A Synthesis of Pollutant Inputs to Buzzards Bay

edited by :
Joseph E. Costa
Victoria Gibson
Judith M. Pederson

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Preface

The four chapters that comprise this volume were developed under the guidance of the Buzzards Bay Project National Estuary Program to fulfill requirements to characterize pollutant inputs to Buzzards Bay, in order to develop and refine recommendations contained in the Buzzards Bay Comprehensive Conservation and Management Plan (CCMP). The CCMP was drafted in 1990, and finalized in 1991. Development of Chapters 1-3 (a synthesis of inputs and impacts of pathogens, nutrients, and toxics in Buzzards Bay) commenced in 1989 and was finalized by 1991. Chapter 4, the "characterization" of pollutant loadings to Buzzards Bay began in 1990 and was finalized in 1991. This latter chapter was based partly on existing drafts of Chapters 1, 2, and 3, land use statistics and loading assumptions from the Buzzards Bay Project, and other data.

Together these four chapters helped lay the framework for the Buzzards Bay CCMP pollution characterization chapter, the Buzzards Bay Monitoring Plan (CCMP Volume III), as well as help set the focus of the Buzzards Bay Project on non-point sources of pollution. These reports and others prepared for the Buzzards Bay Project found that overall, Buzzards Bay had good water quality, particularly offshore. While it was found that the preponderance of pathogen, nutrient, and toxic inputs to Buzzards Bay are discharged from the New Bedford sewer outfall and combined sewer overflows, the impacts from these inputs were localized in the waters surrounding New Bedford. Elsewhere in Buzzards Bay, water quality and living resources were also degraded, and these degraded conditions were most likely to be found within the more than 30 embayments that fringe the shore of the Bay. This uncoupling between pollutants to the Bay as a whole versus the conditions existing in Buzzards Bay embayments led the Buzzards Bay Project to redirect its efforts toward characterization and management of pollution at the embayment level. One of the eventual outcomes of this redirection of effort was the draft final Buzzards Bay Project report "A Buzzards Bay Embayment Subwatershed Evaluation: Establishing Priorities for Nitrogen Management Action" released in May of this year.

As noted in Chapters 1 and 4, one of the key indicators documenting water quality decline of Buzzards Bay embayments was the fourfold increase in shellfish bed closures in Buzzards Bay between 1970 and 1990. While some of the increased closures were due to more diligent monitoring, most of this decline appeared related to increased coastal development during the same period. That is, where degradation occurred, it appeared related to localized sources of pathogens and nutrients from the watersheds surrounding each embayment. In most Buzzards Bay embayments, non-point sources of pollution, rather than permitted point sources, appear to contribute the bulk of the pathogen and nutrient loadings. Among the major findings in these and other Buzzards Bay Project studies was that the primary nitrogen source to Buzzards Bay was wastewater disposal. For most embayments this took the form of groundwater discharges from household onsite wastewater disposal systems. This conclusion helped focus the Buzzards Bay Project's nitrogen mass loading approach. Findings that stormwater related discharges were often the single most important source of pathogens in embayments where shellfish beds were

closed also strongly influenced the direction of future Buzzards Bay Project efforts to implement the CCMP.

Impacts of non-point sources of toxic contamination were not as well defined for Buzzards Bay, and this largely reflects uncertainty about the effects of low level toxic contamination at the ecosystem level. On the other hand, contamination of seafood species by PCBs originating from the Superfund site in New Bedford was identified as the only contaminant problem of bay-wide concern (including the central Bay). This contamination represents a clear potential health threat, but ecosystem level impacts have not been as well documented. Because of the potential health threat of PCB contamination of commercial and recreational seafood species, the Buzzards Bay Project emphatically supported Superfund Site remediation in the CCMP so that PCBs will be eventually reduced below the US FDA action limits for seafood species throughout Buzzards Bay. Because the Project believed that it was vital to document the reduction of PCBs in seafood species bay-wide, this action item was an important element in the CCMP Monitoring Plan. The localized nature of other toxic discharges around New Bedford led the Buzzards Bay Project to commence in 1992 a toxic use reduction technical assistance program for commercial and industrial operators in the greater New Bedford area.

While these reports have been used as "in house" resources for the past several years by the Buzzards Bay Project, our intent has long been to combine the reports into this single volume with a single format and style to be made available to the public, researchers, and libraries. We regret that it has taken this long to release these reports, and we appreciate the patience and support of the authors.

Joseph E. Costa
Victoria Gibson
Judith M. Pederson

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We would like to thank all those who took the time and effort to review these reports. We are especially grateful to Tracy Warncke for pulling together all these reports into a consistent format.

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Chapter 1

Pathogen Issues in Buzzards Bay

by D. Jay Grimes [1], George Heufelder[2]

Introduction

Description of Buzzards Bay

Buzzards Bay is a large coastal embayment located in southeastern Massachusetts between Cape Cod and the southern mainland of the state (Figure 1). The Bay, which is approximately 28 mi long, opens to the south into Rhode Island Sound; from the northeast, passage to Cape Cod Bay is allowed through the Cape Cod Canal. The Bay and its tributaries serve as a substantial economic and recreational resource: both shores have an abundance of harbors and coves used for both recreation and commerce, and more than 4,300 boat slips and moorings are located along the bay (Gil, 1987). Shellfishing is an important recreational and commercial pursuit, with an average annual harvest exceeding 86,000 bushels of shellfish (Alber, 1987). In 1985, the catch of shellfish had a market value of over $6.5 million. The recent increase in the number of shellfish harvesting areas closed due to bacterial contamination of the overlying waters has been defined as a priority issue in the National Estuary Program's Buzzards Bay Project. This report synthesizes the available information on bacterial fecal indicator and human pathogen sources within the bay. This information is needed to develop strategies to properly manage human sanitary wastes near marine resources.

Geomorphology

Buzzards Bay has a surface area of approximately 235 mi^2 and an average depth of about 50 ft in the central basin (Gil, 1988). The western portion of Buzzards Bay has a total drainage area of approximately 350 mi^2 and, in addition to the Buttermilk Bay drainage system at the Bay's head, is drained by seven coastal river basins. From east to west, the major river basins in the Buzzards Bay watershed are the Agawam, Wankinco, Weweantic, Mattapoisett, Acushnet, Paskamanset/Slocums, and Westport. The river basins of the western shore generally consist of series of small freshwater ponds, streams and cranberry bog drainages combining their flows and widening at their mouths as they enter Buzzards Bay.

The easternmost river basin, the Agawam, has a relatively small drainage area. It is bordered on the east by the Buttermilk Bay-Onset Bay drainage system and on the west by the Wankinco drainage basin. The Wankinco and Agawam Rivers join to form, in part, the flow of the Wareham River. The Wareham Publicly Owned Treatment Work (POTW) discharges to the Agawam River through four separate pipes. Before discharge, the effluent has received secon-

1 U.S. Department of Energy, Washington, DC 20545
2 Barnstable County Health and Environmental Department, Barnstable, MA 02630

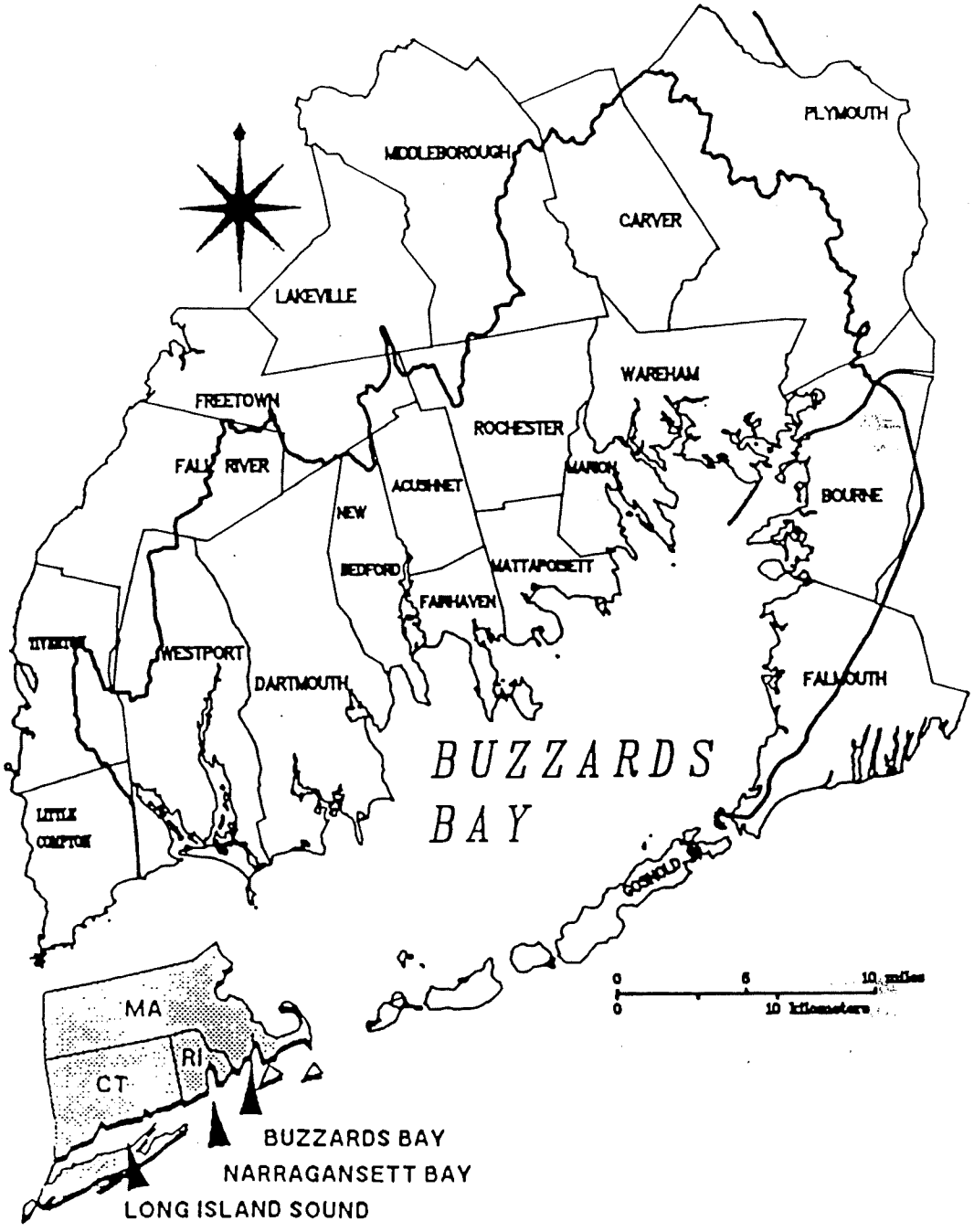

Figure 1. Buzzards Bay and its drainage basin

Source: Town boundaries provided by MassGIS and digitized from 1:25000 scale USGS quadrangle maps. Basin boundary complied by USGS-WRD and digitized by MassGIS. Cape Cod side basin boundary based on interpretation of water table elevation contours published in Hydrologic Atlas No. HA-692

Figure 1. Map of Buzzards Bay area, southeastern Massachusetts. Source: Buzzards Bay Project

dary treatment and is filtered through sand. In addition, the Regional Landfill at Carver is situated along the freshwater reach of the Wankinco River.

To the west of the Weweantic-Sippican and Mattapoisett drainage basins, which are relatively undeveloped for much of their freshwater reach, is the Acushnet River drainage basin. The Acushnet is the most urbanized watershed that drains into Buzzards Bay. Nearly 50% of the population of the Buzzards Bay coastal drainage basin resides within this 15 mi^2 subwatershed (Gil, 1988), which includes the city of New Bedford, portions of Dartmouth, Fairhaven, Acushnet, and Freetown. Headwaters of the Acushnet are at the outlet of the New Bedford Reservoir. The upper reaches of the Acushnet River flow through rural/suburban areas for 3 mi before becoming tidally influenced just below the Main Street Bridge. Below this point, the area is heavily industrialized. Sixty percent (17 of 27) of all National Pollutant Discharge Elimination System (NPDES) permits approved by the Environmental Protection Agency (EPA) for Buzzards Bay are in this basin. The Acushnet River estuary contains two sewage treatment plants. The first, for the town of Fairhaven, discharges secondarily treated wastes (3.0 million gallons per day [MGD]) just inside (north) of the hurricane dike. The second, serving New Bedford, discharges primary effluent into Buzzards Bay some 3,000 ft out from Clark Point at a rate of 30 MGD. In addition to effluent from the New Bedford sewage treatment plant, the estuary receives sanitary wastes through 38 combined sewer overflows (CSOs) located throughout Acushnet river estuary, Acushnet harbor, and Clarks Cove.

To the west of the Acushnet drainage basin lies the Paskamansett/Slocums Rivers. The headwaters of the Paskamansett River drain a largely undeveloped expanse of wetlands that includes the Acushnet Cedar Swamps. Over the past 35 years, agriculture in this drainage basin has been replaced by a mix of residential, commercial, and industrial uses, especially between Routes 6 and 195. The Town of Dartmouth's sewage treatment plant discharges approximately 2.0 MGD of secondary-treated sewage 3,000 ft off Mishaun Point.

Much of the land drained by the Westport River, with its two major branches, is devoted to agriculture. At least 32 active dairy and beef cattle farms (about 3,900 cattle) are located in the watershed (Beskenis, 1989). Along the East Branch of the river, a number of the dairy farms are sited on the steep slopes of the river banks. Although the United States Soil Conservation Service instituted some runoff controls in 1986, a Food and Drug Administration (FDA) sanitary survey (September-October, 1986) indicated that these measures have been largely ineffective in reducing the levels of fecal indicator in the receiving waters.

The eastern shore of Buzzards Bay, from the Cape Cod Canal to Woods Hole and Falmouth, provides approximately 35 additional square miles of drainage from river systems smaller than those on the western shore. The prominent freshwater streams along the eastern shore from north to south are Back River, Pocasset River, Wild Harbor River, and Herring Brook. In general, these rivers meander through areas of low coastal marsh and lack the higher bordering relief common on the western shore of Buzzards Bay. On-site sewage treatment is the only form of sanitary waste disposal in the eastern watershed; thus none of the rivers on this side of Buzzards Bay contain POTW facilities.

Riverine and Tidal Influences

There are a variety of factors influencing the dispersal of contaminants entering Buzzards Bay. In general, the effect of tidal action is fairly small except in the "holes" along the Elizabeth Island chain. Most mixing appears to be caused by wind-driven surface currents (Rocky Geyer, Woods Hole Oceanographic Institution, personal communication). A study being conducted in conjunction with the siting of the New Bedford POTW outfall should clarify the role of these currents in dispersing the sanitary wastes from this source and increase our overall understanding of the transport mechanisms within the bay.

In order to understand how the hydrography of Buzzards Bay affects pathogens and indicator organisms, it should be noted that dilution and dispersal are primary factors that determine the persistence of these organisms in aquatic environments. If bacteria or viruses enter the Bay attached to particulate matter, they may settle to the bottom, where the sediments provide a protected environment and survival is significantly enhanced (Roper and Marshall, 1974; Sayler et al., 1975; Smith, 1978; LaBelle and Gerba, 1980; Liew and Gerba, 1980; LaBelle and Gerba, 1982). If unattached, however, bacteria are presented with a dilute environment from which it is difficult to derive nutrients, and viruses become subject to thermal destabilization as well as other deactivating factors. It has been shown that some bacteria, when presented with this stress, will enter a resistive stage and become nonculturable using standard recovery techniques.

On windless days, tidal movements govern the dispersal of microbes entering Buzzards Bay. Under these conditions, sewage discharges behave as parcels of water oscillating northward and southward at tidal frequency and have less chance of leaving the bay. This phenomenon was demonstrated in the 1987 FDA dye study near the Massachusetts Maritime Academy outfall. Despite a strong westerly flow during the dye release, the returning tide moved the mass of water back toward the head of the bay, even to the mouth of Buttermilk Bay.

Estuarine Pollution

Point Source Pollution

Sources of pollution entering a body of water are commonly classified as either point or nonpoint. Point sources occur at discrete and identifiable points along the estuary, usually through pipeline discharges and direct dumping. The Office of Technology Assessment (OTA) recently estimated that pipeline dischargers to estuaries and coastal waters in the United States number almost 2,000, and have a total volume of 6.44 trillion gallons per year. Most of these discharges (96%) are located in estuaries and are from major industries (66%). Of direct interest to this report was the observation that 43% of the dischargers were concentrated in the northern Atlantic region of the United States (OTA, 1987).

Of the approximately 15,500 POTWs in the United States, only 578 discharge directly into estuaries and coastal waters (OTA, 1987). However, those 578 POTWs account for one-fourth of the nation's wastewater, and 509 discharge into estuaries. Of the 2.3 trillion gallons of municipal wastewater released to

marine waters each year in the United States, 2 trillion gallons go into estuaries and 0.3 trillion into coastal waters.

In the United States, only the sewerage authorities in Los Angeles and Boston discharge their sewage sludge by pipeline into coastal waters and estuaries. Nineteen sewage plants, located in New York City and northern New Jersey, dump their sludge into the ocean at the 106-Mile Deepwater Municipal Sludge Site. This practice will, however, almost cease by January 1992, as a result of the Ocean Dumping Ban Act (ODBA) of 1988. All other POTWs that discharge into marine waters dispose of their sludge by means of incineration, landfill, or land application. Finally, no industrial dumping is now permitted in the United States, either in its territorial waters (12 miles) or in the Exclusive Economic Zone (EEZ, 200 miles).

Nonpoint Source Pollution

Nonpoint sources are diffuse, often unknown or ill-defined inputs to an estuary. Nonpoint sources of pollution include surface runoff, rainfall or rainout, atmospheric fallout or deposition, underground transport, and leaching of materials to the estuary. Good estimates, especially quantitative ones, of the contributions made by nonpoint sources to estuaries are lacking. However, available data do permit some generalizations about nonpoint pollution.

Materials deposited upon surfaces associated with cities, suburban areas, farmland, forests, wetlands, and industry are subsequently removed from those surfaces by rainfall. Included are contributions from both generalized surface runoff and two specific point sources, streams and CSOs. Streams receive generalized runoff from upland areas and convey materials to the estuary. CSOs are sewer interceptors that receive both wastewater and stormwater and, because of inadequate capacity of the POTW to handle the increased volume due to the stormwater, divert the untreated mixture of wastewater and stormwater directly to a receiving body of water. Surface runoff was implicated by OTA (1987) as a major source of fecal coliforms, suspended solids, and nutrients to coastal waters, including estuaries.

Underground transport includes both aquifers and septic systems that have contact with the upper water table that, in turn, connects with the coastal ocean. In some cases, the ground becomes so saturated with water that septic systems fail, wastewater breaks to the surface, and the surfaced material enters as surface runoff.

In a recent report, EPA singled out nonpoint sources as the most important contributor of damaging pollutants in 48% of the cases where estuaries failed to support the key uses of fishing, swimming, and propagation of marine life (EPA, 1984). This report further stated that, except in the Northeast, nonpoint sources were more important than point sources.

Sources of Pathogens

Pathogenic microorganisms are associated with many waste materials, including domestic wastewater, industrial and hospital wastewater, wastes from wildlife, and wastes associated with boats and ships. Many of these pathogens are capable of survival and growth in aquatic habitats, including estuaries.

Waste-associated microorganisms can be pathogenic for both humans and animals. Among the pathogens are viruses, bacteria, algae, protozoans, and fungi. In practice, only the viruses and bacteria are of any great concern to humans, and, of these, only the bacteria can grow in aquatic habitats. Viruses, being obligate intracellular parasites, can only replicate (grow) inside a suitable host.

Because of their parasitic nature, viruses pathogenic to humans derive primarily from human wastes; therefore, POTWs, septic systems, boat-waste holding tanks, and other receptacles for human wastes are the sources from which these pathogens enter estuaries. Many different viruses are associated with human wastes. Among the more frequent ones (Table 1), most cause gastrointestinal illness and most are ribonucleic acid (RNA) viruses.

Most bacteria are not obligate intracellular parasites; in fact, most bacteria do not cause disease of any kind. However, raw sewage can contain large numbers of pathogenic bacteria, which are often discharged into estuaries. In addition, since most bacteria are free-living, some not only survive in estuaries, but many can grow and a few even occur naturally in estuaries. Bacteria that are foreign to a habitat are said to be allochthonous; i.e., they are aliens that enter that habitat from another, unlike habitat. An example of a bacterial species that is allochthonous to estuaries is the fecal coliform, *Escherichia coli*. Autochthonous bacteria, on the other hand, are indigenous to a habitat. *Vibrio parahaemolyticus* is a human pathogen that has the estuary as its normal habitat. Human pathogenic bacteria commonly encountered in estuaries are listed in Table 2.

Table 1. Human viruses found in waste material

Family Virus	Nucleic Acid	Disease(s)	Waste(s)
Adenoviruses			
Human adenovirus	DNA	Acute respiratory,pharngitis, accute hemorrhagicrhagic cystitis	Wastewater
Enteric adenovirus	DNA	Gastroenteritis	Wastewater
Caliciviruses			
Calicivirus	RNA	Gastroenteritis	Wastewater
Norwalk virus	RNA	Gastroenteritis	Wastewater
Coronaviruses			
Enteric coronavirus	RNA	Intestinal disorders	
Orthomyxoviruses			
Influenza virus	RNA	Influenza	Human, swine, & fowl waste
Picornaviruses			
Coxsackievirus A	RNA	Meningitis, herpangia, common cold,	Wastewater
Coxsackievirus B	RNA	Myocarditis, pleurodynia, rash, meningitis, paralysis	Wastewater
ECHO virus	RNA	Paralysis, diarrhea, meningitis	Wastewater
Hepatitis A virus	RNA	Infectious hepatitis	Wastewater
Poliovirus	RNA	Poliomyelitis	Wastewater
Reoviruses			
Reovirus	RNA	Respiratory, gastroenteritis	Wastewater
Rotavirus	RNA	Infantile diarrhea	Wastewater
Astrovirus	???	Gastroenteritis?	Wastewater

Table 2. Pathogenic bacteria frequently detected in estuaries

Species	Diseases(s)	Waste and / or Source
Acinetobacter calcoaceticus	Nosocomial	Water, human skin & mouth
Aeromonas hydrophila	Septicemia,wound infection, diarrhea	Water (fresh & estuarine)
Aeromonas sobria	Same as above	Same as above
Aeromonas caviae	Same as above	Same as above
Chromobacterium violaceum	Septicemia, diarrhea	Soil, water
Citrobacter spp.	Nosocomial	Water
Clostridium botulinum	Botulism	Soil, sediment, fish
Clostridium perfringens	Gangrene, wood abscesses, food poisoning	Animal feces
Clostridium sporogenes	Gangrene	Soil & feces
Clostridium tetani	Tetanus	feces
Enterobacter spp.	Nosocomia	Wastewater
Erysipelothrix rhusiopathiae	Erysipeloid	Animal feces, fish slime
Escherichia coli	Gastroenteritis	Wastewater
Flavobacterium meningosep-ticum	Nosocomial, meningitis	Fresh water
Francisella tularensis	Tularemia	Rodents, fresh water
Klebsiella pneumoniae	Pneumonia, bacteremia, nosocomial	Water, feces, soil plants
Legionella pneumophila	Legionnaires' disease	Fresh water, cooling towers, hot water tanks
Leptospira interrogans	Leptospirosis	Urine
Listeria monocytogenes	Listeriosis	Soil, feces
Morganella morganii	Urinary tract nosocomial nosocomial	Water, feces, decaying animals
Mycobacterium marinum	Swimming pool, granuloma	Water, fish
Plesiomonas shigelloides	Gastroenteritis	Water, fish, aquatic animals
Proteus spp.	Urinary tract, posocomial	Water, feces, decaying animals
Pseudomonas aeruginosa	Burn, wound, corneal, ear, urinary lung, skin GI	Water, wastewater plants, sediment, fish
Staphylococcus aureus	Abscesses, food poisoning	mammalian skin, ocean water
Streptococcus faecalis	Endocarditis	Animal feces
Vibrio alginolyticus	Wound infection	Ocean water, aquatic animals
Vibrio cholerae	Cholera	Wastewater, shellfish, saltwater
Vibrio parahaemolyticus	Gastroenteritis	saltwater, shellfish
Vibrio vulnificus	Septicemia, wound infection	Oysters, seawter
Yersinia enterocolitica	Gastroenteritis, acute mesenteric lymphadenitis	Water, milk, mammalian alimentary canal

Fate of Pathogens

Pathogenic microorganisms differ in their ability to survive in saltwater habitats. Clearly, survival can continue for much longer than previously thought, and this extended survival has significant implications for estuarine pollution. Fate of an organism can include more than its survival; the term fate can also refer to where in the habitat it is found. Pathogens may reside in the water column, at air-water and solid-water interfaces, in estuarine animals (e.g., oysters), and in the sediment. In general, the greatest concentrations of pathogenic microorganisms exist in animals, at interfaces, and in the sediment.

Most saltwater survival studies conducted on human pathogens have used plate count methodology. A typical experiment involves growing a representative pathogen in or on rich medium, harvesting the growth into a sterile suspension liquid (usually sterile salt water or buffer), adjusting the suspension

to a known concentration of cells, and then adding a measured volume of the suspension to salt water in a flask or in a membrane chamber. At various time intervals, aliquots of the saltwater-suspended bacteria are removed and either spread-plated (or pour-plated) onto (into) a rich medium; alternatively, serial dilutions of the bacteria are inoculated into tubes of broth so that a most probable number (MPN) can be determined. Factors that have been cited as contributing to death of bacteria in marine systems include osmotic stress (Carlucci and Pramer, 1960), ultraviolet and visible light (Kapuscinski and Mitchell, 1981), predation, parasitism, competition, antibiosis, toxic organic and inorganic chemicals (Jones, 1964 and 1971), and temperature (Yoshpe-Purer and Shuval, 1972). A thorough discussion of these factors appeared in the recent review of Elliot and Colwell (1985).

Recently, the concept of stress or injury to bacterial cells released to natural aquatic environments has gained popularity. Deriving from similar, earlier work by food microbiologists (Clark and Ordal, 1969; Hobbs and Olson, 1971; Ray et al., 1971), several papers have been published that relate injury to environmental limiting factors (Bissonnette et al., 1975; Zaske et al., 1980; McFeters et al., 1982). Certainly, adverse environmental factors affect bacteria and no doubt kill large numbers in certain situations. Toxic chemicals kill bacteria by such mechanisms as enzyme inactivation, interference with nutrient transport, and osmotic stress which may cause plasmolysis of sensitive cells. In addition to lethal injury, sublethal injury also occurs, resulting from factors such as chlorine (LeChevallier et al., 1985) and acid (Walsh and Bissonnette, 1983; Wortman and Bissonnette, 1985). Most of the evidence supporting sublethal injury is indirect; i.e., it is inferred from observing the loss of some activity or attribute previously exhibited by the uninjured cells. However, there is also some direct evidence of sublethal injury, most coming from electron microscope observations (Tuttle et al., 1977; Zaske et al., 1980; Wortman et al., 1985).

Several studies by McFeters and his colleagues at Montana State University have addressed the phenomenon of injury. They define injury as the difference between colony forming units (CFU) on nonselective medium and CFU on selective medium, divided by the CFU on nonselective medium (Domek et al., 1984; McFeters et al., 1986). Unfortunately, this definition does not include bacteria that are still viable but are incapable of growth on any medium, selective or nonselective.

While injury and stress exact their toll on bacteria released to estuaries and the sea, another phenomenon tends to preserve bacteria faced with less than optimum growth conditions. This phenomenon has frequently been referred to as dormancy and it appears to be widespread among gram-negative bacteria. Dormancy was first described by Novitsky and Morita (1978) and by Stevenson (1978). Recently, Morita described dormancy as the "normal mode of survival" for gram-negative bacteria (Morita, 1985). Dormancy is not synonymous with injury and stress; it is a normal survival strategy that allows for preservation of species in the absence of other more obvious mechanisms, e.g., endospores and cysts. Although dormancy was first described for autochthonous aquatic bacteria, it is now apparent that allochthonous species also become dormant upon entering nutrient-poor aquatic habitats. Unlike native dormant bacteria, which can often be cultivated under proper conditions (Tabor et al., 1981; MacDonell

and Hood, 1982), their allochthonous counterparts usually become nonculturable in and on all conventional culture media. Like native dormant forms, viable but nonculturable bacteria tend to round up and become much smaller than log-phase cells of the same strain (Tabor et al., 1981; MacDonell and Hood, 1982; Baker et al., 1983; Morita, 1985; Grimes et al., 1986).

The "viable but nonculturable" phenomenon was first described for *Escherichia coli* and *Vibrio cholerae* by Xu et al. (1982). Since that time, the phenomenon has beendocumented for *Vibrio cholerae, V. parahaemolyticus, V. vulnificus, Escherichia coli, enterotoxigenic E. coli, Enterobacter aerogenes, Agrobacterium tumifaciens, Aeromonas* spp., *Listonella anguillara,Shigella flexneri, S. sonnei, Salmonella enteritidis, S. typhimurium, Campylobacter jejuni, C. pylori,* and *Legionella pneumophila.* Each of these species exhibits the general response shown in Figure 2, differing only in the slope of the various lines. The slope is influenced by such variables as pH, temperature, salinity, sterility vs. nonsterility of the microcosm, and presence of toxic chemicals. In every case, however, viable cells remain in the microcosms long after the system becomes nonculturable.

Little success has resulted from attempts to grow nonculturable cells in vitro. Roszak et al. (1984), working with a *Salmonella enteritidis* serogroup C_1 isolated from the Potomac River near Washington, DC, was able to culture cells 21 d after they were placed into Potomac River microcosms. This represented 18-19

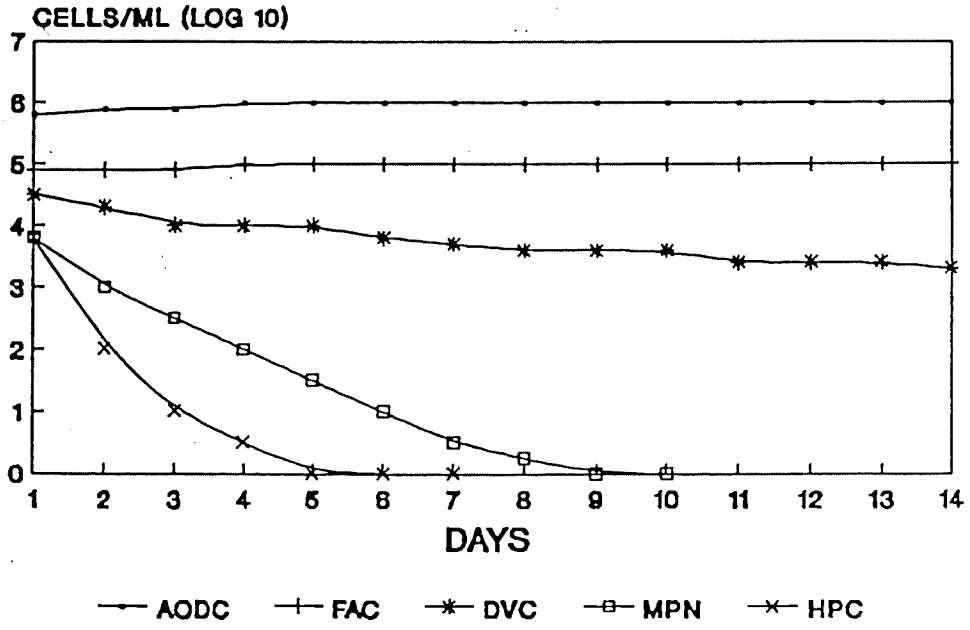

Figure 2. Enteric pathogen survival in marine systems as measured by acridine orange direct count (AODC), fluoresent antibody count (FAC), direct viable count (DVC), most probable number (MPN), and heterotrophic plate count (HPC).

d after the cells became nonculturable on veal infusion agar, on xylose-lysine-desoxycholate (XLD) agar, and in a *Salmonella* enrichment broth. Recovery was accomplished in full-strength heart infusion broth prepared with filtered Potomac River water. This is in contrast to other studies (e.g., Novitsky and Morita, 1978; MacDonell and Hood, 1982) that used dilute media in order to successfully grow dormant autochthonous bacteria recovered from aquatic habitats.

It has been argued that since viable but nonculturable bacteria cannot be grown, they are truly nonrecoverable. In other words, if they are not dead, they are definitely moribund. This hypothesis has not been supported by recent data. Colwell et al. (1985) reported that viable but nonculturable human pathogens could be recovered, in fully virulent form, by passage through appropriate animal models. In one series of experiments with Shigella spp. in Chesapeake Bay water microcosms, viable but nonculturable S. flexneri was recovered by passage through a mouse gut (Brayton et al., 1984). In another experiment, this time with Vibrio cholerae serovar 01 contained in Patuxent River water microcosms, recovery in fully virulent form was accomplished by use of the rabbit ligated ileal loop model of Spira et al. (1981b). Not only was V.cholerae recovered from the ligated loops, but the loops were engorged with bloody fluid, proof that the organism was still virulent. Enterotoxigenic Escherichia coli (strain H10407) was allowed to enter a viable but nonculturable stage of growth while contained in seawater-filled membrane chambers that were suspended in Nixon's Harbor, South Bimini, Bahamas (Grimes and Colwell, 1986). Upon return to the laboratory, contents of the membrane chambers were concentrated and inoculated into ligated rabbit ileal loops. E. coli recovered from engorged ligated loops was shown, by identification of virulence plasmids known to be associated with that particular strain, to be the experimental strain (H10407). The case for viable but nonculturable bacteria has recently been reviewed by Grimes et al. (1986).

It is now well substantiated, based on the experiments described above and on others not described in this report, that gram-negative bacteria pathogenic for humans enter a viable but nonculturable stage of growth when placed into nutrient-poor aquatic habitats. The cells become smaller than normal and appear to lose ability to grow on conventional media that normally support growth of the species in question. The cells do not lyse, but appear to remain intact, as shown by AODC and FA epifluorescent microscopy. In addition, the cells remain physiologically responsive to nutrient and therefore alive, as determined by the DVC of Kogure et al. (1979) and by uptake of ^{3}H- and ^{14}C-labeled substrate (Roszak, 1986 and 1987). Upon introduction to the appropriate animal model, the cells again become culturable, demonstrating all of their normal properties, i.e., cell size, ability to grow on culture media, and virulence (both genotypic and phenotypic). Indeed, a very recent abstract by Brayton et al. (1987) reported successful recovery of fully culturable *V. cholerae* from stool samples collected from two human volunteers who swallowed approximately 10^9 total cells harvested from a microcosm that had tested viable but nonculturable for 48 h. Roszak (1987) has recently coined the terms "viviform" and "somnicell" to describe viable but nonculturable bacteria. It can be concluded from these studies that current indirect enumeration methods, whether based on indicator bacteria or on pathogens themselves, are seriously underestimating the true numbers of viable pathogenic bacteria entering and

accumulating in estuarine and coastal ocean environments throughout the world.

Fecal Indicators as a Measure of Health Risk

The overall health of an estuary can be diagnosed in different ways. Sedimentation rates can be correlated with the efficacy of upland soil management, nitrogen budgets reveal susceptibility to acid rain and agricultural runoff, toxic chemical and metal analysis can pinpoint industrial and agricultural activity, and pathobiology can be used to focus on the health of resident animal life. Each approach provides an indication of anthropogenic impact and each requires some degree of sophistication to conduct the analysis and interpret the data.

Similar to the physicochemical parameters, detection and quantification of specific groups of bacteria, historically referred to as indicator bacteria, have been used to estimate the public health risk (or safety) of water, sediment, shellfish, and other components of an estuary. In almost every case, a given indicator includes more than one taxonomic group, since it is based on the isolation and identification of bacteria with one or more physiological attributes in common. For example, coliform bacteria include several genera and species, including *Escherichia coli, Enterobacter aerogenes, Klebsiella pneumoniae,* and a variety of *Salmonella* species. The feature used to differentiate these from other, closely related, gram-negative bacteria is the ability of coliforms to ferment the sugar lactose with production of acid and gas within 48 hours when incubated at 35°C. Coliforms were originally employed to indicate fecal contamination and, hence, a potential for causing enteric disease. In the early 1900s, it was realized that some coliforms occur naturally in healthy or "normal" terrestrial and aquatic habitats (e.g., *Klebsiella pneumoniae* in plant materials, including redwood trees), thereby making their use as an indicator of enteric disease questionable, if not misleading.

In an attempt to provide public health officials with a more reliable indicator of fecal pollution, the concept of a fecal coliform was proposed by the German bacteriologist Eijkman at the turn of the century. A subset of coliform bacteria, fecal coliforms are defined as gram-negative, rod-shaped bacteria fermenting lactose to produce acid and gas within 24 to 48 hours, at an incubation temperature of 44.5 °C, depending on the method of testing employed. Fecal coliforms were believed to comprise a single taxon, Escherichia coli, but problems of specificity were quickly recognized when fecal coliform testing was done. Today, it is generally accepted that the use of fecal coliforms as an indicator of public health safety has the following disadvantages:

- Both fecal and nonfecally-associated bacteria, e.g., *Klebsiella pneumoniae,* comprise what is recognized as the fecal coliform group;
- Fecal coliforms bear little, if any, quantifiable association with pathogens of concern, including viruses, e.g., hepatitis A, and indigenous aquatic bacteria, such as *Vibrio cholerae, Aeromonas hydrophila,* and *Pseudomonas aeruginosa,* which are potential pathogens;
- Fecal coliforms survive for indefinite periods in aquatic habitats, including estuaries and shellfish, either in a fully detectable and culturable form or in a dormant, i.e., viable but nonculturable form; and
- Fecal coliforms do not provide a meaningful indication of wastewater disinfection efficacy, since commonly employed disinfectants, such as

chlorine, will accelerate the transition of these bacteria from a culturable to a nonculturable form, thereby making them appear to die when, in fact, they are only in a starvation/survival, or dormant, stage.

EPA recently recommended replacing the fecal coliform index with enumeration of *Escherichia coli*, enterococci, or both in fresh water and enterococci in marine and estuarine waters. This recommendation was based on an extensive epidemiological study in which incidence of enteric disease among swimmers was compared with incidence of selected indicator bacteria, including coliforms, fecal coliforms, *E. coli*, and enterococci, in seawater at the swimming site (beach). The results suggest that enterococci may be preferable as an indicator organism, although scientists and regulatory agencies have expressed concern about implementation. Enterococci are gram-positive cocci capable of growth at 10 and 45°C and demonstrating the presence of selected biochemical attributes. The enterococci may or may not be universally adopted by state, county, and local regulatory agencies as a suitable indicator organism; some states, for example Delaware, Hawaii, and Maine, have already adopted the new *E. coli* and enterococcus standards. Nevertheless, the presence of large numbers of enterococci appears to be significantly correlated with the incidence of viral infections.

Other indicator systems are being proposed, some of which are based on molecular genetic methods. For example, viruses of the coliform bacteria, the coliphages (especially the male-specific coliphages) have been suggested as indicative of the risk of infection with human enteric viruses, including hepatitis A. The *Clostridium perfringens* spore test is helpful in tracking long-term build-up or movement of sewage in aquatic habitats. This test was developed by Emerson and Cabelli (1982), and is predicated on the assumption that *C. perfringens* spores are not present in the absence of fecal pollution. Limitations of this test are as follows: (1) lack of host specificity, i.e., *C. perfringens* spores are found in many warm-blooded animals, (2) spores live for indeterminate periods of time and their detection does not define the time of pollution; and (3) the test is generally not useful in the water column, since spores settle into the sediment (Emerson and Cabelli, 1982; Beskenis, 1989). The spore test is best used for tracking and delineating sewage plumes or waste disposal sites in open-water situations. Of great interest are gene probes, now being used to detect pathogenic microorganisms in clinical specimens. At the present time, two problems are associated with gene probes. First, gene probes can be highly specific, that is, they may detect only a small subset, or strain, of a given pathogenic species of bacteria. Therefore, gene probes detecting a genetic sequence that is common, or evolutionarily conserved, to all subsets of the pathogen of concern will need to be developed. Second, most gene probes use a radioactive (^{32}P) detection system, creating testing and waste disposal difficulties. Neither of these problems are insurmountable. In fact, non-radioisotopic detection systems have already been developed, as have probes of broader specificity. In addition, the recent development and perfection of the polymerase chain reaction (PCR) for amplifying nucleic acid concentration has allowed for gene probe detection of very small amounts, and in some cases single copies, of RNA and DNA. Clearly, gene probes will be adapted by regulatory agencies charged with monitoring the health of estuaries. Immunological methods for direct detection of pathogens, especially the fluores-

cent monoclonal antibody detection procedure, also show promise for regulatory use.

Pathways to Hosts

Once in the estuary, disease agents can take either a direct or an indirect pathway to a potential host. Direct pathways allow the pathogen to invade a host directly from the water. They include invasion through body orifices (for example, ingestion while swimming); invasion through both broken and unbroken skin; and transmission by aerosols (for example, the aerosols created by sewage plants and breaking waves). Indirect pathways are those that involve vectors, fomites, and seafood. In each case, the pathogens have colonized something in the water which, in turn, has been used, contacted, or eaten by man. Both direct and indirect pathways are epidemiologically significant and must be given careful consideration. Figure 3 diagrams the major pathways that disease agents take en route to a potential host.

Evidence to document direct pathways is not extensive but does exist. Many reports have involved nonenteric infections, i.e., infections that do not involve the gastrointestinal tract. These have included dermatitis, myositis, otitis, wound infections, endometritis, and vaginitis, and some of the pertinent cases are discussed by Brisou (1975), Pien et al. (1977), Cabelli (1978, 1983), Joseph et al. (1979; 1982), Coolbaugh et al. (1981), Kelly and McCormick (1981), Brook et al. (1982), Tison and Kelly (1984), and Tacket et al. (1984). Several potential agents of these types of disease exist in seawater and include *Pseudomonas aeruginosa, Staphylococcus aureus, Mycobacterium* spp., and several *Vibrio* species (Table 2). Documented nonenteric cases of disease appear to be infrequent. However, nonenteric infections derived from the sea are infrequently reported to health officials. This fact, combined with the knowledge that sewage and nutrient inputs to the ocean are increasing and will continue to increase (Duedall et al., 1983), is sufficient cause for concern about waste disposal in the ocean. Allochthonous pathogens are being added to the oceans of the world at an alarming rate, and both they and autochthonous pathogens are being supplied with increasing amounts of growth-stimulating nutrients present in the wastes.

Enteric disease resulting from direct contact with water is not well documented, for either fresh water or seawater. The best example of enteric disease transmission by swimming is the *Shigella* outbreak that occurred in Dubuque, Iowa, in 1974 (Rosenberg et al., 1976). Forty-five cases of bacillary dysentery (shigellosis) resulted from swimming in a 8-km stretch of the Mississippi River. Water samples collected from that stretch of the river shortly after cessation of the outbreak were found to contain high levels of fecal coliforms (10^4/100 ml) and of *Shigella sonnei* with the same antibiogram, phage, and colicin types as some of the clinical isolates. There was a high correlation between illness and swimming ($p < 0.0001$), and the only non-swimmer that developed shigellosis was a child who had played at the river's edge over a 2-day period. Although *Shigella sonnei* was not isolated from the Dubuque sewage plant located some 8 km upstream, this POTW was strongly suspected as the source of the outbreak. Coincidentally, Grimes (1980) pointed out that dredging was occurring in the vicinity of Nine Mile Island, site of the swimming associated with the outbreak (Rosenberg et al., 1976). Dredging resuspends sediment-bound bacteria

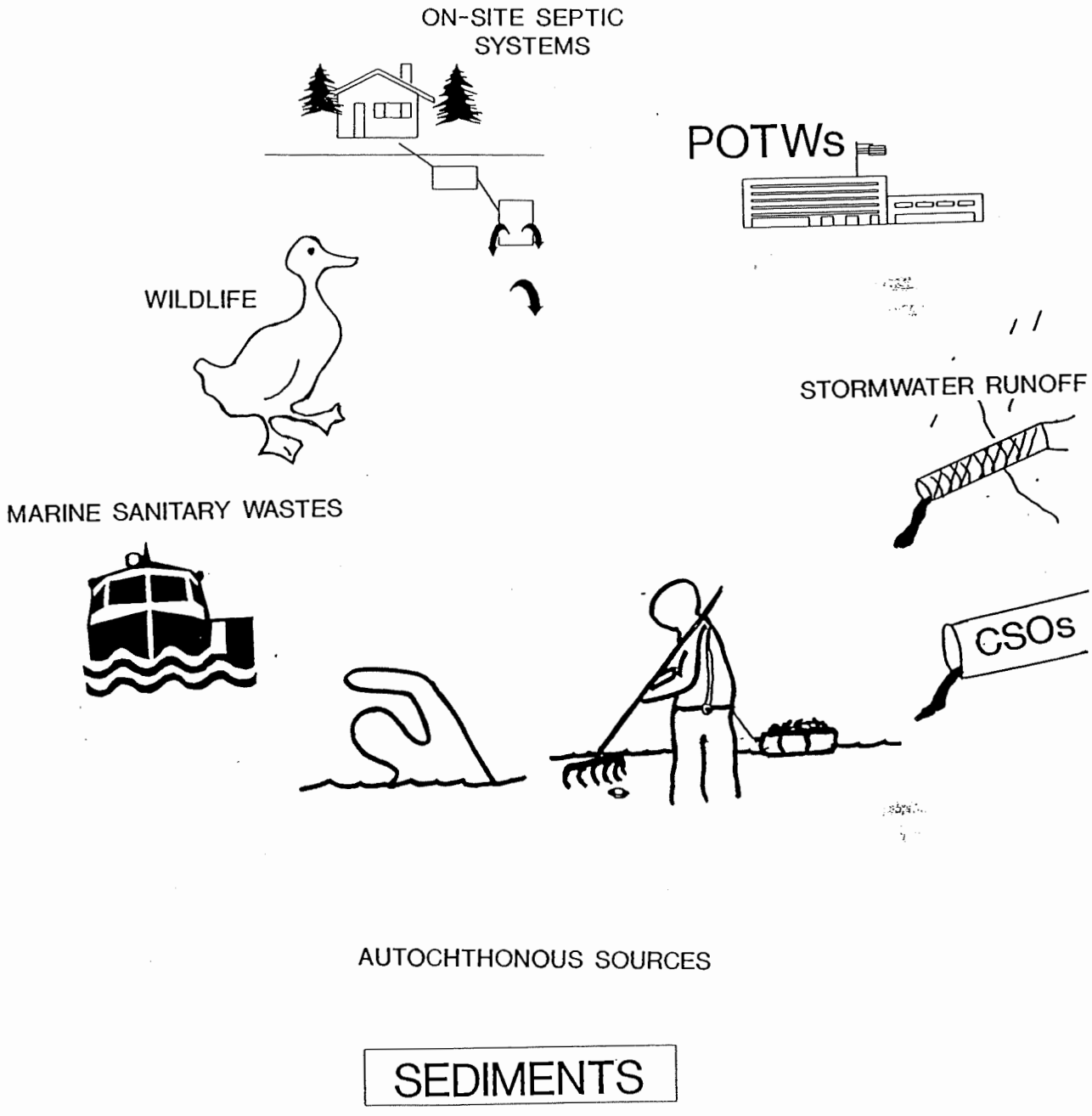

Figure 3. Potential pathways of pathogens to man from various sources in Buzzards Bay.

(Grimes, 1975; 1980), and therefore could have also contributed shigellae to the area.

Disease outbreaks related to swimming in fresh water have also been reported from Vermont and from France. The Vermont outbreak occurred in 1972, and involved a boys' summer camp located on Lake Champlain. Coxsackievirus B5 was recovered from samples cultured from 15 of 33 boys; the same virus was isolated from a water sample (Hawley et al., 1973). Coxsackie A_{16} was implicated in the Niort, France, outbreak that involved five cases. The virus was isolated from lake water that contained 50-1,000 *E. coli* per 100 ml, Group D streptococci, and *P. aeruginosa* (Denis et al., 1974).

Enteric disease resulting from direct transmission has not been reported as frequently from seawater as for fresh water. However, it can not be assumed that seawater is less of a threat than fresh water. Dufour pointed out that illness rates in the Cabelli study of seawater-borne disease (Cabelli, 1983; see below) were significantly greater (p 0.05) than rates in his freshwater study (Dufour, 1984), even though indicator concentrations were comparable. Dufour suggested that differential survival rates of indicators and pathogens in fresh water and seawater might explain this result. It was pointed out earlier in this report that certain viruses may be stabilized in saltwater, but many other factors have an influence. For one thing, most people avoid ingestion of seawater because of the disagreeable, salty taste. If they do get a mouthful of seawater, they will usually expel it. Another factor to be considered is poor reporting of enteric disease to health officials, especially in the past. As pointed out by Cabelli (1983) and Dufour (1984), most of the documented seawater-borne cases have resulted from grossly polluted environments. Furthermore, data from many of the reports in the literature are questionable.

An outbreak of typhoid fever occurred in Australia in 1958, and was linked to swimming in seawater contaminated by a broken sewage outfall (Flynn and Thistlewayte, 1964). Also linked to swimming in heavily polluted seawater was an outbreak of typhoid fever in Alexandria, Egypt (Wahdan, 1979).

In the only properly executed epidemiological study of seawater-borne enteric disease conducted in the United States, significant gastroenteritis was associated with swimming at beaches considered to be polluted, based on fecal indicator counts (Cabelli et al., 1982). The study was undertaken to determine if illnesses could be associated with swimming in sewage-polluted water and, if so, whether rates of illness could be related to some quantitative measure of water quality. The salient features of the study design were proximity of the beaches to large metropolitan areas (Coney Island in New York City, NY, proved to be the best model), use of a prospective cohort, and water quality monitoring during peak weekend swimming activity. With one exception (Stevenson, 1953), all other epidemiological investigations of water-borne disease have been retrospective. In retrospective studies, people diagnosed as having a particular disease are compared to persons free from that disease. The major fault of retrospective studies lies in the selection of a proper control group; in other words, their main weakness is in scientific design. Prospective studies do not usually suffer from improper scientific design; their main problem is

logistical, since they are very large, very expensive, and require a long period of time to complete (Mausner and Bahn, 1974).

Thorough, interpretive discussions of the indicator results are given in Cabelli (1983) and in Cabelli et al. (1979; 1982). Briefly, enterococci proved to have the best correlation with symptoms of gastroenteritis among swimmers, r = 0.96, for the New York study. No other indicator had a correlation greater than 0.65, and fecal coliforms, the currently accepted fecal indicator in the United States, had an r value of 0.51. Cabelli and his associates concluded that swimming in polluted seawater significantly increases the probability of contracting gastrointestinal disease. They further speculated that the cause of the acute illnesses observed could have been rotaviruses or Norwalk virus (Cabelli et al., 1982). However, no attempt was made to isolate and characterize the disease agent. Although this is a fault in the Cabelli study, it is an understandable one. Prospective studies are very expensive, and isolation and characterization of the disease agent would have added several million dollars to the cost of the study.

Indirect pathways have most frequently involved seafood. Both allochthonous and autochthonous pathogens are transmitted by this pathway. Shellfish, largely because of their feeding habits and grounds, are implicated more often than finfish. Bivalve mollusks, e.g., oysters, clams, scallops, and mussels, feed by sieving particles from water passing through their open shell. A single oyster can filter 1,500 liters of water per day in its search for food particles (Goyal, 1984). These particles contain viruses and bacteria, usually adsorbed to particle surfaces, which are transferred to the digestive tract of the mollusk. Once in the digestive tract, they become concentrated, either by accumulation (viruses and bacteria) or growth (bacteria). Both pathogens and nonpathogens become concentrated in this manner, and are therefore available to potential hosts when the mollusks are eaten raw or improperly cooked. Pathogen accumulation is not limited to filter-feeding mollusks (Goyal, 1984). Other shellfish have been shown to contain pathogens, and some of these studies and the organisms involved are discussed by Goyal (1984), Colwell (1985), and Sobsey (1985). Finally, shellfish are attracted to sewage-polluted waters, presumably because these waters are more productive, providing a better source of food. Japanese mariculturists are fully aware of this phenomenon and place oysters in the vicinity of sewage outfalls to achieve more rapid growth. Once a marketable size has been reached, the oysters are depurated with sanitized (usually ultraviolet light irradiated) water and sold.

Outbreaks of shellfish-borne disease have involved both viruses and bacteria. Most have resulted from the consumption of raw and/or improperly cooked meat, although some have been attributed to re-contamination of properly cooked material. Verber (1984) and The General Accounting Office, in a report to The Honorable Thomas J. Downey (GAO/HRD-84-36, June 14, 1984), listed all cases of shellfish illnesses reported since 1900; the total was approximately 12,000 cases. Some of the more recent outbreaks have been summarized by Goyal (1984), Morse et al. (1986) and Sobsey (1985), who noted an absence of *Salmonella typhi*. In past years, typhoid fever outbreaks were common among persons who ate raw oysters (Leake and Velda, 1925; Hart, 1945). However, largely because of the establishment of fecal indicator standards for shellfish

waters, such outbreaks of disease caused by salmonellae are now very rare. Instead, viruses, especially the Norwalk agent, now are responsible for the largest number of cases. The almost complete lack of correlation between fecal indicator bacteria and enteroviruses is a significant factor in the shift in etiology. This, combined with the increased contamination of shellfish beds and the fact that mollusks are frequently harvested from closed areas, a practice known as "bootlegging" (DuPont, 1986; GAO/HRD- 84-36, June 14, 1984), has contributed to an increased incidence of shellfish-borne disease. Indeed, such disease is the most serious immediate threat to human health deriving from the ocean, and the most common water-borne illness in the United States is that associated with consumption of shellfish. If the practice of disposing of pathogen containing wastes is allowed to continue, it will most certainly eliminate the shellfish industry as we now know it.

In the next 3 or 4 years, EPA is expected to publish an assessment of the relationships between gastrointestinal disease among shellfish consumers and the microbiological quality of water from which the shellfish are harvested (Dufour, U.S. EPA, Cincinnati, OH, personal communication). This document will probably provide a scientifically defendable criterion for establishing new microbiological standards for water and sediments in shellfish harvesting areas.

Human disease agents, especially those of small size, can be picked up and carried by other marine organisms, including microorganisms. The extent of this phenomenon in the marine environment is not known because most marine plants, animals, and microorganisms have not been examined for the presence of human pathogens. A case in point is the recent work on sharks and rays (Buck et al., 1984; Grimes et al., 1984, 1985a, 1985b). Sharks are susceptible to vibriosis and the disease agents include *Vibrio* spp. pathogenic to humans (Grimes et al., 1985a, 1985b). In addition, other human pathogens are carried by sharks, apparently as part of their normal (autochthonous) flora (Buck et al., 1984; Grimes, 1990). Transfer of these pathogens to humans by means of shark bites has been discussed (Buck et al., 1984; Grimes, 1990) and public health implications of shark meat consumption by humans has been reviewed (Grimes, 1990).

Bony fish also carry human pathogens, both allochthonous and autochthonous (Colwell and Grimes, 1984). Humans contract pathogens by consuming raw, undercooked, or recontaminated fish. Marine mammals are also capable of contracting and disseminating pathogens. An excellent example involves the bacteria and viruses that infect pinnipeds (Smith et al., 1978). The availability of these pathogens to humans is unclear, and warrants further study.

Acanthamoeba spp. have been shown to be capable of ingesting and maintaining in a viable state certain types of bacteria, including *Legionella* spp. Presumably, other marine and estuarine amoebae can also take up bacteria that they fail to kill for food. The potential of this phenomenon, the harboring and transmittal of pathogens by their predators, is obvious, and must be considered in light of increasing additions of waste to the ocean. It could be argued, for example, that intracellular Legionella pneumophila might be contributing to the disease caused by *Acanthamoeba*. However, recorded incidence of amoebic meningoencephalitis is rare; as of 1980, only 10 cases were known worldwide (DeJonckheere, 1980), and documentation for bacterial disease transmission by *Acanthamoeba* is nonexistent.

