chlor-alkali industry and combustion of fossil fuels also contribute mercurial compounds (Phillips and Russo, 1978).

Human absorption of metallic mercury is approximately 0.01 percent, while methyl mercury is essentially completely absorbed (USEPA, 1984f; NAS, 1991). Chronic and acute effects of mercuric salts and methylated mercury include chromosomal alterations, kidney damage, as well as the well-documented central nervous system impairment as observed at Minimata Bay, Japan. Teratogenic effects and neurological defects have been reported in animal studies. Renal effects are the most sensitive indicator.

Zinc

Zinc is a ubiquitous metal with a multitude of uses, including alloys, electronics, pharmaceutical, and pigments. Elevated concentrations of zinc are added to the environment through corrosion products of galvanized metals, discharges from metals plating industries, and from urban runoff.

Zinc is an essential element for both humans and animals and has a very low toxicity to man overall (Phillips and Russo, 1978). Overexposures through inhalation have resulted in pneumonia-like symptoms (Berry et al., 1974).

III.H.2.b. Metals - Ecological Impacts

For metals, bivalve uptake routes can include both from solution and from food particles. Absorption of dissolved species can be passively accomplished at exposed permeable membranes (mantle, gills, gut) (Rainbow, 1990) and is generally thought to involve the free metal ion (Waldichuk, 1985; Nelson and Donkin, 1985) with metals subsequently sequestered by metal binding proteins such as metallothioneins or sequestered in granular form (Mason, 1988). For oysters, granules containing either copper and sulfur or zinc and phosphorous reduces the toxic effects of these ions and also allows high accumulations of these metals. Other molluscan granules contain calcium phosphate and numerous other metals (Viarengo, 1989). Lipid soluble complexing agents may also increase biological uptake by allowing metal complexes to cross lipophilic membranes.

For those metals subject to uptake from solution, the factors which affect speciation and free ions present (ionic strength, salinity, pH, EH, presence of dissolved organics and other chelating agents, suspended sediment) will influence metal bioavailability (McClusky et al., 1986; Ahsanullah and Florence, 1984; Elder, 1988). While anoxic conditions in pore waters are often accompanied by high metal levels, formation of relatively low solubility metal sulfide compounds can act to offset this increase in free ion concentrations.

Variations in the physiological responses (pumping rates, metabolism, etc.) of the organism to the specific environmental variables will also play a role. In general, high temperature and low salinity increase bioavailability (for cadmium, lead, mercury, zinc) while availability is reduced in the presences of sediments or other natural organic chelators (Langston, 1990). Contaminant uptake from solution generally produces an inverse relationship of body size to tissue concentration, as larger animals have a lower surface area to volume ratio (Boyden, 1985). For *C. virginica,* nickel has been shown to decrease with increasing size (Zaroogian and Johnson, 1984).

Metals sorbed onto particulates can become available to the filter-feeding organism during ingestion and digestion, although the total metal concentration of food may not accurately reflect metal availability due to species differences in digestion, and interactions with other metals (Luoma and Bryan, 1978). If the primary mode is through ingestion, however, a slight increase in contaminant concentration with size is observed (Elder, 1988).

Excretion of metals, in addition to occurring passively as environmental variables change, can also take place via the excretion of metal rich granules from the kidneys of bivalves (Bryan, 1976), although this process is slow. Rates of uptake and excretion are what defines the time period over which species reflect ambient conditions. Half lives, or the time period required for half of the body burden of contaminant to be excreted, are typically longer for metals than for organochlorines or hydrocarbons (Phillips and Segar, 1986) and for *Crassostrea virginica* range between 70 and 180 days (Okazaki and Panietz, 1981).

Accumulation reflects the net results of exposure, uptake, excretion, as well as any degree to which tissue concentrations are "diluted" by increasing size of the organism (Rainbow, 1990). Usual patterns of accumulation indicate that body burdens correlate with ambient metal bioavailability, particularly well illustrated by zinc accumulations in select oyster species (Phillips and Yim, 1981). On the other hand, some species have the ability to regulate body concentrations, maintaining a relatively constant value under varying ambient conditions of exposure. Regulation is most typically observed for essential metals such as copper or zinc, and in bivalves, has been demonstrated for mussels, *Mytilus edulis* (Amiard et al., 1987).

Accumulation of metals also varies with a number of other factors, including size or age, seasonal variation in either physiological processes or contaminant loads, sex and reproductive status, temperature and salinity, and the vertical position on the shoreline (Phillips, 1990; NAS, 1980; Paes-Osuna and Marmolejo-Rivas, 1990).

Bioconcentration factors (BCF), or concentration in tissues divided by the concentration in exposure water, are the net result of all variables to contaminant uptake and excretion. As summarized by Dillon and Gibson (1985), the metals studied in this project can be separated into four differing categories, on the basis of mean results from a number of studies on both fresh and saltwater organisms. The arsenic BCF averaged between 1 and 100, lead between 100 and 1,000, cadmium, zinc, and copper were between 1,000 and 10,000, while mercury BCF was in the 10,000 to 100,000 range. Specifics on these elements are described below.

Toxicity of metals rarely extends to lethal effect, except in a few heavily polluted areas, but in general occurs when excretion, storage and detoxification mechanisms fail to meet or exceed uptake rates (Langston, 1990). These processes, in addition to strong variations between species, also vary with life stage and size of the particular organism, with embryos and larvae being notably more sensitive to contamination (Viarengo, 1989). Toxicity effects can be evidenced through either biochemical or whole animal responses (both growth and morphology). Production of metallothionein and other metal binding proteins are viewed as an indicator of stress and toxic effects may occur when the metal supply exceeds the binding capacity of these proteins (Langston, 1990). Impaired lysosomal function and links to clearance rates and growth have also been demonstrated (Moore et al., 1984)

The relationship of biota tissue contaminant concentrations to sediment or water levels are complicated by the numerous and competing processes controlling solubility and bioavailability. Physiological processes in the organism itself further complicate the picture, with numerous species differences apparent. Some work has illustrated declines in tissue concentrations when organisms were removed from sediment contact (Rubenstein et al., 1984) and others have observed that despite all variables, organisms exposed to increased metal levels in the water column respond with an increase in tissue concentrations (Bender et al., 1989).

Both oysters and clams have been reported to exhibit reduced growth and larval toxicity at water column concentrations of copper, mercury, zinc and cadmium which may be found in the environment (Calabrese, 1977). Copper, mercury, and silver have also been implicated in the formation of abnormal larvae for several bivalve species (Marten et al., 1981; MacInnes and Calabrese, 1978). Reduced fecundity has also been observed in mussels at relatively high concentrations of copper and zinc (Myint and Tyler, 1982). Reduced filtration rates or burrowing behavior have also been demonstrated (Bayne et al., 1985; McGreer, 1979) and impaired settlement and survival of bivalves noted near a sewage outfall with only moderate levels of metals (McGreer, 1982). Antifouling paints, particularly organotin based compounds, have also been observed to produce shell abnormalities, reduced growth and lower recruitments in the oyster, *Crassostrea gigas* (Alzieu and Heral, 1987).

Once in the environment, arsenic is subject to both redox reactions as well as microbiologically mediated methylation. In the aqueous phase, arsenates are strongly sorbed by colloidal humic material, under conditions of low pH, phosphate and mineral content (USEPA, 1980a; Thanabalasingam and Pickering, 1986). Arsenic is accumulated by shellfish to a much greater degree than in fishes, and concentrations in marine species usually contain more arsenic than freshwater organisms (Phillips and Russo, 1978). Lipophilic, arsenic tends to concentrate in fatty tissues.

In freshwater, cadmium ions are relatively mobile, but increasing salinity reduces the number of species and cadmium chloride complexes are thought to dominate with strong absorption to organic materials and suspended particulates (USEPA, 1980b). Bioaccumulation of cadmium has been reported for many aquatic species with bivalve shellfish bioconcentration factors exceeding those for fish and other shellfish (Eisler, 1981). In particular, bivalve tissues were noted as failing to reach a steady state concentration under fixed water column conditions (USEPA, 1980b).

Oysters have been reported to accumulate cadmium at substantially higher rates than many other marine species and indeed than other bivalves (Eisler et al., 1972; Talbot et al., 1976). Cadmium toxicity symptoms in oysters include emaciation, discoloration, followed by death at 100 µg/l of exposure (Shuster and Pringle, 1969). Uptake among a number of bivalves has been reported to increase with increased temperature and to decrease with increasing salinity (Jackim et al., 1977).

Copper uptake in oysters appears to be proportional to the exposure level (Shuster and Pringle, 1969) and bioconcentration factors for this element high. While uptake of copper in fish does not appear to result in tissue concentrations harmful to humans, the high enriching tendency of oysters could pose some concern.

The mobility of lead in natural surface waters is restricted by the number of insoluble compounds formed with carbonate, hydroxide, and sulfate ions. Solubility also decreases with increasing pH (USEPA, 1980d), which is significant for transport in the relatively high pH marine environment.

Lead concentration in *Crassostrea virginica* has also been shown to be proportional to exposure concentrations (Shuster and Pringle, 1969). Again, while fish tissues do not accumulate significant quantities of lead in edible tissues, bivalve bioconcentration factors. can produce unacceptable lead concentrations.

Mercury in the environment is typically oxidized and then methylated by both aerobic and anaerobic bacteria with the methylated form being the most toxic. Methyl mercury not only bioaccumulates, but is directly toxic. Depuration is extremely slow with half-lives of two to three years, and bioconcentration factors are quite high, near 40,000 for oysters (USEPA, 1980e).

Zinc toxicity to fish decreases with increasing pH and increasing water hardness which would indicate reduced toxicities in the marine environment. Concentration factors in Pacific bivalves, *Crassostrea gigas,* reached 15,000 to 25,000 (Seymour, 1966) and in *Crassostrea virginica* and *Venus mercenaria,* the highest zinc concentrations were observed in those tissues with a large exposed surface area (Wolfe, 1970). The half life of zinc in oysters is on the order of eight months.

III.H.3. Toxic Organics

III.H.3.a. Pesticides Environmental Distribution

Chlorinated pesticides are persistent, lipid-soluble, synthetic chemicals that are toxic to a wide variety of aquatic organisms, as well as humans, and in some instances are carcinogenic. Although originally developed for their persistent, long term action, this persistence has resulted in environmental contamination, leading to banning or severely restricting the use of most of the chlorinated pesticides over the past 15 years.

The chlorinated pesticides have been replaced with less persistent, yet often more toxic organophosphate and carbamate pesticides. These pesticides generally do not persist in the marine environment for years; however, they do persist for weeks to months and may have a short-term impact following local applications and stormwater runoff.

Studies of fish and shellfish during the late 1960's and early 1970's showed widespread contamination throughout the nations coastal regions, with specific "hot spots" along California, Mobile Bay, Delaware Bay, and one site in southeast Florida. The most abundant chlorinated pesticides detected were DDT and its metabolites, including the various isomers of DDT, DDD and DDE, as well as dieldrin and chlordane isomers. A number of other pesticides reflected more localized usage (Butler, 1974; Mearns et al., 1988). Concentrations of DDT were found in 63 percent of more than 8,000 tissue samples during the first comprehensive national survey conducted by NOAA's National Pesticides Monitoring Program (NOAA, 1988). The time averaged 180-site mean concentrations ranged from 0.01 µg/g wet weight (lower limit of detection, LLD) to 1.4 µg/g wet weight, and the 180 site median was 0.024 µg/g wet weight (0.24 µg/g dry weight) (Butler, 1973).

Following the ban of DDT in 1972, subsequent studies of fish, shellfish and sediments along the nation's coastal region have shown a marked decline as a direct result of reduction in use (Butler et al., 1978; Mearns et al., 1988). A follow-up study of the same sites in 1977 showed a dramatic reduction in DDT to <0.01 µg/g wet weight in all but a few sites along California and Delaware (Butler et al., 1978).

A summary of the recent NOAA Status and Trends Mussel Watch study of chemical contaminants in tissues from 1986 to 1988 shows DDT concentrations in oysters and mussels ranging from below a detection limit of approximately 0.001 µg/g dry weight to 1.3 µg/g, indicating that at least several sites still contain considerable amounts of DDT and metabolites (NOAA, 1989). Yearly means of oyster samples from estuaries along the southwest Florida coast, from Rookery Bay to Tampa Bay, ranged from <0.001 in parts of Charlotte Harbor and Tampa Bay to a maximum of 0.12 µg/g dry weight in another part of Tampa Bay.

Other pesticides of concern detected in southwest Florida estuaries during the 1986-1988 Mussel Watch study were total chlordane (<0.001 to

0.11 µg/g dry weight), dieldrin (<0.001 to 0.018 µg/g dry weight, γ-BHC (lindane) (<0.001 to 0.003 µg/g dry weight) (NOAA, 1989).

III.H.3.b. Pesticides' Biological Effects

Most toxic effects evaluations are derived from the amount of toxic substance to which an organism is exposed, as in water column concentrations. Assessing biological effects from body burdens in shellfish is complicated by variations in the duration and intensity of exposure and by different rates of metabolism/depuration. Bioconcentration factors (BCF) provide insight to the amount of pesticide to which an organism has been exposed which, in conjunction with body burden data, may provide inferences regarding potential biological effects.

Acute toxicity levels (LC_{50}, or lethal concentration for 50 percent of the test animals) for many chlorinated and other pesticides are in generally in the mg/l range for hard clam-eggs and larvae. Hard clams also accumulate these compounds but more slowly and with higher depuration rates than many species (Roegner and Mann, 1991), including soft shelled clams (Butler, 1973).

Bivalve mollusks concentrate chlorinated pesticides from the water, sediment and food, although not to the extent that many other species do (Roegner and Mann, 1991). The bioavailability of a pesticide represents how available the chemical is for accumulation in an organism, an important factor for exposure, accumulation and potential harmful effects (NOAA, 1990a). Adsorption of DDT to organic particulates in water was shown to be an important mode of transport and accumulation in sediments (Pierce et al., 1974). Adsorption to sediments inhibits bioconcentration in marine organisms that do not ingest the sediment, where as those organisms feeding on sediment and their predators have been found to exhibit bioconcentration via sediment contamination (Funderburk et al., 1991).

Variations in BCF among shellfish species was observed by Butler (1973) reporting that soft shell clams accumulated a variety of pesticides in greater concentrations than did hard shell clams exposed to the same mixture. Hard clams exposed to 1 µg/l DDT in water for seven days contained $6x10^3$ µg/g (BCF of 6,000) (Butler, 1966). Oysters were reported to accumulate chlorinated pesticides from water, with 90 percent depuration in four days (Hansen et al., 1976). BCF for DDT in hard clams was near 2,000 with depuration times of a little over 3 months (Courtney and Denton, 1976). Other work has identified a BCF for DDT at 6,000 (Butler, 1966). Another study showed BCF in *M. edulis* to vary with exposure concentration, with an average BCF of 1.8×10^3. Subsequent depuration of DDT required about three months (Courtney et al., 1976). Mussels did not accumulate high levels of chlorinated hydrocarbons from suspensions of contaminated sediment (Peddicord, 1980; Olsen and Adams, 1984).

Sublethal effects of chlorinated pesticides create stress on *Mercenaria* through interference with enzyme pathways and reduced formation of glucose (Engle et al., 1972). Effective concentrations (EC_{50}, the concentration where effects are first noted in 50 percent of the organisms) range from µg/l to mg/l for *Crassostrea* (Kennedy, 1991). Again eggs and larvae are more susceptible than juveniles and adults.

Concentrations of chlorinated pesticides found during the NS&T Mussel Watch study generally were not high enough to pose an acute toxicity problem; however, the concern for sublethal effects from chronic exposure must be considered. Embryos and larvae of the hard shell clam *M. mercenaria* are very sensitive to low concentrations of pesticides, and clam larval survival is used as a yardstick to determine biological effects of contaminated sediments (NOAA, 1990a).

Although the organophosphate and carbamate pesticides are not as persistent as most organochlorines, their acute toxicity to aquatic organisms could produce considerable impact to populations over a short period of time. Dibrom has been reported to be highly toxic to clam larvae (Lipsey et al., 1987). Acute toxicity of chlorpyrifos to three species of shrimp and one species of crab ranged form 0.2 to 5.2 µg/l (ppb) (Lowe, 1980). The half-life of chlorpyrifos in marine sediment (24 days), coupled with its high acute toxicity, implicate this chemical as a potential hazard for benthic species (Schimmel et al., 1983).

III.H.3.c. Pesticide Human Health Impacts

The greatest health risk associated with shellfish consumption is from viruses and enteric bacteria, followed in order of importance by natural biotoxins (i.e., ciguatera, PSP, NSP, DSP), and then chemical contamination (NAS, 1991). Of the chlorinated hydrocarbon contaminants, PCB's have proven to be the most persistent over the years. Most health impacts documented for these compounds are from acute and chronic exposures at very high levels (agricultural workers, spray operators, factory workers) or accidental exposures through consumption of flour ground from treated seeds (Rathus, 1973).

DDT and its metabolites are persistent, lipophilic substances which tend to bioaccumulate with increasing trophic level. Neurological, liver toxicity and tumors, kidney damage, and estrogenic effects have been observed in rodents and humans at non-acute exposures, while acute exposures act as a central nervous system stimulant and induce tremors and convulsions (Diechmann, 1973).

Dieldrin also effects the neurological system and produces liver tumors in rodents, but is more toxic than DDT and is responsible for more human fatalities. Carcinogenic and fetal toxicity has been observed in mice and rats (ATSDR, 1984). Acute dieldrin intoxication has resulted in epileptic type seizures (Rathus, 1973). Aldrin and endrin are similar in toxicity to dieldrin and portions are metabolized to dieldrin within mammalian systems (Diechmann, 1973). These compounds are also retained in fatty tissues.

Heptachlor and heptachlor epoxide are similar in toxicity to dieldrin, while chlordane is less so (NAS, 1991). The BHC isomers act as neurotoxins, with lindane the most toxic, and can cause excessive central nervous system stimulation and anemia in humans. Chlordane is also listed as a probable human carcinogen with significant incidence of hepatic tumors induced in animal studies, but the compound fails to bind to DNA (Shigenaka, 1990). One aspect of the lipophilic compounds (DDT, dieldrin, and heptachlor and the epoxide) is that when fat reserves are consumed, the newly released compound can then produce toxic results (NAS, 1991).

Organophosphate insecticides have varying mammalian toxicity, but in general act through inhibition of cholinesterase, as do carbamates (Rathus, 1973). In acute exposure, some of these compounds can produce symptoms far more rapidly than do chlorinated compounds such as DDT (Rathus, 1973). LC_{50} values for oral exposure to rats are comparable for all compounds examined in this study, however, generally ranging from 40 to 500 mg/kg with ß-BHC at 6,000 mg/kg and endrin at 3 mg/kg (USEPA, 1984a).

The World Health Organization has set acceptable daily intake levels for several of the compounds addressed in this study, including DDT at 0.02 µg/kg/day, total aldrin and dieldrin 0.1 µg/kg/day, endrin 0.02 µg/kg/day, lindane 12.5 µg/kg/day, total heptachlor and heptachlor epoxide 0.5 µg/kg/day, chlordane 1 µg/kg/day (Rathus, 1973; Diechmann, 1973). The Food and Drug Administration has set formal tolerance levels for PCB's (2.0 µg/g) and action levels to trigger regulatory response (Table 5). These limits are compared with other regulations in Table 6. The National Shellfish Sanitation Program (NSSP) has also proposed expanding the list of regulated compounds with action levels or lowering the action levels to include malathion (0.1 µg/g), mirex (0.05 µg/g), and toxaphene (1.0 µg/g) (Ratcliff, 1971).

More recent approaches to public health protection have been from the standpoint of risk assessment and, for the consumption of shellfish, is based on the concentration of toxic or carcinogenic chemicals and the amount of contaminated tissue that may be ingested. Risk assessments for these compounds suffer from the same uncertainties regarding patterns of, consumption and exposure as was discussed previously and are also hampered by the unknown etiology of this groups carcinogenicity (NAS, 1991).

Human health aspects for toxic chemicals in the marine environment depend on several factors including, the concentration and composition of toxic chemicals in edible tissues, the toxicity and mode of action of the chemicals, and the amount consumed. These factors are considered to provide a risk assessment for potential human health hazards (Kipp, 1990). The potential risks of adverse health effects range from acute toxicity to chronic illness and cancer.

Risk characterization is the estimation of risks of adverse health effects based on dose-response and exposure data. Since any exposure to a carcinogen presents an increased risk of cancer, an acceptable level of

Table 5. FDA action levels for chemical contaminants.

Substance	Action Level (µg/g)	Type of Food
Methylmercury	1.0	Fish, shellfish, crustaceans, and other aquatics
PCBs	2.0	Fish and shellfish
Aldrin	0.3	Fish and shellfish
Chlordane	0.3	Fish
Dieldrin	0.3	Fish and shellfish
DDT, DDE, and TDE[a]	5.0	Fish
Endrin	0.3	Fish and shellfish
Heptachlor and heptachlor epoxide	0.3	Fish and shellfish
Kepone	0.3	Fish and shellfish
	0.4	Crabmeat
Mirex	0.1	Fish
Toxaphene	5.0	Fish

[a]DDT = dichlorodiphenyltrichloroethane;
DDE = dichlorodiphenyldichloroethane;
TDE (DDD) = diphenylethanedichlorophenylethane.
SOURCE: FDA (1987).

Table 6. Regulatory limits for toxic organics in seafood extracted from United Nations, United States, and Canadian regulations.

Contaminant	FAO/WHO (µg/g)[a]	FDA Action Level (µg/g)	Canada Health and Welfare Action Level (µg/g)
DDT/metabolites	1.9	5.0	5.0
Heptachlor/ heptachlor epoxide	1.9	0.3	0.1[b]
Endrin		0.3	0.1[b]
Aldrin/dieldrin		0.3	0.1[b]
Chlordane		0.3	0.1[b]
Mirex		0.1	0.1[b]
PCBs		2.0	2.0
Toxaphene		5.0	0.1[b]
All other agricultural chemicals	–		0.1[b]

[a] Based on conversion of FAO/WHO provisional tolerable weekly intake (PTWI) to U.S. population assuming average per capita seafood consumption to be 18.7 grams per 70 kilogram adult body weight per day.

[b] Characterized by way of "All Other Agricultural Chemicals" in Canadian regulations,

SOURCE: DFO (1988); FAO (1989); FDA (1987).

risk must be established, where as non-carcinogenic chemicals generally have a no-effects level that can be derived experimentally (Kipp, 1990). Although no absolute cancer risk levels have been established, the U.S. EPA uses a risk level of one in 100,000 (10^{-5}) as unacceptable level (Kipp, 1990).

III.H.3.d. Polycyclic Aromatic Hydrocarbons (PAH) - Environmental Occurrence

Polycyclic aromatics include a multitude of compounds containing fused (alkyl naphthalenes, benz(a)pyrene, and phenanthrene) or linked benzene rings (biphenyls) and are an intrinsic component of petroleum hydrocarbons. Crude oil PAH content is near 40 percent (Kennish, 1991). There are biogenic aromatics, synthesized by marine phytoplankton, but these are typically simple, minimally substituted compounds. Most biogenic material appears as unsaturated alkenes and represents 99.98 percent of the total hydrocarbons reaching the sea (Clark, 1989).

Of the approximately five million tons per year of non-biogenic hydrocarbons reaching the sea, roughly 30 percent is estimated to come from shipping and oil production activities. Approximately 12 percent is attributed to natural oil seeps, and another 12 percent to atmospheric deposition. The remaining 46 percent is a combination of sources such as industrial discharges, municipal wastes, and urban and river runoff (Clark, 1989).

The primary sources of PAH to the marine environment can be categorized by whether they are petroleum-derived (petrogenic) and combustion-derived (pyrogenic). Either of these sources may be from human activities, such as oil spills and combustion of fossil fuels, or from natural occurrences, such as oil seeps and forest fires (Farrington, 1980; NAS, 1985). Other sources may have localized importance, such as leaching from creosote-treated wood, waste from industrial facilities or commercial marine operations (Pierce et al., 1985).

Petroleum spills, with their complement of PAH's, begin the process of weathering immediately, and undergo physical, chemical and biological processes that disperse, and degrade the oil. Weathering includes evaporation of lighter, more toxic fractions, dissolution, adsorption to sediments and suspended particles, photooxidation, and microbial degradation. The type of oil or refined product spilled, hydrological and atmospheric conditions as well as the environment into which the oil was spilled will determine the fate of the oil and the resulting ecological damage. Combustion sources present a generally different mix of aromatic compounds as many substitution groups are removed in the combustion process

Because of PAH persistence and toxicity, a number of researchers have dealt directly with this class of compounds rather than all of the petroleum components. Among various estuaries, the amount and type of PAH contamination varies with location and watershed land use. Atmospheric fallout of PAH-containing particulates or particles has been reported to be a major contributor of PAH in some estuaries (NAS, 1985; Youngblood and Blumer, 1974). Urban stormwater runoff and sewage effluent also have been shown to be major sources of PAH to estuaries (Latimer et al., 1986; Hoffman et al., 1987) and used automobile crankcase oil has been implicated as a major contributor (Pruell and Quinn, 1988; Tanacredl and Cardenas, 1991). PAH, of course, also enter the marine environment directly as spills from transport and transfer of oil and petroleum products. Summaries of PAH input to aquatic environments attribute 73 percent to petroleum spills, 21 percent to atmospheric deposition, 3 percent to wastewaters and surface runoff, with 1 percent from biogenic sources (Eisler, 1987). Within the estuary only 33 percent of PAH are dissolved, with most subject to particulate absorption.

Beyond the distinction between biogenic and petroleum/combustion sources, the chemical composition of PAH provides information to distinguish between petroleum and combustion sources. Petroleum-derived PAH contain more of the 2 and 3 ring compounds with alkyl substitution on the rings (Farrington, 1980). The mono-, di- and tri-alkyl homologues of phenanthrene and dibenzothiophene are indicative of petroleum, and the relative abundance of the substituted homologues to parent compound increases with weathering (Boehm et al., 1981; Overton et al., 1981).

Combustion sources are characterized by the presence of unsubstituted three to five ring compounds, including predominantly fluoranthene and pyrene, often found with phenanthrene, benzanthracene, chrysene and benzopyrenes (NAS, 1985). More recently, combustion derived PAH have been identifies as fluoranthene, pyrene, benzo(a)anthracene, chrysene, benzofluoranthenes, benzo(a)pyrene, benzo(e)pyrene, indeno(c,d)pyrene, and benzo(ghi)perylene (Barrick and Prahl, 1987).

III.H.3.e. PAH - Ecological Significance

For hydrocarbons, toxicity increases from alkanes, cycloalkanes, alkenes, to aromatics (Clark, 1989). Within a particular class, the lower molecular weight hydrocarbons tend to be more toxic than the larger ones (Kennish, 1991). The lower molecular weight PAH (2 to 3 rings) are generally acutely toxic but noncarcinogenic, while the 4 to 7 ring higher molecular weight compounds are less toxic but carcinogenic, mutagenic, or teratogenic (Eisler, 1987). For a given ring structure, higher molecular weight PAH have been shown to be more toxic to marine organisms; LC_{50} values for pink shrimp exposed to PAH in aqueous solution was 0.5 mg/l for methylnaphthalene and 2.0 mg/l for naphthalene (NAS, 1985). The low molecular weight toxic compounds include anthracenes, fluorenes, naphthalenes, and phenanthrenes, while some of the carcinogenic compounds are benzo(c)phenanthrene, 3-methylcholanthrene, benzo(a)pyrene, and dibenzo(a,i)pyrene (Kennish, 1991).

PAH compounds are rapidly metabolized by organisms, but that process can activate or increase the carcinogenicity (Mix, 1984). Enzymes to activate these compounds are present in fish, but to a lesser degree in mollusks and other invertebrates. As a whole, PAH are not biomagnified within the food chain, as the result of rapid degradation and depuration, and low absorption in higher organisms (Jakim and Lake, 1978).

Bivalve mollusks, however, due in part to the lack of enzymes that prevent activation to carcinogenic compounds, cannot as effectively metabolize and excrete PAH, and so accumulate higher levels (Lawrence and Weber, 1984; Mix, 1982). Bivalve depuration rates for higher molecular weight compounds are longer, and bioconcentration factors higher, than for the smaller compounds (Mix, 1982; Neff et al., 1976). Seasonal increases in PAH tissue concentrations coincide with periods of lipid storage for spawning (Marcus and Stokes, 1985).

A comparison of PAH accumulation was observed for three species of bivalve mollusks in contaminated sediment from Chesapeake Bay. These studies showed similar uptake of specific PAH by the oyster, *Crassostrea virginica,* the hard clam, *Mercenaria mercenaria,* and the marsh clam, *Rangia cuneata,* with total PAH concentrations ranging from 1.1 to 1.8 µg/g (dry weight) (Bender et al., 1988). The average BCF for 14 select PAH from contaminated sediment, over a 28 day period, was 15,000 for oysters and 4,000 for clams, as clams depurated at a much faster rate (Bender et al., 1988). Other work has resulted in contradictory findings, with little PAH depuration occurring in M. *mercenaria* and 90 percent in C. *virginica* over a 14 day period (Jakim and Lake, 1978). Boehm and Quinn (1977) found only 30 percent of hydrocarbons depurated from *Mercenaria* within 120 days.

More recent studies of the accumulation of PAH from waste crank case oil in water have shown that the clam, *Mercenaria,* did not depurate significant amounts of the PAH over a 45-day period (Tanacredl and Cardenas, 1991). This mixture of results indicates that PAH bioaccumulation and depuration varies with the chemical and physical characteristics of the PAH and with conditions of exposure and that long-term persistence and bioaccumulation could be of concern under certain conditions.

Bioavailability is a major determinant in the accumulation of PAH in shellfish. Analyses of PAH in mussels and adjacent sediment indicate a preference for petroleum-derived PAH in the shellfish, with pyrogenic PAH dominant in sediments (Farrington et al., 1983; Farrington et al., 1985). These and other similar results have been interpreted by Farrington (1986) to suggest that pyrogenic PAH are tightly bound to particles and not readily available for biological accumulation, while petrogenic PAH occur in dissolved and colloidal suspensions, more readily available for biological uptake and microbial decomposition.

Concentrations of PAH in bivalve mollusks were monitored throughout the U.S. coast during the 1986 to 1988 Mussel Watch Program. The low molecular weight (two- and three-ring compounds) ranged from below detectable levels (0.02 µg/g dry weight) to 4.30 µg/g in Elliot Bay, Washington. Oyster samples from estuaries along the southwest Florida coast from Rookery Bay to Tampa Bay ranged from <0.02 to 0.39 µg/g. High

Molecular weight PAH from the same Florida estuaries ranged from <0.02 to 0.20 µg/g (NOAA, 1989).

Analysis of the distribution of PAH in sediment and oysters from the Gulf of Mexico sub-set of Mussel Watch samples revealed similar concentrations of PAH in sediment and oysters (0.507 µg/g dry weight in sediments and 0.536 µg/g dry weight in oysters). Oysters accumulated more of the low molecular weight PAH, while sediments contained more of the high molecular weight PAH (Wade et al., 1988).

Petroleum products (rather than specific PAH compounds) have been the subject of many exposure and toxicity assessments. Effects range from mortality (George, 1970) to gamete degeneration and resorption and reduction in storage levels in *Mytilus* (Lowe and Pipe, 1987) and in general larval and embryonic forms are more susceptible. Smaller oil particles, presumably from increased surface area and bioavailability are more toxic (Stromgren et al., 1986). Petroleum compounds are also incorporated by *C. virginica,* are toxic to larvae, and interfere with fertilization (Renzoni, 1973). Refined petroleum products and waste motor oil are toxic to bivalve embryos and larvae at the low ppm (mg/l) exposure levels (Roegner and Mann, 1991). Chronic exposure to subacute levels may cause delayed larval development, which in turn may inhibit development into viable adults (Calabrese, 1972) and sediments contaminated with oil will reduce the depth to which clams burrow (Olla et al., 1983).

Oyster and mussel larvae are routinely used to test the toxicity of natural sediments, but again little information exists to relate exposure concentrations to body burdens. Percent survival tests for bivalve larvae can be compared with levels of specific toxics in the sediment; however, correlation with sedimentological characteristics have been found to provide better correlation with biological effects that total contaminant concentration (NOAA, 1990a). Since metals and pesticides are more toxic than PAH, interpretation of sediment toxicity data is not specific to PAH impact (NOAA, 1990a). Estimating biological impact from tissue contaminant levels is tenuous, due to the many variables associated with accumulation and depuration. A general trend may be inferred from tissue levels and degree of environmental contamination, but seasonal changes may obscure long term trends.

In general, high concentrations of PAH in shellfish are indicative of a continuous source of bioavailable PAH to the environment. This, in turn indicates the probability for bioaccumulation in fish and other marine organisms that feed on the shellfish (including humans). Although acute toxicities (LC_{50}) can be used to indicate relative toxicities of individual PAH, these have little application for sublethal effects.

III.H.3.f. PAH - Human Health Impacts

Polycyclic aromatic hydrocarbons as a class include some relatively well characterized carcinogenic compounds such as benzo(a)pyrene. This well studied compound, however, typically forms only 1 percent of the total PAH present (NAS, 1991). USEPA's ambient water quality criteria of

28 ng/l of total carcinogenic PAH are set from a 10^{-5} lifetime risk level (Allan et al., 1991). Reviews by Neff (1979) and Stegeman (1981) appear to indicate that the amount of PAH ingested through shellfish consumption are probably minimal in relationship to that associated with smoked or charcoal-broiled foods (Moore et al., 1989). The potential presence of unidentified mutagens (PAH metabolites) in shellfish, however, may be of concern (Parry et al., 1981).

Results of toxic substance analyses in clams from Narragansett Bay, Rhode Island, indicated cancer risks of 0.8/100,000 to 30/100,000, depending on the contaminant level and amount of shellfish consumed (Kipp, 1990). These calculations considered all contaminants including pesticides, PCB's, PAH and metals and PAH concentrations ranged from 0.002 to 0.024 µg/g. FDA has set no action or tolerance levels in seafood for these compounds.

IV. STUDY DESIGN

IV.A. Rationale

To evaluate the contaminant status of an area, water column concentrations are notoriously variable and pose many difficulties in their use. Sessile biomonitors or sediments, however, provide an integrating effect and tissue or sediment contaminant concentrations are generally accepted to provide a representation of long-term or average contaminant loading to an area. The comparative use of tissue concentrations of contaminants can then be used to indicate problem areas or scope of influence of particular pollution sources.

Biomonitors, if they tolerate a wide range of pollutants, should also closely reflect contaminant bioavailabilities. Bivalves have been extensively used for this purpose (Phillips, 1990) and bioaccumulation of metals by bivalves, in particular oysters, has been documented to far exceed that evidenced by teleost fish (Phillips and Russo, 1978). Bioaccumulation of certain compounds by bivalves can, depending on the organism's strategy of detoxification, in some instances magnify pollution gradients, making them more readily detectable. The bulk of this work, however, has been carried out in temperate rather than sub-tropical or tropical waters (Phillips, 1990).

Bivalves are particularly well suited for this role due to their feeding strategy, where they filter large volumes of water and associated particulates. The particulates include not only the preferred phytoplankton food items, but also bacteria and viruses, toxic dinoflagellates ('red tide'), inorganic, and detrital particles. Human health risks in connection with shellfish most often arise from the consumption of raw or inadequately cooked shellfish and the subsequent exposure to pathogens concentrated from the water column.

The particulate fraction concentrated by filter feeders also typically contains the bulk of the anthropogenic contaminants, specifically toxic metals and hydrophobic synthetic organic compounds. Following the physical concentration of particulates, metabolically active compounds can also be further incorporated into the organism and levels of some elements or compounds can reach levels hazardous for consumption.

These factors combined to dictate the thorough assessment of shellfish abundance, sanitation, and contaminant status within the Sarasota Bay study area. This investigation was particularly relevant since, with no commercial fishery present, shellfish would be exclusively collected by recreational harvesters and would not typically be subjected to the regulations designed to ensure safety and sanitary quality. Accordingly, emphasis was placed on recreational efforts and implications.

IV.A.1. Species Selection

Shellfish species selected for the assessment of contaminants within Sarasota Bay were chosen primarily on the basis of abundance and broad geographic distribution to permit bay-wide comparisons and evaluations. In addition, these species (or closely related ones) have been the subject of numerous other contaminant investigations and these existing data are available for comparative purposes. Due to the recreational and public health emphasis brought by the program, the selected species were also edible, preferably with a commercial fishery still in existence elsewhere, against which to evaluate the benefits of restoration measures.

Previous work (Estevez and Bruzek, 1986) identified the five most abundant edible shellfish species as *Mercenaria campechiensis* (hard clam), *Chione cancellata* (cross-barred Venus), *Macrocallista nimbosa* (sunray Venus), *Dinocardium robustum* (cockle), and *Crassostrea virginica* (American oyster). During that study, *Chione* and *Dinocardium* were typically found in deeper waters and thus less accessible to the majority of recreational shellfish harvesters. *Macrocallista,* on the other hand, is primarily associated with areas of comparatively good water quality, high current velocities, coarse grain sized sediments, and seagrass beds. Its distribution would therefore limit its use in evaluation of bay-wide conditions and its collection could result in significant disruption to highly-valued *Thalassia* beds.

Accordingly, *Mercenaria* spp. and *Crassostrea virginica* were selected for investigation during the present effort. Harvest of these organisms within the Bay is limited to recreational efforts at this time and closure of the fisheries is attributed to a combination of a decline in effort and in closure of approved waters by bacterial contamination. Of the two organisms selected, *Mercenaria* is the more common due to the broader range of environmental requirements and occurs in a variety of habitats.

The other selected species, *C. virginica,* is not present in the study area in extensive reefs and only rarely occur subtidally. They do not offer the commercial harvesting possibilities found in other areas of the state such as Appalachicola Bay or Cedar Key. None the less, oysters are found on the shoreline throughout the NEP area and do form small reefs at creek mouths. Their continuing consumer popularity and the massive data base on tissue contaminants justifies their inclusion in this study.

IV.A.2. Potential Health Impacts

The study was not designed to provide a formal risk assessment associated with the consumption of shellfish or to set recommended tissue concentration limits or seafood consumption guidelines. Potential health impacts were to be presented relative to Federal agency guidelines, levels set by various shellfish sanitation organizations, and in relation to the tissue concentrations observed in other geographic regions.

IV.B. Population Survey

Despite a previous extensive species distribution survey (Estevez and Bruzek, 1986), little is known of the seasonal abundance of shellfish within Sarasota Bay. For *Mercenaria,* in particular, the presence of shells from dead organisms and the absence of live individuals at nearly 30 percent of the stations in the earlier study indicate that the populations are not necessarily stable. While changes in water quality can produce mass mortalities, unusually severe conditions are typically required.

Recruitment is reported to be erratic and may be controlled by predation rather than by environmental factors (Mulholland, 1984). Previous experience supports this as there are some areas of the Bay where juvenile *Mercenaria* are not present and only adults larger than 80 mm in length are found. As *Mercenaria* reach a harvestable size (30 to 40 mm) within approximately 18 months, the lack of juveniles would indicate at least one year of poor clam sets.

These factors, coupled with the potential year round availability of clams for harvest, make desireable two population surveys for *Mercenaria* during the study year. These data were not to be used to develop a formal stock assessment or carrying capacity for Sarasota Bay with regards to shellfish harvesting, but for insight into the current relative distribution of *Mercenaria,* any seasonal fluctuations in densities, preferred habitat and substrate, and relative accessibility to recreational shellfishing.

Oyster surveys were to be structured differently due to the differing ecological requirements of these organisms. Surveys primarily consisted of identifying areas of viable and senescent reef, based on both previous mapping and observations during this study. Reef condition and any physical destruction were to be noted, as was dominant oyster length, and the presence of oyster predators.

IV.C. Contaminant Selection

To provide useful information from the standpoint of public health, the entire tissue that is commonly eaten should be analyzed, rather than a specific fraction or organ. Contaminants selected for analysis should be indicative of the possible sources of pollution (i.e., non-point source, boating, domestic effluent).

IV.C.1. Bacteriological Contaminant Selection

In the marine environment, one of the most important routes of infectious diseases in humans is through water contact and the consumption of raw shellfish (SCAG, 1988). The pathogens of most concern are associated with human fecal wastes, such as bacteria and viruses causing gastroenteritis or Hepatitis A. Due to the difficulty and expense of identifying the actual disease-producing organisms, however, the presence of fecal coliforms has been used as an indicators of sewage. The presence of fecal coliform does not always accurately predict illness, however, and they appear to be more susceptible to chlorination or other sanitation treatments than do some viruses.

Sarasota Bay is fortunate in that massive industrial discharges do not exist, major sewage treatment plant effluents are in the process of being rerouted to either upland disposal or deep well injection, a large portion of its residents are on municipal sewer systems, and storm and sanitary sewers are separate systems, thus preventing the combined sewer overflow problems observed in older cities. Within the NEP study area, however, there are still a number of domestic waste effluents reaching the Bay either directly or indirectly. Secondary treated effluent is discharged by numerous small package plants within Sarasota County, primarily to Phillippi Creek, and secondary effluent and sludge is used for agricultural practices in Manatee County on lands directly adjacent to the Bay. The City of Sarasota discharges effluent treated to advanced waste treatment (AWT) standards through Whitaker Bayou on an intermittent basis. There is also a large recreational boating population who is in unknown compliance with Federal regulations regarding overboard discharge of untreated sewage. Stormwater runoff has also been identified as one of the major water quality problems within the Bay (FDER, 1988) and is reported to contain bacterial levels comparable to secondarily treated effluent (Galvin, 1987).

In addition to sewage related bacterial problems, health threats due to *Vibrio* spp. illnesses have received recent attention. These marine bacteria, together with *Aeromonas* spp., are indigenous to marine waters, unrelated to the presence of sewage, and have been identified in both approved and prohibited waters (Blake and Rodrick, 1983) with no correlation to fecal coliform levels. As most deaths attributed to vibrio infections have been associated with shipped shellstock rather than packed meats, "time and temperature abuse" in handling apparently exacerbates the problem (ISSC, 1988).

The presence and numbers of vibrios in oysters do exhibit a seasonal peak during the warmer months of the year (Ruple et al., 1989). This fact is even more noteworthy with respect to the shellfish species collected year-round (i.e., *Mercenaria).* It is deemed extremely important to assess this problem from the standpoint of the recreational shellfish harvester.

A number of microbial analyses were selected to provide information on sources of contaminants and severity of contamination. Sample matrices included both shellfish tissues and water column samples: Aerobic plate counts quantified the entire heterotrophic population. Total and fecal coliform, and fecal streptococci were to be used to evaluate the component of human versus other mammalian feces, as well as distance from the source of contamination. A total of seven pathogenic vibrio species were selected for quantification, together with *Aeromonas hydrophila* and *A. sobria,* which are potential human pathogens. *E. coli* are also potential enterotoxic pathogens, and were also selected for enumeration.

IV.C.2. Metal Contaminant Selection

Sarasota Bay is fortunate in that there are, in comparison to other estuaries within the National Estuary Program, comparatively few industrial point source discharges. One of the major problem sets identified in the nomination document, however, was stormwater runoff. This contaminant source is one of the major mechanisms by which toxic pollutants from non-point sources are transported to the estuarine environment. In addition to the significant amounts of sediment transported in this fashion, pollutants characteristic of stormwater include metals such as arsenic, cadmium, chromium, copper, lead, mercury, toxic organic compounds, and petroleum products. Marinas and boating operations can contribute metals to the environment as well, particularly with respect to anti-fouling paints.

Accumulation of metals in tissues, particularly for benthic organisms exposed more heavily to sediment burdens, has been demonstrated for many metals (Richter, 1988). Sludge from domestic waste operations is also noted for the concentrations of cadmium, lead, arsenic, and mercury and its use as a soil amendment in near-Bay agriculture operations may pose some concern.

Previous sediment work from the northern portion of the study area, in which copper, lead and zinc were examined, indicate that several areas are enriched in these metals beyond state-wide averages for pristine conditions, and lend support for a bay-wide evaluation of metals in both sediments and tissues.

Metals selected for this effort are as follows: arsenic, cadmium, copper, lead, mercury, and zinc. These choices are based on the requirements that the parameters should be indicative of the possible sources of pollution (i.e., non-point source, boating, domestic effluent), should be those with a demonstrated analytical technique (Iyengar, 1989), and should be meaningful with respect to the sediment monitoring to be conducted in the Bay. In considering possible future uses of this information, which may include health risk assessment or epidemiological studies, a substantial existing data base on human health effects, potential uptake through the gastrointestinal tract, and the suitability of non-invasive samples (hair, blood, urine, etc.) to evaluate bioaccumulation is also desireable. Finally, but certainly not the least important, is the significance of the particular metal to the success of the organism (Eisler, 1988), and the availability of existing literature on toxicity levels, bioaccumulation potential, and levels of reproductive or other impairment.

IV.C.3. Pesticides Contaminant Selection

The major sources of pesticide contamination to Sarasota Bay include: agricultural chemicals, industrial and domestic lawn pest control and mosquito control applications. Although these applications generally do not introduce the pesticides directly into the Bay, rapid influx occurs through stormwater runoff.

Pesticides selected for contamination assessment included representatives from three classes of chemicals (carbamate, organophosphate, and chlorinated pesticides) that are used in Sarasota and Manatee Counties. These indicator residues for shellfish contamination include: the organophosphates chlorpyrifos (dursban), used for domestic and industrial insect control and dibrom (naled), used as a mosquito adulticide; and the carbamate bendiocarb (ficam) used on turf and ornamental (Agricultural Chemicals Handbook, 1989). In addition to these pesticides currently in use around the Bay, residues of persistent chlorinated hydrocarbon pesticides (e.g., DDT and derivatives, chlordane, BHC) will be monitored.

IV.C.4. PAH Contaminant Selection

The Southwest Florida coast has been shown to be relatively free of substantial oil pollution except for localized "hot spots" receiving petroleum contamination from certain land-use and marine activities including urban stormwater runoff, industrial drainage, marina activities, and transport of refined petroleum products (Pierce, 1983; Pierce et al., 1985; Van Vleet et al., 1985). Studies by Pierce et al. (1986) in Charlotte Harbor, did find, however, that oysters up to one mile from sources of contamination contained discernable concentrations of petroleum characteristic of the source material. Areas within one mile of shore within the Sarasota Bay study area will certainly encompass the vast majority of recreational shellfishing opportunities.

Although no major oil spills have been observed in Sarasota Bay, the chronic influx of petroleum from tributaries bringing stormwater runoff from a diverse urban watershed and spillage from a number of marinas raises concern for bioaccumulation of toxic petrochemicals in marine organisms. Problems arise not only from the human health aspect of consuming oysters contaminated with petroleum, but also from the adverse effects of petroleum on the survival and reproduction of shellfish (National Academy of Sciences, 1985). Of greatest concern are the polycyclic aromatic hydrocarbons (PAH) which include both toxic and carcinogenic substances (NAS, 1985).

Two major sources of PAH are: petroleum (petrogenic); and combustion (pyrogenic) products (Lake et al., 1979; NAS, 1985). Certain diagnostic characteristics are used to distinguish the predominant source. The total amount of PAH determined from the suite of 16 PAH specified in by EPA, Method 610, will establish the level of contamination. The relative amounts of substituted (the C_0 to C_3 alkyl homologs of naphthalene, phenanthrene, dibenzothiophene and pyrene) vs. parent PAH will determine the relative contribution from petroleum and pyrogenic sources. These data will be used to identify the sources of petroleum contamination and to provide recommendations for alleviating the impact to shellfish.

V. METHODS AND MATERIALS

V.A. Survey Techniques - *Mercenaria*

Stations for the survey of *Mercenaria* spp. populations emphasized those areas accessible (i.e., wading depth) to recreational fishermen and avoided unnecessary destruction of established seagrass beds. A total of 169 station locations were visited during each survey (Figure 6). Stations were distributed along most of the shallow areas of Sarasota Bay and so to some extent, a subdivision of stations by segment (Estevez and Palmer, 1991) proportionally reflect the absolute area of shallows within each segment. Station locations were documented with Loran C readings and compass headings to landmarks which were transferred to base maps using a map of Loran lines previously developed by Mote Marine Laboratory (MML) for the inshore areas.

The technique used in the clam surveys consisted of timed searches for clams through visual and tactile means. Experimentation showed this technique to be effective in locating clams, and also the technique is similar to the treading technique employed by the average recreational shellfisherman. In addition to locating clams by sighting the siphon tube (glass bottom buckets were employed where advantageous) or by feeling the shell, three plots of 1 m^2 were raked at each station to assure the presence of clams at a station was not overlooked. For each survey, three five-minute replicate searches were made per station.

During surveys, live *Mercenaria* were enumerated, measured, and released at all stations and the presence of dead shells be recorded. All other species collected of possible importance were also noted, particularly the edible species, *Macrocallista nimbosa*. Those species known for their predation on *Mercenaria* were also enumerated. The dominant sediment type was noted, as was the presence of any seagrasses.

The initial *Mercenaria* survey was conducted during the early spring, in January and early February 1990. From these results, a size class common to the majority of the Bay was established for preferential collection of organisms for the analysis of tissue contaminants. *In situ* water quality parameters (conductivity, temperature, dissolved oxygen) were recorded during surveys as well. The *Mercenaria* survey was to be repeated a second time during the nominal wet season (June to September 1990) but extremely low rainfall levels forced the postponement of the effort, in hopes of later rains. Rainfall deficits for the Manasota Basin (SWFWMD, 1990) in calendar year 1990 were over 15 inches below normal, while rainfall records on City Island were nearly 22 inches below the amounts received in the preceding eight years. These did not materialize and the survey was conducted during mid to late September 1990. The two surveys were therefore termed spring and fall to avoid the implication of substantial rainfall during the latter collection. Techniques and stations during the second survey were identical.

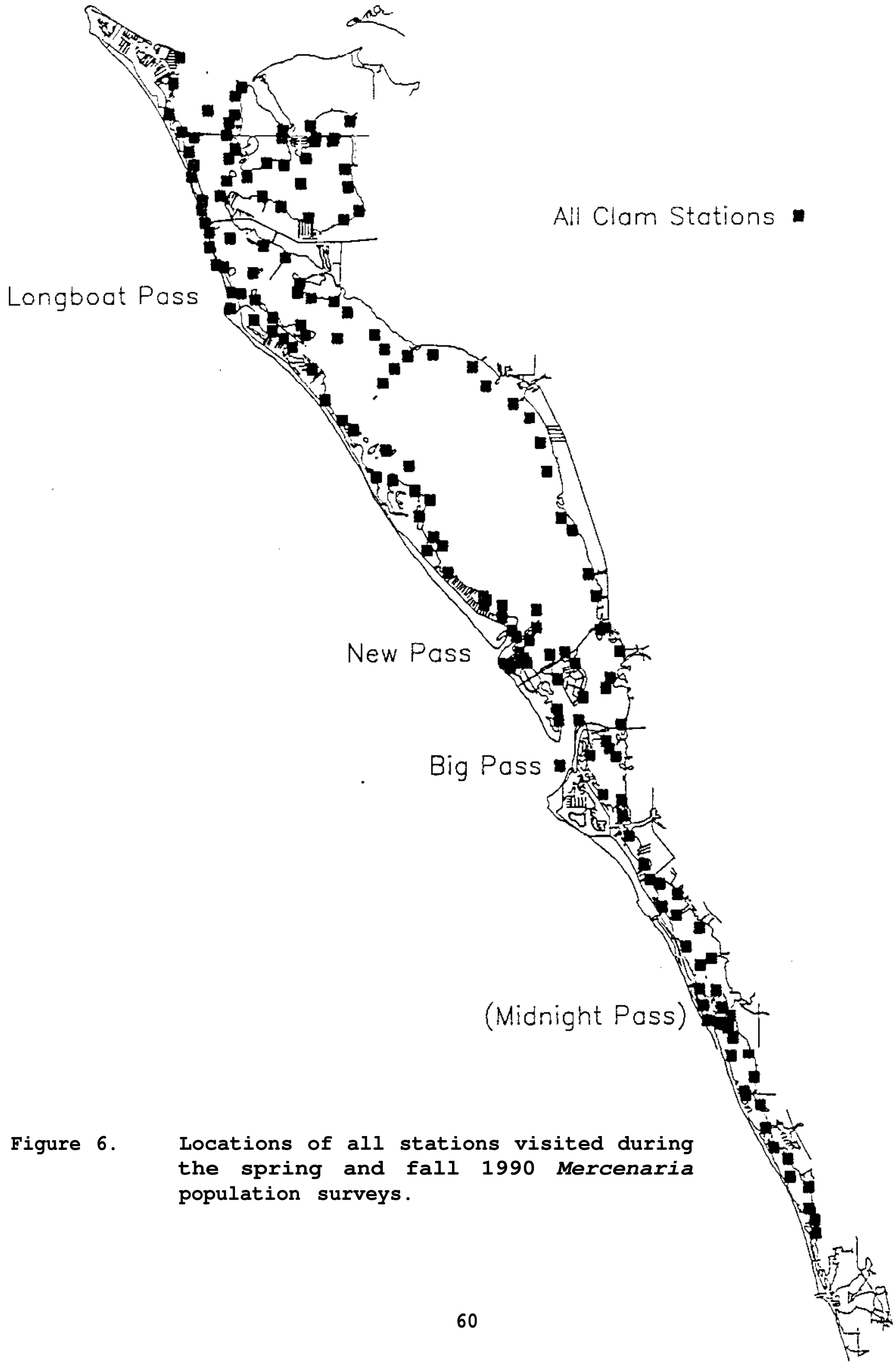

Figure 6. Locations of all stations visited during the spring and fall 1990 *Mercenaria* population surveys.