Marine plants probably have receptor sites for attachment of pathogens, although such associations have not been demonstrated. Indirect support for this hypothesis comes from recent descriptions of plant-microorganism interactions. Enteric bacteria (*Enterobacter* and *Klebsiella*) were shown by Haahtela et al. (1985) to adhere to the roots of bluegrass, *Poa pratensis*. Adhesion occurred by means of type 1 fimbriae (i.e., mannoside-sensitive fimbriae) to unknown carbohydrate receptors on the roots. Spira et al. (1981a) showed that *Vibrio cholerae* attaches to and concentrates on the surface of the water hyacinth, *Eichornia crassipes*. Another water-borne pathogen, *Pseudomonas aeruginosa*, can also associate with aquatic plants, including water lily (*Nymphaea tuberose* Paine), duckweed (*Lemna* spp.), arrowhead (*Sagittaria rigida* Pursh.), lotus (*Nelumbo pentapetala* [Walt.] Fern.), and algae (Pellett, 1977, M.S. Thesis, University of Wisconsin, La Crosse; Pellett, et al., 1983). In all of these examples, attachment to plants occurred in fresh water; however, marine plant-microorganism interactions have been observed and recorded (Sieburth, 1975) and some probably included pathogens. Finally, since many marine plants are harvested for food and food additives (agar, algin, carrageenan, etc.), the availability of pathogens by this route is real.

Buzzards Bay

Sources of Pathogens

Publicly Owned Treatment Works (POTWs)

Buzzards Bay contains numerous point and nonpoint sources of fecal indicator bacteria and pathogens. The most obvious potential point sources of human pathogens are the POTWs. The POTW discharges within the confines of the bay contribute a combined waste load of approximately 37.17 MGD (Table 3).

Although the disinfection performance of each POTW is monitored under the NPDES Permit System and periodically verified by The Massachusetts Department of Environmental Protection, Division of Water Pollution Control, occasional plant malfunctions and failures continue to make these facilities the primary potential source of human pathogens entering the bay. This statement is particularly true for the New Bedford facility, which discharges primary-treated effluent into Buzzards Bay at a rate of 30 MGD. During rain events, a portion of the wastes bypasses this treatment facility and is discharged through at least 29 combined CSOs located in the Acushnet River estuary and harbor and Clarks Cove (see section titled Stormwater Runoff).

Table 3. Summary of POTW discharges to Buzzards Bay, as of January, 1989.

Facility	Location	Volume (MGD)
Wareham	Agawam River	1.75
Marion	Aucoot Cove	0.34
Dartmouth	3,000 ft off Mishaum Point	2.0
Fairhaven	Immediately north of hurricane dike, New Bedford Harbor	3.0
New Bedford	3,000 ft from Clarks Point	30.0
Massachusetts Maritime Academy	Cape Cod Canal	0.08
Gosnold System	Buzzards Bay	<0.01
	Approximate Total	**37.17**

The effect of sewage treatment plants on the human pathogen concentrations in Buzzards Bay must be viewed in two ways. There is a perceived effect, as evaluated by the abundance of fecal indicator organisms discharged at the point source, and there is the actual pathogen loading of the system. A growing body of evidence strongly suggests that traditional fecal indicator organisms are not adequately portraying real pathogen loading to Buzzards Bay and other estuarine and coastal ocean systems (see section titled Fate of Pathogens).

Regardless of their actual pathogen loading, discharges from POTWs will always have an effect on shellfisheries in the vicinity. With approval from FDA, each state's shellfish sanitation program establishes a closure area around sewage outfalls, the extent of which depends on conditions specific to the site. One consideration is that, if the plant fails, allowing untreated sewage to enter the bay, jurisdictions of all affected waters must be rapidly notified to implement closure. In an area where notification and closure could not be carried out quickly enough to prevent possible harvesting after a plant failure, the area is kept permanently closed. These areas may, however, be used for controlled purification programs known as "relays." The closure area near the New Bedford POTW is approximately 5,000 acres.

Septic Systems

Many of the residents of the Buzzards Bay watershed use on-site sewage disposal systems for their sanitary wastes. The construction of these systems is regulated by 310 CMR 15.00 THE STATE SANITARY CODE TITLE 5: MINIMUM REQUIREMENTS FOR THE SUBSURFACE DISPOSAL OF SANITARY WASTES (Title 5), which sets minimum requirements for their design and placement. Three primary components govern the placement of a septic system: (1) the elevation of the site above groundwater, (2) the lateral distance between the leaching component of the facility and a point of water use (well, watercourse, surface waters, etc.), and (3) the suitability of the soils or sediments to receive the liquid effluent from the septic wastes.

The removal of pathogens from septic system wastes occurs primarily by two mechanisms, physical retention, or straining, by the receiving soil, and adsorption of pathogens on soil particles. The effect of these processes has been shown to be enhanced if the waste passes through soil that is not saturated with water. For this reason all states have adopted minimum distances of separation between the bottom of a septic leaching facility and the groundwater. In Massachusetts, this minimum allowable distance between the bottom of a leaching element of a septic system and the groundwater is 4 ft. To obtain better treatment of the wastes for pathogens, the town of Bourne has adopted a supplement to the State Regulations and requires a 6-ft minimum separation.

Although distance to groundwater, and hence treatment of wastes in the vadose, or unsaturated zone, is considered an important aspect of soil treatment, the lateral distance between the point where wastes enter the groundwater and the point where the wastes intercept a point of human contact is also important. For this reason, Massachusetts has adopted minimum lateral distances between the septic system components and points of water use (Table 4). Again, these regulations are designed to provide adequate distance for the effective treatment of the biological wastes either by physical processes (strain-

ing or adsorption) or biological/chemical processes (bacterial die-off and virus deactivation). A typical septic system is diagrammed in Figure 4.

In addition to vertical distance to groundwater and lateral distance to point of water use, the third major consideration in the placement of septic systems is the ability of the soils to receive septic wastes. In Massachusetts, this suitability is determined by results obtained from digging a "deep observation hole" and from performance of soilpercolation tests witnessed by a representative of the local board of health. The purpose of the deep observation hole is to determine and record the kinds of soil in the proposed leaching area and to evaluate groundwater elevation. Observations are generally made when groundwater is at its maximum elevation. In some instances, when testing is allowed outside the time frame of maximum groundwater elevation, observation wells can be used to adjust the observed elevation. Deep observation holes are excavated to a depth at least 4 ft below the bottom elevation of a proposed leaching facility.

Table 4. Minimum lateral distances (in ft) between components of septic systems and certain water supplies and/or sources

Water Supply and/or Source	Septic Tank	Leaching Facility
Well or suction line	50	100
Water supply line (pressure)	10	10
Surface water supplies (reservoirs or tributaries to reservoirs, including open and subsurface drains)	50	100
Watercourses	25	50
Subsurface drains	25	25

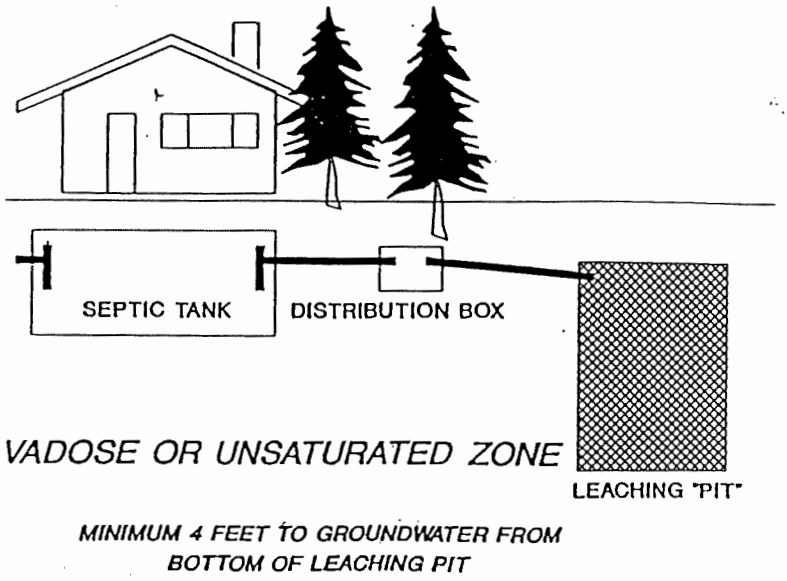

Figure 4. Schema of a typical septic system

Percolation tests at the proposed disposal site are used to determine the ability of the soil to accept water. Percolation tests are performed by digging a 12-in-diameter, 18-in-deep hole in the proposed material and introducing water to the hole in the prescribed manner. The percolation rate is determined by measuring the acceptance of the water (drop in water level in the hole) over a period of time. Under present Massachusetts regulation, any soils with receiving rates slower than 30 min/in (or 20 min/in for systems over 2,000 gal/d) are deemed unsuitable for septic system construction. In general the "faster" the soil, the smaller the surface area required for the leaching facility.

The suitability of the sediments in the Buzzards Bay Region to accept effluent wastes varies considerably. The two broad classifications of sediments present in the watershed, glacial moraine and outwash plain deposits, have quite different suitability characteristics. The eastern shore of Buzzards Bay (Bourne-Falmouth area) is dominated by outwash deposits which, due to their high permeability, are considered suitable for accepting effluent; however, this high permeability also provides less soil-treatment of wastes. On the eastern shore, the two areas not characterized by outwash deposits are the sediments around Hog, Mashnee and Toby's Islands, which consist of Buzzards Bay ground moraine deposits, and the area of North Falmouth towards Woods Hole, which contains Buzzards Bay moraine deposits (Oldale, 1976). These later deposits are primarily till and may contain silts and finer sediments not generally deemed acceptable for septic system placement.

Deposits on the western shore are also outwash and moraine. This shore tends to have a much greater relief above its bays and estuaries. The uplands are generally compact silty and bouldery till with occasional bedrock outcrops not found on the eastern shore. The soils in the till areas are considered of low suitability for septic system placement. The stream valleys, however, contain outwash deposits that are generally of high permeability and septic systems may be installed where the distance to groundwater is adequate.

Although subsurface sewage disposal is governed by state regulations, the enforcement of the regulation takes place at the local level through the boards of health. The degree to which on-site sewage disposal practices result in contamination of surface waters depends, therefore, partly on the diligence and expertise of the local enforcement agent. The health agent is the "first line of defense" to ensure that septic systems accompanying new construction are properly sited to prevent contamination. The health agent's responsibility begins during the planning of a project, when suitability test are performed. It is required that an observation hole and percolation test be witnessed by the agent of the board of health. The results of these tests determine the size and placement of the septic system on the lot. The health agent also reviews the septic system design and inspects the system prior to backfilling. At each of these junctures, the agent of the board of health has the authority to enforce compliance with Title 5.

Bacterial contamination of Buzzards Bay from septic systems can occur in at least three ways. Perhaps the most obvious public health threat occurs when a system experiences overt failure. Failure occurs when soils can no longer receive septic effluent, and sewage pools on top of the septic system, often

breaking out onto the surface of the ground. In some cases, this sewage, which may contain both bacterial and viral pathogens, can be transported to surface waters via stormwater drainage systems or overland flow. In general, overt failures are repaired quickly, since they are often offensive to the property owner and adjacent residents. The local board of health has full authority under Title 5 to require the repair of failing on-site septic systems. As a consequence of this clear authority and a general increase in public awareness of the contamination issue, overt system failure likely plays only a minor role in the overall pathogen contamination of Buzzards Bay.

Closely related to overt failure is the existence of overflow pipes. Such pipes were once connected to the leaching component of septic systems, to prevent failure and subsequent surface break out. Overflow pipes were often designed to empty directly into a major body of water or a connecting ditch or stream. The practice of connecting overflow pipes is thought to have been quite common in past years. Although these connections are illegal, some undoubtedly still exist in Buzzards Bay and will have to be corrected. The amount of contamination entering the bay from this source is uncertain. A series of sanitary surveys conducted on the eastern shore of Buzzards Bay revealed some overflow pipes; these pipes are being eliminated. Sanitary surveys soon to be conducted on the western shore will undoubtedly result in the discovery of more such connections, which will also require correction.

At present, there are no good estimates of fecal coliform loading to Buzzards Bay from failing septic systems or those with illegal overflow connections. The results of recent shoreline surveys by DMF, however, indicate that overt failures play a minor role in the overall fecal coliform contamination in the bay.

The third mechanism by which pathogens can enter Buzzards Bay from septic systems is through the groundwater. Weiskel and Heufelder (1990) found little entrainment of bacterial indicators in groundwater near five septic systems in Buttermilk Bay. However these authors provided a substantial review of literature and site-specific information suggesting the possibility of entrainment of viruses in groundwater entering the bay. On the basis of work performed in Long Island with similar soil types (Vaughn et al., 1983) and other such studies, these authors presented a convincing case that the present practices of siting septic systems in the Buzzards Bay communities, while likely providing adequate treatment of bacterial wastes, are not preventing the entrance of viruses into the bay.

Perhaps the most compelling evidence that current siting practices may allow for entrainment of virus by the groundwater entering the bay is presented by Yates (1987). Using an extensive review of the literature, this author developed a rating system for use in septic system siting. The index uses eight factors to assess the probability of contamination by septic system placement: distance to groundwater, annual recharge, hydraulic conductivity, groundwater temperature, soil type, aquifer medium, effluent application rate, and distance from leaching facility to point of water use. Computation of the index is demonstrated in Figure 5. Even when the requirements of Title 5 are met, for example, when the receiving sediments are fine sand or coarser, and liberal values are used for hydraulic conductivity (100 gpd/ft2) and effluent application rate (cm/day), computation of the index produces a numerical score in the range of "CONTAMINATION PROBABLE." The

example assumes a sediment and aquifer medium of fine sand (S=10, 5S=50 and A=10, 3A=30), a depth to groundwater of 5-10 ft (DTW=9, 5DTW= 45), an annual recharge 10 in (R=9, 2R=18), hydraulic conductivity 100 gpd/ft2 (K=1, 3K=3), a temperature range 10-12.5oC (T=7, 2T=14), an application rate 5 cm/da (AR=1, 4AR=4) and a distance to water use 100 ft (D=8, 5D=40). Thus the equation is:

INDEX = 5DTW + 2R + 3K + 2T + 5S + 3A + 4AR + 5D
INDEX = 45 + 18 + 3 + 14 + 50 + 30 + 4 + 40
INDEX = 204 CONTAMINATION PROBABLE

Stormwater Runoff

Stormwater runoff refers to that portion of the precipitation returned to a waterbody via surface routes from the adjacent land mass. Although precipitation is generally devoid of fecal indicator organisms when it falls, as it flows over the earth's surface it collects debris and sediment and washes them into the surface waters. This debris may be composed of or contaminated with

INDEX = 5DTW + 2R + 3K + 2T + 5S + 3A + 4AR + 5D

DTW = DISTANCE TO GROUNDWATER
Ranges and ratings for depth to water

Range (ft)	Rating
0-5	10
5-10	9
10-30	7

WEIGHT = 5

R = RECHARGE TO AQUIFER
Ranges and ratings for net recharge

Range (in/yr)	Rating
0 - 2	1
2 - 4	3
4 - 7	6
7 - 10	8

WEIGHT = 2

K = HYDRAULIC CONDUCTIVITY
Ranges and rating for hydraulic conductivity

Range (gpd/sq ft)	Rating
1-100	1
100-300	2
300-700	4
700-1000	6
1000-2000	8
2000	10

WEIGHT = 3

T = TEMPERATURE
Ranges and ratings for temperature

Range	Rating
<5 C	10
5 - 10	9
10 - 12.5	7
12.5 - 17	5
17 - 20	4
20 - 25	2

WEIGHT = 2

S = SOIL TYPE
Ranges and ratings for soil type

	DEPTH TO WATER		
Soil Type	45 m	9 m	1.1 m
fractured rock	10	10	10
coarse gravel	9	10	10
coarse sand	8	10	10
fine sand	7	10	10
sandy loam	6	8.6	10
loam	5.2	7.4	10
sandy clay loam	4.2	6.1	10
clay loam	3.1	4.4	7.7
sandy clay	2.5	3.6	6.2
clay	1	1.4	2.5

WEIGHT = 5

continued next page...

Figure. 5. Computation of rating index for determining the probability of virus entrainment in groundwater near septic systems (adapted from Yates (1987)

continued from previous page...

A = AQUIFER MEDIUM

Ranges and ratings for aquifer medium

	DISTANCE TO POINT OF WATER USE			
Soil Type	200 m	20 m	2 m	
fractured rock	10	10	10	
coarse gravel	10	10	10	
coarse sand	8.9	10	10	
fine sand	7.8	10	10	
sandy loam	6.7	8.6	10	WEIGHT = 3
loam	5.8	7.4	10	
sandy clay loam	4.7	6	8.4	
clay loam	3.4	4.4	6.2	
sandy clay	2.8	3.6	5	
clay	1.1	1.4	2	

AR= APPLICATION RATE

Ranges and rating for effluent application rate

Range (cm/day)	Rating
<5	1
5 - 13	2
13 - 45	3
45 - 100	4
100 - 360	5
360 - 920	7
920 - 2000	9
2000 - 3300	10

WEIGHT = 4

D=DISTANCE TO POINT OF WATER USE

Ranges and ratings used for separation distance between septic system and point of water use

Range (ft)	Rating
0 - 50	10
50 - 75	9
75 - 100	8
100 - 125'	7
125 - 150	6
150 - 200	5
200 - 300	4
300 - 500	3

WEIGHT = 5

INDEX = 5DTW + 2R + 3K + 2T + 5S + 3A + 4AR + 5D

INDEX RANGES	
0-75	CONTAMINATION NOT VERY PROBABLE
75-150	CONTAMINATION POSSIBLE
150-255	CONTAMINATION PROBABLE
225	CONTAMINATION VERY PROBABLE

human or animal wastes. In the past, certain activities of man have increased the proble. In attempting to drain road surfaces for safety purposes, the general practice has been to pipe or berm stormwater collecting on roadways to the nearest low point. This has often been a pond, stream, estuary or other surface waterbody. As a result, contaminants from adjacent uplands often find an overland route to surface waters. In agricultural areas, the "sheet" flow from adjacent land masses is of particular significance. In this case, the flow is unconsolidated and enters the receiving water in broad, less defined areas instead of being collected and discharged through pipes.

Investigations in Buzzards Bay have confirmed the findings of the National Urban Runoff Program (NURP) and other studies pointing to stormwater runoff as a major contributor of fecal indicators to surface waters. The contribution of fecal indicators by stormwater has an added implication for the use of the area for shellfishing, because bacteriological sampling to classify the shellfish harvesting areas is conducted during "adverse pollution conditions" (NSSP guidelines), generally following a rain event. Within the Buzzards Bay watershed, both agricultural runoff, which dominates the western portion of the bay near Westport, and urban runoff, which dominates New Bedford and other residential areas near cities and towns, can be found entering the bay at

discrete points such as pipes and open ditches or in broader, less defined areas of sheet flow.

In the Acushnet River Estuary and Clarks Cove, CSOs discharge stormwater runoff mixed with sanitary wastes that bypass the New Bedford POTW during rain events. These CSOs are the most overt source of human pathogens in the bay. In addition, the sediments in the area of the discharges will probably serve as reservoirs for human pathogens for a longer period of time than suggested by fecal indicator measurements of the overlying waters (Goyal et al., 1978; LaBelle and Gerba, 1980 and 1982; Liew and Gerba, 1980; Rao et al., 1984). An annotated bibliography of the literature regarding the persistence of pathogens and fecal indicators in sediments has been compiled for the Buzzards Bay Project by Heufelder and Rask (1989).

To address the CSO problem, the City of New Bedford is beginning the process of upgrading its POTW. As of this date, however, it is still undetermined how many CSOs will be eliminated by the new facility.

Table 5 is a partial list of stormwater pipe discharges entering Buzzards Bay and its tributaries. These data are part of the 1989 sanitary survey conducted by the Massachusetts Division of Marine Fisheries (DMF) and include only samples analyzed at the Barnstable County Health and Environmental Department Laboratory. The remainder of Buzzards Bay sources were analyzed at different laboratories, and the results were not available for this report.

These data show, with few exceptions, that the highest densities of fecal coliform from stormwater pipes investigated came from CSOs. In particular, CSO#003, which flowed even during dry weather (Frank Germano, DMF, personal communication) had fecal coliform densities consistently above 200,000/100 ml. This CSO, which has only recently been diverted into the treatment plant, often had dry-weather flows approximating 50 gal/min. Using this flow, the lowest value observed at this CSO (200,000 fecal coliform/100 ml), would result in an effluent discharge of 72,000 gallons of discharge per day which, is 14,300 times the acceptable level for shellfish harvesting areas. The daily dilution water required to have resultant fecal coliform levels of 14 fecal coliform/100 ml near this discharge would be 3,160 acre-feet (3,160 acres of surface area with a depth of 1 ft or 316 acres with a depth of 10 ft). Effluent quality at this CSO was highest during the rain event of September 15, 1989 (Figure 6), suggesting that the sanitary waste, which consistently flowed even during dry weather, was being diluted by the urban runoff entering the system.

Any CSO that flows continuously probably represents a failed system by which sanitary wastes are bypassing the treatment plant. The Lucas Street CSO in New Bedford appears to be an exception to this rule. Unlike other CSOs sampled on the same date (November 12, 1989), the Lucas Street CSO showed very low densities of fecal coliform (15/100 ml). In this instance, the "continuous flow" may merely represent uncontaminated groundwater infiltrating various parts of the system and subsequently being discharged. In general, however, the highest priority should be placed on the correction of continuously flowing CSOs, and shellfish harvesting should be prohibited within their zones of influence.

Table 5. Fecal coliform densities from selected stormdrains in the Buzzards Bay Watershed. Data are part of the sanitary surveys conducted by the Massachusetts Division of Marine Fisheries under the Shellfish Sanitation Program, 1989.

LOCATION	SAMPLING DATE	FECAL COLIFORM/100 ML
DARTMOUTH - APPONAGANSETT BAY		
Culvert A	08-Aug-89	24,000
Culvert A	15-Aug-89	1600
Culvert A	04-Oct-89	340
Bay Street Storm Drain	08-Aug-89	250
Knowles Pier Pipe	29-Jun-89	2300
Knowles Pier Pipe	08-Aug-89	100,000
Knowles Pier Pipe	15-Aug-89	1500
Knowles Pier Pipe	19-Sep-89	3700
Bridge Street Bridge Pipe	12-Jul-89	2600
Ross Pipe	19-Sep-89	3700
Joy Landing Pipe	19-Sep-89	1700
Culvert - Patrol Boat Dock	27-Jun-89	1090
Prospect Street - 48" Corrugated	19-Sep-89	1300
Fort Street - 48" Cement Pipe	19-Sep-89	1400
Shipyard Lane - 12" Cement Pipe	19-Sep-89	13,000
Russell Mill Road Culvert	16-Aug-89	3500
Russell Mill Road Culvert	19-Sep-89	5400
Gladys Street Stormdrain	04-Oct-89	20
Day Street Stormdrain	04-Oct-89	30
NEW BEDFORD - CLARK'S COVE		
18" Pipe - Mosher Point	12-Nov-89	70
Rogers Street - Right Pipe	28-Aug-89	2400
Rogers Street - Right Pipe	19-Sep-89	11,000
Rogers Street - Right Pipe	04-Oct-89	150
Rogers Street - Right Pipe	22-Oct-89	40
Rogers Street - Right Pipe	12-Nov-89	40
Rogers Street - Left Pipe	28-Aug-89	1100
Rogers Street - Left Pipe	19-Sep-89	15,000
Rogers Street - Left Pipe	04-Oct-89	80
Rogers Street - Left Pipe	22-Oct-89	150
Rogers Street - Left Pipe	12-Nov-89	80
Pipe At Clarks Cove Fish Market	12-Nov-89	200
CSO #003	16-Aug-89	700,000
CSO #003	28-Aug-89	220,000
CSO #003	19-Sep-89	200,000
CSO #003	04-Oct-89	320,000
CSO #003	22-Oct-89	330,000
CSO #003	12-Nov-89	200,000
Woodlawn Street Pipe	12-Nov-89	140
Wall Pipe - End Of Beach	12-Nov-89	10
Lucas Street CSO	12-Nov-89	10
Capital Street CSO	12-Nov-89	1000
Calumet Street CSO	12-Nov-89	68,000
Bellevelle Street CSO	12-Nov-89	410,000
CSO #005	22-Oct-89	10,000
FAIRHAVEN - OUTER HARBOR		
Winsegansett Culvert	26-Jun-89	150
Culvert, Hurricane Barrier	06-Nov-89	190
Culvert At Barrier	11-Oct-89	6100
FAIRHAVEN - WEST ISLAND SOUTH		
Littleneck Lane Culvert	11-Jul-89	70
Storm Drain - Little Neck Road	31-Jul-89	400
Little Neck Road - Culvert	16-Aug-89	4500
Littleneck Road Culvert	17-Sep-89	1000
Littleneck Road Culvert	18-Oct-89	130
1 Nakata St Culvert	12-Jun-89	810
1 Nakata St Culvert	26-Jun-89	1000
Island View Rd. Culvert	12-Jun-89	3800
Island View Rd. Culvert	26-Jun-89	1900
Island View Rd. Culvert	17-Jul-89	19,000
Island View Rd. Culvert	16-Aug-89	6100
Island View Rd. Culvert	17-Sep-89	7000
Island View Rd. Culvert	18-Oct-89	360

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Gull Island Road Culvert	17-Sep-89	200
Storm Drain - Gull Island Road	26-Jun-89	10
Storm Drain - Gull Island Road	11-Jul-89	60
Storm Drain - Gull Island Road	31-Jul-89	150
Storm Drain - Gull Island Road	16-Aug-89	20
Fisherman Road Culvert	11-Jul-89	70
Fisherman Road Culvert	17-Sep-89	900
Fisherman Road Culvert	18-Oct-89	1300
FAIRHAVEN - WEST ISLAND NORTH		
Causeway Rd. Culvert	05-Jul-89	100
Causeway Rd. Culvert	06-Jul-89	930
14" Culvert Jack's Cove	05-Jul-89	100
18 Causeway Rd. Culvert	06-Jul-89	105
18 Causeway Rd. Culvert	11-Jul-89	80
Cement Pipe - Causeway Rd.	12-Jun-89	10
Cement Pipe - Causeway Rd.	29-Jun-89	10
Cement Pipe - Causeway Rd.	11-Jul-89	330
Cement Pipe - Causeway Rd.	13-Jul-89	10
Cement Pipe - Causeway Rd.	17-Jul-89	30
Cement Pipe - Causeway Rd.	17-Sep-89	2000
Metal Pipe At Private Beach Sign	12-Jun-89	4800
Metal Pipe At Private Beach Sign	05-Jul-89	100
Metal Pipe At Private Beach Sign	06-Jul-89	750
Metal Pipe At Private Beach Sign	17-Jul-89	4800
Metal Pipe At Private Beach Sign	7-Sep-89	4000
Metal Pipe At Private Beach Sign	18-Oct-89	370
Storm Drain Blue Point Rd.	06-Jul-89	16,600
Storm Drain Blue Point Rd.	11-Jul-89	440
Storm Drain Blue Point Rd.	31-Jul-89	100
Storm Drain Blue Point Rd.	16-Aug-89	2000
Storm Drain Blue Point Rd.	17-Sep-89	4000
FAIRHAVEN - NASKETUCKET BAY		
Howard Beach Culvert	06-Nov-89	10
Ocean Avenue Culvert	26-Jun-89	380
Ocean Avenue Culvert	11-Jul-89	3300
Ocean Avenue Culvert	17-Jul-89	87,000
Ocean Avenue Culvert	16-Aug-89	2900
Ocean Avenue Culvert	17-Sep-89	8000
Ocean Avenue Culvert	24-Oct-89	100
Ocean Avenue Culvert	05-Nov-89	300
FAIRHAVEN - LITTLE BAY		
East Culvert At Railroad	13-Jun-89	20,000
East Culvert At Railroad	21-Aug-89	2900
East Culvert At Railroad	19-Sep-89	1900
East Culvert At Railroad	24-Oct-89	100
West Culvert At Railroad	13-Jun-89	1700
West Culvert At Railroad	21-Aug-89	260
12" Pipe On Map	24-Oct-89	100
S.d. Opposite 39 Weeden Road	06-Nov-89	30
Culvert Behind STP	13-Jun-89	3100
Raymond Street Culvert	26-Jun-89	430
Raymond Street Culvert	11-Jul-89	2900
Raymond Street Culvert	17-Jul-89	210,000
Raymond Street Culvert	16-Aug-89	11,000
Raymond Street Culvert	21-Aug-89	310
Raymond Street Culvert	17-Sep-89	1000
Raymond Street Culvert	24-Oct-89	100
Raymond Street Culvert	05-Nov-89	10
WAREHAM - BOURNE COVE/LITTLE HARBOR		
Warren Point Road Culvert	28-Nov-89	4800
3" Pipe Flowing	28-Nov-89	39,000
FALMOUTH		
First Culvert West Of Bridge	15-Nov-89	10
12 Uncantena Road - 10" Culvert	02-Oct-89	10
Culvert - North End Of Racing Lane	02-Oct-89	10
6" Pipe Next To Boat Yard	02-Oct-89	2200

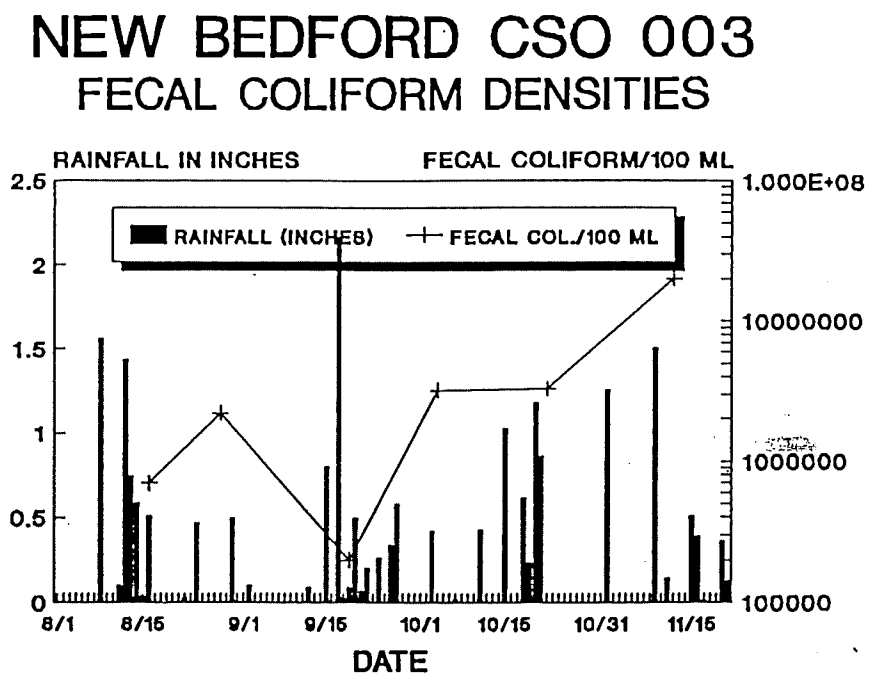

Figure 6. Densities of fecal coliform in effluent from CSO #003 on selected dates. Date from DMF shoreline survey work, 1989.

In addition to the CSOs of the New Bedford area, which have a known connection between the stormwater and sanitary systems, runoff enters the bay from other urban areas in which this type of cross connection has not been found. One such area is Buttermilk Bay, located at the head of Buzzards Bay. Investigating over 20 discharge points for stormwater failed to locate any sanitary facility cross connections with the stormwater systems, yet fecal coliform densities exceeding 104/100 ml in stormwater discharges were common (Heufelder, 1988). This work concurred with that of Olivieri et al. (1977), who obtained similar results in a highly urbanized setting. In Buttermilk Bay, the relative densities of fecal indicators at discharge points depended on factors such as the density of residential development, the period of time since the previous rain event, and the air temperature. In general the highest fecal indicator levels were found in summer, following a long period of dry weather, at discharges serving the highest residential density.

Data from the non-CSO stormwater discharge pipes (Table 5) indicate that these too often contain high densities of fecal coliforms. These pipes exhibit an extreme spatial and temporal variability in the fecal coliform densities observed. In evaluating the relative importance of each discharge pipe, it is important make the comparison with other discharges sampled on the same date. For instance, the Gladys Street and Day Street stormdrains in the Dartmouth - Apponagansett Bay area might, at first glance, appear to be minor in implication

compared with the other stormdrains in the area. However, closer examination shows that the only data reported are from October 4, 1989. Culvert A in the same area and the Rogers Street drains in the New Bedford - Clarks Cove area were sampled on several dates including October 4. Data from these two discharges indicate that values from the range of 2-4 magnitudes less were reported on October 4 from these sites. These results suggest that the October 3 rain event had some characteristic that produced fecal coliform concentrations lower than normal in stormwater discharge pipes. Thus the low fecal coliform densities from the Day and Gladys Street stormdrains were likely at least 2-3 orders of magnitude higher during the warmer months. The generally lower fecal coliform levels on October 4 were probably due to rainfall in the preceding weeks (Figure 7), which could have washed away a significant portion of the contaminant load.

The sources of fecal indicators and pathogens in urban situations where there are no sanitary cross connections have been the subject of considerable speculation. Since the feces of all warm-blooded animals contains E. coli, presumably the most frequent fecal coliform, it is reasonable to conclude that some of the fecal coliforms observed in urban stormwater runoff originate from dogs, cats, birds, and other domestic and wild animals. In addition, a failing septic system where sanitary wastes are pooled on top of the ground may occasionally find a surface pathway to the receiving water during a rain event. Heufelder (1988)

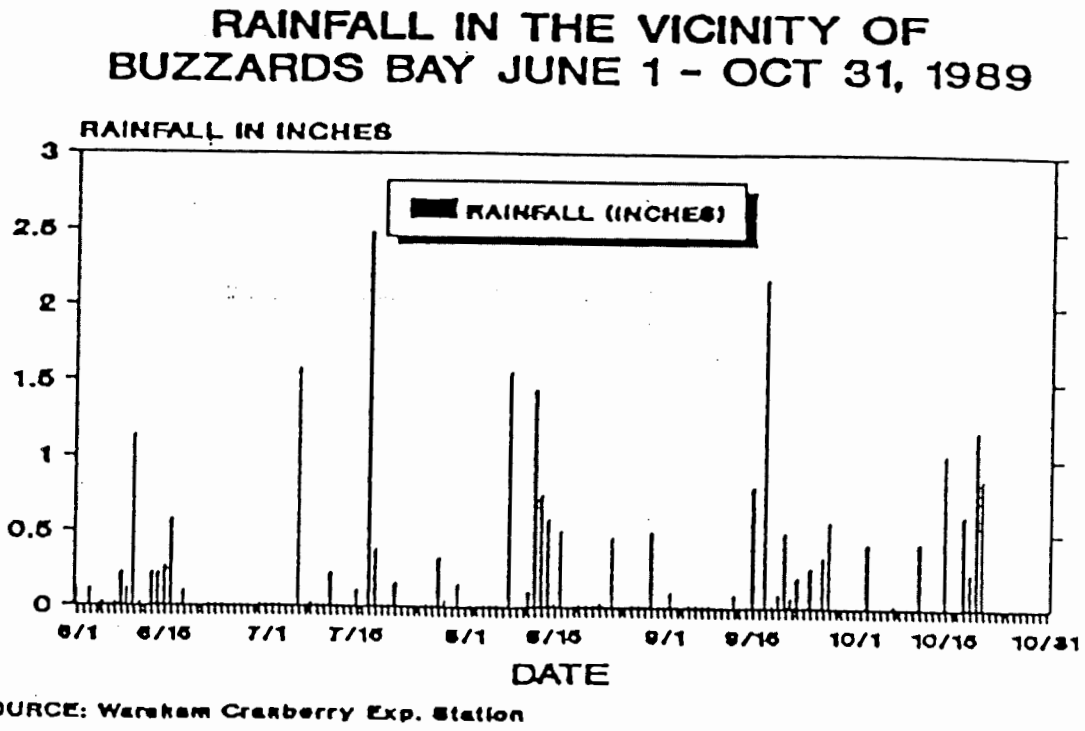

Figure 7. Rainfall in the Buzzards Bay area June 1 - October 31, 1989.

estimated that the fecal coliform loading during a 2.54 cm (1 in) rain into Buttermilk Bay, located at the head of Buzzards Bay, was equal to the fecal coliforms present in 150 kg (331 lb) of dog wastes. This is estimated equal to a 2 to 3-d accumulation of the wastes from dogs within the surface watershed.

The extensive use of the western shore of Buzzards Bay, particularly near Westport, for agriculture makes this area highly susceptible to agricultural runoff. Unlike urban runoff, which is generally consolidated into pipes or ditches, agricultural runoff frequently occurs as sheet flow, which enters streams and receiving waters in broad areas along the bordering steep slopes. Undoubtedly, fecal coliforms from this type of runoff originate primarily in animal feces. Animal raising and use of manure on crops are both sources of feces.

The public health implication of stormwater contamination has been the subject of much controversy. Where waste from failing septic systems is making its way into the stormdrain systems, few would argue that no threat to the public health exists. However, no epidemiological studies have linked the transmission of disease to water contaminated by stormwater runoff. Nonetheless, runoff often gives a "signal" to public health enforcement agencies that is currently indistinguishable from that of sewage. For this reason, the conservative choice -- regulating the resource based on the fecal indicator organism -- appears to be in the best interests of public health.

Discharges from Marine Craft

There are over 4,300 slips and moorings in Buzzards Bay (Gil, 1988), and nearly 20,000 vessels pass through the Cape Cod Canal yearly. Marine sanitary wastes are thus a potential source of contamination to bay waters. Because of the intermittent and often covert nature of the disposal methods, the overall impact of sanitary wastes on Buzzards Bay is difficult to assess. In order to understand the nature of the problem, a review of the regulatory framework and current disposal practices is necessary.

To estimate the fecal coliform loading from marine craft, a number of assumptions are made. Using the estimate of slips and moorings given by Gil (1988) and assuming that this approximates the number of resident boats, there are 4,300 boats in Buzzards Bay. The National Marine Manufactures Association indicates that boats exceeding 26 ft are usually equipped with marine sanitation devices (MSDs). Data from the Massachusetts Division of Law Enforcement indicate that approximately 25% of the registered boats exceed the 26-ft length. Thus the total number of potential dischargers in Buzzards Bay is approximately 1,075. A formula used by the Interstate Shellfish Sanitation Conference (ISSC) for determining the maximum number of boats that waterbodies can sustain while maintaining acceptable water quality is used to estimate the fecal coliform loading. The formula makes the following assumptions:

- 100% occupancy
- Overboard discharge by all occupied boats
- 2 persons to a boat
- A discharge of 2×10^9 fecal coliform/person/day

Using these figures, the fecal coliform loading from resident boats in Buzzards Bay is approximately 4.3 X 10^{12}/d. To place this value in perspective, Table 6 compares the loading from boats with that of selected CSOs sampled during the recent sanitary surveys.

The importance of sanitary wastes from boats is site specific. In poorly flushed situations, the amount of dilution water is limited and the effect may be substantial. Assuming a two-person occupancy, the zone of influence in which the water will reach a fecal coliform density of 14 fecal coliforms/100 ml near a boat discharging in 10-ft deep water is approximately 2.3 acres (22.9 acre-feet or 10^6 ft^3 of water required for dilution). This may be reduced in harborages that receive substantial tidal influence.

Table 6. Comparison of estimated fecal coliform loadings from marine craft discharges and selected CSOs in Buzzards Bay.

Source	Fecal Coliform/Day
Boat Discharge based on assumptions given in text	4.3×10^{12}
CSO#003 based on a mean of six sampling dates	1.3×10^{13}
CSO#010 Bellevelle St. based on 12 Nov. sample	2.3×10^{11}
CSO#008 Calumet based on 12 Nov. sample	1.8×10^{10}
All POTWs combined (37.2 MGD) assuming 5 fecal coliform/100 ml at discharge	7.0×10^{9}
All POTWs combined (37.2 MGD) assuming 200 fecal coliform/100 ml at discharge	1.8×10^{11}

The owner of a boat containing sanitary facilities has several alternatives for the disposal of sanitary wastes. First, whether or not the craft has proper storage facilities for the wastes, discharge into marine waters is allowed beyond the 3-mi limit. Discharges at this point may or may not be macerated and disinfected. Second, discharge within the 3-mi limit is allowed if the waste is properly disinfected so that the resulting effluent does not exceed a fecal coliform count of 200 FC/100 ml. Third, boats equipped with storage tanks may have the tank pumped out and the wastes disposed of properly. The third method is environmentally preferred because it eliminates the potential for contamination of recreational waters. Unfortunately, this option is not available in most cases. Although Massachusetts General Law, Chapter 91 Section 59 B, requires all marinas to provide "adequate facilities for the collection, treatment, and disposal of sewage and other sanitary wastes" in order to be issued a marina license, only a few marinas in Buzzards Bay have such pumpout facilities.

The Executive Office of Environmental Affairs has recently formed a task force to identify the technical and political issues involved in this problem. In general, under the present guidelines and policies, a marina operator wishing to comply with the licensing conditions faces a number of unresolved technical problems and requirements. Until the technical issues are resolved and marine sanitary waste pumpout facilities are installed and used, marine sanitary wastes will continue to be a significant potential source of pathogens in Buzzards Bay.

Data presented by Beskenis (1989) did not conclusively demonstrate the efficacy of pumpout facilities, but some interesting results were obtained. While sampling in Buzzards Bay where pumpout facilities were present, as well as in marinas where these facilities were not available, the author found similar indications of sanitary discharge from marine craft. These results suggest that, in addition to increasing the number of pumpout facilities, ways must be found to foster their use. Any program to eliminate marine sanitary discharges in the bay must combine regulation, enforcement and education.

Wildlife and Waterfowl and Domestic Animals

Buzzards Bay contains a variety of aquatic habitats that provide sanctuary to wildlife. A number of sheltered embayments and marshes on both shores of the bay not only support a year-round population of mammals and birds, but also act as harborage for many migratory waterfowl, particularly Canada geese, which often overwinter there.

It is a common belief that waterfowl, and in particular Canada geese, contribute significant levels of fecal indicators in the shellfishing areas of Buzzards Bay. This contention is not difficult to understand, considering that large rafts of over 150 Canada geese are common to many of the sheltered embayments throughout the winter. In Buttermilk Bay, Heufelder (1988) observed the maximum occurrence of waterfowl during late summer to early spring when it was common to observe over 100 Canada geese and over 200 ducks within a 2-km^2 area of the bay.

The potential for contribution of fecal indicators from waterfowl can be substantial. Daily estimates of 10^7 fecal coliform per goose and 10^9 fecal coliform per duck have been reported (Hussong et al., 1979 and Koppelman; Tanenbaum, 1982). This can be compared with the per diem estimate of 2×10^9 fecal coliform for humans.

Monitoring efforts in Buttermilk Bay have suggested that, in contrast with reports for human sewage discharge, the effects of waterfowl fecal deposits may be cumulative over several days due to persistence of the fecal indicator in the environment (Heufelder, 1988). This author found that fecal wastes from Canada geese remained viable for 2-3 weeks, especially when the wastes became entangled in the wrack or strand line deposits. It was suggested that the wrack line protected these wastes from the bactericidal effects of ultraviolet light and desiccation. It was further suggested that waterfowl wastes deposited in the intertidal zone frequently become resuspended and hence reintroduced into the receiving water with each incoming tide.

The effect of waterfowl waste in Buzzards Bay must be considered on a case-by-case basis. In situations where there is poor flushing, concentrations of waterfowl can undoubtedly elevate fecal coliform counts, especially if the animals defecate in areas of soft organic sediment which harbors the fecal indicators and prolongs their viability. When sediments are resuspended, as during a storm or tidal surge, fecal coliforms may be reintroduced to the water column, decreasing water quality. In addition, there is evidence that when animal waste gets entangled with beach wrack, the survival characteristics of the waste are also altered and may cause a slow diffuse leaching of indicators

from the wrack with each inundating tide. Conversely, when waterfowl defecate in well-flushed areas, the forces of dilution and dispersal act to lessen the impact on the environment.

Despite the contribution of fecal indicators by waterfowl, their concurrent contribution of human pathogens has been questioned. However, many investigators have demonstrated the presence of *Salmonella* in avian feces (Faddoul and Fellows, 1966; Mitchell and Ridgwell, 1971; Berg and Anderson, 1972;) and others have reported *Campylobacter* (Hill and Grimes, 1984).

Autochthonous Pathogens and Indicators

There are no available data that support or refute the hypothesis that indigenous pathogens and indicators exist in Buzzards Bay. Based on reports for other bodies of water, it must be assumed that at least some microorganisms native to the bay are either pathogenic for humans or give a positive fecal indicator test. For example, Vibrio species are ubiquitous in salt water and certain species are human pathogens. The same line of reasoning can be made for autochthonous indicators, including strains of Klebsiella pneumoniae and Aeromonas growing on nutrients present in the bay. Near sewage outfalls and CSOs, it is likely that both pathogens and indicators are building up in sediments, subject to resuspension by various physical mixing forces (e.g., storm events, dredging, boats) as described by Grimes (1980) and by Gerba and McLeod (1976).

Growth-Stimulating Nutrients

Few studies have provided data on the contributions of autochthonous and allochthonous nutrients capable of supporting growth of fecal indicator bacteria; pathogenic bacteria; and other free-living, heterotrophic microorganisms. Aquatic plants provide sugars and other simple carbohydrates that are assimilable by heterotrophic microorganisms, including indicator and pathogenic bacteria (Simidu and Tsukamoto, 1985; Heufelder, 1988). Other aquatic organisms provide more complicated organic compounds, such as chitin, cellulosics, phenolics, and glycoproteins, and these too can be utilized for growth by resident microorganisms. Finally, xenobiotic compounds, including polyaromatic hydrocarbons, herbicides, surfactive agents, and chlorinated hydrocarbons, can stimulate certain metabolically diverse, opportunistic pathogens such as *Pseudomonas aeruginosa* and *Vibrio* species.

Current Microbiological Status of Buzzards Bay

Coliform Surveillance

The microbiological status of Buzzards Bay can be inferred from a number of past and ongoing studies performed by various state and federal agencies. At present, however, there is no sampling program designed to provide long-term analysis of general trends regarding bacteriological quality of the bay. The most frequent bacteriological sampling program is conducted by the Massachusetts Division of Marine Fisheries (DMF), under the Shellfish Sanitation Program. Under this public health program, the agency monitors specific stations within each shellfish harvesting area at least five times per year. In addition, under the National Shellfish Sanitation Program or Interstate Shellfish Sanitation Conference's (ISSC) guidelines, DMF conducts sanitary surveys that sample all potential sources of contamination every 9 y and reviews and updates the

surveys every 3 y. This monitoring program has incorporated an action level which, when reached, causes the closure of an area to shellfishing. Currently, under the FDA-approved program, an area is closed if the geometric mean of the most probable number (MPN) exceeds 14 fecal coliforms/100 ml and not more than 10% of the samples exceed a MPN of 43 for a 5-tube, 3-dilution test. The present status of the shellfish harvesting areas in Buzzards Bay is summarized in Table 7. As of January 1, 1990, nearly 14,000 acres of shelfish harvesting areas were closed due to bacterial contamination.

An analysis of over 10,000 bacteriological samples taken in Buzzards Bay from 1985 to 1989 reveals no significant trends in water quality degradation. Figure 8 indicates no significant trends in the number of samples that exceed the 14 fecal coliform/100 ml of sample standard, nor have there been significant changes in the geometric mean of fecal coliform/100 ml of sample from 1985 to 1988. The apparent improvement in water quality during 1989 is due to the exclusion of summer-fall sampling dates for this analysis.

In order to interpret these data, it is important to ensure that relative sampling effort across the years examined has remained fairly constant. For example, if sampling in one year was skewed toward the more contaminated areas, the geometric or log mean of fecal coliform in samples, as well as the percentage of samples exceeding 14 fecal coliform/100 ml might be elevated in that year. Table 8 summarizes the percentage of samples taken in each area of Buzzards Bay. From 1985 to 1988 there were no significant shifts in sampling effort. In 1989, however, the areas of Dartmouth, New Bedford and Fairhaven received significantly more attention. The results of this shift in effort, even though the entire year's data have not been entered, are perplexing, since this apparent proportional increase in sampling effort in more contaminated areas near New Bedford did increase in geometric mean of densities of fecal coliform (Figure 8).

In addition to routine monitoring of stations in Buzzards Bay, DMF has recently completed shoreline surveys of most areas in Buzzards Bay. Besides testing the water overlying shellfish harvesting areas, these surveys will report on the water quality of stormwater and other discharges (See section titled Stormwater) as well as the streams and rivers of the watershed. Water quality values for some Buzzards Bay tributary waters are presented in Table 9.

These data indicate that, in many cases, these freshwater streams contribute significant fecal coliform inputs to the Buzzards Bay system. For the most part, the sources of fecal coliforms in these tributary streams have not been investigated, but probably result from use of the area by wildlife and waterfowl as well as autochthonous sources.

Table 7. Summary of shellfish harvesting areas closed due to bacterial contamination as of January 1, 1990. Source: Massachusetts Division of Marine Fisheries.

Town	Area Name	Acreage Closed
Acushnet	Acushnet River	58.6
Bourne	Back River	111.5
	Cape Cod Canal	526.4
	Hen Cove	4.0
	Pocasset Harbor	12.9
	Pocasset River	40.0
	Queen Sewell Cove	14.3
	Red Brook Harbor	30.0
	Butlers Cove	192.0
	Buttermilk Bay	44.0
	Onset Bay	4.0
	TOTAL:	**979.1**
Dartmouth	Apponagansett River	69.6
	Apponagansett River	37.2
	Clark's Cove	974.6
	Salter's Point	619.2
	Slocums River	423.7
	TOTAL:	**2124.3**
Fairhaven	Shaw's Cove Creek	5.0
	Acushnet River	354.8
	Little Bay	43.5
	Shaw's Cove Creek	5.0
	Acushnet River	354.8
	New Bedford Outer Harbor	2856.7
	New Bedford, Fairhaven Dartmouth outer Harbor	727.9
	TOTAL:	**3987.9**
Falmouth	Herring Brook	8.6
	Quisset	36.3
	TOTAL:	**44.9**
Marion	Briggs Cove	10.7
	Hammett Cove	20.3
	Weweantic River	105.6
	Aucoot Cove	11.4
	Weweantic River	78.0
	Wings Cove	30.5
	Sippican Harbor	33.0
	TOTAL:	**289.5**
Mattapoisett	Eel Pond	26.3
	Mattapoisett Harbor	75.1
	Mattapoisett River	36.3
	Aucoot Cove	14.3
	Shaw Cove	76.0
	TOTAL:	**241.1**
New Bedford	Acushnet River	689.9
	Clarks Cove & New Bedford Outer Harbor	3478.3
	TOTAL:	**4168.2**
Wareham	Buttermilk Bay	27.0
	Wareham River	**32.9**
	Wareham River	4.3
	Wareham River	256.5
	Weweantic River	281.5
	Butlers Cove	164.5
	Marks Cove	15.0
	TOTAL:	**781.7**
Westport	West Branch	230.2
	East Branch	440.0
	East Branch (seasonal only)	554.8
	Richmond Pond	51.5
	TOTAL:	**1276.5**

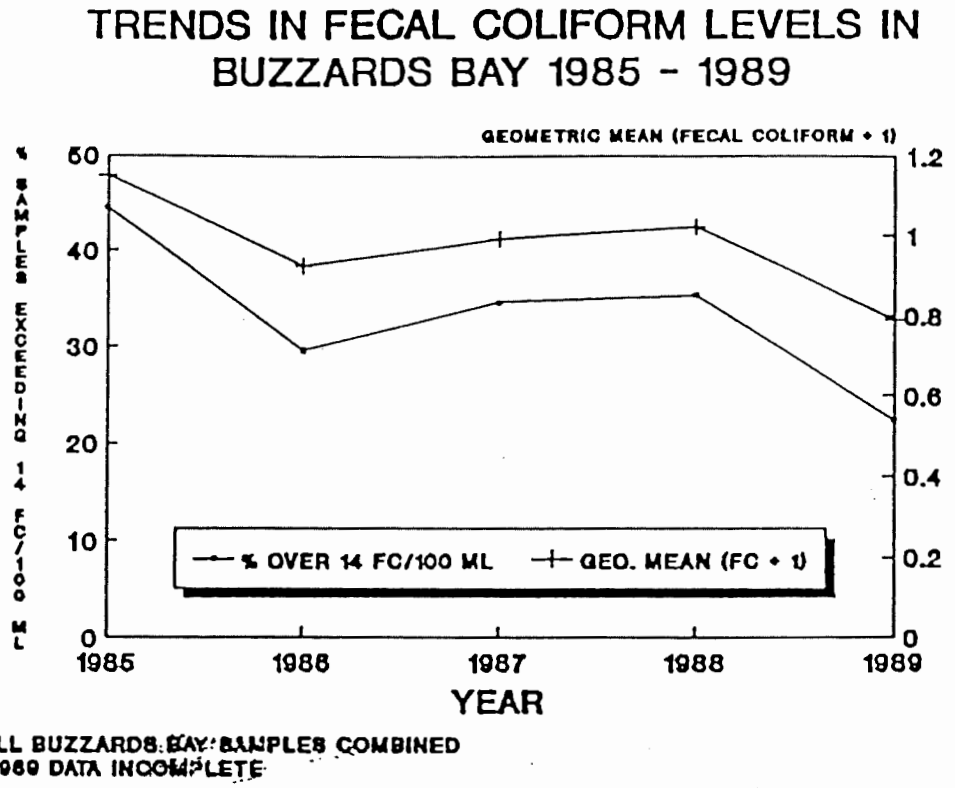

Figure 8. Trends in fecal coliform levels in Buzzards Bay from 1985-1989. All sampling areas combined. Source: Massachusetts Division of Marine Fisheries data file.

Table 8. Percentage of samples taken by the Massachusetts Division of Marine Fisheries a sites in Buzzards Bay, 1985-1989. Note that 1989 data are incomplete

	Percentage of Total Samples Taken				
Area	1985	1986	1987	1988	1989
Westport	3.4	3.6	4.2	6.5	8.5
Dartmouth	2.6	8.3	14.9	6.4	13.1
New Bedford	0.7	1.3	1.9	1.4	9.8
Fairhaven	2.9	4.6	7.7	5.5	16.0
Mattapoisett	13.7	23.3	12.8	15.0	8.4
Marion	7.9	15.8	16.7	11.6	9.4
Wareham	38.5	21.7	18.7	16.6	11.6
Bourne	21.5	7.2	8.2	16.2	8.7
Falmouth	8.7	14.3	15.0	20.8	14.7
Total Number of Samples	1510	2193	1603	3556	1997

Table 9. Fecal coliform densities from selected freshwater streams in the Buzzards Bay Watershed. Data are part of the sanitary surveys conducted by the Massachusetts Division of Marine Fisheries under the Shellfish Sanitation Program, 1989.

LOCATION	SAMPLING DATE	FECAL COLIFORM/ 100 ML
Dartmouth - Slocum River		
Unnamed Creek Station 3A	09-Aug-89	250
Unnamed Creek Station 3B	09-Aug-89	290
Giles Creek	09-Aug-89	190
Shattuck Creek	09-Aug-89	80
Shattuck Creek	15-Aug-89	70
Shattuck Creek	28-Aug-89	10
Unnamed Creek Station D2	09-Aug-89	190
Unnamed Creek Station D2	15-Aug-89	230
Unnamed Creek Station D2	28-Aug-89	20
Unnamed Creek Station D3	09-Aug-89	230
Unnamed Creek Station D3	15-Aug-89	60
Unnamed Creek Station D3	28-Aug-89	10
Creek Pelegs Isle	09-Aug-89	300
Creek Pelegs Isle	15-Aug-89	60
Creek Pelegs Isle	28-Aug-89	10
Unnamed Creek Station D5	09-Aug-89	2500
Unnamed Creek Station D5	15-Aug-89	600
Unnamed Creek StatioN D5	28-Aug-89	220
Unnamed Creek Station D6	09-Aug-89	550
Unnamed Creek Station D6	15-Aug-89	500
Unnamed Creek Station D6	28-Aug-89	90
Giles Creek Ditch	28-Aug-89	10
Dartmouth - Smith Neck		
Meadow Shore Creek	08-Aug-89	20,000
Meadow Shore Creek	15-Aug-89	890
Meadow Shore Creek	04-Oct-89	10
Meadow Shore Creek	26-Oct-89	50
Salter's Pond Creek	08-Aug-89	560
Dartmouth - Apponagansett Bay		
Duck Pond Drain	27-Jun-89	15
Corner Creek	08-Aug-89	150
Unnamed Creek Station 2	29-Jun-89	10
Unnamed Creek Station 24	12-Jul-89	10
Star Of The Sea Creek	08-Aug-89	1550
Bush Point Creek	12-Jul-89	<10
Bush Point Creet	08-Aug-89	>16000
Stanton Pier Creek	12-Jul-89	60
Stanton Pier Creek	08-Aug-89	3400
Unnamed Creek 67	04-Oct-89	10
Unnamed Creek Station 68	04-Oct-89	20
Creek Between N. & S. Pier	12-Jul-89	200
Fairhaven - Outer Harbor		
Egypt Lane West CreeK	26-Jun-89	60
Egypt Lane East Creek	26-Jun-89	260
Mouth Of Creek	11-Oct-89	5000
Fourth West Creek	11-Oct-89	7000
Mouth Of Creek	06-Nov-89	90
Bend - First West Creek	11-Oct-89	5500

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Table 9 continued from previous page...