V.B. Survey Techniques - *Crassostrea*

Surveys of oysters, due to existing mapping (Mangrove Systems, 1988), was structured differently from the *Mercenaria* work. In addition, due to the differing environmental requirements of *Crassostrea* and *Mercenaria,* station locations were not expected to overlap. Otherwise, the general approach of surveys followed by tissue collections during spring and fall, numbers of stations and samples analyzed, and rationales were entirely analogous.

From the location of substantial oyster bars shown in existing mapping (1984 color aerial photographs groundtruthed in 1987), 15-20 stations were selected prior to oyster field work. Station selection criteria was much as that described for the *Mercenaria* work and emphasized shallow areas with high recreational access. The selected stations were visited, reefs described according to the abundance of oysters and characteristics of the bars (senescent, mature, accreting, etc.) as defined by Hines and Belknap (1986). Bars noted during the *Mercenaria* surveys were also described. The presence or absence of legal harvest size oysters was also noted, as were obvious areas of destruction, such as channels, which have been created through areas of reef.

V.C. Tissue Collections Methods

Stations for tissue collections were identified with consideration for broad geographic distribution, for point and non-point sources, recreational access, and, for oysters, endeavored to include the major tributaries to the Sarasota Bay system (Figure 7 and 8). Twenty stations were selected, ten for *Mercenaria* and ten for *Crassostrea.* Stations are described in terms of access, abundance, potential contaminant sources, and other characteristics within Tables 7 and 8 and latitudes and longitudes are listed in Table 9. Following each of the two population surveys, two composite tissue samples were collected from each of twenty locations. Collections took place in April and again in November-December 1990.

The clams and oysters for tissue analyses were collected or pried from the reef structure, rinsed briefly in ambient water, bagged and iced for transport to the laboratory. Apparent density of clams on the tissue collection station visit was, in some instances, substantially less than previously observed. To meet bacteriological holding times, samplers would visit some stations on the day previous, collect sufficient organisms, bag them in mesh bags, and leave the bag at the station for rapid retrieval on the sampling day. At each station, water column samples were also collected for bacteriological analyses. Water samples were deep surface samples, collected as near to the shellfish sampling site as is practical. *In situ* data were also collected during each tissue collection. Sediments were collected from nearby for archival purposes. A total of 80 tissue samples and 40 water samples resulted for the entire study.

Clam Tissue Collection Stations

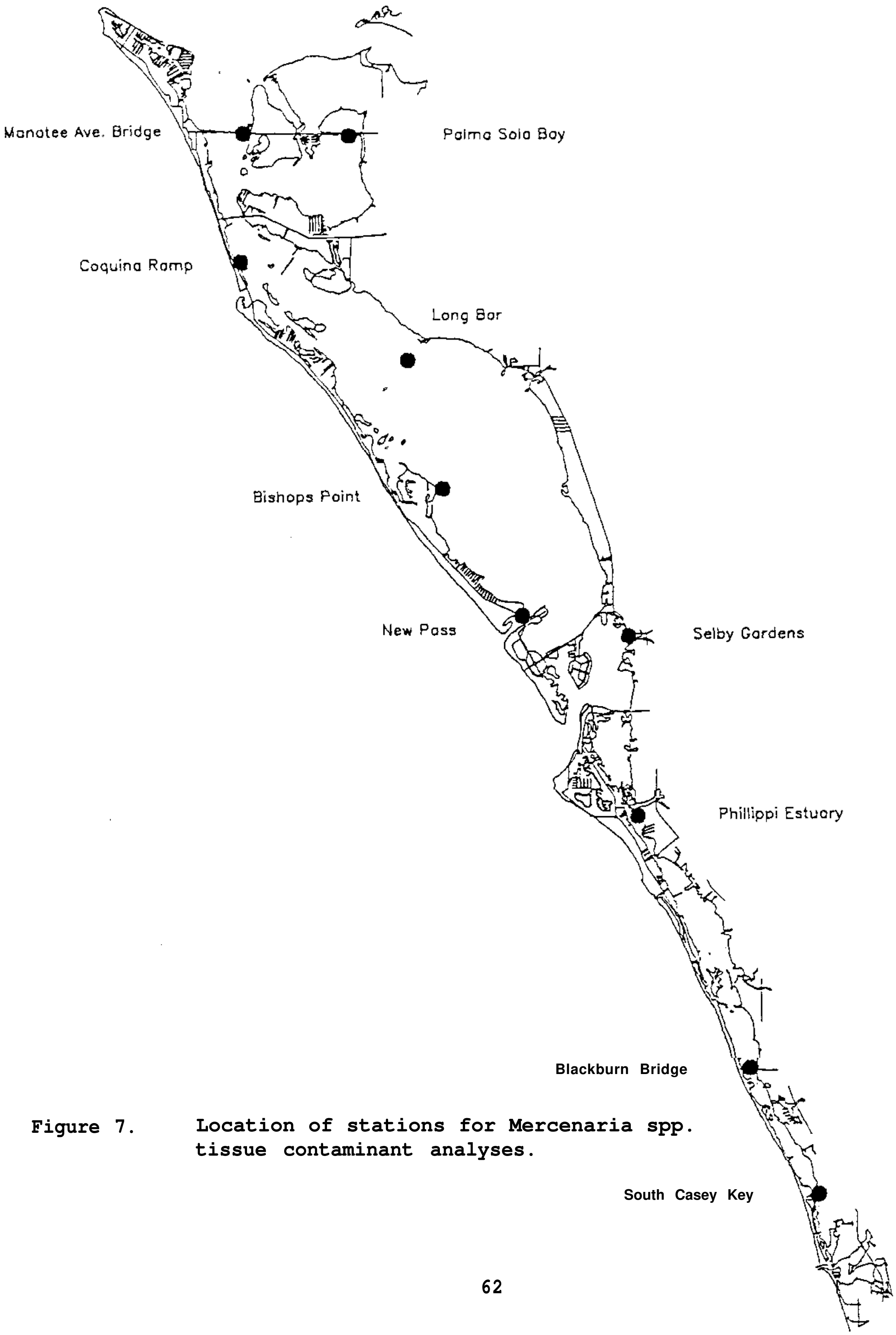

Figure 7. Location of stations for Mercenaria spp. tissue contaminant analyses.

Oyster Tissue Collection Stations

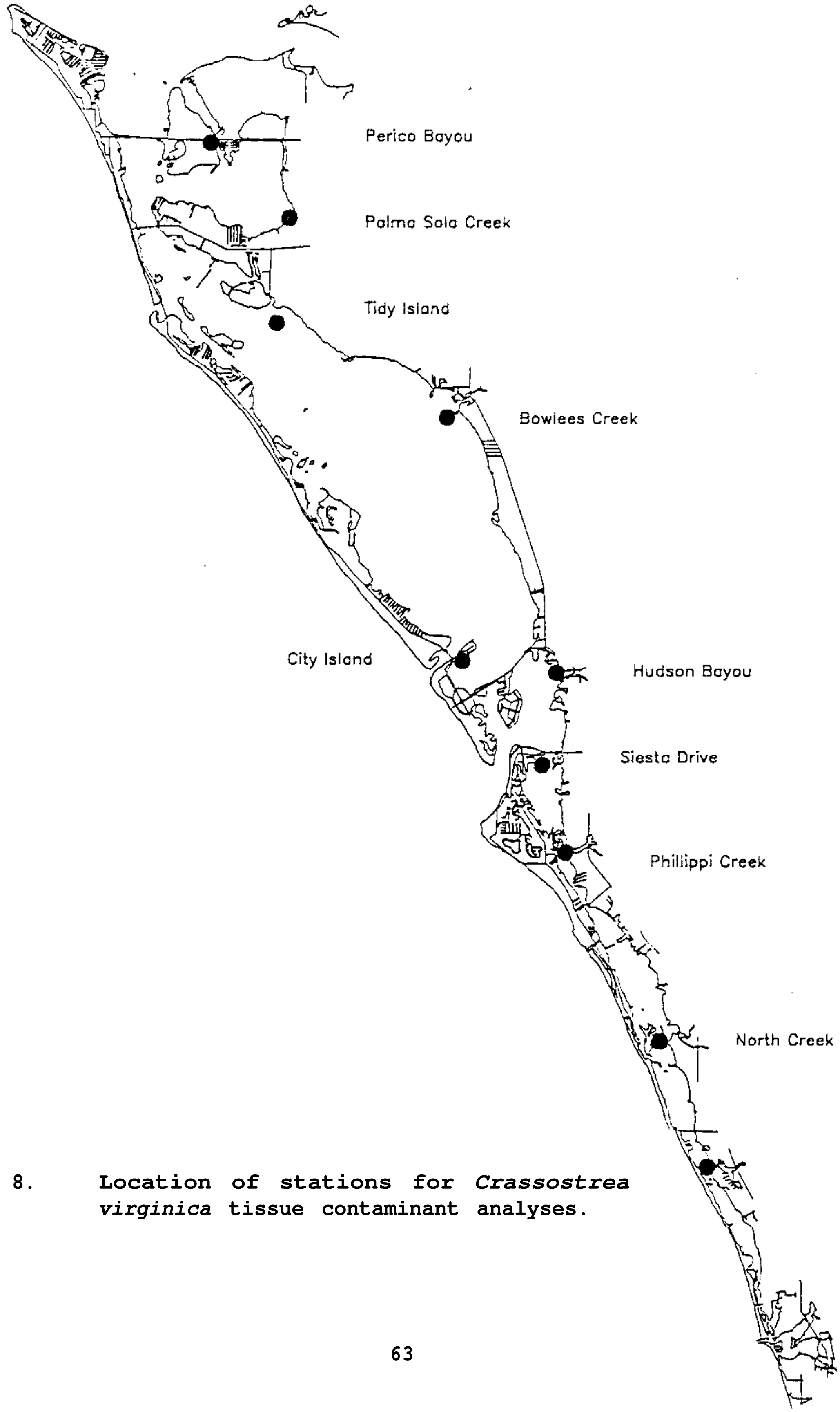

Figure 8. Location of stations for *Crassostrea virginica* tissue contaminant analyses.

Table 7. Location of the ten clam tissue stations.

Bishops Point. This nearshore station is in the area approved for shellfishing. Clams were abundant but access other than by boat would be limited to shoreline residents.

Blackburn Bridge. Clams were collected along the islands north of the bridge on the west side of the ICW. A marina is present on the east side. Access is easy by land but clams were not found to as abundant as in other areas.

Coquina Ramp. Clams were collected along the channel leaving the boat ramp opposite Bradenton Beach. No development exists from the ramp to Longboat Pass. Access is easiest, but can be accomplished by land as well by boat.

Long Bar. Abundant clams were found on Long Bar. The station is well off shore and accessible only by boat.

Manatee Ave. Bridge. The station was north of the bridge on the east side. A marina is present but the open water of Anna Maria Sound may provide much water exchange. The proximity to the mouth of the Manatee River may be considered in assessing shellfish in this area. Parking is available along the bridge.

New Pass. Clams were collected in the pass east of the drawbridge. Public access is easy and evidence of clamming was observed.

Palma Sola Bay. Clams were collected from the small bay north of Manatee Ave. The area has high public use including water skiing and, along shore, dog walking and occasional horseback riding. Ample parking is available. Few boats are moored in the immediate area.

Phillippi Estuary. The station was situated south of the creek mouth on the east side of the ICW. Clams are subjected to runoff from Phillippi Creek and limited flushing. Access will be available only by boat after the last lots are developed along the mainland shore.

Selby Gardens. The station was located on the mud flats and grassbeds off of Selby Botanical Gardens. The area is exposed to runoff from Hudson Bayou and downtown. Also, abundant boats are moored in the bay and at Marina Jacks.

South Casey Key. Spoil on the east side of the ICW was the site of the southernmost clam collection station. Although development is continuous on both shorelines, the proximity of the station to the Venice Inlet may provide flushing from the Gulf of Mexico.

Table 8. Location of the ten oyster tissue collection stations.

Bowlees Creek. This area produces some of the most extensive oyster bars in the study area. A marina and heavily developed area exist upstream. Access is difficult other than by boat.

City Island. Oysters were collected from several round bars located just south of City Island near the boat ramp. The station is not associated with a creek. Access is made easily by land.

Hudson Bayou. The area of collection was under the Orange Avenue bridge. This area is exposed to effects of stormwater runoff, boats and dense human development.

North Creek. Oysters were collected outside the creek mouth. Development is not intense upstream but without Midnight Pass open, water flushing is minimal.

Palma Sola Creek. Scattered clumps of oysters occur along the mangroves in the creek and several small bars lie just outside the creek mouth. The creek is too small for boats. The station is accessible by land.

Perico Bayou. Oysters were collected from scattered clumps north of Manatee Ave. The shallow area may experience limited flushing. Development occurs only to the south side of the road.

Phillippi Creek. Oyster were collected in the creek downstream of US 41. Bars are abundant in the lower creek. Like Bowlees Creek, marinas and intensive development are present. Access is not easily made by land.

Siesta Drive. The station was located south of the Siesta Drive bridge. A small marina is nearby but proximity to Big Pass should result in a high rate of water exchange. Bars along shore are accessible from private property.

South Creek. The station was situated upstream from the ICW and downstream from U.S. 41. Boats line each shore and access is only by water.

Tidy Island. A large circular bar was sampled south of Tidy Island. A small creek which may carry some agricultural runoff is located north of the bar. The area can only be approached easily by boat.

Table 9. Location of clam and oyster tissue collection stations.

Station	Latitude	Longitude
Clam Stations		
Bishops Point	27° 22′ 41"	82° 36′ 40"
Blackburn Bridge	27° 10′ 46"	82° 29′ 39"
Coquina Ramp	27° 27′ 26"	82° 41′ 38"
Long Bar Point	27° 25′ 10"	82° 37′ 31"
Manatee Ave. Bridge	27° 29′ 49"	82° 41′ 22"
New Pass	27° 20′ 12"	82° 34′ 53"
Palma Sola Bay	27° 29′ 48"	82° 38′ 58"
Phillippi Estuary	27° 15′ 56"	82° 32′ 13"
Selby Gardens	27° 19′ 33"	82° 32′ 30"
South Casey Key	27° 08′ 10"	82° 28′ 06"
Oyster Stations		
Bowlees Creek	27° 24′ 43"	82° 34′ 46"
City Island	27° 19′ 57"	82° 34′ 22"
Hudson Bayou	27° 19′ 39"	82° 32′ 21"
North Creek	27° 12′ 54"	82° 30′ 02"
Palma Sola Creek	27° 28′ 22"	82° 28′ 28"
Perico Bayou	27° 28′ 32"	82° 40′ 07"
Phillippi Creek	27° 16′ 16"	82° 32′ 14"
Siesta Drive	27° 18′ 05"	82° 32′ 41"
South Creek	27° 10′ 04"	82° 29′ 09"
Tidy Island	27° 26′ 24"	82° 38′ 20"

No oyster tissue stations were located in conditionally approved waters, under either the existing or proposed classification boundaries. Clams collected from Bishops Point were from conditionally approved waters under both classification systems and those clams gathered at Long Bar Point would be approved if the proposed boundaries are adopted. The remaining tissues analyzed in this study were collected from prohibited or unclassified waters, which would remain so under even under the proposed classification.

Tissue collection stations, while placed with as broad a geographic coverage as possible, were dictated by the presence (or absence) or organisms and the relatively limited number of stations. Of the 16 bay segments (Estevez and Palmer, 1990), there were no tissue stations within three of the four pass segments (Longboat, Big Sarasota, and Midnight Passes) and no stations within Segment 1, on the western side of Anna Maria Sound. Segments 13 and 16, Roberts Bay and Blackburn Bay, had the most number of tissue stations, with three apiece. Station location was also heavily influenced by the presence of tributaries. Because of the comparatively low density of tissue stations by segments, data on tissue contaminants were presented on a tributary rather than segment basis.

For bay-wide comparisons, a single size class was desireable to permit the assumption of equivalent exposure times. Although legally harvestable organisms were preferred, the lack of legal sized oysters in all areas of the Bay, and the possible non-compliance of the recreational harvester, dictated the use of a smaller organism likely for *Crassostrea.* The low numbers of clams in many locations further restricted the size grading possible.

Original study design called for the collection and analysis of tissues during the dry season, followed by two collections and tissue analyses at some priority subset of stations during a significant storm event of the wet season. Tissue levels of pollutants during the dry season, while not worst case, were to allow for bay-wide comparisons of possible problem areas. Data obtained from tissues collected in the wet season were to represent a 'worst case' scenario. As described above, low rainfall amounts forced the redesign of the study, to focus on potential seasonal differences between each of the 20 tissue stations. The two samplings were referred to as a spring and a fall collection to avoid the implication that substantial rainfall occurred.

V.D. Bacteriological Methods

Shellfish for bacterial analyses were collected during the same efforts described above, but were iced and transported by courier to another laboratory for processing within 6-12 hours of collection. As for the other parameters, each replicate sample from a station and date consisted of a composite of a number of individuals. In addition to the tissue samples, deep-surface water samples and some sediment samples were also processed for information on bacterial sources.

The shellfish, scrubbed under water of potable quality and drained on sterile towels, were opened using a sterile thin-bladed knife forced between the shells. Shell liquor was collected and formed part of the subsequent sample. Tissues and liquor were blended with equal weights of either sterile phosphate buffer or 0.1 percent peptone water, and the mixture pipetted for transfer to culture tubes.

Both presumptive total and confirmed fecal coliforms and aerobic plate count were determined by established techniques specified for shellfish meats and seawater in "Recommended Procedures for the Examination of Seawater and Shellfish" (APHA, 1985b), using the Standard Plate Count (SPC) and MPN on seawater and shellfish samples. When high numbers of fecal coliforms were encountered, the method of Richards (1978) was employed for the numeration of total and fecal coliforms. Fecal streptococci and were determined using established procedures described in F.D.A.'s Bacteriological Analysis Manual (1978).

In addition, the seawater and shellfish meats and some sediments were tested by established techniques for the presence of *Escherichia coli, V. alginolyticus V. cholerae, V. damsela, V. fluvialis, V. hollisae, V. metschnikovii, V. mimicus, V. parahaemolyticus, V. vulnificus, A. hydrophila,* and *A. sobria.* All vibrio and aeromonas species were tested by gram stain, negative oxidase test, and for 0/129 sensitivity. *Aeromonas hydrophila* and *A. sobria* were determined using Rimler-Shotts media according to established techniques. Isolated typical colonies were bio-chemically tested using established techniques. All samples analyzed for vibrios were serially diluted with 1 percent peptone, then enriched using an MPN series of 1 percent alkaline peptone at pH 8.4 and incubated at 37°C for 18-24 hours. All inoculated alkaline peptone tubes positive for growth were streaked on to Thiosulfate Citrate Bile Salts Sucrose (TCBS) and Cellobiose Polymyxin Colistin (CPC) agar plates and incubated at 37 and 40° C for 18-24 hours respectively. The appropriate dilutions and sample numbers were noted and isolated colonies from both the TCBS and CPC plates were further identified by biochemical methods using API 20E system (Analy Lab).

Quality control of culture identification was established by using cultures obtained from American Type Culture. Cultures of *V. alginolyticus, V. cholerae, V. damsela, V. fluvialis V. hollisae, V. parahaemolyticus V. metschnikovii, V. mimicus, V. vulnificus A hydrophila,* and *A. sobria* were used for positive controls. Data quality objectives applied and met for all analyses during this project appear in Table A.1 of Appendix A.

V.E. Metals and Toxic Organics Methods

Live shellfish were collected, rinsed briefly in ambient water, bagged and iced for transport to the laboratory. Storage was at 4°C until shellfish were processed within eight hours of collection. Analyses of shellfish for metals and organics began with a thorough scrubbing to remove all loose materials, using deionized laboratory water for rinsing. Organisms were opened using a stainless steel knife and the edible tissues, less shell liquor, removed and weighed. Each sample for analysis was the composite of a number of individuals and replicate samples from the same station and date each consisted of a different group of individuals. Tissues were homogenized using a combination of sonication and blending with a tissue homogenizer. The homogenate for each sample was then aliquoted into portions for metals, pesticide and PAH, and percent solids determination.

Digestion of tissues for metals analyses were those specified by Plumb (1981) and included an extended low temperature mineral acid digestion with nitric acid and peroxide. Digestates were then brought to a standard volume and analyzed via atomic absorption. Arsenic, cadmium, copper, lead, and zinc were analyzed by graphite furnace electrothermal atomization. A separate acid permanganate digestion, utilizing the tissue homogenate directly, was used for cold vapor mercury analyses. Standard curves for quantification were based on standard additions to avoid matrix effects. Tissue concentrations were converted from a wet to a dry weight basis using sample specific percent moisture determinations. Data quality objectives appear in Appendix A. Percent moisture determinations were conducted by drying an aliquot of the homogenized tissue to a constant weight at 105°C.

Toxic organics analyzed included selected carbamate, organophosphate and chlorinated pesticides, and polycyclic aromatic hydrocarbons (PAH). The analytical procedures followed the EPA methods established for water samples (Federal Register 40 CFR part 136, 1984) with modifications for tissue analysis. The analysis of PAH followed EPA-610 using capillary GC-FID with HPLC and GC/MS (EPA 625) for verification. Samples indicating the presence of PAH were further analyzed for mono-, di- and tri-methyl homologues of naphthalene, phenanthrene, dibenzothiophene and pyrene, to aid interpretation of the PAH source. The chlorinated pesticide samples were analyzed by methods EPA-608 and EPA-625, using capillary GC-ECD with verification of select samples by GC-MS. These specific compounds are listed below.

EPA 608 Pesticides:

α-BHC	o,p-'DDD
ß-BHC	o,p'-DDE
τ-BHC (lindane)	o,p'-DDT
aldrin	p,p'-DDD
dieldrin	p,p'-DDE
endrin	p,p'-DDT
heptachlor	α-chlordane
heptachlor epoxide	

EPA 610 PAH (including PAH hydrocarbons listed in EPA 625)

acenaphthene	acenaphthylene
benzo(a)anthracene	benzo(a)pyrene

benzo(b)fluoranthene	benzo(ghi)perylene
benzo(k)fluoranthene	chrysene
dibenzo(a,h)anthracene	fluoranthene
fluorene	indeno-pyrene
naphthalene	phenanthrene
anthracene	pyrene

Although no standard method is available for chlorpyrifos (dursban), dibrom (naled), and bendiocarb (ficam) in tissue samples, modifications of the standard EPA methods 608, 610, and 625 were used for these analyses. The halogenated organophosphates, chlorpyrifos and dibrom, were analyzed by GC-ECD and the carbamate, bendiocarb, by GC-FID.

The internal standards included: hexamethyl benzene (HMB) for PAH; tetrachlorometaxylene (TCMX) for chlorinated pesticides; and tetrachlorvinphos (TCVP) for organophosphate pesticides. The chromatographic columns were 30 m x 0.25 mm DB-5 fused silica capillary with He carrier gas and N_2 make-up gas.

Modifications of the analyses for tissues were according to procedures of Pierce et al., (1985), and the NOAA National Status and Trends Program (1984) and included the tissue extraction process and the column clean-up methodology.

An aliquot of the homogenized tissue sample (10 g) was placed into a 250-ml boiling flask fitted with a Stark and Dean moisture trap. Internal standards for each class of chemicals were added and the samples refluxed in cyclohexane (C_6H_{12}) to remove moisture and initiate extraction of the organic compounds. After removal of all moisture, the C_6H_{12} was removed and the sample refluxed with dichloromethane (CH_2Cl_2) to complete the extraction. The solvents were combined and evaporated and the residue dissolved in hexane for column chromatography cleanup.

The crude extract was separated in to characteristic fractions, and interfering compounds were removed by chromatographic separations through a column of silica gel and alumina, 80-200 mesh, topped with sodium sulfate to remove moisture, and copper powder to remove sulfur. The extract was added to the column in l-ml of hexane and eluted with 25 ml of hexane (Fraction 1). Fraction 1 contains saturated hydrocarbons and were archived for future petroleum analyses if desired. The column was then eluted with 25 ml of hexane/ dichloromethane (20:80) to recover the all three classes of <u>pesticide</u>s and <u>PAH</u> (Fraction 2). The third elution (Fraction 3) was with 25-ml of methanol (CH_3OH) to recover coprostanol and associated sterols which were also archived.

GC/MS confirmation of results were performed on a minimum of 10 percent of the samples by Dr. John P. Toth of the University of Florida, Institute of Food and Agricultural Sciences and MML. Tissue concentrations were converted from a wet to a dry weight basis using sample specific percent moisture determinations. Data quality objectives appear in Appendix A.

VI. RESULTS AND DISCUSSION

VI.A. Surveys - Results and Discussion

VI.A.1. Geographic Distribution and Abundance of Clams

From the 169 stations, a total of 1,273 clams in the spring and 938 in the fall were collected and measured. The number of clams from each station appears in Appendix B Table B.1. These results are summarized as a histogram in Figure 9 and in Table 10 for both clams per station (per 15 minutes effort) and clam lengths. The similarity between the seasons is apparent and indicates no large seasonal variations in populations. Of the 169 sampling stations, more clams were found during the spring than fall at 79 stations (47 percent); more during the fall than spring at 42 stations (25 percent); and an equal number of clams during each sampling at 48 stations (28 percent). The number of stations where no clams were found was nearly equal between surveys: 61 stations during the spring survey and 64 stations during the fall survey. No clams were found during either survey at 42 stations (25 percent).

The average number of clams per station was 7.53 and 5.55 (per 15 minutes effort) for the spring and fall surveys, respectively. The occurrence of clams at a station for only one survey was 19 (spring=0, fall >0) and 22 (spring>0, fall=0), or 11 percent and 12 percent of the stations, respectively. The average size of the clams was also similar between seasons, 96.79 mm and 97.98 mm for the spring and fall surveys, respectively.

Despite the similarity between seasons for numbers, sizes and distributions of clams, the possibility that variations in tides and vegetation between the seasons could produce variable results with the given sampling technique could not be ignored. In spring, the tides were lower and vegetation was less, so clams were more easily located visually. In the fall, clams were located more frequently by feel. This could account for the slight differences in clams per station and total numbers of *Mercenaria* between seasons. In addition, diurnal and day to day variations in tide height, wind, wave, and water clarity could produce similar confounding results within a season. Therefore, for all between station comparisons it was considered best to combine season totals to represent a more complete sampling effort. Clam abundance numbers for each station which follow are the result of a timed 30 minute search (three 5-minute replicates for each survey).

The dominant sediment type was also noted at each sampling station as mud, sandy mud, muddy sand, sand or shell. The sediment notations were qualitative, as any one station area often covered more than one substrate type, but by recording the dominant type it was possible to identify major trends in substrate effects on clam distribution. The number of stations of each type and total and mean number of clams measured is provided in Table 11.

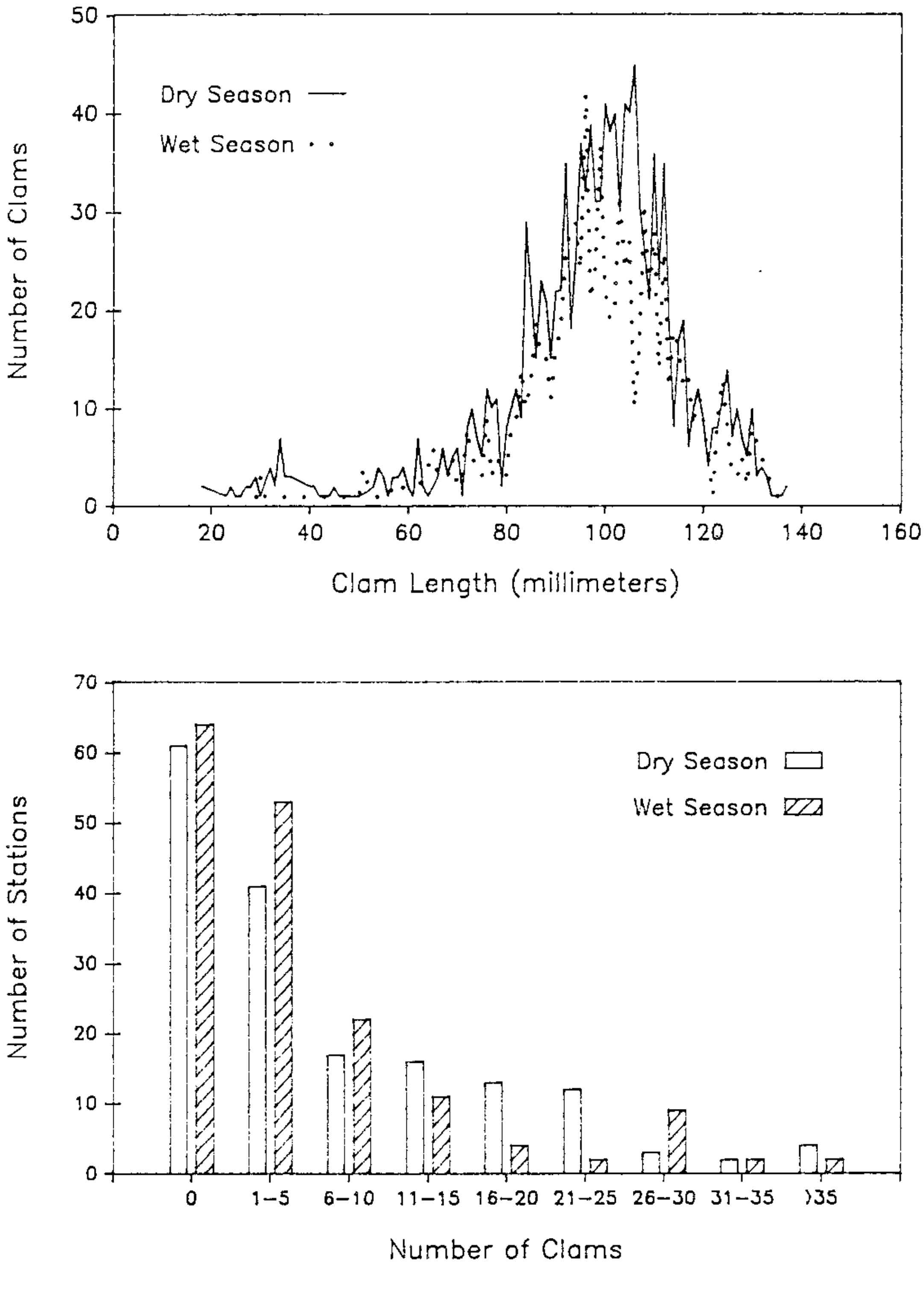

Figure 9. Distribution of *Mercenaria* spp. length and abundance during the spring and fall 1990 surveys.

Table 10. Summary of results of the spring and fall *Mercenaria* spp. surveys.

	Spring	Fall
Sampling Stations	169	169
No. of Clams Collected	1273	938
Average Clams per Station	7.5	5.6
Stations with 0 Clams	61	64
Average Clam Length (mm)	96.8	98.0
S.D.	16.9	19.5
Maximum Clam Length (mm)	137	137
Minimum Clam Length (mm)	18	29

Table 11. Descriptive statistics and ANOVA of mean clam count per station for differing sediment types.

	Mud	Sandy Mud	Muddy Sand	Sand	Shell
Count	52	75	63	119	29
Mean	6.2	4.7	8.5	7.5	3.7
Std Dev	9.9	7.6	10.3	9.7	6.2
Std Err	1.4	.9	1.3	.9	1.2
Variance	98.7	57.9	106.4	94.5	38.5
Coeff Var	161.0	161.2	121.9	129.1	168.2
Minimum	0	0	0	0	0
Maximum	37	36	41	41	24

1-Way ANOVA

	SS	DF	MS
Treatment	839.9	4	210.0
Error	28146.1	333	84.5
Total	28985.9	337	
F-test Ratio	2.484104		

Tukey's Studentized Range Test

Q = 3.858
k = 5
V = 333

Above the diagonal = Minimum significant differences
Below the diagonal P = .05 (***)

	Mud	Sandy Mud	Muddy Sand	Sand	Shell
Mud		4.5	4.7	4.2	5.8
Sandy Mud	-		4.3	3.7	5.5
Muddy Sand				3.9	5.6
Sand	-				5.2
Shell					

Stations described as shell were rarest (29 or 17 percent) and least productive of clams (mean 3.7). The shell classification used in this study referred to coarse substrate that may have prevented clams from burying and subjected them to predation. Stations with the next two coarsest sediment types, sand and muddy sand, were most common and productive. An ANOVA was performed to test for a relationship between substrate and clam abundance, and indicated that clam numbers were significantly different between sediment types ($p<.05$) but a means test (Tukey's HSD) failed to indicate any differences due to differing sensitivities between the statistical procedures (Table 11). The extremes in abundance, however, are represented by the shell stations with an average of 3.7 clams and muddy sand with 8.5 clams per 30 minute effort.

Previous studies also indicate sand substrate as more favorable than mud as clam habitat. In Georgia, clams were most abundant in sand with shell, next abundant in sand and least in mud (Walker et al., 1980). Sand with shell was not reported a common bottom type in South Carolina but supported 68 percent of the hard clams collected (Anderson et al., 1978). More locally, however, adult *M. campechiensis* did not show consistent correlation with particle size in Tampa Bay (Sims and Stokes, 1967).

Stations where no clams were found (Figure 10) in either survey were clustered primarily in two areas, adjacent to former Midnight Pass and on the east side of main Sarasota Bay. There were also a few stations between New Pass and Big Pass which yielded no clams. For Big Pass at least, high sand content and current velocities may prevent larval clams from setting or damage larval shells.

In the present study numerous dead clams near Midnight Pass were observed by sighting the characteristic keyhole shape of the siphon tube. These clams were oriented in the usual manner of a living clam but all tissue was gone and the shells filled with mud. As predation is eliminated based on the undisturbed shells, some relatively abrupt environmental factor is likely responsible. Most probable are freshwater inflows beyond the tolerance of *Mercenaria,* as Midnight Pass is directly opposite North and Catfish Creeks. Located approximately midway between Big Pass and the Venice Inlet, freshwater entering this supposed tidal null zone, and supported by flows from Phillippi Creek and South Creek to the north and south, would tend to have a greater influence than elsewhere in the study area. Salinities were not noted during the surveys, however, but correlations of clam abundance to salinity data gathered during other monitoring programs were made and are discussed below.

An additional factor, which is related to the tidal null zone may be low current velocities. The fact that the "keyholes" remained intact above dead clams indicates extremely low velocities, although currents were not measured during the clam survey. Increased clam density (Wells, 1957) and growth (Kerswill, 1949) has been shown to correlate with water current, and while low currents may not provide enough food for clam survival, this too, is apparently an episodic event as clams set and matured in the area.

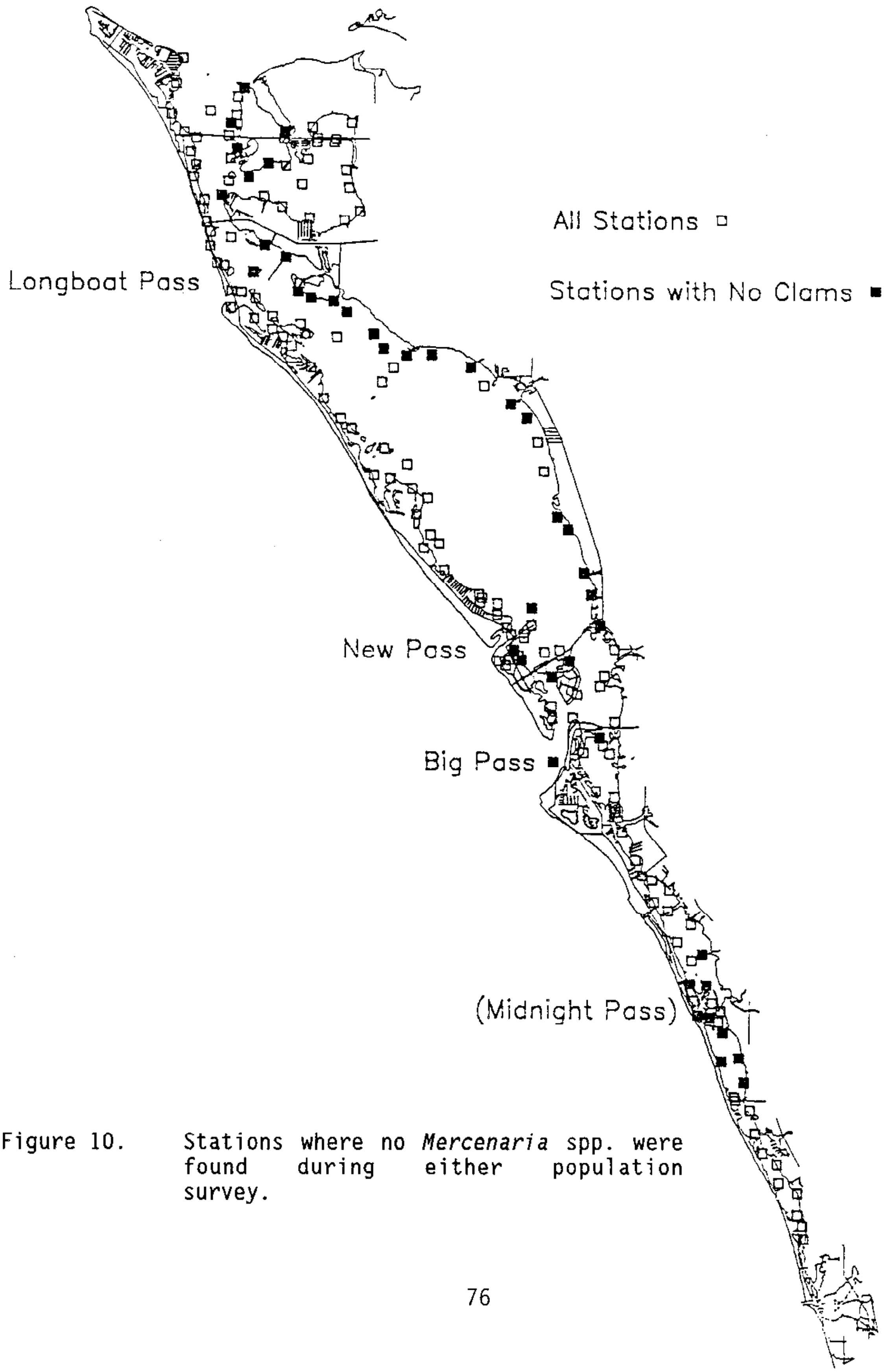

Figure 10. Stations where no *Mercenaria* spp. were found during either population survey.

The lack of clams on the east side of Sarasota Bay is not easily explained. It is reported that this pattern of more clams on the west side of bays is common on the Florida west coast, however (Don Hesselman, personal communication), and could be associated with bathymetry, sediment type, or predation.

Stations where abundant (more than 21 per 30 minute effort) clams were found (Figure 11) in both surveys primarily occurred on the west side of main Sarasota Bay along Longboat Key and southern Anna Maria Island. Only four stations in the abundant clam group were south of Siesta Drive and none in the Midnight Pass area. These results may be partially explained by the predominantly sand substrate in the northern area. The absence of clam stations in the greater than 21 category in the Midnight Pass area is probably not sediment related but a result of freshwater and currents as discussed previously.

The second highest class of clam abundance selected was 8-21 clams (Figure 12). Stations in this group were found in the Longboat Pass area, Palma Sola Bay, from New Pass to Stickney Point, and the most southern extent to the study area. No stations near Midnight Pass and only one station along the east side of Sarasota Bay were in this category.

Stations with low clam yield (1-7 clams) (Figure 13) were found in Palma Sola Bay, along northern Anna Maria Island, Lido Key and Bird Key, and widely distributed in the southern third of the study area, including Midnight Pass.

The presence and density of vegetation was noted at each sampling station during the surveys. As with the sediment types, the vegetation habitats were qualitative assessments of the dominant habitat in the area or the habitat where most clams were found. The vegetation classifications are listed in Table 12 with the number of stations and clams collected. The most clams per station (8.0) were found in sparse *Halodule* (manatee grass), and the least (3.5) in dense *Halodule.* In each habitat the standard deviation exceeded the mean number of clams and as expected with such variability no significant differences occurred between means.

Overall, the greatest number of clams was collected from bare and sparse *Halodule* areas. These two habitats provided the easiest environments for locating clams visually due to the thin and sparse seagrass blades and also the tendency of these stations to be shallower. An association of clam distribution with *Thalassia testudinum* previously reported (Schroeder, 1924; Woodburn, 1962; Sims and Stokes, 1967; Taylor and Saloman, 1968, 1970; Godcharles, 1971) was neither confirmed or denied in the survey. Present results agree partially with those of Peterson (1982) where greater clam densities were observed in sparsely vegetated than bare areas, but not with Peterson's result of greater densities in densely versus sparsely vegetated areas.

It is difficult to separate actual densities from those which may be an artifact of the qualitative survey technique used for this project.

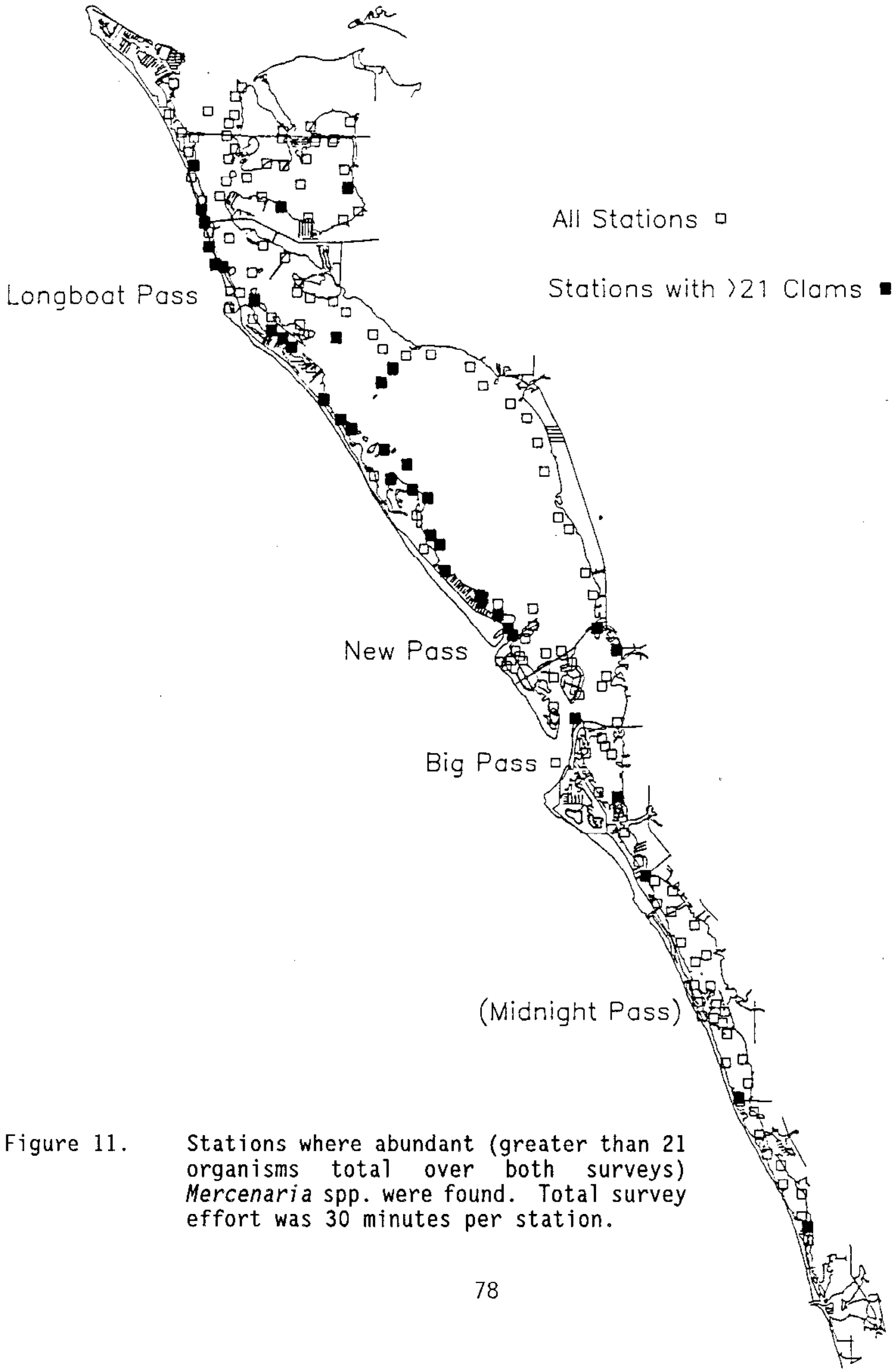

Figure 11. Stations where abundant (greater than 21 organisms total over both surveys) *Mercenaria* spp. were found. Total survey effort was 30 minutes per station.

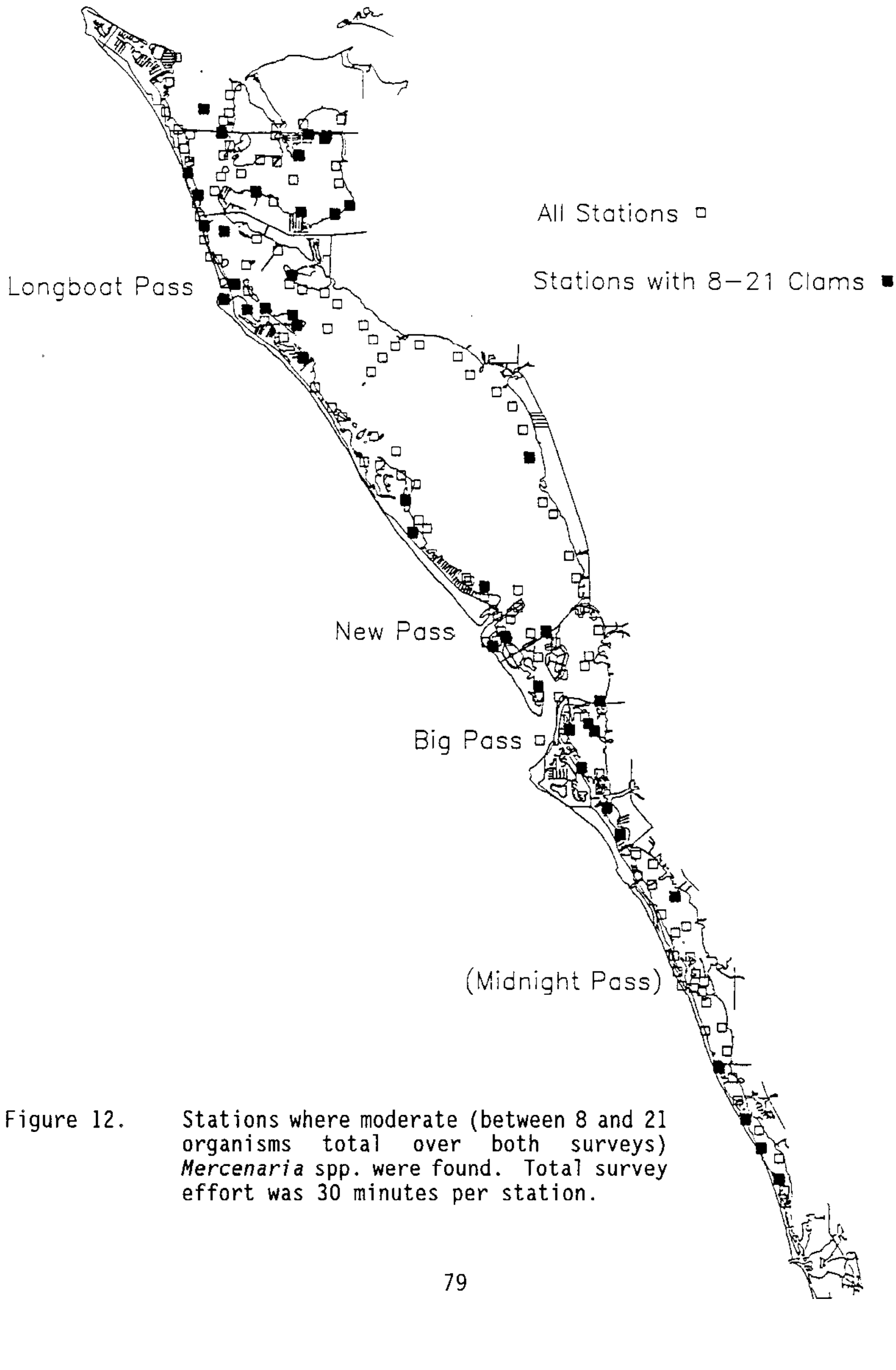

Figure 12. Stations where moderate (between 8 and 21 organisms total over both surveys) *Mercenaria* spp. were found. Total survey effort was 30 minutes per station.

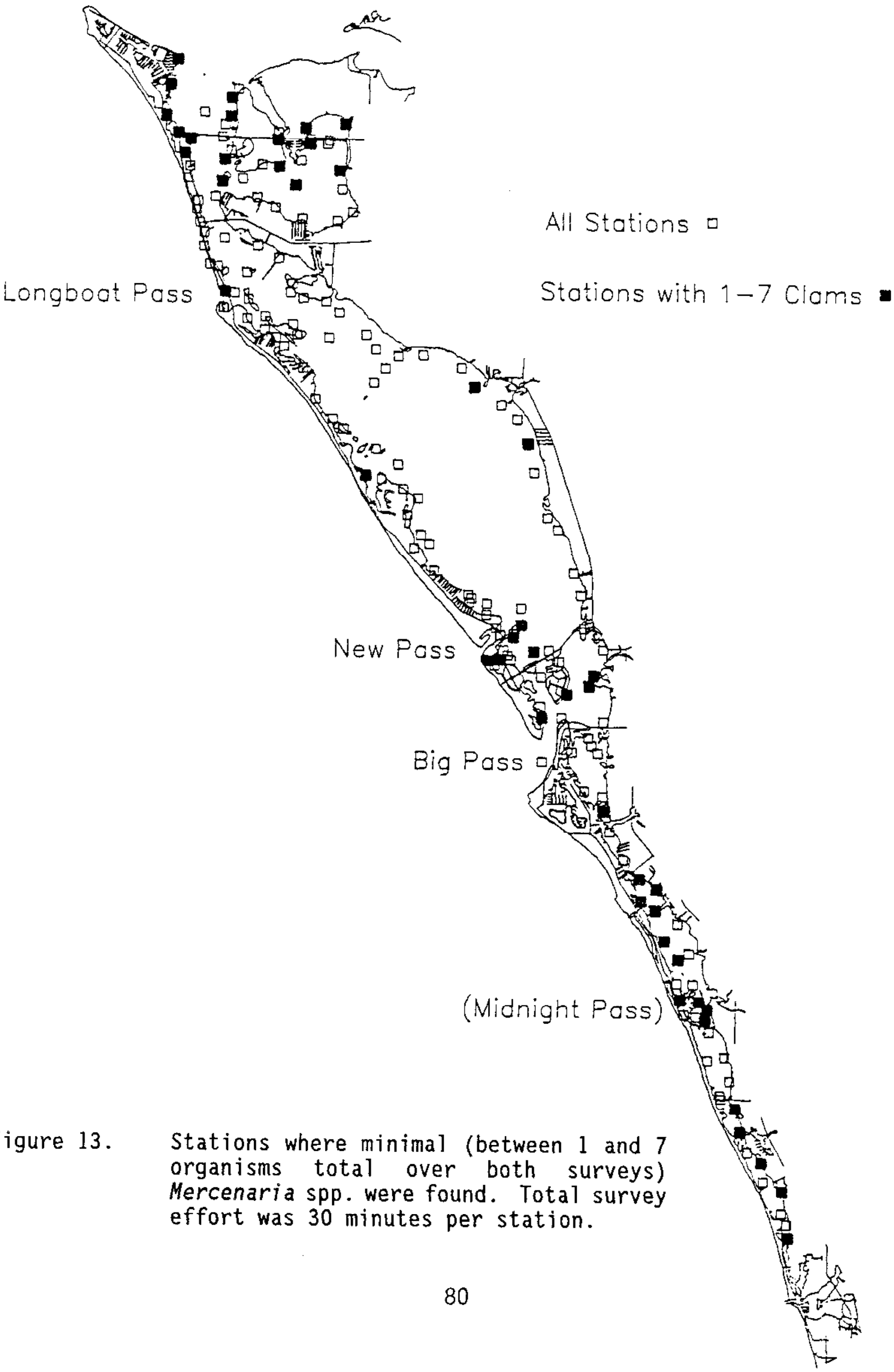

Figure 13. Stations where minimal (between 1 and 7 organisms total over both surveys) *Mercenaria* spp. were found. Total survey effort was 30 minutes per station.

Table 12. Descriptive statistics and ANOVA of mean clam count per station separated into vegetation types.

	Bare	Sparse *Halodule*	Dense *Halodule*	Sparse *Thalassia*	Dense *Thalassia*
Stations	80	51	33	35	32
Mean	5.7	8.0	3.5	7.6	7.7
Std Dev	8.6	10.4	5.7	8.5	8.7
Std Err	1.0	1.5	1.0	1.4	1.5
Variance	73.2	108.7	32.9	72.3	76.5
Coeff Var	150.5	130.0	163.3	112.3	113.3
Minimum	0	0	0	0	0
Maximum	38	41	27	33	34

1-Way ANOVA

	SS	DF	MS
Treatment	551.9	4	138.0
Error	17103.5	226	75.7
Total	17655.4	230	
F-test Ratio	1.8	N.S.	

This possibility was recognized during the design of the project, but a relatively a non-destructive technique was desireable. Even the qualitative procedure used, however, was noted to be destructive, particularly in *Thalassia* beds in soft mud.

Monitoring stations from the first year of the NEP water quality study were paired, where possible, to clam survey stations to compare 25 water quality parameters to the number and size of the clams collected (Table 13). A total of 55 station pairs were selected. ANOVA's were performed between the parameters to evaluate relationships. The number of clams per station did not correlate significantly with any parameter. This was partly a result of the high variability of clam distribution.

Although not observed in the ANOVA analysis, high salinities have a potential effect on clam recruitment. Values greater than the 32.5 ppt maximum for egg to veliger development reported for *M. mercenaria* are common, and, as *M. campechiensis* is commonly found offshore, high salinity in the Sarasota NEP area is not likely to affect adult clams and a large set of juvenile clams was reported in New Pass for 1991 (Don Hesselman, personal communication). As postulated for the Midnight Pass area, low salinities which persist beyond the food reserves of adults may have adverse effects in selected areas.

VI.A.2. Clam Distribution by Segment

A reduction of clam abundances by segments appears in Table 14, together with the approximate segment areas, the areas less than 3 feet below mean lower low water, and the respective percentages of the entire study area. Clam survey stations were generally distributed to represent recreational clamming (wading) and so the number of survey stations is generally proportional to the shallow area rather than the total expanse of the segment.

The average number of clams per station by segment varies from zero to over 35 animals per 30 minute effort at a station. Those segments with abundant clams were western Anna Maria Sound (21.2 clams per 30 minutes), western Sarasota Bay (35.5), and New Pass (23.0). Few clams per station were collected in the Big Pass, Midnight Pass, and Little Sarasota Bay segments. Those segments with comparatively low abundances were located along the eastern shore of the Bay, near passes, or in the southern portion of the study area.

VI.A.3. Clam Predators

The primary emphasis in the *Mercenaria* surveys was to determine the presence, abundance and size of clams at each station, but additionally the presence of clam predators was noted during both surveys. Predation has been indicated as the most important biotic factor regulating hard clam populations (Virstein, 1977; MacKenzie, 1979). The major predators consisted of several large gastropods: *Busycon* spp., *Pleurocantha*

Table 13. ANOVA results for water quality parameters and mean clam count per station. No significant differences occurred.

	Count	Slope	Intercept	r2	F-test
Bottom Dissolved Oxygen	55	.002	7.021	.001	.051
Bottom Temperature	55	-.010	25.543	.012	.651
Bottom % Saturation	55	.048	97.277	.004	.219
Bottom Salinity	55	.028	32.918	.039	2.175
Surface Dissolved Oxygen	55	.000	7.070	.000	.008
Surface % Saturation	55	.019	97.743	.001	.045
Surface Salinity	55	.040	32.171	.032	1.755
Surface Temperature	55	-.003	25.370	.006	.331
Chlorophyll A	55	-.054	7.048	.046	2.527
Chlorophyll B	55	-.012	2.386	.002	.130
Chlorophyll C	55	-.048	4.589	.018	.994
Phaeophytin	55	-.223	79.371	.011	.586
Color	55	-.076	14.636	.017	.898
Particle Count	55	39.584	56653.104	.000	.016
Potassium	54	-.007	.915	.047	2.586
NH3N	55	-.001	.020	.009	.503
NO23N	55	.000	.069	.016	.883
Orthophosphate	55	.000	.048	.031	1.724
Total Phosphorus	55	-.001	.164	.056	3.123
Total Inorganic Carbon	55	-.015	26.658	.011	.602
Total Kjeldahl Nitrogen	55	-.002	.650	.023	1.257
Total Organic Carbon	55	-.006	2.001	.006	.334
Total Suspended Solids	55	-.052	12.237	.024	1.292
Turbidity	55	-.016	4.265	.014	.760
Volatile Suspended Solids	55	-.016	4.663	.015	.785

Table 14. Summary statistics of segmentation areas with clam stations and counts.

Area	Segment Number	Percent of Total Area	Square Miles	Percent of Bay <3 feet	Sq Miles <3 feet	No. of Stations	No. of Clams*	Avg. No. of Clams/ Stations*
West Anna Maria Sound	1	5	2.8	8	1.5	12	254	21.2
West Palma Sola Bay	2	7	3.4	13	2.5	16	82	5.1
East Palma Sola Bay	3	5	2.8	6	1.2	13	151	11.6
Long Boat Pass	4	<1	.2	<1	<.1	2	27	13.5
North Longboat/Jewfish	5	7	3.6	9	1.7	14	339	24.2
Tidy Island	6	14	7.3	21	3.8	19	159	8.4
West Sarasota Bay	7	17	8.8	8	1.6	13	462	35.5
East Sarasota Bay	8	10	5.3	3	.6	4	21	5.3
New Pass	9	1	.3	<1	.1	2	46	23.0
City Island/Bird Key	10	11	5.5	8	1.4	22	233	10.6
Downtown Sarasota	11	8	3.8	2	.3	10	116	11.6
Big Pass	12	1	.7	1	.2	1	0	0.0
Roberts Bay	13	5	2.4	6	1.1	13	162	12.5
Little Sarasota Bay	14	6	2.8	8	1.5	15	67	4.5
Midnight Pass	15	<1	.2	1	.1	3	4	1.3
Blackburn Bay	16	2	1.2	4	1.5	10	88	8.8
	Totals	100	51.1	100	19.1	169	2211	

* per 30 min. effort

gigantea and *Tulippa* spp. Other motile predators such as blue crabs are widely distributed throughout the area and, while particularly important for predation on newly attached larvae, were not noted due to their mobility and avoidance of samplers.

Figure 14 illustrates the clam stations where predacious gastropods were observed. The figure does not coincide closely with any other figure regarding clam size or abundance. Predators were predominantly noted in the northern half of the study area. They also were found at stations on the east side of main Sarasota Bay where no clams were observed. This may imply that other prey species were being utilized, as other bivalves such as *Chione cancellata* were seen in this area.

In summary, clams were most abundant on the west side of Sarasota Bay, clams were conspicuously scarce on the east side of the bay and near Midnight Pass, and clams in other areas generally were present but in lower numbers.

VI.A.4. Size Distribution of Clams

Each clam collected in the surveys was measured prior to release. Figure 9 illustrates the distribution in clam lengths for each survey. Results were similar in each season and reveal an overall lack of small clams. Sampling procedure was somewhat biased towards large clams but numerous small were collected (minimum 18 mm) and the lack of small clams can not be presumed to be an artifact of sampling. Figures 15 and 16 illustrate the clam stations (where greater than 7 clams were collected) for mean length of greater and less than 100 mm. It is evident that the area inside Longboat Key where clams were abundant also produced large clams. The fact that this area on the southeast side of Longboat Key is the only currently approved shellfishing waters in the study area would appear to indicate that harvesting pressure are minimal (assuming that no size selection takes place during harvest).

The smaller, although still large, mean clam lengths 400 mm were found at stations roughly in the area of passes: Longboat Pass; New and Big Passes; and in the far south portion of the study area near Venice Jetties. These may represent more recent sets of cohorts. Stations where clams of less than 50 mm were collected are shown in Figure 17, but no pattern of occurrence is apparent. The rapid growth habits of *Mercenaria* and intense predation on juveniles undoubtedly bias populations towards larger individuals. However, even allowing for this and assuming that the individuals less than 50 mm are less than two years old (Jones et al., 1990), it is apparent that recruitment rates are relatively slow.

Clam lengths were also analyzed according to sediment type Table 15). Results of differences were highly significant ($p>.01$). Mean clam size from sand stations was larger than all others and clams size from shell stations was smaller than all others. The clam sizes for mud, sandy mud and muddy sand did not differ significantly. The result of greater clam size from the sand stations agrees with research of Johnson

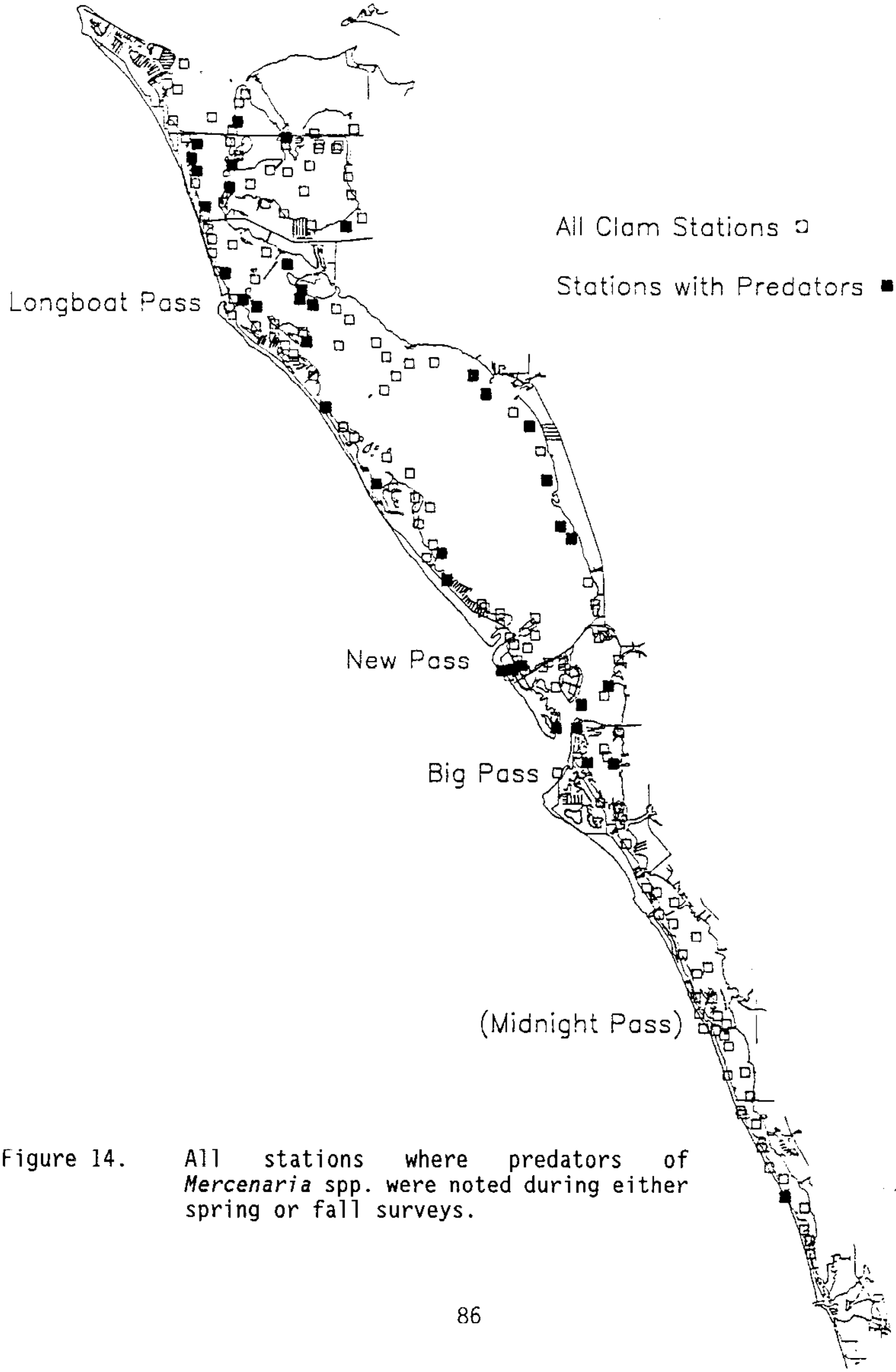

Figure 14. All stations where predators of *Mercenaria* spp. were noted during either spring or fall surveys.

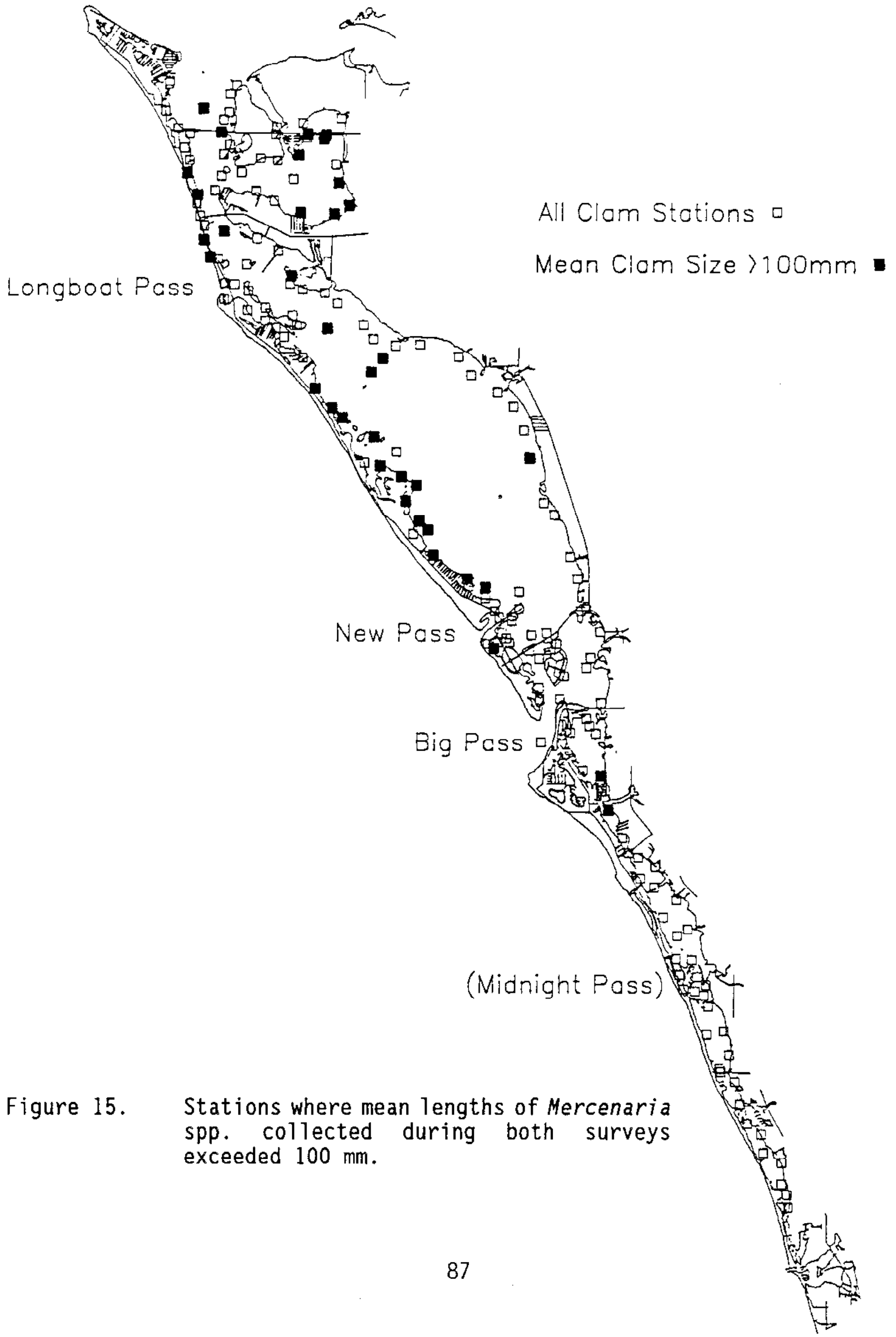

Figure 15. Stations where mean lengths of *Mercenaria* spp. collected during both surveys exceeded 100 mm.

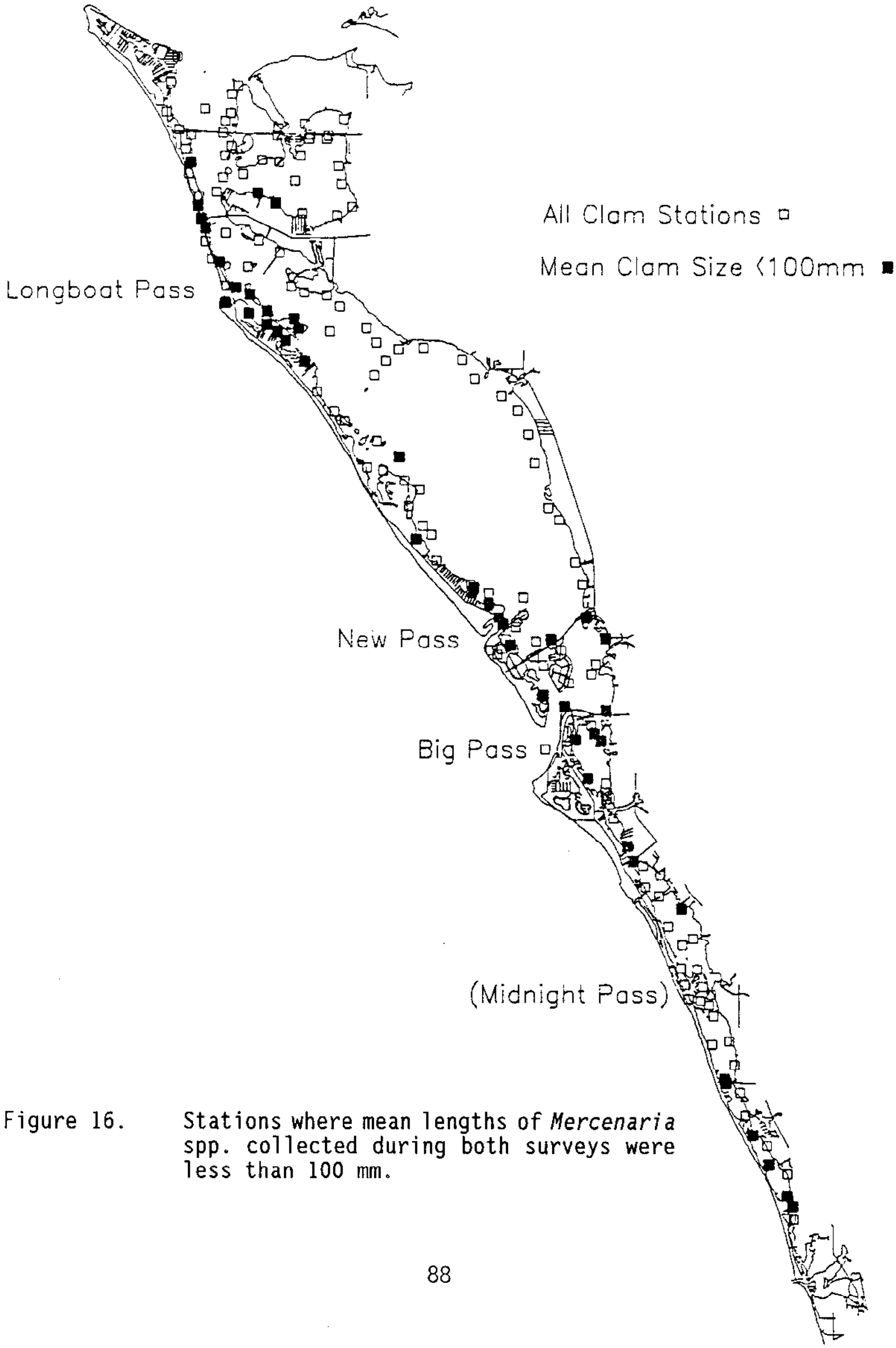

Figure 16. Stations where mean lengths of *Mercenaria* spp. collected during both surveys were less than 100 mm.

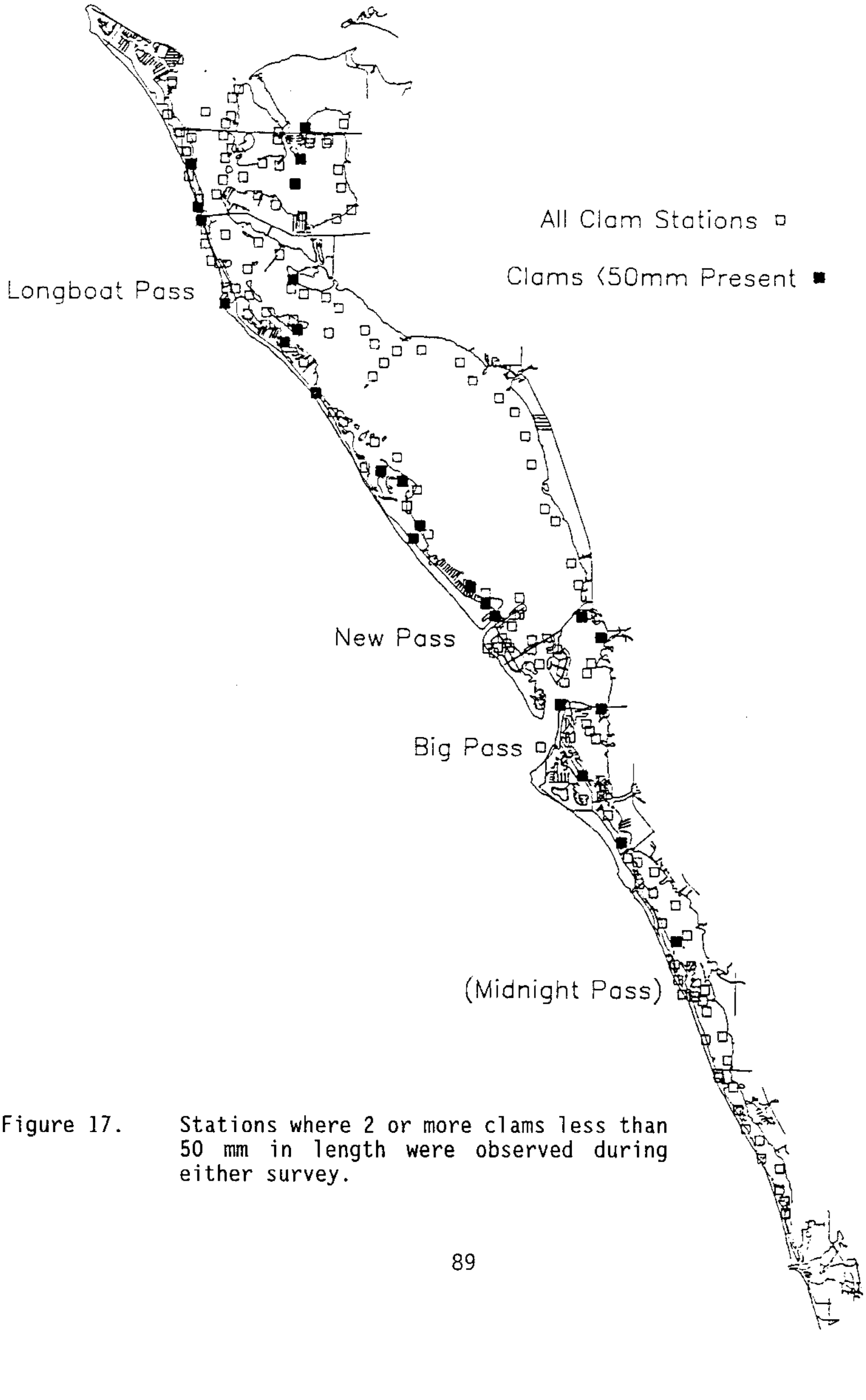

Figure 17. Stations where 2 or more clams less than 50 mm in length were observed during either survey.

Table 15. Descriptive statistics, ANOVA and means comparisons of clam lengths from differing sediment types.

	Mud	Mud	Sandy Sand	Muddy Sand	Shell
Count	305	364	533	896	113
Mean	95.2	97.8	94.6	100.8	86.0
Std Dev	17.4	15.9	20.4	18.0	14.8
Std Err	1.0	.8	.9	.6	1.4
Variance	304.1	252.4	418.1	324.2	219.4
Coeff Var	18.3	16.2	21.6	17.9	17.2
Minimum	28	29	25	18	29
Maximum	135	133	137	137	120

1-Way ANOVA

	SS	DF	MS
Treatment	30987.6	4	7746.9
Error	721232.4	2206	326.9
Total	752220.0	2210	
F-test Ratio	23.69505		

Tukey's Studentized Range Test

Q = 3.858
k = 5
V = 2206

Above the diagonal = Minimum significant differences
Below the diagonal P = .05 (***)

	Mud	Sandy Mud	Muddy Sand	Sand	Shell
Mud		3.8	3.5	3.3	5.4
Sandy Mud			3.4	3.1	5.3
Muddy Sand				2.7	5.1
Sand	***	***	***		4.9
Shell	***	***	***	***	

(1977) who showed a correlation between clam size and sediment particle size and Pratt (1953) demonstrated that clams grew faster in sand than mud. As described above, the shell classification in the present study only referred to coarse sediments which may have prevented clam burial and subjected the clams to predation. This possibly explains the smaller mean size and presumably younger clams from the shell stations.

The size of the clams from each vegetation was also analyzed (Table 16), but the range of mean values was very small (96.0 mm to 99.8 mm) with no significant differences detected.

As with clam abundance, data from the monitoring stations from the first year of the NEP water quality study were also used to correlate 25 water quality parameters to the size of the clams collected (Table 17). Several significant correlations occurred. Only 40 station pairs were used in these analyses since all stations did not produce clams. Nitrate-nitrite nitrogen, orthophosphate, particle count, and total organic carbon correlated positively ($p<.05$) to clam size. Surface dissolved oxygen correlated negatively to clam size ($p<.01$).

The correlations of nitrate-nitrite nitrogen, orthophosphate, total organic carbon and particle count with clam length may simply reflect the results shown in Figures 16 and 17, where smaller mean clam size was observed near the passes to the Gulf of Mexico, rather than implying causality. These parameters are typically those associated with tributary loading to a coastal area and areas of relatively greater flushing will reduce these levels. The interaction of these nutrients with primary producers and food organisms for *Mercenaria is* also complex, but may provide a causal link. The remainder of the water quality parameters showed no significant correlations to clam size.

The negative correlation of dissolved oxygen to clam length is not likely a cause-effect relationship. The smaller clams in the area of passes would not experience low D.O. values, thus a significant, but possibly misleading, statistical correlation occurs. Hard clam tolerance to low D.O. and the D.O. values recorded during the first year of monitoring do not indicate D.O. as a controlling factor of clam size or distribution in the study area. The monitoring plan, however, is designed to provide occasional spatially intensive "snapshots" of water quality and samplings are conducted during midday to obtain light attenuation readings. Nighttime or bloom related D.O. levels can drop substantially (Dr. D. Tomasko, unpublished data). If a causal link is present, larger adult *Mercenaria* may have the food reserves necessary to compensate for periods of valve closure associated with low dissolved oxygen levels.

VI.A.5. Oyster Survey

Oyster stations were tentatively selected for tissue collection prior to field work. Habitat maps generated from aerial photographs (Mangrove Systems, 1988) were examined to determine oyster distribution. These results showed approximately ten notations of oysters between City

Table 16. Descriptive statistics and ANOVA of clam lengths from differing vegetation types.

	Bare	Sparse *Halodule*	Dense *Halodule*	Sparse *Thalassia*	Dense *Thalassia*
Count	455	409	116	265	247
Mean	98.1	96.0	96.1	99.8	97.4
Std Dev	20.9	18.7	19.6	17.5	16.1
Std Err	1.0	.9	1.8	1.1	1.0
Variance	436.2	351.5	384.7	305.1	257.9
Coeff Var	21.3	19.5	20.4	17.5	16.5
Minimum	24	23	30	18	31
Maximum	137	133	137	135	134

1-Way ANOVA

	SS	DF	MS
Treatment	2724.6	4	681.1
Error	529683.5	1487	356.2
Total	532408.1	1491	
F-test Ratio	1.9122105 (N.S.)		

Table 17. ANOVA results for water quality parameters and length of clams. Significant differences (p=.05) are indicated with asterisks.

	Count	Slope	Intercept	r2	F-test
Bottom Dissolved Oxygen	40	-.009	7.871	.028	1.107
Bottom Temperature	40	.020	23.651	.041	1.635
Bottom % Saturation	40	-.021	99.173	.001	.029
Bottom Salinity	40	.023	30.894	.021	.826
Surface Dissolved Oxygen	40	-.018	8.749	.213	10.268*
Surface % Saturation	40	-.193	115.081	.084	3.473
Surface Salinity	40	-.034	35.550	.018	.692
Surface Temperature	40	-.008	26.139	.037	1.473
Chlorophyll A	40	.073	-1.070	.093	3.910
Chlorophyll B	40	-.007	2.531	.001	.052
Chlorophyll C	40	-.001	3.546	.000	.000
Phaeophytin	40	.248	53.314	.012	.468
Color	40	.193	-4.482	.083	3.436
Particle Count	40	703.233	-15445.204	.111	4.751*
Light Attenuation	39	.008	.023	.077	3.108
NH3N	40	.000	.004	.027	1.069
NO23N	40	.001	-.063	.168	7.650*
Orthophosphate	40	.001	-.041	.114	4.907*
Total Phosphorus	40	.001	.039	.069	2.820
Total Inorganic Carbon	40	-.016	28.177	.011	.421
Total Kjeldahl Nitrogen	40	.004	.276	.064	2.580
Total Organic Carbon	40	.032	-1.063	.125	5.411*
Total Suspended Solids	40	.023	8.574	.006	.231
Turbidity	40	.008	3.272	.004	.169
Volatile Suspended Solids	40	.034	.917	.078	3.224

Island and Stickney Point, 40 notations between Stickney Point and Blackburn Point, and about 20 from Blackburn Point to Albee Road. No oysters were noted north of City Island to the Sarasota County line. During the clam survey observations were made on other areas where oysters might be collected. Figure 18 illustrates 38 areas (not including the ten tissue collection stations) where oyster bars were evident and assessed for possible tissue collection. It is clear in the figure that oysters were more abundant in the southern area. The shallow and more enclosed bays provide good habitat, and small bars or scattered clumps of oysters are visible from any point south of Siesta Drive at low tide.

Results of the present study suggest a more ubiquitous distribution than the existing mapping, although north of City Island, oyster bars were small and often only an aggregation of small clumps along shore. Scattered clumps were often observed along mangrove shorelines. Some relict bars were found on the east side of Sarasota Bay but few live oysters were present. The majority of oyster reefs occurred at the mouths of creeks, where presumably favorable salinity gradients reduced predation, with the exception of Whitaker Bayou, where no live oysters were observed. The predator *Melongena corona* (Florida crown conch), however, was present at nearly every oyster area examined.

The ten tissue collection stations were selected from the most productive areas which included seven stations directly associated with a creek. The stations were chosen to cover the area as completely as possible. They are listed below with brief descriptions of the conditions of the oyster bars. Abiotic parameters recorded during these collections appear in Appendix Table B.2. Ten NEP water quality monitoring stations were selected to correspond to the oyster stations and mean values are given for several water quality parameters in Table 18. The results are also presented in Table 19 where the values are represented with symbols indicating the relative ranking, or whether the mean value for a station was among the highest three or lowest three of the means of all ten stations. This table is provided and commented on below to give a relative comparison between the station environments.

Bowlees Creek. Some of the most extensive oyster bars in the study were found here. Oyster growth may be aided by greater currents and predator control through salinity fluctuations. A boat channel bisects the reefs, however, and intense development including marinas exists immediately upstream. Relatively high particulate, nutrient and chlorophyll values occurred.

City Island. Several round senescent (dead crests) bars are located just south of City Island. No source of significant freshwater flow is nearby, and predation may be high.

Hudson Bayou. Scattered oyster clumps line areas of the bayou shore. Although water currents and salinity changes may be favorable, little area is available for reef development. Turbidity and suspended solids values were relatively high.

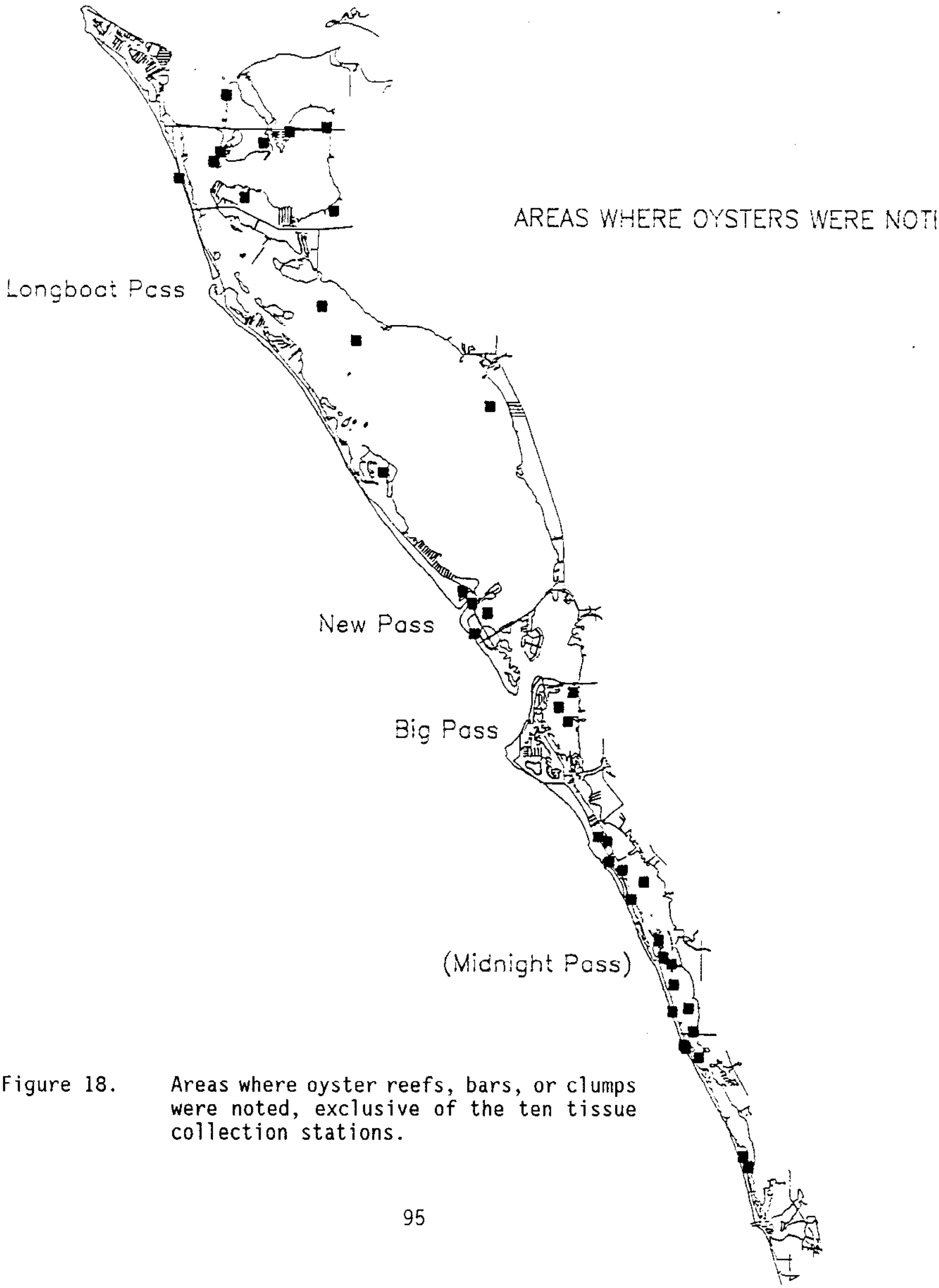

Figure 18. Areas where oyster reefs, bars, or clumps were noted, exclusive of the ten tissue collection stations.

Table 18. Mean and standard deviation for water quality parameters from monitoring stations near oyster tissue collection stations.

		Bowlees Creek	City Island	Hudson Bayou	North Creek	Palma Sola Creek	Perico Bayou	Phillippi Creek	Siesta Bridge	South Creek	Tidy Island
Bottom Dissolved Oxygen	Mean	7.13	7.87	6.03	8.53	6.75	4.50	5.75	8.15	6.63	8.05
	S.D.	1.06	1.34	1.42	1.47	1.77	2.55	1.25	2.11	.31	2.00
Bottom Temperature	Mean	26.50	27.50	25.25	26.03	30.80	30.90	24.37	26.40	25.78	24.90
	S.D.	5.30	5.41	5.98	5.85	4.10	5.80	6.44	4.92	5.47	6.51
Bottom % Saturation	Mean	102.80	113.13	86.35	112.93	102.20	71.75	67.60	103.70	89.08	113.95
	SD.	4.31	15.45	15.82	7.15	45.11	53.53	27.06	16.95	10.75	36.63
Bottom Salinity	Mean	31.03	34.20	29.47	30.10	31.90	30.40	18.60	31.60	31.48	33.25
	S.D.	2.57	.98	3.49	1.37	1.56	5.09	N/A	2.58	2.35	1.88
Surface Dissolved Oxygen	Mean	6.68	7.68	6.08	8.48	6.83	5.60	6.08	8.30	6.80	8.25
	S.D.	1.50	.98	.74	1.52	1.65	1.98	1.51	1.17	.37	1.63
Surface % Saturation	Mean	93.85	107.90	89.58	112.73	92.35	80.70	81.05	108.03	91.50	116.08
	S.D.	36.02	12.35	16.01	7.67	27.73	32.88	9.67	8.14	9.00	31.72
Surface Salinity	Mean	20.00	34.05	30.05	29.75	22.93	31.05	17.30	31.28	31.68	32.95
	S.D.	7.65	1.53	3.65	1.00	14.78	3.40	4.20	2.63	2.44	1.59
Surface Temperature	Mean	26.75	25.38	26.35	26.15	25.98	25.73	25.95	26.75	25.83	25.00
	S.D.	5.35	5.78	6.13	5.79	6.23	6.86	6.57	5.42	5.54	6.67
Chlorophyll A	Mean	19.30	3.67	6.91	7.83	20.68	11.07	8.13	3.68	6.31	6.70
	S.D.	26.22	3.37	4.33	8.89	20.76	17.70	6.16	2.49	5.03	5.43
Chlorophyll B	Mean	9.24	.00	.00	5.51	.12	8.92	.29	.00	.26	.77
	S.D.	18.18	.00	.00	10.46	.23	19.88	.32	.00	.53	1.54
Chlorophyll C	Mean	18.15	.35	.95	9.52	3.42	14.27	1.25	.26	1.14	2.57
	S.D.	35.35	.42	1.04	17.94	3.91	30.08	.80	.30	1.07	2.17
Phaeophytin	Mean	134.33	67.20	80.93	115.68	147.09	97.90	61.27	85.57	102.02	68.55
	S.D.	118.89	40.78	53.22	36.54	107.33	75.43	45.36	24.38	23.80	82.61
Color	Mean	40.00	8.00	15.00	18.75	57.50	21.00	40.00	10.00	13.75	16.25
	S.D.	15.81	4.47	4.08	2.50	37.97	8.22	14.72	.00	2.50	6.29

Table 18. Continued. Mean and standard deviation for water quality parameters from monitoring stations near oyster tissue collection stations.

		Bowlees Creek	City Island	Hudson Bayou	North Creek	Palma Sola Creek	Perico Bayou	Phillippi Creek	Siesta Bridge	South Creek	Tidy Island
Particle Count	**Mean**	**146153**	37744	**71628**	**111240**	97365	73767	**71444**	49021	62800	**49748**
	S.D.	**146304**	15708	**26952**	**117732**	35490	54126	**12802**	29697	25577	46153
Light Attenuation	**Mean**	**.81**	.24	**1.18**	1.09	N/A	2.60	**.97**	.79	.81	.71
	S.D.	**N/A**	.12	.92	**.12**	N/A	1.31	.45	.43	.18	.27
NH3N	**Mean**	-.01	-.01	-.01	-.01	-.01	-.01	.03	-.01	-.01	-.01
	S.D.	.00	N/A	.00	.00	.00	N/A	.06	.00	.00	.00
NO23N	**Mean**	.20	.02	.06	.07	.31	.06	.15	.05	.05	.08
	S.D.	.20	.01	.03	.03	.35	.01	.10	.03	.01	.02
Orthophosphate	**Mean**	.09	.02	.05	.04	.18	.04	.15	.04	.02	.04
	S.D.	.08	.02	.02	.02	.20	.02	.04	.02	.02	.03
Total Phosphorus	**Mean**	.18	.12	.13	.16	.23	.15	.22	.14	.13	.20
	S.D.	.08	.06	.08	.07	.09	.03	.13	.10	.08	.10
Total Inorganic Carbon	**Mean**	33.58	26.48	27.19	27.49	23.35	26.87	28.28	27.11	28.17	25.05
	S.D.	8.68	4.17	4.47	5.41	3.53	3.18	6.68	4.24	5.77	4.61
Total Kjeldahl Nitrogen	**Mean**	1.08	.71	.65	.77	1.18	.93	.99	.68	.38	.80
	S.D.	.37	.47	.21	.38	.14	.24	.21	.48	.27	.39
Total Organic Carbon	**Mean**	5.17	1.40	1.79	2.59	7.99	2.74	4.74	1.60	1.96	1.77
	S.D.	2.36	.63	.26	.54	8.84	1.41	2.95	.48	.77	.55
Total Suspended Solids	**Mean**	7.35	10.72	14.60	10.20	28.30	12.00	7.70	7.60	15.68	14.50
	S.D.	2.70	6.37	7.11	3.66	27.57	5.00	1.97	3.50	6.77	8.00
Turbidity	**Mean**	4.13	3.68	5.20	4.80	7.63	3.74	4.45	3.75	4.85	2.48
	S.D.	1.03	.80	.42	.71	7.22	2.14	.73	1.55	1.89	.88
Volatile Suspended Solids	**Mean**	5.43	3.68	6.60	5.00	12.33	5.20	3.50	2.90	6.00	5.00
	S.D.	2.29	1.78	2.62	2.03	10.06	2.51	1.05	1.32	2.61	2.62

Table 19. Ranking of mean water quality values from NEP monitoring for stations corresponding to oyster tissue collection stations (*** indicates station among highest 3 means; * indicates station among lowest 3 means).

	Tidy Island	City Island	Siesta Bridge	Perico Bayou	Palma Sola Creek	Bowlees Creek	Hudson Bayou	Phillippi Creek	North Creek	South Creek
Abiotic Water Quality Parameters										
Surface Temp.	*	*	***			***	***			*
Bottom Temp.	*	***		***	***			*		*
Surface D.O.	***		***	*			*		***	*
Bottom D.O.	***	***	***	*				*		*
Surface % Saturation	***	***	***	*				*		*
Bottom % Saturation	***	***	***	*				*		*
Surface Salinity	***	***	***		*	*		*		
Bottom Salinity	***	***			***		*	*		*
Light Attenuation	*	*		***			***	***		*
Organic and Inorganic Particulates										
Color		*	*		***	***	*	***		
Turbidity	*	*		*	***		***		***	
Total Suspended Solids	***		*		***	*	***	*		
Volatile Suspended Solids		*	*		***	***	***	*		
Particle Count		*	*	***	***	***				*
Total Inorganic Carbon	*	*			*	***			***	***
Total Organic Carbon	*	*	*		***	***		***		
Nutrients										
Total Phosphorous		*	*		***		*	***		***
Orthophosphorus		*	*		***			***	*	***
Nitrate-Nitrite-Nitrogen		*	*	*	***	***		***		
Total Kjeldahl Nitrogen		*	*		***	***	*			***
Phytoplankton Related Parameters										
Chlorophyll a		*	*	***	***	***				*
Chlorophyll b			*	***	*	***	*		***	
Chlorophyll c			*	***	***	***	*			*
Pheophytina	*	*			***	***		*		***

North Creek. Numerous senescent bars are located at the creek mouth and scattered clumps along mangroves. Alterations in current and temperature regimes may have occurred since the closure of Midnight Pass. Low salinities have not destroyed *Melongena* as they were abundant during sampling. Water quality values in North Creek were generally average.

Palma Sola Creek. Scattered clumps of oysters occur along the mangroves in the creek and several small bars lie just outside the creek mouth. Relatively high chlorophyll readings were observed at the station.

Perico Bayou. Oysters were collected from scattered clumps north of Manatee Ave. Few large oysters were collected, although some shells were found on land where they apparently had been shucked and eaten. The shallow and enclosed nature of the area could result in marginal currents and stressful high temperatures. Relatively high chlorophyll values and low dissolved oxygen values were recorded for the station.

Phillippi Creek. Like Bowlees Creek, Phillippi Creek has relatively extensive bars along the main creek channel and a fork which diverges to the south. Currents appear favorable for oysters but development on both sides of the creek have destroyed mangrove swamp/oyster habitat and a boat channel follows the creek center. Also, the ICW and development to the west have eliminated intertidal habitat. Relatively high nutrient values and low D.O. and salinity values were observed.

Siesta Drive. From the Siesta Drive bridge to the south extent of the study area, oyster bars are numerous on both sides of the bays. Oysters were collected from small bars east of the ICW just south of the bridge. The proximity to Big Pass should produce adequate water currents. High D.O. values and low values for suspended particulates and nutrients were observed. Spoil from the ICW provides some substrate for reef growth.

South Creek. Scattered clumps or small bars are found on either side of the creek upstream of the narrow channel to the ICW. Some of the largest oysters of the study were found here in each collection. Development on both side of the creek limit suitable intertidal habitat. Otherwise conditions appear favorable for oysters. Relatively low D.O. values were reported.

Tidy Island. A singular large circular bar is located south of Tidy Island. Mangroves occupy the center of the bar and live oysters are found only on the outside edges. This is typical of senescent reefs where the crests produce few live oysters. *Melongena* were particularly abundant around this bar. Oysters are prevented from colonizing the crest due to excessive exposure and are limited in lateral expansion by predation. Little freshwater flow is available to control predators. This station is similar to City Island and Siesta Drive stations in the occurrence of higher dissolved oxygen and salinities and lower suspended particulates and nutrients.

VI.A.6. Summary - Clam and Oyster Surveys

The distribution of hard clams in the NEP study area was shown, with a few exceptions, to be fairly ubiquitous. One area of maximum clam abundance was along Longboat Key, largely in conditionally approved waters for shellfishing. No substantial seasonal variation was observed between the two surveys.

The extent of recreational clamming was not a portion of this study, but clammers were observed or reported during the survey in four areas, New Pass, Pansy Lagoon, Selby Gardens area, and the north end of Palma Sola Bay. Bay access is relatively easy at these locations, but all sites with the exception of New Pass are in prohibited shellfishing areas. Much of the harvested organisms may be used for bait in finfishing.

Using the arbitrary minimum value of 7 clams per 30 minutes effort as a criteria for an acceptable recreational harvest rate, the survey results indicated that 50 percent of the 169 sampling stations met that criteria during one of the samplings of the survey. The size distribution of the clams, however, shows that less valuable large (chowder) clams are predominant and a low recruitment rate is indicated, possibly related to more common predation (and potentially harvesting pressure) on small clams. The quantitative effect of harvest pressure on these and on *Mercenaria* populations elsewhere is relatively unknown, however.

Any subsequent opening of new areas to shellfishing should be accompanied by monitoring of clam recruitment and consideration given to seeding small clams. Areas for enhancement should be selected on bottom types of larger sediment particle size and, for the protection of seagrasses, in unvegetated areas. The entire length from Anna Maria Island to the southern tip of Longboat Key was very productive of clams (which includes areas open to shellfishing) and is most likely to deserve efforts of enhancement.

The possibility of reopening large areas would involve an unlikely, drastic reduction of contaminant runoff. In particular, the whole area of Roberts Bay, Little Sarasota Bay and Blackburn Bay receives runoff from a large watershed of high human development into a relatively restricted area. Also, the low number of clams collected near Midnight Pass indicates that portions of the area are presently unsuitable habitat. Considering the extremely small commercial harvest at present, the economic benefits of opening waters to clamming are probably small. However, enhancement of recreational harvesting may be a significant improvement to the environment for many residents.

Oysters are also common in the area but their distribution and limited abundance make them less likely to be exploited for commercial harvest and limited for recreational harvest. As a more estuarine organism, oysters were most abundant in the more enclosed bays south of Big Pass. Phillippi, North, Catfish and South Creeks feed into shallow Roberts Bay, Little Sarasota Bay and Blackburn Bay and may be responsible for salinity fluctuations and nutrient input favorable for oyster survival. Consistently low bacteria levels for safe shellfishing may not be a possibility regardless of human management decisions, however, particularly in view of the fact that most local live oysters are intertidal. Intertidal oysters are generally less desirable for harvest as they are exposed more directly to sunlight and the elevated temperatures which result can allow bacterial populations to multiply substantially. Predation appears to limit oyster habitat to the intertidal range. Enhancement of oyster populations is not without merit, though. Oyster bars serve a function in stabilizing shorelines, preventing resuspension of particulates and providing habitat for other invertebrates.

Destruction of oyster bars can not be accurately ascertained since historical bar locations are not known. Two types of destruction which have occurred are dredge and fill operations and channelization. Both alter current patterns and intertidal area. Since mangrove shorelines throughout the study area usually were associated with at least small oyster areas, it appears that shoreline alterations that resulted in seawalls with no natural intertidal beach have limited oysters with correspondingly lower levels of substrate afforded by seawalls. Construction of the ICW from Big Pass south certainly bisected some of the oyster bars common to the area. Likewise, boat channels in Bowlees Creek and Phillippi Creek have affected oysters there.