Ditch Behind Shack (1st Bend)	06-Nov-89	60
Third West Creek	11-Oct-89	470
Ditch Behind White House	06-Nov-89	10
Ditch Opposite 2-story House	06-Nov-89	170
Ditch Opposite 17 Egypt Lane	06-Nov-89	30
Fairhaven - West Island South		
Unnamed Creek 3S2	12-Jun-89	1900
Unnamed Creek 3S2	26-Jun-89	2000
Mouth Of Nakata Creek	16-Aug-89	1300
Mouth Of Nakata Creek	18-Oct-89	110
North Branch Of Creek	26-Jun-89	900
South Branch Of Creek	26-Jun-89	1100
25' Nakata St. Ditch	26-Jun-89	400
Creek Behind #1 Nakata Ave	17-Jul-89	14,000
North Point Creek	26-Oct-89	10
Fairhaven - West Island North		
Jack Cove Creek	18-Oct-89	30
Ditch #2	05-Jul-89	30
North Point Creek	13-Jul-89	20
North Point Creek	31-Jul-89	10
Corner Creek	26-Oct-89	10
David Ward Creek	05-Jul-89	30
Mouth Of Creek At I.A. Beach	12-Jun-89	230
Mouth Of Creek At I.A. Beach	29-Jun-89	730
Mouth Of Creek At I.A. Beach	05-Jul-89	1200
Mouth Of Creek At I.A. Beach	06-Jul-89	6700
Mouth Of Creek At I.A. Beach	11-Jul-89	350
Creek By Fence	29-Jun-89	330
First Ditch West Of Corner	26-Oct-89	10
First Street Creek	13-Jul-89	10
First Ditch East Of Fir	26-Oct-89	10
2nd Fir Street Creek	13-Jul-89	20
First Ditch East Of Dogwood	26-Oct-89	10
1st Ebony Creek	13-Jul-89	10
2nd Ebony Creek	13-Jul-89	130
Fairhaven - West Island East		
Bass Creek Mouth	05-Oct-89	10
Bass Creek	26-Oct-89	10
Unnamed Creek F1	05-Oct-89	220
Unnamed Creek F2	05-Oct-89	20
Fairhaven - West Island East		
Unnamed Creek F3	05-Oct-89	20
Unnamed Creek F4	05-Oct-89	10
Unnamed Creek F5	05-Oct-89	10
Fairhaven - Nasketucket Bay		
Shaw Cove Creek	26-Jun-89	20
Shaw Cove Creek	24-Oct-89	40
Town Marker Creek	26-Jun-89	80
Town Marker Creek	24-Oct-89	40
East End Creek	26-Jun-89	120
East End Creek	24-Oct-89	40
Mouth, Howard Beach Creek	06-Nov-89	90
Black Duck Creek	06-Nov-89	30

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Table 9 continued from previous page ...

Fairhaven - Little Bay		
Unnamed Creek 21	24-Oct-89	20
Drainage Ditch On Marsh-Weeden Rd.	29-Aug-89	10
Unnamed Creek 22	24-Oct-89	20
Ditch On Marsh Front Of Weeden Rd.	9-Aug-89	50
Unnamed Creek 23	24-Oct-89	40
Unnamed Creek 24	24-Oct-89	50
Unnamed Creek 25	24-Oct-89	60
Unnamed Creek 26	24-Oct-89	40
Unnamed Creek 27	24-Oct-89	100
Creek Behind Aqua And Brown House	24-Oct-89	100
Creek Opposite Station 28	24-Oct-89	100
Ditch North Of Abandoned House	24-Oct-89	300
Drainage Ditch 33	24-Oct-89	100
Drainage Ditch 34	24-Oct-89	100
Drainage Ditch 35	24-Oct-89	100
Drainage Ditch 36	24-Oct-89	40
Knowlmere Creek	21-Aug-89	1400
Knowlmere Creek	24-Oct-89	10
Small Creek. Knowlmere	19-Sep-89	800
Small Knowlmere Creek	24-Oct-89	100
Nonquitt Avenue Creek	06-Nov-89	80
Summit Drive Creek	06-Nov-89	20
Stp Creek	21-Aug-89	700
Creek At Bridge Street	13-Jun-89	1700
Creek Under Slocum Street	13-Jun-89	400
Raymond Street Creek	21-Aug-89	1000
Raymond Street Creek	24-Oct-89	100
Raymond Street Stream	05-Nov-89	10
Knowlmere Beach Creek	21-Aug-89	4800
Wareham-Bourne Cove/Little Harbor		
Head Of Marsh Stream	02-Oct-89	50
Head Of Marsh Stream	28-Nov-89	10,000
Mouth Of Stream 11	02-Oct-89	20
Wareham -Bourne Cove/Little Harbor		
Mouth Of Stream 11	28-Nov-89	3100
Drainage Ditch 12	02-Oct-89	80
Mouth Of Creek	28-Nov-89	5000
Mouth Of Ditch	02-Oct-89	10
Creek At Mouth	02-Oct-89	50
Creek Upland	02-Oct-89	340
Stream Station 21	28-Nov-89	60
Mouth Of Creek	02-Oct-89	10
Mouth Of Ditch On Map	28-Nov-89	230
Creek South Of Finn Residence	02-Oct-89	10
Creek South Of Golf Course	28-Nov-89	90
Creek North Of Mouth	02-Oct-89	20
Creek North Of Golf Course	28-Nov-89	10
Bourne - Cape Cod Canal		
Bournedale Herring River	12-Jun-89	10

Coliform Pollution

Water quality surveys conducted by the Massachusetts Department of Environmental Protection, Division of Water Pollution Control (DWPC) provide additional sources of information regarding bacteriological quality in Buzzards Bay. In 1985, water quality data were collected by DWPC in the subdrainage basins of both the western and eastern shores. Station locations in these studies were selected on the basis of the following criteria:

- Historical data, i.e., stations sampled during previous DWPC surveys or by the United States Geological Survey;
- Station location (upstream or downstream of a known pollution source);
- Location at confluence with a major tributary or within the main channel; and
- Location within the transitional zone between fresh and saline waters when salinity measurements range between 0 and 15 parts per thousand. These stations were most easily accessed from the land and were included under the generic term of intertidal stations.

The fecal coliform densities measured during the DWPC survey are summarized and presented in Table 10.

In addition to the aforementioned work, the U.S. Food and Drug Administration, through its Technical Services Unit, periodically evaluates some shellfish areas in conjunction with the DMF monitoring program. The purpose of these studies is to provide training for individuals involved in the state monitoring program as well as to evaluate the specific sites. In 1985, FDA conducted a sanitary survey in Buttermilk Bay and the Westport River. In 1987, this agency performed a hydrographic study near the Massachusetts Maritime Academy Wastewater Treatment Facility (Carr, 1987) to evaluate effluent effects on an area proposed for a shellfish grant. In general, these surveys confirmed the findings of other studies and classification, relative to the areas status for shellfish harvesting, already given an area. In the coming year, FDA will be conducting a similar survey in the Clarks Cove area.

Disease Outbreaks

As mentioned previously, in recent years the etiology of shellfish - related diseases has shifted from bacterial to viral origin. In general, the most common symptom of shellfish-related illness is a mild gastroenteritis, which generally goes unreported. Without a consistent means of reporting and relating this symptom to shellfish consumption, it is not possible to determine disease outbreaks. An aggressive reporting program in New York revealed that shellfish-related gastroenteritis was sometimes quite common. Because Massachusetts lacks a reporting program, data on shellfish-borne diseases in the Buzzards Bay area are not available.

Present Management Strategies

Domestic Sanitary Wastes

At present, the primary source of human pathogens into Buzzards Bay — domestic human sewage — is treated in one of two ways; by a Publicly Owned Treatment Works or on-site by a septic system. Treatment plants on the bay generally discharge primary- or secondary-treated chlorinated wastes. In some

Table 10. Summary of geometric mean fecal coliform values for DWPC 1985-86 Buzzards Bay survey. N=number of samples comprising the geometric mean. Data from Gil (1987 and 1988).

Watershed and Number of Stations	Geometric Mean (FC/100 ml)
Buttermilk Bay Drainage Basin	
1 Freshwater (N=5)	31
3 Intertidal (N=5)	26, 10.3, 5.1
Onset Bay Drainage Basin	
1 Freshwater (N=4)	89
2 Intertidal (N=3-4)	12.3, 7.2
Agawam River Drainage Basin	
2 Freshwater (N=5)	36, 72
1 Intertidal (N=5)	93
Wankinco River Drainage Basin	
2 Freshwater (N=4)	224, 152
1 Intertidal (N=5)	106
Weweantic River Drainage Basin	
2 Freshwater (N=5)	92, 92
1 Intertidal (N=5)	56
Sippican River Drainage Basin	
2 Freshwater (N=5)	44.6, 103
Wareham River Drainage Basin	
Intertidal (N=5)	231.7
Mattapoisett River Basin	
3 Freshwater (N=4)	263, 107.7, 840
Mattapoisett Harbor Drainage Basin	
2 Freshwater (N=4)	345, 199
1 Intertidal (N=4)	18.6
Acushnet River Drainage Basin	
6 Freshwater (N=4)	81, 116, 662, 45,585
2 Intertidal (N=4)	1,247, 83
Buttonwood Brook/Apponagansett Rivers	
3 Freshwater (N=3)	207, 144, 42
2 Intertidal (N=3)	3.5, 2.5, 640, 7,085
Paskamanset/Slocums Rivers	
7 Freshwater (N= 2-3)	4, 62, 10, 33, 81, 75, 57
4 Intertidal (N=2)	64, 12, 4, 4
Westport River Drainage Basin	
3 Freshwater (N=4)	127, 141, 671
8 Intertidal (N=2)	
Phinney's Harbor Drainage Basin	
1 Freshwater (N=4)	105
1 Intertidal (N=4)	97
Pocasset River Drainage Basin	
1 Freshwater (N=4)	145
2 Intertidal (N=4)	130
Pocasset Harbor Drainage Basin	
2 Intertidal (N=4)	118, 7.1
Redbrook Harbor Basin	
2 Freshwater (N=4)	61, 5
Megansett Harbor Drainage Basin	
3 Freshwater (N=4)	51.8, 333, 96
3 Intertidal (N=4)	328, 6.1, 12.6
Wild Harbor Drainage Basin	
1 Freshwater (N=4)	290
1 Intertidal (N=4)	87.8
Herring Brook Drainage Basin	
1 Intertidal (N=4)	24.5
West Falmouth Harbor Drainage Basin	
3 Intertidal (N=4)	10.8, 12.3, 207
Great Sippewissett Creek Drainage Basin	
1 Intertidal (N=4)	12.6
Little Sippewisset Creek Drainage Basin	
1 Intertidal (N=4)	199

instances, such as discharge from CSOs, primary, untreated wastes flow into the bay. On-site sewage disposal, described elsewhere in this report, discharges sewage to the ground, and wastes may eventually enter the bay via the groundwater. In recent years, the increasing population in shoreline areas has stressed both methods of sewage disposal. In response to the increasing demand, three POTWs are being expanded. Unfortunately, fiscal restraints are limiting the ability of municipalities to address sewage treatment in a comprehensive manner. Management strategies for on-site systems generally comprise adequate enforcement of Title 5 and, in some communities, further regulation of the placement of on-site systems through the adoption of supplements to these regulations.

Stormwater

The relatively recent recognition of the role that stormwater plays in shellfish area closures has prompted some communities to begin attempts at stormwater mitigation. Recent efforts sponsored by the Buzzards Bay Project in Buttermilk Bay and by the town of Bourne are the beginnings of an effective management strategy for stormwater. Citizen action groups have recently made some significant strides toward getting the issue of stormwater addressed in a comprehensive manner in the towns of Bourne, Marion and Westport.

Marine Sanitary Wastes

Recently, a report to the Executive Office of Environmental Affairs outlined the history of marine sanitary requirements at marinas. In the absence of a requirement for pumpout facilities, very few were constructed. Buzzards Bay has no management strategy for handling of sanitary wastes and relies solely on the present regulations prohibiting discharge. Much of the responsibility for handling marine sanitary wastes rests with the boat owner. The town of Bourne operates a pumpout facility and shoreline toilet facilities in an effort to prevent nearshore discharge of wastes.

Disinfection of Seafood

Although the harvest of shellfish for direct consumption is not allowed in contaminated areas, the shellfish resource in these areas is not completely unused. Under a DMF program, contaminated shellfish are transferred into clean (acceptable) areas for depuration. This program of "relays" has benefitted a number of towns in southeastern Massachusetts and may be expanded in the future as areas in Buzzards Bay are properly classified.

Conclusions and Recommendations

Pathogens in Buzzards Bay—General Conclusions

Much of our current knowledge about the bacteriological quality of the waters in Buzzards Bay is derived from data showing shellfish harvesting area closures. These data, collected by DMF, indicate that water quality in Buzzards Bay has declined dramatically in recent years. Although this may indeed be the case, the question of whether human pathogen densities within the bay are actually increasing is not being adequately addressed.

Chapter 2

Nutrients and the Trophic Status of Buzzards Bay

by John Kelley[1], Ivan Valiela[2], and Douglas Hersh[2]

Introduction

That nutrient levels generally influence algal biomass and productivity of a water body is axiomatic. Activities of humans have enhanced nutrient inputs to coastal ecosystems worldwide (Kempe, 1988). Nutrient loading often leads to excessive algal growth, followed by depletion of dissolved oxygen in seawater as a result of rapid decomposition of the algal biomass. Anoxia is perhaps the most dramatic end point of nutrient loading, but it is not the only concern. Elevated nutrients can, for example, lead to changes in water clarity and can alter, directly or indirectly, the abundance, distribution, and mix of organisms (e.g., reduce seagrasses, change the plankton and benthos communities and food webs leading to fish and shellfish).

Nutrient loading is to some extent complicated by flushing by water inflow. Faster water renewal lowers the impact of any given nutrient loading rate: systems that flush faster can, in principle, tolerate higher nutrient inputs. Water motion in coastal bays is complex—influenced by mixing of fresh water and seawaters, strong tidal and wind forcing and hydrodynamical coupling of open deeper waters with associated sub-estuaries, embayments, marshes, and tidal flats. Moreover, marine eutrophication is fundamentally different from freshwater eutrophication (Valiela, 1984), in that nitrogen rather than phosphorus seems to be the more limiting nutrient. For such reasons, our present ability to predict marine and estuarine responses to increasing nutrient loads is less advanced than it is for fresh waters.

Some stations and fringing areas of Buzzards Bay have been studied quite extensively (e.g., Great Sippewissett salt marsh, New Bedford Harbor, Buttermilk Bay, sporadically visited deep water sediment stations). However, spatial and temporal distribution of nutrients through the open waters and sediments of the Bay is poorly known. This basic information is part of that required to assess the consequences of anthropogenic nutrient inputs.

Construction of a nutrient budget is an effective way to assess the condition of a bay. Suitable data are lacking, however, for Buzzards Bay as a whole. Our focus in this report is on a few studied, if still uncertain, aspects of the inputs, outputs, and concentrations of the nutrients nitrogen (N) and phosphorus (P), along with algal biomass and growth. Our synthesis and calculations are gathered from pieces of information dispersed throughout a variety of literature on the Bay. This report is intended to serve as a menu for what needs to be

[1]*Battelle Ocean Sciences, 397 Washington Street, Duxbury, MA 02332*

[2]*Boston University Marine Program, Marine Biological Laboratory, Woods Hole, MA 02543*

learned. If our compilation stirs controversy that stimulates research on the uncertainties, all the better.

This report is organized in the following sequence: (1) a discussion of estimated sources and rates of nutrients going into the bay; (2) an examination of recent data sets on *in situ* nutrient and water quality parameters within the Bay, including a brief historical comparison to some earlier data; (3) perspective on nutrients, plant productivity, and trophic status of Buzzards Bay, by comparison to other coastal bays and estuaries; (4) an exercise to outline whether Buzzards Bay can be defined with respect to its long-term retention of nutrients, an issue relevant to long-term management; and (5) some recommendations for further scientific study.

Nutrients Delivered to Buzzards Bay

There are many paths by which nutrients can enter the Bay (Figure 1). Surface sources from land include sewered and municipal discharges, storm runoff from urban and other areas, stream discharges from the watershed (with or without anthropogenic contributions), and more diffuse runoff from land bordering the edges of the Bay (including wetlands). Groundwater flux can deliver nutrients from septic systems. Rain also falls directly on the Bay, bringing nutrients (mainly N) of both anthropogenic and of terrestrial origin.

If one had accurately quantified all these freshwater hydrological vectors for nutrient input, there would still be missing sources, because in addition to the land-bay and bay-air interfaces, there are water-mass boundaries definable by basin and geography. Water transport across three such boundaries delineating Buzzards Bay can carry nutrients into, as well as out of, the Bay: offshore exchange through the open mouth with Rhode Island Sound to the southwest, exchange

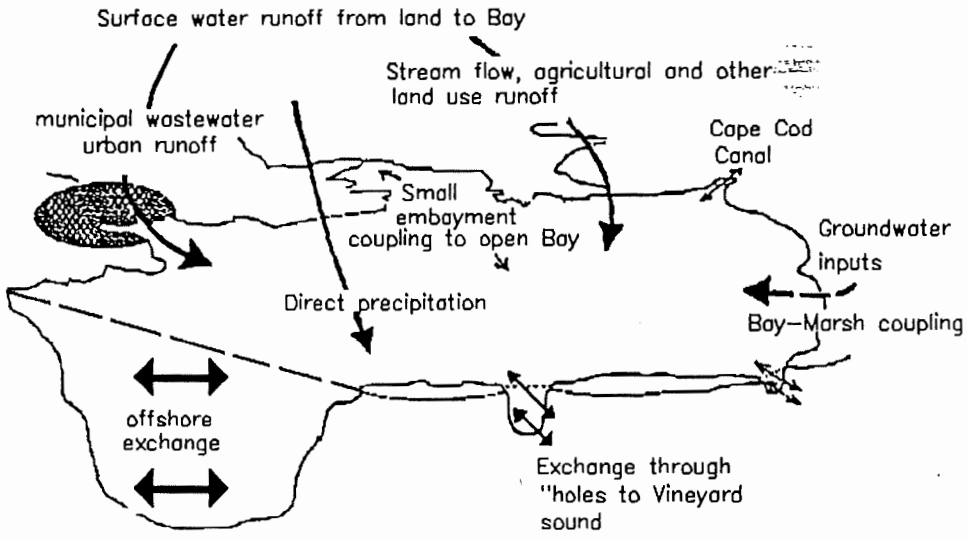

Figure 1. Paths by which nutrients can enter the Bay.

between the island "holes" that connect the Bay to Vineyard Sound, and exchange with the Cape Cod Canal (Figure 1).

There is one published estimate of land nutrient discharge for the whole Bay (NOAA, 1988). The estimate includes "known" (circa 1982) point sources going directly to "surface water in a coastal county," and "nonpoint sources" calculated based on gross land-use categories. Not explicitly quantified, however, are nutrients carried from "upstream sources," groundwater, exchanges with wetlands or "barren lands," offshore exchange, and direct precipitation to the Bay itself. Thus, it is likely that inputs are underestimated, as is probably the case in most coastal waters.

An estimate of precipitation is provided here, but ocean-side and canal nutrient exchanges are unknown. Although total inputs have not been established, sources and their potential importance deserve some individual discussion.

Sources Estimated

Land-Derived Surface Water Sources

The National Oceanic and Atmospheric Administration (NOAA, 1988) gives estimates of total[1] nitrogen and phosphorus discharges for the "estuarine drainage area" of Buzzards Bay and other coastal water bodies in the northeast United States. Nutrient loading[2] from NOAA (1988), using 228 sq. mi. as the area of Buzzards Bay (see Table 1), is 53 mmol N $m^{-2} y^{-1}$ and 10 mmol P $m^{-2} y^{-1}$, the majority of the load contributed by point sources.

1 Nitrogen and phosphorus occur in many forms: a primary distinction is operationally defined: dissolved (passes through a nominal 0.45 μ filter) vs. particulate (retained by the filter). Dissolved inorganic nitrogen (DIN) is the major plant nutrient and includes three forms: ammonium (NH_4), nitrate (NO_3), and nitrite (NO_2). Dissolved organic nitrogen (DON) is nitrogen in certain organic compounds such as urea, amino acids, etc., and may be taken up or excreted by plants or animals. Dissolved inorganic phosphorus, another primary plant nutrient, is mostly in seawater in the form of orthophosphates, abbreviated phosphate (PO_4). Dissolved organic phosphorus (DOP), is phosphorus in certain organic compounds and may be taken up or excreted by plants and animals. Particulate nutrients mostly are in organic compounds (i.e., particulate organic nitrogen (PON) or phosphorus (POP), but may be inorganic—adsorbed to the outside of biological cells or inert material such as silts or clay particles. If both organic and inorganic forms are measured by a technique, it may be termed particulate N or P (i.e., PN or PP). Dissolved organic or particulate nutrients may not all be available for biological use. Usable portions must be decomposed first (i.e., "recycled," "remineralized" or "regenerated") to the inorganic, or very simple organic, forms, and thus are not immediately available as plant nutrients without modification (which, however, can be rapid). Total N or P usually refers to all forms found in a bulk analysis of unfiltered seawater or to the amount found by adding up all various forms analyzed separately. If analyzed by a specific chemical technique which may or may not recover all inorganic, organic, dissolved or particulate forms, the term may be qualified by the technique—such as with N analyzed by a Kjeldahl chemical digestion, termed Total Kjeldahl Nitrogen (TKN). The NOAA (1988) estimates do not specify technique, but include inorganic and organic forms, presumably both particulate and dissolved.

2 Loading is the amount of nutrient added into a receiving body per unit time and generally expressed per unit area or volume. Standard reporting units for estuarine and seawater samples, following oceanographic convention, are metric and for chemical units, the atomic measure of moles (abbrev. mol) is used. Freshwater convention has been mass (grams, abbrev. g). For conversion, there are ~14 grams of N per mole and 31 grams of P per mole. Annual loading is expressed in mol $m^{-2} y^{-1}$. The prefix m (as mmol or mg [used in DEP data set later]) is for milli (one part in a thousand; 10^{-3}) moles or milligrams. A prefix μ (as mols) is for micro (one part in a million; 10^{-6}) moles. When concentrations in seawater are discussed, these may be in micromoles per liter, abbreviated as μmol l^{-1} or more commonly, μM (both abbreviations mean micro moles in a liter of solution).

Table 1: Physical statistics reported for Buzzards Bay[a]

Parameter	Value	Reference[b]
Estuarine drainage area (EDA)	1492 km^2 (576 sq. mi.)	1
Watershed area	997 km^2 (385 sq. mi.)	4
Watershed area	780 km^2 (301 sq. mi.)	5
Drainage basin area	1120 km^2 (432 sq. mi.)	7
Bay surface area	590 km^2 (228 sq. mi)	1
	608 km^2 (235 sq. mi.)	4
	550 km^2 (212 sq. mi.)	5
	590 km^2 (228 sq. mi.)	7
Drainage area/surface area	2.53	1
	1.64	4
	1.42	5
	1.90	7
Average depth	10 m	1
	13-16m	3
	~15 m "in central portion"	4
	11 m	5,7
Max depth	generally 20 m, excepting localized depressions and a trough at the mouth up to ~ 43 m	2,4
Volume	6.02×10^9 m^3	1
	6.1×10^9 m^3	5
Freshwater input	mean 34 m^3 sec^{-1}	1
	mean 27 m^3 sec^{-1}	6
	mean 15 (3.9) m^3 sec^{-1}	5
—Acushnet River, New Bedford harbor	mean 0.2 m^3 sec^{-1}	6
Tides	mean range ~1.128 m	from 6
	spring tide ~1.40 m	from 6

aThe table shows that even for something as simple as the dimensions of the bay, values derived by different studies or quoted by different authors can differ, sometimes substantially. In calculating nutrient budgets, this can lead to errors when rates or pool sizes are compared from study to study. Calculations in text specify the dimensions used, some calculations use a range for illustration. For some parameters such as surface area, the value depends on how the bays boundaries are defined. For other parameters, new information has resulted in improved estimates. For example, the drainage basin reported by EPA and EOEA (1990) were based on recent groundwater recharge area delineations from USGS and represents the most accurate estimate for this parameter. See also footnote 3.

[b]1: NOAA (1988); 2: Young (1971); 3: Rhoads (1973); 4: Hough (1940); 5: Signell (1987), 6: Summerhayes et al. (1985); 7: from the Buzzards Bay Comprehensive Conservation and Management Plan (EPA and EOEA, 1991)

The NOAA estimate of inputs delivered in surface waters off the land "circa 1982" was to provide a rough guide[3], but more recent efforts supported by the Buzzards Bay Project (EPA & EOEA, 1991 and SAIC 1991) suggest the sources of surface discharge, groundwater, and major point-source effluents to be 1935 metric tons per year or 235 mmol N m^{-2} y^{-1} (Buzzards Bay Project, 1991 and SAIC, 1991). A major-point source discharge is effluent from the New Bedford

3 For example, NOAA data yield estimates, for nearby Naragansett Bay and Long Island Sound, of nitrogen loading that are 74% N and 245% of estimated DIN loading, respectively, made by Nixon (1981; also Nixon and Pilson, 1983). Some of the difference in loading estimates can arise from disagreement among published reports of the physical statistics of a given estuary (see Pilson, 1985; Nixon, 1987); some may arise because estimates were made in different years. Precision in source estimates is useless if different dimensions are used in calculating aspects of a nutrient budget, and one must be alert to this problem (see Table 1).

wastewater treatment facility, on the order of 960 metric tons y^{-1} (SAIC, 1991), which alone amounts to 121 mmol N m^{-2} y^{-1} to the Bay as a whole. Although the NOAA values must be an underestimate, and the actual value may exceed 235, the range 53 to 257 mmol N m^{-2} y^{-1} sets bounds for calculations and comparisons discussed later in the report.

New Bedford Harbor represents only a small fraction of the total area of the Bay (~3.43 x 10^7 m^2; Camp Dresser and McKee, 1990), but loading to this one area is high. We calculated a value of 2000 mmols N m^{-2} y^{-1}, using 960 metric tons N y^{-1} (above); detailed harbor budgeting (in Camp, Dresser, and McKee 1990) calculated that effluent Dissolved inorganic nitrogen (DIN) and remineralizable organics, groundwater, and precipitation provide somewhat less than this, about 1300 mmol N m^{-2} y^{-1}.

Direct precipitation

The National Acid Deposition Program (NADP, 1988) provides data on the NH_4 and NO_3 content of wet precipitation; dry deposition and dissolved organic nitrogen (DON) may add more nitrogen. Data of NADP for PO_4 is not summarized because values often are below limits of detection. For the calendar year 1987, precipitation-weighted mean nitrogen concentrations, and total precipitation (cm) are reported for only two NADP sites in Massachusetts (Quabbin and Middlesex). Annual mean values for NH_4 were 6.1 and 7.8 mM, for NO_3 were 17.8 and 19.4 mM; thus, DIN (NH_4 + NO_3) concentrations were 23.9 and 27.2 mM respectively. Using total precipitation of 102.8 and 98.2 cm at these sites, the annual wet deposition amounts to 24.6 to 26.7 mmol DIN m^{-2} to the Bay. Concentrations during 1987 seem typical of the period of record (1982-1987) for these sites and another Massachusetts coastal site (N. Truro—without a 1987 summary [NADP, 1988]) and the estimate is similar to that made for Buttermilk Bay at the northeastern end of Buzzards Bay—derived from data for an earlier period (Valiela et al., 1978).

Sum of Two Estimated Sources

The range for nitrogen loading on a Baywide basis is calculated as >79 (~26 + 53) to ~283 (~26 + 257) mmol N m^{-2} y^{-1}. Precipitation thus delivers about 10 to 33% of nitrogen inputs ultimately derived from the land, which is similar to many coastal areas.

"Acid rain" (with high nitrate levels) has stirred controversy (especially in Chesapeake Bay), because the nitrogen content of rain has shown a trend of increase and now contributes a substantial fraction of the input to some coastal waters. In Buzzards Bay, nitrogen input from precipitation might have some impact on the oligotrophic waters of the open Bay because it is delivered directly to them; on the other hand, it constitutes a minor fraction of inputs to a area like New Bedford Harbor, which receives large point-source discharges.

Potential sources

Groundwater

Groundwater nutrient inputs characteristically have not been estimated for larger estuaries and bays, and this is true for the whole of Buzzards Bay. Inputs are often presumed to be small relative to other sources, but the importance of groundwater nitrogen discharge may depend in part on the spatial scale of the system. Where

individual septic systems predominate rather than sewered wastewater, groundwater flux can be significant. For Buttermilk Bay, about 85% of the estimated DIN input was from groundwater (Valiela and Costa, 1988); septic systems (along with use of fertilizers in the watershed) are a mechanism for input to groundwater. While it is unclear how groundwater inputs may compare in magnitude to surface water inputs when considering all of Buzzards Bay, groundwater is a significant nitrogen source at local scales, including the case of salt marshes bordering the eastern edge of the Bay (Valiela and Teal, 1979; Valiela, 1983).

Offshore Exchange

The potential input of nutrients to Buzzards Bay via offshore exchanges (Figure 1) is most difficult to assess, and, like most coastal ecosystems, is little studied. Garside et al. (1978) suggested significant import of NO_3 from ocean exchange was possible in the Sheepscot estuary (Maine) from up-estuary bottom layer flow of denser, saltier, nutrient-rich Gulf of Maine water in a classic two-layered estuarine circulation. Summerhayes et al. (1985) have discussed the net movement of both mud and detritus into the New Bedford Harbor area by similar mechanism, and at least portions of Buzzards Bay may be "inverse" (Meade, 1965)—to an extent they become filled by sediment coming from the seaward margin, as well as from land erosion of the continent. Given a net tidal drift into the western end (Signell, 1987), it may be that the Bay functions as a nutrient importer from offshore systems. For example, phytoplankton can deplete surface water nutrient concentrations and the newly formed particulate nutrients sink to bottom waters. If surface Bay waters are advected offshore while bottom waters are brought in, the Bay could act not only to retain some of its own recyclable nutrients into its own bottom waters, but also to bring in nutrients sedimented by phytoplankton blooms in surface waters offshore.[4]

Given especially the openness of Buzzards Bay to the southwest, significant water exchange between the shallow shelf and open waters of the Bay might occasionally occur by both wind-driven events and semidiurnal tides, but net effects on nutrient budgets are unknown. Winds prevailing from the southwest (much of the year [Signell, 1987]) would tend to drive in low-nutrient surface waters, forcing out bottom waters sometimes richer in nutrients. Yet Signell also noted that some winds may drive surface waters out of the Bay, forcing a bottom water input in order to conserve mass. He suggested (using a reasonable scenario for direction, duration, and wind strength) that this mechanism could induce a 15% Bay volume exchange by offshore waters in about 3 days (with sustained wind conditions). In contrast to temporal variations in wind patterns and the strength or direction of wind-driven currents, regular bottom—water exchange by the tides may continually import more nutrient-rich waters.

4 Nixon and Pilson (1984) estimated from a stoichiometric model that a significant input of nutrients to nearby Narragansett Bay may come from offshore water exchange. Amounts of nitrogen and phosphorus roughly equivalent to land-derived nutrient sources may enter Narragansett Bay via this route even though there is a *net* nutrient loss to offshore because there is a net seaward advection of water (if bay salinity is at steady state, then the advected amount equals the runoff + precipitation - evaporation) into the bay. If the magnitude of nutrient influx by offshore exchange suggested for Narragansett Bay occurred in Buzzards Bay, it could take on greater significance, because nutrient discharges from land to Buzzards Bay are lower. Data necessary to use Nixon and Pilson's model in the case of Buzzards Bay are not available.

Exchanges At Other Boundaries: Sources or Sinks, and Potential Significance?

Strong tidal currents move water in and out of the canal at the head of the Bay, through the "holes," and interact with marshes (Figure 1). As in the case of exchange at the open western end of the Bay, effects on nutrient import and export are unknown.

Tidal exchange of nitrogen is significant in the case of regular flooding of the bay's salt marshes (Valiela and Teal 1979). For Great Sippewissett marsh, the tides carry exported particulate N and NH_4^+ back to the Bay. Valiela (1983) estimated that all marshes surrounding Buzzards Bay (~9.2×10^6 m^2), if similar to Great Sippewissett, might export about 60,700 kg N y^{-1} resulting in a baywide (564 km^2, Table 1) input of 7.7 mmols m^{-2} y^{-1}. This value is less than direct precipitation and only 2.7 to 9.7% of the source strength range given above; as with precipitation, marsh input may occur to some relatively unenriched waters and have impact larger than suggested by amount alone (Valiela, 1983).

Daily tidal influx may be about 1/10 of the volume of the whole Bay (Table 1), which potentially could provide nutrients to be utilized within the Bay. Movement of resuspended bottom particles by tidal advection into the bay (Rhoads 1973; Rhoads et al., 1975) from offshore may bring a high proportion of particulate (as well as dissolved) nitrogen into the Bay. The "dietary" value of this nitrogen detritus brought in along the bottom would be less than the nitrogen in organic matter of phytoplankton. Moreover, it would tend to feed an animal foodweb directly, in contrast to imported dissolved nutrients that would first stimulate the metabolism of the phytoplankton community. Thus, retention of such detrital inputs would be significant relative to food web structure and algal biomass. Not only the magnitude of the nutrient source, but also its form can have relevance to determining the eutrophication potential of the bay.

Physical Controls Associated with Nutrient Inputs: Tidal vs. Freshwater Flushing

Emphasis on the tidal forces in Buzzards Bay has been longstanding, and the importance to ecology recognized in more than passing reference. Moore (1963) noted that "as the bay is protected from large, long period, open ocean waves, 'this system' [i.e., tidal currents as sorting energy for sediments] is the one main mechanical distributor of detritus." Moore further cites an estimate (Sumner et al., 1913): "It would require about 10 days to completely renew the bay water mass, a feature of interesting biological implication." Some simple calculations next suggest that flushing caused by tidal forces may not be quite this fast, but still has potential be faster than freshwater inputs.

Assuming an average tidal height of about 1.128 m per tidal cycle and an average depth of about 10 m for all of Buzzards Bay (see Table 1), a first-order dilution model that mixes the tidal flood and standing Bay volume, with 11.28% of the standing Bay volume replaced by a tidal flood-ebb cycle, would suggest a half-life for replacement of the volume in 3.07 days. Based on this tidal prism approximation method, about 97% of the volume could be tidally exchanged in about 15.4 days—roughly half of the lunar cycle of spring and neap tides. The

assumption of complete mixing of inflowing water with water already within the Bay surely makes this an underestimate, because some water must just slosh back and forth without mixing. However, studies (Rhoads 1973; Rhoads et al., 1975; Roman and Tenore, 1978; Roman, 1978, 1980) suggest incoming bottom water does mix actively, at least within a bottom turbid layer of 2 to 2 m, and the body of the bay is well-mixed and "not thermally stratified" (Roman and Tenore, 1978), or typically only slightly thermally stratified during warmer months (Rhoads et al., 1975).

A hypsographic[5] curve of the Bay to derive estimates of the actual *volumes* (not just tidal *height*) involved at ebb and flood are required also to improve the calculation, which is intended only as a guide in comparison to freshwater flushing, estimated next. [The interested reader is further referred to Signell (1987) for detailed discussions of tidal advection.] A method for calculating freshwater residence time used in nearby Narragansett Bay (Pilson, 1985) is used here for Buzzards Bay. A major difference between Narragansett Bay and Buzzards Bay is the 3-fold or greater input of fresh water in the former,[6] to the freshwater flushing time calculated for Nar-ragansett Bay (average of 26 days), must be no less than three times as fast as Buzzards Bay.[7]

If one assumes an average salinity of 29 ppt and an ocean endmember of 32 ppt for Buzzards Bay,[8] using the fraction of freshwater method for flushing time (Pilson, 1985) and a 34 $m^3 s^{-1}$ input, the estimated freshwater flushing time is 188 days. As the difference between "ocean" and "bay" salinity increases, the calculated time gets longer;[9] as it decreases the time shortens. If the freshwater input estimate of Signell (1987) is more correct (15 $m^3 s^{-1}$), then the calculated flushing time must stretch to at least a year.

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5 A hypsographic curve shows the area of bay above a given depth of water. It is useful for determining volumes of different areas and for baywide weighted estimates of concentrations for budgeting purposes.

6 With a drainage area of ~1781 sq. mi. (4613 km^2) (NOAA, 1988)[or 4708 km^2 (Pilson, 1985)] in Narragansett Bay, compared to about 1492 km^2 (or less; see Table 1) for Buzzards Bay, it is not surprising that Narragansett Bay receives at least 3 times the freshwater discharge (long-term average of 105 $m^3 s^{-1}$ [Pilson, 1985] vs. Buzzards Bay: ~34 $m^3 s^{-1}$; perhaps 15 $m^3 s^{-1}$; Table 1). The two bays have a reasonably similar salinity (perhaps slightly higher for Buzzards Bay) and, thus, a similar freshwater volume percent: mean volume-weighted salinity in Narragansett Bay ranges from about 28 to 31 ppt (Pilson 1985); for Buzzards Bay a volume-weighted salinity is not available. Salinity within the Acushnet River estuary may reach as low as 23 ppt (seasonal range ~23 to 30) in New Bedford Harbor, the Harbor "approach" range is about 27.5 to 30.7 ppt (Summerhayes et al., 1985), and most of Buzzards Bay is greater than 28 ppt.

7 Narragansett Bay surface area depends on definition of the boundaries of the system (for example, NOAA [1988] gives an area of 165 sq.mi., or 427 km^2, but Pilson [1985] gives a value of 328 km^2). It is smaller than Buzzards Bay (228 sq. mi., or 590 km^2 — NOAA, 1988; but see Table 1). The two bays have a similar average depth: almost 9 m for Narragansett Bay, although Buzzards Bay is slightly deeper (10 m — NOAA, 1988; but see Table 1). Thus Buzzards Bay volume is nearly twice that of Narragansett Bay and even with similar freshwater input would take longer to be flushed.

8 Salinity values throughout the open Bay commonly are reported in the range 29 to 32 ppt (data sets discussed later in this report; also see Signell, 1987). Presumably, both Buzzards Bay and Narragansett Bay have a similar Rhode Island Sound ocean endmember, ~32 to 33 ppt (Pilson, 1985).

9 For example, assuming 28 and 33 ppt as bay and ocean salinities, the time to replace fresh water in the bay is 300 days at 34 $m^3 s^{-1}$.

There are other more sophisticated methods for calculating the tidal flushing, and the ones here are crude calculations, ignorant of the complexities of vertical and horizontal mixing and advection. Occasional wind and stormwater events may enhance flushing over short periods (Signell, 1987), but on a regular basis, freshwater input seems to replace the freshwater component in the open Bay on a time scale of months, whereas tidal forces may exchange the entire bay's water in weeks. Thus, ocean exchange by tidal mixing may be the more defining physical influence upon water quality maintained in much of Buzzards Bay and is highly relevant to the nutrient dynamics and productivity of the Bay.

Nutrients in the Bay

A principal data set gives nutrient concentrations in Buzzards Bay and its associated bays and estuaries (Gil, 1987, 1988). These data were obtained during monitoring activities carried out during 1985 and 1986 by the Massachusetts Department of Environmental Protection (DEP) (then the Department of Environmental Quality Engineering [DEQE]) (cf. Table 2). These data were a primary source for a water quality assessment (Stenner, et al. 1988).

More recent data come from a study of water quality covering stations throughout the Bay, carried out by Dr. Jefferson Turner of Southeastern Massachusetts University (SMU) and colleagues (Turner et al., 1989). For additional recent data related to nutrients around the New Bedford area, we consulted Camp Dresser and McKee (1990).

DEP Monitoring Data

Here we examine whether the DEP data set can be useful to evaluate nutrient conditions in Buzzards Bay. To summarize, the utility for our purpose of baywide assessment was limited, due to (1) analytical problems, especially with regard to detection limits and lack of filtration of samples, and (2) sampling that was not sufficiently interspersed over time or representative over space, having been focused on many of very near shore and lower salinity areas.

Salinity and Inorganic Nutrients

The DEP collected water samples from stations distributed widely over the Bay, with most samples taken in the nearshore. There were no systematic efforts to sample at regular intervals over the seasons, nor over tide cycles. To see if the DEP samples encompassed the range from freshwater to seawater, we plotted the frequency distribution of measurements of salinity (Figure 2). We calculated salinity from the reported measurements of chloride concentrations, since the salinity measurements actually reported by DEP appeared to have irregularities. We calculated the salinities corresponding to the chloride concentrations as [0.03 + 1.805 x chloride (mg/l)/1000 (Riley and Chester 1971)].

There were many stations in freshwater, some stations in mixes of freshwater and seawater, and a substantial number of stations in salinities that spanned the range of Buzzards Bay seawater (28-32 parts per thousand [ppt]).

The intent of the sampling was to assess water quality, and to pinpoint areas where nutrient loading (and other contamination) might be a problem. Nutrient loading from land to coastal waters occurs principally because of human

Table 2. Ranges of concentrations (μM) of nutrients in Buzzards Bay and, for comparison, in nearby coastal waters.

	NO_3	NH_4	DIN	PO_4	Reference
Buzzards Bay[a]	0.006-0.11		0.27-3.5		Roman & Tenore (1978) Roman (1980)
Buzzards Bay	0-0.4	0-5.5		0.3-1.7	Valiela et al. (1978)
Buzzards Bay[b]	1.4-785.7	0.71-478.3		0.32-126.7	DEP study (Gil, 1987, 1988) also Stenner et al. (1988)
Buzzards Bay[c], including					
New Bedford Harbor area	0.01-4.16	0.07-70.53		0.14-6.72	SMU study (Turner et al., 1989)
New Bedford Harbor area	0.1-3	0-14		0.3-6.2	Camp, Dresser & McKee
Undiluted sewage effluent[d]	1.4-119	271-546		33-97	Report (1990)
Buttermilk Bay[e]	0.1-38	0-11.3		0.05-2.2	Valiela & Costa (1988)
Vineyard Sound	0-1	0-0.8			Goldman & Dennett (1983)
Vineyard Sound	0.3	1.9		0.7	Mlodizinska, WHOI unpubl. data 1988
Waquoit Bay	0-2.4	0.2-3.1		0.5-1.1	Valiela & Costa, unpubl. data 1986-1989
Siders Pond					
Oxic layer	0-0.2	0.6-0.8		0.04-0.06	Caraco (1986)
Anoxic layer	up to 38	up to 1000		to 60	

[a] 13 m open bay station. Seasonal range (1974-1975) of average integrated water column values, calculated from data in papers.

[b] Summer months during 1985 and 1986; many samples in freshwater reaches and tidal embayment zones near land sources of nutrients.

[c] Sampling through an annual cycle (1987-1988) in 8 stations in more saline open waters of the bay.

[d] Samples taken October 1987 to May 1988.

[e] Sampling throughout 1985 and 1986 at numerous "open water" and "nearshore" stations.

activities in the adjoining watersheds (Jaworski, 1981; Nixon et al., 1986; Valiela and Costa, 1988). Nutrients from watersheds are transported to receiving waters by freshwater. Hence, our first step to examine the available data was to plot nutrient concentrations versus salinity for all the DEP samples, to see if we could find a correlation of lower salinity and higher nutrient concentrations.

None of the nutrients measured as part of the DEP monitoring (cf. Table 2 in Stenner et al., 1988) showed a clear relation to salinity. We show some scatter plots that are representative of the lack of relationship (Figure 3).

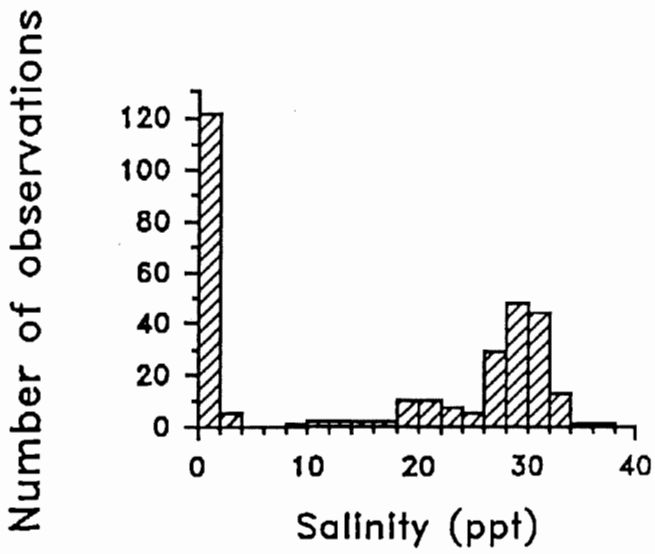

Figure 2. Frequency distribution of salinity in DEP water samples from Buzzards Bay

Scatter plots such as those of Figure 3 include geographical variation of the variables that might mask significant relationships within a locality. To examine this possibility, we graphed scatter plots of data within each of the regions of Buzzards Bay used by Stenner et al. (1988). There were no trends of nutrients and salinities within individual regions or estuaries, and for the sake of brevity we do not include the scatter plots.

The lack of relation of nutrients and salinity is unusual. Even where, as in Buttermilk Bay, there are a multitude of point sources for nutrients, we still found a significant relation between nutrient concentrations and salinity (Valiela and Costa, 1988).

Since the usual nutrient/salinity approach to study loading and distribution did not work, we next tried to define most frequent (modal), upper, and lower concentrations for each of the nutrients measured in the DEP sampling. To do this, we plotted the frequency distributions of the specific measurements, irrespective of date or station (Figure 4). For the nitrogenous species modal concentrations were less than 0.2 mg/l of nitrate, and less than 0.08 mg/l of ammonium (Figure 4a,b,c). These values translate into 14.3 μM nitrate and 5.7 μM ammonium, concentrations higher than others have reported for water of Buzzards Bay and for nearby Vineyard Sound (Table 2).

Even in Buttermilk Bay, a very shallow embayment with many houses in the watershed, and in the New Bedford Harbor area, where an outfall and a contaminated river add nutrients, upper values of nutrient concentrations are considerably below those reported for the DEP data (Table 2). The only concentrations of NH_4 that reach values comparable to those of the DEP data are the anoxic lower layers of Siders Pond, a very eutrophic pond, and undiluted sewage effluent (Table 2). Buttermilk Bay, New Bedford Harbor, and Siders Pond are examples of places we would expect that nutrient concentrations in our general area would be highest and water quality lowest. Concentrations of such magnitude may certainly occur here and there, and at one time or another

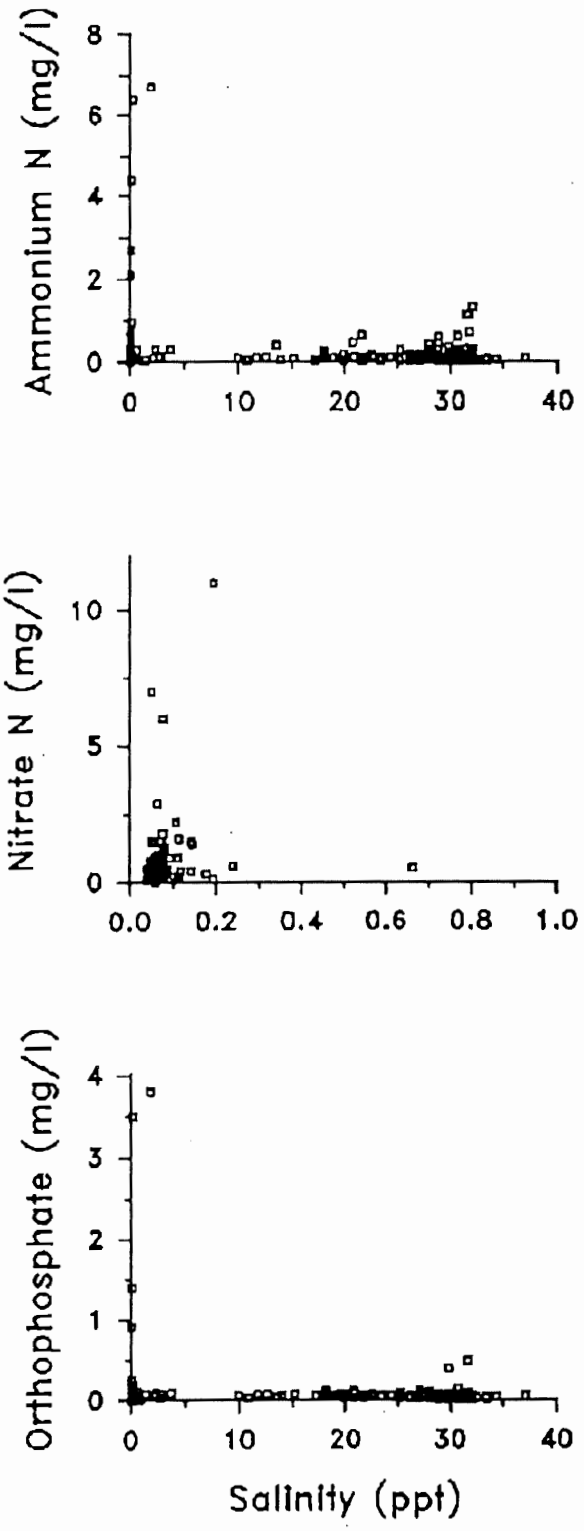

Figure 3. Relation of ammonium, nitrate, and phosphate to calculated salinity in DEP water samples from Buzzards Bay.

in Buzzards Bay. It seems unreasonable, however, to find that modal concentrations and lower bounds (Fig. 4) in the Bay exceed or resemble the highest concentrations recorded in the region.

Fresh water often contains much greater concentrations of dissolved inorganic nutrients than does seawater (Valiela, 1984). To check whether the large concentrations of NH_4 and NO_3 in Figures 4a,b were not merely because freshwater samples dominated the data set, we sorted the DEP samples into two categories, fresh (0-1 ppt salinity) and seawater (28-32 ppt salinity). Then we plotted frequencies of concentrations of NH_4 and PO_4. Unfortunately, there were very few reported nitrate concentrations from samples of seawater (cf. lower axis of Figure 3), so nitrate was not included. At any rate, there were no major differences in the distribution of NH_4 or PO_4 concentrations in fresh or seawater samples (Figure 5a,b). This suggests that the high concentrations of nutrients (cf. Figure 4) could not be attributed merely to freshwater samples.

The unusually high concentrations of nutrients may be evidence of truly eutrophic conditions in the Bay; this seems to be unreasonable considering the rather good water quality that is evident over most of Buzzards Bay. The collection of samples primarily from nearshore locations is probably the reason that the DEP data may not be representative of the main body of the Bay. Alternatively, the high concentrations may be evidence of inappropriate methodology. For example, from what we understand of DEP protocols, unfiltered water samples are routinely fixed by adding sulfuric acid, and this is not a standard procedure for the standard seawater analyses. More significant perhaps is that the lower detection limit of the methods used seems too high for the concentrations of nutrients we might expect in seawater (Table 3). For samples of

seawater, methods than can measure at least 0.05 µM are needed for inorganic nutrients.

Total Nitrogen and Phosphorus

The DEP data on total nitrogen show a range of 0 to 4.5 mg N/l (Figure 6a). The most frequent concentration is about 1 mg N/l; once again this seems high. For instance, the target concentrations for total nitrogen for the most enriched class of receiving water in the nutrient-loading bylaw of the Town of Falmouth is 0.75 mg N/l. This threshold was derived based on scientific experience with coastal bays and ponds on Cape Cod, and was established on the basis of our experience about the sum of NH_4, NO_3, DON, and particulate nitrogen present in waters of coastal lagoons and bays. This target concentration exists in water that is frequently turbid and soupy green, such as that of the eutrophic Siders Pond (Caraco, 1986). A lower threshold of 0.32 mg N/l is also included in the bylaw and is applicable to cleaner waters. Both of these limits are exceeded by most of the measurements of total nitrogen in the DEP data (Figure 5a). This seems unreasonable knowing the overall quality in the Bay and surrounding waters.

Once again to see if the high total nitrogen and phosphorus concentrations were associated with freshwater samples, we sorted the DEP data into fresh (0-1 ppt) and seawater (28-32 ppt) types (Figure 7a,b). Although the modal freshwater total N was a bit higher than the mode for seawater, the difference was small (Figure 7a). Seawater tended to have somewhat greater total phosphate content than freshwater (Figure 7b).

The ratio of N to P has been suggested as a rough index of the relative importance of nitrogen or phosphorus in limiting rates of primary production (Jaworski, 1981). In theory, N:P

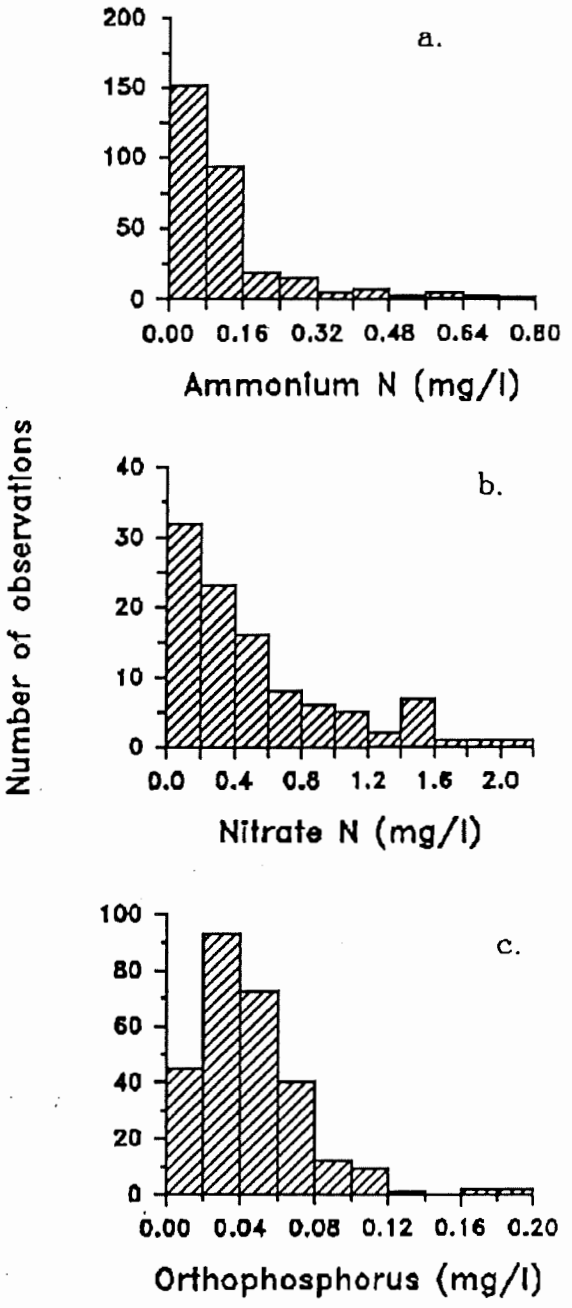

Figure 4. Frequency distribution of concentrations of a) ammonium, b) nitrate, and c) phosphate in DEP water samples from Buzzards Bay.

Table 3. Minimum and maximum concentration of three nutrients in DEP data set.