A simple technique for creating oyster bars is by providing suitable substrate such as shell (cultch) for larval settlement. Sediment type for cultch application is also important and qualitative data from surveys during this project, together with sediment data from the benthic habitat assessments and sediment monitoring conducted under the NEP Program, could be employed in this regard. Cultch placement in the NEP area would require some preliminary experimentation to assure success. The importance of water current, elevation, and seasonality of cultch application (to coincide with oyster spawning) must be determined or verified. The results of small scale experiments would determine the magnitude of oyster bar creation and enhancement possibilities. Areas for testing most likely should be in the southern portion of The NEP areas where oysters are more prevalent. This enhancement should be considered an environmental rather than an economic improvement in view of the limited abundance of oysters in the Sarasota Bay NEP area.

V1.B. Bacteriological Contaminants - Results and Discussion

VI.B.1. Water Column Bacteriological Results

Total coliform bacteria, fecal coliforms, *E. coli* and fecal streptococcus were analyzed in the water column to compare human to non-human inputs of these indicator species (Tables 20 and 21). Total coliform bacteria showed the greatest between station and between season variation and ranged from <3.2 to 1,600 per 100 ml in the spring and from 16 to 32 per 100 ml in the fall. In general, total coliforms were more numerous at stations with lower total aerobic plate counts. This pattern

Table 20. Total and fecal coliforms, *E. coli,* fecal streptococci in spring 1990 water samples (#/100 ml). Blanks indicate values less than detection limits of 3.2 organisms/100 ml.

Station	Total Counts	Coliforms Total	Coliforms Fecal	*E. coli*	Fecal strep
***Mercenaria* Stations**					
Bishops Point	340	1600	32	16	
Blackburn Bridge	420	1200	16	16	
Coquina Ramp	180	120	64	16	
Long Bar Point	190	1410	16	16	
Manatee Ave. Bridge	2100				
New Pass	2300				
Palma Sola Bay	1600				
Phillippi Estuary	1200	64	32	16	
Selby Gardens	620	1100	16	16	
South Casey Key	410	64	32	16	
***Crassostrea* Stations**					
Bowlees Creek	420	1500	64	64	16
City Island	2200	32	32		
Hudson Bayou	1800	175	16	16	
North Creek	120	32	32	16	
Palma Sola Creek	1500				
Perico Bayou	1100				
Phillippi Creek	180	32			
Siesta Drive	160	32			
South Creek	310	64			
Tidy Island	140	1112	16	16	

Table 21. Total and fecal coliforms, *E. coli,* fecal streptococci in fall 1990 water samples (#/100 ml). Blanks indicate values less than detection limits of 3.2 organisms/100 ml.

Station	Total Counts	Coliforms Total	Coliforms Fecal	*E. coli*	Fecal strep
***Mercenaria* Stations**					
Bishops Point	180	16	16		
Blackburn Bridge	1900	32			
Coquina Ramp	3100	32	32		
Long Bar Point	160	16	16		
Manatee Ave. Bridge	3300	32	32		
New Pass	3100	32	32		
Palma Sola Bay	2600	32	32		
Phillippi Estuary	1200	32			
Selby Gardens	2900	32	32		
South Casey Key	1000	32			
***Crassostrea* Stations**					
Bowlees Creek	100	16	16		
City Island	1400	32			
Hudson Bayou	110	16	16		
North Creek	1800	32			
Palma Sola Creek	4100	32	32		
Perico Bayou	210	32	32		
Phillippi Creek	140	16	16		
Siesta Drive	160	16	16		
South Creek	1100	32			
Tidy Island	120	16	16		

was the most obvious in the spring. On the other hand, levels of fecal coliform bacteria in the water column at the stations in Sarasota Bay showed little variation either seasonally or geographically and ranged from <3.2 to 64 per 100 ml in the spring and from <3.2 to 32 per 100 ml in the fall. During the spring, 12 of the 20 stations exceeded the NSSP standard for shellfish harvesting waters of 14 per 100 ml for fecal coliform, while 14 of the 20 stations exceeded this value in the fall. It should be remembered, however, that only the clams from near Bishops Point came from within approved shellfishing areas.

Fecal strep, with the exception of the Bowlees Creek station during the fall which showed 16 per 100 ml, were undetectable in the water column (<3.2 per 100 ml) in both the fall and spring samples. *E. coli* samples varied from <3.2 to 16 in the fall and from <3.2 to 64 per 100 ml in the spring.

Analyses were performed for seven species of *Vibrio* during this study. Results from the water samples taken from Sarasota Bay during the fall and spring appear in Tables 22 and 23. The highest number of a vibrio species isolated from a sample was *V. vulnificus (640* per 100 ml) and if *V. vulnificus* was detected, it was typically more numerous than other co-occurring species. Six stations of the twenty stations had measurable numbers of *V. vulnificus* present during the spring.

The species of *Vibrio* found in the largest number of spring water samples were *V. parahaemolyticus* followed by *V. alginolyticus* and V. *vulnificus. V. parahaemolyticus* and *V. alginolyticus,* however, when present, were much less numerous than *V. vulnificus,* ranging from <3.2 to only 64 per 100 ml. *V. cholerae* and *V. metschnikovii* were also detected during this sampling but at relatively few stations. During the fall, *V. alginolyticus* and *V. parahaemolyticus* were the only vibrios detected in the water column, and concentrations were again low, ranging from <3.2 to 32 per 100 ml. *V. damsela* and *V. metschnikovii* were not found in any of the water samples during this study.

In addition to the vibrios, analyses for two species of *Aeromonas, A. hydrophila* and *A. sobria,* were performed, with *A. hydrophila* being the only species isolated from the water column and in numbers ranging from <3.2 to 285 per 100 ml. *No Aeromonas* of either species were observed at 50 percent of all clam and oyster stations during the spring, while all stations displayed equivalent levels (16 per 100 ml) of *A. hydrophila* during the fall. There were no geographic patterns noted in the distribution of any of either the *Aeromonas* or *Vibrio* species during either sampling, although during the fall, the waters from more of the clam stations reported the presence of *V. vulnificus* and *A. hydrophila* than did the waters at the oyster stations. This may be related to the differing salinity structure of the clam and oyster stations, with the clam stations of overall higher salinity.

The low counts returned for water column samples made difficult an re-evaluation of the fecal coliform standard for shellfishing waters with respect to vibrio counts. The ubiquitous nature of some of the vibrio

Table 22. *Vibrio* and *Aeromonas* species in spring 1990 water samples (#/100 ml). Blanks indicate values less than detection limits of 3.2 organisms/100 ml.

Station	Va	Vc	Vd	Vm	Vn	Vp	Vv	Ah	As
***Mercenaria* Stations**									
Bishops Point						16	640	32	
Blackburn Bridge						16		16	
Coquina Ramp						16	182	64	
Long Bar Point						16	175	64	
Manatee Ave. Bridge		32				32		32	
New Pass		32				64			
Palma Sola Bay								64	
Phillippi Estuary	32	32				32	32		
Selby Gardens						32		16	
South Casey Key	16					32			
***Crassostrea* Stations**									
Bowlees Creek						16	584	285	
City Island		32				64			
Hudson Bayou						16		16	
North Creek	16					16			
Palma Sola Creek									
Perico Bayou									
Phillippi Creek	32			16		64			
Siesta Drive	16					32			
South Creek	32			16		32			
Tidy Island						16	112	132	

Va *Vibrio alginolyticus*
Vc *V. cholerae*
Vd *V. damsela*
Vm *V. mimicus*
Vn *V. metchnikovii*
Vp *V. parahaemolyticus*
Vv *V. vulnificus*
Ah *Aeromonas hydrophila*
As *A. sobria*

Table 23. *Vibrio* and *Aeromonas* species in fall 1990 water samples (#/100 ml). Blanks indicate values less than detection limits of 3.2 organisms/100 ml.

Station	Va	Vc	Vd	Vm	Vn	Vp	Vv	Ah	As
***Mercenaria* Stations**									
Bishops Point	16					16		16	
Blackburn Bridge	32					16		16	
Coquina Ramp	16					16		16	
Long Bar Point	16					16		16	
Manatee Ave. Bridge	16					16		16	
New Pass	16					16		16	
Palma Sola Bay	16					16		16	
Phillippi Estuary	32							16	
Selby Gardens	16					16		16	
South Casey Key	32					16		16	
***Crassostrea* Stations**									
Bowlees Creek	16					16		16	
City Island	32							16	
Hudson Bayou	16					16		16	
North Creek	32					16		16	
Palma Sola Creek	32					16		16	
Perico Bayou	16					16		16	
Phillippi Creek	16					16		16	
Siesta Drive	16					16		16	
South Creek	32					16		16	
Tidy Island	16					16		16	

Va *Vibrio alginolyticus*
Vc *V. cholerae*
Vd *V. damsela*
Vm *V. mimicus*
Vn *V. metchnikovii*
Vp *V. parahaemolyticus*
Vv *V. vulnificus*
Ah *Aeromonas hydrophila*
As *A. sobria*

species, however, does support previous findings that its occurrence is not linked to fecal pollution. Although few samples contained *V. vulnificus,* those positive for the species were not associated with any tributary where fecal contamination would be most expected. Aeromonas found, however, did appear to have a positive correlation with both *E. coli* and total coliforms counts in the water column, which is likely a result of the preference of both for fresh and brackish waters (West and Coldwell, 1984).

VI.B.2. Oyster Tissue Bacteriological Results

For the oyster tissues, coliform bacteria remained low during both the spring and fall (Tables 24 and 25). Total coliform ranged between <3.2 to 1067 per 100 g in the spring to <3.2 to 32 per 100 g in the fall. For fecal coliform bacteria, counts ranged from <3.2 to 100 per 100 g of meat. These higher counts occurred during the fall of 1990. No oyster tissue exceeded the 230 per 100 g tissue NSSP standard for fecal coliform. For the fall samples, half of the oyster tissues reported fecal coliform counts higher than the total coliform. This phenomena was not apparent in the water samples, thus eliminating methodological problems, and is most likely the result of the MPN (most probable number) technique and the extremely low levels of bacterial counts.

E. coli counts were also low, ranged from <3.2 to 24 per 100 g tissue and were detected in more tissue samples during the spring sampling. Fecal streptococci was always less than 3.2 per 100 g of oyster meat. Total plate counts ranged between 100 and 51,000 per 100 g, but never exceeded the 50,000,OOO per 100 g NSSP standard. The within-station variability was particularly evident during the spring sampling when counts were high, and samples from the same station could vary by as much as two orders of magnitude.

Of the seven species of *Vibrio,* only four species were detectable in any sample of oyster tissue. All of these counts remained low ranging from <3.2 to 70 per 100 g of oyster meat (Tables 26 and 27). Of these, *V. vulnificus* showed the highest counts, which only occurred during the spring 1990 samples, and was present in four of the 20 oyster samples for this season. *V. alginolyticus, closely* followed by *V. parahaemolyticus* were detected the most frequently during the spring and occurred in roughly 60 percent of the spring oyster samples. *V. cholerae* were not found in oysters from two stations with a measurable water column presence of these organisms, nor were any *V. mimicus* detected in oyster tissue. *Aeromonas* concentrations were low during the spring and *A. hydrophila* detected in relatively few samples. Fall occurrences of both vibrios and aeromonas closely parallelled water column distributions with low and comparable levels of *V. alginolyticus, V. parahaemolyticus,* and *A. hydrophila* present in almost all oyster tissue samples. Similar to the results for the water column samples, no *A. sobria* were detected in tissues in either season.

Table 24. Total and fecal coliforms, E. coli, fecal streptococci in spring 1990 oyster meats (#/100 g wet tissue). Blanks indicate values less than detection limits of 3.2 organisms/100 g.

Station	Total Counts	Coliforms Total	Coliforms Fecal	*E. coli*	Fecal strep
Bowlees Creek	410	210	16	16	
	360	219	16	16	
City Island	2400				
	5200	64	32	16	
Hudson Bayou	210	1027	16	16	
	180	1067	16	16	
North Creek	130	32			
	3100	116	64	16	
Palma Sola Creek	7100	32	32		
	2200				
Perico Bay	3100				
	6100	64	32		
Phillippi Creek	410				
	51000	132	64	24	
Siesta Drive	240				
	3200	132	64	24	
South Creek	110	16			
	6100	132	64	24	
Tidy Island	330	169	16	16	
	390	179	16	16	

Table 25. Total and fecal coliforms, *E.* coli, fecal streptococci in fall 1990 oyster meats (#/100 g wet tissue). Blanks indicate values less than detection limits of 3.2 organisms/100 g.

Station	Total Counts	Coliforms Total	Coliforms Fecal	*E. coli*	Fecal strep
Bowlees Creek	100		100		
	100		100		
City Island	100	32			
	100	32			
Hudson Bayou	100		100		
	100		100		
North Creek	100	32			
	100	32			
Palma Sola Creek	1600	32	32	16	
	18000	32	32	16	
Perico Bayou	2100	32	32	16	
	11000	32	32	16	
Phillippi Creek	100		100		
	100		100		
Siesta Drive	100		100		
	100		100		
South Creek	100	32			
	100	32			
Tidy Island	100		100		
	100		100		

Table 26. *Vibrio* and *Aeromonas* species in spring 1990 oyster meats (#/100 g wet tissue). Blanks indicate values less than detection limits of 3.2 organisms/100 g.

Station	Va	Vc	Vd	Vm	Vn	Vp	Vv	Ah	As
Bowlees Creek						16			
						32			
City Island	32					32		32	
	32	32				64		64	
Hudson Bayou						16			
						16			
North Creek	64					16			
	64						44		
Palma Sola Creek	32								
	32	32							
Perico Bayou	32								
	32	32						32	
Phillippi Creek						32			
	64						70		
Siesta Drive						32			
	16						16		
South Creek						32			
	64						70		
Tidy Island	64					16			
	32					16			

Va *Vibrio alginolyticus*
Vc *V. cholerae*
Vd *V. damsela*
Vm *V. mimicus*
Vn *V. metchnikovii*
Vp *V. parahaemolyticus*
Vv *V. vulnificus*
Ah *Aeromonas hydrophila*
As *A. sobria*

Table 27. *Vibrio* and *Aeromonas* species in fall 1990 oyster meats (#/100 g wet tissue). Blanks indicate values less than detection limits of 3.2 organisms/100 g.

Station	Va	Vc	Vd	Vm	Vn	Vp	Vv	Ah	As
Bowlees Creek	16					16		16	
	16					16		16	
City Island	32							16	
	32							16	
Hudson Bayou	16					16		16	
	16					16		16	
North Creek	32							16	
	32							16	
Palma Sola Creek	16					16		16	
	16					16		16	
Perico Bayou	16					16		16	
	16					16		16	
Phillippi Creek	16					16		16	
	16					16		16	
Siesta Drive	16					16		16	
	16					16		16	
South Creek	32							16	
	32							16	
Tidy Island	16					16		16	
	16					16		16	

Va *Vibrio alginolyticus*
Vc *V. cholerae*
Vd *V. damsela*
Vm *V. mimicus*
Vn *V. metchnikovii*
Vp *V. parahaemolyticus*
Vv *V. vulnificus*
Ah *Aeromonas hydrophila*
As *A. sobria*

VI.B.3. Clam Tissue Bacteriological Results

For clam tissues, the intolerance of *Mercenaria campechiensis,* the presumed study species, to refrigeration and transport is highlighted by the fact that five of the 20 samples died (gaped) between collection and transport during the fall sampling. These were not the clams that had been harvested the day prior to collection, enclosed in mesh bags, and left in ambient waters until collection. As no samples died during the initial spring sampling, and as transport times and techniques were identical between samplings, this is attributed to the physiological intolerance of the organism rather than methodological flaws and may be more evident when ambient temperatures are substantially different from transport temperatures. This is reported to be a frequent occurrence for similar investigations (Dr. G. Rodrick, personal communication).

Fecal coliform and fecal streptococci bacteria in clam meats also remained low in both spring and fall samples (Tables 28 and 29). Counts for fecal coliform bacteria in clam meats ranged from less than 3.2 to 100 per 100 g of meat, well below the 230 per 100 g NSSP standard. Total coliform bacteria counts in the clam samples were more erratic and during the spring were as high as 2,421 per 100 g of clam meat at some stations. As for the oyster tissues, one samples reported a fecal coliform count higher than the total coliform, and is thought to be the product of the MPN procedure and the very low counts found. A general pattern of higher total coliforms was associated with lower total plate counts. Total plate counts ranged between 100 and 36,000 per 100 g clam tissue, in no instance exceeded the 50,000,000 per 100 g NSSP standard, and during the fall were substantially higher than in oysters taken during the same season. Within-station variation was again more pronounced during the spring sampling for these organisms.

Only two species of vibrio, *V. parahaemolyticus* and *V. alginolyticus,* were detected in the clam meats from the spring and fall samples (Tables 30 and 31). *No V. vulnificus* was found during either season, despite detectable water column counts in the spring at four of the clam stations. Vibrio counts ranged from <3.2 to 64 per 100 g of clam meat. Only one species of aeromonas was detected, *A. hydrophila,* and this ranged from <3.2 to 32 per 100 g of clam meat. Fall counts again parallelled oyster and water column distributions, in that *V. alginolyticus V. parahaemolyticus,* and *A. hydrophila* were uniformly low and present at almost all stations. Unlike oyster or water column samples, no *V. cholerae* or *V. mimicus* were found.

VI.B.4. Summary - Bacteriological Contaminants

During the two sampling periods in 1990, all bacterial counts in both tissue and water column samples remained exceptionally low in Sarasota Bay. This may have resulted from the relatively low rainfall that occurred during this sampling year. Additionally, as sampling was delayed in anticipation of increased rainfall amounts, the study was not

Table 28. Total and fecal coliforms, *E. coli,* fecal streptococci in spring 1990 clam meats (#/100 g wet tissue). Blanks indicate values less than detection limits of 3.2 organisms/100 g.

Station	Total Counts	Coliforms Total	Coliforms Fecal	*E. coli*	Fecal strep
Bishops Point	270	1617	16	16	
	310	1817	16	16	
Blackburn Bridge	190	2421	16	16	
	410	2218	16	16	
Coquina Ramp	170	174	32	16	
	210	1121	16	16	
bong Bar Point	140	210	32	16	
	280	1488	16	16	
Manatee Ave Bridge	1700				
	3400	118	64		
New Pass	1800				
	4100	64	32	16	
Palma Sola Bay	2100				
	6400	64	32		
Phillippi Estuary	2100				
Selby Gardens	190	1146	16	16	
	180	164	64	16	
South Casey Key	140	16			
	36000	132	64	16	

Table 29. Total and fecal coliforms, *E. coli*, fecal streptococci in fall 1990 clam meats (#/100 g wet tissue). Blanks indicate values less than detection limits of 3.2 organisms/100 g.

Station	Total Counts	Coliforms Total	Coliforms Fecal	*E. coli*	Fecal strep
Bishops Point	Died	---	---	---	---
	Died	---	---	---	---
Blackburn Bridge	1000	32			
	1000	32			
Coquina Ramp	1800	32	16		
	4100	32	16		
Long Bar Point	100		100		
	Died	---	---	---	---
Manatee Ave Bridge	28000	32	32	16	
	Died	---	---	---	---
New Pass	1400	32	16		
	1700	32	16		
Palma Sola Bay	26000	32	32	16	
	21000	32	32	16	
Phillippi Estuary	Died	---	---	---	---
	1000	32			
Selby Gardens	4100	32	16		
	2800	32	16		
South Casey Key	1000	32			
	1000	32			

-- No data

Table 30. *Vibrio* and *Aeromonas* species in spring 1990 clam meats (#/100 g wet tissue). Blanks indicate values less than detection limits of 3.2 organisms/100 g.

Station	Va	Vc	Vd	Vm	Vn	Vp	Vv	Ah	As
Bishops Point						16			
						16			
Blackburn Bridge						16			
						16			
Coquina Ramp						32			
						32			
Long Bar Point						32			
						16			
Manatee Ave Brdg.	64								
	32					64		32	
New Pass	64								
	32							32	
Palma Sola Bay	32								
	32					64		32	
Phillippi Estuary	32					32		32	
Selby Gardens						32			
						16			
South Casey Key						40			
	64					32			

Va *Vibrio alginolyticus*
Vc *V. cholerae*
Vd *V. damsela*
Vm *V. mimicus*
Vn *V. metchnikovii*
Vp *V. parahaemolyticus*
vv *V. vulnificus*
Ah *Aeromonas hydrophila*
As *A. sobria*

Table 31. *Vibrio* and *Aeromonas* species in fall 1990 clam meats (#/100 g wet tissue). Blanks indicate values less than detection limits of 3.2 organisms/100 g.

Station	Va	Vc	Vd	Vm	Vn	Vp	Vv	Ah	As
Bishops Point	Died	--	--	--	--	--	--		
	Died	--	--	--	--	--	--	--	--
Blackburn Bridge	32							16	
	32							16	
Coquina Ramp	16					16		16	
	16					16		16	
Long Bar Point	16					16		16	
	Died	--	--	--	--	--	--	--	--
Manatee Ave Bridge	16					16		16	
	16					16		16	
New Pass	16					16		16	
	16					16		16	
Palma Sola Bay	16					16		16	
	16					16		16	
Phillippi Estuary	Died	--	--	--	--	--	--	--	
	32							16	--
Selby Gardens	16					16		16	
	16					16		16	
South Casey Key	32							16	
	32							16	

Va *Vibrio alginolyticus*
Vc *V. cholerae*
Vd *V. damsela*
Vm *V. mimicus*
Vn *V. metchnikovii*
Vp *V. parahaemolyticus*
Vv *V. vulnificus*
Ah *Aeromonas hydrophila*
As *A. sobria*

-- No data

conducted during the warmest portion of the year, when ambient bacterial counts are expected to be at a maximum.

While fecal coliform standards for shellfishing waters were exceeded at many stations, only one of the stations, Bishops Point, was within a conditionally approved area. Counts at this station were 32 and 16 per 100 ml during spring and fall respectively. Many of the stations from within prohibited or unclassified waters recorded water column counts below the 14 per 100 ml standard during one sampling or the other, but only the waters at South Creek were below this standard during both sampling events.

Bacterial counts at sampling times did not indicate highly polluted conditions or for that matter the presence of high numbers of vibrios unrelated to pollution. The low levels of vibrios are also a likely result of sampling during the spring and fall rather than during the summer months. While specific dose-response information is generally lacking for vibrio infections, the vibrio counts determined during this study are approximately four orders of magnitude less than either total vibrios or *V. vulnificus* alone as documented at a Gulf Coast oyster processing plant (Ruple et al., 1989).

The erratic counts of total coliform bacteria may indicate that some non-human inputs may be significant at some locations, but further analyses (identification of actual fecal coliform species) would have been required to determine the source and nature of these non-human inputs. As no fecal streptococci were found above a background level of 100 per 100 ml (APHA, 1985a), fecal coliform to fecal strep ratios could not be employed to gain information on sources of fecal matter, even providing the contamination source was less than 24 hours distant and that the ratios could be tentatively employed in the marine environment. Overall, the most prevalent bacteria identified were *V. alginolyticus, V. parahaemolyticus* and *A. hydrophila,* with *V. vulnificus* occurring only in the spring and at selected stations in the water column and oyster tissue samples. Results suggest that the major groups of vibrios and aeromonas are a part of the normal ecosystem and not of human fecal origin.

It should be pointed out that sampling during wetter conditions and warmer months would provide a clearer picture of the bacteriology of Sarasota Bay, in addition to obtaining a better estimate of "worst case" conditions. The study was originally designed to capture such an event, but the low rainfall amounts received during the sampling year and subsequent study redesign forced the sacrifice of this information.

V1.C. Toxic Metals - Results and Discussion

Summaries, as station means, of tissue concentrations for the six heavy metals are presented in Table 32 on a dry weight basis and all further data reductions were carried out on dry weight data, unless noted otherwise. Complete data sets appear in Appendix C. Percent solids

Table 32. Tissue metal concentrations µg/g dry weight). Station means and standard deviations.

Clam Tissue - *Mercenaria campechiensis*

	Arsenic		Cadmium		Copper		Lead		Mercury		Zinc	
	Mean	s.d.	Mean	s.d	Mean	s.d	Mean	s.d	Mean	s.d.	Mean	s.d.
Bishops Point	27.0	24.5	1.7	0.3	14.4	1.6	4.3	1.4	0.14	0.02	142	46
Blackburn Bridge	12.1	7.8	0.5	0.1	20.8	6.1	5.9	1.8	0.10	0.01	191	54
Coquina Ramp	16.5	10.2	1.2	0.2	18.1	4.4	4.7	3.9	0.12	0.04	122	64
Long Bar Point	22.2	6.2	1.4	0.3	9.4	2.5	2.8	0.8	0.14	0.01	91	20
Manatee Ave Bridge	1.7	2.6	1.0	0.2	15.4	3.1	3.9	1.7	0.12	0.02	113	30
New Pass	16.7	6.1	1.3	0.5	17.8	4.7	2.0	0.7	0.14	0.05	91	12
Phillippi Estuary	19.5	10.6	1.5	1.7	20.4	4.0	10.8	5.1	0.21	0.04	246	55
Palma Sola Bay	23.8	31.6	0.8	0.6	11.7	1.9	4.4	2.1	0.13	0.02	94	12
South Casey Key	23.0	9.3	0.9	0.3	12.9	4.7	4.5	1.6	0.12	0.05	170	62
Selby Gardens	4.0	0.9	3.0	3.2	22.9	6.6	8.7	1.9	0.13	0.02	102	27
Mean	16.6		1.3		16.4		5.2		0.13		136	

Oyster Tissue - *Crassostrea virginica*

	Arsenic		Cadmium		Copper		Lead		Mercury		Zinc	
	Mean	s.d.	Mean	s.d	Mean	s.d	Mean	s.d	Mean	s.d.	Mean	s.d.
Bowlees Creek	8.8	9.6	0.7	0.1	153.7	12.1	3.3	2.6	0.12	0.03	2939	250
City Island	26.3	21.9	1.3	1.0	66.1	13.4	1.3	0.8	0.13	0.01	2455	651
Hudson Bayou	11.9	7.1	0.8	0.2	556.1	40.7	6.9	0.4	0.11	0.03	4752	1319
North Creek	8.8	6.5	1.5	0.4	143.4	19.7	1.6	0.6	0.16	0.01	2826	177
Perico Bayou	3.6	3.9	0.9	0.9	57.2	15.1	0.5	0.2	0.17	0.03	1326	221
Phillippi Creek	11.0	3.4	1.8	0.6	300.8	82.2	2.1	0.9	0.17	0.05	5254	1093
Palma Sola Creek	9.6	7.5	0.6	0.2	59.8	8.4	1.1	0.3	0.14	0.01	1385	322
Siesta Drive	30.7	12.8	0.8	0.3	105.5	9.8	1.5	0.5	0.15	0.01	2716	256
South Creek	34.8	26.3	1.0	0.4	179.2	36.2	1.0	0.7	0.21	0.04	3969	281
Tidy Island	18.1	15.8	1.3	0.7	31.6	13.0	0.9	0.4	0.15	0.04	802	93
Mean	16.4		1.1		165.3		2.0		0.15		2842	

values of the wet tissue used for the wet to dry weight conversion and average size data per station (as length) appear in Table 33.

VI.C.1. Seasonal Variations

For metals concentrations overall, there was no statistically significant seasonal variation apparent for either hard clams or oysters. For clams this is consistent with other work in the northeast on *Mercenaria mercenaria* (Pratt and Hale, as reported in Bender, 1989). The April and October samplings were generally grouped closely with one another in any ranking of data, particularly for the metals encompassing large ranges. Selected stations evidenced substantive changes between samplings but these were distributed between increases and decreases with time, were not consistent within species, and so were attributed to intersample variation or varying loadings to an area rather than any physiological processes.

The relative lack of seasonal variation may, to some extent, be a product of the extremely dry conditions experienced during the study year. It is conceivable that these results may differ substantively during years of normal rainfall when loadings are predicted to be higher.

VI.C.2. Species Differences

Of the two species analyzed during this study, oysters concentrated significantly higher concentrations of some metals within their tissues. Copper and zinc in particular reached a factor of 20 or more higher in oyster tissues than in the maximum clam tissues concentrations. Higher bioconcentration factors for these metals were also observed in oysters when compared to clam tissues taken from nearby and under presumably similar influences of contaminant loadings. These inter-specific differences are probably the result of physiological differences in metal detoxification strategies of the two species.

VI.C.3. Geographic Variations

Significant geographic variations (Freidman's ANOVA) were present for all metals and each species with the exception of mercury in both clams and oysters. Oysters typically displayed stronger significance levels for a given metal than did clam tissues. Cadmium and arsenic exhibited the lowest significance of variation by station while still significant at the 0.05 level.

Since the ability of shellfish to bioaccumulate mercury has been extensively documented, it can be inferred from these results that mercury is not associated with any major point sources or loadings from the basins represented by stations within this study. If mercury is not being transported to Sarasota Bay by basin tributaries, then this would indicate aerial deposition to the Bay as a substantive transport mechanism.

Table 33. Mean percent solids and organism lengths by station for the spring and fall 1990 tissue collections.

Station	Spring Survey Percent Solids (%)	Spring Survey Length (mm)	Fall Survey Percent Solids (%)	Fall Survey Length (mm)
Mercenaria spp.				
Bishops Point	19.6	104.5	18.6	104.7
Blackburn Bridge	18.5	81.0	15.9	82.9
Coquina Ramp	19.9	104.2	18.3	98.2
Long Bar Point	21.2	104.5	19.9	102.7
Manatee Ave Bridge	20.0	103.2	20.3	109.6
New Pass	18.3	79.7	18.7	78.3
Phillippi Estuary	15.4	83.9	18.0	100.3
Palma Sola Bay	20.7	101.6	19.1	113.7
South Casey Key	16.9	89.0	18.7	81.1
Selby Gardens	19.4	94.3	20.0	89.3
C. virginica				
Bowlees Creek	20.4	55.1	17.4	54.7
City Island	22.0	54.2	18.0	58.4
Hudson Bayou	20.9	59.0	14.8	57.1
North Creek	16.0	49.2	17.8	49.2
Perico Bayou	20.9	52.3	19.4	57.9
Phillippi Creek	16.8	42.3	12.1	42.3
Palma Sola Creek	27.3	44.9	18.0	45.8
Siesta Bridge	18.2	43.0	16.5	43.0
South Creek	15.0	67.0	18.3	67.0
Tidy Island	23.3	53.5	16.3	54.7

For the remaining metals with significant geographic variation, selected stations or regions were consistently high during both of the samplings, and to some extent for both of the species sampled. Table 34 expresses tissue metal content as a percentage of the maximum concentration observed during this study for any station and for a particular species. The percentages for all metals are then summed by station and stations presented in rank order. This overall ranking equally weights all metals, making no distinction between relative human or molluscan toxicities.

Of the metals sampled, and for the ranges of oyster tissue concentrations observed, Hudson Bayou exhibited the highest overall concentrations of metals (expressed as a percentage) than any of the other stations. Metals that averaged the highest at this station included both copper and lead. Oysters from Phillippi Creek and South Creek recorded the highest concentrations of cadmium and zinc and arsenic and mercury, respectively. In comparison, oysters from Palma Sola Creek and Perico Bayou were low in overall metal concentration.

For clam tissues, those gathered from the` Phillippi Creek estuary were highest in lead, mercury and zinc, while those from near Selby Gardens were highest in cadmium and copper. Arsenic concentrations in clams were highest in tissues collected off of Bishops Point. Clams from the northeast side of the Manatee Avenue Bridge were the lowest overall in metals percentages.

Figures 19 through 30 illustrate these geographic variations for both oysters and clams.

VI.C.4. Comparison with Predicted Loadings

Differences in tissue concentrations can be the product of many variables. The factors contributing to bioconcentration have been discussed in preceding sections but include not only water and sediment concentrations of contaminants, but also sedimentary characteristics and chemistries, salinity, temperature, physiological differences between species, and size, sex, and reproductive condition differences among individuals of the same species.

As the focus of this study was to determine contamination status as it may affect the recreational shellfish harvester rather than to elucidate the processes of bioconcentration, this multitude of factors was not addressed in any quantitative fashion. Sediment characteristics and contaminant loads are being determined under a separate study and are not yet available, while samples of tissues in this study were composed of a number of individuals from a location in order to obtain a population estimate across a range of sizes and including both sexes. As a result, there can be no definitive or quantitative link between tissue concentrations and these variables and watershed loadings. There is, however, the strong probability that sediments and tissue concentrations,

Clam Tissue - Arsenic

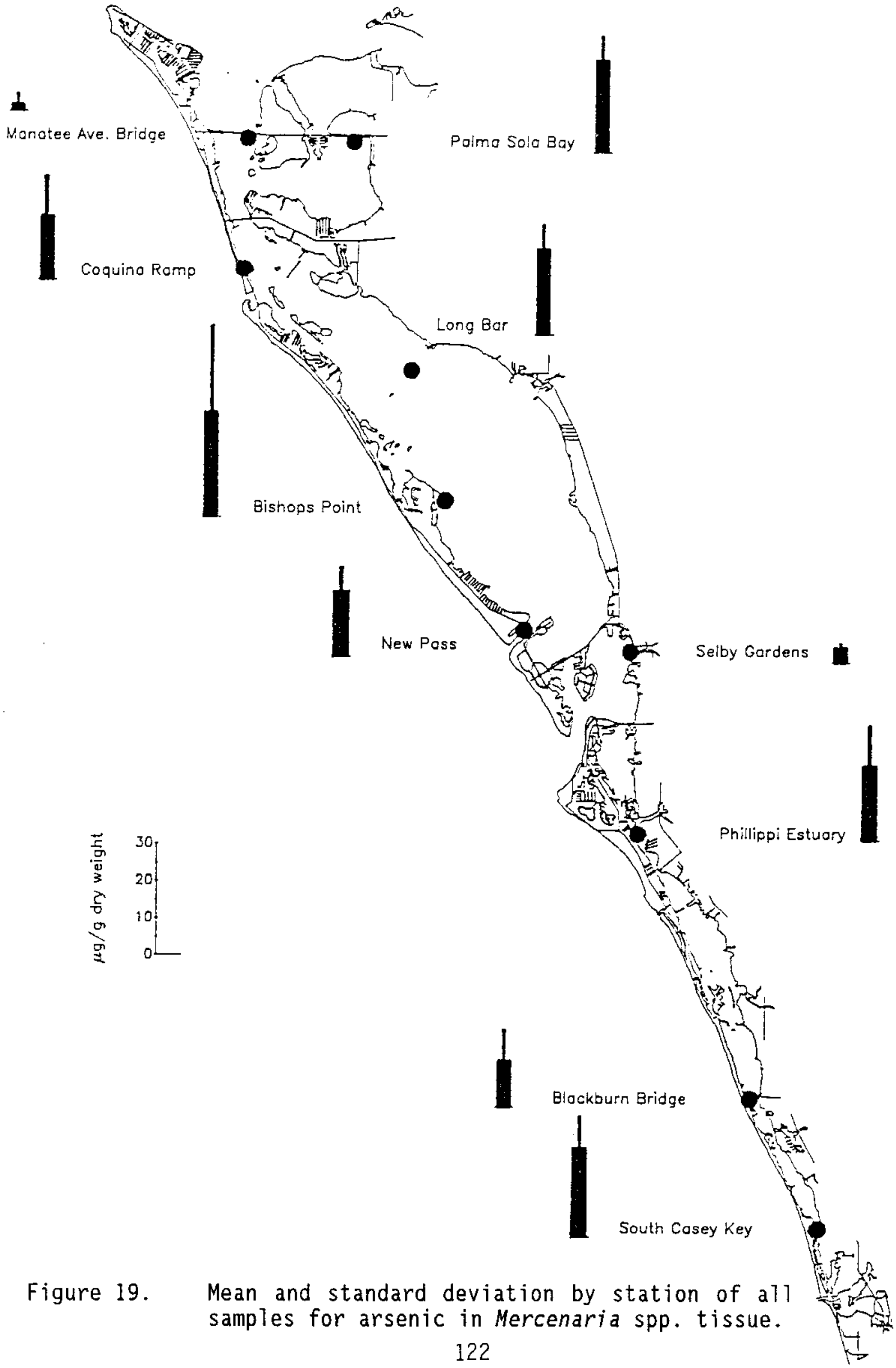

Figure 19. Mean and standard deviation by station of all samples for arsenic in *Mercenaria* spp. tissue.

Clam Tissue - Cadmium

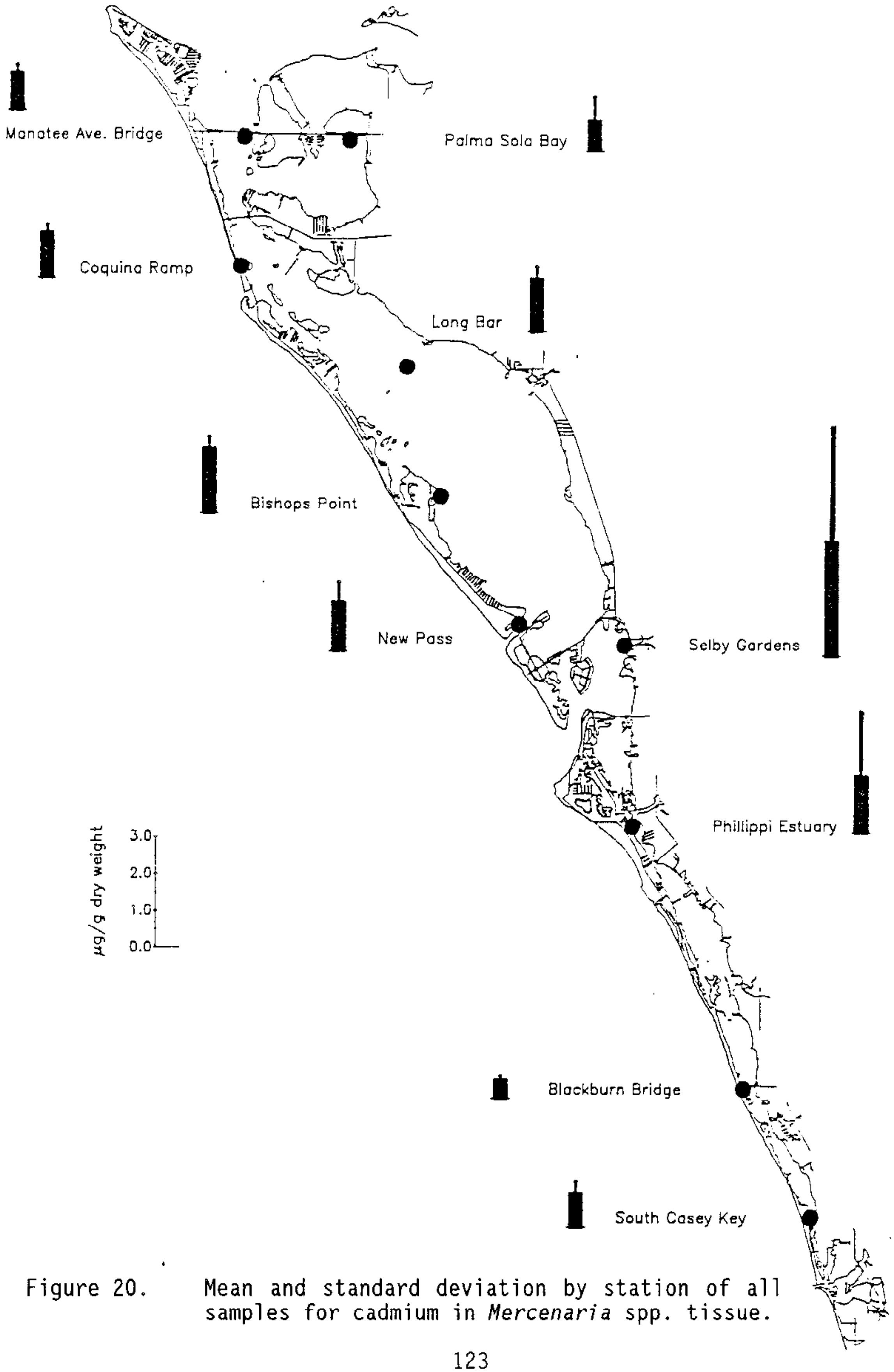

Figure 20. Mean and standard deviation by station of all samples for cadmium in *Mercenaria* spp. tissue.

Clam Tissue - Copper

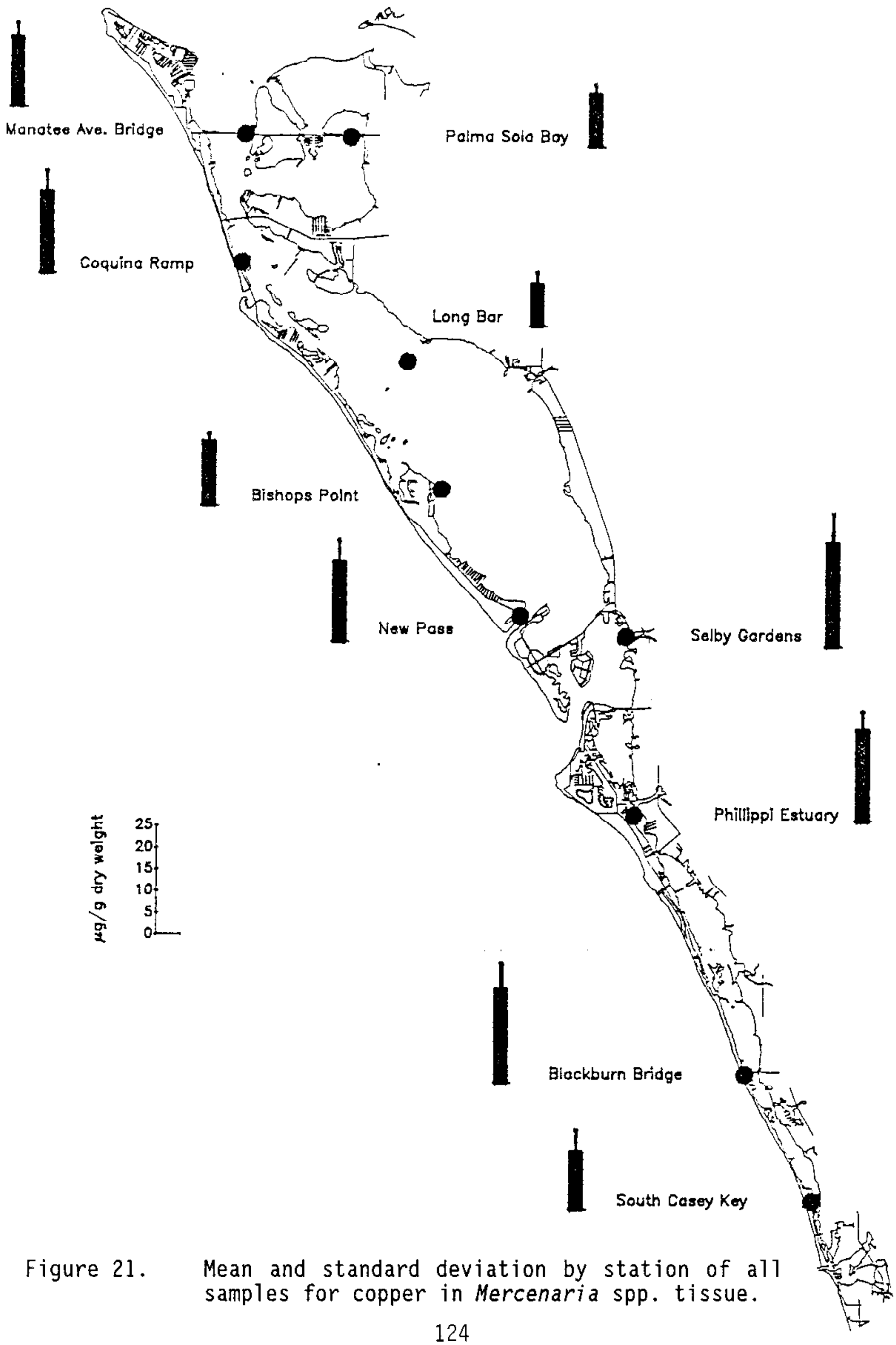

Figure 21. Mean and standard deviation by station of all samples for copper in *Mercenaria* spp. tissue.

Clam Tissue - Lead

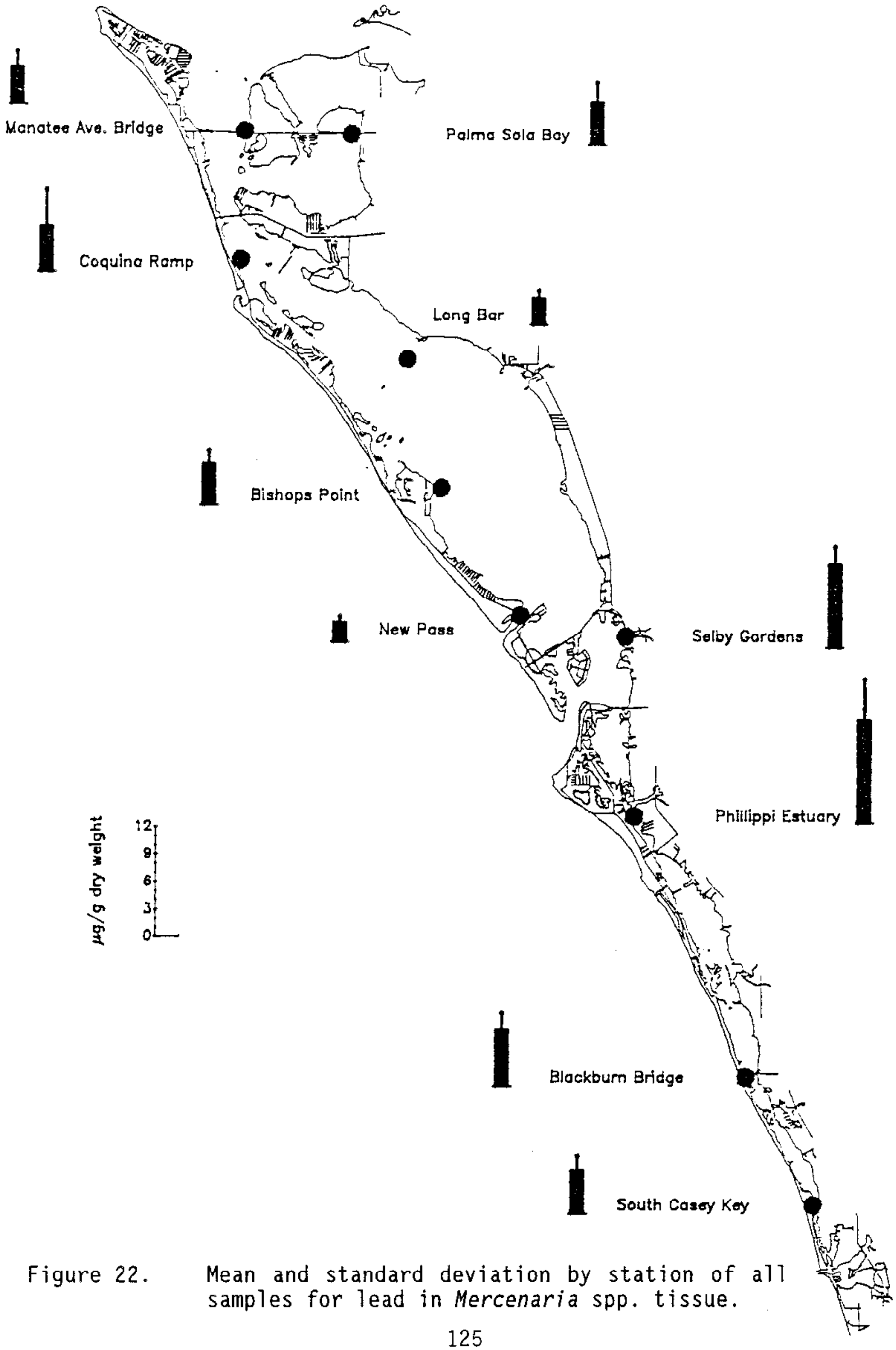

Figure 22. Mean and standard deviation by station of all samples for lead in *Mercenaria* spp. tissue.

Clam Tissue - Mercury

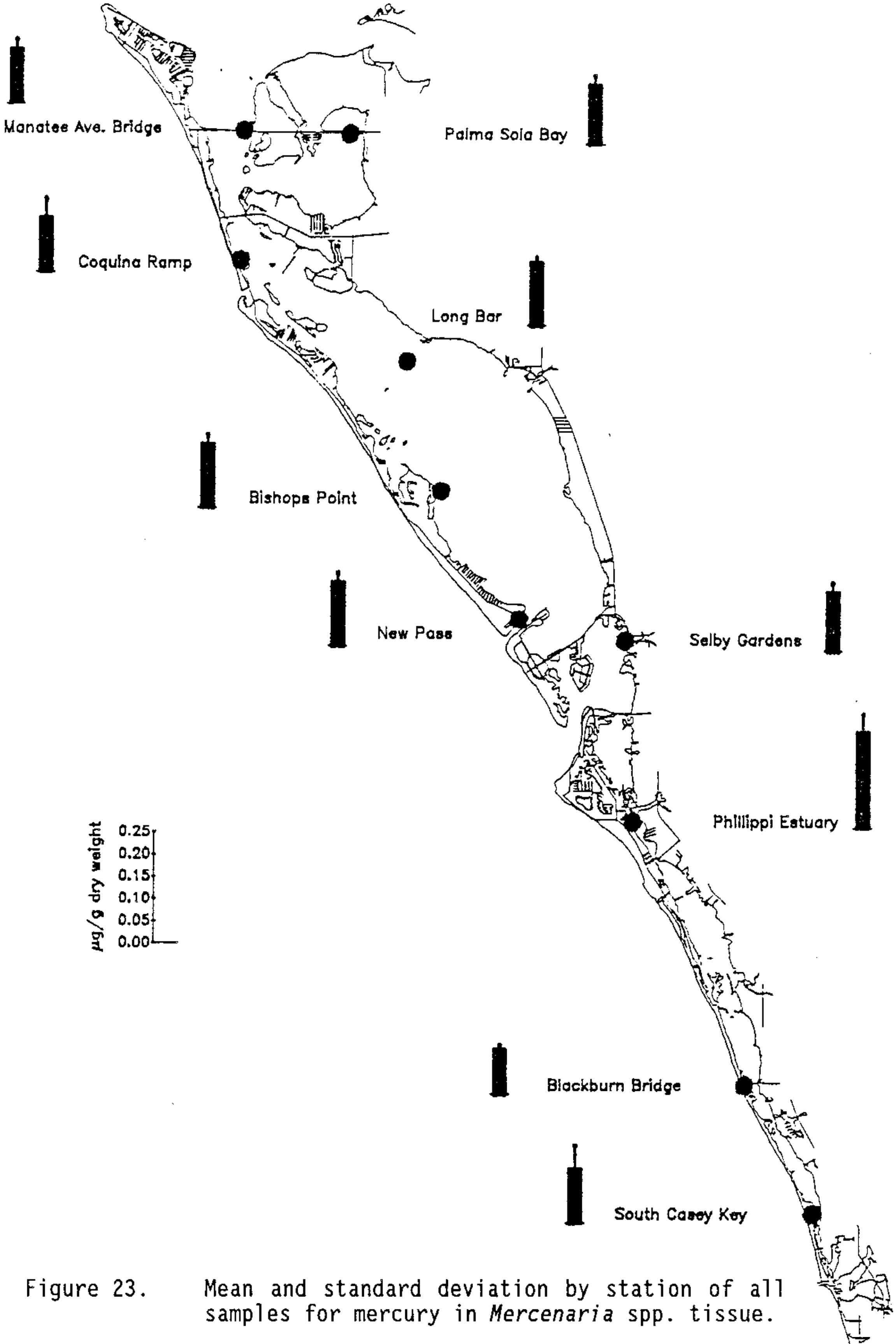

Figure 23. Mean and standard deviation by station of all samples for mercury in *Mercenaria* spp. tissue.