Units	NO_3	PO_4	NH_4
mg/l	0.01-6.7	0.02-11	0.01-3.8
mM	0.71-478.3	1.43-785.3	0.32-126.7

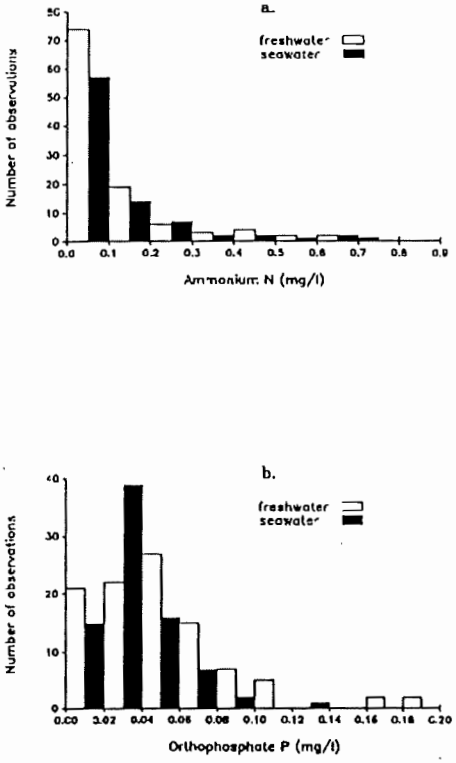

Figure 5. Frequency of concentrations of a) ammonium and b) phosphate in DEP water samples sorted into freshwater (salinity 0-1 ‰) and seawater (28 to 32 ‰ salinity samples).

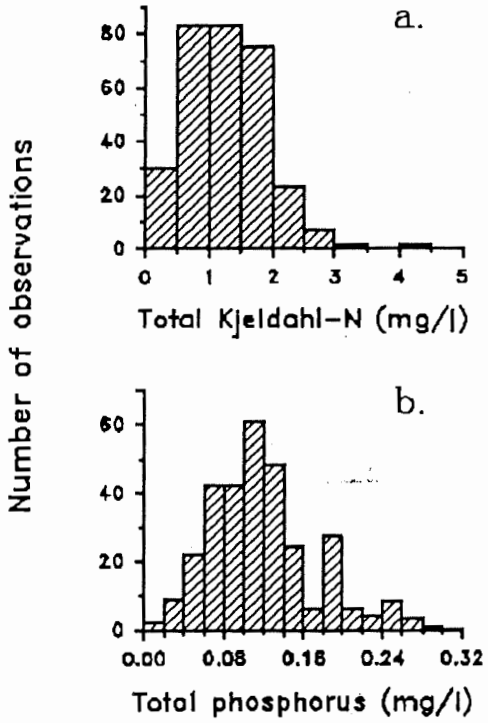

Figure 6. Frequency distribution of concentrations of a) total nitrogen and b) total phosphorus in DEP samples from Buzzards Bay.

<16 (often found in seawater) should suggest nitrogen limitation, while N:P >16 (more frequent in freshwater) suggests that phosphorus is the limiting nutrient. For the DEP data for both fresh water and seawater, the frequency of low total nitrogen values in seawater is a bit greater than for freshwater (Figure 7a). The frequency of total phosphorus, on the contrary, is greater for fresh water than for seawater (Figure 7b). These trends are as might be expected. There is something anomalous, however, about the N to P of these samples (Figure 7c). Values of N:P distributed very broadly on either side of the Redfield value of 16:1. It is unclear what this means. More of the freshwater than seawater values lie below 16:1, suggesting that more of the freshwater samples may contain nitrogen-limited phytoplankton, contrary to what might be expected.

Oxygen, Coliforms, and Nutrients

Dissolved oxygen [and our calculation of percent oxygen saturation based on temperature of the water (Weiss, 1970)] seems reasonably high for most samples collected by DEP (Figure 8a,b). Sustained oxygen concentrations lower than 5 mg/l are unsuitable for animal survival, and are of concern. Table 4 shows the list of sites where such low oxygen values were recorded, although not much should be made of such a list. Low oxygen events tend to be short-lived, and the sampling schedule would have certainly missed most anoxic events. In addition, many shallow ponds and lagoons may become anoxic at certain times of the year and day, even without anthropogenic nutrient loadings. Sampling of waters of Buzzards Bay should consider the possibility of such events.

Oxygen concentrations were not related to concentrations of NH_4 or any other nutrient; nor were they related to salinity (data not shown). This is disappointing, since no attribution or correlation to any loading factor is therefore possible.

Fecal coliforms reported by DEP varied broadly (Figure 9), with most samples showing a very low number. Further examination of this indicator is found in a companion report. Unfortunately fecal coliforms are unrelated in the data to either oxygen saturation (Figure 8a), or to ammonium or nitrogen content (Figures 10b,c).

Conclusions from the DEP Data

If the DEP data were to be taken at face value, Buzzards Bay would have to be considered a highly eutrophic water body, and one in which anomalous relations among variables are common. It seems unlikely that is the case. The failure to find any relation to salinity, the very high reported concentration of nutrients in water that appears to be of good quality, the high reported concentrations of total nitrogen and phosphorus where other reports find much lower concentrations, and the anomalous values of N:P all suggest that the data examined in this DEP report need to be considered cautiously. There seem to be few interpretable or reasonable relationships among water quality indicators (oxygen concentrations and saturation, coliforms, or nutrients) in the DEP data set, although these are indicators that usually can be correlated in some way.

The variability, generally high concentrations, and lack of expected relationships are probably the result of at least two features. The relatively high values for NH_4, NO_3, PO_4, total N and P may have been the result of application of methods not best suited for the low-nutrient, buffered, salty character of seawater samples. Monitoring of Buzzards Bay waters requires sampling of waters of very different kinds. This makes the choices for strategy of sampling and methods of analysis understandably difficult.

The lack of expected relationships among variables that ought to show some relation, and the apparent random and high variability of the data are probably the result of the sampling schedule. Tidal, seasonal, hydrological, and spatial factors are confounded, and their combined variations make it very difficult to interpret the data.

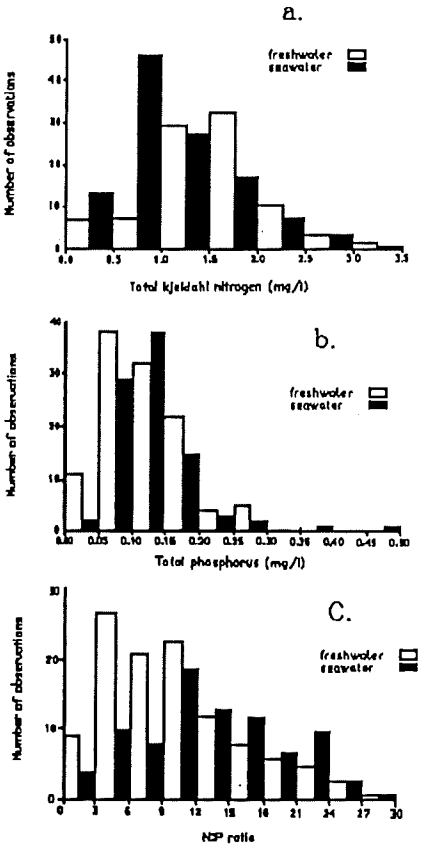

Figure 7. Frequency distribution of a) total nitrogen and b) total phosphorus, and c) N:P in fresh (0-1 ‰) and seawter (28-32 ‰ samples collected by DEP.

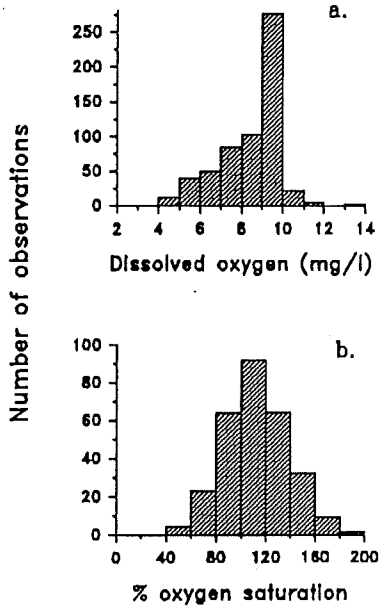

Figure 8. Frequency distribtuion of b) dissolved oxygen and b) percent oxygen saturation in DEP water samples.

Table 4. Oxygen concentrations for each station observed in DEP 85-86 Survey

Station ID	Station Type	Station Basin	Area	Temperature Date(°C)	DO (mg/l)	Salinity (ppt)	% O_2 Saturation
19MH110		FW	10	327-Aug-85	17.2	4.2	0.0846.6
19WEO110		FW	6	222-May-85	17.0	4.6	0.0550.8
19WEO110		FW	6	213-Aug-85	21.1	3.8	0.0645.6
19WEO110		FW	6	213-Aug-85	23.3	4.4	
19WEO110		FW	6	214-Aug-85	25.6	3.1	
19WEO110		FW	6	214-Aug-85	22.2	3.6	0.0544.2
1PR01		FW	2	122-Jul-86	20.0	4.2	0.0849.4
1PR01		FW	2	122-Jul-86	24.4	4.2	
1PR01		FW	2	123-Jul-86	20.0	4.3	0.0750.5
20MH140		FW	10	327-Aug-85	23.3	4.2	
20MH140		FW	10	327-Aug-85	21.6	4.9	0.0659.4
20MH140		FW	10	328-Aug-85	21.1	3.3	0.0639.6
20MH140		FW	10	328-Aug-85	23.3	4.1	
29SRO160		FW	6	214-Aug-85	23.3	3.7	
29SRO160		FW	6	214-Aug-85	21.7	4	0.0748.6
2BB020		FW	8	213-Aug-85	22.2	4.9	0.0560.1
2BB020		FW	8	214-Aug-85	22.2	4.5	0.0555.2
2DB010		FW	4	115-Oct-86	12.0	4.7	0.0546.6
2PR02		FW	2	123-Jul-86	23.3	4.9	
30HB010		SW	10	327-Aug-85	21.6	4.6	18.9862.3
31WSH020		SW	10	328-Aug-85	18.9	4.6	11.7656.7
35GSC020		SW	10	327-Aug-85	22.2	4.5	30.7266.0
36LSC020		SW	10	327-Aug-85	22.2	4.4	30.7264.5
36MR080		FW	5	213-Aug-85	20.5	4.6	0.0554.6
36MR080		FW	5	214-Aug-85	21.1	4.7	0.0556.5
37PI010		SW	5	214-Aug-85	19.4	4.8	0.1155.8
38MHO300		SW	5	213-Aug-85	20.0	4.2	29.1858.6
38MHO300		SW	5	214-Aug-85	23.3	4.4	28.9165.2
4ACR02		FW	4	114-Oct-86	18.0	4.3	0.0648.5
4ACR02		FW	4	115-Oct-86	12.0	3.8	0.0737.6
4ACR02		FW	4	116-Oct-86	7.0	3.6	0.0731.6
4PR04		FW	2	122-Jul-86	21.7	4.7	
4PR04		FW	2	122-Jul-86	18.9	4.9	0.0856.3
4PR04		FW	2	123-Jul-86	22.2	4.7	
4PR04		FW	2	123-Jul-86	19.4	4.8	0.0855.7
6GB040		FW	8	214-Aug-85	23.3	4.9	0.0661.4
6PR06		FW	2	123-Jul-86	20.6	4.9	0.0758.3
9BWB02		FW	3	122-Jul-86	18.9	3.6	0.0841.4
9BWB02		FW	3	122-Jul-86	23.9	3.9	
9BWB02		FW	3	123-Jul-86	19.4	3.3	0.0838.3
9BWB02		FW	3	123-Jul-86	25.0	4.6	
9PR040		SW	9	327-Aug-85	20.6	4.6	22.5962.5
9PR040		SW	9	328-Aug-85	20.6	4.7	24.4064.5

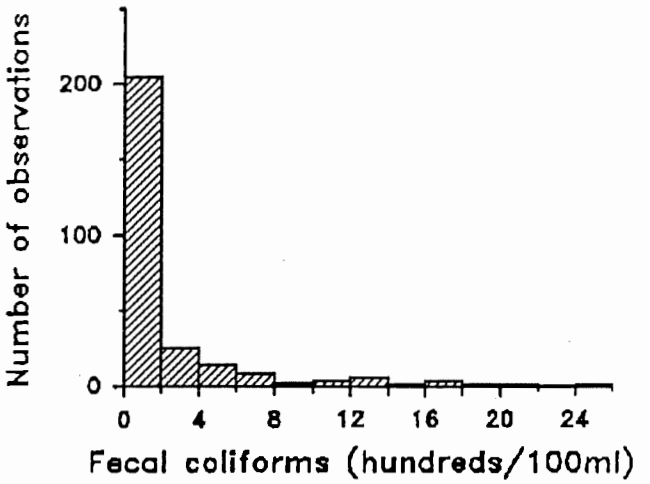

Figure 9. Frequency distribution of fecal coliform samples in DEP water samples.

The SMU 1987-1988 Survey

The bay survey of Turner et al. (1989) offers an extensive data set, based on accepted techniques that are commonly used for coastal waters. The study seems a rich source of information, worthy of further analyses, which the SMU group are conducting. The survey covered eight stations occupied repeatedly through an annual cycle (Figure 11); unlike the DEP survey, these stations all had high salinity.

Since these researchers have not yet published their data, we limited our analysis of the SMU data set. We chose four stations, which covered highest to lowest nutrient conditions of the full set (Figure 12). We combined nitrate and ammonium values (DIN), but most of the DIN usually was ammonium. We also averaged the surface, midwater, and bottom samples at each station.

We had four specific objectives. The first objective was to compare the seasonal range of variation of nutrients at selected locations. The second objective was to display nutrient concentrations vs. salinity and concentrations as frequency distributions as we had for the DEP data, to see if the data set provided a different picture of the Bay. The third ojective was to look at the data for evidence on whether nitrogen seems more limiting to production than phosphorus. The fourth objective was to examine a gradient from the lesser enriched to the more enriched nutrient conditions, with the aim of identifying any patterns related to enrichment.

Variability Through Season And Across Stations

In the central Bay (Station 5), the range in nutrient and chlorophyll concentrations was about double the annual mean of each variable (Figure 13 middle). A strong seasonal temperature cycle did not coincide with an obvious seasonal cycle of either nutrient, which is unlike neighboring bays. There was suggestion of a chlorophyll minimum in mid-spring, after the maximum reached during early winter, but there was no clear summer maximum. A chlorophyll peak during winter-spring (the "spring bloom," initiated by increasing solar angle and day length at a time when nutrients are available), followed by a minimum (depletion of nutrients), is common for coastal and ocean waters. The winter-spring bloom in Figure 13 middle seems early, but timing varies from year to year. Many coastal areas do have a broad seasonal peak in chlorophyll during summer, although there may be no standard pattern (Nixon, 1986).

For DIN, there was temporal variation in concentration at each station, but there was no clear seasonal pattern across the four stations of Turner's survey we examined (Figure 13 bottom). Turner et al. (1989) suggest this is true baywide.

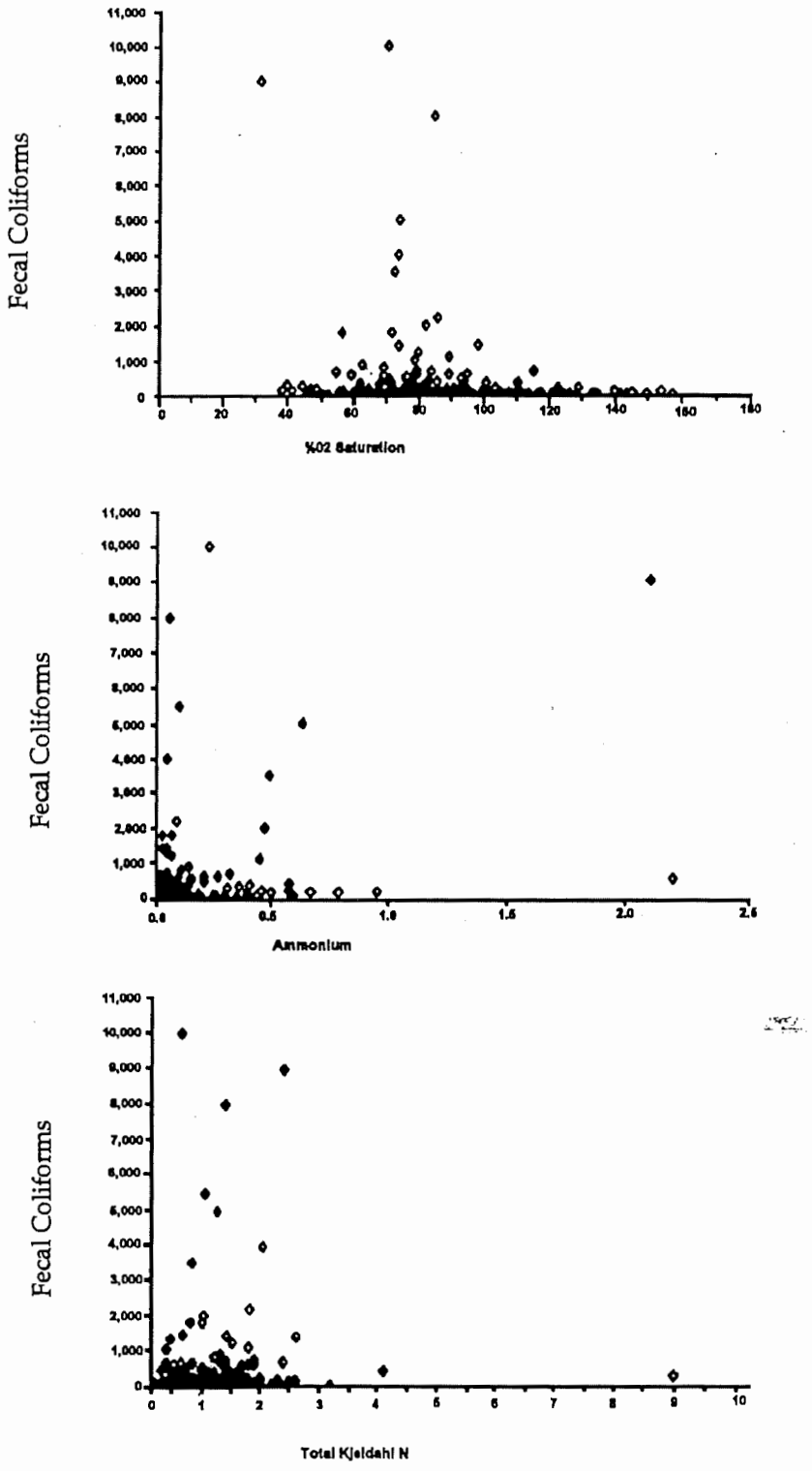

Figure 10. Relation of fecal coliforms and ‰ O_2 saturation, ammonium, and total nitrogen in DEP water samples.

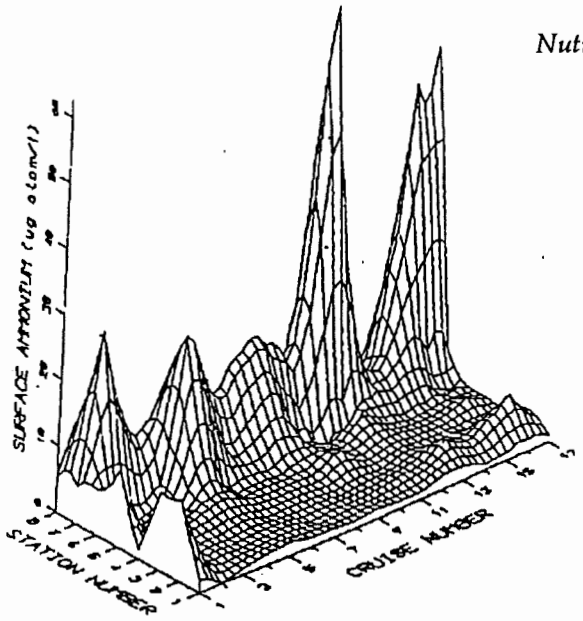

Figure 11. Surface ammonia concentrations measured at 8 stations on 17 dates (Data summarized from Turner etal. 1989; compare to Figure 12).

Even adjusting for different water depths at the four stations, DIN (as mmol m^{-2}) reached highest values at the New Bedford outfall followed by the Inner Harbor stations (Figure 13 bottom). Variability at these New Bedford sites was such that values were sometimes in the range of the central Bay station, but on the average DIN was still higher at these two sites proximal to the major population center of the Bay. Lowest integrated water column DIN values were seen in Mattapoisett Harbor, which had concentrations similar to those at the central Bay station, although the water in the harbor was shallower.

For chlorophyll, highest values occurred at the two New Bedford sites. Peak values generally were found during summer in the Inner Harbor, not adjacent to the outfall (Fig. 14). Across the stations, chlorophyll *a* concentrations had no regular seasonal pattern, but some did show a late summer maxima. Values at the central Bay station and in Mattapoisett Harbor were comparable.

The only distinct vertical profile for either chlorophyll or nutrients was for nutrients near the New Bedford outfall. Turner et al. (1989) and Camp Dresser and McKee (1990) reported DIN and PO4 values in this area over some of the same time period (Figure 15). Main differences seemed to be (1) higher surface values often recorded in the Turner survey—a difference, that seemed to apply to both DIN and PO4—whereas (2) the 25% isolume depth samples (mid-water) reported in Camp Dresser and McKee (1990) were occasionally lower than mid- or bottom-water samples of Turner et al. (1989). The two studies did not necessarily occupy precisely the same station. Given the vagaries of effluent plume rise and dispersion, it is reasonable to assume that the differences, especially at the surface, represent spatial variability rather than fundamental methodological differences. The wider net of sampling stations and parameters encompassed by the two studies in the area could be further compared, but the levels seem broadly comparable, lending additional confidence to the data sets.

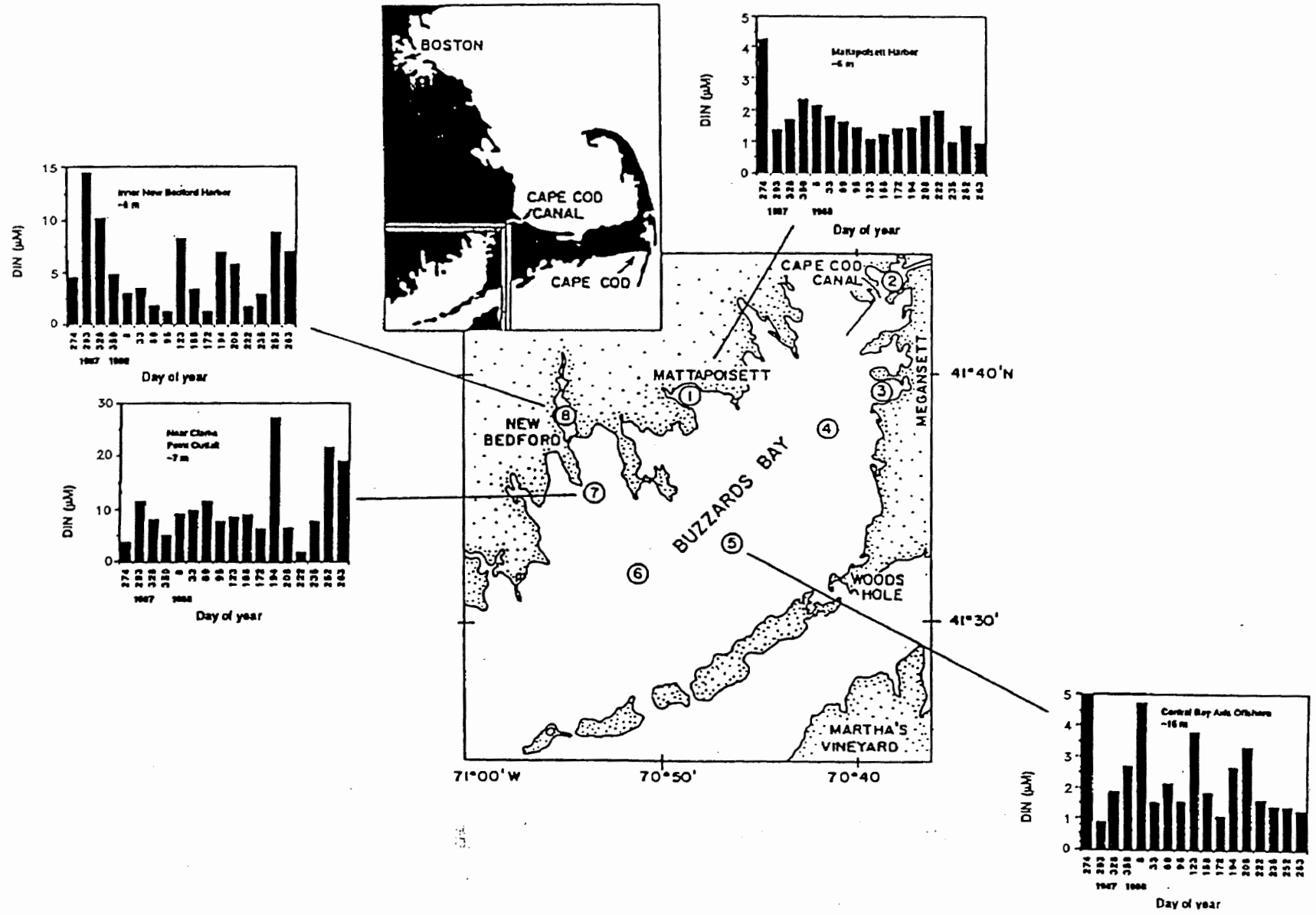

Figure 12. Dissolved inorganic nutrient (DIN) cycles at four stations are overlain on the station map reproduced from Turner et al. (1989). Data were summarized from Turner et al. (1989).

Bay Nutrient Concentration Patterns Based On The 87-88 Survey

The plot of nutrients vs. salinity is confined to a small salinity range, mostly from 27 to 31 ppt. The Turner data depict seawater locations *within the Bay,* accessible only boat, and do not represent tributary or freshwater sites on the Bay's perimeter. There was no striking relation of nutrients to salinity and no large seasonal variation in salinity at any station. At a given salinity, the DIN and PO_4 concentrations varied greatly (Figure 16); for example, the range for DIN and PO_4 at 30 ppt (~0.3 to 40 μM N, ~0.23 to 7 μM P) is nearly as broad as across all salinities. The station near the New Bedford outfall and the station in Inner New Bedford Harbor were often higher in nutrients than the central Bay station and Mattapoisett Harbor, but none of the individual stations showed a strong pattern of nutrients varying with salinity.

Point sources of nutrients apparently cause geographic variations of nutrient-related variables without also leaving a strong salinity co-signature in areas of Buzzards Bay. Thus, it is possible that the ability to distinguish patterns with salinity within a data set like the DEP compilation is confounded in part by such features. This again points out the need to plan a series of selected local and regional scale studies that do not suffer from such confounding factors.

The concentrations of nutrients shown in frequency plots (Figure 17), show that most samples for DIN were less than 4 μM and for PO_4 were less than about 1 to 1.5 μM. If one considers only the central Bay offshore station and Mattapoisett Harbor, the above concentrations were rarely exceeded. High values were found principally at the two stations near New Bedford.

Insets in Figure 17 are in mg/l, rather than μM, to present a scale similar to Figure 6 for comparison to the DEP data. The frequency distributions for the two data sets on a mg/l classification scale are remarkably similar in modal values, but note that only the New Bedford area samples extend to the much higher classes that appeared in the DEP set. In this case, it is clear that the scale, chosen on the

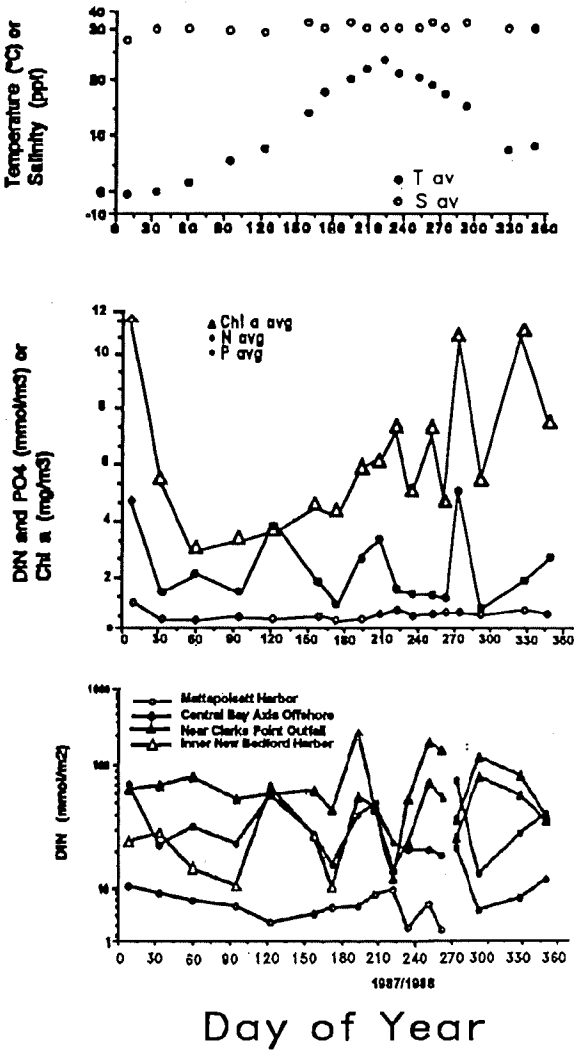

Figure 13. A seasonal cycle for temperature salinity, nutrients and chlorophyll for central Buzzards Bay, 1987-1988. Based on Turner etal (1989) data for their station 5.

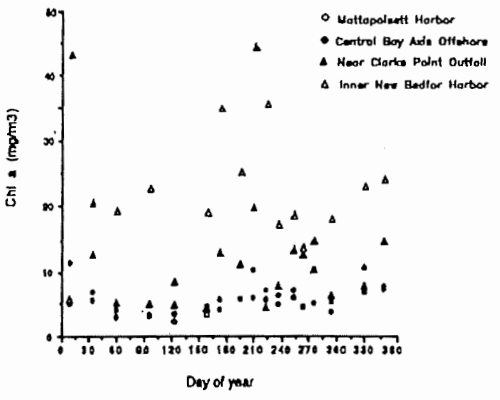

Figure 14: Comparison of DIN and chlorophyll concentrations at 4 stations in Buzzards Bay, 1987-1988. Based on data of Turner et al. (1989).

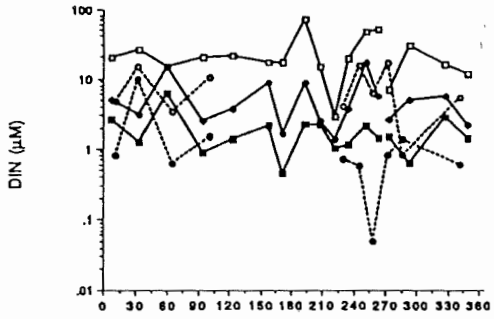

Figure 15: Comparison of nutrient concentrations measured near the New Bedford outfall during 1987-88. Data from Turner et al. (1989) for surface (s), mid (m) and bottom (b) samples at their station 7. Data from Camp, Dresser, and McKee (1990) for surface and 25% isolume depth for station 2 of the outer Harbor.

basis of minimum concentrations for the data reported in mg/l, is misleading. The sensitivity of the standard chemical analyses for inorganic nitrogen in seawater (methods used by Turner et al. 1989) allows discrimination of smaller classes, even smaller than the 2 µM increments shown in the main plots of Figure 17. When the Turner data are arrayed in this way, as is the convention for seawater samples, it is evident that the majority of samples had low nutrient concentrations.

On the basis of this data set, Buzzards Bay is not highly nutrient-enriched. This is contrary to the conclusion that could be drawn from the DEP data. The Turner values are broadly comparable to others in the region (Table 2), and the body of points, with selected exceptions related to the New Bedford outfall, suggests the main portion and most of the volume, of Buzzards Bay has nutrient levels typical of less enriched coastal ecosystems.

Inorganic N and P: Potential Limiting Nutrients

For the four stations of Turner et al. (1989), the ratio of DIN to PO_4 was consistently lower than a 16:1 Redfield model for plankton tissue (Figure 18). Indeed, there was substantial PO_4 in waters where DIN approached detection limits (Figure 19; note that the insert excludes New Bedford area points and uses a linear scale). A pattern so strong and consistent, with little seasonality, can indicate that nitrogen is more limiting to productivity than phosphorus.

If relative nutrient limitation is to be determined, supplies (i.e., not concentrations) of nitrogen and phosphorus, including those from recycling, and including all forms (i.e., total of the nutrients), also need consideration. Only a portion of that information is available for Buzzards Bay. For example, sewage effluents from New Bedford (Figure 19) also had low DIN:PO_4 ratios (~5.9:1). Ratios at the outfall station often reflected this signature (Figure 19), since the mean annual N:P value at this station was 5.7.

Most samples tended towards even lower ratios (Figures 18, 19), and the mean annual N:P at the three other stations ranged from 2.9 to 3.9 N:1 P. The NOAA loading estimates earlier in this report suggested an N:P input ratio of 5.3:1; if precipitation is included (very high in nitrogen relative to phosphorus) the ratio could be over 7:1. (The Buzzards Bay

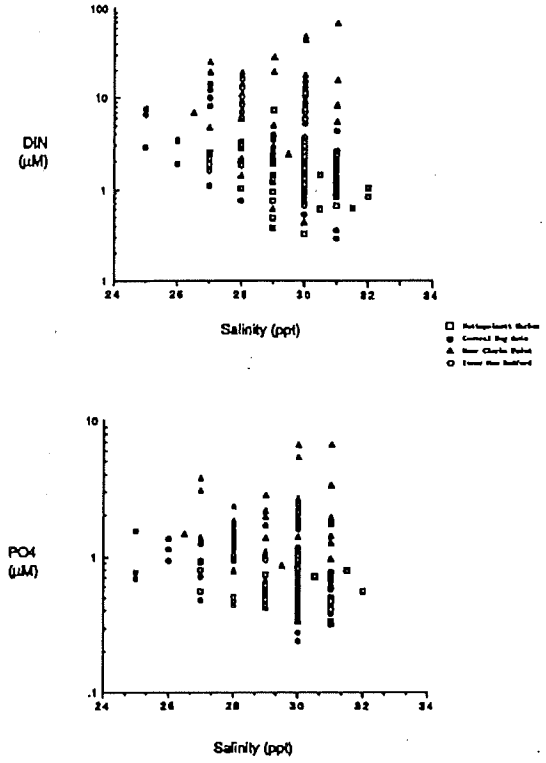

Figure 16. Nutrients vs. salinity during 1987-88 at four Buzzards Bay stations. Data from Turner et al. (1989).

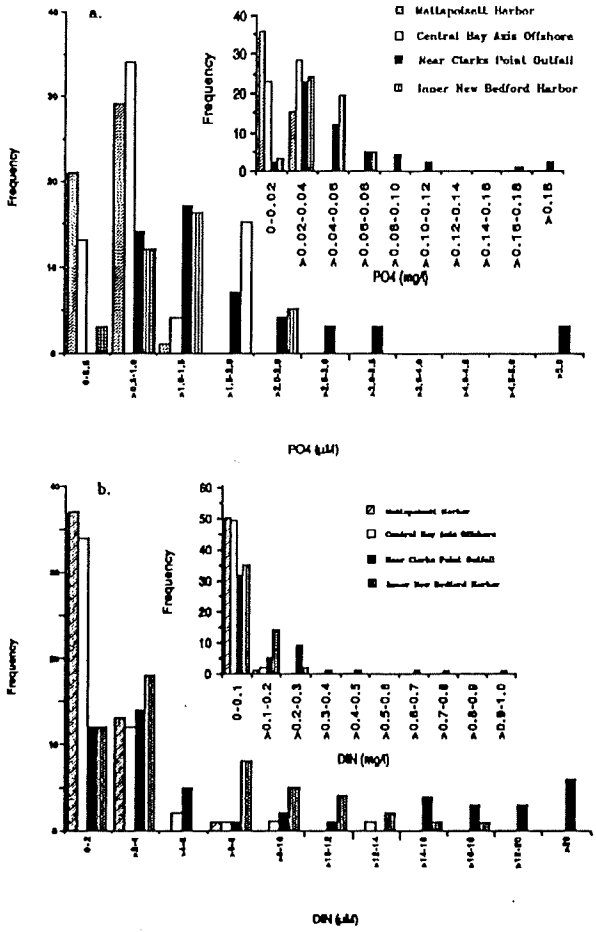

Figure 17. Frequency distribution of nutrients. (a) DIN concentrations as (mM) or mg/l (inset). (b) PO4 concentrations as (µM) or mg/l (inset). For 4 stations sampled during 1987-88 by Turner et

Project has not yet provided estimates of phosphorus loading for comparison to their estimate of nitrogen loading.) The available data on inputs, when compared to *in situ* N:P ratios (Figures 18, 19) suggest that activity within Buzzards Bay (or the import from offshore waters) may lower the incoming DIN:PO_4 ratio even more. Similar results have been found in other bays (Figure 20), and suggest nitrogen (rather than phosphorus) limitation. Nutrient interaction between water and sediments is one mechanism that promotes low N:P ratios in coastal waters (Nixon et al., 1980; Kelly, 1990).

Turner et al. (1989) suggest the possibility of silicate limitation of phytoplankton production in the Buzzards Bay. The ratio of silicate to nitrogen has not yet been shown to affect the metabolic consequences (and prospect for low-oxygen events) of nutrient enrichment, but there may be consequences for the structure of the food chain (Doering et al., 1989).

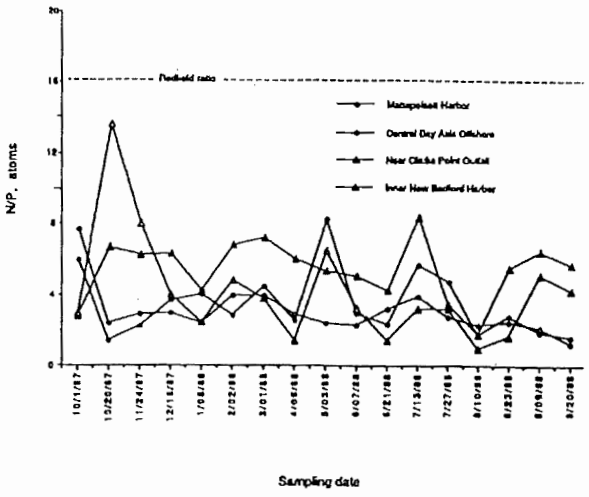

Figure 18:. DIN/PO_4 ratios in Buzzards Bay in 1987-88. Data from Turner et al. (1989).

Pattern of Chlorophyll with Enhanced Nutrient Levels

Chlorophyll increases as average nitrogen concentrations increase (Figure 21) in Buzzards Bay, but only roughly following a rule of thumb that each factor of 10 increase in DIN (μM) may bring a fourfold to fivefold increase in chlorophyll *a* (μg l^{-1}) (derived from Nixon et al. 1986). Near the outfall, chlorophyll levels appear depressed relative to the enriched nitrogen levels, perhaps a consequence of inhibition of primary producers by sewage effluent (Camp Dresser and McKee, 1990). In contrast, the Inner Harbor of New Bedford has a higher chlorophyll level than might be suggested by the DIN. Currents may focus particulate materials into this area, leading to accumulation of chlorophyll higher than sug-

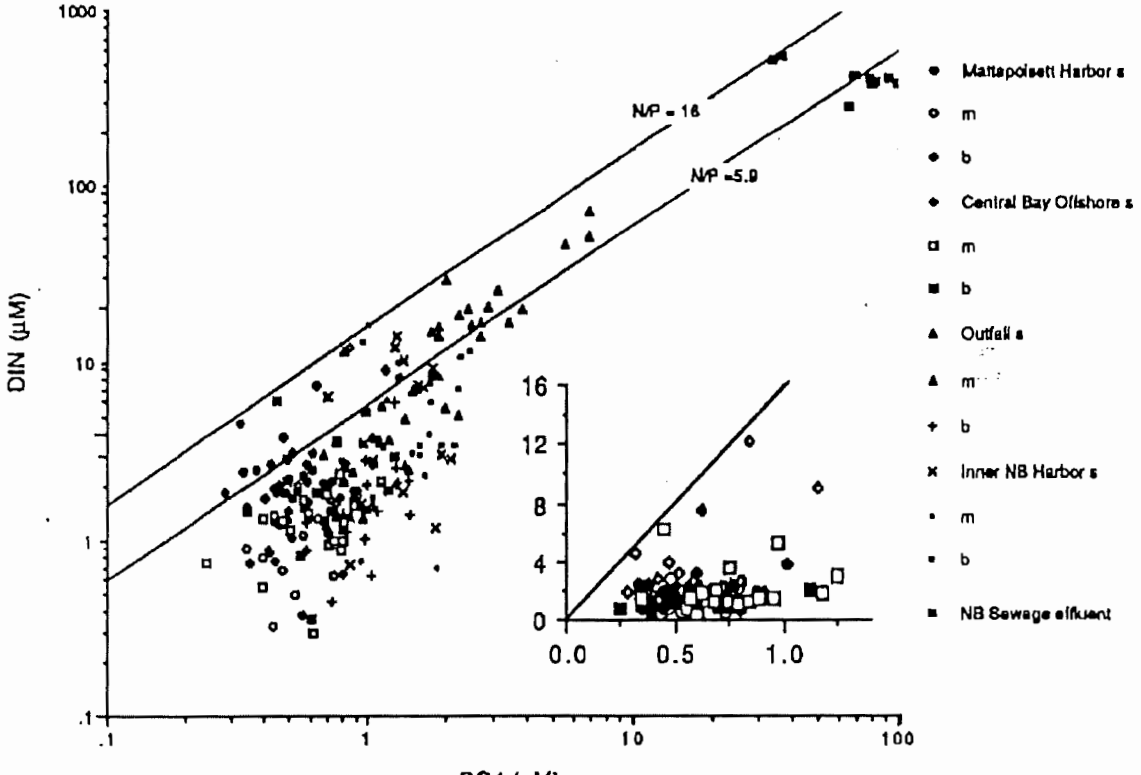

Figure 19: DIN vs. PO_4 in different areas of Buzzards Bay and in New Bedford sewage effluent. Legend: s, m, b for stations are for surface, mid, and bottom water samples from Turner et al. (1989); NB = New Bedford effluent concentrations (Camp, Dresser, and McKee 1990). Isopleth lines show Redfield ratio of 16 N:1 P (atoms) and average N/P ratio (5.9 N:1 P) for composite effluent samples, October 1988 to May 1989. Insert shows linear scale for Mattapoisett Harbor and Central Bay samples only, with 16 N:1 P isopleth line.

gested by the indigenous nutrient levels within waters restricted by the hurricane barrier (Turner et al., 1989).

Historical Trends of Concentration in Open Water

Major research and management concerns for any coastal area center on the way that water quality and biological resources have changed or might change over time as a consequence of human activities. This is a critical issue; unfortunately there is a paucity of historical data for Buzzards Bay. Rather than dismiss the issue entirely, we have included a brief illustration relative to determining temporal trends.

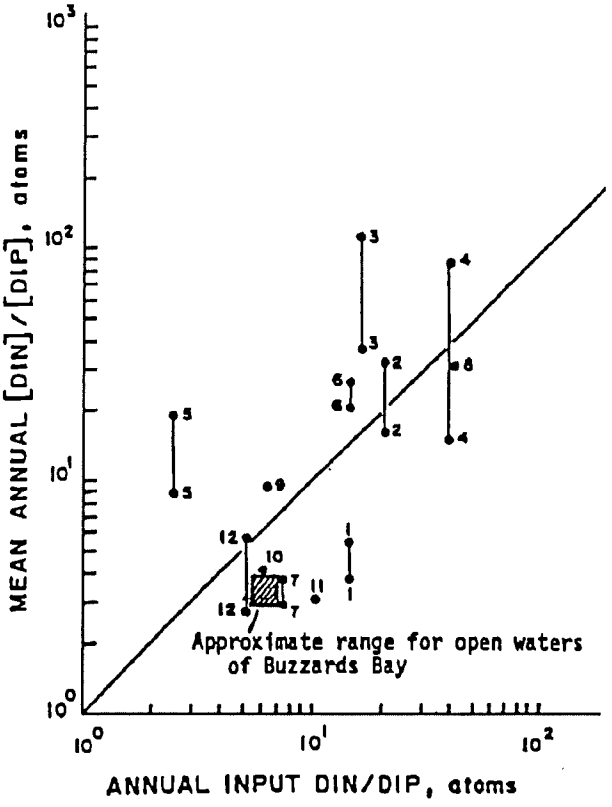

Figure 20. DIN/PO_4 ratios in water as a function of input to various U.S. estuaries. Modified from Nixon and Pilson (1984). Buzzards Bay estimated range is based on NOAA (1988) inputs, modified for precipitation (this report), and four stations of Turner

Annual cycles of chlorophyll and nutrients were reported for the 1970s (Rhoads et al., 1975; Roman and Tenore, 1978; and Roman, 1980), for a station representative of silt-clay deeper water areas of Buzzards Bay.[10] Concentrations of nutrients and chlorophyll at Turner's Station 5[11] seemed representative of the three central-axis open-water stations of the 1987-88 survey, owing to the well mixed character of the open Bay; and we compare them to the earlier studies by converting all data into the same concentration units (Figure 22). Comparing data gathered by different investigators at different times is an exercise that must always be interpreted very cautiously, even when the analytical techniques appear comparable. The problem of identifying what point in space was occupied for sampling arises if one must rely on past studies that were not designed specifically to assess long-term trends and were not documented accordingly.

If the data offered in Figure 22 were a valid comparison, it would appear that average values and maximum values of *both* dissolved inorganic nitrogen and

10The position of "Station H-7" (Rhoads et al., 1975, from Rhoads, 1973) or the "Black and White Gong Station" (Roman and Tenore, 1978) is shown on maps that have mislabeled axes for latitude and longitude. The position of H-7, judged from Rhoads (1973) looks to be roughly 7044'W, 4032'N, or south of a central Bay axis, roughly 3 nautical miles offshore of the Woods Hole area. The reported depth is 13 m as given by Roman and Tenore (1978), but 15.5 m by Rhoads et al. (1975).

11Station 5 was at a depth of about 14-15 m (depending on tidal stage) along the central axis of the Bay, apparently to the west of the earlier station. Stations 4, 5, and 6 were all at buoys along the central axis (10-15 m) and were located by Loran. Presumably, station latitude and longitude be provided in a future report.

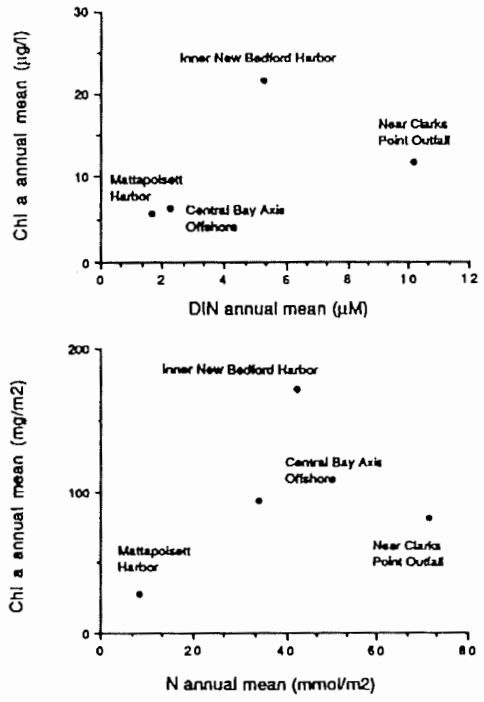

Figure 21. Annual mean chlorophyll *a* concentration as a function of annual mean DIN. Summarized from data of Turner et al. (1989). Difference between top (volumetric) and bottom (area-based) plots is due to differences in station

chlorophyll *a* were higher during the late 1980s than they were during the period of 1972-1976. Mean values of Turner et al. (1989) are roughly a factor of 2 higher. We hasten to point out that even the 1987-88 data do not suggest a highly enriched ecosystem.

Given other aspects of the Bay nutrient budget, would a doubling of dissolved nutrients between the mid 1970s and late 1980s be possible? Using the conservative input estimate of 79 mmol N m^{-2} y^{-1}, one can calculate that the mass of DIN within the entire Bay (based on the 1987-88 survey concentrations; volume = ~6×10^9 m^3, Table 1) could be replenished several times each year; thus, an increase of twofold theoretically is possible within the timeframe of a year. Over a 12-year period, if loading had averaged only 79 mmol N m^{-2} y^{-1}, an increase in nutrient concentrations Baywide by a factor of 2 would constitute water column DIN retention of only a small percentage (1 to 2%) of the total nitrogen input. Given that Bay water renewal by tidal action appears as fast or faster than nutrient mass replenishment, an increase in nutrients or chlorophyll could imply that rates of nutrient input had increased significantly, and/or that retention of nutrients and algal biomass, and the nutrient budget, was altered during these decades. It is not possible to comment further on either likelihood, and these calculations can not be used to confirm a time trend, only to suggest that the apparent concentration increases are possible.

The most striking lesson for future planning may be that, given the seasonal variations against which a relatively smaller secular trend must be judged, a convincing demonstration of an open-water Baywide trend would seem to require a consistent data set purposefully gathered over a time period on the order of a decade (unless loading levels jumped by orders of magnitude across annual periods). Moreover, most trends would be detected first at embayment-scale levels (which act as primary land discharge receiving systems) unless diffuse atmospheric loading were the major cause.

We chose the data sets for Figure 22 as among the best for a historical comparison. Yet, it is obvious that the validity of the comparison can be challenged; any dedicated effort at historical reconstruction must assess such concerns. Rather than providing a definitive answer to long-term changes, this exercise was intended to arouse discussion regarding the difficulties of making strong inferences without having systematically designed studies to answer questions on temporal change.

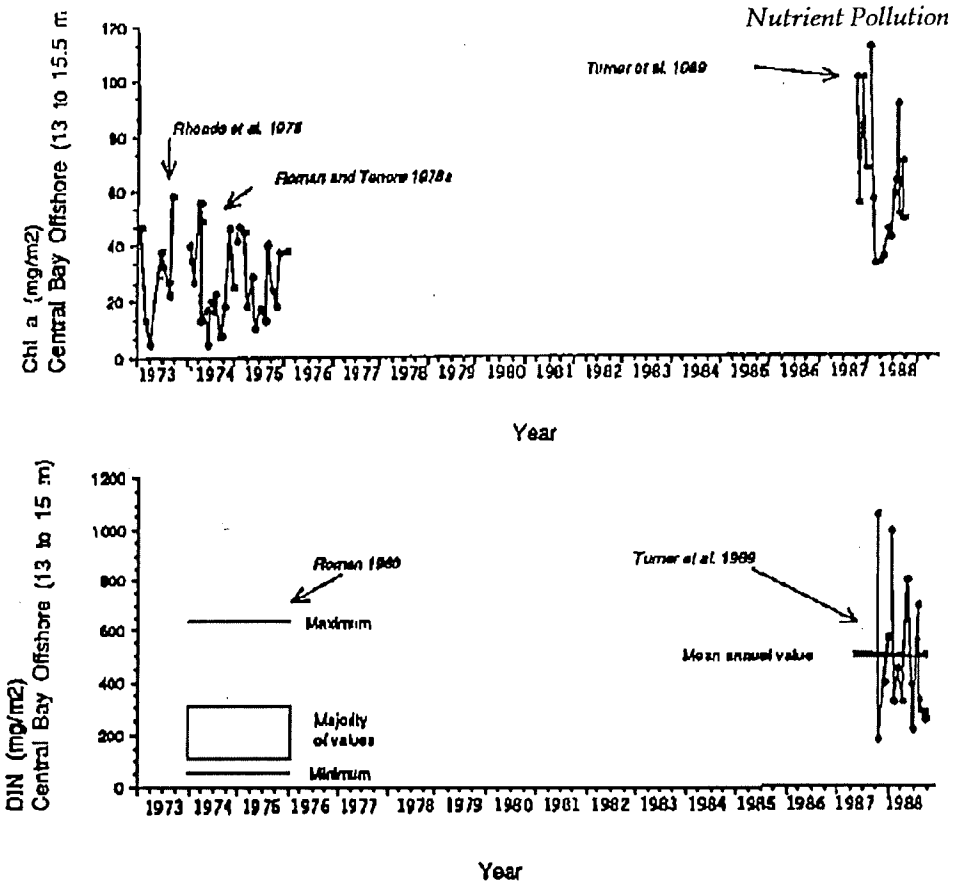

Figure 22. Historical comparison of chlorophyll and DIN concentrations in open waters of central Buzzards Bay. Data sources indicated; see text for position of stations.

Buzzards Bay in Comparison to Other Coastal Areas

Nutrient Inputs

If the two neighboring southern New England bays, Narragansett and Buzzards, received fresh water with equal loads of nitrogen (i.e., differences in human populations and land use did not result in different effective input concentrations), the input, unless weighted for water turnover, would be over 3 times higher in Narragansett Bay. In fact, however, the land-derived nitrogen loading to Narragansett Bay may be greater by a factor of 4 to 16 (i.e., 2^2 to 2^4, Figure 23), reflecting its more populated watershed. Buzzards Bay as a whole has lower input rates than its northeastern end subestuary, Buttermilk Bay (Valiela and Costa, 1988), and than Massachusetts Bay (NOAA, 1988). Indeed, Buzzards Bay, using the range discussed earlier, appears less loaded than most of the five dozen or so coastal areas in the Northern Hemisphere for which one can find published input values of nitrogen and phosphorus. New Bedford Harbor, with values of perhaps 1300 to 2000 mmol m^{-2} y^{-1} (see earlier), would be about in the midrange of coastal areas surveyed, but still falls only in the same class as the whole of Narragansett Bay (Figure 23). Some ecosystems may

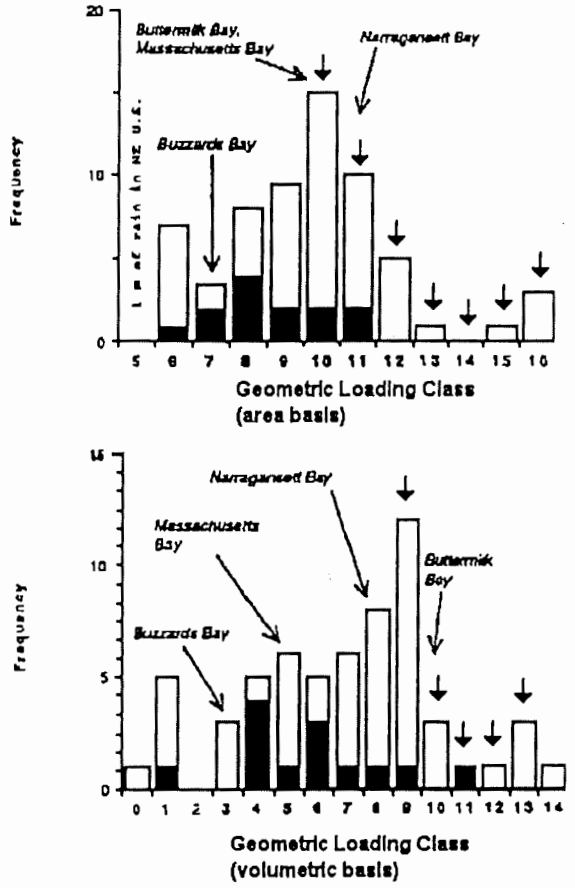

Figure 23: Geometric N loading class frequency distribution for 62 coastal lagoons, estuaries, bays, bights, and seas. Modified from Kelly (1990). Range for loading to Buzzards Bay is discussed in text. Buttermilk Bay is from Valiela and Costa (1988), Massachusetts Bay is from NOAA (1988), Narragansett Bay is from Nixon and Pilson (1983, 1984). A geometric class, x, is an interval range, defined as greater than 2x-1, but less than or equal to 2x. For example, class 7 is 26 and 27, 65 to 128 mmols N m-2 (or m-3) y-1. An uncertainty of about 1 loading class is expected at the mid range of the distribution. Arrows indicate classes represented by MERL microcosm gradient experiment (Nixon et al. 1986). The dark bar histogram is the frequency distribution of loading classes for which annual benthic fluxes in the field (n=13) also have been measured. For reference, the inorganic nitrogen content in about one meter of rain input per year to a flat surface would provide a loading represented by about class 5.

have particularly rapid flushing rates (like Buttermilk Bay). A water-turnover-weighted picture (Valiela and Costa, 1988) could give a slightly different picture than Figure 23, by bringing a few of the fast-flushing, high-load sites towards low-load, low-flushing sites. Nevertheless, Buzzards Bay would still be seen as among those receiving low nutrient inputs from land sources.

With a few exceptions, nutrient loading estimates for other coastal ecosystems do not include, explicitly, groundwater and offshore exchange as inputs. Even neglecting these potential inputs, a majority of coastal waters receive tremendous loads of nutrients, sometimes exceeding intensively fertilized agricultural fields, intensive ocean upwellings, and highly eutrophic freshwater lakes (Figure 24). The nitrogen loading range for the whole of Buzzards Bay is perhaps more typical of an oligotrophic lake than it is of the 'average' coastal area (Figure 24). The estimate of the Bay's phosphorus input, being much higher than to those same oligotrophic lakes, gives Buzzards Bay a very low N:P of inputs, and as discussed, reinforces the image of nitrogen as the more limiting nutrient.

In Situ Conditions: Nutrients and Chlorophyll

The central axis of Buzzards Bay has chlorophyll levels similar to midregions of many coastal systems. The two New Bedford sites are at the higher end of the range summarized for 39 coastal estuarine ecosystems (Boynton et al., 1982), and they indicate that algal biomass responds to increased nutrients (Figure 21).

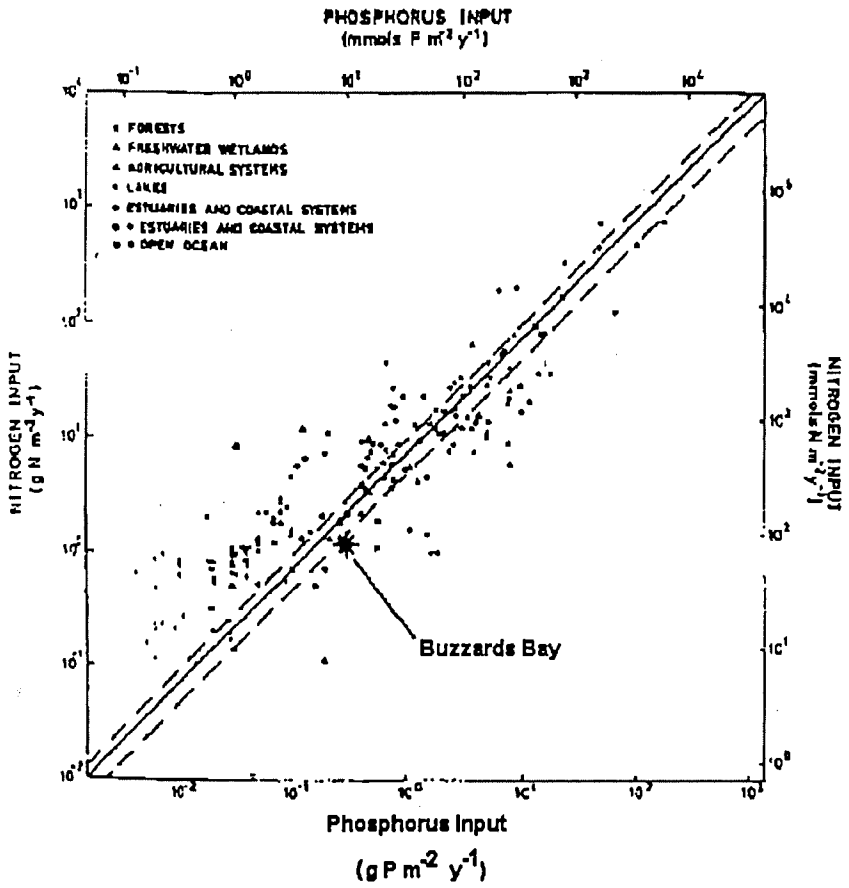

Figure 24. Nitrogen inputs vs. phosphorus inputs to aquatic and terrestrial ecosystems. Modified from Kelly and Levin (1986). Point estimate of input N/P for Buzzards Bay is based on NOAA (1988), modified for precipitation; also indicated is the range for N loading to Buzzards Bay as discussed in the text. Solid line shows Redfield ratio (16N : 1P), dotted line above is for 20N : 1P), dotted line below is for 10 N: 1 P.

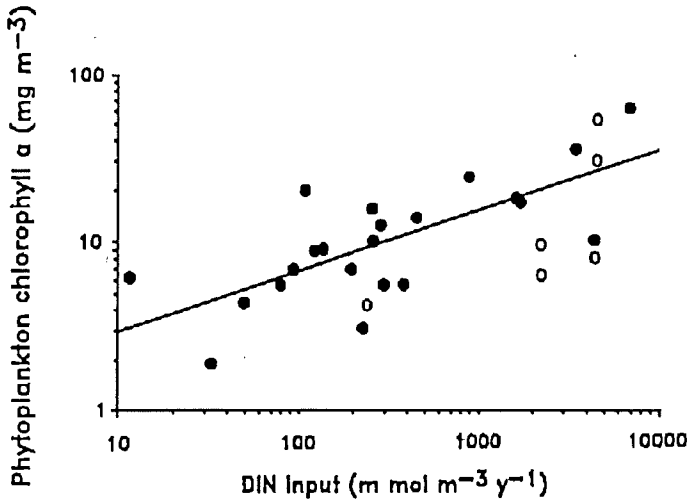

Figure 25. Chlorophyll a vs. nitrogen loading to coastal ecosystems. Modified from Nixon and Pilson (1983). Buzzards Bay is based on data of Turner et al. (1989) and loading range discussed in this report. Note that data from 1970's (Fig 21.) showed lower chlorophyll levels than depicted in this figure.

The relationship between nutrient inputs and chlorophyll varies from system to system, due to physical and biological factors. Annual mean values for many coastal areas lie within a band of about 5 to 10 Chl a mg m-3, which includes Buzzards Bay. This range is found in ecosystems receiving a wide range of nutrient loading (Figure 25). Maximum chlorophyll values measured recently in Buzzards Bay do not approach those encountered in more enriched systems, but it is intriguing that mean values similar to Buzzards Bay can be found in much more highly loaded areas. It could suggest very efficient nutrient recycling within Buzzards Bay or less efficient herbivore grazing of phytoplankton, or it could arise because the Bay is quite clear compared to many others and light is not limiting algal growth.

Primary Production and Nutrient Recycling Processes

Primary production reported for the central Bay (Table 5) is very low compared to many other shallow coastal areas (a range of 165 to 925 is given for systems summarized by Nixon et al., 1986). New Bedford Harbor sites have high primary production rates, but this area is small, and the Baywide estimate is largely set by the open-water rate (Table 5). Production estimates fall within the scatter seen for other aquatic ecosystems in reference to nutrient loading (Figure 26), although we caution that values indicated for any ecosystem, Buzzards Bay and New Bedford Harbor areas included, have large uncertainty.

Open-water rates may have increased since measured in the 1970s (see Figure 21) and rates in nearshore waters may also be higher. Therefore, the Table 5 Baywide estimate is most likely conservative; for illustration we will assume a plankton production of 150 g C m^{-2} y^{-1}.

Production of 150 g C m^{-2} y^{-1} implies the need for about 1900 mmol N m^{-2} y^{-1}, assuming a Redfield ratio (Redfield et al., 1963). Nitrogen input of ~79 to 283 mmol N m^{-2} y^{-1} represents only 4 to 15% of this need. Thus, as typically found unless a coastal system is quite eutrophic (Nixon, 1981; Kelly, 1990), other nutrient sources are required. The implication is that the primary production is dependent on nutrients provided by recycling processes within the Bay or from sources not yet assessed.

Table 5: Primary production estimates for areas in Buzzards Bay

Area	Production (g C m^{-2} y^{-1})	Reference
Open water, 13 m	89 to 123	Roman and Tenore (1978, 1974 and 1975) (+50% by benthic diatoms?)
New Bedford area	~360 to 828	From Camp Dresser and McKee (1990), near present outfall and at Outer Harbor (1987/88)
Harbor area average	~408	""
Whole Bay*	~124	* Assumes water column average (106) of Roman and Tenore (1978) for 94% of the Bay, and 408 for area of New Bedford Harbor 3.43 x10^7 m^2 (Camp Dresser and McKee 1990).

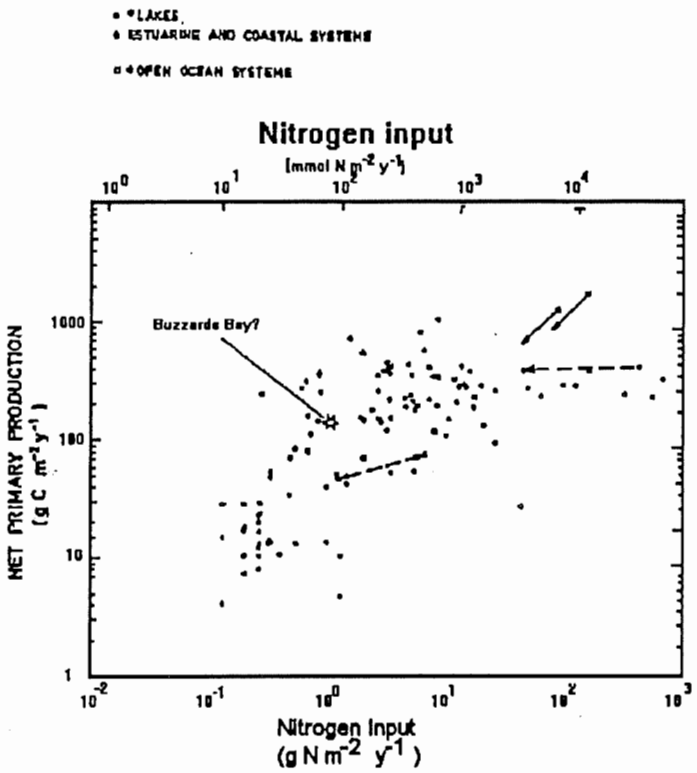

Figure 26. Time-weighted annual mean concentrations of pelagic chlorophyll 'a' and dissolved inorganic nitrogen (DIN) in the MERL mesocosms during the nutrient addition experiment. Measurements were made approximately weekly; analytical methods are described in Frithsen, Keller & Pilson (1985a).

Nutrient recycling occurs both in water ("pelagic") and in bottom sediments ("benthic"). To our knowledge, pelagic recycling has been little studied, but there has been longstanding emphasis on the benthos of Buzzards Bay.

Measurements (Rowe et al., 1975; Florek and Rowe, 1983; Christensen et al., 1983) indicate Buzzards Bay sediments may release 39 to 125 mol $m^{-2} h^{-1}$ of DIN during summer. Compared to the summer nitrogen assimilation needs of phytoplankton (Roman, 1980), this range of benthic flux could supply about 20 to 64% of the nitrogen needed for production, a normal range for a shallow water coastal site (Kelly, 1990). Banta and Giblin (in preparation) have made benthic flux measurements through an annual cycle, and these will add significantly to understanding of the internal nutrient dynamics in Buzzards Bay.

Benthic recycling and water column productivity are usually strongly coupled, although nonplankton organic inputs (sewage, vascular detritus, resuspended sediments) can alter the relationship (Nixon, 1981; Kelly and Nixon, 1984; Kelly, 1990). In general, sediment nutrient fluxes are rapidly reset (i.e. ~years or less) as a function of nutrient loading changes that affect production (Smith et al., 1981; Oviatt et al., 1984; Kelly et al., 1985). Sediment nutrient recycling therefore helps sustain production, but cannot be responsible for driving a system to a highly eutrophic state or to anoxia for a sustained period; such a state is more dictated by the water column and inputs to it.

Retention of Nutrients and Burial in the Sediments

One of the fates of nutrients that enter a coastal ecosystem is burial in bottom sediments. Nutrients in the sediments are part of a bay's historical legacy, but it is not always easy to read these traces in reconstructing the past.

Particulate nitrogen and phosphorus falls to the bottom of the bay, where decomposition of the organic matter takes place and most of the nutrients are regenerated to the water column. Only a small fraction of the organic matter becomes buried in the sediments over time. For example, sediments of both Narragansett Bay and Chesapeake Bay appear to retain on a long-term basis only a few percent of the nitrogen and phosphorus that enter them (1.7% of N, 2.6% of P in Narragansett Bay; 3 to 5% of N, 11 to 16% of P in the Chesapeake — Nixon, 1987, 1988).

We can only approximate nutrient retention in sediments of Buzzards Bay. Assuming Benninger's[12] discharge rates per unit of basin apply as a maximum to the Buzzards Bay drainage basin, one can calculate that Buzzards Bay receives, from the watershed, about 3.26×10^{10} g solids y^{-1}, or about 55 g m^{-2} y^{-1} of the Bay. If spread evenly to the sediments of the Bay, this would amount to less than 0.07 mm of sediment deposition per year as a Baywide average.[13]

The silt-clay depositional regions of the Bay are fairly stable (cf. Moore, 1963 vs. Hough, 1940) and are primarily found along a central axis in deeper water, biased towards the southerly border defined by the Elizabeth Islands. Rhoads (1973) gives the silt-clay bottom typical of the central axis of Buzzards Bay as 160 km^2 (about 28% of the Bay area). Assuming 28% is the area of accretion that collects solids, about 203 g m^{-2} is retained in the "depositional area" per year, equal to about 0.24 mm y^{-1}. This estimate would not include any fine-grained transport into the Bay with tides. Fine sediments in Buzzards bay contain about 0.15 to 0.25% nitrogen.[14] In comparison, three stations across central Long Island Sound had background levels of 0.1%, ~0.15%, and ~0.2% N (Rosenfeld, 1977). In the mid to upper Chesapeake Bay, sediments seem to be about 0.15 to 0.25%

12 Benninger (1978) calculated sediment load to Long Island Sound to be about 9×10^{11} g solids y^{-1} eroded from a drainage basin of 41,180 km^2. Benniger's solids/drainage area ratio is higher by about a factor of 2 than what Gordon (1979) estmated for all of New England. Santschi et al. (1984) found Benniger's rato estimated an input value only slightly higher than an independently-derived estimate of deposition in Narragansett Bay.

13 In comparison, deposition estimate for Naragansett Bay as a whole is 250 g m^{-2} y^{-1}, an average accumulation of about 0.3 mm y^{-1} (Santschi et al., 1984). Sediment input to Long Island Sound as a whole is about 157 g m^{-2} y^{-1}, (Benninger, 1978), and input to Chesapeake Bay as a whole is on the order of 260 to 300 g m^{-2} y^{-1} (Nixon, 1987). All these systems have larger watersheds and greater freshwater input than Buzzards Bay.

14 Driscoll (1975) gives sediment nitrogen at four stations in northwestern Buzzards Bay: about 0.026 to 0.147% N by weight. The deepest station, with the highest percent nitrogen, had finest sediments and may be most representative of an area of active deposition. Christensen et al. (1983) report sediment nitrogen content for an "organic rich" site in 17 m of water in Buzzards Bay; there was about 0.3% N at the surface, with deeper "background" (buried sediment below the bioturbation zone) values slightly about 0.2% N. Young (1971) reported nitrogen content (0 to 5 cm in sediment) in central Buzzards Bay as about 0.21 to 0.25% N.

N (Nixon, 1987). A value of 0.15 to 0.2% N is found as a "background" level in depositional areas of mid Narragansett Bay, with much lower values as one approaches Rhode Island Sound (Oviatt et al., 1984). Upper Narragansett Bay sediments near Providence and the major point source of nutrient input may have background nutrient levels approaching 0.4% (Oviatt et al., 1984).

Coarser sands in Buzzards Bay may have 0.025% N, and this may represent standing benthic biomass of microbes, although there are also benthic fauna, particularly suspension feeders and epifauna, common in this type of sediment (Rhoads, 1973). Over the long-term, it is unlikely this nitrogen is significantly retained and buried, for these are not sediments that are accumulating organic materials. Therefore, we calculate about 5.9 to 9.8 mmol N m^{-2} y^{-1} buried as a Baywide average, which represents a long-term retention of about 2 to 12% of the annual input, using the range of 79 to 283 mmol N m^{-2} y^{-1} for loading. This low percentage perhaps should not be surprising given the strength of tidal currents and the degree of offshore exchange possible.

The value we obtained for solids input is much lower than mud accrual rates obtained at specific sites (Table 6). It is unclear for some of dating methods whether the impact of surface bioturbation has been included; omission will yield an apparent sedimentation rate higher than realized (Livingston and Bowen, 1979; Santschi et al., 1980, 1984). Moreover, core-based, or site-specific, burial estimates can lead to overestimates because of errors of scale: these include inappropriate spatial extrapolations to a whole bay when the area of deposition represents only a fraction of the bay, or inappropriate temporal extrapolations where long-term rates of deposition (including flood or hurricane events) measured by tracers are compared to shorter term measures of nutrient input (Nixon, 1987).

This is related final calculation relative to New Bedford Harbor accumulation. We assume the post-1966 accumulation rate of 4 cm per year to the Inner Harbor (~3.42 km^2) (Table 6), and that sediment nitrogen content is about 0.5% N (Summerhayes et al., 1985). If we use the mean deposition rates of Summerhayes et al. (1985), this area alone would accumulate 4.84×10^{10} mmol N y^{-1}. Something is much amiss — the whole Bay receives perhaps 4.45 to 15.96×10^{10} mmol N y^{-1}. Could *at least* 30% and maybe *all* of the input come to permanent rest in this one small area? The vagaries of drift of fine muds, by which some areas of the Bay accumulate at the expense of erosion of others, must be playing a trick upon us in this case; but this most clearly highlights problems with deposition estimates and the difficulties of extrapolating from small areas to large bays without averaging over time and space.

Retention in sediments is difficult to assess, but the suggested implication of Buzzards Bay and other coastal bays as poor sinks for nutrients is profound. It suggests that eutrophication problems are more a function of recent nutrient inputs (on the order of years)rather than a consequence of a "burden" accumulated over decades or more of growing human cultural use (Smith et al., 1981; Oviatt et al., 1984; Kempe 1988).

Table 6. Deposition of solids in Buzzards Bay

Method	Station	Accumulation, y-1 (Deposition)	Reference, Notes
Radiocarbon dating	Station R, 20 m lower Bay 4129.2'N,7053.8'W	0.52 to 0.84 mm (~208 to 336 g $m^{-2}y^{-1}$)	Young (1971), note (1)
Rough calculation	Many cores; fine-grained sediment area	Porosity at depth ~1.5 mm Surface porosity ~2.3 mm	Hough (1940), note (2)
Summarized by Summerhayes et al. (1985)			Note (3)
Radionuclids	--	1mm	From Bowen et al. (1976)
^{210}Pb	--	2.95 mm	From Farrington et al. (1977)
Average assumption		3 mm	Summerhayes et al. (1985)
for Buzzards Bay		(1200 g $m^{-2}y^{-1}$ of clay) (2400 g $m^{-2}y^{-1}$ of silt)	Note (3)
Acushnet River Area			
^{210}Pb	New Bedford Harbor 8 feet of water	1.7 cm, top 17cm of core 2 mm, deeper in core	Summerhayes et al. (1985)
14C dating of shell layer at depth	17 feet of water	2.5 to 6.2 mm	Summerhayes et al. (1985)
Accumulation	area dredged in 1968-69	4 cm	Summerhayes et al. (1985)
in dredged areas	(Post-Hurricane Barrier Period Only)		
	area dredged in 1953	2.5 cm	Summerhayes et al. (1985)
Accumulation	Clark's Point		
of outfall sewage since 1920	3 cm from outfall		Summerhayes et al. (1985)
"mound" 5 to 6 ft high	3 mm within mi. of outfall		
Average assumption for Harbor area	Pre-hurricane barrier (constructed 1964-66)	1.5 cm (2500 g $m^{-2}y^{-1}$ of clay) (14,000 g $m^{-2}y^{-1}$ of silt)	Note (3)
	Post-hurricane barrier	4 cm (6900 g $m^{-2}y^{-1}$ of clay) (32,600 g $m^{-2}y^{-1}$ of silt)	
Input of solids from land			
	spread to whole Bay	~0.066 mm (55 g $m^{-2}y^{-1}$)	See text
	Accumulation in silt-clay sediments only (as 28% of whole Bay area)	~0.24 mm (203 g $m^{-2}y^{-1}$)	Note (4)

(1)"An estimate of the rate of deposition by radiocarbon dating (Geology Department, Yale University)" p. 561, p. 566; Station R has clayey silt to silt-sized sediments typical of the lower bay in 12 to 20 m; it is unlikely that this estimate considered the activity of biota in mixing surface sediments within the 5 to 20 cm upper layer, and thus would provide an overestimate (cf. Santschi et al., 1980;Livingston and Bowen, 1979). Deposition rate relative to 920 g m^{-2} y-1 were calculated for 2.3 mm y^{-1} rate of Hough (1940).

(2)Hough had noted layers, about 1 to 2 cm thick, and performed a calculation to see if these might be annual deposits. His calculation assumed original maximum bay depth (10,000 years before present) of 100 ft, where 50 ft of sediment then deposited to bring depth to observed current depth.

(3)We have not ascertained whether bioturbation generally has been taken into account by these estimates. Calculations for average values assumes 50% water of surface sediments, 35-40% water is appropriate for buried sediments below animal mixing zones, and the solids deposition would be higher if that correction were made.

(4) 28% estimate was derived from Rhoads (1973) as discussed in text. As also judged by maps of Hough (1940) and Moore (1963), the proportion of fine-grained accumulating sediments is of this order.

Perspective

The recent Buzzards Bay bibliography (Tripp, 1985) gives a historical clue to the scientific focus in the Bay. The number of citations grouped under the topic "benthos" (152) is about three times those for either "plankton" (49) or "fisheries" (53). An additional 87 or so citations relate to "circulation and sediment transport, tides," and tidal flushing/offshore exchange may have very strong influence on the biology and chemistry of Buzzards Bay. Scientifically, the open Bay commonly has been viewed not from the land seaward, nor from the water column down, but from the bottom up and from the offshore in (e.g. Sanders, 1960; Moore, 1963; Rhoads et al., 1975; Rowe et al., 1975; Roman and Tenore, 1978; Summerhayes et al., 1985). Not until recently has there been strong emphasis on nutrients from the perspective of "water quality" issues, nutrient budgets, and effects on the Bay from activity within its watershed.

Buzzards Bay is apparently dominated by low rates of primary production in the water, weak stratification because of the dynamic physical oceanographic setting, and biologically and geochemically active and tidally mixed near-bottom waters. The historical influence of land-originated loading of nutrients and solids on the Bay as a whole seems relatively small, and our rough calculations suggest the Bay exports most of its nutrient loading. Primary production in the bay is largely dependent on nutrients furnished by recycling of nutrients within the Bay.

The newest water quality data sets suggest Bay waters respond to enhanced nutrient inputs. The influence of the population around New Bedford is evident as higher nutrient concentrations and plankton biomass, and primary production; waters along the fringes of the Bay may be at similar risk. Much of the Bay does not seem greatly affected and has the appearance of a relatively oligotrophic ecosystem compared to many coastal areas.

The Bay, on the whole, is not excessively enriched now. Enrichment effects could easily increase, because many portions of Buzzards Bay still may lie on the steep part of the nutrient stimulation curve (Figure 26), and substantial changes in water quality, ecology, and living resources are possible even with small increases in nutrient loading. The open Bay, as defined in large measure by intense biological and physical interactions of its bottom with its surface water, could change slowly, and perhaps has (Figure 22). Whether deemed good or bad, these changes to the Bay as a whole would require large changes in the use of the watershed, and first would be seen in local situations. Thus, to produce an adequate survey of nutrients or other water quality parameters, well chosen sampling sites, schedules, and strategies are needed.

Recommendations

A fundamental decision is whether to (a) select a few simple key parameters measured at many stations and on many occasions or (b) select a greater suite of parameters at a few selected sites, for the luxury to do both is rarely granted. These choices offer clearly different advantages and limitations. The former choice allows better identification of the temporal and spatial pattern of

whatever is measured and, for example, could help to define volume- and area-weighted values that are critical to whole-bay budgeting. The latter choice can provide a better sense of the ecological character at the chosen site, is likely to allow more informed guesses as to the probable response to change, and is more often the scale at which public involvement happens. Where potential mitigation options are part of the debate, there is often a high priority for ascribing cause and effect, and this also tends to dictate the latter approach. To an extent, it matters most what one cares about in the bay, but careful selection of several areas to be routinely monitored seems to us the better choice given the patterns noted in this report for Buzzards Bay areas.

A strategy is needed in which we (a) first work out an objective way to establish priorities for sampling by estimating the degree to which water quality is threatened in each estuary or watershed of Buzzards Bay, (b) design appropriate sampling strategies for those estuaries or watersheds that merit high-priority status, and (c) suggest appropriate water sampling and analysis methods to reliably apply to the samples.

Determining Sampling Priorities

One way to set priorities on where to sample is to use geographical information systems (GIS) to describe the landscape use mosaic of each watershed of Buzzards Bay and, by applying methods of Frimpter et al. (1988), calculate the expected nutrient loading rates, given the development present, in each of the watersheds. These two steps will produce a hierarchy of watersheds, from the most to the least loaded unit of coastal landscape. We can then, with the participation of state and municipal officials, decide on how many of the top-ranked (on the basis of loading) watersheds to include in the monitoring schedule.

Preliminarily, one might, for example, choose five watersheds of the top ranked and one with a low ranked. The latter might be used for long-term monitoring in a nondeveloped watershed, to check whether factors in addition to development are changing our water quality.

In principle, there must also be a commitment to conduct a Baywide resurvey along the lines established by Turner et al. (1989) at appropriate time intervals. This need not be done with high frequency — probably once a decade is enough for monitoring purposes — but each instance should encompass a full seasonal cycle of measurements and mainain the physical relationship to previously surveyed stations.

Designing the Sampling Strategy

Once we have selected the estuaries or watersheds for monitoring, we must decide how to sample them. We need to consider down-estuary gradients, hydrography, sources of loading, seasons, and tidal state. These considerations will — after some planning and preliminary sampling to determine transect orientation, number of stations, number of samples per station, depth profiles, etc. — produce a sampling scheme. The scheme should be fairly uniform for each estuary, to make inter-estuary comparison easy, but may need slight alteration to suit local conditions. These modifications will require some reconnaissance and initial sampling.

Determining What Variables to Measure and How to Measure Them

Trying to obtain a simple index that can be used to denote the "health" of a bay is unrealistic (Kelly and Harwell, 1989). Even in the simple case of nutrients, concentration does not tell all. The best approach is to use suites of different measures to gain better understanding. Yet some measurements are considerably more valuable than others as indicators of water quality. To save time and funds, and to allow more intensive sampling of the more important variables, we ought to produce a list of the key variables to be measured.

Most of the basic choices for indicators of eutrophication are well known and have been used, if sparingly, in Buzzards Bay: dissolved oxygen concentration (and processes in water and sediments that affect it); water column nutrient concentrations (nitrogen, phosphorus, perhaps silicon); chlorophyll; phytoplankton species composition; distribution of submerged aquatic vegetation that are sensitive to turbidity and nutrients; concentrations of nutrients and other compounds as recorded in the sediment; and zooplankton and benthic species/biomass changes. Additionally, it would seem a rewarding exercise to carefully inspect the historical record, in the literature as well as in the sediments, of the Bay's living resources (plankton, macrophytes, benthos, fish and shellfish).

Once a list of variables has been agreed upon, we should work out the best methods that are (1) adequate for the range of samples and concentrations we are likely to encounter, and (2) feasible for a state agency to carry out.

The plan outlined above will ensure that state funds are spent in a way that will provide useful results. The plan (including the GIS, sampling, and methods) should be designed with mechanisms for updating it say every 5 years or so. It also should contain thresholds that will serve as alarm signals and incorporate mechanisms to interpret and convey such information to the appropriate decision makers.

Acknowledgements

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References

Benninger, L.K. 1978. ^{210}Pb balance in Long Island Sound. Geochim. Cosmochim. Acta 42:1165-1174.

Bowen, V.T., H.D. Livingston, and J.C. Burke. 1976. Distribution of transuranium nuclides in sediment and biota of the North Atlantic Ocean. Pp. 107-120 in Transuranium Nuclides in the Environment, Interat. Atom. Energy Agency, Vienna.

Boynton, W.R., W.M. Kemp, and C.W. Keefe. 1982. A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. Pp. 69-89 in V.S, Kennedy (ed.), Estuarine Comparisons, Academic Press, New York.

Camp Dresser and McKee. 1990. Phase 2 Facilities Plan. City of New Bedford, Massachusetts.

Caraco, N. 1986. Phosphorus, Carbon, and Iron Cycling in a Salt Stratified Coastal Pond. Ph.D. Thesis. Boston University, Boston, MA.

Christensen, J.P., G.T. Rowe, and C.H. Clifford. 1983. The possible importance of primary amino nitrogen in nitrogen regeneration by coastal marine sediments in Buzzards Bay, Massachusetts. Int. Revue ges. Hydrobiol. 68: 501-512.

Doering, P. H., C.A. Oviatt, L.L.Beatty, V.F. Banzon, R. Rice, S.P. Kelly, B.K. Sullivan and J.B. Frithsen. 1989. Structure and function in a model coastal ecosystem: silicon, the benthos and eutrophication. Mar. Ecol. Prog. Ser. 52: 287-299.

Driscoll, E.G. 1975. Sediment - animal - water interaction, Buzzards Bay, Massachusetts. J. Mar. Res. 33: 275-302.

Farrington, J.W., S.M. Henrichs, and R. Anderson. 1977. Fatty acids and Pb-210 geochronology of a sediment core from Buzzards Bay, Massachusetts. Geochim. Cosmochim. Acta 41: 289-296.

Florek, R.J. and G.T. Rowe. 1983. Oxygen consumption and dissolved inorganic nutrient production in marine coastal and shelf sediments of the Middle Atlantic Bight. Int. Revue ges. Hydrobiol. 68:73-112.

Frimpter, M.H., J.J. Donahue, and M.V. Rapacz. 1988. A mass balance nitrate model for predicting the effects of land use on groundwater quality in municipal wellhead protection areas. Cape Cod Aquifer Management Project.

Garside, C., G. Hull, and C.S. Yentsch. 1978. Coastal source waters and their role as a nitrogen source for primary production in an estuary in Maine. Pp. 565-575 in M.L. Wiley (ed.), Estuarine Interactions, Academic Press: New York.

Gil, L.W. 1987. Buzzards Bay 1986 Water Quality Survey Data: Part A. Massachusetts Department of Environmental Quality Engineering, Division of Water Pollution Control. Publication #14,712-158-75-2-87-CR. 147 pp.

Gil, L.W. 1988. Buzzards Bay 1986 Water Quality Survey Data: Part B. Massachusetts Department of Environmental Quality Engineering, Division of Water Pollution Control, Westborough, MA. Publication #15,462-116-75-5-88-CR. 104 pp.

Goldman, J.C. and M.R. Dennett. 1983. Nitrogen-plant dynamics. Pp. 39-76 in J.M. Teal (ed.) The Coastal Impact of Groundwater Discharge: an Assessment of Anthropogenic Nutrient Loading in Town Cove, Orleans, Ma. Unpub. Rep. Woods Hole Oceanographic Institution, Woods Hole, MA.

Gordon, R.B. 1979. Denudation rate of central New England determined from estuarine sedimentation. Amer. J. Sci. 279: 632-642.

Hough, J.L. 1940. Sediments of Buzzards Bay, Massachusetts. J. Sedim. Petrol. 10: 19-32.

Jaworski, N.A. 1981. Sources of nutrients and the scale of eutrophication problems in estuaries. Pp. 83-110 in B.J. Neilson and L.E. Cronin (eds.) Estuaries and Nutrients, Humana Press, Clifton, NJ.

Kelly, J.R. 1990. Paradigms of benthic-pelagic coupling of carbon and nutrients in coastal ecosystems as derived from annual field studies. ERC-124b, Ecosystems Research Center, Cornell University, Ithaca, NY 14853. Unpub. Rep. 26^{+} pp.

Kelly, J.R. and S.W. Nixon. 1984. Experimental studies on the effect of organic deposition on the metabolism of a coastal marine bottom community. Mar. Ecol. Prog. Ser. 17: 157-169.

Kelly, J.R. and S.A. Levin. 1986. A comparison of aquatic and terrestrial nutrient cycling and production processes in natural ecosystems, with reference to ecological concepts of relevance to some waste disposal issues. Pp. 165-203 in G. Kullenberg (ed.), The Role of the Oceans as a Waste Disposal Option, NATO Advanced Research Workshop Series, D. Reidel Publishing Co., Dordrecht, Holland.

Kelly, J.R. and M.A. Harwell. 1989. Indicators of ecosystem response and recovery. Pp. 9-35 in S.A. Levin, M.A. Harwell, J.R. Kelly, and K.D. Kimball (eds.), Ecotoxicology: Problems and Approaches, Springer-Verlag, New York.

Kelly, J.R., V.B. Berounsky, C.A. Oviatt, and S.W. Nixon. 1985. Benthic-pelagic coupling and nutrient cycling across an experimental eutrophication gradient. Mar. Ecol. Prog. Ser. 26: 207-219.

Kempe, S. 1988. Estuaries — their natural and anthropogenic changes. Chapter 13, Pp. 252-285 in T. Rosswall, R.G. Woodmansee, and P.G. Risser (eds.), Scales and Global Change, SCOPE, John Wiley and Sons Ltd., Chichester, England.

Livingston, H.D. and V.T. Bowen. 1979. Pu and ^{137}Cs in coastal sediments. Earth Planet. Sci. Letters 43: 29-45.

Meade, R.H. 1965. Landward transport of bottom sediments in estuaries of the Atlantic Coastal Plain. J. Sedim. Petrol. 39: 222.

Moore, J.R., III. 1963. Bottom sediment studies, Buzzards Bay, Massachusetts. J. Sedim. Petrol. 33: 511-558.

NADP. 1988. National Atmospheric Deposition Program. NADP/NTN Annual Data Summary. Precipitation Chemistry in the United States. 1987. Natural Resource Ecology Laboratory, Colorado State University, Fort Collins, CO. 353 pp.

NOAA. 1988. National Oceanic and Atmospheric Administration. Strategic assessment of near coastal waters, northeast case study. Chapter 3: Susceptibility and concentration status of northeast estuaries to nutrient discharges. Strategic Assessment Branch, National Oceanic and Atmospheric Administration; Office of Marine and Estuarine Protection and Environmental Results Branch, U.S. Environmental Protection Agency; and College of Marine Studies, University of Delaware. Rockville, MD: NOAA, 49 pp.

Nixon, S.W. 1981. Remineralization and nutrient cycling in coastal marine ecosystems. Pp. 111-138 in B.J. Neilson and L.E. Cronin (eds.), Estuaries and Nutrients, Humana Press, Clifton, NJ.

Nixon, S.W. 1986. Nutrient dynamics and the productivity of marine coastal waters. Pp. 97-115 in R. Halwagy, D. Clayton, and M. Behbehani (eds.), Marine Environment and Pollution, The Alden Press, Oxford.

Nixon, S.W. 1987. Chesapeake Bay nutrient budgets — a reassessment. Biogeochemistry 4: 77-90.

Nixon, S.W. 1988. Physical energy inputs and the comparative ecology of lake and marine ecosystems. Limnol. Oceanogr. 33: 1005-1025.

Nixon, S.W. and M.E.Q. Pilson. 1983. Nitrogen in estuarine and coastal marine ecosystems. Pp. 565-648 in E.J. Carpenter and D.G. Capone (eds.), Nitrogen in the Marine Environment, Academic Press, New York.

Nixon, S.W. and M.E.Q. Pilson. 1984. Estuarine total system metabolism and organic exchange calculated from nutrient ratios: an example from Narragansett Bay. Pp. 261-290 in V.S. Kennedy (ed.), The Estuary as a Filter, Academic Press, Orlando, FL.

Nixon, S.W., J.R. Kelly, B.N. Furnas, C.A. Oviatt, and S.S. Hale. 1980. Phosphorus regeneration and the metabolism of coastal marine bottom communities. Pp. 219-242 in K.R. Tenore and B.C. Coull (eds.), Marine Benthic Dynamics, University of South Carolina Press, Columbia.

Nixon, S.W., C.A. Oviatt, J. Frithsen, and B. Sullivan. 1986. Nutrients and the productivity of estuarine and coastal marine ecosystems. J. Limol. Soc. South Afr. 12: 43-71.

Oviatt, C.A., M.E.Q. Pilson, S.W. Nixon, J.B. Frithsen, D.T. Rudnick, J.R. Kelly, J.F. Grassle, and J.P Grassle. 1984. Recovery of a polluted estuarine system: A mesocosm experiment. Mar. Ecol. Prog. Ser. 16: 203-216.

Pilson, M.E.Q. 1985. On the residence time of water in Narragansett Bay. Estuaries 8:2-14.

Redfield, A.C., B.H. Ketchum, and F.A. Richards. 1963. The influence of organisms on the composition of seawater. Pp. 26-77 in M.N. Hill (ed.), The Sea, Volume 2, The Composition of Sea-Water, Interscience Publishers, John Wiley and Sons, New York.

Rhoads, D.C. 1973. The influence of deposit-feeding benthos on water turbidity and nutrient recycling. Amer. J. Sci. 273: 1-22.

Rhoads, D.C., K. Tenore, and M. Browne. 1975. The role of resuspended bottom mud in nutrient cycles of shallow embayments. Pp. 563-579 in L.E. Cronin (ed.) Estuarine Research, Volume 1: Chemistry, biology, and the estuarine system, Academic Press: New York.

Riley, J.P. and R. Chester. 1971. Introduction to Marine Chemistry. Academic Press.

Roman, M.R. 1978. Tidal resuspension in Buzzards Bay, Massachusetts. II. Seasonal changes in the size distribution of chlorophyll, particle concentration, carbon and nitrogen in resuspended particulate matter. Est. Coast. Mar. Sci. 6: 47-53.

Roman, M.R. 1980. Tidal resuspension in Buzzards Bay, Massachusetts. III. Seasonal cycles of nitrogen and carbon:nitrogen ratios in the seston and zooplankton. Est. Coast. Mar. Sci. 11: 9-16.

Roman, M.R. and K.R. Tenore. 1978. Tidal resuspension in Buzzards Bay, Massachusetts. I. Seasonal changes in the resuspension of organic carbon and chlorophyll a. Est. Coast. Mar. Sci. 6: 37-46.

Rosenfeld, J.K. 1977. Nitrogen Diagenesis in Nearshore Anoxic Sediments. Ph.D. dissertation, Yale University, New Haven. 191 pp.

Rowe, G.T., C.H. Clifford, K.L. Smith, Jr., and P.L. Hamilton. 1975. Benthic nutrient regeneration and its coupling to primary productivity in coastal waters. Nature 255: 215-217.

Sanders, H.L. 1960. Benthic studies in Buzzards Bay III. The structure of the soft-bottom community. Limnol. Oceanogr. 5: 138-153.

Santschi, P.H., Y-H. Li, J.J. Bell, R.M. Trier, and K. Kawtaluk. 1980. Pu in coastal marine environments. Earth Planet. Sci. Letters 51: 248-265.

Santschi, P.H., S.W. Nixon, M.E.Q. Pilson, and C. Hunt. 1984. Accumulation of sediments, trace metals (Pb, Cu) and total hydrocarbons in Narragansett Bay, Rhode Island. Est. Coast. Shelf Sci. 19:427-449.

Signell, R.P. 1987. Tide- and Wind-Forced Currents in Buzzards Bay, Massachusetts. WHOI-870-15. Woods Hole Oceanographic Institution, Woods Hole, MA. March. 86 pp.

Smith, S.V., W.J. Kimmerer, E.A. Laws, R.E. Brock, and T.W. Walsh. 1981. Kaneohe Bay sewage diversion experiment: perspectives on ecosystem responses to nutritional perturbation. Pacific Sci. 35:279-395.

Stenner, T.W., T.C. Gulbransen, and E.M. Baptiste. 1988. Water Quality Data Assessment for Buzzards Bay, Mass. Final Report to Mass. Exec. Office Env. Affairs. Battelle Ocean Sciences, Duxbury, MA. 69 pp.

Summerhayes, C.P., J.P. Ellis, and P. Stoffers. 1985. Estuaries as sinks for sediment and industrial waste — a case history from the Massachusetts coast. Contributions to Sedimentology (eds. H. Fuchtbauer, A.P. Lisitzyn, J.D. Milliman, and E. Seibold) 14:1-47.

Sumner, F.B., R.C. Osburn, L.J. Cole, B.M. Davis. 1913. A biological survey of the waters of Woods Hole and vicinity. Bull. U.S. Bureau of Fisheries, Part 1, 31. 544 pp.

Tripp, B. 1985. Buzzards Bay Bibliography, A Reference Collection of Scientific and Technical Reports Published on Buzzards Bay. Technical Report WHOI-85-27, CRC-85-1 Coastal Research Center, Woods Hole, MA. July. 96 pp.

Turner, J.T., D.G. Borkman, W. Lima, and R.W. Pierce. 1989. A Seasonal Study of Plankton, Larval Fish, and Water Quality in Buzzards Bay, Massachusetts. Interim Data Report. Massachusetts Division of Water Pollution Control Research and

Demonstration Project 87-15. May. Prepared for Technical Services Branch, Division of Water Pollution Control, Westborough, MA. Pub. #15988-250-50-6-89-CR. 248 pp.

Valiela, I. 1983. Nitrogen in salt marsh ecosystems. Pp. 649-678 in E.J. Carpenter and D.G. Capone (eds.), Nitrogen in the Marine Environment, Academic Press, New York.

Valiela, I. 1984. Marine Ecological Processes. Springer-Verlag, New York. 546 pp.

Valiela, I. and J.M. Teal. 1979. The nitrogen budget of a salt marsh ecosystem. Nature 280: 652-656.

Valiela, I. and J.E. Costa. 1988. Eutrophication of Buttermilk Bay, a Cape Cod coastal embayment: concentrations of nutrients and watershed nutrient budgets. Environ. Manage. 12: 539-553.

Valiela, I., J. M. Teal, S. Vokmann, D. Shafer, and E.J. Carpenter. 1978. Nutrient and particulate exchanges in a salt marsh ecosystem: Tidal exchanges and inputs by precipitation and groundwater. Limnol. Oceanogr. 25: 187-192.

Weiss, R.F. 1970. The solubility of nitrogen, oxygen and argon in water and seawater. Deep-Sea Res. 17:721-735.

Young, D.K. 1971. Effects of infauna on the sediment and seston of a subtidal environment. Vie et Milieu (Suppl) 22, Vol II: 557-571.

Chapter 3

Toxic Chemicals in Buzzards Bay: Sources, Fates, and Effects

by John W. Farrington[1] and Judith McDowell Capuzzo[1]

Introduction to Toxic Chemicals of Environmental Concern

Buzzards Bay receives a wide range of chemical contaminants (also called toxic chemicals or chemical pollutants) from society's wastes. These contaminants come from industrial and municipal discharges, dredged material, atmospheric fallout, riverine inputs, and other nonpoint sources of pollution. The distribution, fate, and effects of chemical contaminants in coastal marine environments are governed by natural biogeochemical processes that influence the persistence of the contaminant and its availability to living organisms. Accumulation of contaminants in biological resources may occur through aqueous, dietary, or sedimentary pathways. In the long term, chemical contaminants of biological concern, such as metals and organic compounds, are associated primarily with particulate matter. In coastal areas transport of contaminants bound to particles coincides with sediment transport; thus, there are numerous locations around the world where sediment deposits in coastal areas reflect waste-disposal histories. Transfer of contaminants to marine biota and humans and disturbance of ecological systems are dependent on the availability and persistence of contaminants within sediments and transport within benthic ecosystems.

Environmental concern about contaminant input to coastal waters is focused on

- The accumulation and transfer of metals and organic contaminants in marine food chains, including accumulation in commercial resources and potential impacts on human health;
- The toxic effects of such contaminants on the survival and reproduction of marine organisms and the resulting impact on marine ecosystems.

Chemicals of environmental concern are those that have known or potentially deleterious effects on natural resource populations and on humans. Modern societies introduce chemicals to the environment via many processes and under several conditions. These include production, use, and disposal of synthetic chemicals (e.g., pesticides); mobilization of chemicals that exist naturally in the environment (e.g., mining); transport and use of metals; and transport, production and use of petroleum (accompanied by occasional oil spills). Many chemicals did not exist prior to synthesis and production by modern societies. These are often referred to as "xenobiotic" compounds. Examples are chlorinated pesticides such as DDT and chlordane and the common dry cleaning and degreasing solvents carbon tetrachloride and tetrachloroethylene. Some chemicals, such as xylenes and phenols, are present in very low concentrations in

[1] *Woods Hole Oceanographic Institution, Woods Hole, Massachusetts, 02543*

many aquatic environments; but relatively large amounts of other chemicals are produced, distributed and subsequently released to the environment by modern society. Other chemicals of environmental concern are those created as by-products of human activities (e.g., chlorinated compounds produced as a result of the chlorination of sewage effluents).

The total number of organic chemicals synthesized easily exceeds 100,000 of which 60,000 are in common use. Since 1978 approximately 1,000 have been added to this total each year (CEQ, 1984). It is important to remember that the synthesis and use of these chemicals was, and is, intended to provide benefits to modern societies in products such as pharmaceuticals, pesticides, and plastics. The vast majority of these chemicals are not threats to human health or to natural resource populations. However, it is estimated that on a worldwide basis approximately 1,000 chemicals are of environmental concern (Butler, 1978) because they are produced in amounts that could cause environmental concern if released indiscriminantly. Two characteristics that influence the effect of a chemical on natural resource populations and humans are persistence and mobility (Miller, 1984).

Persistence of a chemical in the environment is caused by particular properties of the chemical that prevent its easy chemical or biological degradation into more basic component chemicals found naturally in the environment. An example is the conversion of a synthetic organic chemical such as DDT to carbon dioxide and water. Several chemicals of environmental concern were designed to persist in the environment in order to be effective. The chlorinated hydrocarbon pesticides such as DDT needed to be persistent to provide effective and economical site-specific pest control over long periods of time to eradicate insects that carry human diseases or to protect valuable crops. Polychlorinated biphenyls (PCBs) needed to resist alteration during use in electrical components in order to maintain their desirable dielectric properties.

Unfortunately, both groups of compounds, plus others, are not only persistent but also mobile when released to the environment. During the late 1960s and early 1970s, these compounds were detected worldwide. They were spread around the world by atmospheric, riverine, groundwater, and oceanic transport processes. In most areas, the concentrations of these compounds were very low, but widespread distribution has made them available for uptake by many natural populations. Accumulation of these chemicals in various components of selected world ecosystems, and in a few cases in people, resulted in concentrations high enough to have unintended adverse effects.

Considerable progress was made in the 1970s and 1980s in understanding which chemicals are of concern because of their possible effects on human health and on valuable living natural resources (e.g., shellfish, fish, birds, and marine mammals). We have now identified how these chemicals enter and move through the environment, and the types of adverse impacts that they might have on various biological systems. Using polynuclear aromatic hydrocarbons (PAHs) as an example of toxic contaminants, we present a diagram of the biogeochemical cycle — i.e., how the chemicals enter and move through the environment and the Four Rs of a biogeochemical cycle: Routes, Rates, Reactions, Reservoirs — in Figure 1. Examples of chemical structures of PAHs are

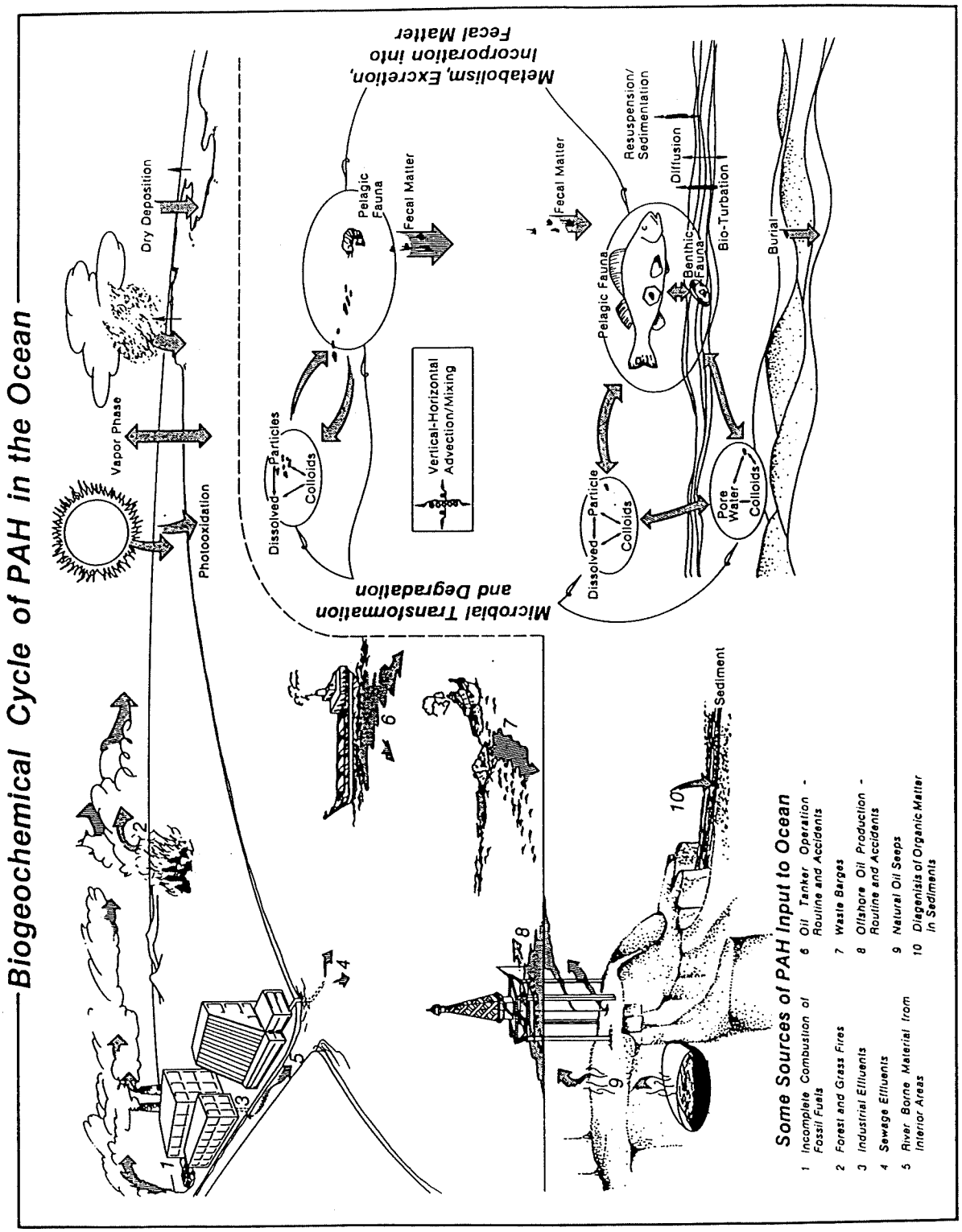


Figure 1. Biogeochemical cycle of polynuclear aromatic hydrocarbons in coastal ecosystems (from McElroy et al., 1989; Farrington, 1989).

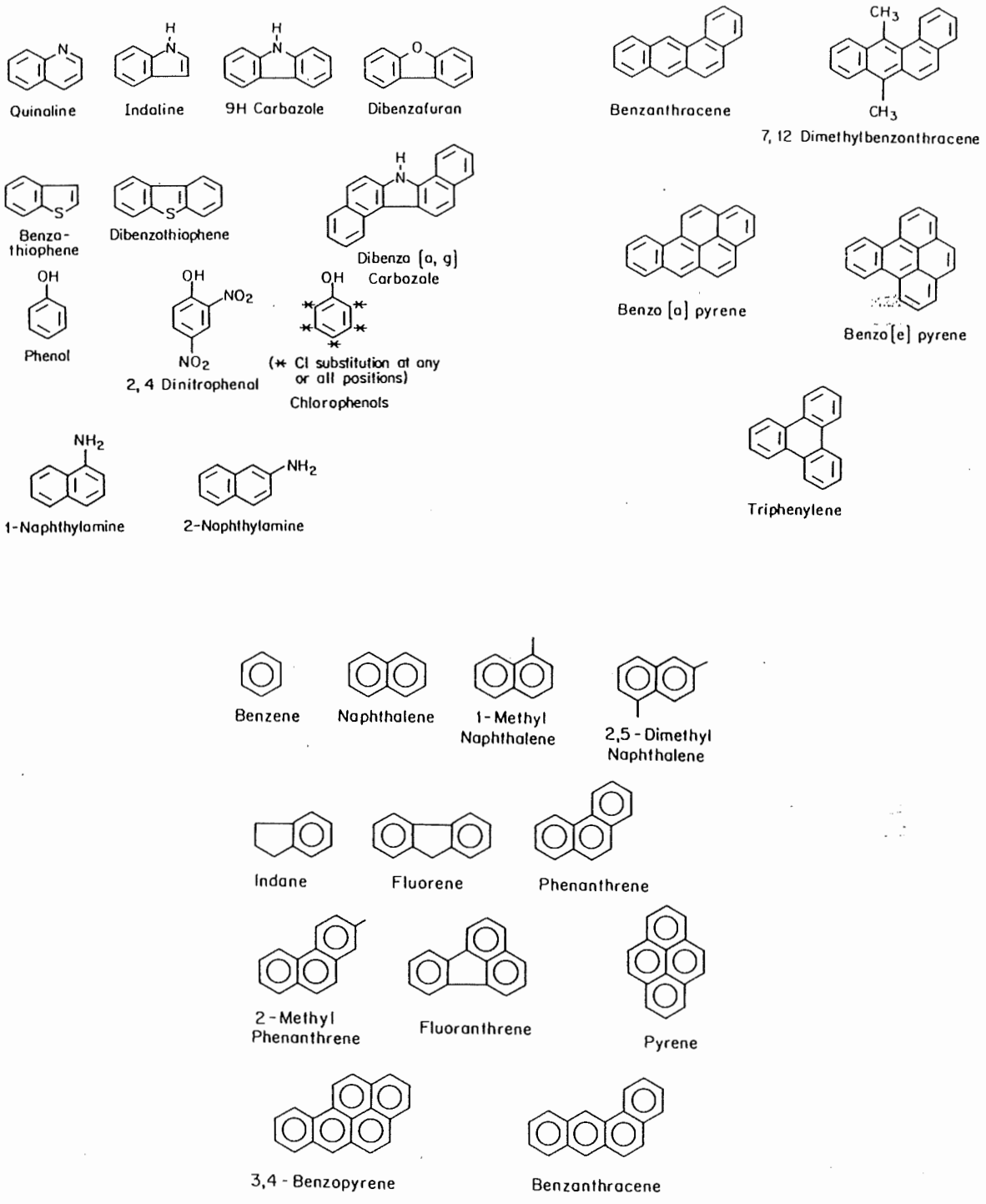


Figure 2. Examples of chemical structures of polynuclear aromatic hydrocarbons and other polynuclear aromatic compounds.

given in Figure 2. Representative chemical structures of other organic chemical contaminants that will be discussed in this report are also given in Figure 2. The types of adverse impacts that can be caused by chemical contaminants are outlined in Table 1.

Several examples of human health problems associated with chemical contaminants were identified in the 1950s and 1960s and alerted us to potential future increases in problems if the patterns and practices of chemical loss continued unchanged. Well-studied examples include Minimatta disease associated with eating coastal fish polluted with mercury in Minimatta Bay, Japan, and Yusho disease associated with the consumption of PCB-contaminated rice oil used for cooking (Goldberg, 1976). The unexpected and inadvertent damage resulting from the use of the persistent chlorinated pesticides such as DDT and other pesticides was eloquently predicted in Rachel Carson's book *Silent Spring* (Carson, 1962). The resulting scientific investigation and debate brought about the reduced use and subsequent banning of several of the more persistent chemical contaminants such as DDT and PCBs in the United States by the mid 1970s. Concerns about adverse effects also resulted in more careful practices for

Table 1. Response levels of Marine Organisms to Chemical Contaminants.

Level	Types of Responses	Effects at Next Level
Biochemical-Cellular	Toxication	Toxic metabolites
	Metabolic impairment	Disruption in energetics and cellular processes
	Cellular damage	
	Detoxication	Adaptation
Organismal	Physiological changes	Reduction in population performance
	Behavioral changes	
	Susceptibility to disease	
	Reproductive effort	
	Larval viability	
	Adjustment in rate functions	Regulation and adaptation of populations
	Immune responses	
Population	Age/Size structure	Effects on species productivity and coexisting species and community
	Recruitment	
	Mortality	
	Biomass	
	Adjustment of reproductive output and other demographic characteristics	Adaptation of population
Community	Species abundance	Replacement by more adaptive competitors
	Species distribution	Reduced secondary production
	Biomass	Ecosystem adaptation
	Trophic interactions	No change in community structure and function

chemical use and disposal as new national and state legislation and regulations were implemented.