Clam Tissue - Zinc

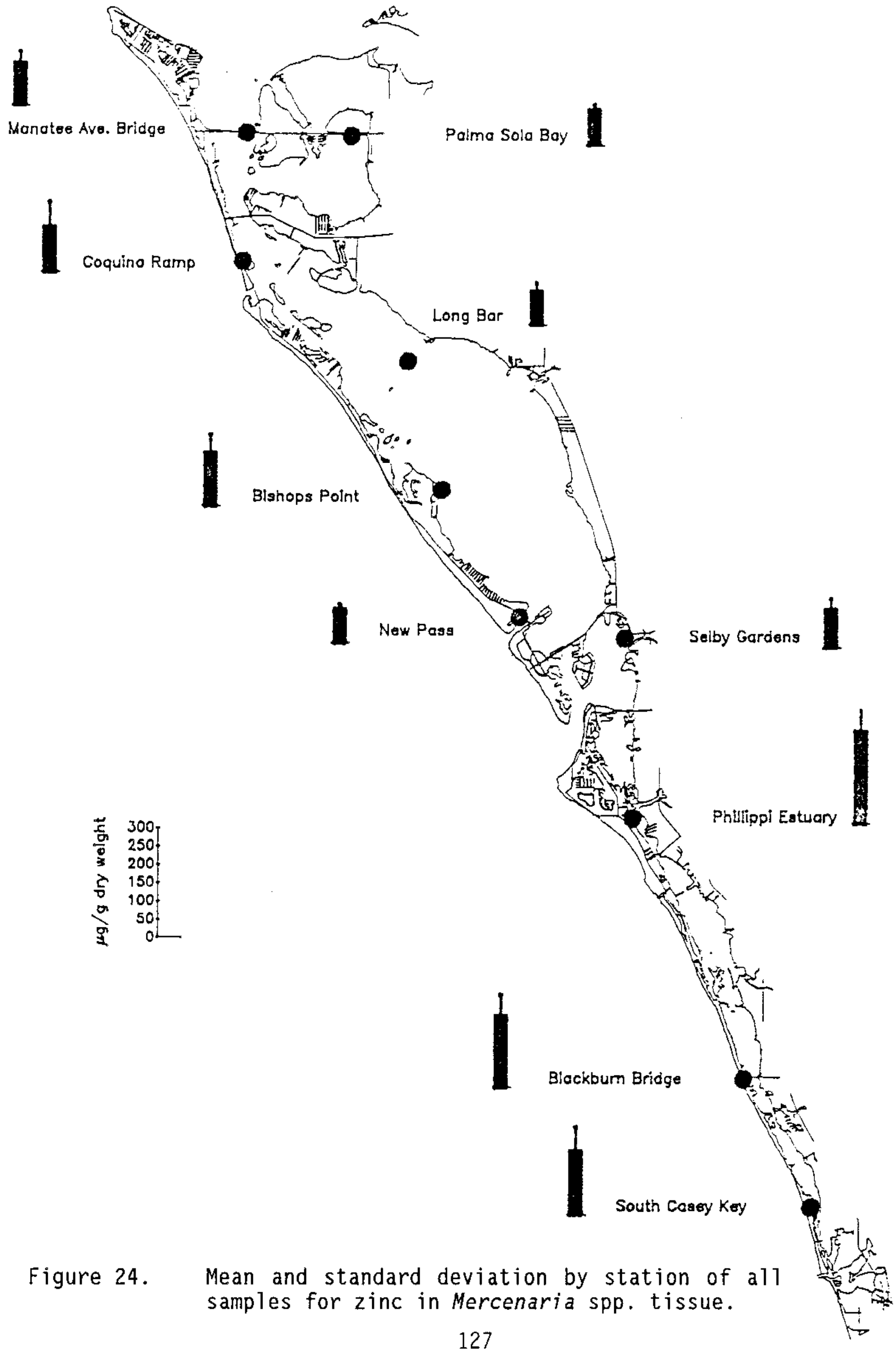

Figure 24. Mean and standard deviation by station of all samples for zinc in *Mercenaria* spp. tissue.

Oyster Tissue - Arsenic

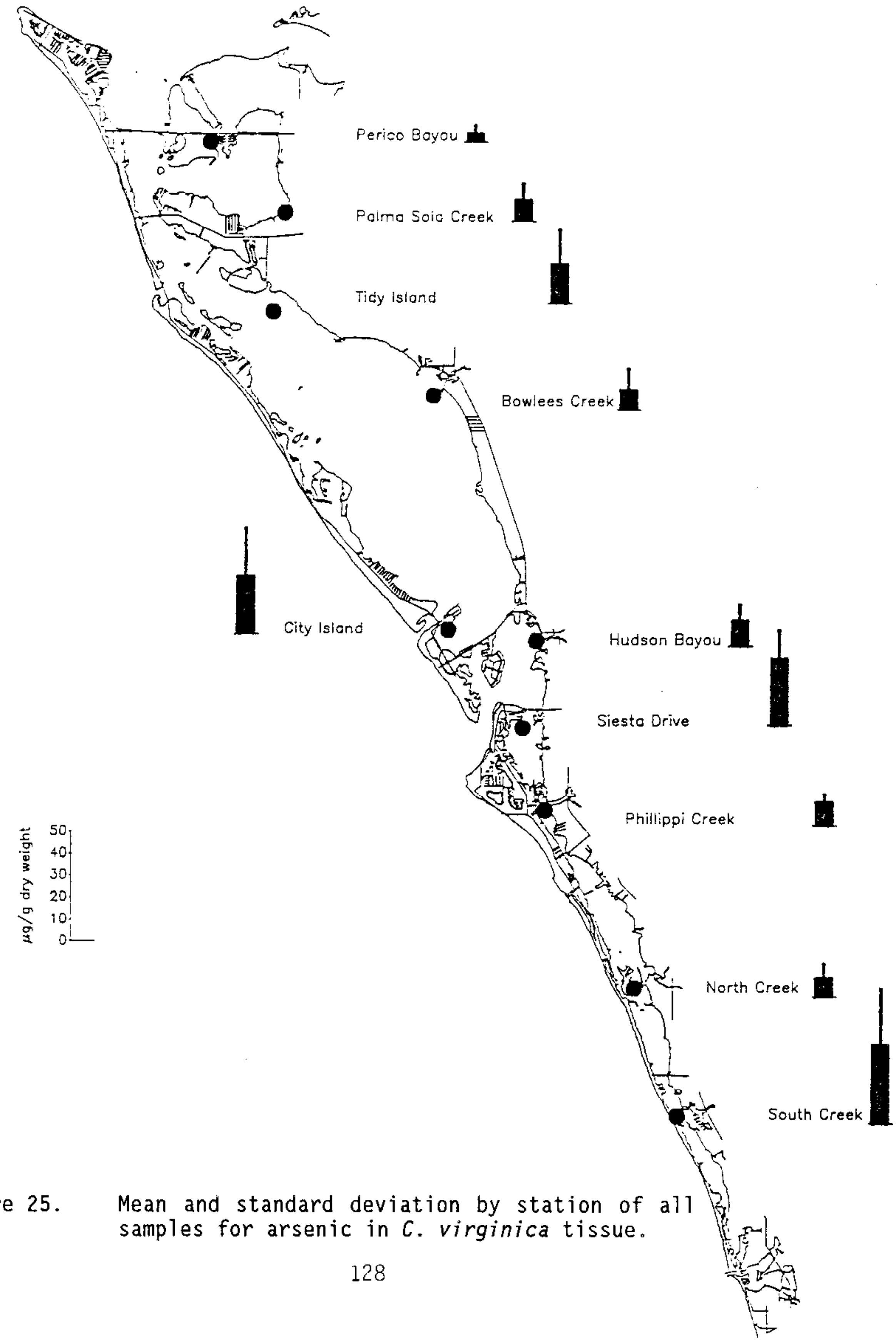

Figure 25. Mean and standard deviation by station of all samples for arsenic in *C. virginica* tissue.

Oyster Tissue - Cadmium

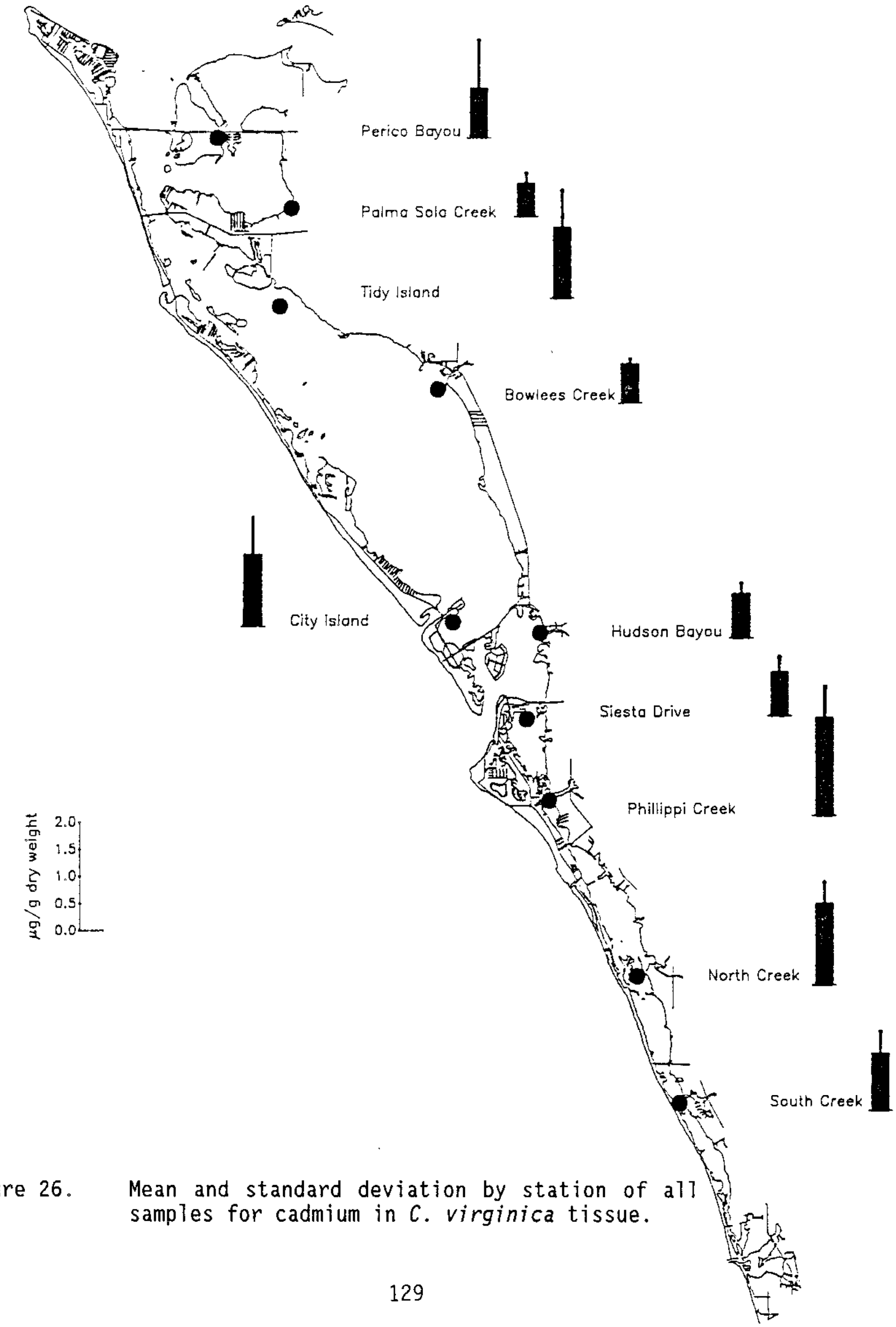

Figure 26. Mean and standard deviation by station of all samples for cadmium in *C. virginica* tissue.

Oyster Tissue - Copper

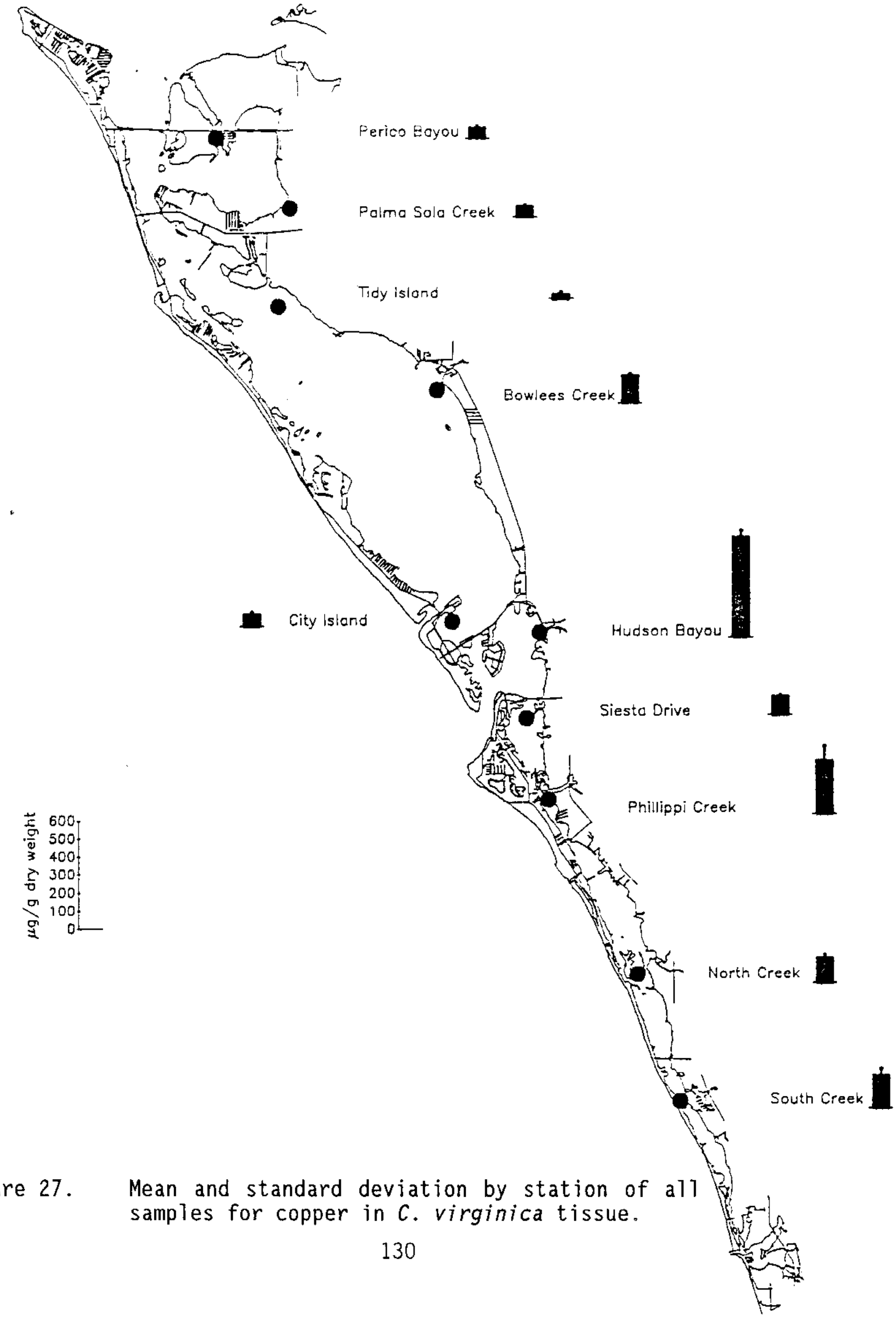

Figure 27. Mean and standard deviation by station of all samples for copper in *C. virginica* tissue.

Oyster Tissue - Lead

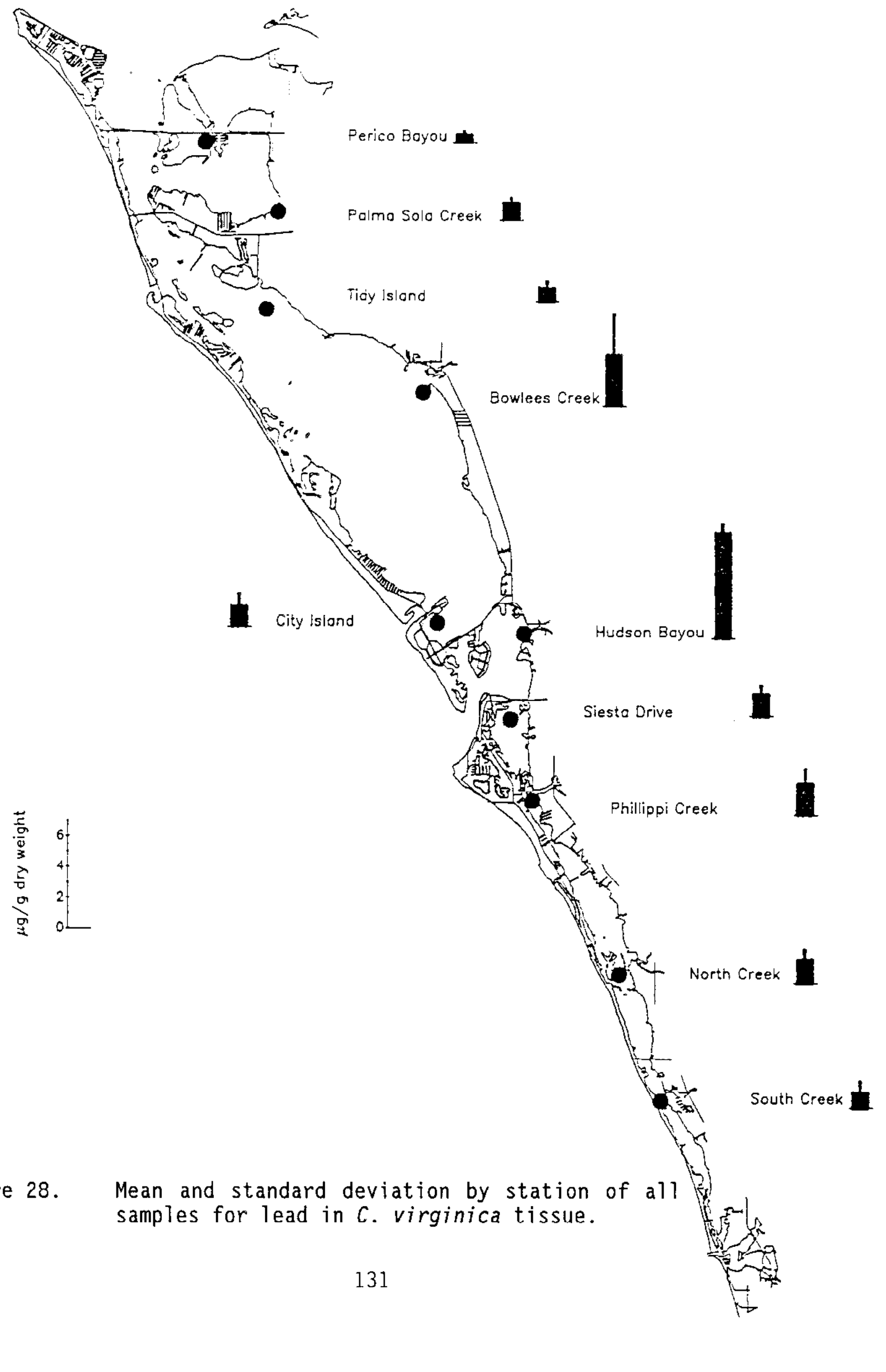

Figure 28. Mean and standard deviation by station of all samples for lead in *C. virginica* tissue.

Oyster Tissue - Mercury

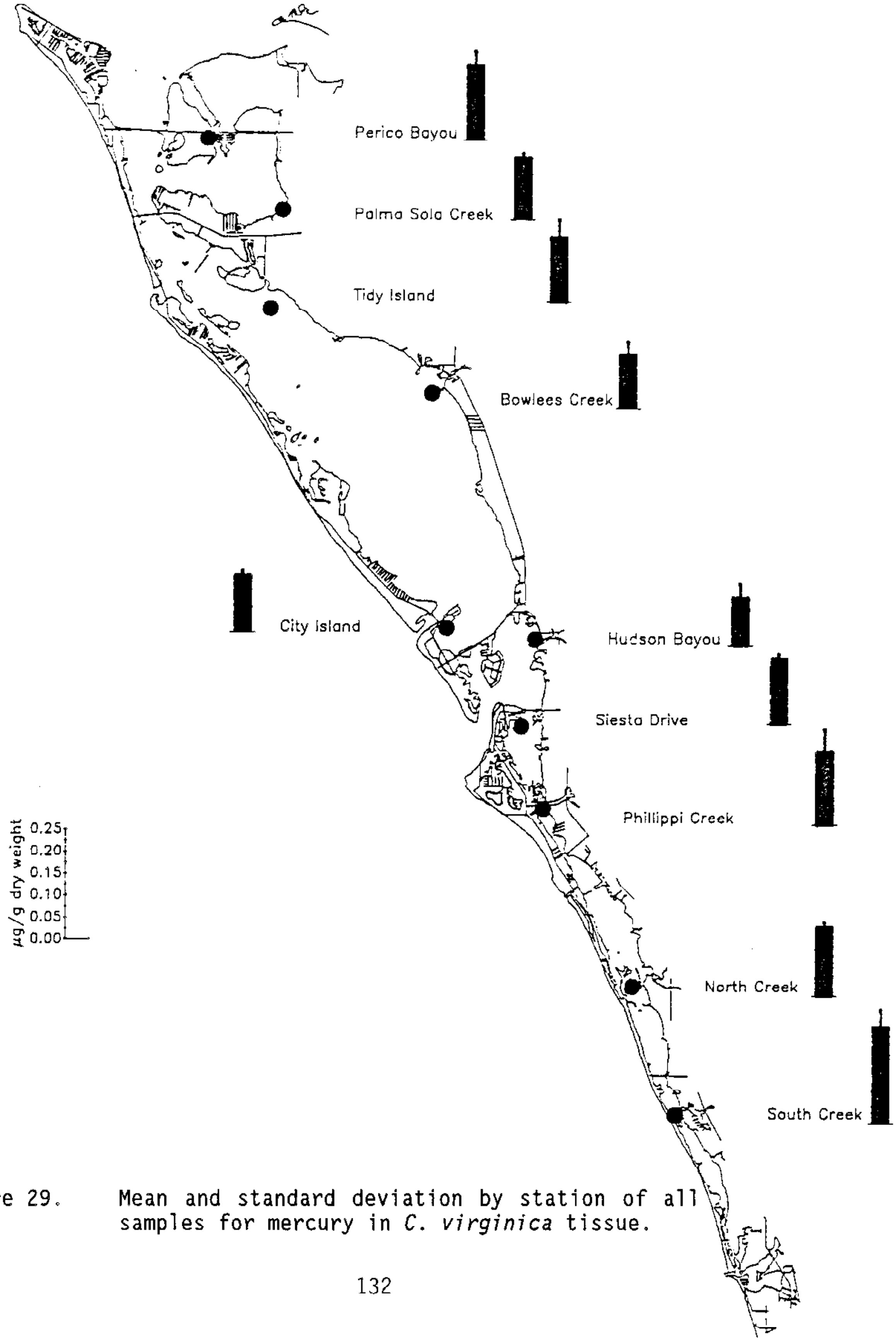

Figure 29. Mean and standard deviation by station of all samples for mercury in *C. virginica* tissue.

Oyster Tissue - Zinc

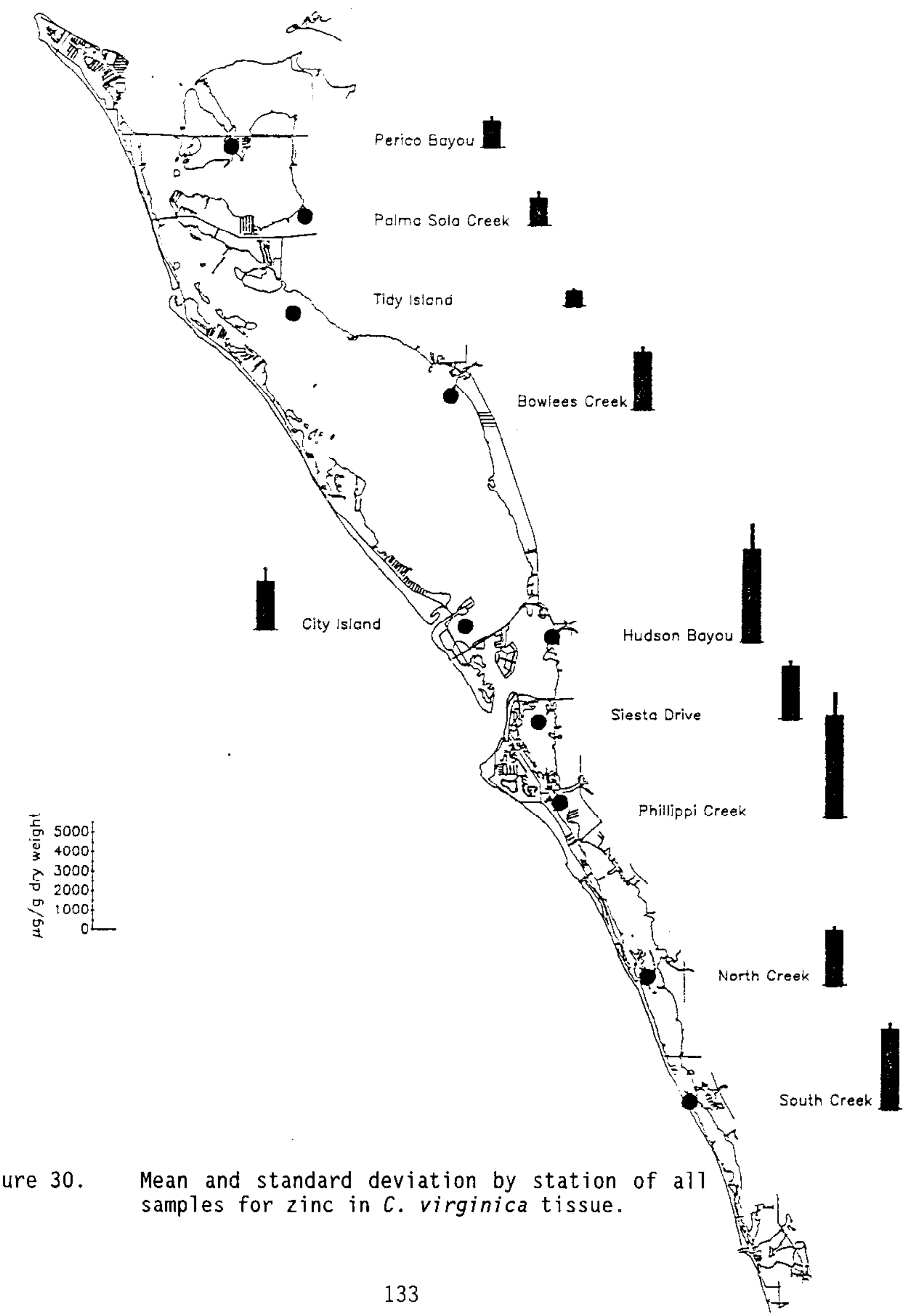

Figure 30. Mean and standard deviation by station of all samples for zinc in *C. virginica* tissue.

Table 34. Station mean tissue concentrations µg/g dry weight) and as a percentage of the maximum value observed for any one species.

Metals in Clam Tissues - Mercenaria campechiensis µg/g dry weight)

	Arsenic	Cadmium	Copper	Lead	Mercurv	Zinc
Manatee Ave Bridge	1.7	1.0	15.4	3.9	0.12	113
Long Bar Point	22.2	1.4	9.4	2.8	0.14	91
Palma Sola Bay	23.8	0.8	11.7	4.4	0.13	94
New Pass	16.7	1.3	17.8	2.0	0.14	91
Coquina Ramp	16.5	1.2	18.1	4.7	0.12	122
Blackburn Bridge	12.1	0.5	20.8	5.9	0.10	191
South Casey Key	23.0	0.9	12.9	4.5	0.12	170
Bishops Point	27.0	1.7	14.4	4.3	0.14	142
Selby Garden	4.0	3.0	22.9	8.7	0.13	102
Phillippi Estuary	19.5	1.5	20.4	10.8	0.21	246

Metal Percent in Clam Tissues (%)

	Arsenic	Cadmium	Copper	Lead	Mercury	Zinc	Mean
Manatee Ave Bridge	6	34	67	36	55	46	41
Long Bar Point	82	45	41	26	67	37	50
Palma Sola Bay	88	25	51	41	60	38	51
New Pass	62	43	78	19	66	37	51
Coquina Ramp	61	39	79	44	58	50	55
Blackburn Bridge	45	16	91	55	49	78	56
South Casey Key	85	30	56	42	59	69	57
Bishops Point	100	57	63	46	67	57	64
Selby Garden	15	100	100	81	60	41	66
Phillippi Estuary	72	49	89	100	100	100	85

Table 34. Continued. Station mean tissue concentrations µg/g dry weight) and as a percentage of the maximum value observed for any one species.

Metals in Oyster Tissues - Crassostrea virginica µg/g dry weight)

	Arsenic	Cadmium	Copper	Lead	Mercury	Zinc
Palma Sola Creek	9.6	0.6	59.8	1.1	0.14	1385
Perico Bayou	3.6	0.9	57.2	0.5	0.17	1326
Tidy Island	18.1	1.3	31.6	0.9	0.15	802
Bowlees Creek	8.8	0.7	153.7	3.3	0.12	2939
North Creek	8.8	1.5	143.4	1.6	0.16	2826
City Island	26.3	1.3	66.1	1.3	0.13	2455
Siesta Drive	30.7	0.8	105.5	1.5	0.15	2716
South Creek	34.8	1.0	179.2	1.0	0.21	3969
Phillippi Creek	11.0	1.8	300.8	2.1	0.17	5254
Hudson Bayou	11.9	0.8	556.1	6.9	0.11	4752

Metal Percent in Oyster Tissues (%)

	Arsenic	Cadmium	Copper	Lead	Mercury	Zinc	Mean
Palma Sola Creek	28	36	11	15	67	26	31
Perico Bayou	10	53	10	8	80	25	31
Tidy Island	52	73	6	13	70	15	38
Bowlees Creek	25	40	28	48	55	56	42
North Creek	25	86	26	22	76	54	48
City Island	76	74	12	18	63	47	48
Siesta Drive	88	43	19	22	71	52	49
South Creek	100	54	32	14	100	76	63
Phillippi Creek	32	100	54	30	82	100	66
Hudson Bayou	34	43	100	100	53	90	70

other variables being comparable, are higher in contaminants in areas receiving higher contaminant loadings.

With this qualifier, however, the draft predicted loadings for a dry year (CDM, 1992) are presented for comparison together with the tissue data from the nearest station likely to represent the dominant influence of the tributary. Due to the salinity requirements of oysters, and the range of metal concentrations exhibited in this species, the oyster stations correspond most closely to the mouths of the various tributaries, although there are many more tributaries than oyster stations as the result of the shellfish study design and the absence of oysters in many locations. In some instances the loading from two smaller basins (North Creek and Catfish Creek) were combined and concentrations averaged as the respective oyster station was near the confluence of these two tributaries.

The oyster stations at the mouths of tributaries included those at Phillippi and Bowlees Creeks, North and South Creek, Hudson Bayou and Palma Sola Creek. Predicted loadings of lead and zinc under dry year conditions appear in Table 35 and Figures 31 and 32 together with the tissue concentrations of these two metals. For lead, Hudson Bayou oysters appear anomalously high and Phillippi and Bowlees Creek organisms low in relation to watershed loadings. Size or age of the organism may contribute to these results, as the Phillippi Creek oysters were the smallest observed and Hudson Bayou individuals among the largest.

Zinc results indicated that Hudson Bayou and South Creek oysters were high in relation to predicted loadings. These two sites also recorded the largest oysters taken for tissue samples, but Bowlees Creek oysters were not significantly smaller than Hudson Bayou individuals and had substantially lower concentrations. These data, taken with the concentrations of other metals observed in Hudson Bayou tissues, would indicate that sediment or other physical parameters make metals more available to oysters in Hudson Bayou or that loads may be underestimated for this watershed. Clam tissue data from near the mouth of Hudson Bayou, in front of the Selby Botanical Gardens, were also among the highest in overall metal concentrations.

Fine scale loadings to an area, such as that produced by the concentration of boats in the mouth of Hudson Bayou, could produce quantities of bioavailable zinc, lead, and copper. The fact that these influences are also present at the Bowlees Creek station, yet tissue concentrations are relatively low, would seem to indicate that sediment controls of metal bioavailability and watershed loading play a more dominant role in controlling tissue concentrations. Additional information will be available from the sediment contaminant data.

VI.C.5. State and National Perspective - Oysters

To place Sarasota Bay shellfish in perspective with the other regions, both local state and national, the emphasis was on the oyster

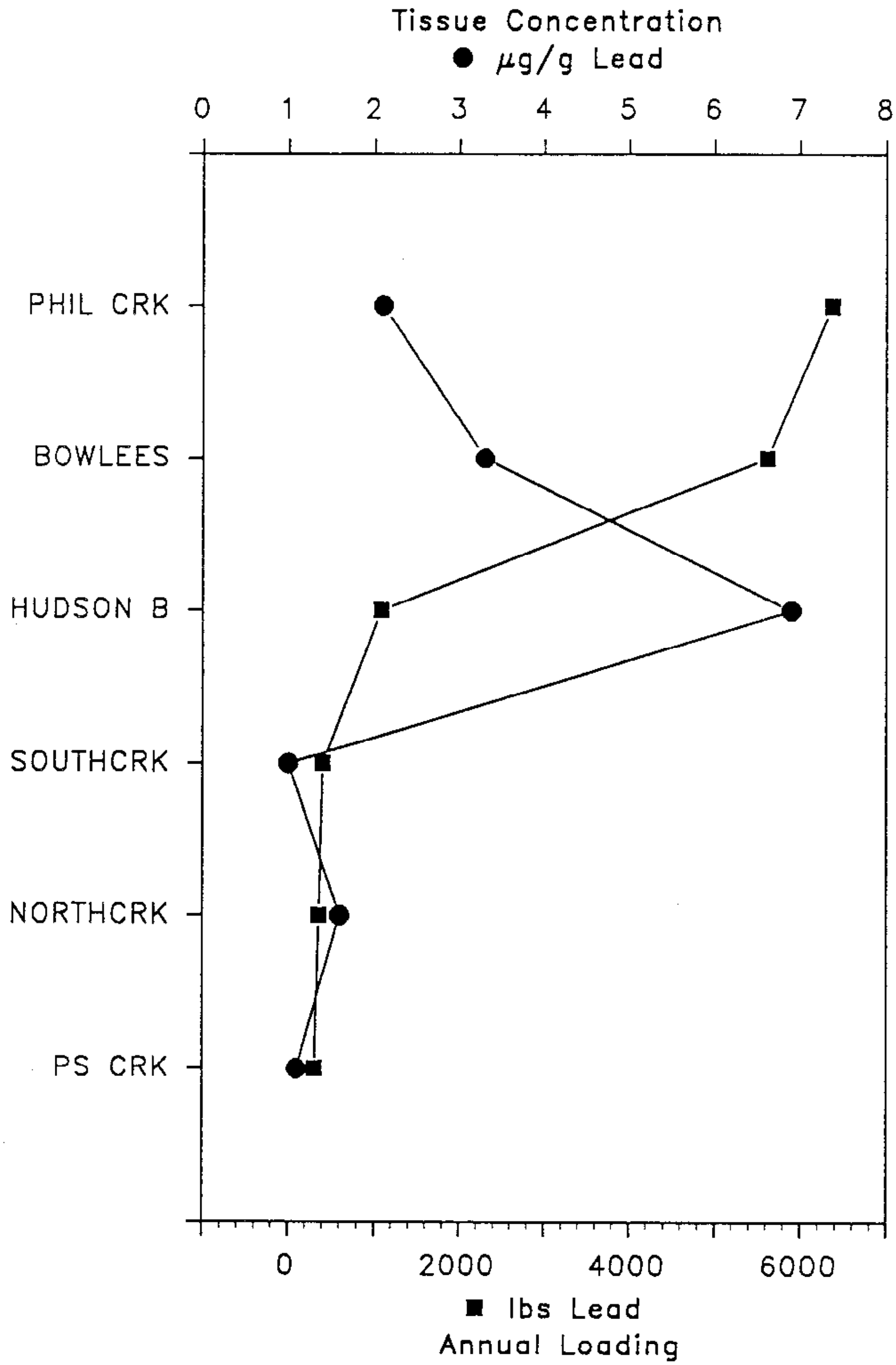

Figure 31. Predicted dry year loadings of lead (CDM, 1992) and lead tissue concentrations reported during this study for *C. virginica* collected from tributary mouths.

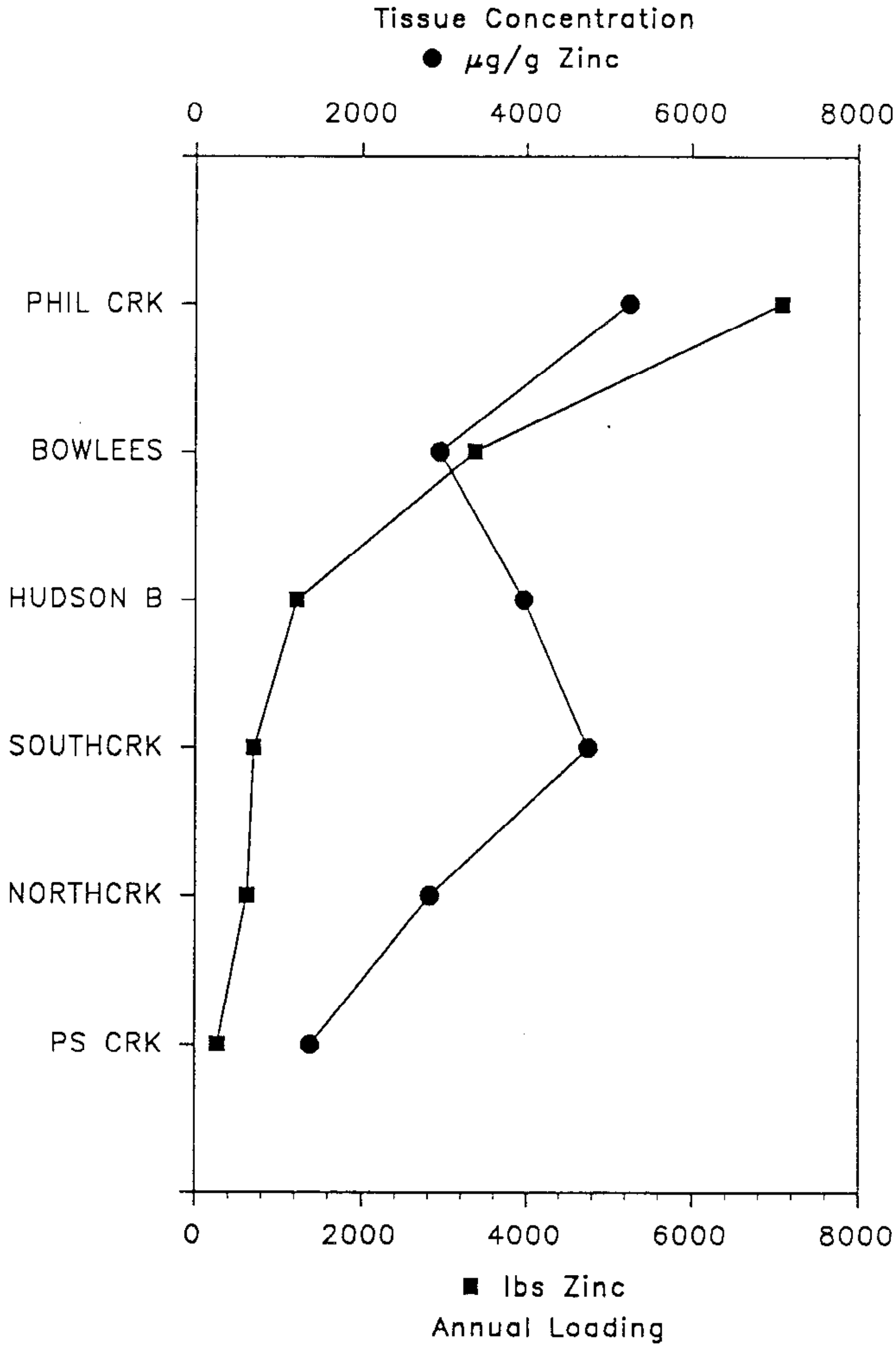

Figure 32. Predicted dry year loadings of zinc (CDM, 1992) and zinc tissue concentrations reported during this study for C. virginica collected from tributary mouths.

Table 35. Watershed loadings of lead and zinc and shellfish tissue concentrations of corresponding stations. (Draft loadings from CDM, 1992.)

	LEAD	
Watershed	Loading (lbs)	Tissue (µg/gdw)
Phillippi Creek	6370	2.1
Bowlees Creek	5620	3.3
Hudson Bayou	1080	6.9
South Creek	400	1.0
North and Catfish Creeks	360	1.6
Palma Sola Creek	310	1.1

	ZINC	
Watershed	Loading (lbs)	Tissue (µg/gdw)
Phillippi Creek	7090	5254
Bowlees Creek	3370	2939
South Creek	1220	3969
Hudson Bayou	700	4752
North and Catfish Creeks	620	2826
Palma Sola Creek	270	1385

tissue data base developed by the National Oceanic and Atmospheric Administration's National Status and Trends Program (NS&T) (NOAA, 1989). Data include dry weight tissue metal values for oysters collected in 1986-1988 at 93 stations from Delaware to Texas with 20 stations located along Florida's Gulf coast. Samples for this program generally consisted of 20 individuals of a preferred size of 70-100 mm, which was larger than the organism available within the Sarasota Bay study area. Sites which were sampled in 1988 but not in previous years, including five of the 20 in Florida, were generally added to investigate more urban areas.

Mean oyster tissue values for the entire Sarasota Bay study are presented in Tables 36 and 37 in comparison to NS&T data for the nation and for the Florida stations in the Gulf of Mexico. The Sarasota Bay data are calculated as a percent of the mean, median, and maximum value recorded in the NS&T data sets, and as a percentile within the entire NS&T data set.

Sarasota Bay metal concentrations in oysters approximate or are lower than national averages for cadmium, copper, mercury and zinc. Arsenic levels are slightly higher than the national average, 16.4 µg/g compared to 11.0 µg/g. Lead concentrations are substantially higher than national averages, 2.0 µg/g compared to 0.6 µg/g, although the Sarasota Bay average is still less than 40 percent of the maximum lead concentration recorded for any of the NS&T stations. Cadmium concentrations are approximately one fourth of the national average. Sarasota Bay arsenic, lead, and mercury concentrations are all above the 80'" percentile of the national data set, while lead for Sarasota Bay is at the 99"" percentile, nationally.

In relation to Florida Gulf coast values, Sarasota Bay oysters are lower than average in cadmium and mercury, average for arsenic, slightly above average for copper and zinc, and well above average for lead (2.0 compared to 0.8 µg/g). The average lead was still less than 40 percent of the maximum observed in the Florida NS&T stations, but the Sarasota Bay Hudson Bayou concentration of 6.9 µg/g exceeded the highest lead value reported (5.4 µg/g) for either Florida or the nation.

Individual stations within the Sarasota Bay study area were contrasted with the 20 Florida Gulf coast stations and appear in Figures 33 through 38. Individual stations are noted for their comparatively high arsenic (South Creek and Siesta Bridge), copper (Hudson Bayou), lead (Hudson Bayou and Bowlees Creek), and zinc (Phillippi Creek, Hudson Bayou, and South Creek) concentrations. Cadmium and mercury concentrations were low to average in comparison.

The mean concentrations at the 20 Florida Gulf coast NS&T stations together with the ten Sarasota Bay oyster stations were ranked by individual metal, and the ranks summed by station and converted to a percentage of the maximum possible (i.e., 100 percent indicating the most contaminated station for all metals). This again equally weights all metals without regard to toxicity. The values appear in Table 38 and indicate that tissues from Phillippi Creek and South Creek have relatively

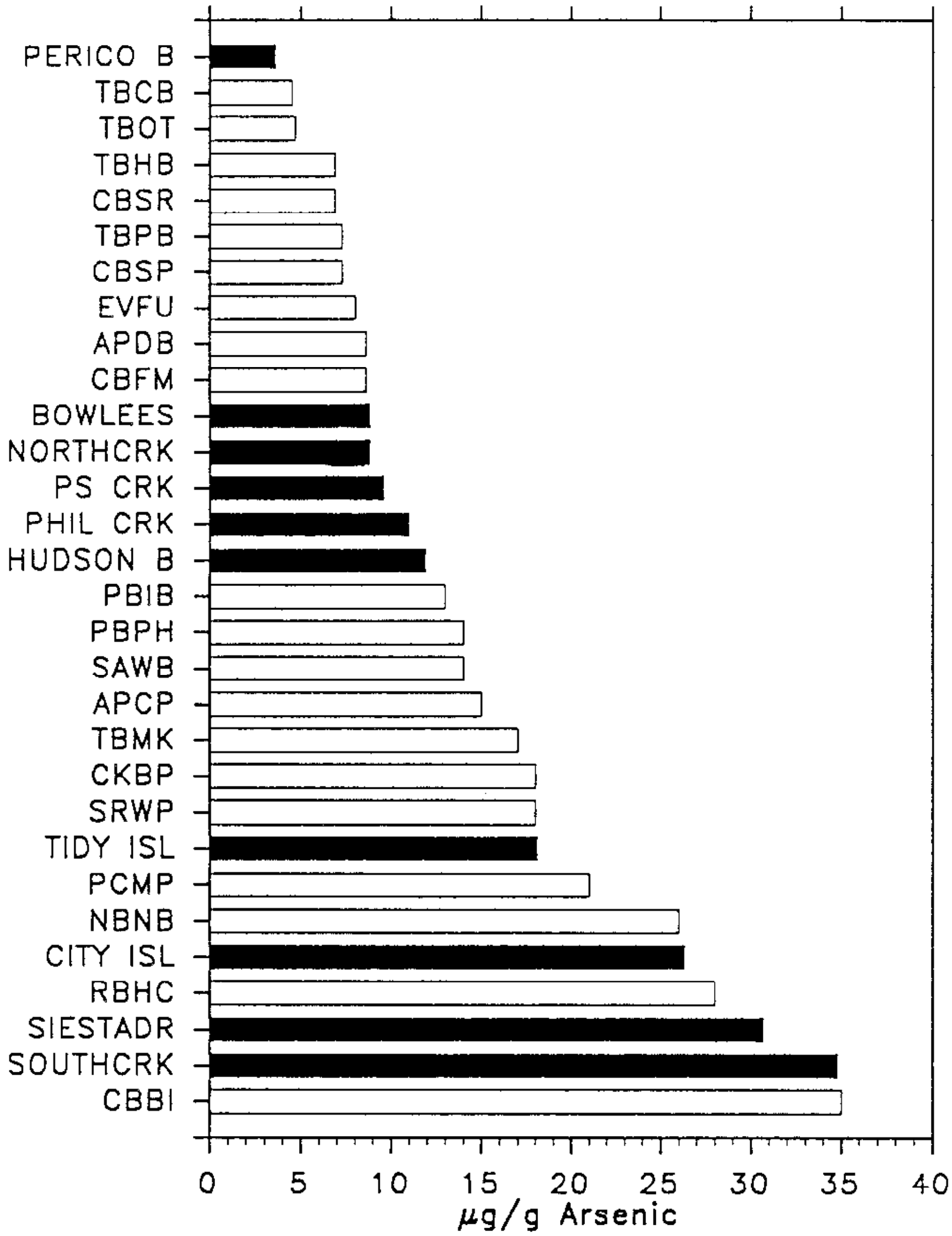

Figure 33. Comparison of study mean values of arsenic in *C. virginica* tissues from Sarasota Bay (dark bars) to the station means computed from 1986-1989 National Status and Trends data (open bars), Florida Gulf coast stations.

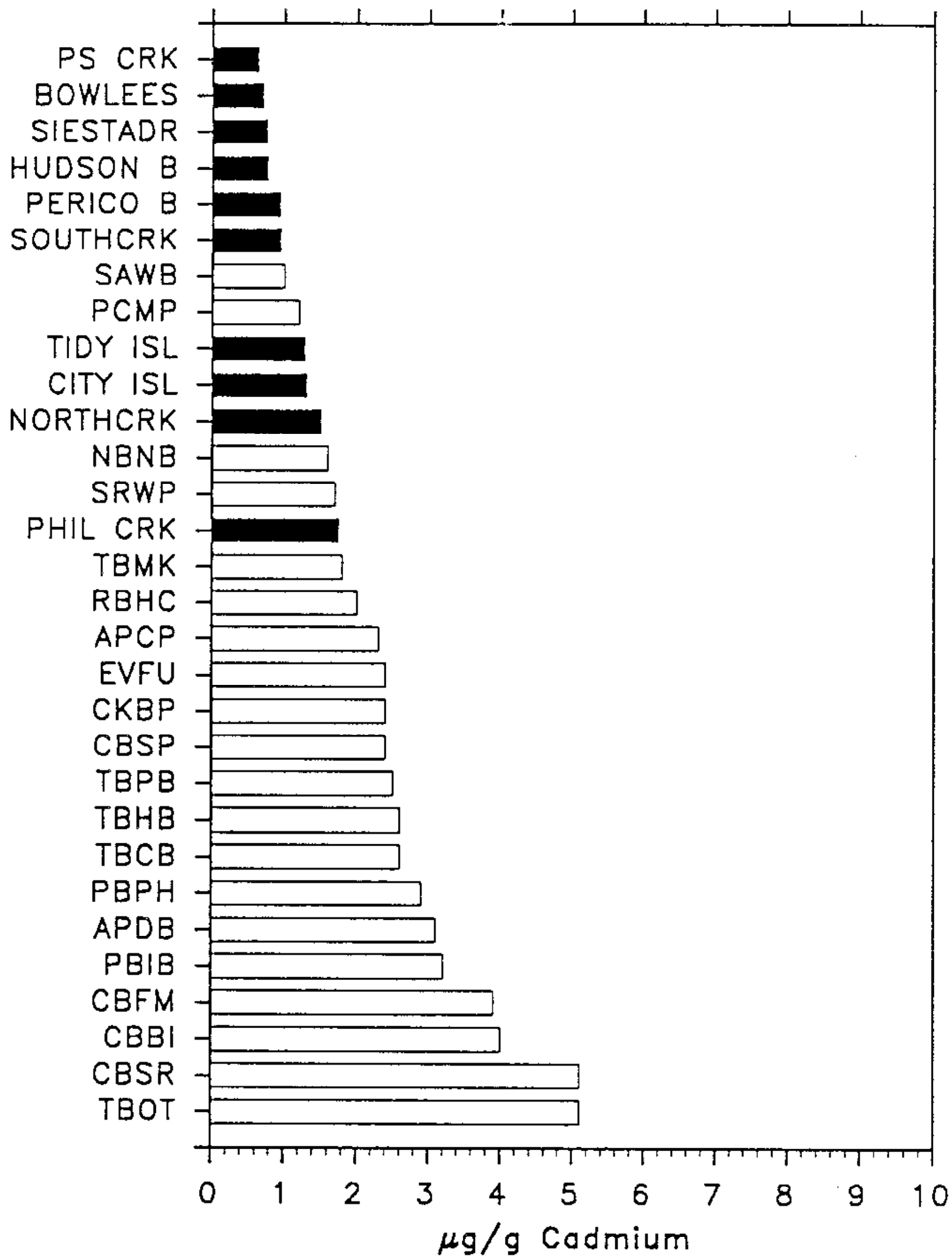

Figure 34. Comparison of study mean values of cadmium in *C. virginica* tissues from Sarasota Bay (dark bars) to the station means computed from 1986-1989 National Status and Trends data (open bars), Florida Gulf coast stations.

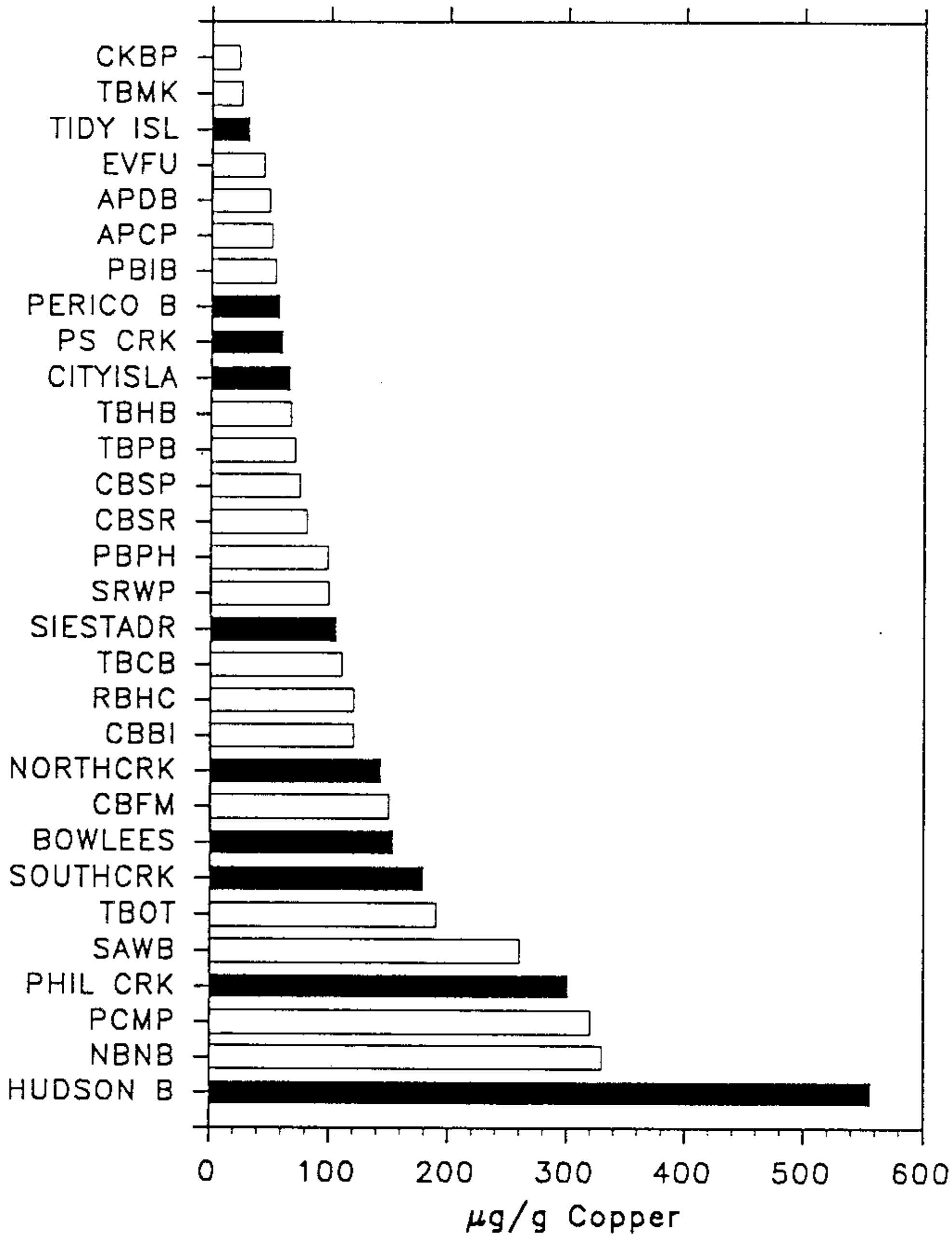

Figure 35. Comparison of study mean values of copper in *C. virginica* tissues from Sarasota Bay (dark bars) to the station means computed from 1986-1989 National Status and Trends data (open bars), Florida Gulf coast stations.

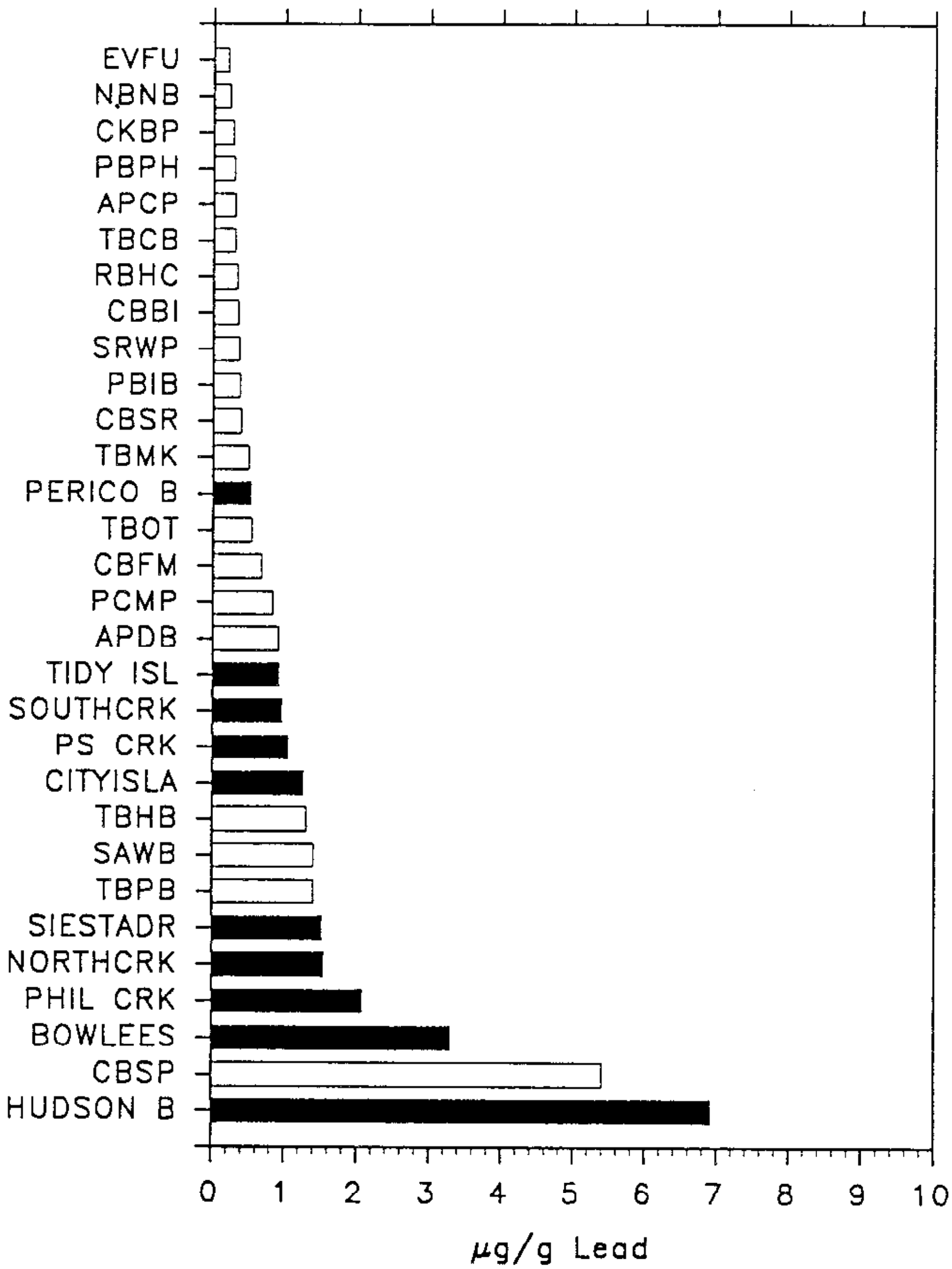

Figure 36. Comparison of study mean values of lead in *C. Virginica* tissues from Sarasota Bay (dark bars) to the station means computed from 1986-1989 National Status and Trends data (open bars), Florida Gulf coast stations.,

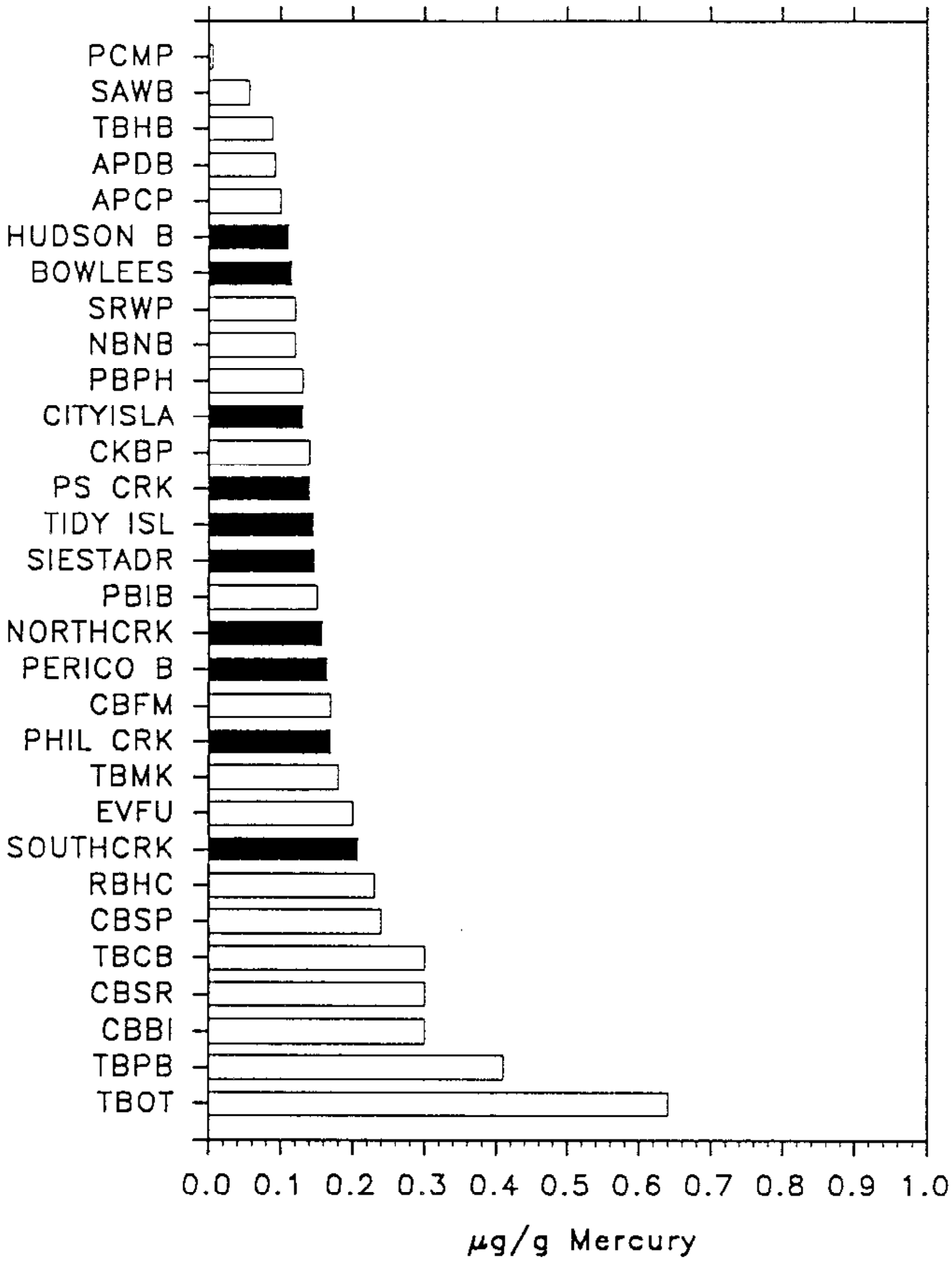

Figure 37. Comparison of study mean values of mercury in *C. Virginica* tissues from Sarasota Bay (dark bars) to the station means computed from 1986-1989 National Status and Trends data (open bars), Florida Gulf coast stations.

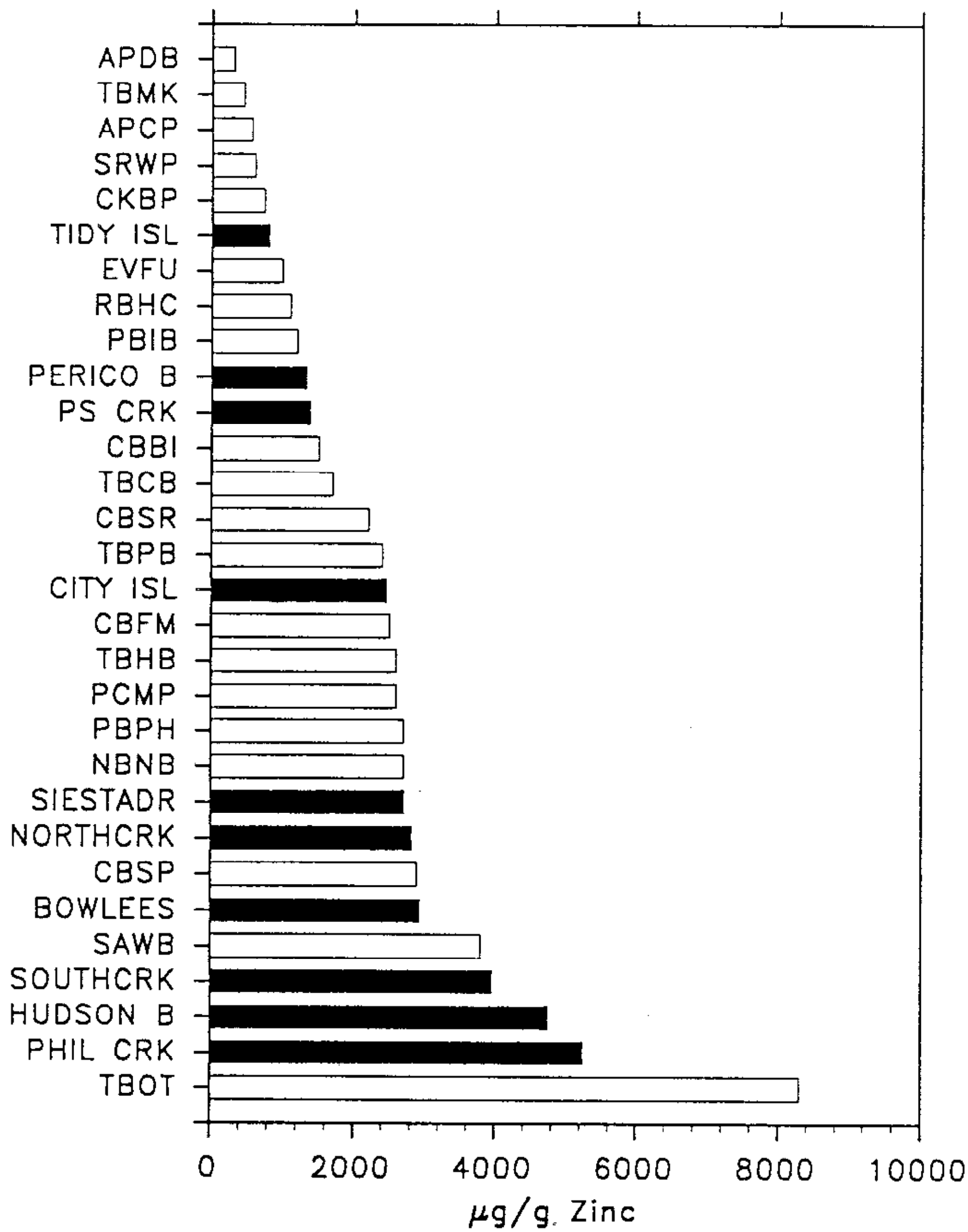

Figure 38. Comparison of study mean values of zinc in *C. Virginica* tissues from Sarasota Bay (dark bars) to the station means computed from 1986-1989 National Status and Trends data (open bars), Florida Gulf coast stations.

Table 36. Sarasota Bay oyster tissue data compared to National Status and Trends concentrations (93 stations). NS&T data consist of station means of annual average concentrations, 1986-1988 data.

	Arsenic (µg/g)	Cadmium (µg/g)	Copper (µg/g)	Lead (µg/g)	Mercury (µg/g)	Zinc (µg/g)
Minimum NS&T	2.6	0.4	23	0.2	0.01	310
Sarasota Bay station minimum	3.6	0.6	32	0.5	0.11	802
Sarasota Bay study mean	16.4	1.1	165	2.0	0.15	2842
Sarasota Bay station maximum	34.8	1.8	556	6.9	0.21	5254
Maximum NS&T	41.0	13.0	820	5.4	0.64	11000
Mean NS&T	11.0	3.9	158	0.6	0.11	2630
Median NS&T	8.1	3.3	130	0.5	0.09	2200
	(%)	(%)	(%)	(%)	(%)	(%)
Sarasota Bay Percentile	83	6	62	99	82	68
Sarasota Bay as % of mean	149	28	104	332	131	108
Sarasota Bay as % of median	202	33	127	400	170	129
Sarasota Bay as % of maximum	40	8	20	37	23	26

Table 37. Sarasota Bay oyster tissue data compared to National Status Trends concentrations (20 stations of Florida Gulf Coast). NS&T data consist of station means of annual average concentrations, 1986-1988 data.

	Arsenic (µg/g)	Cadmium (µg/g)	Copper (µg/g)	Lead (µg/g)	Mercury (µg/g)	Zinc (µg/g)
Minimum NS&T (FL data)	4.5	1.0	23	0.2	0.01	310
Sarasota Bay station minimum	3.6	0.6	32	0.5	0.11	802
Sarasota Bay study mean	16.4	1.1	165	2.0	0.15	2842
Sarasota Bay station maximum	34.8	1.8	556	6.9	0.21	5254
Maximum NS&T (FL data)	35.0	5.1	330	5.4	0.41	3800
Mean NS&T (FL data)	14.7	2.5	116	0.9	0.17	1819
Median NS&T (FL data)	13.5	2.4	90	3.8	0.15	1950
	(%)	(%)	(%)	(%)	(%)	(%)
Sarasota Bay Percentile	67	10	81	95	52	86
Sarasota Bay as % of mean	112	44	143	225	86	156
Sarasota Bay as % of median	121	46	183	53	100	146
Sarasota Bay as % of maximum	47	22	50	37	37	75

Table 38. Individual and mean ranking of Sarasota Bay stations relative to Florida NS&T data. NS&T data are the station mean of 1986-1988 annual averages.

Station	Arsenic As (µg/g)	As Rank	Cadmium Cd (µg/g)	Cd Rank	Copper Cu (µg/g)	Cu Rank	Lead Pb (µg/g)	Pb Rank	Mercury Hg (µg/g)	Hg Rank	Zinc Zn (µg/g)	Zn Rank	Mean Rank
TBOT	4.7	28	5.1	2	190.0	6	.5	17	.64	1	8300		9
* Phillippi Creek	11.0	17	1.8	17	300.8	4	2.1	4	.17	11	5254	2	9
* South Creek	34.8	2	1.0	25	179.2	7	1.0	12	.21	8	3969	4	10
CBBI	35.0	1	4.0	3	120.0	11	.3	23	.30	4	1500	19	10
CBSP	7.3	24	2.4	13	75.0	18	5.4	2	.24	6	2900	7	12
* Hudson Bayou	11.9	16	.8	27	556.1	1	6.9	1	.11	25	4752	3	12
* Siesta Drive	30.7	3	.8	28	105.5	14	1.5	6	.15	16	2716	9	13
* North Creek	8.8	19	1.5	20	143.4	10	1.6	5	.16	14	2826	8	13
CBFM	8.6	22	3.9	4	150.0	9	.7	16	.17	12	2500	14	13
TBPB	7.3	25	2.5	10	71.0	19	1.4	8	.41	2	2400	16	13
CBSR	6.9	26	5.1	1	81.0	17	.4	20	.30	3	2200	17	14
SAUB	14.0	14	1.0	24	260.0	5	1.4	7	.06	29	3800	5	14
RBHC	28.0	4	2.0	15	120.0	12	.3	24	.23	7	1100	23	14
* Bowlees Creek	8.8	20	.7	29	153.7	8	3.3	3	.12	24	2939	6	15
NBNB	26.0	6	1.6	19	330.0	2	.2	29	.12	23	2700	11	15
PCMP	21.0	7	1.2	23	320.0	3	.8	15	.01	30	2600	13	15
* City Island	26.3	5	1.3	21	66.1	21	1.3	10	.13	20	2455	15	15
PBPH	14.0	13	2.9	7	99.0	16	.3	27	.13	21	2700	10	16
TBCB	4.5	29	2.6	9	110.0	13	.3	25	.30	5	1700	18	17
PBIB	13.0	15	3.2	5	54.0	24	.4	21	.15	15	1200	22	17
TBHB	6.9	27	2.6	8	67.0	20	1.3	9	.09	28	2600	12	17
* Tidy Island	18.1	8	1.3	22	31.6	28	.9	13	.15	17	802	25	19
SRUP	18.0	9	1.7	18	99.3	15	.4	22	.12	22	600	27	19
TBMK	17.0	11	1.8	16	25.0	29	.5	19	.18	10	450	29	19
* Palma Sola Creek	9.6	18	.6	30	59.8	22	1.1	11	.14	18	1385	20	20
CKBP	18.0	10	2.4	11	23.0	30	.3	28	.14	19	730	26	21
APDB	8.6	21	3.1	6	49.0	26	.9	14	.09	27	310	30	21
EVFU	8.0	23	2.4	12	44.0	27	.2	30	.20	9	990	24	21
APCP	15.0	12	2.3	14	51.0	25	.3	26	.10	26	560	28	22
* Perico Bayou	3.6	30	.9	26	57.2	23	.5	18	.17	13	1326	21	22

* - Sarasota Bay stations, 1990 data.

high concentrations of all metals, exceeded only by a station in Old Tampa Bay. Hudson Bayou, while high in copper, lead, and zinc, is low enough in the other metals analyzed to appear comparable to North Creek and Siesta Bridge. Oysters from Perico Bayou are extremely low overall in metal concentrations.

VI.C.6. State and National Perspective - Clams

No integrated data set of the magnitude of the NS&T data was available to compare clam tissues. Selected studies (WAR, 1988; MML, 1990; Beach and Hittinger, 1989; Caspar, 1988) were used (Table 39) which represented a very similar species *(Mercenaria mercenaria)* in Narragansett Bay, an extremely urbanized estuary of the northeast, and *Mercenaria campechiensis* tissue concentrations in a relatively pristine location in Cumberland Sound, Georgia. Both of these data sets were relatively recent (1986-1990).

As might be expected, the Sarasota organisms were intermediate in tissue concentrations of copper and zinc between the pristine and urbanized areas and zinc concentrations were approximately double those observed in Cumberland Sound. The mean cadmium concentrations from Sarasota Bay, while averaging higher than the data from Narragansett Bay, were not significantly different (t-test). Potential species differences (between *M. mercenaria* and *M. campechiensis)* in metal detoxification strategies may also play a role. Higher lead levels in Sarasota Bay were also not significantly different from the Narragansett Bay data set. The higher means, however, may be another symptom of the apparently elevated lead concentrations of Sarasota Bay as identified by the oyster tissue samples.

VI.C.7. Ecological Implications

There is a comparative lack of data sets in which biological effects data (mortality, physiological processes, reproductive impairment, or other sublethal effects) are presented together with tissue concentrations, most being evaluated as a function of water column or sediment concentrations. Long et al. (1991) have compiled this information for oysters.

For copper, a reduced shell thickness is associated with 450 µg/g (Frazier, 1976), and mortality increases from 6 to 10 percent with an increase in tissue concentrations from 607 to 1,458 µg/g (Shuster and Pringle, 1969). Oxygen consumption, normal at 500 µg/g of copper in tissues, increases when tissue concentrations reach 800 µg/g (Engel and Fowler, 1979) and abnormal larvae production increases above 270 µg/g. Based on these criteria, the oysters in Hudson Bayou may suffer from reduced shell thickness, and those in Hudson Bayou and Phillippi Creek, a higher incidence of abnormal larvae.

Table 39. Concentrations of metals in clam tissues from other locations. Units are in µg/g dry weight.

Station/Site	Cadmium	Copper	Lead	Mercury	Zinc
Average of 11 states (Caspar, 1988)	0.4	18.5	1.2	-	90
Cumberland Sound - 1987-1988 (WAR, 1989)*	1.1	3.3	1.4	0.35	62
Cumberland Sound - 1990 (MML, 1991)*	0.3	12.8	0.7	0.39	63
Narragansett Bay, RI - 1985-86 (Beach and Hittinger, 1989)**	0.7	25.2	3.6	0.26	190
Sarasota Bay - 1992 (This document)	1.3	16.4	5.2	0.13	135

* Values reported as wet weights. Converted to dry weights using 20% solids.

** *Mercenaria mercenaria,* remaining studies with *M. campechenisis.*

Zinc tissue concentrations above 4,100 µg/g again reduced oyster shell thickness (Frazier, 1976) in comparison to organisms with tissue concentrations of 1,700 µg/g. Mortalities increased from 6 to 10 percent when tissue concentrations increased from 8,540 to 10,164 µg/g (Shuster and Pringle, 1969). For Sarasota Bay, oysters in Hudson Bayou, Phillippi Creek and possibly South Creek may suffer from reduced shell thickness, although mortalities due to zinc in tissue do not appear to be a problem.

Growth rates of oysters varied between arsenic levels of 5 and 8 µg/g in the Chesapeake Bay but the slight difference in arsenic concentration implies that other factors may contribute. All but one station (Perico Bayou) averaged arsenic concentrations greater than 8 µg/g.

There were no comparable data specific from oysters for cadmium, lead, or mercury, but data from other species (mussels) indicate that 3 µg/g of lead or 0.4 µg/g of mercury in tissues may represent a threshold at which effects can be observed. (The physiological differences between species should be kept in mind and this information used with caution.) Hudson Bayou and Bowlees Creek oysters both had lead concentrations greater than 3 µg/g, although no station had mercury concentrations greater than 0.21 µg/g.

For clam tissues, those data available for mussels were used for an indication of ecological effects. Copper less than 36 µg/g generally show no deleterious effects for mussels (Long et al., 1991) and no clam station averaged greater than 30 µg/g in Sarasota Bay. In contrast, six of the ten clam stations had tissue concentrations averaging greater than 3 µg/g of lead, with those in the Phillippi Creek Estuary, near Selby Gardens, and at the Blackburn Point Bridge the highest. No clam station had mercury tissue concentrations which averaged greater than 0.21 µg/g, while all but two stations exceeded the 7 µg/g threshold at which effects were noted for arsenic in mussels.

VI.C.8. Human Health Implications

A formal risk assessment of shellfish consumption was beyond the scope of this present study and methodological summaries (Brown, et al., 1990) as well as case studies (Metcalf & Eddy, 1988) identify a number of uncertainties and needs for the proper applications of these techniques, which include evaluation of carcinogenicity, mutagenicity, developmental/reproductive toxicity, systemic toxicity, and exposure. Foremost among the shortfalls is the lack of information on amounts and patterns of seafood consumption (including types of species and mode of preparation), as well as the uncertainties regarding the reproductive and developmental toxicities of the various elements and compounds. In addition, FDA guidelines address only a few of the toxic materials known to occur in shellfish.

For this study, to evaluate the potential human health impacts of metals ingestion via shellfish consumption, guidelines for exposure were necessary. FDA action levels were incomplete, missing some metals, as were acceptable intake limits established by Gartrell and others (1985), or the oral reference doses established by USEPA (USEPA, 1985, Metcalf & Eddy, 1988). A composite table was prepared (Table 40) which incorporated data from these sources, as well as the normal dietary intakes given by Iyengar (1989). Arsenic exposures are based on arsenic as the trivalent (most toxic) form rather than the organic forms present in fish and shellfish tissues. Where data were given in µg/kg/day, a 70 kg individual was assumed. For these screening purposes, the lowest of any of the intake levels was conservatively used for reference, but even these levels may not adequately reflect reproductive and developmental toxicities or hazards for particular at-risk individuals.

Seafood consumption estimates vary widely with region and population studied and range from 1.2 to 165 g/day of mixed seafood (fish and shellfish). For this analysis, a value of 5 g/day of shellfish (oysters or clams) was selected, which would represent 1,825 g/year (4 lbs/year or approximately 8 cups of shucked oyster or clam meats) of wet weight shellfish consumed. These weights were corrected to dry weights with a percent solids value of 20 percent and metals consumed per day was evaluated as a percentage of the lowest acceptable or normal daily intake value derived from Table 40.

The metals ingested through an average 5 g/day consumption of clams or oysters, as a percentage of a daily intake standard, are presented in Table 41. For the purposes of this study, an arbitrary level of 30 percent of the acceptable maximum daily metal intake through shellfish was set as a conservative screening criteria. A more quantitative assessment would obviously require more detailed dietary investigations as to the amounts of metals received through alternate foods and exposure routes.

Arsenic levels, as a percent of the acceptable intake level, were quite high for both clam and oyster tissues, averaging 120 percent and 117 percent, respectively. It should be restated, however, that acceptable intake levels of arsenic are established based on the consumption of the inorganic trivalent form of this metal rather than the comparatively non-toxic organic forms found in seafood tissues. The National Academy of Sciences (1990), in particular, reports that arsenic consumed in seafood has not been found to be toxic.

For the remaining metals, no clam tissues, consumed at a average rate of 5 g/day, would exceed 30 percent of the acceptable maximum intake level. For oyster tissues, Phillippi Creek and Hudson Bayou were above 30 percent for zinc, but below 30 percent all other stations and metals. The maximum percentage of any metal or tissue was 38 percent for zinc in oysters at Phillippi Creek. Fortunately, zinc and arsenic, as they exist in fish and shellfish tissues, are among some of the least toxic to humans.

A more restrictive criteria (but again arbitrarily selected) of 10 percent was next examined. All tissues, clam and oyster alike,

Table 40. Average, acceptable, or reference dosages for metals assuming a 70kg individual and without reference to potential reproductive or developmental toxicities. Lowest value selected for evaluation of Sarasota Bay Shellfish.

ELEMENT	AVERAGE DIETARY[a] INTAKE (MG/D)	ACCEPTABLE DAILY[b] INTAKE (MG/D)	ORAL[c] REFERENCE	ACCEPTABLE DAILY[d] DOSE
Arsenic	100	--	--	14
Cadmium	40	57-72	20.3	38.5
Copper	2,500	--	2590	2800
Lead	300	429	98	42
Mercury	10	43	140	7
Zinc	13,000	15,000	--	14,000

[a] Iyengar, 19.
[b] Landolt et al., 1987.
[c] Metcalf & Eddy, 1988. Reference dose is the level where the critical effect is unlikely to occur.
[d] Brown et al., 1990.

Table 41. Metal content of shellfish, assuming average consumption of 5 g/day, as a percentage of a maximum acceptable intake rate for individual metals.

	Arsenic (µg/g)	% of Max	Cadmium (µg/g)	% of Max	Copper (µg/g)	% of Max	Lead (µg/g)	% of Max	Mercury (µg/g)	% of Max	Zinc (µg/g)	% of Max
Clam Tissue												
Bishops Point	27.0	193	1.7	8	14.4	1	4.3	10	0.14	2	142	1
Blackburn Bridge				2	21.3	1	5.9	14	0.10	1	181	1
Coquina Ramp	16.5	118	1.2	6	18.1	1	4.7	11	0.12	2	122	1
Long Bar Point	22.2	159	1.4	7	9.4	0	2.8	7	0.14	2	91	1
Manatee Ave Bridge	1.7	12	1.0	5	15.4	1	3.9	9	0.12	2	113	1
New Pass	16.7	119	1.3	6	17.8	1	2.0	5	0.14	2	91	1
Phillippi Estuary	19.5	139	1.5	7	20.4	1	10.8	26	0.21	3	246	2
Palma Sola Bay	23.8	170	0.8	4	11.7	0	4.4	10	0.13	2	94	1
South Casey Key	23.0	164	0.9	5	12.9	1	4.5	11	0.12	2	170	1
Selby Gardens	4.0	29	3.0	15	22.9	1	8.7	21	0.13	2	102	1
Mean Clam	16.8	120	1.3	7	16.4	1	5.2	12	0.13	2	135	1
Oyster Tissue												
Bowlees Creek	8.8	63	0.7	3	153.7	6	3.3	8	0.12	2	2939	21
City Island	26.3	188	1.3	6	66.1	3	1.3	3	0.13	2	2455	18
Hudson Bayou	11.9	85	0.8	4	556.1	22	6.9	16	0.11	2	4752	34
North Creek	8.8	63	1.5	7	143.4	6	1.6	4	0.16	2	2826	20
Perico Bayou	3.6	26	0.9	5	57.2	2	0.5	1	0.17	2	1326	9
Phillippi Creek	11.0	79	1.8	9	300.8	12	2.1	5	0.17	2	5254	38
Palma Sola Creek	9.6	69	0.6	3	59.8	2	1.1	3	0.14	2	1385	10
Siesta Bridge	30.7	219	0.8	4	105.5	4	1.5	4	0.15	2	2716	19
South Creek	34.8	249	1.0	5	179.2	7	1.0	2	0.21	3	3969	28
Tidy Island	18.1	129	1.3	6	31.6	1	0.9	2	0.15	2	802	6
Mean Oyster	16.4	117	1.1	5	165.3	7	2.0	5	0.15	2	2842	20

exceeded to 10 percent level for arsenic. For zinc, all oyster tissues with the exception of those from Tidy Island and Perico Bayou exceeded this value, although no clam tissues were in excess. Again, toxicity levels of zinc are extremely low and arsenic intake levels assume that all arsenic is present in the trivalent form. Clams from Selby Gardens exceeded the 10 percent criteria for cadmium as did copper in oysters from Hudson Bayou and Phillippi Creek. For lead, consumption of oysters from Hudson Bayou would exceed 10 percent of acceptable daily intake levels, while all clam tissues with the exception of those from Long Bar Point, Manatee Avenue Bridge and New Pass would exceed 10 percent.

VI.C.9. Comparison to Regulatory Standards/Guidelines

A number of regulatory limits on contaminant concentrations in foods and shellfish were compiled and presented in Table 42. Although not expressly stated with the exception of the NSSP guidelines, these data were assumed to be on a wet weight basis. Guidelines from both international and national sources are included although some (NSSP data) appear to be based on percentile values from previous surveys rather than on documented health impacts. The minimum values of any presented, however, were used in reference to Sarasota Bay shellfish, which appear as station means by wet weight in Table 43.

Arsenic as a contaminant is not addressed by any of these organizations. The minimum alert or action level for cadmium, copper, mercury, or zinc was not exceeded by any of the Sarasota Bay station averages. For lead, all of the clam stations, with the exception of New Pass, exceeded the Canadian action level of 0.5 µg/g, as did Hudson Bayou and Bowlees Creek for oysters. The less restrictive NSSP criteria of 2.0 µg/g for oysters and 4.0 µg/g for hard clams was not violated by any station, but the validity of a standard established by a percentile of observed concentrations is open to criticism.

V.C.10. Summary - Toxic Metals

In summary, there was no consistent seasonal variation in tissue metal concentrations, although this result may differ under years with more rainfall. Comparisons between species support other literature in that oysters are noted for high concentrations of copper and zinc. There were significant variations between stations for all metals and each species with the exception of mercury. Oysters typically displayed a larger range between stations than did Mercenaria and the least geographic variation, although still significant, was evidenced by cadmium values.

Overall, tissue metal concentrations were most notable in Hudson Bayou, Phillippi Creek, and South Creek. Tissue concentrations of zinc and lead do not correlate particularly well with predicted loads from the various basins, which may reflect varying bioavailability of metals.

Table 42. Regulatory limits for toxic contaminants in seafood extracted from United Nations, United States, and Canadian regulations and unofficial NSSP Alert Levels (µg/g wet weight).

Contaminant	FAO/WHO (µg/g)[a]	FDA Action Level (µg/g)[a]	Canada Health and Welfare Action Level (µg/g)	NSSP Alert Levels for Trace Metals (µg/g)	
				Oysters	Hard-Shell Clams
Arsenic					
Cadmium	3.6-4.4			3.5	0.5
Copper				175.0	10.0
Lead	26.7		0.5	2.0	4.0
Mercury	2.7	1.0	0.5		
Methylmercury	1.8				
Zinc				2,000.0	65.0

[a] Based on conversion of FAO/WHO provisional tolerable weekly intake (PTWI) to U.S. population assuming average per capita seafood consumption to be 18.7 grams per 70 kilogram adult body weight per day.

Source: DFO (1988); FAO (1989); FDA (1987); Ratcliffe and Wilt (1971).

Table 43. Mean tissue metal concentrations (µg/g wet weight).

Mean Clam Tissue	Arsenic	Cadmium	Copper	Lead	Mercury	Zinc
Bishops Point	5.2	0.33	2.8	0.82	0.027	27.1
Blackburn Point Bridge	2.0	0.08	3.4	0.96	0.017	31.2
Coquina Beach	3.1	0.23	3.4	0.90	0.023	23.2
Long Bar Point	4.6	0.28	1.9	0.56	0.029	18.6
Manatee Ave Bridge	0.3	0.21	3.1	0.77	0.023	22.8
New Pass	3.0	0.24	3.2	0.36	0.025	16.6
Palma Sola Bay	4.7	0.15	2.3	0.87	0.025	18.6
Phillippi Creek Estuary	3.3	0.25	3.4	1.80	0.035	41.2
Selby Garden	0.8	0.59	4.4	1.69	0.024	19.8
South Casey Key	4.1	0.16	2.3	0.80	0.022	30.2
Mean Oyster Tissue						
Bowlees Creek	1.7	0.13	29.0	0.62	0.022	554.7
City Island	5.3	0.26	13.2	0.25	0.026	490.4
Hudson Bayou	2.2	0.14	103.1	1.28	0.020	881.1
North Creek	1.5	0.25	24.2	0.26	0.027	476.2
Palma Sola Creek	2.2	0.14	13.5	0.24	0.032	313.1
Perico Bayou	0.7	0.19	11.6	0.11	0.033	269.0
Phillippi Creek	1.5	0.25	42.4	0.29	0.024	740.8
Siesta Dr Bridge	5.3	0.13	18.3	0.26	0.026	471.2
South Creek	5.8	0.16	29.7	0.16	0.034	658.8
Tidy Island	3.5	0.25	6.1	0.18	0.028	155.8

Sarasota Bay metal concentrations in oysters approximate or are lower than national averages for cadmium, copper, mercury and zinc. Arsenic levels are slightly higher than the national average. Lead concentrations are substantially higher than national averages, although the Sarasota Bay average is still less than 40 percent of the maximum lead concentration recorded for any of the NS&T stations. In relation to Florida Gulf coast values, Sarasota Bay oysters are lower than average in cadmium and mercury, average for arsenic, slightly above average for copper and zinc, and well above average for lead. The Sarasota Bay Hudson Bayou concentration of 6.9 µg/g exceeded the highest lead value reported (5.4 µg/g) for either Florida or the nation. Metal concentrations in clam tissues were similar to other urban areas.

There was little information available on ecological impacts on *Mercenaria* or *Crassostrea* as a function of tissue concentrations of toxic metals. What was available indicated that oysters in Phillippi Creek, Hudson Bayou, and possibly South Creek may suffer from impairments such as altered shell thickness and abnormal larvae.

Shellfish tissues did not violate any applicable FDA action levels, but lead from most clam stations and a few oyster stations did exceed the more restrictive Canadian limits. Rough assumptions on shellfish consumption rates (5 g/day), various agency recommendations on acceptable daily intake (ADI) rates of toxic metals, and Sarasota Bay tissue concentrations were used to estimate what percent of an acceptable intake would be constituted by shellfish consumption. Shellfish from most sites would not exceed 20 percent of an ADI for all metals except arsenic. Of these metals the most exposure was from zinc in oysters. For arsenic, quantities reported exceeded 100 percent of the ADI. The ADI, however, is based upon inorganic arsenical compounds and the organically bound arsenic found in seafood is considered to be substantially less toxic.

VI.D. Toxic Organics - Results and Discussion

VI.D.1. Pesticides

Results of the spring and fall 1990, pesticide analyses are given in Table 44, reporting the average and standard deviation of duplicate samples collected from the same site. The concentrations are given in ng/g dry weight tissue, with a lower limit of detection in shellfish tissue of 3 ng/g, and a limit of quantification of 10 ng/g, in which trace amounts were detected but were too low for reliable quantification. Where

quantification and the remaining sample had undetectable quantities, the average of the two samples is presented even if the resulting number is at or below the limit of quantification.

No samples had detectable or quantifiable concentrations of the following pesticide compounds: α-BHC, aldrin, endrin, o,p-'DDD, o,p'-DDE, o,p'-DDT, p,p'-DDT, heptachlor, dibrom, or bendiocarb.

Table 44. Pesticide concentrations (ng/g dry wt.) in Sarasota Bay shellfish, mean and standard deviation of duplicate samples.

Sample	Species	ß-BHC	τ-BHC	Chlorpyrifos	Heptachlor Epox.	α-Chlordane	p,p-DDE	Dieldrin	p,p-DDD
SPRING, 1990									
	Mercenaria spp.								
Blackburn Bridge		ND	ND	ND	ND	35,49	ND	ND	ND
	Crassotrea virginica								
Phillippi Creek		ND	ND	ND	ND	ND	370,353	ND	ND
FALL. 1990									
	Mercenaria spp.								
Blackburn Bridge		ND	ND	ND	ND	ND	ND	10,14	ND
Coquina Ramp		ND	Tr	Tr	Tr	ND	ND	ND	ND
Longbar		ND	ND	Tr	60,70	ND	ND	ND	ND
Manatee.Ave.		ND	ND	5,7	ND	ND	Tr	5,7	ND
Phillippi Estuary		ND	597	ND	ND	ND	ND	ND	ND
	Crassotrea virginica								
Hudson Bayou		ND	ND	ND	ND	ND	195,92	ND	70,42
Palma Sola Creek		ND	ND	ND	ND	ND	Tr	ND	ND
Phillippi Creek		70,49	ND	Tr	ND	ND	ND	ND	ND
South Creek		ND	ND	ND	ND	ND	ND	95,106	ND
Tidy Island		15,29	25,35	ND	ND	ND	ND	ND	ND

ND = Not detected: limit of detection = 3 ng/g dry wt.

Tr = Trace (below limit of quanititation) ≤ 10 ng/g dry wt.

Of the 18 pesticides analyzed, the spring sampling resulted in only three samples from two stations with concentrations above detectable levels. Only two pesticides, α-chlordane and p,p'-DDE, were found. A metabolite of DDT, p,p'-DDE, was found in both oyster samples from Phillippi Creek, at an average level of 370 ng/lg. One clam sample from Blackburn Bridge recorded α-chlordane at a level of 70 ng/lg, whereas the duplicate sample had no detectable amount, resulting in an average for the two samples of 35 ng/lg. No detectable levels of the organophosphates or carbamate pesticides were found during the spring.

Results of the fall pesticide analyses exhibited a greater number of samples with pesticides, although no station with detectable pesticides in the spring reported those same compounds during the fall sampling. Most concentrations were low, near the detection limits. Of the 20 stations sampled, quantifiable (>10 ng/lg) amounts of chlorinated pesticides were found in samples from eight locations, which included both clam and oyster stations. Dieldrin, followed by p,p'-DDE, , ßBHC, and τ-BHC were the most prevalent compounds. Samples from five stations had trace amounts of compounds that were detected but were too little for quantification. Unlike spring, there was no α-chlordane detected during the fall.

During the fall, trace amounts of the organophosphate chlorpyrifos (dursban) was detected in three clam samples and one oyster sample, indicating some influx of pesticides currently in use. Only one clam sample, from the Manatee Ave. site, however, contained quantifiable amounts of chlorpyrifos, averaging 5 ng/lg.

For the study as a whole, eight of the 18 pesticides under analysis were found in shellfish. There were no obvious geographic, or species distributions of the tissue concentrations, trace amounts of pesticides being found in most regions of the study area and in both species. Of the station averages of the pesticides detected, the highest concentrations were usually contained in oysters, but these organisms were also more directly exposed, as stations were preferentially near the mouths of tributaries.

Dieldrin was the most prevalent compound during the study (occurring in the most number of samples) and ranged between <3 to 95 ng/lg dry weight. The next most prevalent were ß-BHC (<3 to 70 ng/lg), τ-BHC (<10 to 25 ng/lg), and p,p'-DDE (<10 to 370 mg/g). The remaining compounds quantified were found at concentrations of 70 ng/lg dry weight or less.

Results of these studies were consistent with previous studies of chlorinated pesticides in shellfish, with the most prevalent chemicals being DDT metabolites, chlordane and dieldrin (NOAA, 1989). No pesticide exceeded the FDA action levels listed in Table 6, nor did any violate the slightly more stringent Canadian Action levels. The oysters collected in the spring from Phillippi Creek, however, did contain DDE in concentrations equal to 7 percent of the FDA action level of 5,000 ng/g. *Mercenaria* from Blackburn Bridge also contained approximately 12 percent of the FDA action level for chlordane. All other shellfish samples from the spring sampling contained less than detectable levels. One sample of oyster tissue from Hudson Bayou contained 5 percent of the total DDT (the sum of all DDT, DDE, and DOD) allowed by FDA during the fall sampling. The remaining samples contained smaller to non-detectable amounts of the various pesticides.

The concentrations of p,p'-DDE in Phillippi Creek oysters during the spring sampling represent anomalously high amounts relative to other shellfish samples throughout the southwest Florida coast (NOAA, 1989), but is still well below the 5,000 ng/g FDA Action Level for fish. The fact that p,p'-DDE was not found in oyster samples from the same site during the subsequent collection, however, indicates no continuous contamination problem. The DDE may have resulted from resuspension of contaminated sediment during digging or dredging operations that were completed prior to the subsequent sampling period. Anomalously high amounts of DDE in conjunction with quantifiable amounts of the other pesticides observed in 14 of the 40 sites analyzed (35 percent), indicate the need for continued monitoring of selected areas, ideally expanding the analyses to include select PCB isomers known for their toxicity.

The concentrations of total chlorinated contaminants (pesticides from the EPA 608 series) in shellfish tissue averaged over the two seasonal sampling episodes are illustrated as mean ± standard deviations in Figure 39 for clam tissue and in Figure 40 for oyster tissue samples. Pesticide contamination was detected (in five of the ten clam sampling sites, with the greatest amount found at the Long Bar site (due to the heptachlor levels found). Blackburn Bridge clams were the next most contaminated overall, again due to a single compound (α-chlordane). Oysters exhibited greater concentrations of pesticides and residues were detected at 5 of the 10 sites sampled, with the greatest amounts found at Phillippi Creek and Hudson Bayou. This ranking was again primarily the product of concentrations of single compounds, p,p'-DDE for both locations.

Chlorpyrifos residues were similarly averaged over the study and presented in Figures 41 and 42. The *Mercenaria* from the Manatee Avenue site and the oysters in Phillippi Creek had the highest average chlorpyrifos residues for the study.

VI.D.2. Polycyclic Aromatic Hydrocarbons

Results of the 1990 spring and fall PAH analyses are given in Table 45 reporting amounts in ng/g dry weight tissue above the lower limit of detection, 15 ng/g, and the limit of quantification, 50 ng/g.

Analyses of shellfish collected during spring detected PAH at one clam station and at three oyster stations. In all cases only trace amounts were detected, or between 15 and 50 ng/g. Of these trace amounts, clams at south Casey Key contained the most number of compounds (naphthalene, 2-methyl naphthalene (a C_1-naphthalene), an ethyl naphthalene (C_2), and fluorene) which were indicative of petroleum origin. Oyster

Clam Tissue - Total Chl.

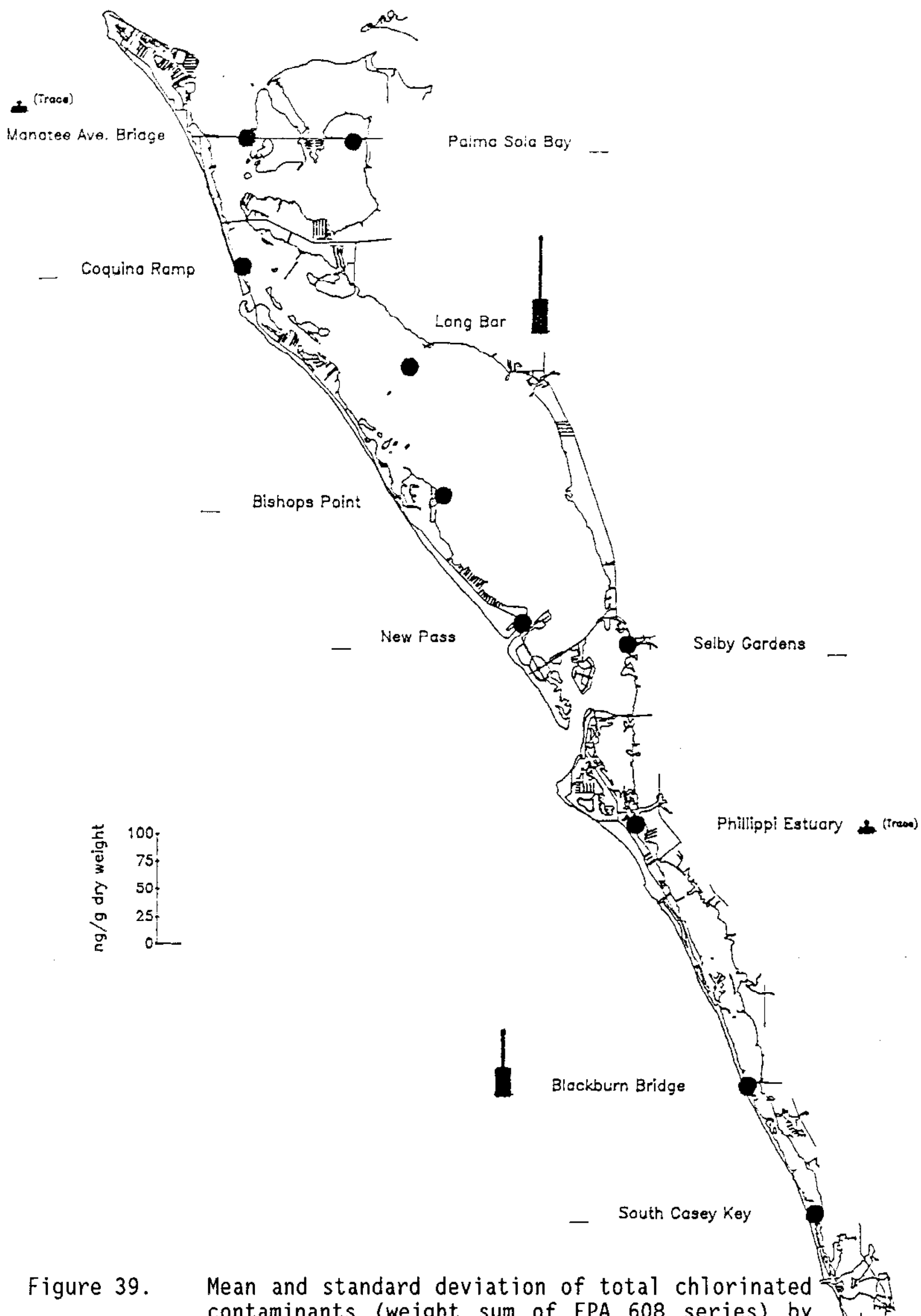

Figure 39. Mean and standard deviation of total chlorinated contaminants (weight sum of EPA 608 series) by station in *Mercenaria* spp. tissues.

Oyster Tissue - Total Chl.