Despite increased understanding of the environmental risks associated with chemicals and the resulting regulatory responses, there is no room for complacency. Much of our knowledge is qualitative and we need more detailed quantitative information if we are to make effective decisions on the utilization and protection of our coastal and estuarine resources. Although we generally understand the fate of chemical contaminant discharges on global and regional scales, this understanding is not always specific enough to be used in decision-making at the local scale. We must also be prepared for surprises. For example, it is common practice to chlorinate sewage effluent for disinfection prior to discharge to coastal waters. The chlorination has the beneficial intent of reducing human-disease vectors to protect bathing beaches and harvestable seafood resources. However, the chlorination process also produces many chlorinated organic compounds because the chlorine reacts with organic chemicals in the sewage effluent. The exact chemical structures of most of these compounds and their potential for long-term adverse effects are unknown. We must be prepared for as yet unknown problems from among these chemicals.

We must also keep a wary eye on the future while we deal with the problems of the present. Unfortunately, past practices have left a legacy of locations where chemical contaminants were discharged or disposed in such a manner as to present an environmental threat long after the discharges ceased. Such a situation exists today in the western portion of Buzzards Bay where sources such as industrial discharges, combined sewer overflows, and municipal sewage treatment plants have contaminated the New Bedford Harbor area with PCBs, trace metals, and other chemicals.

Given the severe pollution problems in New Bedford Harbor and adjacent portions of western Buzzards Bay, we might ask how the sediments and biota became so contaminated and what is the prognosis for the future. As illustrated in Figure 1, there are many sources of PAHs to coastal waters and each (with the exception of oil spills) involves entry of fossil fuel hydrocarbons into the environment in a reasonably dilute form. If concentrations of compounds in discharges and receiving waters are in the renge of parts per million (ppm) to parts per billion (ppb), then why are there problems with accumulation of these chemicals in sediment and biota? Later sections of this report provide an overview of inputs and fates of chemical contaminants in Buzzards Bay and discuss how it was possible for dilute inputs of chemical contaminants to become concentrated to high levels in some portions of the Buzzards Bay ecosystem.

More importantly, why do we care about such low concentrations in the environment? This question is discussed in detail in the report section titled Biological Effects of Chemical Contaminants but a simple analogy may illustrate the point. The amount of active ingredients in a single aspirin is sufficient to attain concentration of 1 ppm if evenly distributed in the body of the average adult. Other pharmaceuticals are equally or more potent in that 1 ppm may have a pronounced effect on our bodies.

General Description of Buzzards Bay

Buzzards Bay is an important estuary of the Massachusetts coastline — important for both the economic and aesthetic resources it offers. The economic resources of the Bay range from the harvest of its fisheries to its use as a transit route for shipping traffic through the Cape Cod Canal and for the New Bedford offshore fishing fleet. Its aesthetic resources include the recreational opportunities it offers in bathing beaches, boating, and recreational fishing. It also offers educational and research opportunities to the research laboratories centered in Woods Hole and the academic institutions (Massachusetts Maritime Academy and University of Massachusetts at Dartmouth) that are located on its shoreline. In recent years, Buzzards Bay has become the subject of a multidisciplinary study of coastal and estuarine dynamics that will add significantly to our knowledge of nearshore processes. The combination of these factors necessitates the development of a management plan for Buzzards Bay that will ensure the rational allocation of the Bay's resources for future generations.

Renowned since colonial times for its whaling and fishing industries, Buzzards Bay is now an estuary in transition. Its western shore has been one of the most troubled economic regions in Massachusetts over the last several decades. Its eastern shore is faced with widespread coastal development. The legacy of industrial pollution from the western shore and the accelerated housing development on the eastern shore combine to threaten the environmental and economic health of Buzzards Bay and typify the multiple-use impacts on many estuaries of the northeastern United States. The wise management of Buzzards Bay requires an increasingly sophisticated knowledge of bay wide processes and an improved understanding of the effect of land-use practices on water quality.

Evaluation of the fate and effects of toxic contaminants of environmental concern in the marine environment requires an understanding of (l) the temporal and spatial distribution of contaminants; (2) the partitioning of contaminants to different compartments of the ecosystem (e.g., sediments and biota), including assessment of contaminant bioavailability; and (3) the level of damage imposed by accumulation of contaminants in biotic resources. Such an evaluation requires the development of a risk assessment or characterization model that couples an understanding of contaminant distribution in the environment with an understanding of the mechanisms of toxic action and the transfer of contaminants to the human consumer. A conceptual model for describing ecological and human health risks must successfully relate contaminant distribution and bioavailability to the probability and magnitude of biological impact. The ultimate fate of chemical contaminants, the location and duration of exposure of marine biota to the contaminants, and the transport of contaminants back to people via food is strongly influenced by many factors, including the way the water circulates and mixes in the Bay. Thus, it is important to briefly review the physical nature of the Bay.

The geomorphology of the Bay is a product of geological and climatic forces acting over hundreds to thousands of years to produce the three-dimensional shape with which we are familiar from reading maps or charts (Figure 3).

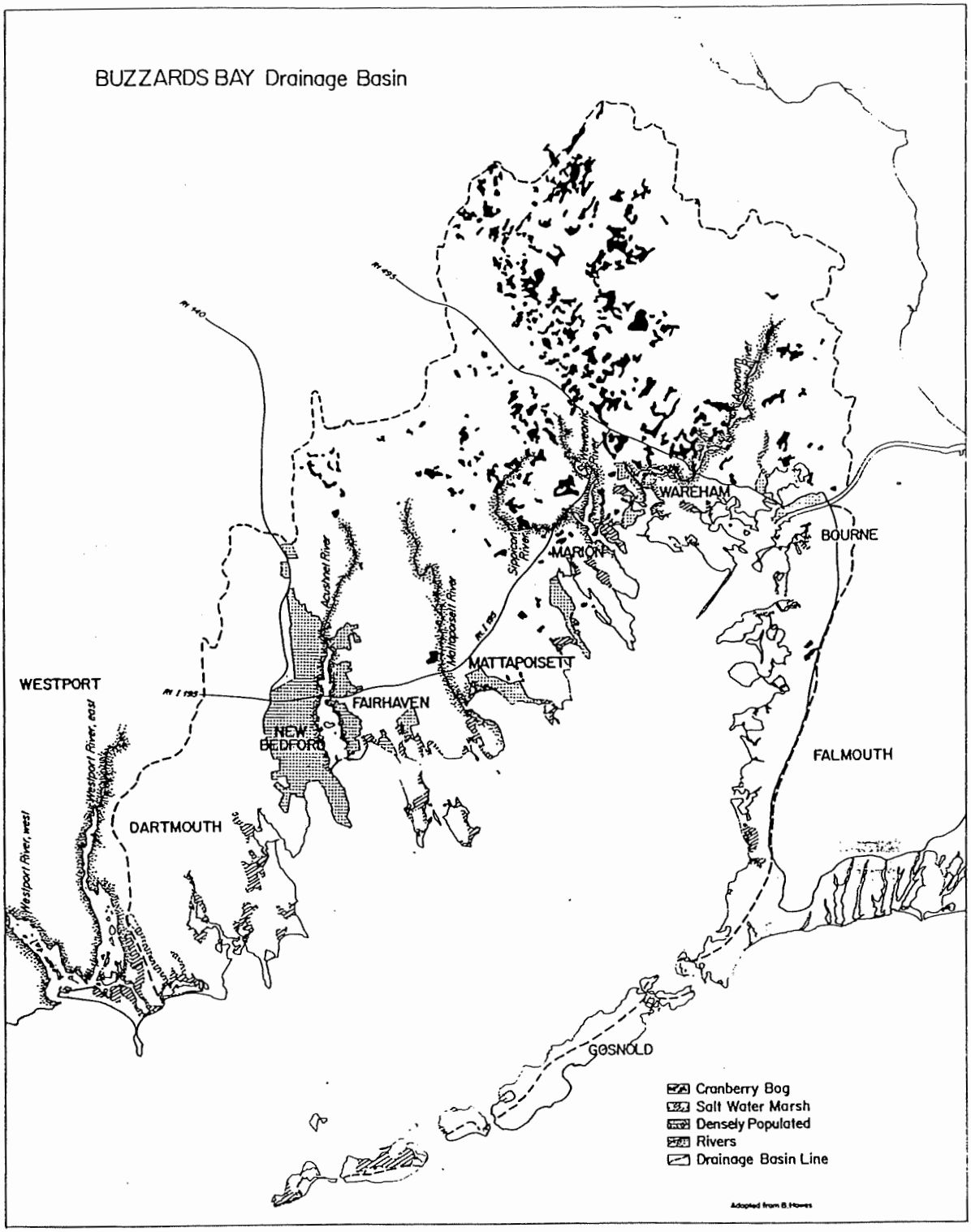


Figure 3. Buzzards Bay: Drainage basin and rivers. [Editor's note: This drainage basin is different than that adopted by the Buzzards Bay Project and the Commonwealth of Massachusetts.]

Advancing and retreating glaciers have shaped the basement rocks that were deposited as sediments in preglacial times — over a million years ago in the late Tertiary and Early Pleistocene periods of the earth's history. This basement rock is covered with a layer of sediment that has been contributed from nearby land via streams, rivers, and stormwater runoff from the drainage area of the Bay. This deposition of sediment began during postglacial times and continues to modern times.

The Bay is approximately 50 km long and 15 km wide, with an average depth of 11 m. It has a drainage basin of approximately 800 km^2 (Figure 3) and a water surface area of about 550 km^2. The volume of water in the Bay is about 6 billion cubic meters or about 1.6 trillion gallons. The general distribution of surface sediment types (e.g., the more commonly known mud and sands) is depicted in Figure 4.

Buzzards Bay is classified as an estuary because of the admixture of fresh water and salt water that yields a salinity of approximately 31 to 33 salt per thousand parts of water. Ordinary seawater from the North Atlantic Ocean has a salinity of about 35 ppt and river water, or fresh water, has a salinity of zero. The salinity in the Bay can become as low as 20 ppt in the New Bedford Harbor area and there can be subtle but important differences in salinity in both the horizontal and vertical directions, throughout the Bay at certain times of the year. These subtle differences in salinity combined with temperature govern the density of the Bay. Density differences can, in turn, cause a layering of less dense water over more dense water. Such layering can have a dramatic effect on the mixing and transport of chemical contaminants.

The freshwater input to the Bay occurs mostly from the eastern end of the Bay and the New Bedford Harbor area. In Figure 3, the rivers and streams of consequence in terms of freshwater input are depicted. Data on water discharges are presented in Table 2. Generally, there is an average of 27 m^3/of freshwater input to the Bay. Actual addition of fresh water can vary significantly from place to place and from season to season. Examples of salinity gradients in the bay are presented in Figure 5. These profiles of salinity show the influence of freshwater input near New Bedford Harbor and from the small rivers at the eastern end of the Bay.

A master's degree thesis by Mr. Richard Signell (Signell, 1987) has recently provided a significant advance in knowledge about the physical oceanography (i.e., the water circulation and mixing) of the Bay. His work builds on earlier work to improve our knowledge of the physical transport and mixing processes that are important in understanding how dissolved chemical contaminants and chemical contaminants bound to particulate matter will be concentrated, diluted, or spread throughout the Bay. Continuing research to improve our knowledge in this regard is reported in the recent U.S. Geological Survey report by Butman et al. (1988).

Signell (1987) concluded, on the basis of the best available data, that the major factors influencing water circulation and mixing in the Bay were the winds blowing across the surface and tidally forced water movement. The direction and speed of the wind has a marked influence on water circulation. If the wind is blowing from the mouth of the Bay (southwest), it has a much greater impact

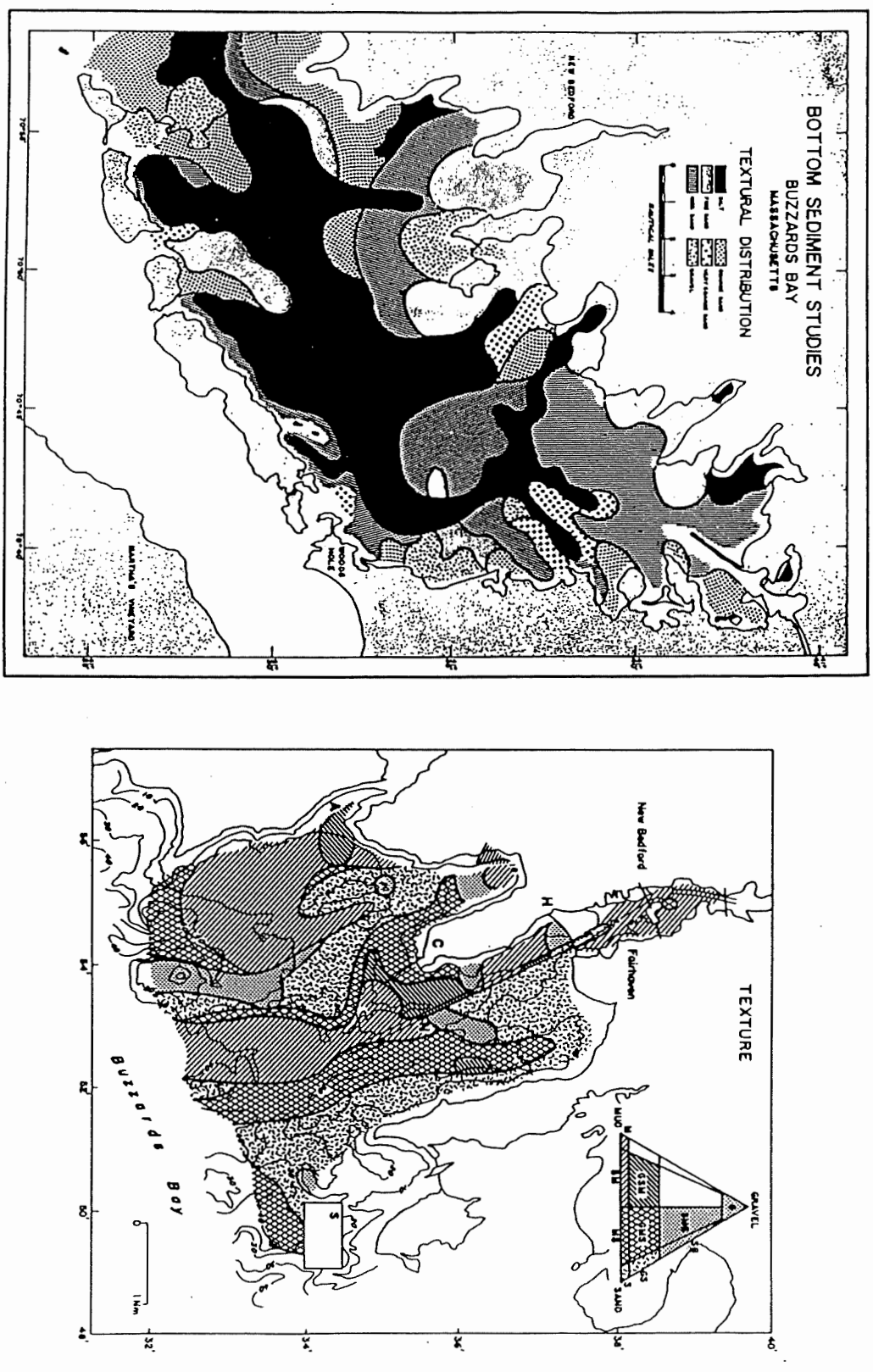


Figure 4. Chart of areal distribution of sediment types in Buzzards Bay (see compilation of data and figures in Summerhayes et al., 1977).

Table 2. River drainage areas and water flow; inferred annual average freshwater volume flux into Buzzards Bay (from Signell, 1987).

Rank	River	Drainage Area km^2	Inferred Inflow $m^3 s$	Percent Total %
1	Weweantic	145.3	2.9	18.6
2	Sippican	72.8	1.4	9.3
3	Paskamansett	67.6	1.3	8.7
4	Mattapoisett	62.2	1.2	8.0
5	Wankinko	53.1	1.1	6.8
6	Agawam	44.1	0.9	5.7
7	Acushnet	42.5	0.8	5.4
8	Red Brook	23.5	0.5	3.3
	Smaller Rivers and Groundwater	266.9	5.3	34.2
	Total	780.9	15.4	100.0

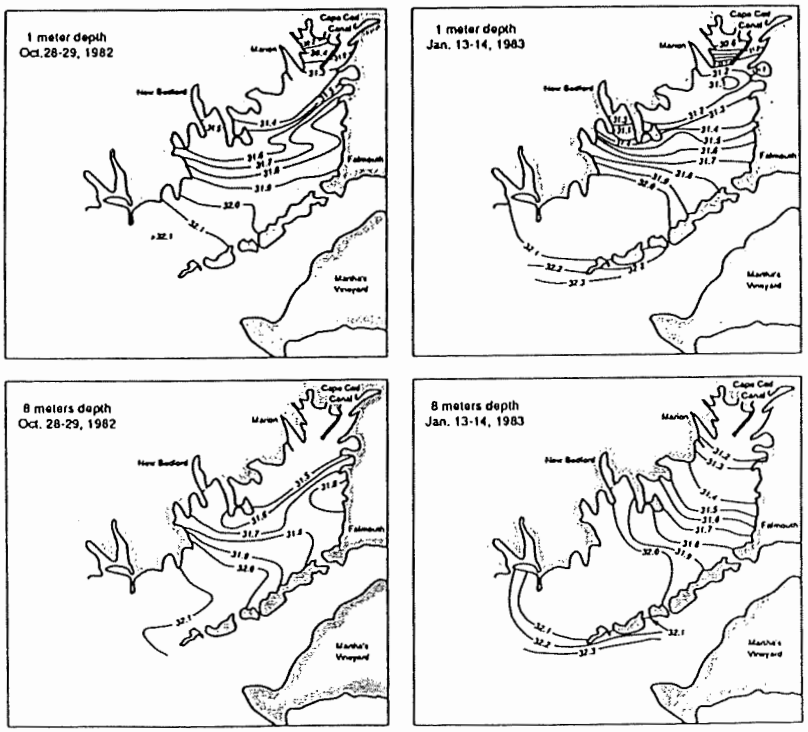


Figure 5. Examples of salinity gradients in Buzzards Bay (from Rosenfeld et al., 1984).

than if it is blowing from other directions. On the average, the waters of the Bay are well-mixed in the fall and winter; thus, it is difficult to detect any substantial gradients of temperature or dissolved constituents in the water (for example, those measured as salinity). Vertical layering (stratification) is apparent in the spring and summer when warmer, less dense and less salty water overlays colder, denser and saltier water.

Tidal currents in Buzzards Bay can be as fast as 10 to 50 cm/s (that is, up to 1 knot) in some sections of the Bay. It has been known for at least two decades (Rhoads, 1963, 1967) that tidal currents can resuspend the top few millimeters of fine-grained sediment even in the deepest portions of the bay. Since the fate of several chemical contaminants is governed by their sorption (binding) to fine grained sediments, the importance of this knowledge is apparent. How, when, and where water circulation and mixing in the near-bottom regions of the Bay resuspend and transport sediment has become an important focus of research.

Major storms can have an important impact on the movement of sediment because storms can be a major factor in mixing and resuspension. These underwater events are analogous to what happens above water on a beach where sands shift markedly from season to season and major storms can produce dramatic shifts. The ability of the forces of nature to move sediments, especially fine-grained muds such as those found in some of the harbor areas, is illustrated by the following example. Nautical charts of Buzzards Bay have areas indicated as "spoils areas." One such area is off West Island, and where anecdotal evidence suggests that dredged spoils (sands and muds dredged from an area) from New Bedford Harbor were dumped there during the 1950s and 1960s. Because these dredged spoils were from areas of the Harbor that contained significant proportions of fine-grained sediment (mud), it would be reasonable to expect that sampling at or near the dumping location would reveal the presence of these types of sediments. This is not the case, as pointed out by Summerhayes et al. (1977) and as observed by investigators from Woods Hole during sediment sampling for the Superfund investigation in August 1982. Summerhayes and his colleagues explain that intense tidal currents and occasional storm conditions resuspended the fine-grained sediments, allowing them to be resuspended and distributed elsewhere. In fact, if we examine the map of sediment distributions (Figure 4), it is clear that the central portions of the Bay or some of the sheltered harbor areas contain the fine-grained sediments. Despite some major technical differences, the fate of fine-grained sediments, (i.e., mud particles) in the Bay can be compared to the movement of snow during winter storms. Snow is swirled up from the ground in areas of high wind turbulence and high velocities and redeposited in more quiescent or protected areas.

The activities of modern civilization such as dredging channels, altering the shoreline with fill and sea walls, and building hurricane barriers like the one across New Bedford Harbor can have profound influences on the transport of sediments in and around the Bay. For example, the rate of sediment accumulation in New Bedford Harbor has doubled in some places as a result of the construction of the hurricane barrier (Summerhayes et al., 1977). We will return to the issue of sediments and sediment transport in a later section of the report

when we discuss the fate of chemical contaminants and their present distribution in the Bay in more detail.

Inputs of Chemical Pollutants to Buzzards Bay

Chemical contaminants enter Buzzards Bay by means of accidental discharges (e.g., oil spills), effluent discharges, river discharges, atmospheric transport and deposition to the waters of the Bay, or deposition to land and direct runoff to the Bay. There may also be some contaminant input along the shores of Buzzards Bay via groundwater exchange with the saline Bay water at the interface of the sediment and water. The latter source is speculative at this point, although some salinity profiles in the Bay (Rosenfeld et al., 1984) are consistent with some freshwater input via groundwater in the eastern part of the Bay off the Sippewissett-West Falmouth shore. It is also fairly common to observe freshwater wetland plants growing in seepage areas, immediately adjacent to the high-tide line.

Chemical contaminants associated with urban and industrial activities enter the Bay primarily in the western portion near the communities of New Bedford, Fairhaven, and Dartmouth. Chemicals associated with agricultural practice, such as pesticides used on cranberry bogs and farms, are more likely to enter the Bay from runoff, creeks, and small rivers near the communities of Westport, Dartmouth, Fairhaven, Mattapoisett, Marion, Wareham, Bourne, and Falmouth. Although we do not have exact input data for many chemicals, we can make some reasonable estimates by extrapolating from research conducted in Buzzards Bay and from data collected in similar coastal areas.

Trace Metal Inputs

A detailed study of many trace metals in the sediments of Buzzards Bay, particularly in the western portions including New Bedford Harbor, has been reported by Summerhayes et al. (1977), who also reviewed earlier work on the same subject. Samples were analyzed for trace metals such as copper, zinc, chromium, mercury, cadmium, nickel, lead, vanadium, arsenic, and aluminum. Many activities associated with modern society are potential sources that could increase concentrations of trace metals above background concentrations from natural sources. Metal plating and other industrial activities are common sources of input, as are municipal urban sewer inputs. Copper was a common constituent of marine antifouling paints during the 1950s and 1960s and is still present in some antifouling paints.

Summerhayes et al. (1977) concluded that "large quantities of potentially toxic trace metals have been discharged to the harbor [New Bedford Harbor] and its approaches." The metals included copper, chromium, lead, and zinc with lesser amounts of silver, arsenic, cadmium, and mercury. U.S. EPA records indicate that during at least part of the 1970s about 90 kg per day of copper (about 33 tons per year) were discharged to the New Bedford Harbor area (Summerhayes et al., 1977). The report on concentrations of trace metals in the New Bedford municipal sewer outfall prepared by Camp Dresser and McKee (CDM, 1979) indicates values of trace metals as listed in Table 3. These data are probably

reliable, but it behooves us to keep in mind the very difficult nature of any trace chemical analysis when confronted with the difficult problems of analyzing chemicals in a natural water matrix. Assuming that these data are reasonably close to the correct values, we can estimate some loading factors for New Bedford sewage effluent inputs to Buzzards Bay. If we consider copper as an example, then we have 0.12 to 0.31 mg/liter x 88 million liters per day x 365 days per year = about 3,850 to 9,960 kg per year or about 4 to 10 tons of copper per year. Similar calculations can be made for other trace metals (Table 3).

Many metals are emitted to the air in industrial countries such as the United States by a variety of activities associated with a modern industrial and urban society. These metals are mixed into the atmosphere and transported long distances of several hundred to several thousand kilometers. Along the route they are deposited to the surface of the earth by several processes including dry fallout (similar to dust deposits), rain and snow, and exchange of vapors for portions of some metals that are in the vapor phase. For example, it has been estimated by Bruland et al. (1974), Edginton and Robbins (1976), Thomason et al. (1975), and several other authors that the "fallout" from the atmosphere attributable to industrial releases is 0.5 micrograms (a millionth of a gram) of copper/cm^2/yr. We know that the drainage basin for the Buzzards Bay area is about 800 km^2 and we can calculate that 4000 kg/yr of copper is deposited to the land of the Buzzards Bay drainage basin. How much of this makes its way to the waters and muds and sands of Buzzards Bay via rivers, streams and land runoff is unknown at present. However, we can safely assume that no more than the total reaches the Bay. If we compare this number of 4,000 kg (or 4 tons)/yr with the amount entering via the New Bedford sewage effluent of 4 to 10 tons/year, then it is obvious that the sewage input is a very important input. The natural background flux of copper to Buzzards Bay has been estimated by Summerhayes et al. (1977) to be on the order of 2.9 micrograms/cm^2/year for natural clay deposited to the muddy sediment areas of the bay. Since these mud sediments occupy about 150 km^2 in central Buzzards Bay (Figure 4), we estimate

Table 3. Trace metal discharges from the New Bedford sewage treatment plant as Reported in the 301h Waiver Application Report (Camp Dresser and McKee,Inc., 1979); data from wastewater effluent in April and May of 1979.

Metal	Concentration (mg/liter)	Approximate Annual[a] Discharge (metric tons)
Cadmium	0.11-0.15	4-5
Copper	0.12 - 0.31	4-10
Lead	0.06 - 0.11	2-4
Selenium	0.03 - 0.056	1-2
Silver	0.009	0.3
Arsenic	0.03	1
Chromium	0.17 - 0.42	5-13
Mercury	0.0004 - 0.0026	0.01-0.08
Nickel	0.01 -0.20	.3-6
Zinc	0.016 -0.38	.512

[a] Calculated by the authors of this report using 88 million liters per day as the average sewage discharge. Note that this calculation is a rough estimate and also assumes that the trace metal concentration data are correct despite problems with similar types of data from measurements of trace metals in other similar studies.

that 4 tons of copper are deposited each year to the Bay as a result of natural clay mineral inputs.

Similar types of calculations can be carried out for other metals. Summerhayes et al. (1977) conducted a very thorough investigation that was state of the art at the time in 1975-1976. They focused primarily on measurements of metals in sediments since the sediments are a principal sink for trace metals, although as they state and we will discuss further in a later section of this report, the sediments are a leaky sink.

It is important to realize that these estimates are very rough estimates and many more careful measurements are required before we can be confident that we know the present inputs of these trace metals to Buzzards Bay from any input source. The measurements of trace metals to date in Buzzards Bay have been primarily aimed at obtaining total metal concentrations. Modern scientific research indicates that the trace metals associated with natural clay minerals may not be as easily taken up by marine organisms (i.e., bioavailable) as the trace metals discharged from the sewage effluent; the industrial trace metal component delivered via the atmosphere may be intermediate in terms of bioavailability. Furthermore, each of the trace metals has a unique chemical and biological reactivity and each has a varying degree of environmental concern as will be discussed later in this report.

There is ample evidence to show that the New Bedford Harbor area of Buzzards Bay, especially the inner Harbor area, has received substantial contaminant inputs of trace metals such as copper, nickel and zinc in the past. Dredging the Harbor and disposal of spoils in the main part of the Bay has certainly contributed contaminant trace metals to the Bay. Dissolved and particle associated trace metals leave the Harbor with exchanges of water between the Harbor and the Bay. Summerhayes et al. (1977) estimated that up to 24 percent of the metals discharged to the Harbor at the time of their study (1975-76) had been transported out of the Harbor to the Bay. They concluded that elevated concentrations of copper, lead and zinc detected in mud areas of the main part of the Bay (other than New Bedford Harbor) did not result from natural processes. The Harbor area itself, as we will discuss in more detail in the next section, has very high concentrations of copper, nickel and zinc in surface sediments.

Petroleum and Fossil Fuel Hydrocarbon Inputs

The general assessment of fossil hydrocarbon inputs for the world's oceans is presented in Table 4, taken from a National Academy of Sciences document titled *Oil in the Sea* (NAS, 1985). Accidental spills of oil garner much attention from the news media and the public, as demonstrated most recently by the spill from *Exxon Valdez* in Prince William Sound, Alaska. The occurrence of several oil spills during June 1989 — in the Houston ship channel, in Narragansett Bay, and the Delaware River — is a reminder that such accidents can happen in the marine environment. However, and this cannot be overemphasized, accidental oil spills account for a small percentage of the total input of fossil fuel hydrocarbons to the world's oceans.

Buzzards Bay is not an exception to this generalization. There are no inputs to Buzzards Bay for several of the categories listed in Table 4. There are no known

Table 4. Input of petroleum hydrocarbons into the marine environment (from NAS,1985)

	Units of Million Metric Tons per Year	
Source	**Probable Range**	**Best Estimate**
Natural Sources		
Marine Seeps	0.02 - 2.0	0.2
Sediment erosion	0.005 - 0.5	0.05
(Total natural sources)	(0.025 - 2.5)	(0.25)
Offshore Production	0.04 - 0.06	0.05
Transportation		
Tanker operations	0.4 - 1.5	0.7
Dry-docking	0.02 - 0.05	0.03
Marine terminals	0.01 - 0.03	0.02
Bilge and fuel oils	0.2 - 0.6	0.3
Tanker accidents	0.3 - 0.4	0.4
Nontanker accidents	0.02 - 0.04	0.02
(Total transportation)	(0.95 - 2.62)	(1.47)
Atmosphere	0.05 - 0.5	0.3
Municipal and industrial wastes and runoff		
Municipal wastes	0.4 - 1.5	0.7
Refineries	0.06 - 0.6	0.1
Nonrefining industrial wastes	0.1 - 0.3	0.2
Urban runoff	0.01 - 0.2	0.12
River runoff	0.01 - 0.5	0.04
Ocean dumping	0.005 - 0.02	0.02
(Total wastes and runoff)	(0.585 - 3.12)	(1.18)
TOTAL	**1.7 - 8.8**	**3.2**[a]

[a] Total best estimate, 3.2 metric tons per year is a sum of the individual best estimates.

inputs associated with offshore production, refineries, or ballasting practices of large tankers at this time or in the past. There are stories of tankers being torpedoed and sunk offshore of Buzzards Bay during World War II, but it is beyond our capability to assess the magnitude of the input from these spills. We lack accurate information on the cargo capacity and how much oil burned or went down with the ship. Certainly, the offshore ballasting practices of coastal tankers during the 1950s and 1960s might have resulted in the release of oil to the waters of Buzzards Bay and the washing ashore of tar balls. Oil tankers discharge residual petroleum when their tanks are cleaned and ballasted with seawater for the return voyage. This discharged material accumulates into thumbnail to fist-sized blobs of oil that drift around until they strand ashore or sink.

It is also likely that some tar balls found in the Sargasso Sea as a result of the tank cleaning and ballasting of large open-ocean oil tankers (NAS, 1985) have been entrained in the Gulf Stream and then mixed from the Gulf Stream into coastal waters off New England, eventually finding their way into Buzzards Bay. The amount of oil that enters the Bay from this source is difficult to estimate, but there are good reasons to assume that it is not large compared to other sources for fossil fuel hydrocarbon inputs.

Since the 1940s tanker and barge traffic through Buzzards Bay has been heavy because the industrial oil and gasoline from refineries in the South passes through the Cape Cod Canal and associated sea lanes to the Greater Boston market. Several tankers per year also arrive at the port of New Bedford and at the Cape Cod Canal Electric Power Plant. U.S. Coast Guard records for Buzzards Bay and nearby waters indicate that the total input of all petroleum and petroleum products from accidental spills was 261,000 gallons, or close to 1 million liters (1,000 metric tons), between 1973 and August 1989. Numerous small spills of less than 10 gallons each accounted for 1,200 gallons. There were 13 spills greater than 1,000 gallons each. Almost one-half of the total spill input was No. 2 fuel oil.

Baxter et al. (1978) reviewed the history of oil spills in Buzzards Bay up to 1978. Two spills of note were (1) a spill of No. 2 fuel oil in the late 1940s on Horseneck Beach in Westport; and (2) a spill of No. 2 fuel oil in the winter of 1963 off Cleveland Ledge, which came ashore at Nye's Neck, Falmouth. There is no record, however, of the amount of oil spilled, and no scientific studies of these spills have been published or recorded to our knowledge.

The barge *Florida* went aground in 1969 off West Falmouth and spilled 650,000 liters, or about 650 metric tons of No. 2 fuel oil into Buzzards Bay and along the shoreline of West Falmouth. This spill was intensely studied and will be discussed later. In October 1974 the barge *Bouchard 65* struck a submerged object at the west end of the Bay and was towed to an anchorage off Scraggy Neck at the east end of the Bay. Rough seas prevented containment of the leaking oil and some No. 2 fuel oil came ashore in North Falmouth and Bourne, principally in Red Brook Harbor, Bassett's Island, and Winsor Cove. The same barge ran aground again in January 1978 and spilled 307,000 liters, or about 307 metric tons, into ice covered waters at the east end of the Bay. Smaller spills have occurred every few years in the Bay or in the Cape Cod Canal.

No accurate measurements are available for the inputs of fossil fuel hydrocarbons from sewage effluent, stormwater runoff, and industrial effluents to Buzzards Bay. However, we can use per capita per year averages from other cities and extrapolate to the urban areas of Buzzards Bay. Hoffman et al, (1983) determined that about 0.969 kg of hydrocarbons per capita entered Narragansett Bay from the greater Providence area due to sloppy use and disposal of oil (e.g., crankcase-oil drippings on the road); products of partial combustion of fossil fuel that fell on the land and then were mobilized by rain and runoff to storm sewers; and effluents from sewage treatment plants. This value is close to the 0.875 kg of hydrocarbons per capita per year that Eganhouse and Kaplan (1981) determined for Los Angeles stormwater runoff.

The population of the City of New Bedford was 98,478 according to the 1980 census, and the population of the Buzzards Bay drainage area is about 260,000. Using the per capita hydrocarbon input previously quoted produces a total input in the range of 90 to 250 tons of fossil fuel hydrocarbons per year. As a rouch check on this estimate, we can multiply the flow of the New Bedford sewage treatment plant (approximately 24 million gallons per day, or 88 million liters) by the concentration, as measured for sewage treatment plants, of fossil fuel hydrocarbons in sewage effluent and stormwater runoff (1 to 100 mil-

ligrams per liter; Farrington and Quinn, 1973; NAS, 1985; Eganhouse and Kaplan, 1981; and Hoffman et al, 1983). This calculation produces an estimated annual input of 32 to 3,200 metric tons of fossil fuel hydrocarbons to the Bay from the New Bedford sewage treatment plant and compares favorably with the estimates calculated on a per capita basis. Over the 1969 to 1989 time period this would total almost 1,800 to 5,000 tons, a value within the range estimated from accidental spills.

Small boats with inboard diesels and outboard engines also dribble oil into the Bay. Although we cannot estimate the amount of oil from this source, we know that there are 4,300 slips and moorings in the Buzzards Bay area and approximately 20,000 vessels transit the Cape Cod Canal each year. Another source of fossil fuel hydrocarbons to the Bay is creosote for treatment of pilings. Creosote, a liquid created from washing of partially combusted coal and/or wood, has very high concentrations of PAH. Creosote-treated wooden pilings have been a common feature along the shores and in the harbors of Buzzards Bay for many years. Their use is now discouraged, but creosote pilings have not been eliminated entirely and creosote from previous use probably remains in the ecosystem.

In summary, in Buzzards Bay, as elsewhere in the world (Table 4), the chronic inputs of hydrocarbons — from the insidious, everyday dribbling of fossil fuels and partial combustion products — is equal to or greater than input from accidental spills.

Pesticides

During the 1950s and 1960s, chlorinated pesticides were undoubtedly in heavy use around Buzzards Bay. These pesticides were used to combat insects that could carry human disease. In the 1960s we realized that pesticides such as DDT and dieldrin tended to persist in the environment and could affect not only the targeted pests but also other organisms such as birds. RachelCarson's book *Silent Spring* (Carson, 1962) was instrumental in alerting the public to these dangers. The inceased awareness of the problems associated with use of the chlorinated pesticides led to a gradual reduction in their use and eventual ban on many of the most persistent forms. It also led to their replacement by a second and third generation of more sophisticated and less persistent chemicals such as the organophosphate-based pesticides (e.g., malathion, parathion, diazinon) and the carbamate pesticides (e.g., carbaryl, carbofuran). The designations "organophosphate" and "carbamate" refer to the basic chemical structure, or moiety, in these two groups of pesticides.

There are no data available that provide an accurate assessment over time of the total input of chlorinated hydrocarbon pesticides to Buzzards Bay. Data from a recently published report (NOAA, 1989a) can be used to estimate application of the more modern pesticides. Approximately 33,000 lb of pesticides are applied each year to crop lands in the drainage basin of Buzzards Bay. Cranberries account for the largest amount at almost 20,000 lb, and the rest is used primarily on crops such as feed corn, sweet corn, potatoes, and squash. It is unlikely that major amounts of these pesticides reach Buzzards Bay as a result of leaching from drainage areas or as a result of atmospheric transport and deposition during or after applications. Organophosphate and carbamate

pesticides have a more rapid chemical or biological degradation rate than the older more persistent chlorinated hydrocarbon pesticides. Thus, these modern pesticides will not persist for years and decades as have the chlorinated hydrocarbon pesticides and some of their immediate environmental degradation products (e.g., DDT and the degradation product DDE). Nevertheless, it is important to consider that improper doses or improper applications of the organophosphate and carbamate pesticides can result in adverse impacts over short periods of time (days to months) and may even be fatal to aquatic life in immediate drainage areas. A compilation of the toxicological properties for most of these pesticides is presented in NOAA (1989a); some pesticides have fairly high toxicities (LC_{50} values for 24-96 hours of one to hundreds of micrograms per liter), whereas others have very low toxicities (LC_{50} values greater than hundreds of milligrams per liter). The use of insecticides and herbicides on cranberry bogs has been implicated as the causative factor in several reported fish kills according to the Massachusetts Division of Marine Fisheries (DMF, 1985, as cited in Gil, 1988) but definitive data were not available for review.

The predominant use of pesticides on cranberry bogs leads naturally to questions about the extent of pesticide input from bogs to the Bay. Gil (1988) reports results of a 1986 special water quality study of the Massachusetts Department of Environmental Quality Engineering (now the Department of Environmental Protection) for Nye Bog and Garland Bog in the Buttermilk Bay freshwater drainage area. Sediments from the estuarine portion of the outlet stream from these bogs were analyzed by the Massachusetts Pesticide Analytical Laboratory at Amherst, MA, for four pesticides (chlorpyfiros, methyl parathion, dichlorobenil, and diazinon) and for several of their metabolites. The samples were also analyzed for the herbicide glyphosate and its primary metabolite. Traces of the pesticide chlorpyrifos and the herbicide glyphosate were recovered in the sediments of the outlet but not downstream in the estuary. According to the grower, the chlorpyrifos was last applied a year before the study was conducted (Gil, 1988). These results, although from a very limited study, are consistent with our present knowledge of the relative nonpersistence and localized environmental impacts of the organophosphate and carbamate pesticides.

The use of pesticides in urban and suburban residential areas is also of concern when we consider the dribbling of these pesticides into municipal waste sewers and storm sewers and the volume of water discharged from the New Bedford sewage treatment plant to Buzzards Bay. Unfortunately, reasonable estimates of these inputs are not available. There have been some analyses for the persistent chlorinated hydrocarbon pesticides in the sewer effluent as a result of the application of the City of New Bedford to EPA for a waiver for installation of secondary treatment (301h waiver application). The analyses were limited by the detection limits specified as acceptable by EPA at the time of the application (1978-1979). Only beta-endosulfan was detected above the detection limits of 1 microgram per liter (1 ppb); aldrin, dieldrin, DDT and common metabolites, alpha-endosulfan, endrin, heptachlor and heptachlor epoxide, lindane and other hexachlorocyclohexanes, toxaphene, and chlordane were not detected. For illustrative purposes only, if we use the beta-endosulfan concentration of 1 microgram per liter and multiply by the volume of wastewater discharge per

year from the plant (which has a daily volume of 88 million liters), we arrive at an annual input value of 32 kg (or 70 lb) of beta-endosulfan to Buzzards Bay. This value can be considered only a very rough estimate.

To our knowledge there are no data for use and inputs of the organophosphate and carbamate pesticides in urban areas of Buzzards Bay, nor is there a procedure by which a rough estimate can be made. We suspect, as noted above, that the relatively nonpersistent nature of these pesticides results in very little input to the Bay. Almost no data exist for homeowner's use for lawn and landscaping applications or in home gardens. Nor is there a good record of pesticide use or standard practices of lawn-care consultants. With some residential development in the flood zone, even to the high-tide line, input from these sources remains a concern.

Polychlorinated Biphenyls (PCBs)

PCBs are a class of chemicals that have been synthesized since the 1930s for a variety of uses. They have ideal or nearly ideal properties for use in electrical components because of their dielectric constant and their reduced flammability. The PCBs have been used in marine antifouling paints, in other paints, in ink formulations, as specialized lubricants, and in hydraulic fluids. The major use of PCBs has been in electrical and electronic components. These compounds were among the "miracle" chemicals of the past several decades. The relatively inexpensive transmission, distribution and storage of electrical power and the use of much electric and electronic equipment would not have been as readily possible without the use of PCBs. What we did not anticipate were the environmental consequences of these "miracle" chemicals.

The chemical structure of a biphenyl molecule and the chemical structure of a single polychlorinated biphenyl are given in Figure 6. There are 209 possible individual chlorobiphenyl molecules, which chemists call congeners (i.e., molecules like one another). Some examples of the congeners are given in

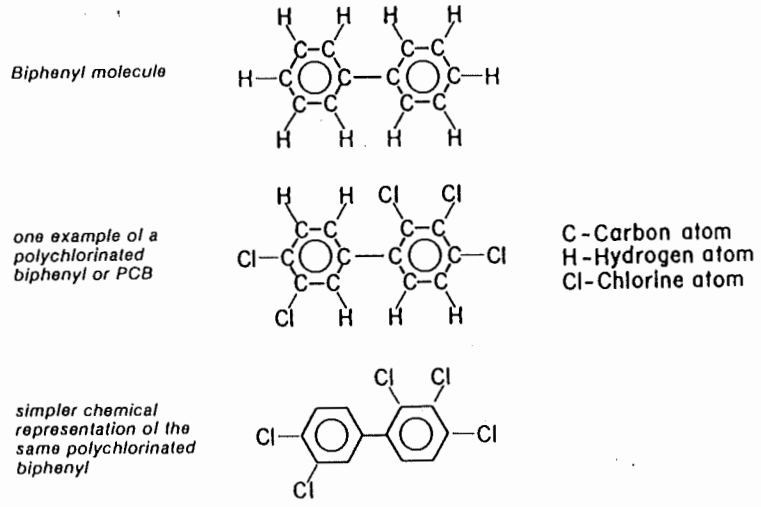
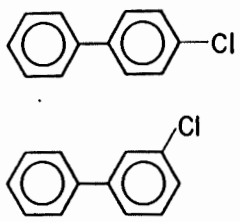
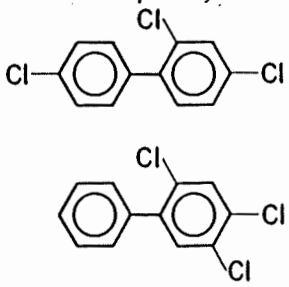


Figure 6. Examples of general chemical structures for polychlorinated biphenyls.

monochlorobiphenyls

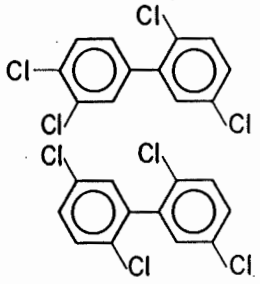


trichlorobiphenyls



2,4,5 trichlorobiphenyl

tetrachlorobiphenyls



dichlorobiphenyls

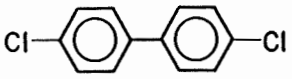
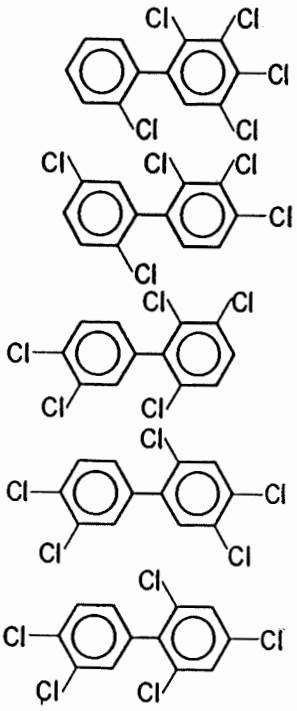
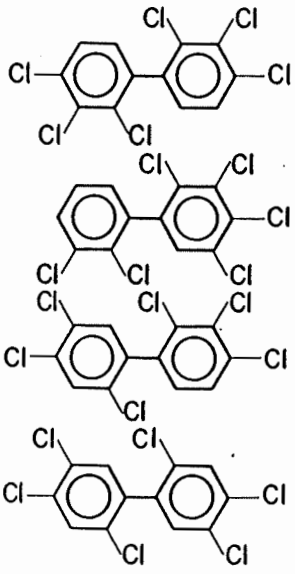


Figure 7. Examples of congeners of PCBs - chemical structures for individual chlorinated biphenyls in commercial mixtures of polychnorinated biphenyls.

pentachlorobiphenyls



hexachlorobiphenyls



hept-[7], octa-[8], nona-[9] chlorobiphenyls and the single decachlorobiphenyl are not as common in use or in the environment

Figure 8. Further examples of congeners of PCBs - chemical structures for individual chlorinated biphenyls in commercial mixtures of polychlorinated biphenyls.

Figures 7 and 8. Note that some molecules have the same number of chlorine atoms substituted on the biphenyl rings (although they are more like hexagons, chemists refer to them as "rings"). For example, trichlorobiphenyls each have three chlorines but the chlorines can be located at different positions or corners of the biphenyl rings. Those chlorobiphenyls with the same number of chlorine atoms are referred to as isomers. Thus, as a group the PCBs are polychlorinated biphenyls; in this scheme, all isomers are congeners but not all congeners are isomers.

The manufacturing of PCBs in commercial formulations yields mixtures of PCBs that have code numbers and a commercial designation. Monsanto Chemical the only U.S. company to manufacture PCBs, designated their PCBs by the trade name Aroclor, derived from the fact that the biphenyl rings are aromatic in chemical character ("Aro") and there are chlorines in the molecule ("clor"). Each commercial formulation was given a code number (e.g., 1242 or 1254), in which the first two digits represent the 12 carbon atoms of the biphenyl molecule and the last two digits represent the weight of chlorine in the formulation. Aroclor 1016 is the exception to the coding scheme. It was produced after the initial formulations and is a slightly reprocessed or changed version of Aroclor 1242. The literature on PCBs is replete with designations of commercial formulations such as Aroclor 1242, Aroclor 1254, and Aroclor 1016. In Europe, the Aroclor designation is replaced by "Clophen" and in Japan it is replaced by "Kaneclor."

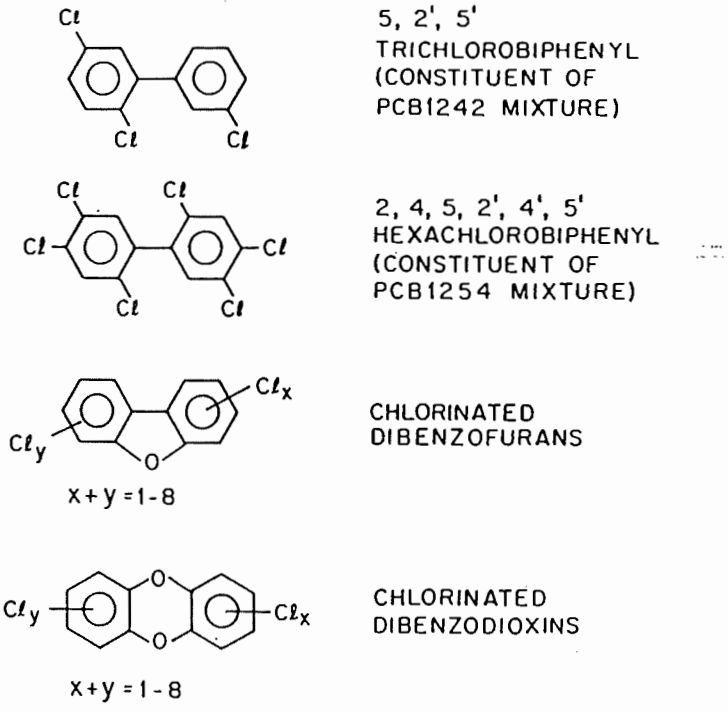


Figure 9. Examples of structures of polychlorinated biphenyls, chlorinated dibenzofurans, and chlorinated dibenzodioxins.

Each of the commercial formulations is composed of 50 to 150 individual chlorobiphenyl congeners. Thus, they are mixtures of chemicals that share some similar properties but may differ in other characteristics such as solubility in water, toxicity, and ability to be biologically metabolized or degraded. Previously, environmental scientists regarded PCBs as a group of similar chemicals; given our limited knowledge on ecosystem processes and transport of PCBs in the 1960s to mid-1970s, this grouping of chemicals was probably adequate. Since the mid-1970s, however, we have realized that we must be more careful in interpreting chemical analyses and understand the fate and effects of the individual chlorobiphenyl congeners and groups of congeners. In addition, we have known for many years that some commercial mixtures of PCBs were contaminated with by-products of their manufacture, such as the chlorinated dibenzofurans (Figure 9). Although found only in trace concentrations, these latter compounds are much more of an environmental concern than PCBs. Furthermore, we know that partial combustion of PCBs in boilers and in municipal waste incinerators gives rise to another class of chemicals of serious environmental concern — the chlorinated dibenzodioxins, often erroneously referred to as "dioxin."

The first substantive hint that PCBs were an environmental problem came in 1968 and 1969 when scientists in Sweden and California noted that PCBs were present as extraneous signals in their analyses of bird eggs for DDT and other chlorinated hydrocarbon pesticides. Because the samples in which the PCBs were found were far removed from any direct source of PCBs, these scientists deduced, by comparison and analogy with DDT that PCBs might be worldwide contaminants. Between 1969 and 1972, surveys and experiments on toxicity and other adverse impacts demonstrated that PCBs were indeed a significant environmental threat, a finding that has been confirmed by a large number of field observations and experiments (NAS, 1979; U.S. EPA, 1977). Open use of PCBs was curtailed (i.e., use in inks, paints, carbonless paper) and substitutes were found starting in 1972 and 1973. A complete ban on new manufacture of PCBs in the United States went into effect in 1977. Since that time, there has been a program to collect PCBs and replace them with other chemicals. For example, collection of PCBs from large power and railroad power transformers has been in progress for several years. Of the total quantity of PCBs manufactured, much is still in use in functioning electrical components (e.g., ballasts for some fluorescent lights). Another portion is in "temporary storage" in landfills, where it accumulated along with discarded electrical and electronic components.

Cornell-Dubilier Electric began manufacturing electrical components using PCBs in 1941 at a site just outside the current New Bedford hurricane barrier. Aerovox, Inc., began to produce PCB-containing components in 1947 at a site further north on the Acushnet River. We will refer in this report to these two companies as Cornell-Dubilier and Aerovox, although we recognize that their exact titles and corporate affiliations have changed several times over the years. There is little doubt that both of these companies contributed to the PCB pollution as it exists today in New Bedford Harbor and Buzzards Bay. The exact routes of input and the amounts contributed by each are unknown.

The concentrations of PCBs found in various components of the ecosystem of New Bedford Harbor have caused the Harbor to be designated an EPA Super-

fund site, one of the few marine Superfund sites in the nation. We can speculate that during the 1950s and 1960s, when the environmental threat of PCBs was largely unknown, drippings and dribblings of PCBs from the manufacturing process were washed into floor drains, then discharged into municipal waste and storm sewers, or into a direct discharge to the Harbor. Anecdotal evidence suggests that some unused PCB compounds were occasionally donated to New Bedford, and perhaps nearby towns and cities, for "oiling" of dirt roads. Some of the rejected electrical components were stored on the grounds of the plants until quantities were sufficient to warrant trucking to the municipal landfill (Weaver, 1984). It is possible that some of these components were washed into the Harbor when severe storms such as the 1954 Hurricane Carol caused extensive coastal flooding.

EPA has estimated the total use of PCBs by Cornell-Dubilier and Aerovox from records of Monsanto and the two New Bedford companies. Aerovox used 19,236,217 lb between 1958 and 1977 (extrapolating the 1958 use for the missing 1959, 1960, and 1961 figures), and Cornell-Dubilier used 19,027,180 lbs between 1962 and 1977. If we take the records for the earliest two years for each company and extrapolate that annual use back to the respective start of operations for each company—almost certainly an overestimate—then we add an additional use of only 1,050,000 lb for Cornell-Dubilier and 990,000 lb for Aerovox. The major use of PCBs for both companies was during the period 1960 to 1977. Overall, both companies together used about 32,000,000 lb (almost 15 million kg or 15 thousand metric tons) of PCBs, or about 4% of Monsanto's total production. If we speculate that the loss rate of PCBs during the electrical manufacturing process was only 1%, then approximately 150 metric tons was lost, some of which could have made its way to the Harbor.

A few measurements of PCBs in the effluent of the New Bedford sewage treatment plant in the late 1970s yielded a value of 7 to 9 micrograms PCB per liter for "total" PCBs. Applying these limited data and the 88 million liters per day of effluent, we obtain a very rough calculation of an input of about 250 kg of PCBs per year (550 lb per year). Measurement of one of Aerovox's wastewater streams yielded a PCB concentration of 73 to 400 micrograms per liter during the late 1970s and early 1980s (Weaver, 1984).

The exact pathways by which PCBs entered the Harbor and the relative contributions from each pathway are largely unknown. It suffices for our purposes in this report to know that discharges from the New Bedford sewer plants and the storm sewers and the direct runoff from plant sites all seemed to contribute PCBs to the Harbor.

Past sources of PCBs to areas of Buzzards Bay outside New Bedford Harbor were marine paints; atmospheric transport from the New Bedford area, the Northeast urban industrial areas, and probably the Mid-West urban industrial areas; water and sediment transport from the New Bedford Harbor area; and dumping of dredged spoils from New Bedford Harbor and perhaps from marinas in some other smaller harbors. The relative proportions of each are unknown, but it is a reasonable assumption that the combination of present-day release of PCBs from New Bedford Harbor and the past disposal of dredge

spoils from New Bedford Harbor has been and continue to be, the major source of PCBs to Buzzards Bay.

The chronology of the PCB pollution in New Bedford Harbor and a description of the situation up to 1982 is presented by Weaver (1984) and Farrington et al. (1985). Since that time many documents have been prepared as part of the Superfund activities and many data have been collected. Because litigation is in progress, these data are not available for review, but some of them have been summarized in a very recent report (Ikalainen and Allen, 1989).

Other Organic Contaminants

New Bedford is an old industrialized city which has, over time, adopted waste-disposal practices common to the region. Analysis of the New Bedford sewage plant effluent has established that several of the synthetic organic compounds that are on EPA's Priority Pollutant list, in addition to PCBs, petroleum and fossil fuel hydrocarbons, and chlorinated pesticides, are present in measurable quantities. These include volatile organic compounds associated with solvent degreasers, cleaning fluids, and other similar products (e.g., dichloromethane, trichloroethane, benzene, and plasticizers such as phthalates common in plastics). These and many more compounds are typical of the compounds found in sewage effluents of urban industrial areas of developed nations.

No data were available for the effluent discharges from the Acushnet Process Companies (Golf Ball Division and Rubber Division), but we suspect that during the 1950s and 1960s these effluents contained petrochemicals, some of which are of environmental concern. The various small and large textile operations located in mills along the New Bedford waterfront almost certainly had effluents that discharged to the harbor during period from the 1940s to the 1960s. Some of these operations used dyestuffs that are derivatives of coal tars and would be considered quite harmful from an environmental perspective. No records on use or discharge are available to us, nor do we have a means of estimating the inputs.

Recent research has shown that a new type of antifouling paint using tributyltin (TBT) as the toxic agent can be harmful to estuarine organisms at extremely low concentrations when it is leached from boat bottoms. There is no way of estimating, nor are there any records of, TBT use or distribution in Buzzards Bay at this time. TBT use in antifouling paints for small boats was banned in Massachusetts in 1988.

Present Status of Toxic Chemicals in Buzzards Bay

Two aspects of the present status of toxic chemical concentrations in components of the Buzzards Bay ecosystems are important to consider: (1) how does the Bay compare with other coastal areas of the United States, particularly the Northeast; and (2) what is the distribution of toxic chemicals within the Bay. There are few reliable values for toxic chemicals in the water column of the Bay because it is difficult to measure these materials at the relatively low concentrations normally found in water. During the past few years analytical measurements have become more tractable, although they continue to demand care and expertise. Skill is also needed to measure chemical contaminats in sediments

and biota, but these analyses are generally less demanding than those for contaminants in the water column. Analysis of sediments sheds light on the status of chemical contamination in an intermediate and long-term reservoir and may provide a long-term history of contamination. Analysis of toxic chemicals in biota helps establish the degree of bioavailability of the contaminants and the level of contamination in valuable living natural resources, and also provides an assessment of potential risks to the human consumer.

Sediments

Trace Metals

The studies of Summerhayes et al. (1977) carefully document and review the extent of trace metal contamination in New Bedford Harbor and western Buzzards Bay. Very high concentrations of copper, chromium, and zinc were detected in the surface sediments of New Bedford Harbor inside the hurricane barrier. In some places these concentrations are as high as 1%, making these sediments ore-grade deposits under other conditions. Most uncontaminated coastal sediments from regions similar to Buzzards Bay have trace metal concentrations many times less than 1%. Typical concentrations are in the range of parts per million rather than the parts per hundred to parts per thousand found in New Bedford Harbor. Cores that penetrate 50 cm to 1 m into the sediment have documented the historical record of the distribution of trace metals with depth. Summerhayes et al. (1977) calculated that at prevailing market prices, there was $2 million worth of trace metals in the sediments of the Harbor inside the hurricane barrier. This estimate does not consider whether the metal is technically or economically recoverable from a soft, wet, highly organic sediment.

Between 1974 and 1987, the National Status and Trends Program for Marine Environmental Quality of NOAA (the National Oceanic and Atmospheric Administration) surveyed about 200 sites in the United States coastal area for concentrations of 12 trace metals. The trace metals analyzed were antimony, arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, tin, and zinc. All chemical contaminants measured in the NOAA Status and Trends Program are listed in Table 5. The four sampling sites in Buzzards Bay were outside of the New Bedford Harbor area (Figure 10). In the ranking of the highest 20 sites in terms of each trace metal analyzed, Buzzards Bay had only one station, Angelica Rock (BBAR), in the top 20 for one metal, silver. In comparison, Narragansett Bay, RI, had three of four stations in the top 20 for tin, one for silver, one for antimony, and one for mercury. However, if we include the Summerhayes et al. (1977) trace metal data for the Inner New Bedford Harbor sediments, then values for copper, chromium, and zinc are among the highest values reported nationwide. This helps us to understand that the trace metal content of some areas of the surface (upper 20-50 cm) muds inside the Harbor is extraordinarily high.

The comparative data from the NOAA Status and Trends Program document that the main part of the Bay, away from the New Bedford Harbor area, is not as extensively contaminated with trace metals as are other areas of the U.S. coast. The data for the four Buzzards Bay stations of the NOAA program are in the same concentration range as data for 11 stations of a survey for trace metal contamination in Buzzards Bay reported over 15 years earlier by Gilbert et al. (1973) and 11 years ago by Summerhayes et al. (1977).

Table 5. Chemicals Analyzed in the NOAA Status and Trends Program.

DDT and its metabolites	Polcyclic aromatic hydrocarbons (PAH)	Major elements
o,p'-DDD	2-ring	Aluminum
p,p'-DDD	Biphenyl	Iron
o,p"-DDE	Naphthalene	Manganese
p,p'-DDE	1-Methylnaphthalene	Silicon
o,p'-DDT	2-Methylnaphthalene	
p,p'-DDT	2,6 Dimethylnaphthalene	
Acenaphthene		
Chlorinated pesticides other than DDT	3-ring	Trace elements
Aldrin	Fluorene	Antimony
Alpha-Chlordane	Phenanthrene	Arsenic
Trans-Nonachlor	1-Methylphenanthrene	Cadmium
Dieldrin	Anthracene	Chromium
Heptachlor	Copper	
Heptachlor epoxide	4-ring	Lead
Hexachlorobenzene	Flouranthene	Mercury
Lindane (gamma-BHC)	Pyrene	Nickel
Mirex	Benz(a)anthracene	Selenium
		Silver
	5-ring	Tin
	Chrysene	Zinc
	Benzo(a)pyrene	
	Benzo(e)pyrene	
	Perylene	
	Dibenz(a,h)anthracene	

Polychlorinated biphenyls
(measured as levels of chlorination in 1984-1987 and as 18 individual chlorobiphenyl congeners in 1988).

Fossil Fuel Hydrocarbons

PAH concentrations in Buzzards Bay sediments are typical of the concentrations found in coastal waters of highly populated and industrialized areas of the Northeast United States (e.g., Long Island Sound, Narragansett Bay, and Massachusetts Bay). In the NOAA Status and Trends Program, sediments were analyzed for 18 aromatic hydrocarbons (Table 5). The Angelica Rock station in Buzzards Bay ranked ninth nationwide in terms of high concentrations of total PAHs (Figure 11).

Much of the pioneering research that has led to our ability to measure PAHs in marine sediments and to understand the significance of these data was conducted in Buzzards Bay by the late Dr. Max Blumer, a Senior Scientist at Woods Hole Oceanographic Institution. Blumer and Youngblood (1974) and Youngblood and Blumer (1975) showed clearly that PAHs from combustion products had important compositional differences compared to PAHs in unburned petroleum (e.g., fuel oils and crude oils). They noted that the PAHs in Buzzards Bay surface sediments appeared to be primarily from combustion-product sources, with the exception of one or two areas in Outer New Bedford Harbor. Concentrations of fossil fuel hydrocarbons were higher in New Bedford

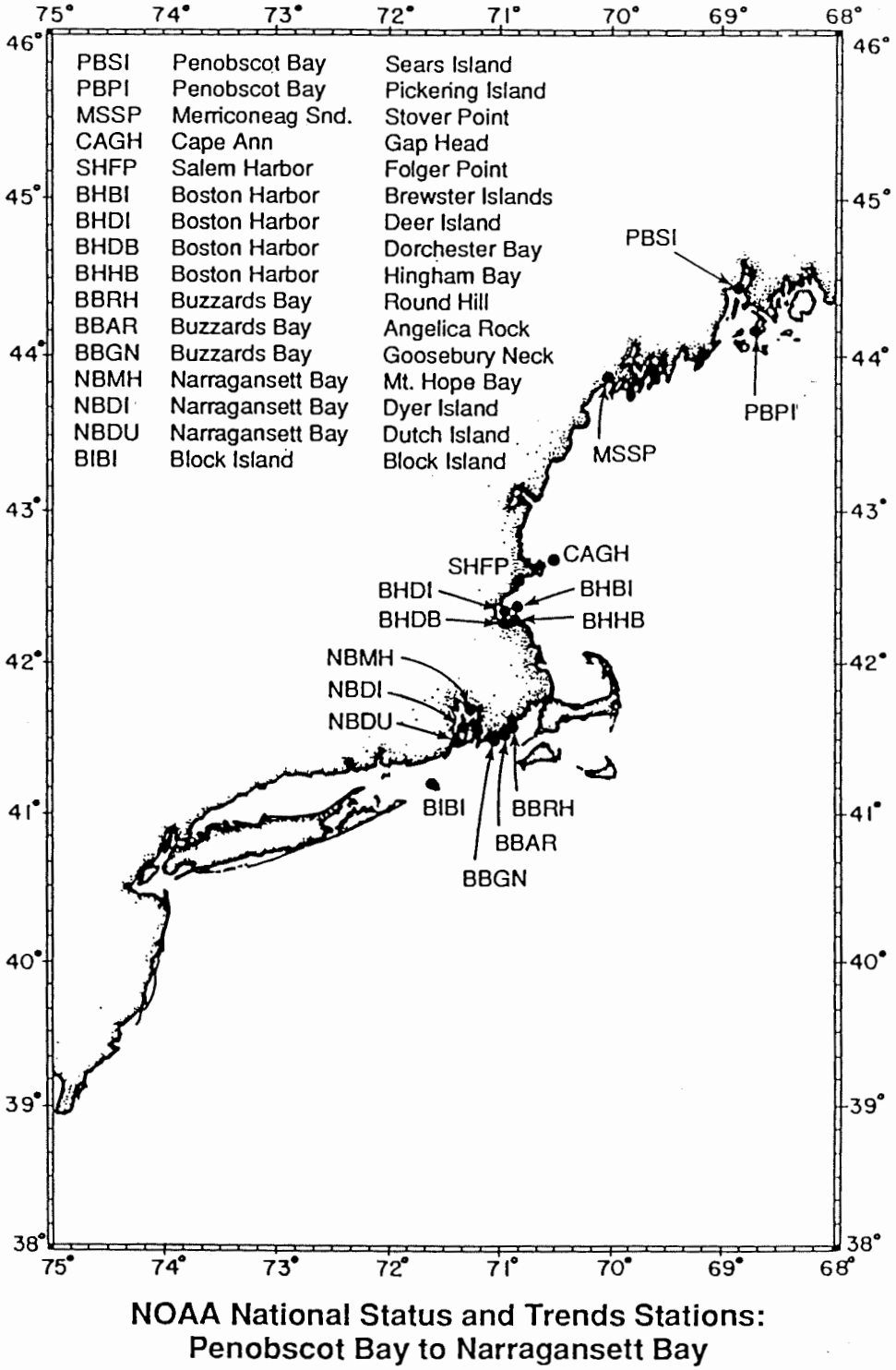


Figure 10. NOAA National Status and Trends Program sampling sites, including Buzzards Bay sites, in the New England area (NOAA, 1988).

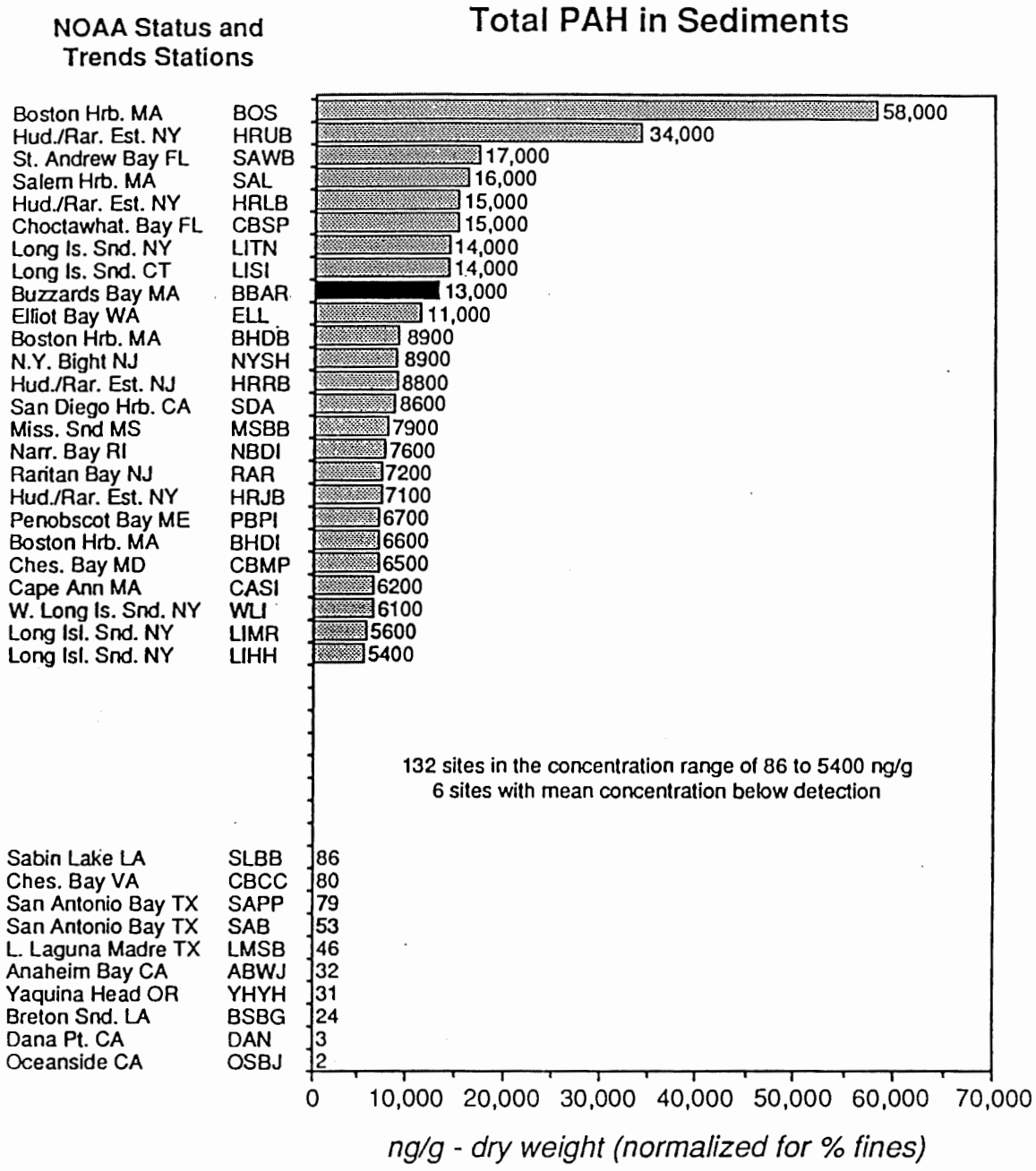


Figure 11. NOAA National Status and Trends Program compilation of totals of selected polynuclear aromatic hydrocarbon concentrations in surface sediments (NOAA, 1988).

Harbor surface sediments, as expected for an urban harbor, and these hydrocarbons had compositions that indicated contamination by petroleum as well as by combustion-product PAHs from fossil fuel burning. More recent studies of the New Bedford Harbor surface sediments by Pruell et al. (1990) document that concentrations of PAHs in surface sediments in several areas of the harbor inside the hurricane barrier are typical of PAH concentrations in urban harbor areas.

Hites et al. (1977) and Farrington et al. (1977) measured the aromatic hydrocarbons in sediment cores from the middle of the Bay and showed that there was a historical record of increasing aromatic hydrocarbon inputs that corresponded with the industrialization and use of fossil fuels for heat and energy in the Northeast United States. It appears that most of the PAHs in sediments deposited after 1850-1900 were derived from incomplete combustion of fossil fuels (coal, oil, and gas). Prior to 1850 most of the PAHs were derived from forest fires and grass fires. The PAHs are released to the atmosphere and then deposited on land in the watershed of Buzzards Bay and to the Bay itself. Rain and runoff wash the PAHs deposited on land into the Bay directly or via rivers and streams. This process has been verified in several coastal areas and lake watersheds around the world (Farrington and Wakeham, 1980). Petroleum hydrocarbons other than PAHs have been measured in surface sediments of the Bay and most probably come from the constant dribbling input of oil from the sewage effluent of New Bedford and from the many small boats that navigate the waters of the Bay (Farrington et al, 1977; Youngblood and Blumer, 1975). Several very important studies of the fate of petroleum hydrocarbons from oil spills in the eastern part of the Bay along the West Falmouth and Bourne coastal areas have resulted in a reasonably good understanding of petroleum hydrocarbons in sediments in those areas. Sediments taken some distance from the spill area and analyzed for petroleum hydrocarbons show very low concentrations in comparison with Outer New Bedford Harbor. Marinas may be an exception to this observation. Few accurate and detailed modern analytical measurements have been made of petroleum hydrocarbons in sediments of small congested harbors and boat marinas.

Chlorinated Pesticides

Chlorinated pesticides in surface sediments have also been measured by the NOAA Status and Trends Program (Table 5). The Angelica Rock station in Buzzards Bay ranks 14th among the top 20 stations in the Status and Trends Program for sediment contamination by non-DDT chlorinated pesticides. No Buzzards Bay stations are in the top 20 for pesticides of the DDT family.

Polychlorinated Biphenyls

Sediment from the Angelica Rock station in Buzzards Bay has the highest ranking for PCBs of any NOAA Status and Trends sediment station nationwide (Figure 12). This ranking is only a weak indication of the degree of PCB contamination in this region of Buzzards Bay. Many analyses of surface sediments in the New Bedford Harbor-Western Buzzards Bay area (e.g., Commonwealth of Massachusetts, 1982; Weaver, 1984; Farrington et al., 1985, 1986a; Brownawell and Farrington, 1986, and references therein) have documented very high concentrations of PCBs in the sediments of these areas compared to

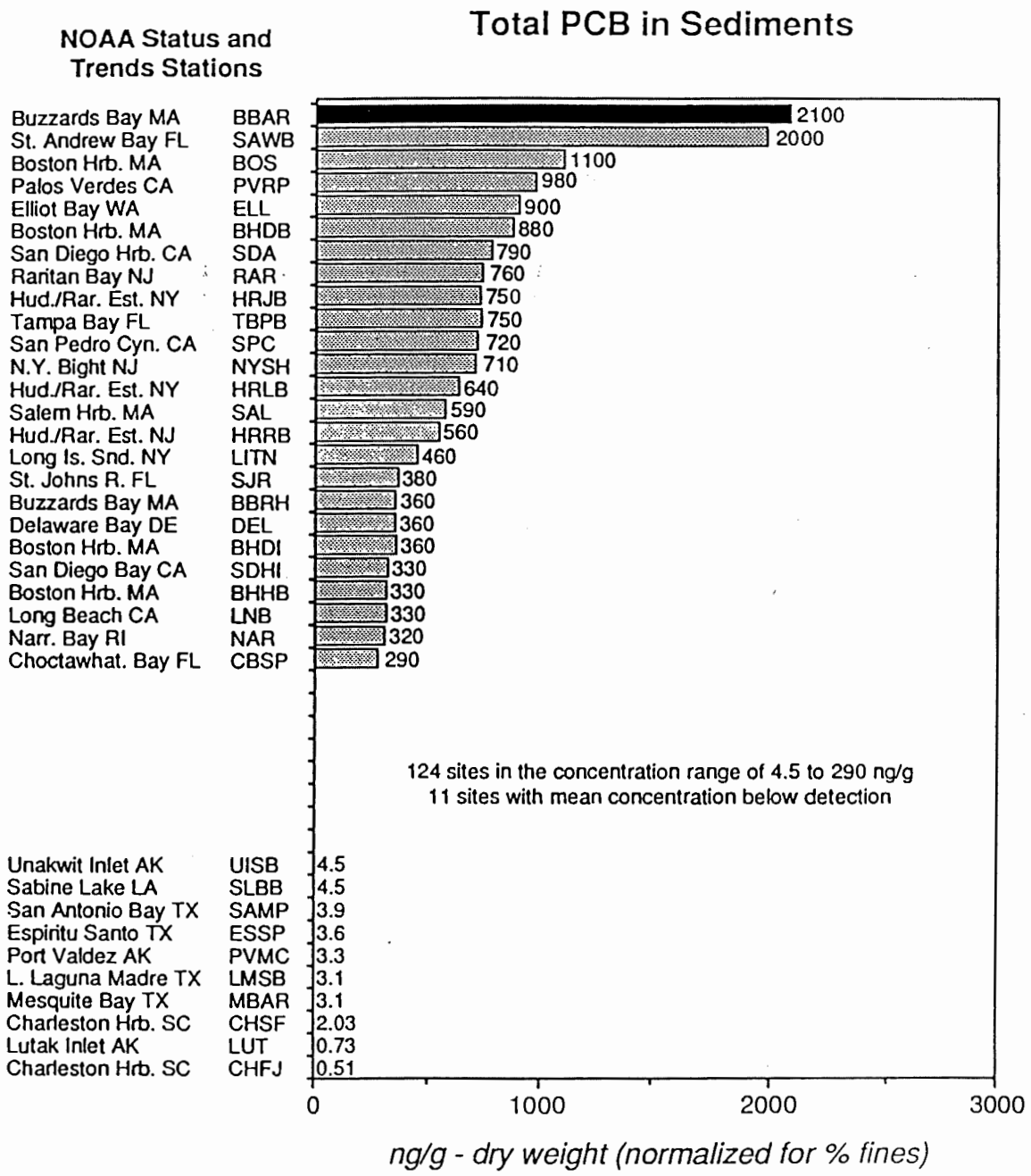


Figure 12. NOAA National Status and Trends Program compilation of total polychlorinated biphenyl concentrations in surface sediments.

coastal sediments elsewhere in the world. Sediments in the upper part of the Harbor above the Interstate Route 195 Causeway have a PCB content as high as an astounding 1%. Spatial contours for the PCB concentrations in the surface sediments indicate where the highest concentrations of PCBs are found (Figure 13).

A depth profile, obtained by Dr. Bruce Brownawell, of PCBs in a sediment core from the channel near Butler's Flat Light in the Outer Harbor is presented in Figure 14. It has been possible to use core samples and analyses of PCBs to show that most of the PCBs in the sediments are in the upper 50 cm, a situation similar to that of trace metals. Professor Jacek Sulanowski used the core data from several reports, the spatial contours, and some basic geological information about the sediments to estimate that the upper 50 cm of harbor sediments inside the hurricane barrier contain between 110 and 300 tons of PCBs (Farrington et al., 1985). Subsequent work under the auspices of the EPA Superfund activity suggests that the estimate is reasonable. As we discussed earlier in this report, if 1% of all the PCBs used by Cornell-Dubilier and Aerovox reached the environment, the total would be 150 tons of PCBs, an amount bracketed by the 110 to 300 tons estimated for the sediments. The percentage of PCBs that entered the Harbor is probably higher than 1% because some would have been lost to the atmosphere via vapor exchange from the water, some would have been mixed out of the Harbor by tidal exchange with Buzzards Bay, and some would have been removed with sediments that were dreged and deposited outside of the Harbor. If we include the estimated 300 tons of PCBs that were disposed and buried in the municipal landfill for New Bedford (Weaver, 1984), then the estimated release to the environment (sediments and landfill) is between 400 and 600 tons, or about 2 to 4% of the total PCBs used by the two companies.

In his Ph.D. dissertation, Brownawell (1986) analyzed PCB concentrations in a core from the mud area in the middle of Buzzards Bay and found evidence of a buildup of PCB concentrations in the sediment of this area over time (Figure 15). He pointed out that it is not clear from the data available whether the increase was gradual and steady or resulted from periodic events such as dredged-spoil disposal and storm resuspension and transport followed by mixing of the PCBs deeper into the sediments by small organisms living in the surface muds. Brownawell also demonstrated conclusively (Brownawell, 1986; Brownawell and Farrington, 1985, 1986) that PCBs were present in relatively high concentrations in the pore waters of sediments. These are the waters that are present with the mineral and organic matter in wet sediment. He further showed that the exchange of these PCBs between the waters in the sediment and the overlying bottom waters of the Bay could be a long-term process. If so, the reservoir of PCBs in the sediments might continue to contaminate the waters and biota of Buzzards Bay, and especially the New Bedford Harbor area, long after input of PCBs from the manufacturing plants to the Harbor had ceased.

Polychlorinated Dibenzofurans and Polychlorinated Dibenzodioxins

An important study of the distribution of the polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) in surface sediments of New Bedford Harbor has recently been completed by Pruell et al. (1990). PCDDs and PCDFs are more toxic than PCBs or PAHs and can cause adverse human

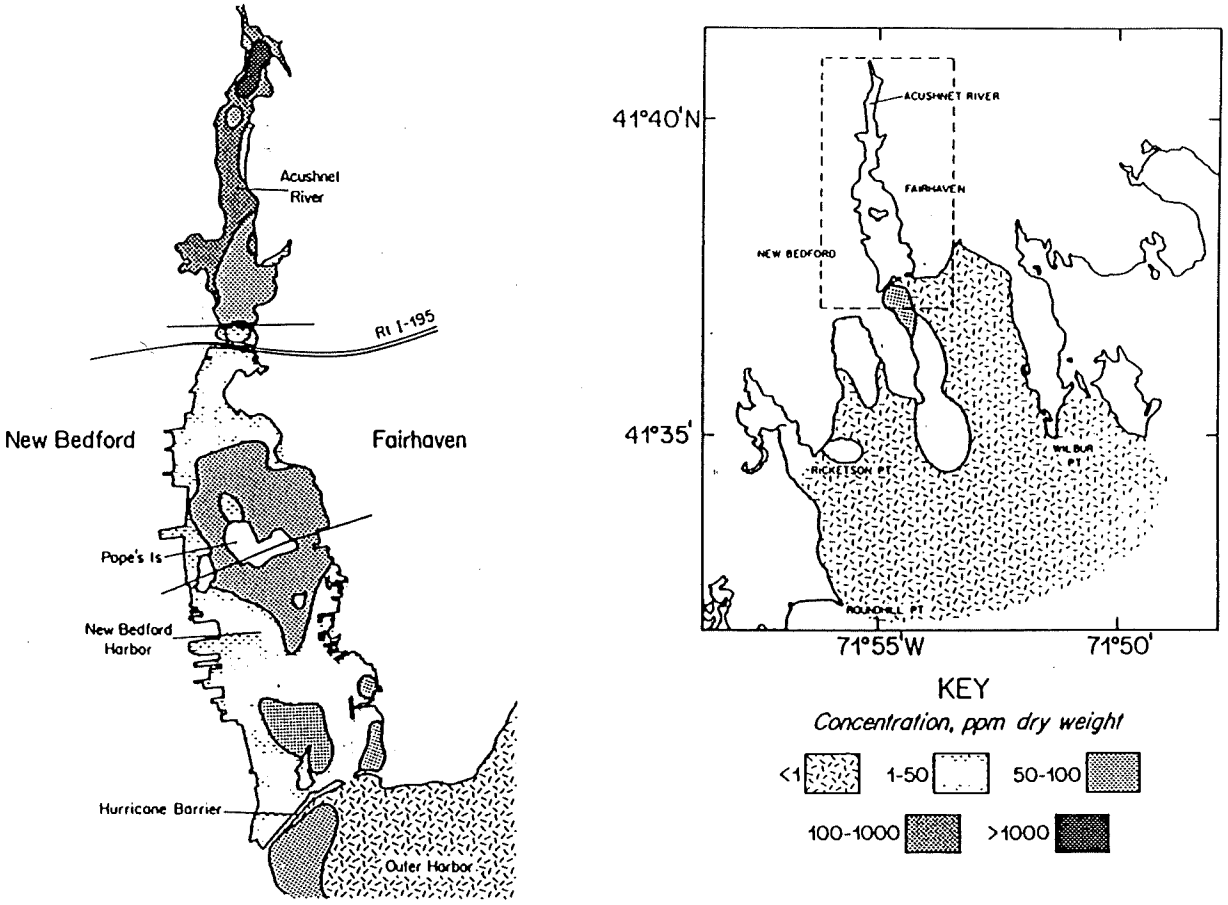


Figure 13. Contour map of polychlorinated biphenyls concentraions in surface sediments of New Bedford Harbor and western Buzzards Bay (from Farrigton et al., 1986).

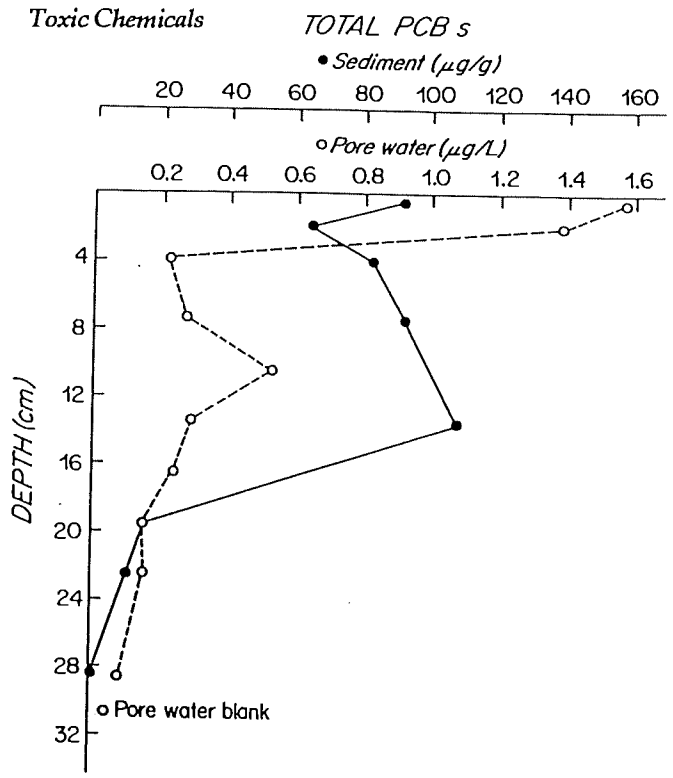


Figure 14. Depth profiles of polychlorinated biphenyls in sediments from a sediment core taken in Outer New Bedford Harbor near Butler's Flat Lighthouse (from Brownawell, 1986 and Farrington et al., 1986).

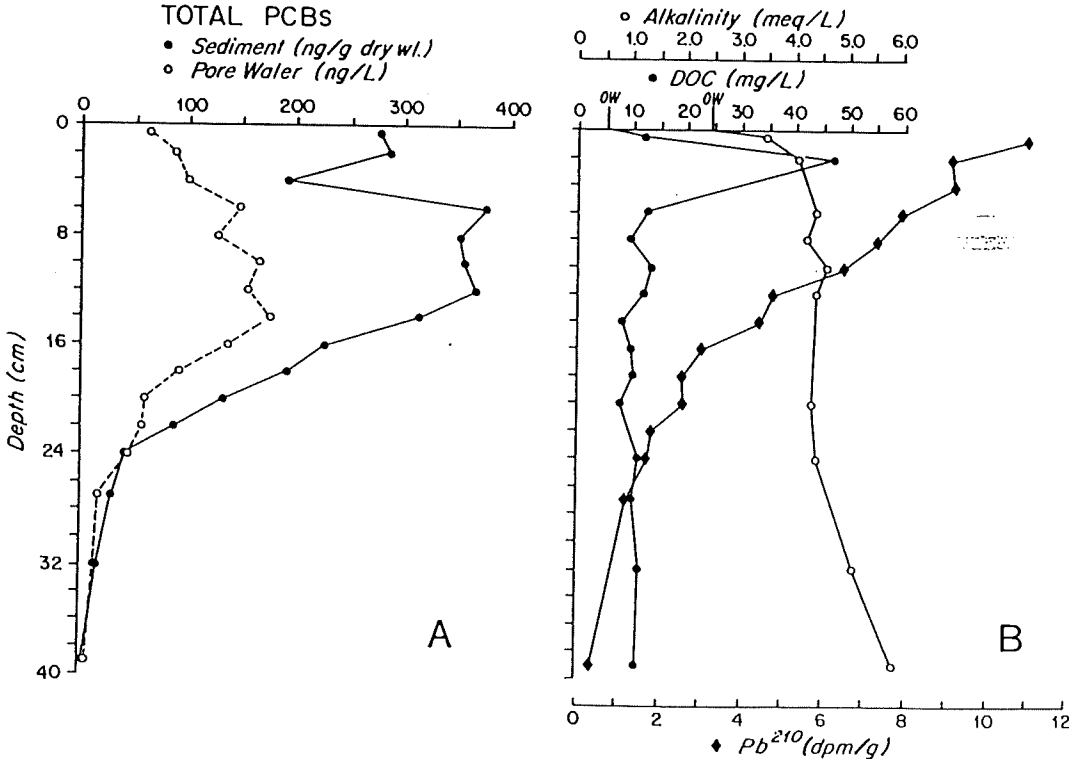


Figure 15. Depth profiles of polychlorinated biphenyls in a sediment core taken near the middle of Buzzards Bay in a patch of mud north of the Weepeckett Island (from Brownawell, 1986).

health problems more readily on a molecule-to-molecule basis. In other words, whereas we are concerned about PAHs and PCBs at concentrations of parts per million and hundreds of parts per billion, we are concerned about PCDDs and PCDFs at concentrations as low as 1 to 10 parts per billion.

The concentration and distribution of PCDDs in the surface sediments of the Harbor appears to be typical of urban areas in the Northeast United States. The distribtuion of PCDFs, on the other hand, suggests that there is a source near the mouth of the Harbor. Pruell et al. (1990) hypothesized that the New Bedford sewge treatment plant could be this local source. Incomplete combustion of PCBs is known to produce PCDFs. If PCBs accumulated in the sludge of the treatment plant as a result of releases to the sewer system from the manufacturing plants, burning of the sludge in the onsite incinerator could have released partial combustion products from the stack. This, plus any runoff associated with the release of PCDFs and cooling water in the plant itself, could account for most of the PCDFs in the surface sediments of the Harbor.

Biota

Studies of chemical contaminants in biota from Buzzards Bay area include information for several classes of organic contaminants and trace metals. Although analytical methods and the ranking of contaminants of most concern have changed over the past 15 to 20 years, there are several sets of data that provide a reasonable overview of the concentrations of chemical contaminants in various biota of the Bay's ecosystems and for various areas of the Bay. Species that are harvested in the commercial and recreational fisheries of the Bay are studied.

Data for chemical contaminants in biota may be expressed in a number of different units. Units such as ppm (parts per million) or ppb (parts per billion) are frequently used. These can be translated into micrograms (10^{-6} grams) per gram for ppm or nanograms (10^{-9} grams) per gram for ppb. Research scientists often report data on the basis of dry tissue weight (i.e., on the weight of a tissue from which the water has been removed by heating in an oven or by vacuum drying). Public health officials and government agency scientists often use wet weights. Both groups may report data on the basis of lipid weight (the amount of fat and oil in the sample) because most organic contaminants such as DDT, PAH, and PCB are "lipophilic" (lipid-, or fat and oil, loving) and concentrate in the fat-rich portions of organisms. Thus, fatty tissues often have much higher concentrations than non-fatty tissues, and organisms with more fat per unit of body weight usually have higher amounts of these lipophilic contaminants than organisms that have a lower proportion of fats. We have specified the basis of the concentrations we discuss, but this is not always the case. The reader should keep this in mind when comparing concentrations of chemical contaminants from different sources.

A considerable portion of the data on chemical contamination of living marine resources in the nation's coastal areas is for bivalve shellfish. In the United States, Butler (1973) pioneered the idea that bivalves were a reasonable species for regional monitoring of chemical contaminants in coastal areas. The U.S. EPA "Mussel Watch" Program improved on the concept and tested it nationwide (Goldberg et al., 1978; Farrington et al., 1983). The EPA prototype program

resulted in the bivalve shellfish portion of the monitoring program known as the NOAA National Status and Trends Program. The sediment component of this program has been discussed above. Bivalve shellfish are good sentinels for chemical contamination because they are relatively sedentary. Thus, unlike fish, crabs, and lobsters which move from place to place, we can be reasonably certain that the chemical contamination of the bivalve shellfish reflects the chemical contamination in the habitat where they were sampled. This assumption has been supported by the results of many experiments in large aquaria and with transplanted shellfish. Because similar species of shellfish can be sampled in many areas of the U.S. coast, data from local, regional, and national databases can be compared with less concern about differences caused by physiology and biochemistry of shellfish species. Farrington (1983) discussed the use of bivalves as sentinel organisms for coastal pollution assessment in more detail in an article written for the nonscientist.

Trace Metals

During the past 12 years studies on trace metals in Buzzards Bay fish and shellfish have included values for copper, zinc, lead, cadmium, nickel, chromium, and — to a lesser extent — mercury, silver, arsenic and selenium. The organisms most frequently sampled are the American lobster (*Homarus americanus*), the blue mussel (*Mytilus edulis*), the quahog (*Mercenaria mercenaria*), and the winter flounder (*Pseudopleuronectes americanus*). The predominance of samples of bivalve molluscs in the data sets is the result of (1) their potential to bioaccumulate metals to levels that are orders of magnitude above water column concentrations; and (2) their use as biological indicators of water quality in large-scale monitoring programs such as the Mussel Watch program and the National Shellfish Sanitation Program (NSSP).

Distinguishing between natural and enhanced levels in marine biota is extremely difficult without detailed information on background levels for different species and the extent to which those background levels vary naturally as a result of environmental and biological factors. The current database for trace metal levels in fish and shellfish from Buzzards Bay is inadequate to make such a distinction with a high degree of confidence. Background levels of trace metals between species can vary as much as several orders of magnitude; whereas levels in individuals of the same species sampled along a gradient of habitats from uncontaminated to contaminated may vary by less than an order of magnitude. Differences in metal concentrations in *Mytilus* vary seasonally by a factor of 2 to 4 due to changes in physiological and/or reproductive condition. Few of the samples in the data sets for trace metals exceed any of the human health standards or the U.S. FDA/NSSP alert levels for trace metals (see discussion and tables in Section VII of this report). The few exceptions include cadmium levels in several species collected in New Bedford Harbor (Kelly, 1978).

In 1988, concentrations of copper in mussels at the Round Hill and Angelica Rock stations were among the top 20 highest concentrations in the National Status and Trends Program sampling of U.S. coastal areas (15th in 1988 for both stations). However, these stations were not in the top 20 in the two previous years. During 1986 and 1987, concentrations of copper were 10 to 15 mg/g dry weight tissue — not much different than the average values for the entire U.S.

coast. A similar situation occurred for lead in mussel tissue for which the Angelica Rock station was ranked 16th in 1986 and 15th in 1987.

In 1984, concentrations of trace metals were analyzed in fish liver samples as part of the National Status and Trends Program. Of the 43 states sampled nationwide, the one sampling site in Buzzards Bay ranked as follows for each metal: 15th for antimony, 5th for arsenic, 29th for cadmium, 22nd for chromium, 36th for copper, 5th for lead, 26th for mercury, 30th for nickel, 36th for selenium, 19th for silver, 35th for tin, and 15th for zinc. However, the comparison between geographic locations was hampered by the fact that different species were sampled in many areas of the U.S. coastline. Thus the concentrations of trace metals in the livers were influenced not only by the concentrations in the habitat, but also by the biochemical and physiological factors that were different for each species.

Fossil Fuel Hydrocarbons

Petroleum hydrocarbon (PHC) — including PAHs — may be derived from a variety of different sources including accidental oil spills and chronic inputs from municipal discharges and marinas. The general distribution of PHC and PAHs is similar to trends observed for PCBs, but the contaminants do not co-vary (in other words, high concentrations of PHC are not always found in samples that have high concentrations of PCBs). In comparing data from the U.S. Mussel Watch program, Farrington et al. (1982) observed distinct geographical differences in hydrocarbon mixtures between stations. These differences reflect variations in the realtive amounts of hydrocarbon pollution from pyrogenic sources (derived from burning of fossil fuels) and petrogenic sources (derived from petroleum discharges). Shellfish collected near harbors had elevated levels of PAHs, with a compound distribution indicative of petroleum pollution, presumably derived from chronic inputs from municipal wastewater and small spills of fuel oil.

PAH and PHC levels tend to follow the same trends as seen in PCBs. Mussels from New Bedford Harbor have higher body burdens than organisms collected from other areas of the Bay. PAH concentrations in mussels collected along the Massachusetts coast as part of the Mussel Watch program were generally <0.1 ppm wet weight, with the exception of the Boston Harbor station where values ranged from 0.3 to 0.5 ppm. PAH concentrations in fish muscle tend to be much lower (<0.005 ppm), even in Boston where a high incidence of fin rot and neoplastic lesions in finfish species has been suggested to be related to high PAH concentrations in sediments.

Out of 132 sites nationwide in the National Status and Trends Program, the Angelica Rock station in Buzzards Bay ranked 19th in 1987 and 17th in 1988 in terms of the total of lower molecular weight aromatic hydrocarbons in bivalve (mussel) tissue, but was not in the top 20 in 1986. The total of lower molecular weight hydrocarbons is a sum of the concentrations of the following compounds: naphthalene, 1-methylnaphthalene, 2-methylnaphthalene, biphenyl, dimethylnaphthalene, acenaphthene, fluorene, phenanthrene, 1-methylphenanthrene, and anthracene. None of the bivalves from three National Status and Trends Program stations in Buzzards Bay had concentrations of the higher molecular weight PAHs high enough to be included among the stations ranked in the top 20. The higher molecular weight PAHs include fluoranthene, pyrene,

benzanthracene, chrysene, benzo(a)pyrene, benzo(e)pyrene, perylene, and dibenzanthracene.

We (Farrington et al. and Capuzzo et al., in preparation) recently analyzed tissue from mussels transplanted from a clean site in Sandwich, MA, to two stations in Buzzards Bay — one just inside the hurricane barrier and the other near Cleveland Ledge light — and a third station in Nantucket Sound. We include these unpublished data because there are so few high-resolution analyses of fossil fuel hydrocarbon data for the shellfish of Buzzards Bay. The concentrations of PAHs were higher in mussels from the Harbor area than in the mussels from Cleveland Ledge and Nantucket Sound (see Appendix). The composition of PAHs in the mussels transplanted to the Harbor indicated that the PAHs were derived from both a petroleum source, probably fuel oil, and a combustion source. Samples of quahog (*Mercenaria mercenaria*) and soft-shelled clams (*Mya arenaria*) from inside the hurricane barrier and the Inner Harbor also had elevated PAH concentrations, as would be expected for urban harbor areas; the compositions also indicated a mixed source of PAHs, both petroleum and combustion product PAHs (Farrington, Gooch, and Capuzzo, in preparation).

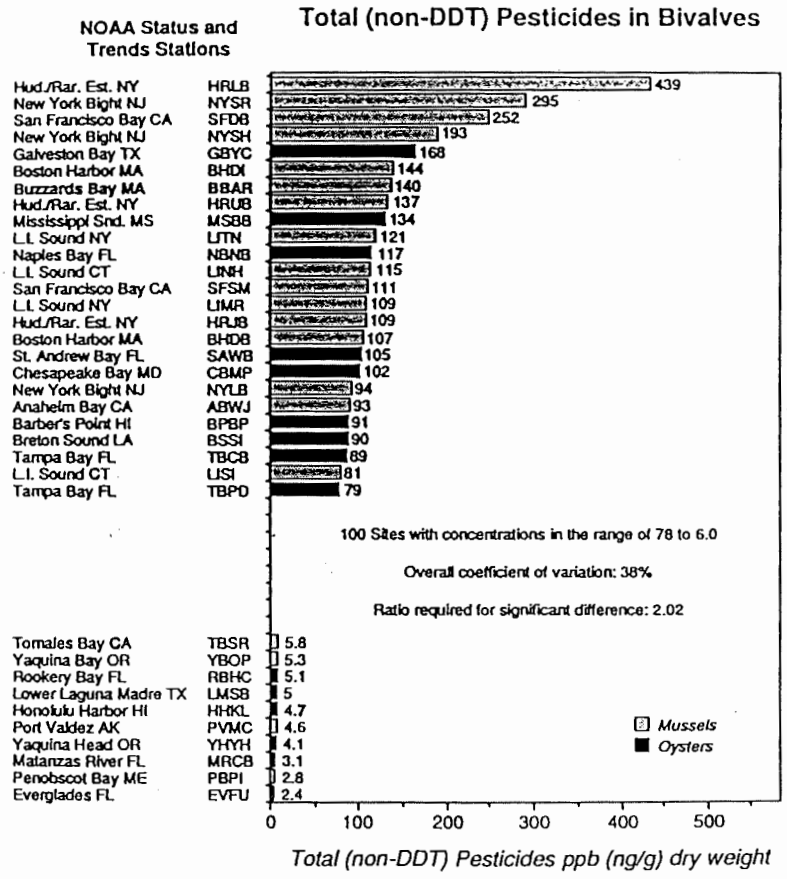


Figure 16. NOAA Status and Trends Program compilation of concentrations of chlorinated pesticides (except for DDT family compounds) in bivalve samples (1986 sampling).

There are species-to-species differences in the concentrations and composition of PAHs when mussels, quahogs, and soft-shelled clams are compared. These differences are probably due to the habitats of the three species. For example the soft-shelled clams were found in muddy-sandy sediments whereas the mussels were suspended in the water column.

During the early 1970s, Blumer and coworkers analyzed oysters and clams as part of the studies of the West Falmouth oil spill of 1969. They clearly documented that the shellfish in the immediate area of the spill were contaminated by the spilled fuel oil. Teal, Burns, and coworkers also documented that several species of biota in the West Falmouth marshes, especially in the Wild Harbor marsh, were contaminated by the spilled oil for almost two years. The fuel oil hydrocarbons are not a consistent contaminant in the western portion of the bay, but the several spills that have occurred have resulted in periodic contamination of the shellfish in the area (Blumer et al., 1970; Burns and Teal, 1971; 1979; Burns, 1975; Farrington et al., 1982; 1986b).

Chlorinated Pesticides

The Angelica Rock station in Buzzards Bay ranked 7th in total DDT concentrations in bivalve mollusc (mussel) tissues in 1986 (Figure 16) but was not in the top twenty ranking in 1987 and 1988. The same station ranked 10th for chlordane concentrations in 1986, 16th for lindane concentrations in 1986, 2nd for lindane concentrations in 1987, 6th and 7th respectively for 1986 and 1987 for dieldrin concentrations. The Buzzards Bay stations were not among the top twenty stations with the highest concentrations of chlorinated pesticides for all other years not noted. Over the three years of sampling, 1986-1988, the three stations in Buzzards Bay had mean concentrations in the mussel tissues as follows: total DDT family pesticides - 13 to 270 nanograms per gram dry weight; chlordane - 5.4 to 90 nanograms per gram dry weight; dieldrin - not detected (<0.1) to 64 nanograms per gram dry weight; lindane - not detected to 9.8 nanograms per gram dry weight. For comparison, the highest concentrations measured for each compound in bivalve mollusc tissues in the U.S. coastal areas as part of the National Status and Trends Program were as follows: total DDT family - 500 to 1000 nanograms per gram dry weight range for samples from the Hudson River-Raritan Estuary of the New York-New Jersey area and near San Pedro Harbor, Anaheim Bay and Palos Verdes, California, and Choctawatchee Bay, Florida; chlordane - 200 to 300 nanograms per gram dry weight for the Hudson River-Raritan Estuary, Marina Del Ray, California, and Choctawatchee Bay, Florida; dieldrin - 100 to 170 nanograms per gram dry weight range for the Hudson River-Raritan Estuary and San Francisco Bay, California; lindane - 20 to 40 nanograms per gram dry weight for the New York Bight and the Hudson River-Raritan Estuary.

In summary, the chlorinated pesticide concentrations in mussels from Buzzards Bay are in the low-concentration groupings compared to many other areas of the U.S. coast. For the few areas of the U.S. coast for which there are data, the concentrations of chlorinated pesticides in tissues of bivalve molluscs and other coastal marine organisms have been declining since the late 1960s and early 1970s (Mearns et al., 1988).

Nisbet and Reynolds (1983) reported on chlorinated hydrocarbon pesticides in the eggs and some food organisms of the common tern (*Sterna hrundo*). They

also provided a brief review of data up to that time for chlorinated pesticide analyses for coastal Massachusetts marine organisms, including some collected from the Buzzards Bay area. The common tern (*Sterna hirundo*) is a migratory species that winters in South America and spends two to four weeks feeding in local inshore waters prior to egg laying (Nisbet and Reynolds, 1983, and references therein). Concentrations of DDE and TDE in the eggs were highly correlated with concentrations in prey fish from the same areas. For Buzzards Bay samples, the concentrations of DDE, dieldrin, and hexachlorobenzene declined significantly between 1971 and 1981.

Krebs et al. (1974) and Krebs and Valiela (1977) reported on chlorinated pesticides in marsh samples taken from the Sippewissett Marsh where pesticides had been introduced in small experimental areas and caused adverse effects in fiddler crabs. This is one of many studies that clearly demonstrated the wisdom of decisions to ban and reduce widespread use of chlorinated hydrocarbon pesticides.

As a result of the ban on the use of some chlorinated hydrocarbon pesticides and the reduced and restricted use of others, concentrations of these chemicals generally declined in coastal marine organisms between the late 1960s and the late 1970s. Data showing this trend are not extensive, but they are sufficient to make chlorinated pesticides less important in measurement and monitoring programs. For this reason, there are few data from the 1980s for chlorinated hydrocarbon pesticide residues in marine organisms from Buzzards Bay.

Little information is available on the distribution of currently used pesticides in fish and shellfish. Monitoring of current pesticide use, particularly from cranberry bogs and other agricultural activities in coastal areas, is warranted. Chemicals of concern include organophosphate and organocarbamate compounds used as herbicides and pesticides. Diazinon, parathion, Sevin, and malathion fall into this category.

Polychlorinated Biphenyls

There are many sets of data for PCB concentrations in marine organisms in Buzzards Bay, especially the western portion. This is the result of widespread recognition that a severe PCB pollution problem existed in New Bedford Harbor and the western Bay. The PCB data have been compiled and summarized by Metcalf and Eddy (1983), Capuzzo et al. (1987) and Brown et al. (1987). Every organism analyzed in the Harbor area has contained substantial quantities of PCBs. The concentrations of PCBs in edible tissue of lobsters sampled in the Outer Harbor have been in the range of 2 to 5 ppm (wet-weight basis) and higher. Two parts per million is the current level recommended by the U.S. FDA as the upper limit for edible portions of fish marketed for human consumption. In 1979 the Massachusetts Department of Public Health enacted a fishery closure in New Bedford Harbor (Figure 17). Three areas were designated on the basis of the previous FDA-recommended level of 5 ppm:

Area I: closed to all fishing

Area II: closed to bottom fishing and lobstering

Area III: closed to lobstering.

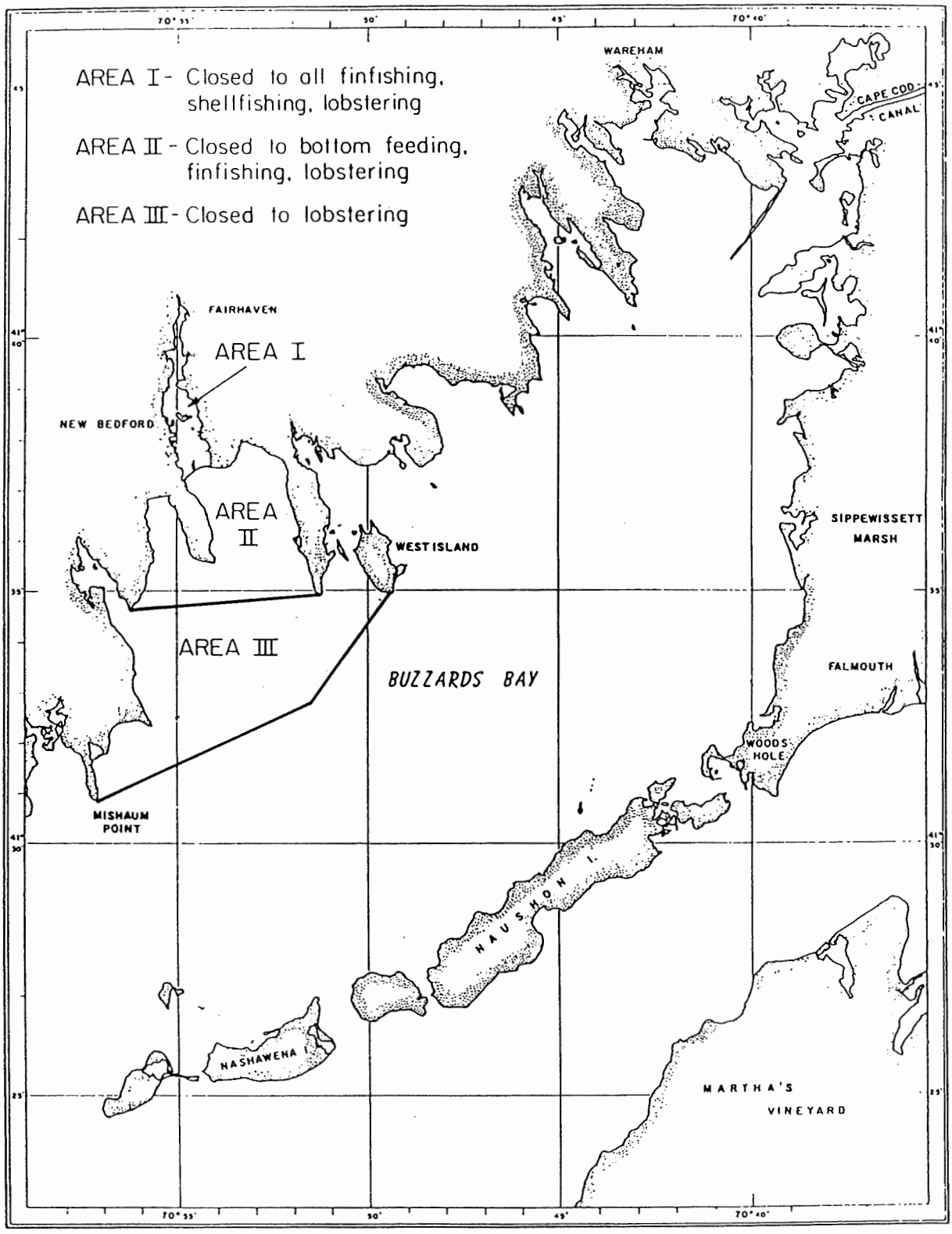


Figure 17. Areas of fishery closure in New Bedford Harbor and western Buzzards Bay due to polychlorinated biphenyls concentraitons in fish, shellfish, and lobster.

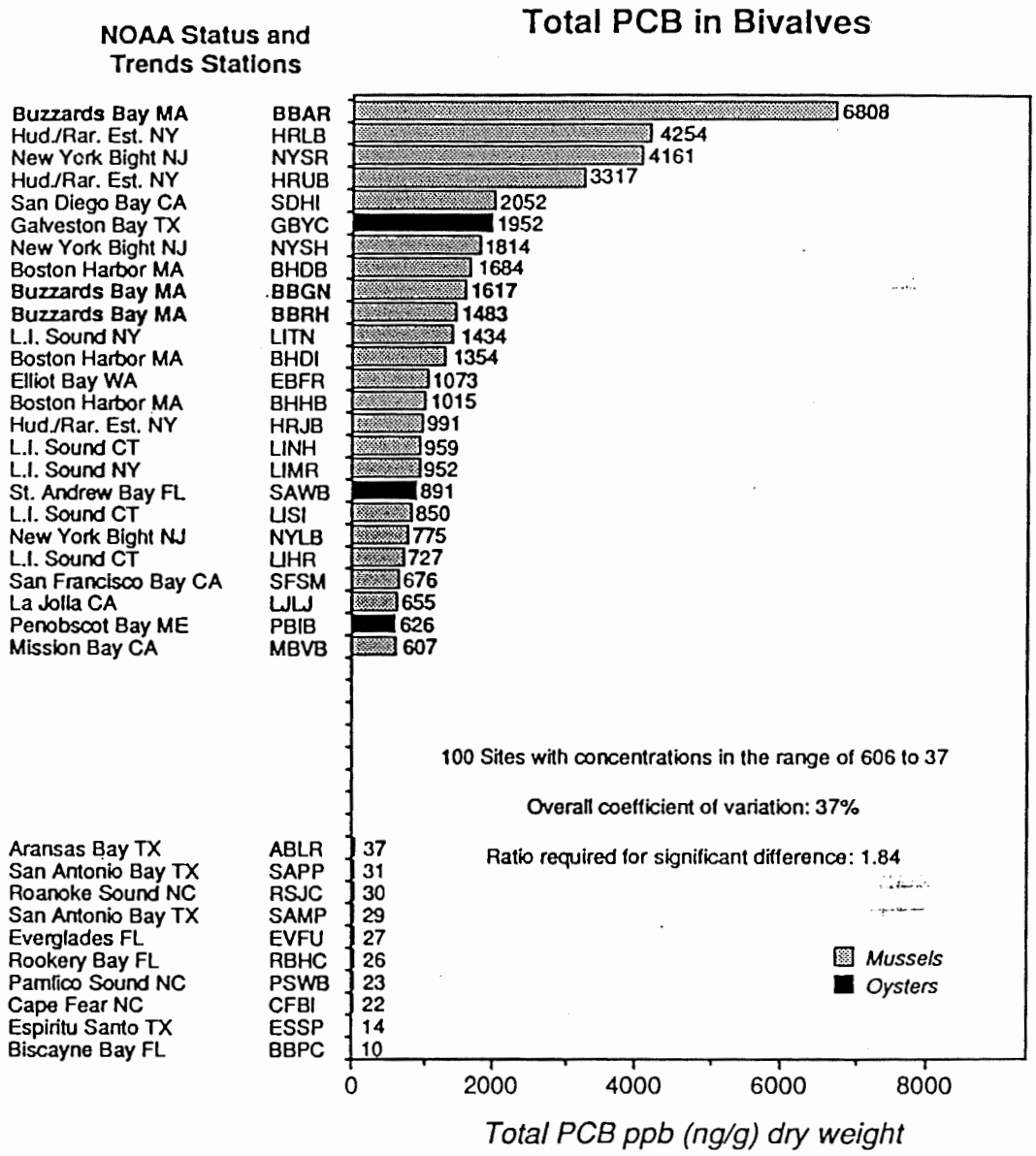


Figure 18. NOAA National Status and Trends Program compliltion of polychlorinated biphenyls concentrations in bivalves (1986 sampling).