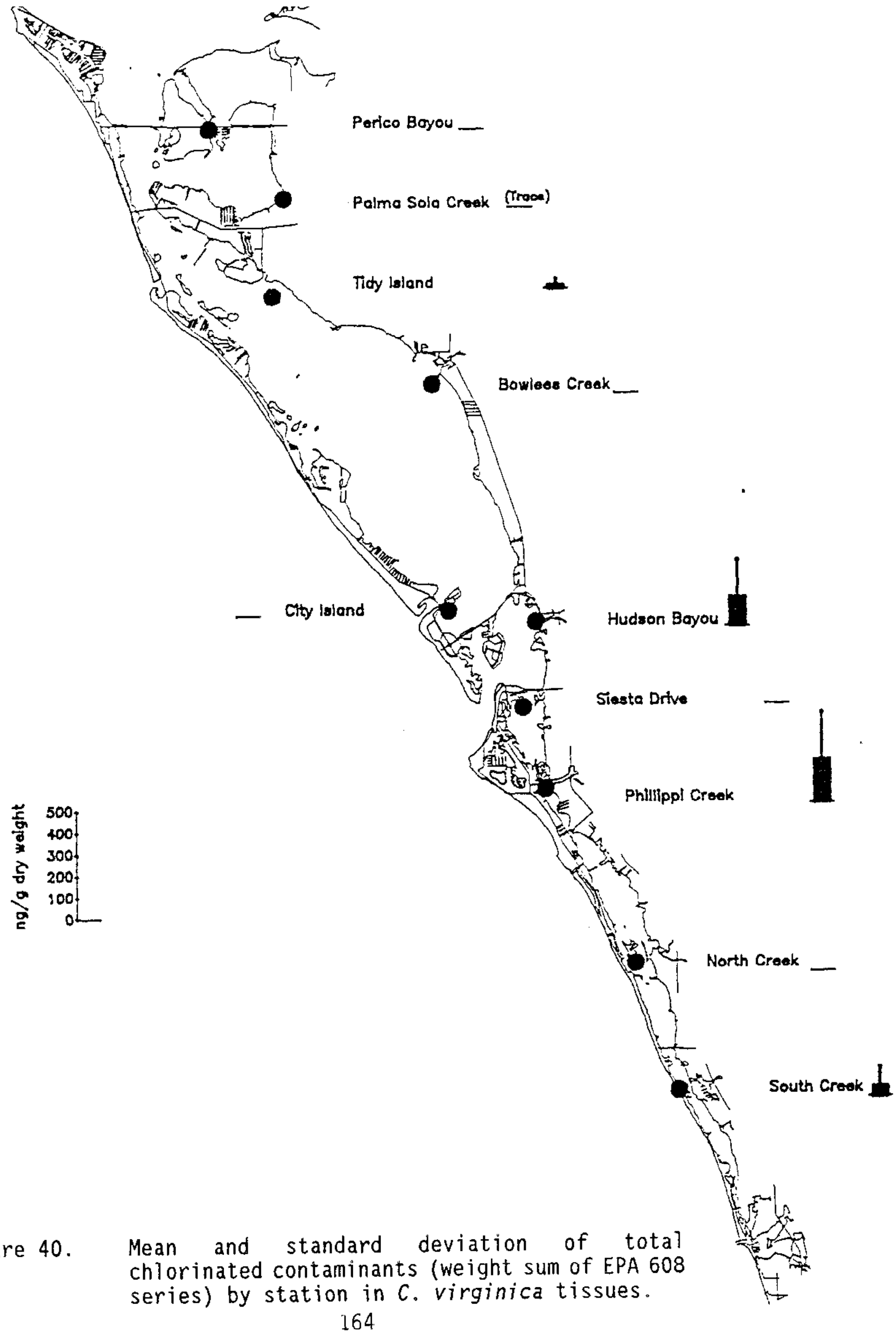

Figure 40. Mean and standard deviation of total chlorinated contaminants (weight sum of EPA 608 series) by station in *C. virginica* tissues.

Clam Tissue - Total Chlorpyrifos

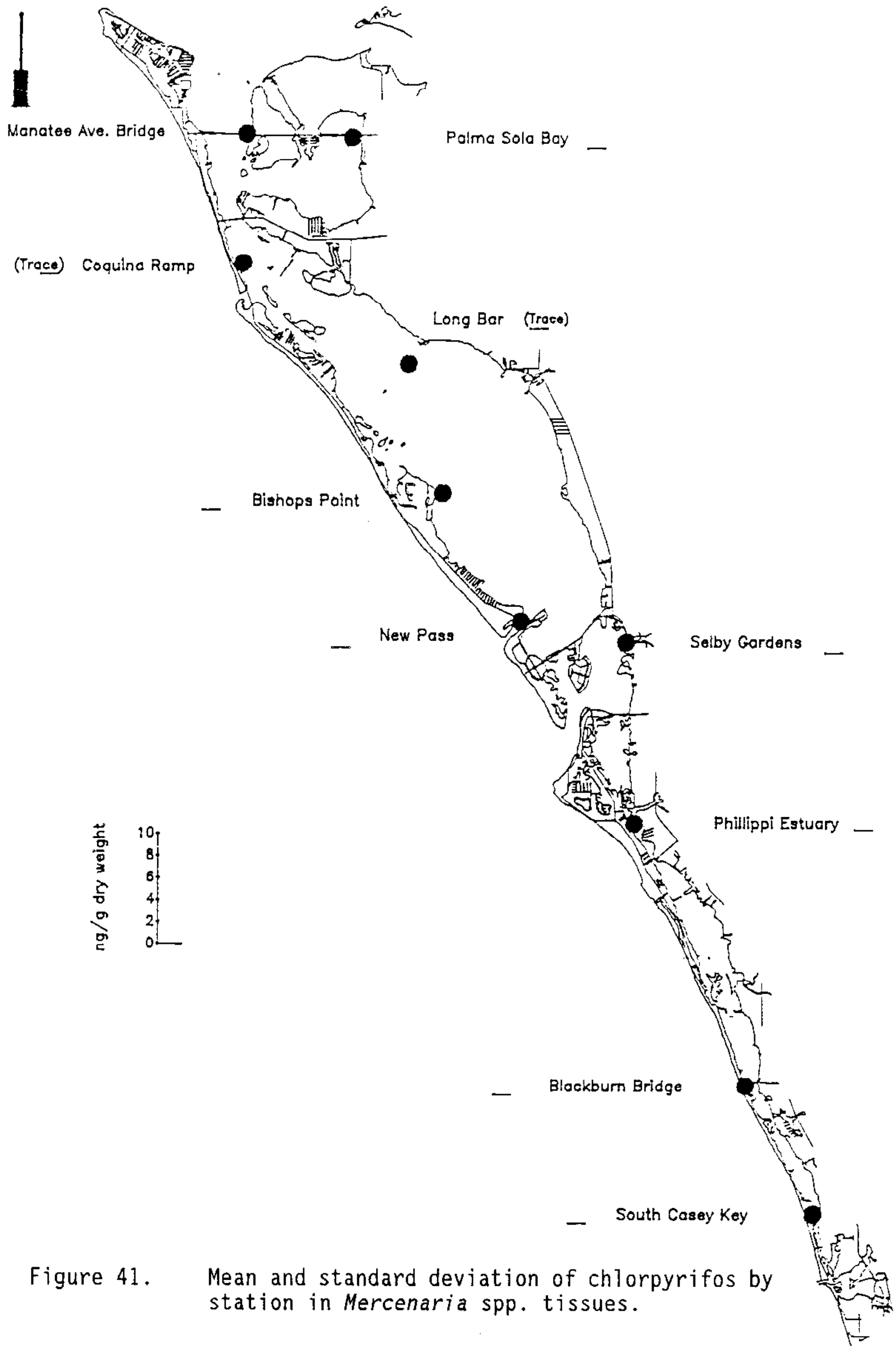

Figure 41. Mean and standard deviation of chlorpyrifos by station in *Mercenaria* spp. tissues.

Oyster Tissue - Total Chlorpyrifos

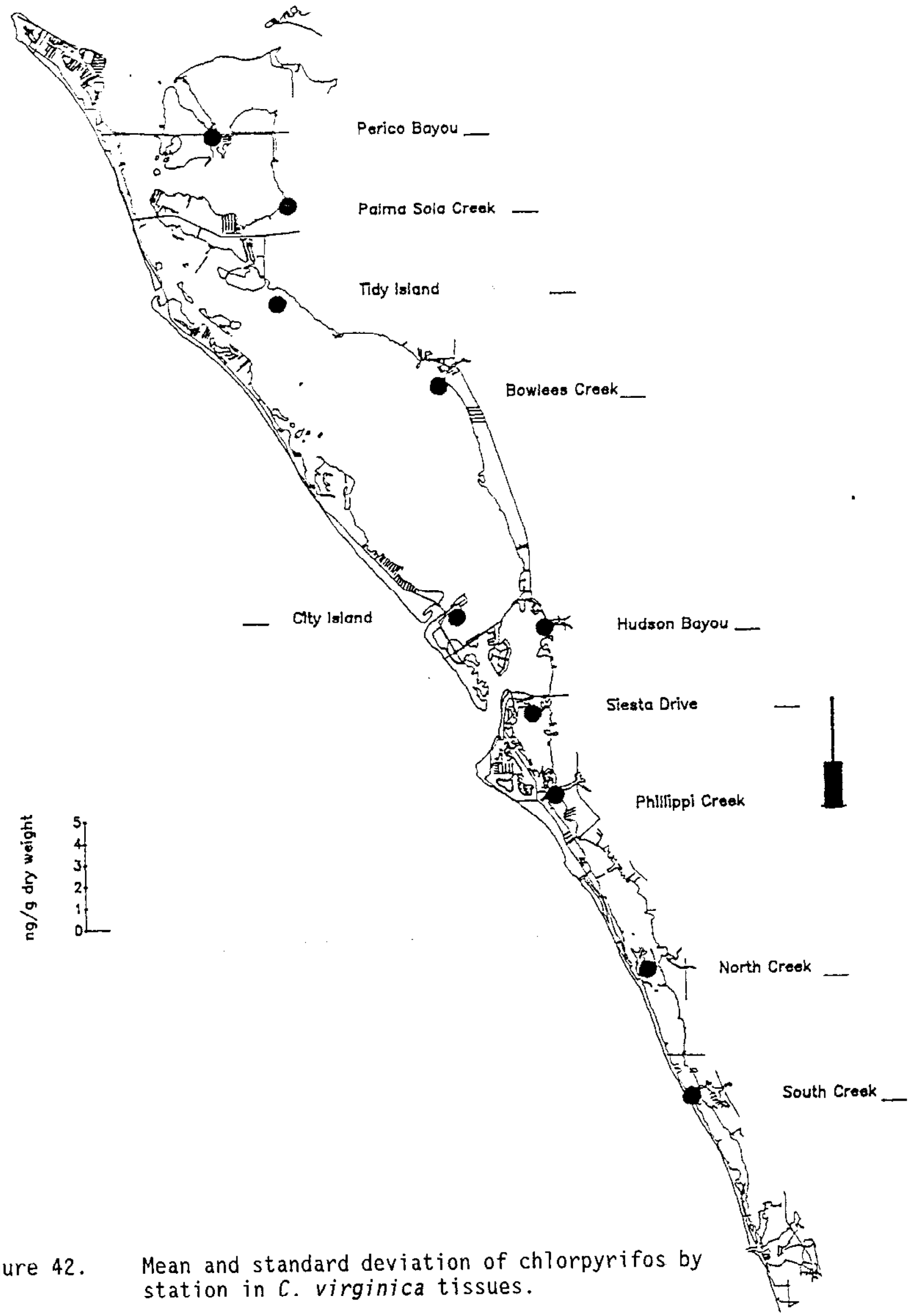

Figure 42. Mean and standard deviation of chlorpyrifos by station in *C. virginica* tissues.

Table 45. PAH Concentrations (ng/g dry wt.) in Sarasota Bay Shellfish, Mean of Duplicate Samples.

Sample	Species	Low Molecular Weight (LMW) PAH					High Molecular Weight (HMW) PAH		
		Naph-thalene	C_1-naph thalenes	C_2-naph thalenes	Fluorene	Phen-anthrene	Floranthene	Benzo(a) anthracene	Chrysene
SPRING 1990									
	Mercenaria spp.								
South Casey Key		Tr	Tr	Tr	Tr	ND	ND	ND	ND
	Crassostrea virginica								
City Island		ND	ND	ND	ND	Tr	ND	ND	ND
Hudson Bayou		ND	ND	ND	ND	Tr	ND	ND	ND
Siesta Drive		ND	ND	ND	ND	Tr	ND	ND	ND
FALL 1990									
	Mercenaria spp.								
Blackburn Bridge		ND	ND	ND	ND	ND	Tr	ND	ND
Coquina Beach		ND	ND	ND	Tr	Tr	ND	ND	ND
Long Bar		ND	ND	ND	ND	ND	Tr	ND	ND
New Pass		ND	ND	ND	Tr	ND	ND	Tr	ND
Palma Sola Creek		ND	ND	ND	ND	Tr	ND	ND	Tr
Phillippi Estuary		Tr	ND	ND	ND	ND	ND	ND	ND
	Crassostrea virginica								
Bowlees Creek		Tr	Tr	Tr	ND	ND	ND	ND	ND
Perico Bayou		ND	ND	ND	ND	ND	ND	ND	Tr
Tidy Island		ND	ND	ND	Tr	ND	ND	ND	ND

ND = Not detected: limit of detection = 15 ng/g dry wt.

Tr = Trace (below limit of quanititation) $\leq$ 50 ng/g dry wt.

samples from City Island, Hudson Bayou and Siesta Drive contained only trace amounts of phenanthrene, indicative of combustion-derived PAH. However, none of the other higher molecular weight PAH were detected, nor were any low molecular weight compounds.

Shellfish samples collected during the fall exhibited a greater diversity of PAH at a greater number of stations, as was observed for the chlorinated pesticides. However, all concentrations were still below levels of quantification. Clams from five different sites (Blackburn Bridge, Coquina Beach, Long Bar, New Pass, and Palma Sola Creek) exhibited high molecular weight PAH with no naphthalene or alkyl substituted naphthalenes, indicating primarily combustion and stormwater (crankcase oil) sources. Clams at the Phillippi estuary station contained only trace amounts of naphthalene, or primarily petroleum derived.

During the fall, oysters from Bowlees Creek contained trace amounts of naphthalene and alkyl substituted naphthalenes, indicative of small amounts of fuel oil. One of the oyster samples from Perico Bay exhibited trace amounts of chrysene and one near Tidy Island contained trace amounts of fluorene.

VI.D.3. Summary - Toxic Organics

In summary, these results indicate that oysters and clams from the majority of the sites sampled throughout Sarasota Bay in spring and fall 1990, did not contain sufficient pesticides to indicate substantial ecological or human health risks, yet low levels of chlorinated pesticides were evident.

Notable concentrations of p,p'-DDE were found in oysters collected from the mouth of Phillippi Creek during the spring and from Hudson Bayou in the fall, with lower concentrations of dieldrin, chlordane, BHC and the organophosphate pesticide, chlorpyrifos. Phillippi Creek represents the largest watershed basin within the study area and the loadings are high. The Hudson Bayou watershed, while small, is highly developed with both residential and commercial areas and was also exceptional for metal contaminants in oysters.

The predominance of DDE is indicative of long-term contamination from DDT pesticides applied to the watershed more than 20 years ago. Periodic disruption and erosion of contaminated soil or resuspension of contaminated sediments would cause the inconsistent pattern of minute amounts of DDT metabolites to be found. The presence of the relatively labile organophosphate pesticide, chlorpyrifos, reflects input from pesticides currently in use within the watershed area. Due to the low precipitation throughout 1990, and correspondingly lower inputs of stormwater to the Sarasota Bay system, the tissue analyses presented here do not represent a maximum contaminant scenario. It is quite possible that during a wetter season or immediately following a major rainfall, influx of current-use pesticides in stormwater runoff would be greater and tissue concentrations proportionally greater.

The low concentration of PAH detected in Sarasota Bay shellfish indicates no chronic petroleum or pyrogenic contamination problems at the sampled sites and indicate a broad geographic distribution of trace amounts of PAH. These levels of PAH apparently pose little substantial ecological or human health risk. A few sites exhibited trace amounts of petroleum-derived PAH, probably from small spills during refueling or from contaminated bilge water.

Sarasota Bay shellfish PAH were derived primarily from pyrogenic sources, rather than from direct input from petroleum products. Primary sources would include atmospheric deposition of PAH-containing particles from automobile and boat engine exhaust, coal and oil combustion industrial processes and forest fires, as well as used crankcase oil washed into the estuary with stormwater runoff. Since many of the PAH in estuaries come from stormwater runoff, a better understanding of the impact from runoff could be gained from monitoring the PAH composition of stormwater at select runoff sites and at select shellfish beds following a major rain event.

Because of dry conditions throughout 1990, the environmental conditions represent a minimum case scenario for stormwater-derived contaminants. In general, the results are indicative of estuarine environments with no consistent, widespread influx of petroleum contamination. PAH analyses in shellfish collected at periodic intervals following a major rain event are needed to provide a better assessment of stormwater-derived PAH in shellfish.

VII. PROTECTION, REESTABLISHMENT, AND ENHANCEMENT

VII.A. Protection of Resource

In evaluating protection and enhancement measures for bivalved shellfish, a number of threats to the resource can be identified immediately. Some threats have a direct human link, i.e., toxic contaminants, gross bacterial contamination, alteration in timing or quantity of freshwater flows, nutrients, and/or particulate loads, or overharvesting. Other threats to a managed resource may include catastrophic weather events (hurricanes, flooding, freezes), red tides, pandemic diseases or parasite infestation, and predator abundance. While there are little quantitative data on this subject, the case can also be argued that human influence may in some way reduce the ability of some organisms to combat disease, or may unduly favor opportunistic predators rather than the desired resource.

Managing the impacts of hurricanes, freezes, and disease or parasites have not been considered in this document. For severe weather events, human control is not possible; for diseases, there is a lack of knowledge of the severity of the impact in this area, the causal agents, and their requirements. In particular, vibriosis in juvenile shellfish is quite lethal but poorly understood. As vibrios appear to be endemic in the estuarine environment and as their ecological requirements are poorly understood, control of these bacteria is limited at this time to reducing human exposure.

The issues of toxic contaminants, altered flows, loadings, and overharvesting, however, are pertinent issues for the National Estuary Program and control measures may be more readily designed and implemented.

VII.A.1. Toxic Contaminants and Bacterial Contamination

To effect the protection of shellfish from toxic contaminants is simple in concept but has proved difficult in execution. Reducing the escape of toxic compounds into the environment and their eventual transport to the estuarine environment will reduce the exposure of all marine organisms, not only filter feeding, and therefore most heavily affected, shellfish. This reduction, of course, is most readily applied to materials released as a result of human activities and does not address the toxic compounds which result from 'natural' events such as forest fires, or bacterial contamination from wildlife.

When setting toxic contaminant reduction goals, there is also a severe lack of information on target levels which are desireable, synergism between compounds, and species-specific ecological susceptibilities. There is a decided lack of information on the ecological effects of specific compounds when measured as tissue concentrations. While there is more information on acute and chronic toxicological effects of the various compounds in both sediments and water, organisms vary in susceptibility by life stage, and sediment physical and mineralogical parameters affect the bioavailability of the many toxic materials.

As a result, there is no readily definable, quantitative link between watershed loading, water column, sediment, or tissue concentrations and ecological health of a population. Control of point and non-point sources and subsequent improvement in quality of water reaching the estuary may well be beneficial to shellfish (and other organisms) and should also reduce the tissue concentrations of contaminants.

Control of particulates and suspended solids (with which most contaminants are associated) will be the most effective technique in reducing loads of toxic materials to the estuary. Bacterial populations are also typically associated with suspended materials. Any eventual reduction in bacterial contamination to legal requirements and subsequent opening of shellfishing beds can be evaluated from an economic or recreational standpoint. It will be difficult, however, to translate any load reductions of toxic compounds into an increase or improvement in the shellfish population, or into a cost-benefit framework, if tissue levels are already below the statutory limits.

Other approaches have included the establishment of sediment criteria for biological protection. The criteria can be statistically based, for example not exceeding the 75th or other percentile of all sediment concentrations in some reference data base. Alternatively, the criteria could be biologically based, using a specific organism response as a threshold. These criteria might be applied bay-wide or may be restricted to areas designated for shellfish harvesting, recruitment areas, or seed beds. Incidentally, human consumers might also receive additional protection if sediment concentrations and shellfish tissue concentrations are monitored and controlled.

Biologically based criteria might use the apparent effect threshold technique (AET) employed in Puget Sound (PTI, 1988), which identifies the species-specific sediment concentration of a particular chemical above which statistically significant biological effects are always expected. The bioassay could also be focused on a particular life stage of the species of concern, developing techniques which identify the acute sublethal effects for the presumably more sensitive larval *Crassostrea* and *Mercenaria.* This approach is obviously easiest to define for acute toxic effects and much more difficult for population parameters.

Certainly from this study, one can identify areas of the Bay in which shellfish have higher tissue concentrations of selected contaminants and therefore would be likely to benefit more from load reduction activities or establishment and enforcement of sediment criteria.

For the overall Sarasota Bay area, pesticide and PAH concentrations were quite low, as were the fecal indicator bacteria and vibrios. For the organics, tissue concentrations were also erratic, implying an event-associated source of these materials. The appearance of chlorinated compounds which have not been in use in many years (and their metabolites) would indicate that older sediments have been resuspended and incorporated into shellfish tissues. The appearance of the more labile compounds would indicate a new application of pesticide which could perhaps be controlled through either better application practices, improved detention of runoff, or both. There were a number of watersheds noted for the occurrence of these compounds, but again Hudson Bayou and Phillippi Creek reported some of the highest tissue concentrations.

Overall, bacteriological parameters were also low. No sample exceeded the fecal coliform standard for shellfish tissues (230 per 100 g wet tissue), despite the fact that numerous stations had greater than 14 fecal coliform per 100 ml in the water column. No one area consistently demonstrated elevated coliforms, which would indicate that there was no regular effluent source near the twenty stations sampled. In the absence of domestic wastes, it is to be expected that the presence of these fecal indicator organisms would be highly correlated with rainfall and runoff amounts. These conditions were unfortunately not sampled during this study, but the control, detention, and particulate reduction recommendations in runoff would serve to control these bacteria as well.

Although human dose response information is lacking, the counts observed for vibrios during this study were substantially lower than those reported for major commercial shellfishing areas. Based on this, the human impacts of vibrios might not have been substantial during this study, but as the etiology of vibriosis on juvenile shellfish is unknown, their ecological impact cannot be assessed.

The most geographic variation in tissue concentrations appeared in the metals data and so offers the most options for focussing management efforts, although the ecological impacts of tissue metal concentrations are poorly understood. For clam tissues, those from the Phillippi estuary, followed by Selby Gardens, and Bishops Point were the highest in metals overall, using an unweighted ranking technique. The relatively high rank of Bishops Point is the product of the arsenic concentrations at this station. In light of the relatively low toxicities of the organic arsenic present in shellfish, and the lack of a tributary nearby, this area should be ranked far behind Phillippi Creek and Hudson Bayou for attention.

For oyster tissue, Hudson Bayou, followed by Phillippi Creek and South Creek were the stations with the highest unweighted ranking of metals. The co-occurrence of Hudson Bayou and Phillippi Creek in the top ranked of both sets of tissue data does not appear to be coincidental, as preliminary sediment data from these tributaries (MML, 1992) indicate that they are heavily enriched in many metals. These basins are also heavily urbanized and lead, zinc, and copper tissue concentrations are elevated with respect to other Florida sites in the National Status and Trends Program. The control of non-point source runoff would serve to reduce the loadings to the tributaries and subsequent tissue concentrations. This effort would be most effective in the Hudson Bayou and Phillippi Creek watersheds.

VII.A.2. Flow Alterations

Freshwater flow and its timing is more directly linked to oyster ecology than to clams, because of the known benefits of *Crassostrea* predator control in reduced salinity environments. Undoubtedly, during historical times, freshwater flows to the Bay were in general less variable, as larger wetland and pervious areas provided for runoff attenuation and the controlled delivery of higher base flows to the estuary. Increasing impervious areas increases the speed with which runoff occurs and freshwater pulses occur on a short term basis. Salinity gradients developed under these conditions are short term and usually of smaller spatial extent as the 'slug' of freshwater forces the estuarine waters to recede or flows out in a sheet over the top of the denser salt water. The lower base flows available under these conditions then maintain a smaller salinity gradient between storm events, as base flows are reduced from pre-impervious conditions.

Physical alterations to channels, such as the dredging of the Intracoastal Waterway through the extensive oyster bars in Roberts and Little Sarasota Bay, also increase flushing. This would serve to transport freshwater from the area more rapidly, reducing the extent of the salinity gradient, increasing salinity, and permitting the increased invasion of oyster predators. Removal of existing bars, event if senescent, should be discouraged from the standpoint of preventing further flow or flushing alterations. In addition, reef removal would reduce the available preferred substrate for oyster spat settlement.

Any action which could return the flow regime to a more unaltered state would also likely benefit oyster populations, providing nutrient content was sufficiently low to prevent eutrophication problems under the increased base flow conditions. In addition, the actions taken to reduce peak flows and increase base flows (runoff detention) are typically those which are used to control non-point source loads and improve water quality in stormwater. The solids removals effected by these methods would also serve to protect oyster reefs from heavy siltation and would improve the quality of substrate for spat settlement.

VII.A.3. Overharvest

Protection of shellfish populations from overharvesting may include either reduction of harvest pressure through either access or harvest restrictions, or activities to increase the amount of the resource. Increasing access may initially decrease harvest pressure at any one site by distributing participating harvesters over a broader area, but also could encourage additional recreational participants for a net increase in harvests.

Harvest pressure on shellfish for recreational consumption appears to be low in the Bay area. While consisting of incidental observations only, no clammers were observed in the only conditionally approved shellfishing area within the study bounds. In addition, this region on the western side of Sarasota Bay recorded some of the highest clam densities per survey effort of anywhere in the study area. Small clams (<50 mm) were also found in this area, which would indicate that legal harvesting pressure for consumption of clams does not reduce the population below that in other areas of the Bay, or prevent (through the removal of reproductive adults) the occurrence of smaller individuals. (If recruitment is higher in this region from environmental factors, this generalization may not hold for other regions of the Bay if they are reclassified in the future.)

The only conditionally approved waters are difficult to reach without a boat and lack of shore access undoubtedly plays a role in the lack of harvest pressure. Clammers were observed in other regions, but these areas appeared to be related more to easy access than to classification of waters. Oysters are also almost uniformly below legal size (3 inches) and the effort to (illegally) collect and shuck small individuals for consumption may deter most recreational harvesters. Other evidence for low harvesting pressure is that no clams were reported during any of the 3,010 interviews conducted with either shore or boat based anglers under the Fisheries Resource Assessment Project of the Sarasota Bay National Estuary Program.

One aspect of shellfish harvesting which is difficult to assess may be the harvest pressure that recreational anglers exert on clam populations through their use as bait for finfishing. This pressure should be most closely related to easy shore access, and increasing shore access around the Bay may serve to increase interest and therefore pressure on this resource. An examination of densities of clams (no clams, 1-7 clams, 8-21 clams, etc.) as a function of public access points, however, demonstrated no consistent pattern which would indicate that populations are reduced as a function of harvesting by shore based shellfishermen. Those areas with no clams reported appeared instead to reflect regional environmental conditions, and there is no obvious justification for reducing access to protect populations.

VII.A.4. Enhancement

Increasing shellfish populations to generate more recreational opportunities may be feasible, but one may question whether the public interest in the activity is present based on the apparently low harvest pressure. Increased *Mercenaria* densities may spur public interest, but the lack of shore access to the conditionally approved waters is a problem that is difficult to surmount and limits the numbers of individuals who could participate in this activity. Increasing access may al so place additional stresses on the fringing seagrass beds of Sarasota Bay.

Any direct enhancement activity undertaken for recreational purposes should preferably be carried out in conditionally approved waters. This could include such activities as seeding of small clams, although some

work has indicated that seeding merely supports a increased population of predators. The approved area, however, also encompasses a portion of one of the largest contiguous grassbeds within the study area and the seagrass destruction which accompanies clam harvesting in grassbeds reduces the desirability of enhancing the interest in this fishery. Throughout the Bay, clams were found in grassy areas and increased traffic, both boat and pedestrian, would be similarly deleterious.

Increasing opportunities for recreational oyster gathering appear to be unfeasible, due primarily to the salinity requirements, the small size of the existing individuals, and the bacterial counts which may be expected near the tributaries with more optimal salinities. The small size of the existing individuals appears to be the result of the intertidal growth habits, which could in itself be the result of predation pressures and less favorable salinity regimes.

Cultch placement along tributary shorelines, in the absence of improvements in flow regimes, is quite feasible, but bacterial counts will likely continue to exceed sanitary criteria during rainfall events, even if non-point loadings are reduced substantially. The availability of harvestable oysters would likely remain low unless 1) salinity regimes were radically restored, 2) subtidal growth habits were encouraged, 3) non-point source loadings were reduced, 4) regions near tributaries reclassified for harvest, 5) sufficient monitoring was supported to guarantee sanitary quality.

While the advantages of increasing <u>recreationa</u>l opportunities for both *Mercenaria* and *Crassostrea* harvesting are minimal or feasibility slight, there are valid <u>ecological</u> reasons for supporting these populations which are consistent with the goals of the Sarasota Bay Program. As filter feeders, both clams and oysters have the theoretical potential to improve the water clarity of Sarasota Bay, particularly if water clarity impairments are linked to chlorophyll and therefore phytoplankton levels. These water quality improvements, while intuitive, are difficult to quantify except in a theoretical milieu, and may also not apply if the attenuating substances in the water column are primarily inorganic in nature or are not the preferred size of food organisms.

The large numbers of larvae produced by these shellfish also form a portion of the productivity of the estuary. Increasing the numbers of reproducing adults and consequently larvae would also support higher numbers of larval and juvenile predators which could include decapods and molluscs, as well as finfish. Any *Mercenaria* seeding or enhancements for *Crassostrea* spat settlement should be coupled with investigations to assess its success and subsequent predator relationships. Oyster reefs support a diverse invertebrate fauna which contribute to the estuary biomass and the reef itself serves as a habitat for juvenile fishes. In addition, the reefs themselves can act to reduce sediment resuspension by reducing wave energies. The ecological inducements provided by increased shellfish biomass, however, are difficult to predict and measure quantifiably, and make the economic benefits also difficult to assess.

The substrate stabilization afforded by oyster reefs is more amenable to economic valuation in that any reduction in erosion can be assessed against real property values. This may be particularly applicable in the southern portion of the study area, where oysters were historically the most prevalent, salinity regimes the most favorable, and where recreational boat traffic creates substantial wave climates in the narrow waterways. The results of the sediment surveys conducted by the Benthic Habitat Assessment Project of the Sarasota Bay Program may provide sufficient information for optimal areas of cultch placement. Coupled with small scale investigations for optimal timing and depth of placement of cultch, this enhancement measure could be readily accomplished.

VII.A.5. Commercial Harvests

The enhancement of the bivalved shellfishery to commercial levels for large numbers of fishermen is unlikely for all of the reasons detailed which make increases in the recreational fishery unlikely or undesirable. In addition to the small area of conditionally approved waters, this region of the Florida Gulf coast experiences a high incidence of red tides which prevent shellfish harvesting altogether. These toxic phytoplankton blooms also typically occur during the late summer and early fall, during oyster harvesting seasons. While one or two individuals may receive some commercial benefit from the regions *Mercenaria,* the uncertainty of possible harvest would prevent this from being a primary source of income.

Opportunistic harvesting of natural populations of *Mercenaria* on a commercial scale is also incompatible with the Program goals of improving water clarity to the maximum extent possible, due to the resuspension of sediments during tonging operations. Destruction of seagrass beds during this process should also remain a concern due to the velocity reduction afforded by dense beds, the improved settling obtained, and the subsequent improvement in water clarity.

Increases in commercial harvesting would only follow improvements in non-point source loading and water quality and subsequent expansion of approved waters by FDNR. Unfortunately the current economic situation and shrinking tax base experienced by the State of Florida has forced the Department to even consider the elimination of the office responsible for shellfish sanitation in this area. While sanitary monitoring may be conducted by other sections of the agency, or even potentially supported by the commercial harvesters themselves, an expansion of approved waters in the near future is unlikely.

Aquaculture activities remain a possibility, but the difficulties associated with obtaining bottom leases of subtidal State lands for this purpose are many. The current non-point source loading and bacterial counts of the area, the small area currently approved, and the high recreational uses of the area make obtaining a lease even more unlikely. The uncertainties associated with harvests (red tides, high recreational traffic, potential vandalism, and poaching) make any monetary investment highly uncertain.

VII.B. Human Consumer

In the Sarasota Bay study area, the coliform standard in shellfish tissues (230 per 100 g) was not exceeded while the standard for the water column (14 per 100 ml) was violated in several instances. For coliform< at least, this would appear to imply that the more readily and more frequently applied water column criteria are more conservative than the actual tissue standards and would effectively allow for some increase in bacterial counts of tissues during transport and processing. Based on a long history of epidemiological work and reduced instances of shellfish related illnesses, these standards are apparently effective at controlling most bacterial diseases of fecal origin. The utility of the fecal coliform standard with regards to viral contaminants has been previously discussed and its limitations are well known.

Less is known about the *Vibrio* species. *Vibrio* spp. and *Aeromonas* spp* counts bear little if any correlation to coliform numbers, and the factors controlling the presence of these endemic estuarine organism are little known. The presence of these organisms in Sarasota Bay, however, is well below that reported for major shellfishing areas, and it is possible that these organisms do not pose a substantial threat to any but susceptible individuals with certain types of existing blood, liver, or immunological disorders. The relatively tedious enumeration process for the *Vibrio,* the lack of dose-illness information, and the difficulty of determining virulence of the isolated strains dictate that control of human exposure to these organisms be conducted chiefly through continued education efforts, aimed particularly at individuals at risk. The Marine Extension agents offer brochures on seafood safety addressing these matters, but this education could be expanded to include selected categories of physicians, or could be incorporated into the recreational fishing license or boat registration programs.

More extreme measures could include the development of a recreational shellfishing licensing program for distribution of information and generation of revenue, information made available at public access points, and restricting the harvest of *Mercenaria* during warm months when *Vibrio* counts are expected to be high. These first of these measures is certain to be unpopular and does not seem justified in view of the low *Vibrio* counts observed.

Beyond these measures, the activities of FDNR, the sanitary surveys and classification of waters based on fecal coliform and red tides, appear to more than adequately protect the human consumer of shellfish from bacterial contaminants. The area approved, while small in relation to the study area as a whole, undoubtedly errs on the side of conservatism, to allow for continued safety during all but the most extreme rain events. Expansion of classified waters would require additional resources and more frequent monitoring, which, at this time at least, do not appear to be available from FDNR and which may not be justified based on the apparently low demand for recreational shellfishing.

With the relative lack of industrial effluents in Sarasota Bay, the coliform standard and resulting classification of waters appears to be effective in limiting human exposure to toxic contaminants as well. The lack of wet season and therefore potentially worst case data during this study, however, should be recognized. The most contaminated sites in the study area were in areas currently unapproved or closed to shellfish harvesting and in areas adjacent to tributary mouths. Some of the tissues examined during this study were higher than many other sites in Florida for copper, lead, and zinc (contaminants known for their association with urban runoff).

While formal health risks were not performed, evaluation of tissue concentrations of metals against daily intake maxima from a variety of agency and scientific sources and assumed consumption rates indicated that moderate consumptions generally constituted less than 20 percent of maximum acceptable daily intake (ADI) rates. The exception to this statement was for arsenic, of which a 5 g/day consumption rate exceeded 100 percent of the maximum ADI. The ADI's for this element are based upon inorganic compounds, however, and the forms of arsenic found in fish and shellfish are reported to be substantially less toxic.

Aside from arsenic, most exposure (as a percentage of ADI) comes from zinc in oysters (20 percent). Eliminating oysters from consumption and ignoring arsenic, clam consumption could constitute 12 percent or less of the ADI for the metals studied. While these estimates may appear low, it should be reemphasized that this study was not a formal risk assessment. Consumption rates are estimates only, the remainder of the individuals' diet is not accounted for, and that ADI are not specific for more sensitive risk groups such as pregnant women. In addition, data were lacking from wet season and potentially worst case conditions.

Pesticides and polycyclic aromatic hydrocarbons were generally present in extremely low levels, if at all, at most sites in Sarasota Bay. Selected stations, however, during some periods were comparatively high in relation to nationwide NOAA Status and Trends data, particularly for DDT and metabolites. The erratic nature of their presence at the same site indicates that loadings are intermittent and associated with either resuspension of older contaminated sediments or with new applications of approved compounds. No shellfish exceeded FDA criteria for any regulated compound, and given assumed consumption rates, oral reference doses, where effects were known to occur, were not exceeded either. Again, no formal health risk assessment was performed.

As for metal contaminants, the closure of shellfishing waters for bacterial reasons appears to effect protection of human consumers from substantial toxic organic exposure. Data were lacking, however, from wet season and high runoff conditions.

It is possible that non-point source controls implemented in the future could reduce the bacterial loading to the estuary and allow FDNR to expand the extent of conditionally approved waters. These waters would likely be opened or closed based on rainfall amounts received in the watershed, in addition to red tide outbreaks. While reduction in new loadings will reduce both new contributions of metals and toxic organics, the material already associated with the sediments in the system may continue to produce elevations in tissue concentrations.

In the event that approved waters are expanded, it may be appropriate at that time to conduct a formal health risk assessment of either the tissue levels determined during this study, or with new tissues with presumably lower concentrations of contaminants. At that time as well, other toxic compounds not examined during this study (such as PCB's) may be examined. In any event, control of the risk to human consumers would be most likely accomplished through restricting access to either the most affected geographical regions or the most affected species (oysters).

Depending on the results of this future health risk assessment for toxic compounds, it may be necessary to geographically restrict the harvest in FDNR approved waters to protect human consumers from unacceptable risks. Other harvest limitation techniques either seasonally or species could also reduce human exposure, but offer their own difficulties with public support, education, and enforcement.

VII.C. Research Needs and Recommendations

Research needs which have developed from the execution of this study are presented below, without regard to either prioritization, timing, or costs.

The vagaries of weather prevented the collection of wet season and potentially worst-case tissue concentrations. While there was little seasonal variation on the collected data set which would indicate seasonal physiological responses of the organisms or differing bioavailability based on temperature, the relative lack of rain during this year may have prevented the typical wet season loadings from being delivered to the estuary. This information is still worthwhile to examine what may be the maximum concentrations of tissue contaminants.

The ecological impacts of tissue and sediment contaminants, together with the synergistic effects of a multitude of toxic compounds are poorly known for these species. Much information is available on acute and chronic (96 hour) toxicity to various life stages, but species specific data on the effects of sediment contaminants on larval settling, on reproductive success, and other physiological parameters which may affect population success are lacking. While the Apparent Effect Threshold approach and data developed for other species in the Puget Sound region could be initially applied for approximate information, more species-specific information is needed if resource enhancement measures are seriously contemplated.

Initial results from the sediment monitoring tasks of the Sarasota Bay Program should also be examined to determine if any obvious relationship between sediment and tissue concentrations exist. These relationships are frequently obscured by differing sediment characteristics which result in varying bioavailability, but if a relationship could be found for the restricted area of Sarasota Bay, then sediment analyses might prove to be a useful surrogate for tissue concentrations.

Preliminary information on contaminant loading from the non-point source assessment in Sarasota Bay indicate that substantial amounts of airborne metals (zinc in particular) could be reaching the estuary through routes not typically considered. Aerial deposition is also reported to play a major role in the transport of pyrogenic polycyclic aromatic hydrocarbons. An evaluation of the airborne loads in relation to surface runoff would indicate whether conventional non-point source controls (retention, detention, other surface water management strategies) would achieve significant reductions, or whether control of airborne sources should be considered.

For relating human exposure to actual risks of disease, a formal health risk assessment could be conducted. Similar to the lack of information on the ecological impacts of chronic levels of contaminants, risk assessment is hampered by lack of quantitative relationships between water and sediment concentrations and tissue levels, lack of relationships of pathological evidences to tissue concentrations of selected contaminants, lack of knowledge regarding the applicability of data for one species to others.

A risk assessment would be particularly necessary, however, if FDNR expands the approved shellfishing waters to include those near the mouths of the tributaries, and could be based either on the tissue data in this study, or upon new data gathered under presumably improved conditions. Other classes of known toxic compounds could be evaluated at this time. As a portion of the risk assessment, the actual recreational consumption should be quantified. A portion of this recreational survey could also include the harvest pressure on shellfish for use as fishing bait, and the sum of the two harvests could be employed in an evaluation of population dynamics.

Population dynamics of both *Crassostrea* and *Mercenaria* are not well understood. Recruitment rates are reported to be highly erratic and may be a function of environmental variables (of either the water column or the sediments/substrate) coupled with physiological requirements. Currents also play an undoubted role in larval distribution. The degree of predation and harvest pressure which the various ages of a stable population can support is difficult to assess, but of interest in managing this resource. Comparison of these parameters with other areas, preferably those regions supporting larger fisheries, will be essential to evaluate the potential success of seeding, or substrate enhancement.

Cultch placement, like seeding, would require some initial small scale investigations to determine optimum times and depth of placement, together with optimal sediment types to prevent settling, burial, or fouling of the newly placed material. Other activities related to the oyster resource would be to update the spatial mapping of this resource. Much of the information included in the Sarasota County Habitat Trend Analysis (Mangrove Systems, 1988) on oyster reefs appears outdated, and the Manatee County portion of the study area is unmapped. While this project was to update the condition of the oyster resource by identifying damaged areas and noting relative health and condition, an actual mapping effort was not planned. This could be readily accomplished and routine surveys of condition would allow trends in oyster health and abundance to be monitored.

The limitations that salinity regimes are known to place upon oyster populations and their predators would also justify the inclusion of paleoenvironmental determinations. Oyster growth habits and shell morphology vary with the degree of intertidal exposure, which appears controlled by increasing predator response to increased salinities. If flow alterations (particularly in the Phillippi Creek area) have played a major role in decreasing the oyster resource in this area, then examination of shell morphologies from both current organisms and from those gathered from Indian shell middens could provide information on historical salinity regimes. These older shells are known to exist and are already radio-carbon dated, thus making this effort very inexpensive. The resulting information may likely support the restoration of more 'natural' flow regimes to the estuary.

VII. LITERATURE CITED

Abbott, R.T. 1974. American Seashells, 2nd ed. Van Nostrand Reinhold Company, New York. 663 pp.

Agency for Toxic Substances and Disease Registry (ATSDR). 1987. Toxicological profile for aldrin/dieldrin (draft). ATSDR, Oak Ridge National Laboratory. U.S. Public Health Service.

Agency for Toxic Substances and Disease Registry. 1989. Toxicological profile for arsenic. ATSDR/TP-88/02. Prepared by Life Systems, Inc. for ATSDR, U.S. Public Health Service in collaboration with U.S. Environmental Protection Agency. 125 pp.

Ahsanullah, M. and T.M. Florence. 1984. Toxicity of copper to the marine amphipod *Allorchestes compressa* in the presence of water- and lipid-soluble ligands. Mar. Biol. Berlin. 84:41.

Allan, R.J., A.J. Ball, V.W. Cairns, G.A. Fox, A.P. Gilman, D.B. Peakall, D.A. Piekarz, J.C. Van Oostdam, D.C. Villeneuve and D.T. Williams (eds). 1991. Toxic Chemicals in the Great Lakes and Associated Effects. Volume II: Effects. March. Environment Canada, Department of Fisheries and Oceans, and Health and Welfare, Canada. 755 pp.

Alzieu, C. and M. Heral. 1987. Ecotoxicological effects of organotin compounds on oyster culture. *In:* Ecotoxicological testing for the marine environment. Vol. 2 (G. Persoone, E. Jaspers and C. Claus, eds.). State University Ghent and Institute Marine Scientific Research, Belgium. 187 pp.

American Public Health Association. 1985a. Standard methods for the examination of water and wastewater. Wash. D.C.

American Public Health Association. 1985b. Recommended procedures for the bacteriological examination of seawater and shellfish. 5th edition (A.E. Greenberg and D.A. Hunt, eds.). American Public Health Association, Washington, D.C. pp. 37-63.

Amiard, J.C., C. Aimard-Triquet, B. Berthet and C. Metayer. 1987. Comparative study of the patterns of bioaccumulation of essential (Cu, Zn) and non-essential (Cd, Pb) trace metals in various estuarine and coastal organisms. J. Exp. Mar. Biol. Ecol. 106:73.

Anderson, R.D. and J.W. Anderson. 1975. Effects of salinity and selected petroleum hydrocarbons on osmotic and chloride regulation of the American oyster, *Crassostrea virginica.* Physiol. Zool. 48(4):420-430.

Anderson, W.D., W.J. Keith, F.H. Mills, M.E. Bailey and J.L. Steimeyer. 1978. A survey of South Carolina's hard clam resources. SC Wildl. Mar. Resour. Dept., Mar. Resour. Ctr., Technical Report 32. 17 pp.

Ansell, A.D. 1968. The rate of growth of the hard clam, *Mercenaria mercenaria* (L.) throughout the geographical range. J. du Conseil. 31:364-409.

Ansell, A.D. and K.F. Lander. 1967. Studies on the hard-shell clam, *Venus mercenaria,* in British waters. III. Further observations on the seasonal biochemical cycle and on spawning. J. Appl. Ecol. 4:425-435.

Arnold, W.S. Review summary of the hard clam, *Mercenaria* spp. in Florida. Unpublished report, Florida Department of Natural Resources.

Arnold, W.S. 1984. The effects of prey size, predator size, and sediment composition on the rate of predation of the blue crab, *Callinectes sapidus* Rathbun, on the hard clam, *Mercenaria mercenaria* (Linne). J. Exp. Mar. Biol. Ecol. 80:207-219.

Arnold, W.S., D.C. Marelli, T.M. Bert, D.S. Jones and I.R. Quitmyer. 1991. Habitat-specific growth of hard clams *Mercenaria mercenaria* (L.) from the Indian River, Florida. J. Exp. Mar. Biol. Ecol. 147:245-265.

Baden, D.G. 1983. Marine food-borne dinoflagellate toxins. Int. Rev. Cytol. 82:99.

Baross, J. and J. Liston. 1970. Occurence of *Vibrio parahaemolyticus* and related hemolytic *vibrios* in the marine environment of Washington State. Appl. Microbial. 20:179-186.

Barrick, R.C. and F.G. Prahl. 1987. Hydrocarbon geochemistry of the Puget Sound region. III. Polycyclic aromatic hydrocarbons in sediments. Est. Coastal Shelf. Sci. 25:175.

Bayne, B.L., D.A. Brown, K. Burns, D.R. Dixon, A. Ivanovici, D.R. Livingston, D.M. Lowe, M.N. Moore, A.R.D. Stebbing and J. Widdows. 1985. The effects of stress and pollution on marine animals. Praeger.

Beach, R. and R. Hittinger. 1989. Trace metals in quahog clams from Narragansett Bay. Report #NBP-90-26 to the Narragansett Bay Project by Thibault/ Bubly Associates. Providence, R.I. 30 pp.

Bender, M.E., W.J. Hargis, Jr., R.J. Huggett and M.H. Roberts, Jr. 1988. Effects of polynuclear aromatic hydrocarbons on fishes and shellfishes: An overview of research in Virginia. Mar. Environ. Res. 24:237-241.

Bender, M., D. Kester, D. Cullen, W. King, S. Bricker and W. Miller. 1989. Distribution of trace metals in the water column, sediments and shellfish of Narragansett Bay. Report #NBP-89-25 to the Narragansett Bay Project. 118 pp.

Blake, N.J. and G.E. Roderick. 1983. "Correlation of Coliform Bacteria with Vibrios in Apalachicola Bay." pp 17-19, *In:* Apalachicola Oyster Industry: Conference Proceedings (S. Andree, ed.). Apalachicola, FL.

Boehm, P.D. 1983. Coupling of particulate organic pollutants between the estuary and continental shelf and the sediments and water column in New York Bight region. Can. J. Fish. Aquat. Sci. 40:262-276.

Boehm, P.D., D.L. Feist, D. MacKay and S. Paterson. 1981. Physical-chemical weathering of petroleum hydrocarbons from the Ixtoc-I blowout chemical measurements and a weathering model. *In:* 1981 Oil Spill Conf. Proc. Pub. #4334, American Petroleum Institute, Washington, DC.

Boehm, P.D. and J.G. Quinn. 1977. The persistence of chronically accumulated hydrocarbons in the hard shell clam, *Mercenaria mercenaria.* Mar. Biol. 44:227-233.

Boyden, C.R. 1975. Effect of size on metal content of shellfish. J. Mar. Biol. Assoc. U.K. 57:675.

Bricelj, V.M., J.H. Lee, A.D. Cembella and D.M. Anderson. 1990. Uptake of *Alexandrium fundyense* by *Mytilus edulis* and *Mercenaria mercenaria* under controlled conditions. pp. *269-274, In:* Toxic Marine Phytoplankton (E. Graneli, B. Sundstrom, L. Edler and D.M. Anderson, eds.). Proc. 4th Intrntl. Conf. Toxic Mar. Phytopl. New York: Elsevier/North Holland.

Broutman, M.A. and D.L. Leonard. 1988. The quality of shellfish growing waters in the Gulf of Mexico. National Oceanic and Atmospheric Administration, Strategic Assessments Branch. Rockville, MD.

Brown, H.S., R. Goblee and L. Tatelbaaum. 1990. Methodology for assessing hazards of contaminants in seafood. Final Report to Narragansett Bay Project for the year 1986-1987. NBP-90-37. Narragansett, RI. 49 pp.

Bryan, G.W. 1976. Heavy metal contamination in the sea. *In:* Marine Pollution (R. Johnson, ed.). Academic Press, London. 185 pp.

Burke, V., M. Gracey, J. Robinson, D. Peck, J. Beaman and C. Burdell. 1983. The microbiology of childhood gastroenteritis: Aeromonas species and other infective agents. J. Infect. Dis. 148:68-74.

Burrell, V.G., Jr. 1977. Mortalities of oysters and hard clams associated with heavy runoff in the Santee River system, South Carolina, in the spring of 1975. Proc. Natl. Shellfish. Assoc. 67:35-43.

Butler, P.A. 1966. Pesticides in the marine environment. Pesticides in the Environment and their Effects on Wildlife. J. Appl. Ecol. 3(Suppl.)243-259.

Butler, P.A. 1973. Organochlorine residues in estuarine mollusks, 1965-1972. National Pesticide Monitoring Program. Pt. I. General summary and conclusions. pp. 238-362, *In:* Residues in Fish, Wildlife and Estuaries. Pest. Monit. J. Vol. 6. Pt. II. Residue data - Individual States. Sect. C: Delaware: pp. 263-267. Pt. III. Sect. J: New York: pp. 303-315.

Butler, P.A., CD. Kennedy and R.L. Schutzmann. 1978. Pesticide residues in estuarine mollusks, 1977 vs. 1972. Pest. Monit. J. 12(3) :99-101.

Byrne, C. 1989. Effects of the water-soluble fractions of No.2 Fuel Oil on the cytokinesis of the quahog clam, *Mercenaria mercenaria.* Bull. Environ. Contam. Toxicol. 42:81-86.

Byrne, C.J. and J.A. Calder. 1977. Effects of water soluble fractions of crude, refined and waste oils on the embryonic and larval stages of the quahog clam *Mercenaria* sp. Mar. Biol. 40:225-231.

Calabrese, A. 1972. How some pollutants affect embryos and larvae of American oyster and hard-shell clam. U.S. Natl. Mar. fish. Serv. Mar. Fish Rev. 34(11-12):66-77.

Calabrese, A., J.R. MacInnes, D.A. Nelson and J.E. Miller. 1977. Survival and growth of bivalve larvae under heavy metal stress. Mar Biol. 41:179.

Camp Dresser & McKee, Inc. 1992. Sarasota Bay National Estuary Program point/non-point source loading assessment. Phase I. Draft. Sarasota, FL.

Canadian Department of Fisheries and Oceans (DFO). 1988. Fish inspection regulations, Amendment List, June 13. Ottawa.

Carriker, M.R. 1951. Ecological observations on the distribution of oyster larvae in New Jersey estuaries. Ecol. Monogr. 21(1):19-38.

Carriker, M.R. 1956. Biology and propagation of young hard clams, *Mercenaria mercenaria.* J. Elisha Mitchell Sci. Soc. 72:57-60.

Carriker, M.R. 1961. Interrelation of functional morphology, behavior, and autecology in early stages of the bivalve *Mercenaria mercenaria.* J. Elisha Mitchell Sci. Soc. 77:168-241.

Caspar, S.G. 1988. Lead, cadmium and other elements in domestic shellfish. Program Report.

Castagna, M. and P. Chanley. 1973. Salinity tolerance of some marine bivalves from inshore and estuarine environments in Virginia waters on the western mid-Atlantic coast. Malacologia. 12:47-96.

Clark, R.B. 1989. Marine Pollution 2nd Edition. Clarendon Press. Oxford.

Colwell, R.R., J. Kaper and S.W. Joseph. 1977. *Vibrio cholerae, V. parahaemolyticus* and other *Vibrios:* occurrence and distribution in Chesapeake Bay. Science. 198:394-396.

Committee on Evaluation of the Safety of Fishery Products. 1991. *In:* Seafood Safety (F.E. Ahmed, ed.). National Academy Press, Washington, DC.

Courtney, W.A.M. and G.R.W. Denton. 1976. Persistence of polychlorinated biphenyls in the hard-clam *(Mercenaria mercenaria)* and the effect upon the distribution of these pollutants in the estuarine environment. Envir. Pollut. 10:55-64.

Cummings, J. and A. Stevens. 1970. Investigations of *Gymnodinium breve* toxins in shellfish. Public Health Service Bulletin no. 3, U.S. Dept. Health Educ. & Welfare, Washington, DC.

Cunningham, P.A., J.M. McCarthy and D. Zeitlin. 1990. Results of the 1989 census of state fish/shellfish consumption advisory programs. Prepared for Office of Water Regulations and Standards, U.S. Environmental Protection Agency. Research Triangle Park, NC.

Dalton, R. and W. Menzel. 1983. Seasonal gonadal development of young laboratory-spawned southern *(Mercenaria campechiensis)* and northern *(Mercenaria mercenaria)* quahogs and their reciprocal hybrids in northwest Florida, U.S.A. J. Shellfish Res. 3:11-18.

Davis, H.C. 1958. Survival and growth of clam and oyster larvae at different salinities. Biol. Bull. (Woods Hole). 114:296-307.

Davis, H.C. 1960. Effects of turbidity-producing materials in sea water on eggs and larvae of the clam *(Venus (Mercenaria) mercenaria).* Biol. Bull. (Woods Hole). 118:48-54.

Davis, H.C. and A. Calabrese. 1964. Combined effects of temperature and salinity on development of eggs and growth of larvae of *M. mercenaria* and *C. virginica.* U.S. Fish Wildl. Serv. Fish. Bull. 63:643-655.

Deichmann, W.B. 1973. The Chronic toxicity of organochlorine pesticides in man. pp. 347-420, *In:* Pesticides and the Environment: A Continuing Controversy (W.B. Deichmann, ed.). Intercontinental Med. Book Corp., New York.

Dillon, R.T., Jr. and J.J. Manzi. 1989. Genetics and shell morphology in a hybrid zone between the hard clams *Mercenaria mercenaria* and *M. campechiensis.* Mar. Biol. 100:217-222.

Dillon, T.M. and A.B. Gibson. 1985. Bioaccumulation and effects on reproduction in aquatic organisms: An assessment of the current literature. Waterways Experiment Station, U.S. Army Corps of Engineers. Miscellaneous Paper D-85-2. Vicksburg, MS. 18 pp.

Dixon, L.K. 1992. Sediment contaminants in selected Sarasota Bay tributaries. Mote Marine Laboratory Technical Report # 242. Sarasota, FL. 12 pp.

Eisler, R. 1981. Trace metal concentrations in marine organisms. Pergamon Press, New York. 687 pp.

Eisler, R. 1987. Polycyclic aromatic hydrocarbon hazards to fish, wildlife, and invertebrates: A synoptic review. Biol. Rept. 85 (1.11). U.S. Fish and Wildlife Service. Washington, D.C.

Eisler, R. 1988. Arsenic hazards to fish, wildlife and invertebrates: a synoptic review. Biol. Rpt. 85(1.12). 92 pp.

Eisler, R., G.E. Zaroogian and R.J. Hennekey. 1972. Cadmium uptake by marine organisms. J. Fish. Res. Board Can. 29(9):1367-1369.

Elder, J.F. 1988. Metal biogeochemistry in surface-water systems: A review of principles and concepts. U.S. Geological Survey Circular 1013. Denver. 43 pp.

Eldridge, P.J. and A.G. Eversole. 1982. Compensatory growth and mortality of the hard clam, *Mercenaria mercenaria* (Linnaeus, 1758). Veliger. 24:276-278.

Engel, D.W. and B.A. Fowler. 1979. Copper and cadmium induced changes in the metabolism and structure of molluscan gill tissue. *In:* Marine pollution: Functional Responses (W.B. Vernberg, et al., eds.). Academic Press, New York, NY. 454 pp.

Engle, R.H., M.J. Neat and R.E. Hillman. 1972. Sublethal chronic effects of DDT and Lindane on glycolytic and gluconeogenic enzymes of the quahog, *Mercenaria mercenaria.* pp. 257-260, *In:* Marine Pollution and Sea Life (M. Ruivo, ed.). Mar. Poll. & Sea Life. Fishing News (Books) Ltd., London.

Epifanio, C.E. and R.F. Srna. 1975. Toxicity of ammonia, nitrite ion, nitrate ion, and orthophosphate to *Mercenaria mercenaria* and *Crassostrea virginica.* Mar. Biol. (Berl). 33:241-246.

Estevez, E.D. and D.A. Bruzek. 1986. Survey of mollusks in Southern Sarasota Bay, Florida, Emphasizing Edible Species. Tech. Report No. 102. 97 pp.

Estevez, E.D. and C.L.P. Palmer. 1990. A segmentation system for the Sarasota Bay Project National Estuary Program. Sarasota Bay NEP Report: SEG. Mote Marine Laboratory Technical Report No. 161. Sarasota, FL.

Estevez, E.D., J. Miller and J. Morris. 1984. A review of scientific information: Charlotte Harbor estuarine complex and the Peace River. Mote Marine Laboratory Technical Report submitted to the Southwest Florida Regional Planning Council.

Farrington, J.W. 1980. An overview of petroleum hydrocarbons in the marine environment. *In:* Petroleum in the Marine Environment. Advances in Chemistry Series No. 185, Chapter 1 (L. Petrakis and F. Weiss, eds.). American Chemical Society, Washington, DC.

Farrington, J.W. 1986. Fossil Fuel Aromatic Hydrocarbon Biogeochemistry in the Marine Environment: Research Challenges. pp. 113-142, *In:* Strategies and Advanced Techniques for Marine Pollution Studies: Mediterranean Sea (C.S. Giam and H.J.M. Dou, eds.).

Farrington, J .W., E.D. Goldberg, R. W. Risebrough, J.H. Martin and V.T. Bowen. 1983. U.S."Mussel Watch" 1976-1978: An overview of the trace metal, DDE, PCB, hydrocarbon, and artificial radionuclide data. Environ. Sci. Technol. 17:490-496.

Farrington, J.W., S.G. Wakeham, J.B. Livramento, B.W. Tripp and J.M. Teal. 1985. Aromatic hydrocarbons in New York Bight polychaetes: U-V.-fluorescence analyses and GC/MS analyses. Environ. Sci. Technol. (submitted).

Finucane, J.H. and R.W. Campbell, II. 1968. Ecology of American oysters in Old Tampa Bay, Florida. Quart. J. Fla. Acad. Sci. 31(1):37-46.

Fish and Wildlife Service. 1984. Habitat Suitability Index Models: Hard Clam., U.S. Dept. of Interior, FWS/OBS-82/10.77. 21 pp.

Florida Department of Natural Resources. 1986. Summary of Florida Commercial Marine Landings. Tallahassee, FL.

Florida Department of Natural Resources. 1991. Rule chapter 16R-17, Florida Administrative Code, Comprehensive Shellfish Control Code.

Florida Department of Environmental Regulation. 1988. National Estuaries Program Analysis: Identification of Resource Management Problems and Issues. A Governor's Nomination Report. Submitted to the U.S. Environmental Protection Agency by FDER in Cooperation with Mote Marine Laboratory.

Fonseca, M.S., J.S. Fisher, J.C. Zieman and G.W. Thayer. 1982. Influence of the seagrass, *Zostera marina* L. on current flow. Estuarine Coastal Shelf Sci. 15:351-364.

Food and Agriculture Organization (FAO). 1989. Food safety regulations applied for fish by major importing countries. FAO Circular No. 825. Food and Agricultural Organization of the United Nations and World Health Organization. Rome. 107 pp.

Science 17(3):188-197.

(Gmelin). U.S. Fish Wildlife, Serv. Bull. 64:1-480.

Gartrell, M.J. 1985. Pesticides, selected elements and other chemicals

68(95):868-875.

George, J.D. 1970. Mortality at Southend. Mar. Pollut. Bull. 1:187.

Godcharles, M.F. 1971. A study of the effects of a commercial hydraulic

Department Nat. Resour. Mar. Res. Lab. Tech. Ser. #64. 51 pp.

Godcharles, M.F. and W.C. Jaap. 1973. Exploratory Clam Survey of Florida Nearshore and Estuarine Waters with Commercial Hydraulic Dredging Gear. Fla. Dep. Nat. Resour. Mar. Res. Lab. Prof. Pap. Ser. 21. 77 pp.

Godwin, W.F. 1968. The growth and survival of planted clams, *Mercenaria mercenaria,* on the Georgia coast. GA Game Fish. Comm., Mar. Fish. Div., Brunswick, GA. Contrib. Ser. 9. 15 pp.

Goyer, R.A. 1986. Toxic effects of metals. *In:* Casarett and Doull's Toxicology (C. Klassen, M. Amdur and J. Doull, eds.). Third edition. Macmillan, New York.

Habitat Requirements for Chesapeake Bay Living Resources. Second Edition. 1991. S.L. Funderburk, J. Mihursky, S. Jordan and D. Riley, eds. Chesapeake Research Consortium, Inc., Solomons, MD.

Hansen, J. 1990. Human exposure to metals through consumption of marine foods: A case study of exceptionally high intake among Greenlanders. *In:* Heavy metals in the marine environment (R.W. Furness and P.S. Rainbow, eds.). CRC Press, Boca Raton, FL.

Haskin, H.H. 1955. Further growth studies on the hard clam, *Venus mercenaria.* Proc. Natl. Shellf. Assoc. 42:181-187.

Haven, D.S., J.S. Loesch and J.P. Whitcomb. 1973. An investigation into commercial aspects of the hard clam fishery and development of commercial gear for the harvest of molluscs. Final contract report for the period 1 July 1970 through 30 June 1973. Commercial Fisheries Research and Development Act. Va. Inst. Mar. Sci., Gloucester Point, Virginia. 112 pp.

Herbert, J.P. 1988. A study on the effects of the closure of Midnight Pass on the clam population of Little Sarasota Bay. The Midnight Pass Society, Inc.

Hine, A.C. and D.F. Belknap. 1986. Recent Geological History and Modern Sedimentary Processes of the Pasco, Hernando and Citrus County Coastline: West Central Florida. Florida Sea Grant, Proj. No. 79. 166 pp.

Hobbs, C.H,, R.J. Byrne, R.A. Gammisch and R.J. Diaz. 1985. Sand for beach renourishment in lower Chesapeake Bay. pp. 790-811, *In:* Proceedings of Fourth Symposium on Coastal and Ocean Management. Amer. Soc. Civil. Eng.

Hoffman, E.J,, G.L. Mills, J.S. Latimer and J.G. Quinn. 1983. Annual Input of Petroleum Hydrocarbons to the Coastal Environment via Urban Runoff. Proceedings of the Conference on Pollution in the North Atlantic Ocean, Halifax, Nova Scotia, Oct. 19-23, 1981. pp. 41-53.

Hoffman, E.J., G.L. Mills, J.S. Latimer and J.G. Quinn. 1984. Urban Runoff as a Source of Polycyclic Aromatic Hydrocarbons to Coastal Waters. Environ. Sci. and Technol. 18:580-587.

Hunt, D.A. 1979. Microbial standards for shellfish growing waters - Past, present and future utilization. Proceedings of the National Shellfisheries Association. 69:142-146.

Hurlbert, S.H., M.S. Mulla and H.R. Willson. 1972. Effects of an organophosphorus insecticide on the phytoplankton, zooplankton and insect populations of freshwater ponds. Ecological Monographs. 42:269-299.

Ingle, R.M. 1949. Oyster and clam culture in Florida. Florida State Board of Conservation - Division of Oyster Culture.

Interstate Shellfish Sanitation Conference. 1986. National Shellfish Sanitation Program Manual of Operations. Part 1. Sanitation of Shellfish Growing Areas. FDA. Washington, DC. 135 pp.

Iyengar, G.V. 1989. Elemental analysis of biological systems. Vol I. Biomedical, environmental, compositional and methodological aspects of trace elements. CRC Press, Boca Raton, FL. 242 pp.

Jackim, E. and C. Lake. 1978. Polynuclear aromatic hydrocarbons in estuariune and nearshore environments. p. 415, *In:* Estuarine Interactions (M.L. Wiley, ed.). Academic Press, New York.

Jackim, E., G. Morrison and R. Steele. 1977. Effects of environmental factors on radiocadmium uptake by four species of marine bivalves. Mar. Biol. 40:303-308.

Jain, H.K., N.P. Agnihotri and A.K Gupta. 1987. Persistence of naled and propetamphos in soil, water and sediment. J. Pesticides. 21(10):43-45.

Johnson, J.K. 1977. A study of the shell length of *Mercenaria mercenaria* in relationship to bottom sediments of Little Bay, New Jersey. Bull. NJ Acad. Sci. 22(2):52. Abstract.

Joint Subcommittee on Aquaculture. 1983. National Aquaculture Development Plan. Vol. 2. Washington, DC. 196 pp.

Jones, D.S., I.R. Quitmyer, W.S. Arnold and D.C. Marelli. 1990. Annual shell banding, age, and growth rate of hard clams *(Mercenaria* spp.) from Florida. J. Shellfish Res. 9:(1):215-225.

Keck, R., D. Maurer and R. Malouf. 1974. Factors influencing the setting behavior of larval hard clams, *Mercenaria mercenaria.* Proc. Natl. Shellfish. Assoc. 64:59-67.

Kennedy, V.S., W.H. Roosenburg, M. Castagna and J.A. Mihursky. 1974. (Mollusca: Bivalvia): temperature-time relationships for survival of embryos and larvae U.S. Natl. Mar. Fish. Serv. Fish. Bull. 72:1160-1166.

Kerswill, C.J. 1949. Effects of water circulation on the growth of quahaugs and oysters. J. Fish. Res. Board Can. 7:545-551.

chemically contaminated seafood - Briefing paper. Narragansett Bay Project. Narragansett, RI.

Klassen, C.D., M.O. Amdur and J. Doull. 1986. Toxicology: The science of poisons. Third Edition. New York, NY. 974 pp.

Lake, J.L., C. Norwood, C. Dimock and R. Bowen. 1979. Origins of polycyclic aromatic hydrocarbons in estuarine sediments. Geochim. Cosmochim. Acta 43. 1847-1854.

Landolt, M., D. Kalman, A. Nevissi, G. van Belle, K. Van Ness and F. Hafer. 1987. Potential toxicant exposure among consumers of Recreationally caught fish from urban embayments of Puget Sound: Final report. National Oceanic and Atmospheric Administration Technical Memorandum NOS OMA 33. Rockville, MD. 111 pp.

Langston, W.J. 1990. Toxic effects of metals and the incidence of metal pollution in marine ecosystems. *In:* Heavy metals in the marine environment (R.W. Furness and P.S. Rainbow, eds.). CRC Press, Boca Raton, FL.

Lawrence, J.F. and D.F. Weber. 1984. Determination of polycyclic aromatic hydrocarbons in some Canadian commercial fish, shellfish, and meat products by liquid chromatography with confirmation by capillary gas chromatography-mass spectrometry. J. Agric. Food Chem. 32:789.

Lederer, W.H. 1985. Regulatory chemicals of health and environmental concern, Von Nostrand Reinhold, N.Y.

Leibovitz, L. 1979. A study of vibriosis at a Long Island shellfish hatchery. NYSG-RR-79-02, Sea Grant Reprint Series. Albany, New York.

Lipsey, R.L. 1987. Adverse effects of dibrom in the environment, special publication, Lipsey and Co., Jacksonville, FL. 1 p.

Long, E.R., D. MacDonald and C. Cairncross. 1991. Status and trends in toxicants and the potential for their biological effects in Tampa Bay, Florida. NOAA Technical Memorandum NOS OMA 58. Seattle, WA. 77 pp.

Loosanoff, V.L. 1962. Effects of turbidity on some larval and adult bivalves. Proc. Gulf & Carib. Fish. Inst. 14th Ann. Sess. pp. 80-95.

Loosanoff, V.L. 1965. Gonad development and discharge of spawn in oysters of Long Island Sound. Biol. Bull. (Woods Hole) 129(3):546-561.

Lores, E.M., J.C. Moore, J. Knight, J. Foster, J. Clark and P. Moody. 1985. Determination of fenthion residues in samples of marine biota and seawater from laboratory exposures and field applications. J. Chrom. Sci. 23:127-127.

Louisiana Universities Marine Consortium (LUMCON). 1991. National shellfish pollution indicator study: Chronology and progress to date.

Lowe, D.M. and R.K. Pipe. 1987. Mortality and quantitative aspects of storage cell utilization in mussels, *Mytilus edulis,* following exposure to diesel oil hydrocarbons. Mar. Environ. Res. 22:243.

Lowry, P.W., L.M. McFarland, B.H. Peltier, N.C. Roberts, H.B. Bradford, J.L. Herndon, D.F. Stroup, J.B. Mathison, P.A. Blake and R.A. Gunn. 1989. Vibrio gastroenteritis in Louisiana: A prospective study among attendees of a scientific congress in New Orleans. J. Infect. Dis. 160:978-984.

Luoma, S.N. and G.W. Bryan. 1978. Factors controlling the availability of sediment bound lead to the estuarine bivalve, *Scrobicularia plana.* J. Mar. Biol. Assoc. U.K. 58:793.

MacInnes, J.R. and A. Calabrese. 1978. Response of the embryos of the American oyster, *Crassostrea virginica,* to heavy metals at different temperatures. *In:* Physiology and behavior of marine organisms (D.S. McClusky and A.J. Berry, eds.). Pergamon Press, Oxford. 195 pp.

MacKenzie, C.L., Jr. 1977a. Predation on hard clam *(Mercenaria mercenaria)* populations. Trans. Am. Fish. Soc. 106:530-537.

MacKenzie, C.L., Jr. 1977b. Sea anemone predation on larval oysters in Chesapeake Bay (Maryland). Proc. Natl. Shellfish Assoc. 67:113-117.

MacKenzie, C.L., Jr. 1979. Management for increasing clam abundance. Mar. Fish. Rev. 41:10-22.

Mangrove Systems, Inc. 1988. Marine habitat trend analysis Sarasota County, Final Report. Prepared for Sarasota County Natural Resources Dept.

Marcus, J.M. and T.P. Stokes. 1985. Polynuclear aromatic hydrocarbons in oyster tissue around three coastal marinas. Bull. Environ. Contam. Toxicol. 25:835.

Marganian, V.M., W.J. Wall, Jr. 1972. Dursban and diazinon residues in biota following treatment of intertidal plots on Cape Cod, 1967-69. Pest. Monitor J. 6:160-165.

Martin, M., K.E. Osborn, P. Billig and N. Glickenstein. 1981. Toxicities of ten metals to *Crassostrea gigas* and *Mytilus edulis* embryos and *Cancer magister* larvae. Mar. Pollut. Bull. 12:305.

Mason, A.Z., K.D. Jenkins and P.A. Sullivan. 1988. Mechanisms of trace metal accumulation in the Polychaete *Neanthes arenaceodentata.* J. Mar. Bio. Assoc. U.K. 68:61.

McClusky, D.S., V. Bryant and R.Campbell. 1986. The effects of temperature and salinity on the toxicity of heavy metals to marine and estuarine invertebrates. Oceannogr. Mar. Biol. Annu. Rev. 24:481.

McGreer, E.R. 1979. Sublethal effects of heavy metal contaminated sediments on the bivalve *Macoma balthica* (L). Mar. Pollut. Bull. 10:259.

McGreer, E.R. 1982. Factors affecting the distribution of the bivalve, *Macoma balthica* (L) on a mudflat receiving sewage effluent, Fraser River estuary, British Columbia. Mar. Environ. Res. 7:131.

McKenzie, J., T. Kjellstrom and R. Sharma. 1986. Cadmium intake via oysters and health effects in New Zealand: Cadmium intake, metabolism and effects in people with high intake of oysters in New Zealand. EPA 6OO/S1-86/004. Research Triangle Park, NC.

Menzel, R.W. 1961. Seasonal growth of the northern quahog, *Mercenaria mercenaria* and the southern quahog, *M. campechiensis,* in Alligator Harbor, Florida. Proc. Natl. Shellfish Assoc. 52:37-46.

Menzel, R.W. 1964. Seasonal growth of northern quahog, *Mercenaria mercenaria* and the southern quahog, *M. campechiensis,* and their hybrids in Florida. Proc. Natl. Shellfish Assoc. 53:111-119.

Menzel, R.W., E.W. Cake, M.L. Haines, R.E. Martin and L.A. Olsen. 1976. Clam mariculture in northwest Florida: field study on predation. Proc. Natl. Shellfish. Assc. 65:59-62.

Menzel, R.W. and H.W. Sims. 1964. Experimental farming of hard clams, *Mercenaria mercenaria,* in Florida. Proc. Natl. Shellfish. Assoc. 53:103-109.

Metcalf & Eddy, Inc. 1988. Analysis of risks from consumption of Quincy Bay fish and shellfish. Task IV Report. Prepared for USEPA, Water Quality Management Section, Water Management Division. Boston, MA.

Miller, T.P. 1982. Investigations of residual activity, toxicity and sorption of chlorpyrifos formulations (Killmaster (Registered Trademark) and Dursban (Registered Trademark). Volume 43/03-B of Dissertation Abstracts International. p. 622. 130 pp.

Mix, M.C. 1982. Polynuclear aromatic hydrocarbons and cellular proliferative-disorders in bivalve molluscs from Oregon estuaries. Rep. 600/4-82-026. U.S. Environmental Protection Agency, Washington, DC.

Mix, M.C. 1984. Polycyclic aromatic hydrocarbons in the aquatic environment: occurrence and biological monitoring. p. 51, *In:* Reviews in Environmental Toxicology, Vol. 1 (E. Hodgson, ed.). Elsevier, Amsterdam.

Moore, M.N., D.R. Livingstone and J. Widdows. 1989. Hydrocarbons in marine mollusks: biological effects and ecological consequences. pp. 291-328, *In:* Metabolism of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment (U. Varanasi, ed.). CRC Press, Inc., Boca Raton, Florida.

Moore, M.N., J. Widdows, J.J. Cleary, R.K. Pipe, P.N. Salkeld, P. Donkin, S.V. Farrar, S.V. Evans and P.E. Thompson. 1984. Responses of the mussel *Mytilus edulis* to copper and phenanthrene: interactive effects. Mar. Environ. Res. 14:167.

Morris, J.G., Jr. and R.E. Black. 1985. Cholera and other vibrioses in the United States. N. Engl. J. Med. 312:343-350.

Morrison, G. 1971. Dissolved oxygen requirements of embryonic and larval development of the hard-shelled clam, *Mercenaria mercenaria.* J. Fish. Res. Brd. Can. 28:379-381.

Mote Marine Laboratory. 1985. Final Report for Crystal River 316 Studies. Submitted to Florida Power Corporation. Subcontractor to Stone & Webster Engineering Corp.

Mote Marine Laboratory. 1991. Final Report: Environmental monitoring program, Naval Submarine Base, Kings Bay, Georgia, July 1989 through June 1990. MML Technical Report No. 210. Submitted to Department of the Navy, Southern Division. 116 pp.

Mulholland, R. 1984. Habitat suitability index models: Hard clam. FWS/OBS-82/10.77. U.S. Fish and Wildlife Service. Department of the Interior.

Music, S.I., J.T. Howell and C.L. Brumback. 1973. Red tide, its public health implications. J. Fla. Med. Assn. 60:27.

Myint, V.M. and P.A. Tyler. 1982. Effects of temperature, nutritive and metal stressors on the reproductive biology of *Mytilus edulis.* Mar. Biol. 67: 209

Nagtmajtenyi, L., A. Selypes and G. Berencsi. 1985. Chromosomal aberratons and fetotoxic effects of atmospheric arsenic exposure in mice. J. Appl. Toxicol. 5:61-63.

NAS. 1980. The international mussel watch. U.S. National Academy of Sciences. 248 pp.

NAS. 1991. Seafood safety: Committee on evaluation of the safety of fishery products (F.E. Ahmed, ed.). Food and Nutrition Board, Institute of Medicine, National Academy of Sciences. National Academy Press, Washington, D.C.

National Academy of Sciences. 1985. Oil in The Sea: Inputs, Fates and Effects. National Academy Press Washington, DC. p. 601.

National Oceanic and Atmospheric Administration. 1984. Standard analytical procedures of the NOAA National Analytical Facility 1984-85: Extractable organic compounds. NOAA Technical Memorandum NMFS F/NWC-64. Seattle, WA. 110 pp.

National Oceanic and Atmospheric Administration. 1988. PCB and chlorinated pesticide contamination in U.S. fish and shellfish: A historical assessment report. Technical Memorandum NOS OMA 39. Nat. Oceanic and Atmos. Admin. Seattle, WA.

National Oceanic and Atmospheric Administration. 1989. Progress report: A summary of data on tissue contamination from the first three years 1986-1988) of the Mussel Watch Project. Technical Memorandum NOS OMA 49. NOAA. Rockville, MD. 22 pp.

National Oceanic and Atmospheric Administration. 1990a. The potential for biological effects of sediment-sorbed contaminants tested in the national status and trends program. Technical Memorandum NOS OMA 52. NOAA, Seattle, WA. March.

National Oceanic and Atmospheric Administration. 1990b. Chlordane in the marine environment of the United States: review and results from the national status and trends program. Technical Memorandum NOS OMA 55. NOAA, Seattle, WA. September.

National Oceanic and Atmospheric Administration. 1991. The 1990 national shellfish register of classified estuarine waters. NOAA. Rockville, MD. 100 pp.

National Oceanic and Atmospheric Administration and National Marine Fisheries Service. 1991. Recreational shellfishing in the United States. Addendum to 1985 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation. 22 pp.

National Research Council of Canada (NRCC). 1978. Effects of arsenic in the Canadian environment. Natl. Res. Coun. Canada Publ. No. NRCC 15391. 349 pp.

National Shellfish Sanitation Program. 1990. Manual of Operation. Part I* Sanitation of Shellfish Growing Areas. pp. C1-C34.

Neff, J.M. 1979. Polycyclic Aromatic Hydrocarbons in the Aquatic Environment: Sources, Fates and Biological Effects. Elsevier, London.

Neff, J.M., B.A. Cox, D. Dixit and J.W. Anderson. 1976. Accumulation and release of petroleum derived aromatic hydrocarbons by four species of marine animals. Mar. Biol. 38:279.

Nelson, A. and P. Donkin. 1985. Processes of bioaccumulation: the importance of chemical speciation. Mar. Pollut. Bull. 16:164.

Okazaki, R.K. and M.H. Panietz. 1981. Depuration of twelve trace metals in tissues of the oyster *Crassostrea gigas* and *C. virginica.* Mar. Biol. 63:113.

Olla, B.L., A.J. Bejda and W.H. Pearson. 1983. Effects of oiled sediment on the burrowing behavior of the hard clam, *Mercenaria mercenaria.* Mar. Environ. Res. 8:183-193.

Olsen, L.A. and K. Adams. 1984. Effects of contaminated sediment on fish and wildlife. FWS/OBS-82/66. U.S. DOI, Washington, DC. pp. 101.

Otwell, W.S., J.A. Koburger and S. Andree. 1986. Survival of southern hard clams in refrigerated storage. Florida Sea Grant Report No. 83. 17 pp.

Overton, E.B., J.McFall, S.W. Mascarella, C.F. Steel, S.H. Antoine, I.R. Politzer and J.L. Laseter. 1981. Petroleum residue sources identification after a fire and oil spill. Proceedings, 1981 Oil Spill Conference, Amer. Petroleum Inst., Washington, DC. pp. 541-546.

Paez-Osuna, F. and C. Marmolejo-Rivas. 1990. Occurrence and seasonal variation of heavy metals in the oyster *Saccostrea iridescens.* Bull. Environ. Contam. Toxicol. 44:129.

Parry, J.M., M. Kadhim, W. Barnes and N. Danford. 1981. Assays of marine organisms for the presence of mutagenic and/or carcinogenic chemicals. p. 141, *In:* Phyletic Approaches to Cancer (J.J. Dawe, J.D. Harshbarger, S. Kondo, T. Sugimura and S. Takayama, eds.) Japan Scientific Societies Press, Tokyo.

Pearse, A.S. 1936. Estuarine animals at Beaufort, North Carolina. J. Elisha Mitchell Sci. Soc. 52:174-222.

Peddicord, R.K. 1980. Direct effects of suspended sediments on aquatic organisms. pp. 501-536, *In:* Contaminants and Sediments, Ann Arbor Sci., Ann Arbor, MI (R.A. Baker, ed.) Vol. 1.

epidemiological appraisal. Naturvardsverket Rapp. SNV PM 1128, Liber Tryck, Stockholm. 265 pp.

(Busycon
experimental tests of the importance of prey size, prey density, and seagrass cover. Mar. Biol. 66:159-170.

Peterson, C.H. 1983. A concept of quantitative reproductive senility: *Mercenaria mercenaria*
Oecologia. 58:164-168.

Peterson, C.H., H.C. Summerson and P.B. Duncan. 1984. The influence of seagrass cover on population structure and individual growth rate of a suspension-feeding bivalve, Mercenaria *mercenaria.* J. Mar. Res. 42:123-138.

Phillips, D.J.H. 1990. Use of macroalgae and invertebrates as monitors of metal levels in estuaries and coastal waters. *In:* Heavy metals in the marine environment (R.W. Furness and P.S. Rainbow, eds.). CRC Press, Boca Raton, FL.

Phillips, D.J.H. and D.A. Segar. 1986. Use of bio-indicators in monitoring conservative contaminants: programme design imperatives. Mar. Pollut. Bull. 17:1O.

Phillips, D.J.H. and W.W.S. Yim. 1981. A comparative evaluation of oysters, mussels and sediments as indicators of trace metals in Hong Kong waters. Mar. Ecol. Prog. Ser. 6:285.

Phillips, G.R. and R.C. Russo. 1978. Metal bioaccumulation in fishes and aquatic invertebrates: a literature review. EPA-600/3-78-103. U.S. EPA Environmental Research Laboratory. Duluth, MN.

Pierce, R.H. 1986. Red tide toxin aerosols: A reveiw. Toxicon. 24:955-965.

Pierce, R.H, 1988. Pesticide residues in mangrove communities and cistern water of the Cayman Islands. Final Report. 9 pp.

Pierce, R.H. and R.C. Brown. 1983a. Hydrocarbon characterization of urban runoff to the Hillsborough River, Tampa, FL. Final Report submitted to the City of Tampa, FL.

Pierce, R.H. and R.C. Brown. 1983b. Hydrocarbon analysis of surficial sediment in support of southwest Florida shelf regional biological communities survey. Final Report submitted to Mineral Management Services.

Pierce, R.H., D.C. Anne, F.I. Saska and B.A. Weichert. 1985. Fate of organic compounds from spent drilling fluid discharged into the sea. Chapter 7. *In:* Wastes in the Ocean, Volume 4: Energy Wastes in the Ocean (I.W. Duedall, D.R. Kester and P.K. Parks, eds.). John Wiley and Sons, Inc., New York.

Pierce, R.H., R.C. Brown, E.S. Van Vleet and R.M. Joyce. 1986. Hydrocarbon contamination from coastal development. PP. 229-245, *In:* Organic Marine Geochemistry (M.L. Sohn, ed.). American Chemical Society, Washington, DC. Chapter 14.

Pierce, R.H., C.E. Olney and G.T. Felbeck. 1974. pp'-DDT adsorption to suspended particulate matter in seawater. Geochim. Cosmochim. Acta. 33:1061-1073.

Pline, M.J. 1984. Reproductive cycle and low salinity stress in adult *Mercenaria mercenaria* L. of Wassaw Sound, Georgia. M.S. Thesis, Georgia Institute of Technology, Atlanta. 74 pp.

Plumb, R.H., Jr. 1981. "Procedure for handling and chemical analysis of sediment and water samples". Tech. Rpt. EPA/CE-81-1. 1 p.

Porter, H.J. 1962. Incidence of *Malacobdella* in *Mercenaria campechiensis* off Beaufort Inlet, North Carolina. Proc. Natl. Shellfish. Assoc. 53:133-145.

Pratt, D.M. 1953. Abundance and growth of *Venus mercenaria* and *Callocardia morrhuana* in relation to the character of bottom sediments. J. Mar. Res. 12:60-74.

Pratt, D.M. and D.A. Campbell. 1956. Environmental factors affecting growth in *Venus mercenaria.* Limnol. Oceanog. 1:2-17.

Pruell, R.J. and J.G. Quinn. 1988. Accumulation of Polycyclic Aromatic Hydrocarbons in Crankcase Oil. Environ. Pollut. 49:89-97.

PTI Environmental Services. 1988. The apparent effects threshold approach: Briefing Report. Submitted by the Puget Sound Estuary Program. Seattle, WA.

Rainbow, P.S. 1990. Heavy metal levels in marine invertebrates. *In:* Heavy metals in the marine environment (R.W. Furness and P.S. Rainbow, eds.). CRC Press, Boca Raton, FL.

Ratcliffe, S. and D.S. Wilt, eds. 1971. Proceedings Seventh National Shellfish Sanitation Workshop, Washington, D.C. October 20-22. 412 pp.

Rathus, E.M. 1973. The Effect of Pesticide Residues on Humans. pp. 23-31, *In:* Pesticides and the Environment: A Continuing Controversy (W.B. Deichmann, ed.). Intercontinental Med. Book Corp., New York.

Renzoni, A. 1973. Influence of crude oil, derivatives, and dispersants on larvae. Mar. Pollut. Bull. 4:9.

Reynolds, W. 1976. Botanical, geological and sociological factors affecting the management of the barrier islands adjacent to Stump Pass. New College (USF) Environmental Studies Program.

Rhoads, D.C. and G. Pannella. 1970. The use of molluscan shell growth patterns in ecology and paleoecology. Lethaia. 3:143-161.

Richards, G.P. 1978. Comparative study of methods for the enumeration of total and fecal coliforms in the eastern oyster, *Crassostrea virginica.* Appl. Environ. Microbial. 36:975-978.

Richter, J. 1988. Puget sound water quality authority, Rpt. State of the sound. 225 pp.

Roberts, B.S., G.E. Henderson and R.A. Medlyn. 1979. The effect of *Gymnodinium breve* toxins on selected molluscs and crustaceans. p. 419, In: Toxic Dinoflagellate Blooms (D.L. Taylor and H.H. Seliger, eds.). New York: Elsevier/North Holland.

Robertson, J. and C. Mazzella. 1989. Acute toxicity of the pesticide diazinon to the freshwater snail, *Gillia altilis.* Bull. Environ. Contam. Toxicol. 42:320-324.

Rodrick, G. 1990. Indigenous bacterial pathogens. *In:* Microbiology in marine food products (D. Ward and C. Hackney, eds.). Van Nostrand Reinhold, New York. 285-301.

Roegner, G.C. and R. Mann. 1991. Hard clam, *Mercenaria mercenaria.* pp. 5-1 -5-17, *In:* Habitat Requirements for Chesapeake Bay Living Resources, 2nd Ed. (S.L. Funderburk, J.A. Mihursky, S.J. Jordan, and D. Riley, eds.). Chesapeake Resarch Consortium, Inc., Solomons, MD.

Rubenstein, N.I., E. Lores and N. Gregory. 1983. Accumulation of PCB's, mercury and cadmium by *Nereis virens, Mercenaria mercenaria* and *Palaemonetes pugio* from contaminated harbor sediments. Technical Report D-83-4. U.S. Environmental Protection Agency. Gulf Breeze, FL.

Rubenstein, N.I., W.T. Gilliam and N.R. Gregory. 1984. Dietary accumulation of PCBs from a contaminated sediment source by a demersal fish *(Leiostomus xanthurus).* Aquatic Toxicology. 5:331-342.

Ruple, A.D., D.W. Cook and R. Dusho. 1989. *Vibrio vulnificus* in raw Gulf Coast oysters. *In:* Conference Proceedings. Fourteenth Annual Conference - Tropical and Subtropical Fisheries Technological Society of the Americas. Atlanta. pp. 118-127.

Saloman, C.H. and J.L. Taylor. 1969. Age and growth of large southern quahogs from a Florida estuary. Proc. Natl. Shellfish Assoc. 59:46-51.

San Francisco Bay: Issues, resources, status and management. 1987. Proc: NOAA Estuary-of-the-month, Sem. Ser. No. 6. (D.M. Goodrich, ed.). 159 pp.

Sander, J.G. 1986. Direct and indirect effects of arsenic on the survival and fecundity of estuarine zooplankton. Can. J. Fish. Aquat. Sci. 43:694-699.

Schimmel, S.C., R.L. Garnas, J.M. Patrick and J.C. Moore. 1983. Acute toxicity, bioconcentration and persistence of AC 222, 705, bendiocarb, chlorpyrifos, fenvalerate, methylparathion and permethrin in the estuarine environment. J. Agric. Food Chem. 31:104-113.

Schroeder, W.C. 1924. Fisheries of Key West and the clam industry of southern Florida. Rep. U.S. Comm. Fish. for 1923, Append. 12, Doc. 962. 74 pp.

Schropp, J.S. and H.L. Windom (ed). 1987. A guide to the interpretation of metal concentrations in estuarine sediments. Florida Department of Environmental Regulation Coastal Zone Management Section. 44 pp.

Seymour, A.H. 1966. Accumulation and loss of zinc-65 by oysters in a natural environment. *In:* Disposal of radioactive wastes into seas, oceans and surface waters. International Atomic Energy Agency. Vienna.

Shukla, S.S. and H.V. Leland, 1973. Heavy metals: A review of lead. J. Water Poll. Control Fed. 45:1319-1331.

Shuster, C.N. and B.H. Pringle. 1969. Trace metal accumulation by the American eastern oyster, *Crassostrea virginica.* Proceedings of the National Shellfisheries Association. 59:91-103.

Sims, H.W., Jr., and R.J. Stokes. 1967. A survey of the hard shell clam *(Mercenaria campechiensis)* (Gmelin) population in Tampa Bay, Florida. Fla. State Brd. Conserv. Mar. Lab. Spec. Sci. Rpt. No. 17. 8 pp.

Sindermann, C.J. and A. Rosenfield. 1967. Principal diseases of commercially important bivalve Mollusca and Crustacea. Fish. Bull. 66:335-385.

Sirota, G.R. and J.F. Uthe. 1981. Polynuclear aromatic hydrocarbon contamination in marine shellfish. p. 329, *In:* Chemical Analysis and Biological Fate: Polynuclear Aromatic Hydrocarbons (M. Cook and A.J. Dennis, eds.). Battelle Press, Columbus, OH.

Sobsey, M.D. 1990. Viruses during controlled and natural depuration. Presentation to the Gulf and South Atlantic Shellfish Sanitation Conference. Wilmington, N.C.

Southern California Association of Governments. 1988. The state of Santa Monica Bay: Executive Summary. 58 pp.

Southwest Florida Water Management District. 1990. Summary of hydrologic conditions. Brooksville, FL. January through December.

Sporstol S., N. Gjos, R.G. Lichtenthaler, K.O. Gustavsen, K. Urdal, F. Oreld and J. Skei. 1983. Source identification of aromatic hydrocarbons in sediments using GC/MS. Environ. Sci. Technol. 17:282-286.

Sprinkel, J. and J. Gorzelany. 1985. Oyster predatory inventory and oyster reef associated fauna. *In:* A Wet Season Characterization of the Tidal Myakka River. E.D. Estevez, Editor. Mote Marine Laboratory Technical Report. Submitted to Sarasota County, FL.

Stanley, J.G. 1985. Hard clam, a species profile. U.S. Fish and Wildlife Service Biological Report 82(11.41).

Stanley, J.G. and R. Dewitt. 1983. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (North Atlantic) -- hard clam. U.S. Fish Wildl. Serv. FWS/OBS-82/11/18. U.S. Army Corps of Engineers, TR EL-82-4. 19 pp.

Stegeman, J.J. 1981. Polynuclear aromatic hydrocarbons and their metabolism in the marine environment. p. 1, *In:* Polycyclic Hydrocarbons and Cancer, Vol. 3 (H.V. Gelboin and P.O.O. Ts'O, eds). Academic Press, New York.

Steidinger, K.A. 1983. A re-evaluation of toxic dinoflagellate biology and ecology. Prog. Phycol. Res. 2:148.

Steidinger, K.A. and K. Haddad. 1981. Biologic and hydrographic aspects of red tides. Bioscience. 31:814.

Steidinger, K.A., M.A. Burklew and R.W. Ingle. 1973. The effects of *Gymnodinium breve* toxin on estuarine animals. pp. 179-220, *In:* Marine Pharmacognosy: Action of Marine Biotoxins at the Cellular Level. Academic Press, New York.

Stevely, J., E.D. Estevez, and J.K. Culter. In Press. The benthic fauna of Sarasota Bay with emphasis on commercially and recreationally important species. Presentation to SaraBasis. Sarasota, FL.

Stone and Webster Engineering Corporation. 1985. Final report Crystal River 316 studies. Prepared for Florida Power Corporation with Mote Marine Laboratory. Boston, MA.

Stromgren, T., M.V. Nielsen, and K. Ueland. 1986. The short term effect of microencapsulted hydrocarbons on shell growth of *Mytilus edulis.* Mar. Biol. 91:33.

Sykes, J.E. and J.R. Hall. 1970. Comparative distribution of mollusks in dredged and undredged portions of an estuary, with a systematic list of species. U.S. Natl. Mar. Fish. Serv. Fish. Bull. 68:299-306.

Talbot, V.W., R.J. Magee and M. Hussain. 1976. Cadmium in Port Phillip Bay mussels. Mar. Pollut. Bull. 7(5):84-86.

Tamplin, M. 1990. The ecology of *Vibrio vulnificus* in *Crassostrea virginica.* Abstracts Annual Meeting. National Shellfisheries Association. April 1-6, Williamsburg, Va. 451 pp.

Tamplin, M., G.E. Rodrick, N.J. Blake and T. Ciba. 1982. Isolation and characterization of *Ubrio vulnificus* from two Florida estuaries. Appl. Environ. Microbial. 44(6):1466-1470.

Tanacredl, J.T. and R.R. Cardnas. 1991. Biodepuration of polynuclear aromatic hydrocarbons from a bivalve mollusc, *Mercenaria mercenaria L.* Environ. Sci. Technol. 25:1453-1461.

Taylor, F.J.R. 1990. Red tides, brown tides and other harmful algal blooms: The view into the 1990's. pp. 527-533, *In:* Toxic Marine Phytoplankton (E. Graneli, B. Sundstrom, L. Edler and D.M. Anderson, eds.). Proc. 4th Intrntl. Conf. Toxic Mar. Phytopl. New York: Elsevier/North Holland.

Taylor, J.L. and C.H. Saloman. 1968. Benthic project. Pages 3-5, *In:.* Report of the Bureau of Commercial Fisheries Biological Laboratory, St. Petersburg Beach, Florida, fiscal year 1967 (J.E. Sykes, ed.). U.S. Fish Wildl. Serv. Circ. 290.

Taylor, J.L. and C.H. Saloman. 1970. Benthic project. Pages 3-10, *In:* Report of the Bureau of Commercial Fisheries Biological Laboratory, St. Petersburg Beach, Florida, fiscal year 1969 (J.E. Sykes, ed.). U.S. Fish Wildl. Serv. Circ. 342.

Taylor, J.L. and C.H. Saloman. 1972. Nereid shell blisters in the southern quahog clam. Quart. J. Fla. Acad. Sci. 35:21-26.

Tettlebach, S.T., L.M. Petti and W.J. Blogoslawski. 1984. Survey of *Vibrio* associated with a New Haven Harbor shellfish bed, emphasizing recovery of larval oyster pathogens. pp. 495-509, *In: Vibrios* in the environment (R.R. Colwell, eds.). John Wiley & Sons. New York.

Thanabalasingam, P. and W.F. Pickering. 1986. Arsenic sorption by humic acids. Environ. Pollut. 12B:233-246.

Thompson, R.E. 1988. Proc. of Conf. of Interstate Shellfish Sanitation Conference. 11 pp.

Thompson, C.A. and C. Vanderzant. 1976. Serological and hemolytic characteristics of *Vibrio parahaemolyticus* from marine sources. J. Food Sci. 41:204-205.

Thorson, G. 1955. Modern aspects of marine level-bottom animal communities. J. Mar. Res. 14:387-397.

Thorson, G. 1966. Some factors influencing the recruitment and establishment of marine benthic communities. Neth. J. Sea Res. 3:267-293.

Tiffany, W.J., III and M.G. Heyl. 1978. Invertebrate mass mortality induced by a *Gymnodinium breve* red tide in Gulf of Mexico waters at Sarasota, Florida. Environ. Sci Health. A13(9):653-662.

Twedt,R.M., J.M. Madden, J.M. Hunt, D.W. Francis, J.T. Peeler, A.P. Duran, W.O. Herbert, S.G. McCay, C.N. Roderick, G.T. Spite and T.J. Wazenski. 1981. Characterization of *Vibrio cholerae* isolated from oysters. Appl. Environ. Microbial. 41:1475-1478.

U.S. Department of Interior. 1988. Metal biogeochemistry in surface-water systems -- a review of principles and concepts. U.S. Geol. Surv. Circ., No.1013. 43 pp.

U.S. Environmental Protection Agency. 1980a. Ambient water quality criteria documents. Arsenic. EPA 440/5-80-021. USEPA Office of Water Regulation and Standards. Criteria Division. Washington, D.C.

U.S. Environmental Protection Agency. 1980b. Ambient water quality criteria documents. Cadmium. EPA 440/5-80-025. USEPA Office of Water Regulation and Standards. Criteria Division. Washington, D.C.

U.S. Environmental Protection Agency. 198Oc. Ambient water quality criteria documents. Copper. EPA 440/5-80-036. USEPA Office of Water Regulation and Standards. Criteria Division. Washington, D.C.

US. Environmental Protection Agency. 1980d. Ambient water quality criteria documents. Lead. EPA 440/5-80-057. USEPA Office of Water Regulation and Standards. Criteria Division. Washington, D.C.

U.S. Environmental Protection Agency. 1980e. Ambient water quality criteria documents. Mercury. EPA 440/5-80-058. USEPA Office of Water Regulation and Standards. Criteria Division. Washington, D.C.

U.S. Environmental Protection Agency. 1984a. Analytical reference standards and supplemental data: The pesticides and industrial chemicals repository. EPA-600/4-84-082. October. Washington, D.C.

U.S. Environmental Protection Agency. 1984b. Guidelines establishing Test Procedures for the analysis of pollutants under the Clean Water Act; Final Rule and Proposed Rule. Federal Register Part VIII, 40 CFR Part 136. 210 pp.

U.S. Environmental Protection Agency. 1984c. Health effects assessment documents. Cadmium. PB86-134455. Washington, D.C.

U.S. Environmental Protection Agency. 1984d. Health effects assessment documents. Copper. PB86-134368. Washington, D.C.

U.S. Environmental Protection Agency. 1984e. Health effects assessment documents. Lead. PB86-134335. Washington, D.C.

U.S. Environmental Protection Agency. 1984f. Health effects assessment documents. Mercury. PB86-134533. Washington, D.C.

U.S. Environmental Protection Agency. 1985. Verified reference doses of the U.S. Environmental Protection Agency. Office of Research and Development.

U.S. Environmental Protection Agency. 1987. Health advisories for *Legionalla* and seven inorganics. PB87-235586. Washington, D.C. 125 pp.

U.S. Food and Drug Administration. 1978. Bacteriological Analytical Manual. 6th edition. Association of Analytical Chemists, Arlington, Va. pp. VI.1-29.

U.S. Food and Drug Administration. 1986. National Shellfish Sanitation Program Manual of Operation Part: Sanitation of Shellfish Growing Areas. Revision.

U.S. Food and Drug Administration (FDA). 1987. Compliance policy guide, Chapter 41-Pesticides. 7141.01, April 1. Food and Drug Administration. Washington, D.C.

U.S. Food and Drug Administration. 1989. NSSP Shellfish Sanitation Program, Manual of operations: Part I - Sanitation of shellfish growing areas.

U.S. Department of Health, Education and Welfare. 1971. Proposal for levels of additional pesticides in shellfish. pp. 36-37, *In:* Seventh National Shellfish Sanitation Workshop Proceedings (S.D. Ratcliff and D.S. Windt, eds). October 20-22.

Van Vleet, E.S., R.H. Pierce, R.C. Brown and S.B. Reinhardt. 1984. Sedimentary hydrocarbons from a subtropical marine estuary. Org. Geochem.6:249-257.

Verber, J. 1984. Shellfish-borne disease outbreaks. U.S. Food and Drug Administration review paper. North Kingstown, RI. 36 pp.

Viarengo, A. 1989. Heavy metals in marine invertebrates: mechanisms of regulation and toxicity at the cellular level. Reviews in Aquatic Sciences. 1(2):295-317

Virstein, R.W. 1977. The importance of predation by crabs and fishes on benthic fauna in Chesapeake Bay. Ecology. 58:1199-1217.

Wade, T.L., E.L. Atlas, J.M. Brooks, M.C. Kennicutt II, R.G. Fox, J. Sericano, B. Garcia-Romero and D. DeFreitas. 1988. NOAA Gulf of Mexico Status and Trends Program: Trace organic contaminant distribution in sediments and oysters. Estuaries. 11(3):171-179.

organisms. Mar. Pollut. Bull. 16:7.

Walker, R.L. and K.R. Tenore. 1984. The distribution and production of the hard clam, *Mercenaria mercenaria* in Wassaw Sound, Georgia. Estuaries. 7:19-27.

Walker, R.L., M.A. Fleetwood and K.R. Tenore. 1980. The distribution of the hard clam, *Mercenaria mercenaria* (Linne) and clam predators in Wassaw Sound, Georgia. GA Mar. Cent. Tech. Rep. Ser. 80-8. 57 pp.

Water and Air Research, Inc. 1988. Environmental monitoring program naval submarine base, Kings Bay, Georgia, July 1987 through June 1988. Final Report submitted to the Dept. of the Navy.

Wells, H.W. 1957. Abundance of the hard clam *Mercenaria mercenaria* in relationship to environmental factors. Ecology 38:123-128.

West, P.A. and R.R. Coldwell. 1984. Identification and classification of Vibrionaceae - An overview. pp. 285-363, *In: Vibrios* in the environment (R.R. Colwell, ed.). John Wiley & Sons. New York.

Windom, H.L., S.J. Schropp, F.D. Calder, J.D. Ryan, R.G. Smith, Jr., L.D. Burney, F.G. Lewis and C.H. Rawlinson. 1989. Env. Sc. & Tech. 23:314.

Wolfe, D.A. 1970. Levels of stable Zn and 65Zn in *Crassostrea virginica* from North Carolina. J. Fish. Res. Board Can. 27(1):47-57.

Woodburn, K.D. 1962. Clams and oysters in Charlotte County and vicinity. Fla. State Board Conserv. Mar. Lab., Mimeo Rep. No. 62-12. 29 pp.

World Health Organization (WHO). 1972. Food Addition Series No. 4. World Health Organization. Geneva.

Youngblood, W.W. and M. Blumer. 1975. Polycyclic aromatic hydrocarbons in the environment: homologous series in soils and recent marine sediments. Geochim. Cosmochim. Acta. 39:1303-1314.

Zaroogian, G.E. and M. Johnson. 1984. Nickel uptake and loss in the bivalves *Crassostrea virginica* and *Mytilus edulis.* Arch. Environ. Contam. Toxicol. 13:411-418.

APPENDICES

AVAILABLE UPON REQUEST