In 1985 this closure was amended to the new FDA level of 2 ppm under the jurisdiction of the Massachusetts Division of Marine Fisheries with enforcement by the Division of Environmental Law Enforcement. Enforced closures to shellfishing in Area II are maintained on the basis of coliform contamination.

Nonmigratory resources outside of the Harbor region do not show elevated levels of PCBs. Migratory species, however, present a different type of problem —both in analysis and jurisdiction. Striped bass and bluefish populations have chronically high levels of PCBs, in part as a result of their feeding habits and the high fat content of their tissues. Advisories have been issued by several New England states warning people to limit their intake of these species; special caution is advised for children and for women of child-bearing age. Massachusetts has not closed its striped bass fishery because mean concentrations of PCBs do not exceed the U.S. FDA action level of 2 ppm wet weight.

The severity of the PCB pollution in the New Bedford Harbor and Buzzards Bay region is illustrated by comparison with other coastal sites. Mussels taken from the hurricane barrier in 1978 and for several years thereafter had PCB concentrations higher than any U.S. Coastal site in the 1976, 1977, and 1978 collections of the U.S. EPA Mussel Watch Program (Farrington et al., 1982; 1983; 1985). The recent NOAA National Status and Trends Program verified this finding for the years 1986, 1987, and 1988 (NOAA, 1989b). Two of the three stations in Buzzards Bay were in the top 10 for each year, and one (Angelica Rock) had the highest PCB concentrations for all three years (Figure 18). PCB concentrations in mussels transplanted to the hurricane barrier in New Bedford Harbor during 1984-1985 (Capuzzo et al., 1989) and our earlier data for PCB concentrations in mussels (cited above) exceed the concentrations for the Angelica Rock station by a factor of 3 to 4.

Dr. Jack Schwartz of the Massachusetts Division of Marine Fisheries has recently reported on PCB concentrations in winter flounder, lobster, and quahog from Buzzards Bay (Schwartz, 1988). Samples were taken during 1985 and 1986; station locations and average PCB concentrations (reported on wet- weight basis) for edible portions of tissue from lobsters are presented in Figure 19. Compared to other areas of Buzzards Bay, the outer New Bedford Harbor area has elevated concentrations of PCBs in lobster. Similar patterns were evident for winter flounder, with fish from the New Bedford Harbor area having the highest concentrations (see Appendix). Station locations and concentrations of PCBs (wet-weight basis) for quahogs are presented in Figure 20. The contrast in PCB concentrations between samples from New Bedford Harbor and samples from other areas of Buzzards Bay is generally more pronounced for quahog than for the lobster and winter flounder. These differences may be due to the migratory behavior of winter flounder and lobster; i.e., winter flounder and lobster caught in the New Bedford Harbor area may have migrated there shortly before being caught, and thus might reflect the lower concentrations of PCBs in the water and sediment elsewhere in the Bay. Quahogs on the other hand, are sedentary and should be more representative of the longer term concentrations of PCBs in their habitats.

Nisbet and Reynolds (1984) reported on concentrations of PCBs in common tern eggs in the same study as cited above for the chlorinated hydrocarbon pesticides. High PCB concentrations were noted in samples from Buzzards Bay and attributed to the "industrial effluents" in the New Bedford Harbor area.

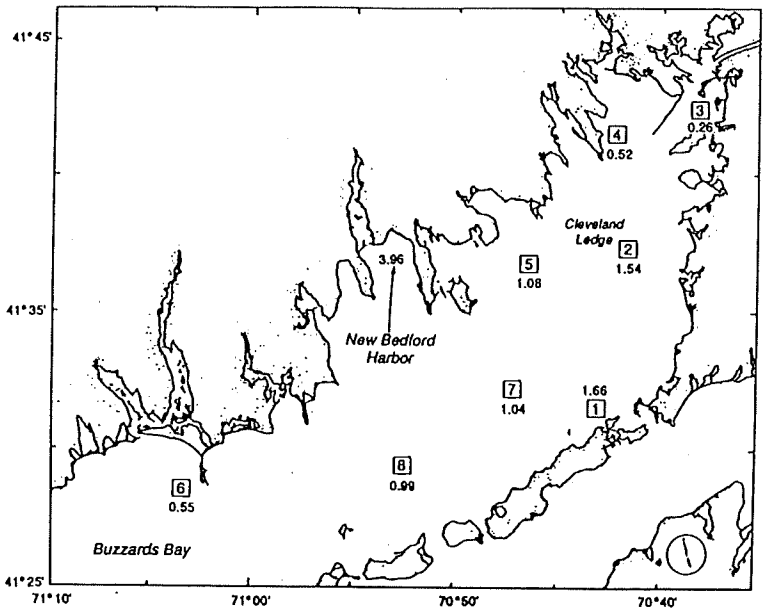


Figure 19. Concentrations of polychlorinated biphenyls in lobster in Buzzards Bay from a survey by the Massachusetts Division of Marine Fisheries (Schwartz et al., 1988).

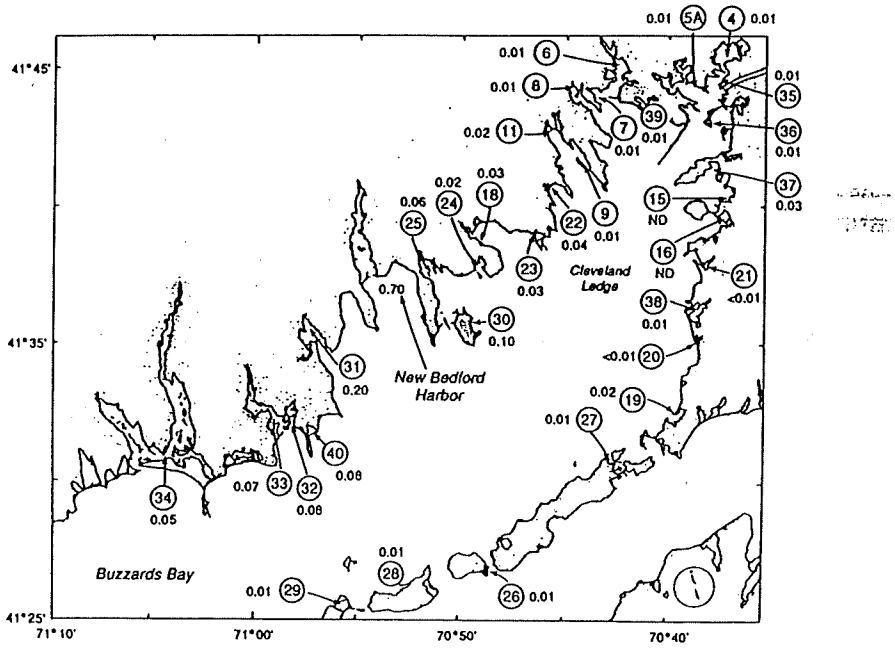


Figure 20. Concentrations of polychlorinated biphenyls in quahog (hard-shell clams) from a survey by the Massachusetts Division of Marine Fisheries (Schwartz etal., 1988).

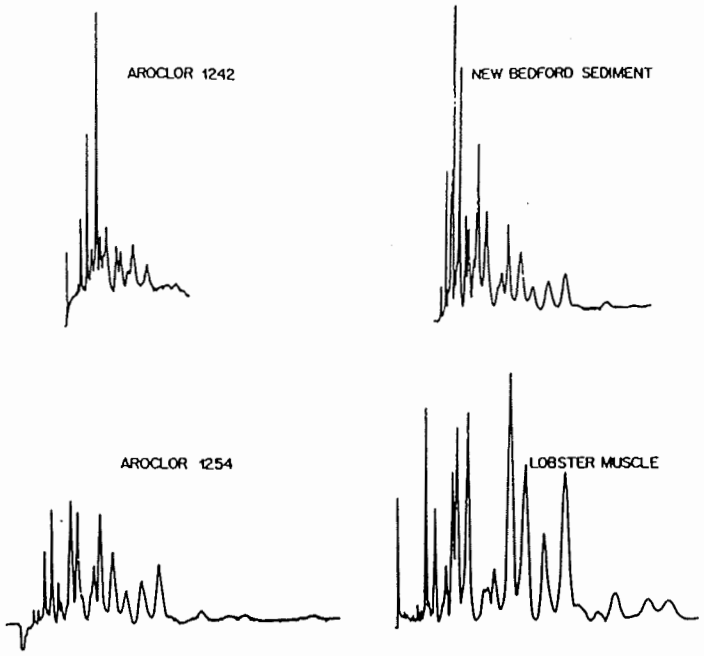


Figure 21. Gas chromatograms from the analysis of polychlorinated biphenyls in selected commercial mixtures and samples from New Bedford Harbor using older methods of packed column gas chromatography (the horizontal axis of the graph is time of elution of the PCBs from the instrument; the height of the peak on the vertical axis is related to the amount of PCB that is present in the sample).

Most analyses of PCBs to date have reported PCB data as the total PCB concentration or as the concentrations of the industrial Aroclor groupings as discussed earlier in this report. These analyses are conducted using the packed-column method of gas chromatography, a technique that has not changed appreciably since 1968. This method could not separate more than 10 or 12 groupings of the 50 to more than 100 individual chlorobiphenyl congeners present in a commercial PCB mixture. A more modern and much higher resolution analytical method has been available since about 1976. This involves a much longer and thinner glass, or fused silica, column in the gas chromatograph with a chemical coating that almost completely separates all individual chlorobiphenyl congeners. The exact details of these analyses are beyond the scope of this report. However, the improved resolution of the second method is evident if we compare the chart-paper outputs produced when the two techniques are used to analyze similar types of samples (Figures 21 and 22). The axis along the bottom (x-axis)of the figures represents time and the peaks that extend up the side axis (y-axis) indicate the presence of groups of chlorobiphenyls. In Figure 21, the peaks are relatively wide and overlap due to the moderate resolution of the packed-column method. In Figure 22, the higher resolution glass capillary method produces peaks that are sharper and more numerous and identifies many more individual chlorobiphenyls. It is also obvious from these figures that the PCBs present in the lobster and crab are of much different composition than the PCBs of the commercial Aroclor mixture and the PCBs found in the sediment and in bivalve tissue (as represented by the mussel sample).

Thus, to understand the present status of PCB pollution in various segments of the Buzzards Bay ecosystem, it is important to realize that the practice of analyzing for total or commercial groupings of PCBs conceals some valuable data about the fates of PCBs. Furthermore, since we know that individual

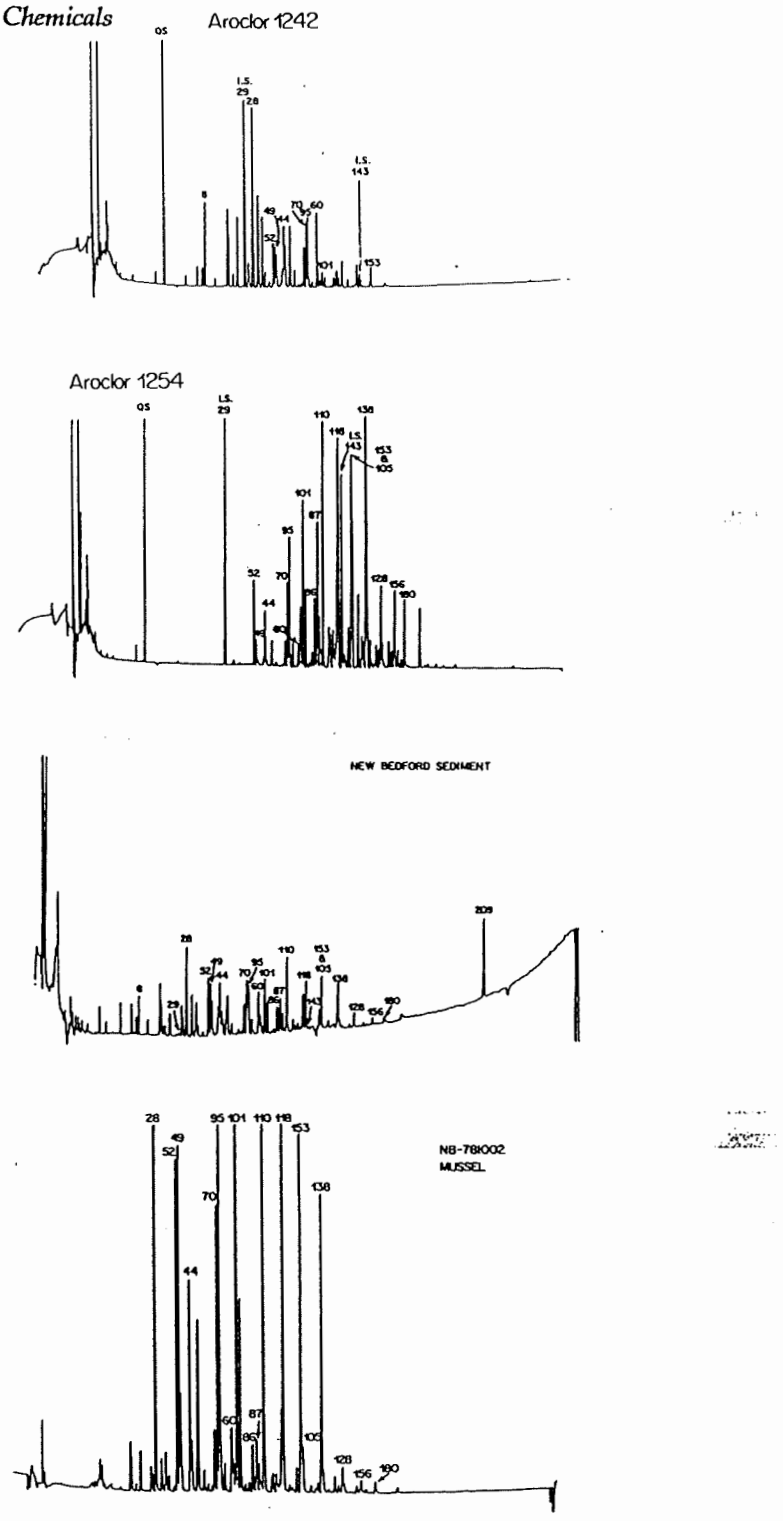


Figure 22. Gas chromatograms from the analysis of polychlorinated biphenyls in selected commercial mixtures and samples from New Bedford Harbor using the newer higher resolution capillary column method of gas chromatography (the graph of axes are the same as for Figure 21; note the greater number of peaks and the sharpness of the peaks, indicating that many of the individual chlorobiphenyls that constitute the PCB mixtures can be measured quantitatively; also note the differences in the pattern of the peaks between some of the samples, clearly indicating that there are major differences in the composition of the PCBs when comparing some of the samples).

chlorobiphenyls have different biological effects, attempts to estimate ecological and human health risks should take into account the different compositions of PCBs as well as the concentration of total PCBs in these samples.

Other Organic Chemical Contaminants

Other classes of organic contaminants may also accumulate in fish and shellfish, but our understanding of their distribution is by far less complete. Industrial contaminants may be discharged to coastal waters and accumulate in biological resources, yet few of these chemicals are monitored or regulated. In Puget Sound, Washington, chlorinated butadienes and other xenobiotic compounds have been detected in sediments (Malins et al., 1984) and, in concert with PAHs and PCBs, have been linked to pathological conditions in English sole (*Parophrys vetulus*). Other contaminants of concern include tributyltin compounds, a major constituent of antifouling paints.

Water

There are very few measurements of chemical contaminants in water samples from Buzzards Bay. Data for trace metals and fossil fuel hydrocarbons are sparse and the methods of sampling and anlysis are not well documented. For PCBs, Brownawell and Farrington (1986), Brownawell (1986) and Farrington et al. (1985, 1986a) report concentrations in waters from Inner and Outer New Bedford Harbor that are in the range of hundreds of nanograms to one microgram per liter. These concentrations are very high for natural waters and are consistent with the other data documenting high concentrations of PCBs in the New Bedford Harbor area.

Summary of the Status of Chemical Contaminants in Buzzards Bay

At present, the problem of chemical contaminants (i.e., trace metals, fossil fuel hydrocarbons, and PCBs) in Buzzards Bay centers on New Bedford Harbor and nearby portions of the western Bay. The Harbor is very contaminated with trace metals and PCBs because industrial discharges during the past five decades have resulted in accumulation of these chemicals in surface sediments. The PCB contamination is so severe that the area has been designated an EPA Superfund site. Chlorinated pesticide contamination appears to have declined over the years, as has been the case for other sections of the U.S. coastline.

Our data on chemical contaminants are limited. We have some measurements for a few chemicals, in a few areas of the Bay — mostly the New Bedford Harbor area. Time-series information that would allow us to keep track of many chemical contaminants Baywide is inadequate. The NOAA Status and Trends Program has provided an initial set of time-series data, but the three to four stations of that program do not provide comprehensive coverage of the Bay. Local problems may thus go unnoticed until some drastic effect causes us to look more closely at a particular location or event.

Scientists have a reasonably good idea of how to monitor the status and trends of chemical contaminants in coastal areas. Using current analytical methods, the effort is very expensive. We estimate that a $600,000 (1990 dollars) annual

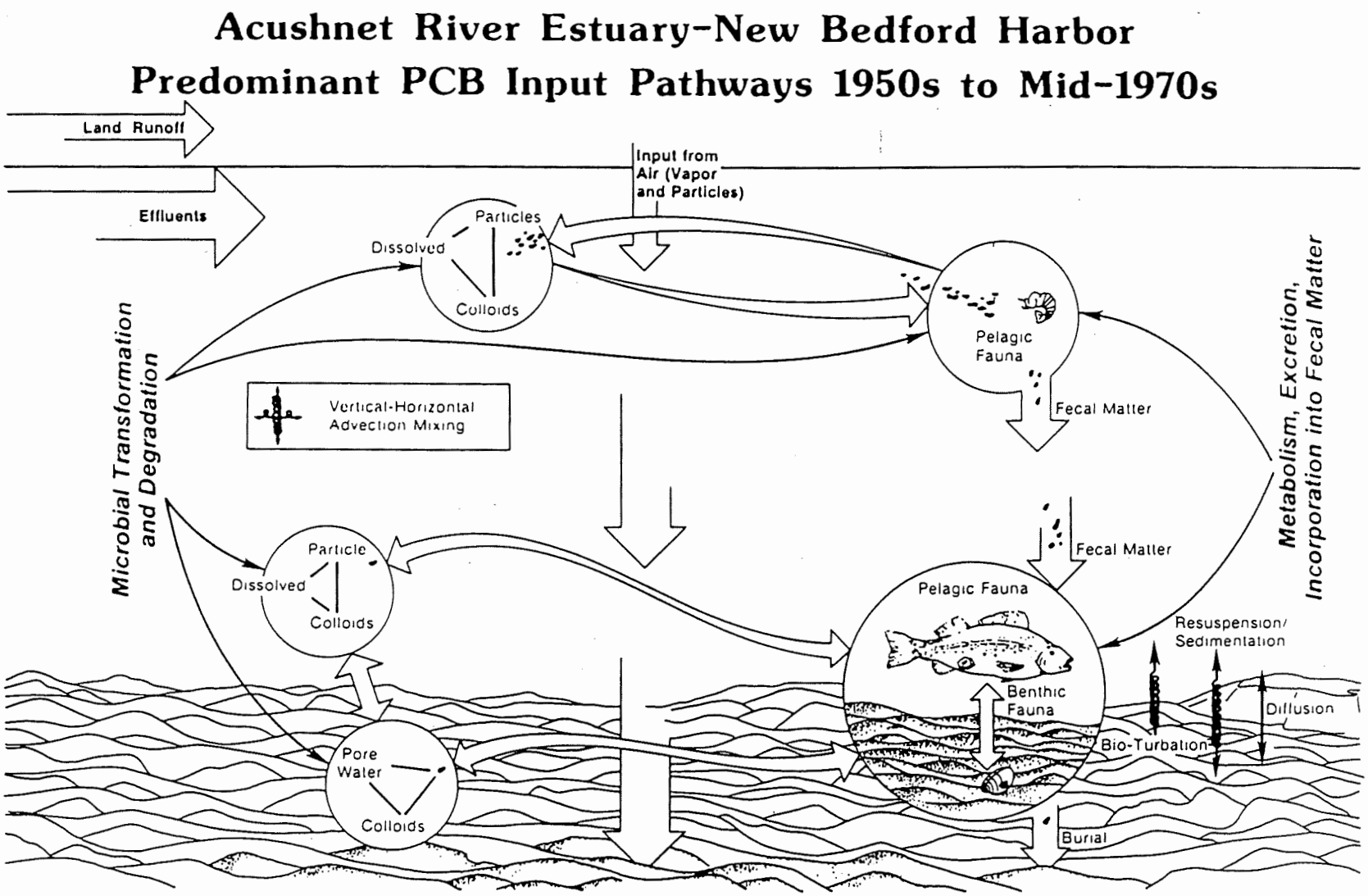


Figure 23. Cartoon of the biogeochemical cycle of polychlorinated biphenyls before the andated reduction in releases of PCBs to the environment of New Bedford Harbor.

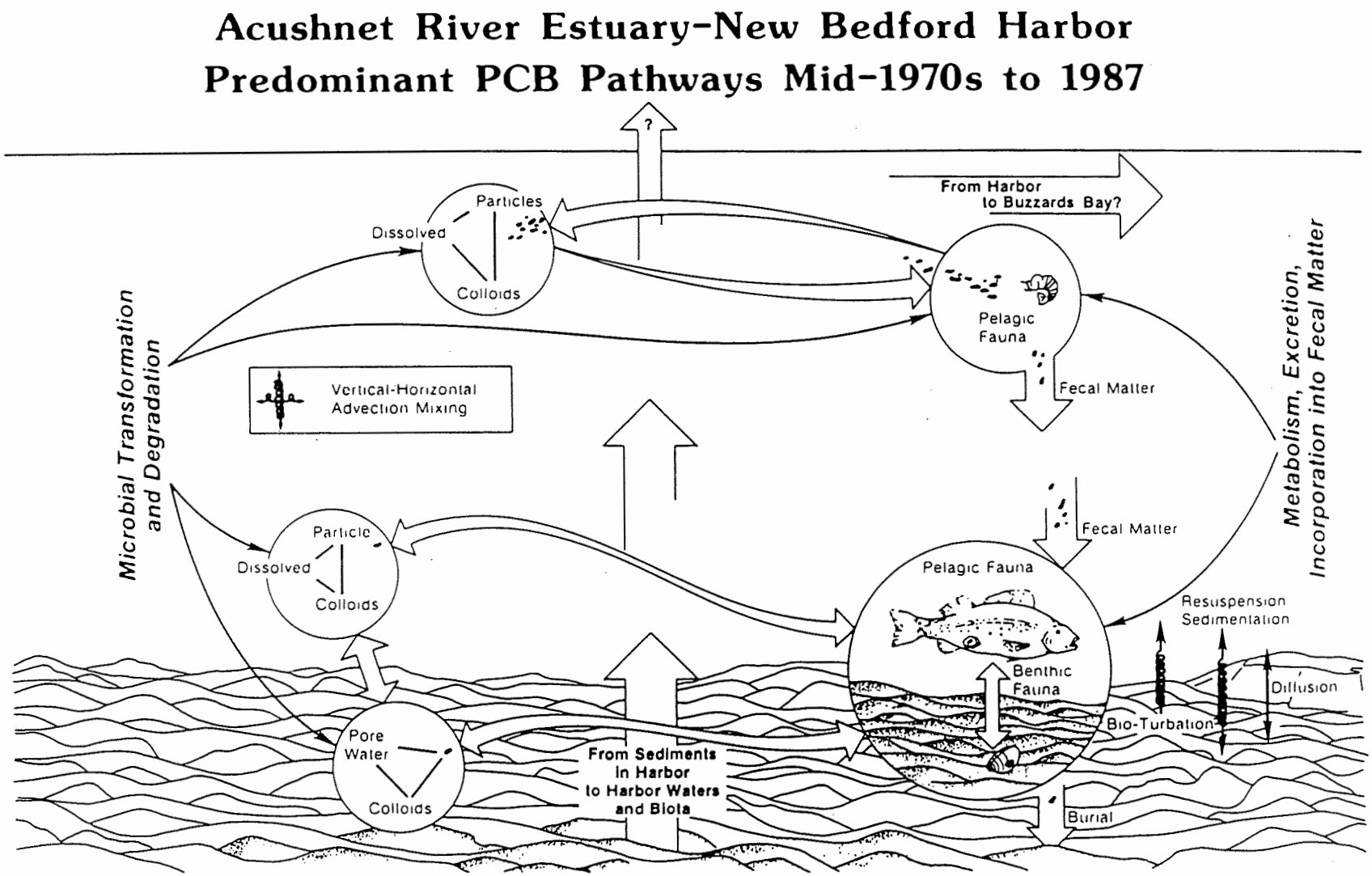


Figure 24. Cartoon of the biogeochemical cycle of polychlorinated biphenyls after the mandated reduction in release of PCBs to the enviornment of New Bedford Harbor.

as TBT in Buzzards Bay. Given the recent concern about effects of TBT-containing bottom paints on shellfish and the importance of shellfish to the Buzzards Bay region economy and recreation, it is unfortunate that no surveys of TBT concentrations have been completed for Buzzards Bay as of the fall of 1990.

Contaminant Fate and Biogeochemical Cycles

An illustration of the biogeochemical cycle for PAHs in the marine environment was presented earlier in this report (Figure 1) to introduce the dynamics of chemical contaminant inputs, transport, and fate in coastal marine ecosystems. Figures 23 and 24 depict the biogeochemical cycle of PCBs in New Bedford Harbor before and after the cessation of releases from industrial effluents and storm sewers. The same processes apply with only minor modifications to other chemical contaminants in New Bedford Harbor (e.g., trace metals and PAHs). PCBs enter the Harbor waters from industrial and sewage discharges and atmospheric deposition. Concentrations in the water are relatively low, in the range of parts per billion to parts per million. PCBs are not very soluble in water and will attach to organic matter-coated mineral and biological particles in the water column. PCBs will also be taken up from the water column by organisms as they respire by passing water across their gill surfaces. Organisms feeding on particles in the water or in sediments will also consume the PCBs that are attached to the nutritious organic matter.

The "partitioning" of PCBs between the water and the particles and between the water and the organisms results in concentration increase of PCBs by several orders of magnitude for the particles and organisms as compared to concentrations in water. Thus, concentrations of parts per billion or even parts per trillion of some PCBs (and tracemetals and PAHs) in water can become concentrations of parts per million in organisms.

Particles contaminated with PCBs settle and become part of the sediment of the Harbor. The organisms living in the sediment feed on the particles and are exposed to PCBs by this route and by exposure to high concentrations of PCBs in the water contained in the sediment — the pore, or interstitial, water. Bottom-dwelling organisms (such as small worms, crabs, and shellfish) are prey for larger organisms such as fish and lobsters. By these two mechanisms (feeding on contaminated prey and uptake across membrane surfaces from the water) PCBs can be passed to larger organisms that become food for people via recreational, subsistence, or commercial fisheries. The interconnections are dynamic and complicated, but we know approximately how they work. We do not know with great accuracy the rate at which they work and how a change in conditions in one compartment cascades through the biogeochemical cycle and changes things in other parts of the system.

The bioaccumulation of organic contaminants may be strongly influenced by chemical factors such as solubility and particle adsorption-desorption kinetics of specific compounds. It is also affected by biological factors such as the transfer of compounds through food chains and the amount of body lipid in exposed organisms (Chiou, 1985; Neff, 1979). Species differ in their rate of uptake due to differences in habitat, trophic status, and physiological condition. Differences

in contaminant concentrations among species from different habitats may be the result of differences in the availability of sediment-bound contaminants and capacity for biotransformation (i.e., the conversion of one chemical to another by action of a biological process). Feeding habits of carnivorous fish will also influence potential contamination.

Once contaminated with chemicals, surface sediments may remain contaminated and a source of these contaminants to the overlying water for many years. The sediment reservoirs in New Bedford Harbor have high concentrations of PCBs, trace metals, and PAHs. Now that the major inputs of toxic chemicals have been curtailed, to what extent are the sediments a source of chemical contaminants to the overlying water and to the living parts of the ecosystem? To what extent are the Harbor sediments a" leaky sink" and a source of continuing contamination from the Harbor to the rest of the Bay? These questions can only be answered in general terms, given the present state of knowledge. However, we do know that natural and man-induced disturbances (e.g., storms, dredging) will resuspend contaminated sediments, rendering them more available for uptake throughout the ecosystem.

Groups of bacteria that live in sediments slowly degrade PCBs, PAHs, and some other chemical contaminants. Although these bacteria may be responsible for altering the composition of contaminants in sediments, bacterial degradation may not be rapid enough to decontaminate the Harbor and Bay area. The focus of the EPA Superfund efforts has been to determine if, how, and when to dredge the most contaminated sediments of the Harbor; and how to treat, contain, and/or dispose of the contaminated sediment. Research investigations and feasibility studies are continuing in an attempt to provide answers that are within the realm of feasible costs. Among the options considered is one to take no action — to do nothing. It is clear, however, that this alternative would perpetuate the present problem for many decades. It would take this long for natural sedimentation to bury the contaminated sediments to such a depth that bioturbation by burrowing organisms and resuspension by storms would not bring the chemical contaminants in contact with surface sediments.

Long-term effects of organic contaminants may be linked to the carcinogenic or mutagenic effects of the biotransformation of specific compounds (Stegeman, 1981). The rate of biotransformation is highly dependent on the chemical structure of the compound and varies from one species of biota to another. This is why we see such differences in the PCB compositions in different species sampled from the same general area of the Harbor. For example, lobster appear to have considerable ability to metabolize PCBs but bivalve shellfish have only minimal capability. This lack of metabolic capability accounts for the close resemblance between the PCBs in mussels and the PCBs in the water and sediment, and the marked difference between the PCB composition in lobsters and the PCBs in the water and sediment reported by Farrington et al. (1986a).

The bioaccumulation of trace metals by marine organisms is also influenced by chemical factors, such as the chemical form and bioavailability of metal, as well as by biological factors, such as physiological condition and the ability of the organism to regulate metal uptake (Phillips, 1977; George, 1982; O'Connor and Rachlin, 1982; Farrington et al., 1983). Marine animals differ in their capacity to

store, remove, and detoxify metal contaminants. Thus, metal content may vary considerably among different species collected from a single location. Metal storage may involve deposition into tissues, skeletal material, and concretions, or within intracellular matrices. Removal may take place through excretion or through the production of particulate products such as feces, eggs, and molts.

Organisms may be classified according to their relative metal-regulating ability; crustaceans and fish have better metal regulating capabilities than molluscs, and some metals (such as copper and zinc) can be regulated to a greater extent than others (such as cadmium, mercury, and lead). Therefore, exposure to heavily contaminated areas may or may not result in increased bioaccumulation of contaminants in commercial fishery resources, depending on the metals and species considered.

Certain metals may be detoxified by binding to metallothionein proteins (Jenkins et al., 1982). These are naturally occurring proteins that are used to regulate metal levels in metabolic reactions; these proteins exert their protective effect by sequestering free metal ions and partitioning them away from potential sites of toxic action. If the binding capacity of metallothioneins is exceeded, toxic effects of metal contaminants may be induced.

Phillips (1980) stressed the importance of understanding the seasonal variability in trace metal burdens of marine organisms if we wish to distinguish between clean and contaminated habitats on the basis of contaminant burdens in commercial resources. In marine bivalve molluscs, natural seasonal variation can account for as much as 15 to 60% of the variability in observed values. National and international monitoring programs for trace metal levels in marine shellfish have demonstrated that trace metal burdens rarely differ by more than a factor of 10 between highly polluted and relatively clean locations (Holden, 1973; Goldberg et al., 1978; Phillips, 1980). Metal concentrations in fish and crustaceans vary even less in response to environmental gradients of metal contamination. This lack of variation is a consequence of the migratory behavior of these species and their metabolic regulation of trace metals. Mercury and selenium, which may accumulate to high levels in finfish and shellfish, are the exception to this observation (Gardner, 1978; MacKay et al., 1975).

For several decades, toxic chemicals have been discharged to Buzzards Bay from a variety of sources. These chemicals have become part of complex biogeochemical cycles and have moved through the ecosystem at varying rates, accumulating usually in fine-grained, organic-rich sediments. We do not know the specific details of the biogeochemical cycling of every contaminant, but we have enough information to begin monitoring the fate and transport of some of the contaminants and to estimate how much exposure marine biota and human consumers of fish and shellfish have to these chemicals.

Biological Effects of Contaminants

Ecological concerns about contamination in the marine environment include changes in species distributions and abundance, alteratons in habitat, and shifts in energy flow and biogeochemical cycles. The toxic effects of chemical contaminants on marine organisms are dependent on the bioavailability and per-

sistence of the chemicals, the ability of organisms to accumulate and metabolize it, and how it interferes with specific metabolic or ecological processes.

The responses of organisms to toxic chemicals can be manifested at four levels of biological organization: (1) biochemical and cellular; (2) organismal, including the integration of physiological, biochemical and behavioral responses; (3) population, including alterations in population dynamics; and (4) community, resulting in community structure and dynamics (Table 1). Biological effects can be manifested at biochemical, cellular, and organismal levels of organization before disturbances at the population level develop (Capuzzo, 1981). All responses are not disruptive and do not necessarily result in degeneration at the next level of organization. Only when compensatory or adaptive mechanisms at one level begin to fail do deleterious effects become apparent at the next level. To compare responses at various levels of biological organization, it is important to learn the degree to which adaptive responses at each of the four levels persist as the concentration of contaminants increases. The initial responses in each case are the triggering of mechanisms to reduce or resist the impact of the toxicant; these mechanisms may include the induction of toxicant-metabolizing processes (at the biochemical level) or the selection of toxicant-resistant forms (at the population level). Adaptive processes are capable of countering disruptive processes until the system reaches a threshold for the toxicant; at this point the adaptive potential is completely overridden by the degeneration imposed on the system by disruptive effects.

To predict the effects of contamination, we must understand the early signs of stress at each level of organization before compensatory mechanisms are surpassed. From the biochemical level to the community level, the degree of system complexity, the number of compensatory mechanisms available, and the lag time to measure a response increase dramatically, thereby increasing the predictive difficulties at each level. Chronic exposure to chemical contaminants can alter the reproductive and developmental potential of populations of marine organisms, resulting in possible changes in population structure and dynamics. It is difficult to ascertain, however, the relationship between chronic responses of organisms to contaminants and large-scale alterations in the functioning of marine ecosystems or the sustainable yield of harvestable species. Cairns (1983) argued that our ability to detect toxic effects at higher levels of biological organization is limited by the lack of reliable predictive tests at population, community, and ecosystem levels. Much research is needed in these areas before we can adequately address environmental hazards that result from contamination. Koojiman and Metz (1985) suggested that the sublethal effects of contaminant exposure should be interpreted in light of the survival probabilities and reproductive success of populations, thus bridging the gap between individual and population responses. Although a wide range of sublethal stress indices have been proposed, few have been linked to the survival potential of the individual organism or the reproductive potential of the population. Experimental studies directed at determining effects on energy metabolism or effects that influence growth and reproduction would be most appropriate for linking effects at higher levels of organization.

When investigating biological effects of contaminants, many variables must be recognized and assessed. Differential sensitivity of different species of or-

ganisms, of various life history stages, and of species from different habitats may be related to the bioavailabilty of the contaminant, the capacity for the biotransformation of the contaminant, and the metabolic consequences of exposure to the contaminant. The increased sensitivity of early developmental stages and the seasonal differences in the responses of adult animals may be related to stage-specific or seasonal dependency on particular metabolic processes (e.g., storage and mobilization of energy reserves, hormonal processes), and may alter developmental and reproductive success (Capuzzo, 1987).

Recent studies of the incidence of tumors and other histopathological disorders in demersal fish from the Duwamish River, near Seattle, Washington, Boston Harbor, Massachusetts, the Hudson-Raritan Estuary (New York-New Jersey), Southern California, New York Bight, German Bight, and Danish coastal waters have suggested a possible link between contaminant levels and increased incidence of histopathological conditions (McCain et al., 1978; Smith et al., 1979; Christensen, 1980; Dethlefsen, 1980; Sindermann et al., 1980; Stegeman, 1981; Perkins et al., 1982; Murchelano and Wolke, 1985). In New Bedford Harbor, increased prevalence of hematopoietic neoplasia (leukemia) in soft-shelled clams (Reinisch et al., 1984; Leavitt et al., 1990) and black gill and shell disease in lobsters (Estrella, 1984) have been reported, but chemical contamination has not been clearly shown to be the cause.

In addition to histopathological damage, sublethal toxic effects of organic contaminants in marine animals include impairment of physiological processes that may alter the energy available for growth and reproduction; other effects on reproductive and developmental processes; and direct genetic damage. Biological effects of contaminants have been attributed to the uptake of specific compounds and/or their metabolites, rather than to the total body burden of contaminants (Capuzzo et al., 1984; Widdows et al., 1982; Malins and Hodgins, 1981). Empirical data suggest that linkages clearly exist between (l) developmental and reproductive abnormalities; (2) the physiological and molecular processes involved in uptake, retention, and loss of contaminants; and (3) the toxicity and/or transformation of contaminants (Capuzzo et al., 1988). An understanding of reproductive and developmental processes provides the critical link between responses to contaminants at the organismal and suborganismal levels and population consequences. Alterations in bioenergetics linked with observations of reduced fecundity and viability of larvae, abnormalities in gamete and embryo development, and reduced reproductive success provide a strong empirical basis for examination of population responses.

Although there are numerous laboratory studies on the effects of contaminants on marine organisms, there have been relatively few field studies in Buzzards Bay. Sanders et al. (1980) conducted one of the most extensive studies to date on the effects of an oil spill on benthic communities. They examined the effects of the spill of 650,000 liters of No. 2 fuel oil from the barge *Florida* on shallow-water benthic communities in Buzzards Bay. Changes in faunal composition of benthic communities were linked to the duration and severity of hydrocarbon exposure, with the most severe impacts observed in the enclosed embayment of Wild Harbor. Recovery of benthic populations correlated with the disappearance/degradation of hydrocarbons in soft sediments. For example, Krebs and Burns (1977) observed changes in fiddler crab (*Uca pugnax*) populations for

seven years after the spill, including long-term reductions in recruitment and population density; and changed in female:male ratios of adult crabs; behavioral aberrations; and high overwintering mortality. Recovery of crab populations was correlated with the disappearance of naphthalenes and alkylated naphthalenes from contaminated sediments.

Recent studies in Buzzards Bay and New Bedford Harbor have examined bioavailability, bioconcentration, and biological effects of lipophilic organic contaminants on marine animals. These local studies add to our understanding of the complex relationship between observed biological effects and contaminant distributions in the ecosystem. Capuzzo et al. (1989; in preparation) examined the effects of PCBs, PAHs, and trace metals on populations of *Mytilus edulis* transplanted to New Bedford Harbor for one year (November 1984 to November 1985); studies were directed at understanding how bioenergetic, biochemical, and histological changes relate to chemical data on body burdens of specific compounds and at assessing how seasonal variability in contaminant distributions is related to reproductive and metabolic activities. Estimates of bioenergetics (condition indices and scope for growth) for mussels at New Bedford Harbor are lower than values measured from mussels from other stations during the pre-spawning period, but are equal after spawning when values for mussels at all stations decline. Scope-for-growth measurements are positive during early stages of the reproductive cycle (November - December) and just prior to spawning as populations take advantage of the spring plankton bloom. Values are negative during the spawning period, indicating that energy is being expended in the production and release of gametes. The components of the energy budget that most strongly influenced the decline in scope for growth observed among mussels at the New Bedford Harbor station were a decrease in the amount of carbon ingested and assimilated and an increase in respiratory expenditures.

Bioenergetic parameters measured for mussels from New Bedford Harbor during the pre-spawning period may have been influenced by lower molecular weight hydrocarbons associated with a spill of No. 2 fuel oil. Donkin and Widdows (1986) suggested that reductions in feeding rates were associated with the narcotizing properties of lower molecular weight hydrocarbons, such as naphthalenes and phenanthrenes. The rapid loss of these compounds prior to spawning, however, would suggest that their toxic effects are probably short-lived and that sustained effects on bioenergetics may be the result of exposure to multiple classes of contaminants. Bioenergetic estimates of reproductive effort also indicate a significant reduction for mussels from the New Bedford Harbor station. This reduction appears to be the result of both diminished allocation of energy to gamete production and degeneration and premature resorption of oocytes.

Black et al. (1988) examined the distribution of PCBs in eggs from winter flounder *(Pseudopleuronectes americanus)* collected in Buzzards Bay and Narragansett Bay. Eggs collected from winter flounder collected in New Bedford Harbor had PCB levels of 39.6 ppm (dry weight), and larvae that hatched from these eggs were significantly smaller than those hatched from eggs collected at relatively uncontaminated sites. The PCB concentrations in winter flounder eggs from New Bedford Harbor were much higher than concentrations ob-

served by von Westernhagen et al. (1981) to cause reductions in viable hatch in Baltic flounder *(Platichthys flesus).* Stegeman and his colleagues are examining the biochemical basis for response to PCBs and their interference with hormonal control of reproduction and gonadal function, as well as metabolic processes. Of the various congeners of PCBs, only those with a coplanar form are capable of inducing cytochrome P-450E (Elskus et al., 1989; Hahn et al., 1989). The distribution of these congeners and coplanar congeners of polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) may be extremely important in eliciting biochemical and reproductive responses in marine fishes. Norwood et al. (1989) and Pruell et al. (1990) found New Bedford sediments to contain high concentrations of not only PCBs but also PCDDs and PCDFs. The significance of these contaminants as environmental toxicants in Buzzards Bay is still being explored.

Sediment bioassays on a gradient of sediment samples from New Bedford Harbor have been conducted using the infaunal amphipod *Ampelisca abdita* and settling larvae of the oysters *Crassostrea virginica* and *C. gigas* (K.J. Scott, unpub. data; Warner et al., ubpub. data); acute LC50 values ranged from 13 to 16 ppm dry weight total PCB. Using these data and the ratio of acute:chronic effects for *A. abdita* (15) one can approximate an apparent no effects level (NEL) of 1 ppm. This is similar to provisional sediment criteria based on an apparent effects threshold (AET) of 1.1 to 2.5 ppm.

Human Health Concerns

The transfer of toxic chemicals through marine food chains can result in bioaccumulation in commercial fishery resources and transfer to the human consumer. Of specific concern is the uptake and transfer of metals, halogenated hydrocarbons, and other organic contaminants including petroleum hydrocarbons derived from accidental oil spills, municipal discharges, and urban runoff. Contaminants that demonstrate mutagenic, carcinogenic, or teratogenic potential are of particular concern because they pose direct threats to human health.

Chemical contamination of fishery resources has recently led to fishery closures in several areas of the U.S. coastline (Capuzzo et al., 1987). For example, striped bass fisheries in New York and Rhode Island were closed in 1986 as a result of PCB contamination; the State of California developed health advisories warning the public against frequent consumption of fish caught in southern California waters; and the Department of Public Health in the Commonwealth of Massachusetts has issued a state-wide advisory against the consumption of lobster tomalley (hepatopancreas) because of its exceedingly high levels of PCBs and other contaminants. In Buzzards Bay, approximately 28 square miles are closed to finfishing and shellfishing as a result of PCB contamination (Figure 17). These recent actions illustrate a growing concern for the impact of chemical contamination on resources in coastal waters. Concern about coastal environmental degradation and its impact on fishery resources is particularly critical in southern New England, where fishing has historically been of great economic significance. Landings from both commercial and recreational fisheries are important.

Human health risks from the consumption of contaminated fish and shellfish can be interpreted on the basis of either (1) Accepted Daily Intake (ADI) or Provisional Tolerable Weekly Intake (PTWI), as recommended by the World Health Organization - Food and Agriculture Organization (WHO-FAO) or (2) accepted concentrations based on an average per capita consumption of seafood products, as recommended by the U.S. FDA. The two concepts differ slightly in the acceptance of a threshold (acceptable concentration) or non-threshold (PTWI) basis for environmental concern. Both are derived by applying various uncertainty factors to appropriately selected exposure levels from studies with humans or animals. U.S. FDA regulatory limits are established by considering not only human health risks but also economic factors, including the economic hardship imposed by adherence to stricter health standards and the benefits derived from the use of a specific chemical (e.g., pesticide application in agriculture).

Lifetime cancer risks and non-cancer risks from consumption of contaminated resources can also be estimated (Connor, 1984, 1989; U.S. EPA, 1988). The first is derived from the use of carcinogenic potency factors (CPF) to estimate the implied finite risk of cancer at various doses of a specific chemical. The second is derived from determination of a reference dose expected to produce adverse health effects. Each assessment has a high degree of uncertainty as a result of the need to extrapolate data from animal feeding studies and the scarcity of human toxicological data (Connor, 1984, 1989). The greatest human health risk is derived from the consumption of nearshore and estuarine species harvested from contaminated habitats; these species include contaminated shellfish and migratory estuarine fishes such as bluefish and striped bass. The U.S. FDA estimates that the largest burden of chemical contaminants to the seafood consumer is derived from these sources, yet only one-third of the average U.S. per capita seafood consumption is from estuarine species (6.5 grams / 18.7 grams total per day for a 70 kg adult; U.S. FDA, 1982a, 1982b).

Table 6. Standards for toxic contaminants in seafood

Contaminant	WHO-FAO PTWI mg/kg body wt.	µg/g eq.	U.S. FDA Action Levels µg/g wet wt.
Cadmium	0.0067-0.0083	3.6-4.4	-
Lead	0.05	26.7	-
Mercury	0.005	2.7*	1.0
Methyl Mercury	0.0033	1.8	-
DDT/metabolites	0.0035	1.9	5.0
Heptachlor H.epoxide	0.0035	1.9	0.3
Endrin	-	-	0.3
Aldrin/Dieldrin	-	-	0.3
Chlordane	-	-	0.3*
Hexachlorobenzene	-	-	0.6 (ADI)
Kepone	-	-	0.3
Mirex	-	-	0.1*
PCBs	-	-	2.0
Toxaphene	-	-	5.0*

µg/g equivalents for U.S. population based on average per capita seafood consumption of 18.7 grams of seafood per 70 kg adult per day.

* indicates for fish only.

Exposure standards for human health protection exist for only a few contaminants. Thus, no regulatory action can be taken for most of the contaminants discussed in this report. Action levels issued by the U.S. FDA and PTWI issued by the WHO-FAO are presented in Table 6. The two sets of standards can be compared for the U.S. population by using an average per capita consumption of 18.7 g of seafood products per person of 70 kg weight per day. Standards do not exist for many other potentially toxic and carcinogenic organic contaminants, particularly those that may accumulate in commercial resources following oil spills or those for which few analytical data are available. A fishery may be closed as a result of tainting with oil, but a more well-defined approach to dealing with oil contamination is needed.

In addition to action levels, the U.S. FDA, through the National Shellfish Sanitation Program (NSSP) has recommended alert levels for several trace metal contaminants. These levels are not based on human health/epidemiological concerns but were developed to provide a baseline of background concentrations for individual species and to be used in evaluation of shellfish growing/harvesting areas. Baseline values have been determined in surveys of shellfish species from representative areas of the U.S. coast. Considerable variation in baseline values is noted among different species and within the same species from different geographical regions and from different salinity regimes. Alert levels for *Crassostrea, Mercenaria,* and *Mya* from the Northeast region of the U.S. are presented in Table 7. Values in excess of these levels indicate that further investigation of the nature of increases in metal burdens is warranted, but do not imply a public health risk.

Another approach in evaluating contaminated resources is to consider the level of contamination in fish and shellfish in relation to contaminant levels in discharged effluents. Water quality criteria, issued by EPA for regulation of contaminant input, also consider the potential for bioaccumulation of individual contaminants in edible resources. If properly implemented, these criteria can be used to prohibit further contamination of coastal resources.

Table 7. National Shellfish Sanitation Program Alert Levels

Metal	Oyster µg/g wet wt.	Hard Shell Clam	Soft Shell Clam
Cadmium	3.5	0.5	0.5
Lead	2.0	4.0	5.0
Chromium	2.0	1.0	5.0
Zinc	2000	65	30
Copper	175	10	25

Recommendations for Future Monitoring Programs

Monitoring programs for measuring chemical contaminants in fish and shellfish species should be designed and executed to provide meaningful information on (1) spatial distribution of contaminants; (2) temporal variability in contaminant distributions, as a result of both natural variability and changes in

chemical-use patterns or pollution abatement; and (3) the relationship of contaminant inputs to ecological consequences, including habitat alterations of valuable resources and human health concerns. Current state and federal monitoring efforts, however, fall short of meeting these goals. In general, the frequency of sampling is too limited — both on temporal and spatial scales — to meet these goals.

At present, the Commonwealth of Massachusetts has no comprehensive statewide monitoring program. Development of such a program is being considered in response to issues raised in a white paper issued in November 1985 on the status of fishery resources (Massachusetts Division of Marine Fisheries, 1985) and to growing concern about coastal degradation. Specific classes of contaminants, including trace metals and PCBs, are monitored on a regular basis by the DMF, at least within specific coastal regions or in specific commercial resources. The Division of Water Pollution Control also spot checks trace metals and PCBs, as part of the water quality component of its basin management plans. In addition, a pesticide monitoring program has been initiated by DWPC for evaluation of pesticide use and discharge from cranberry bogs on Buzzards Bay.

On the federal level, two large-scale monitoring efforts have included Buzzards Bay or New Bedford Harbor as sampling sites: (1) the U.S. EPA Mussel Watch Program conducted from 1976 to 1978; and (2) NOAA's National Status and Trends Program conducted from 1984 to the present. Although these programs have provided a broad regional approach to understanding contaminant distribution, they have not provided detailed evaluations of specific local contaminant problems and should not be viewed as replacements for state wide monitoring efforts.

Ecological effects of contaminants in coastal environments include impairment of feeding, growth, development, and recruitment of living resources . These impairments may alter the reproductive and developmental success and change community structure and dynamics. The human health concerns resulting from contaminated resources are obvious. Yet, it is difficult to ascertain the relationship between chronic responses of organisms to contaminated habitats and large-scale alterations in the functioning of marine ecosystems and large-scale contamination of fishery resources. The sensitivity of early developmental stages, the impairment of reproductive processes, and the long-term effects on populations suggest that chronic exposure to many contaminants may certainly alter the dynamics of populations, including populations of valuable commercial resources.

The environmental objectives of monitoring must be defined before a monitoring program is initiated. For example, keeping track of contaminant levels in commercial resources alone provides a very incomplete picture of environmental degradation. On the other hand, if the primary goal is to define the level of a particular contaminant in edible resources (e.g., PCBs in bluefish), then a simple monitoring program based on analyses of market samples is perfectly adequate. To understand long-term impacts of contamination in coastal areas, it is important to understand the conditions under which contaminants persist in benthic environments, the bioavailability of contaminants to commercial

resources, and the sublethal effects of contaminants that lead to reduced growth, delayed development, and reduced reproductive effort, with resulting impacts on population stability. The synergistic effects of complex chemical mixtures must be understood if contaminant impacts are to be predicted realistically. None of these parameters are well understood and any monitoring program should be closely linked to existing research efforts.

A comprehensive monitoring program requires understanding of (1) the physical processes (flow characteristics) that influence the partitioning of contaminants between sediments and water; (2) the chemical processes that influence the availability, persistence, and degradation of contaminants in sediments and water; and (3) the long-term biological effects that alter population stability and function and the consequences of such effects on resource utilization. The first two aspects are important in establishing realistic exposure scenarios — both in time and space — and the third is important in linking ecological effects to concerns about contamination of resources.

In designing such a monitoring program for evaluating human health risks and coastal environmental degradation, the following components must be considered:

- Define the sources of contamination — both point and nonpoint sources — and the degree to which those sources can be controlled. Without this knowledge, attempts to redue contaminant input through recycling or changes in patterns of use cannot be targeted effectively.
- Determine the persistence, degradation rates, and biogeochemical cycling of contaminants within coastal marine sediments and the flux of those contaminants between sediments, water, and organisms.
- Relate contaminant content of sediments to ecological changes and extent of contamination of commercial resources.
- Expand market-basket surveys (i.e., samples from fishery harvests) to provide a comprehensive evaluation of the temporal and spatial extent of chemical contamination in fish and shellfish resources.
- Using controlled populations (e.g., indigenous bivalve populations or demersal fish populations with little migratory behavior), define seasonal patterns in contaminant concentrations and their relationship to reproductive activity and/or aberrations in physiological condition.
- Improve analytical methodology and experimental design of monitoring efforts to better detect temporal and spatial trends.

Such an approach would lead to a better understanding of the causal relationship between contaminant inputs and environmental degradation in coastal waters and allow the development of predictive approaches to marine environmental monitoring.

Summary and Overview

A pessimist will read our report and be concerned about the lack of information concerning several aspects of toxic chemical pollution in the Bay. The pessimist will also be appalled at the snail's pace of progress in rectifying the pollution problems already identified as significant and in need of remedial action. An optimist, however, will be thankful for the knowledge that we do have and will look forward to accelerated progress in correcting existing pollution problems.

We have documented the limits of our knowledge in comparison to the complexity of what we seek to know about the sources, inputs, fates, and effects of toxic chemicals in the Buzzards Bay ecosystem. Buzzards Bay, like many other coastal areas in modern industrialized nations, receives toxic pollutants primarily in small amounts from the everyday sloppy use of chemicals. We have a reasonable qualitative understanding of the way chemicals move through the Buzzards Bay ecosystem, their ultimate fate, the means by which marine organisms are exposed to these chemicals, and how humans, through the consumption of the edible living resources of the Bay, again come into contact with these chemicals. A quantitative understanding of these biogeochemical cycles is evolving from continuing research efforts in Buzzards Bay and other coastal ecosystems.

We know effects of toxic chemicals on marine organisms and on human health only at the most rudimentary level — the point at which toxicity becomes acute. Long-term chronic effects on marine organisms and ecosystems are largely unknown and are the topics of numerous research programs. Research of the past few decades has taught us that our knowledge is limited compared to what we need to know to adequately predict risks to ecological systems and human health from exposure to toxic chemicals.

We have already seen that the activities of modern society can affect the Bay and its resources. Only through vigorous pursuit of additional knowledge can we correct past mistakes and enter into a phase of truly wise management of this magnificent natural resource — Buzzards Bay.

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References

Baxter, B., P.C. Deslauriers, and J. Morson. 1978. The *Bouchard* #65 Oil Spill, January 1977. NOAA Environmental Research Laboratories, MESA. MESA Special Report. U.S. National Oceanic and Atmospheric Administration, U.S. Department of Commerce, Rockville, MD, 80 pp. plus appendices.

Black, D.E., D.K. Phelps, and R.L. Lapan. 1988. The effect of inherited contamination on egg and larval winter flounder, *Pseudopleuronectes americanus*. Marine Environmental Research 25: 45-62.

Blumer, M. and W.W. Youngblood. 1974. Polycyclic aromatic hydrocarbons in soils and sediments. Science 188: 53-55.

Blumer, M., G. Souza, and J. Sass. 1971. Hydrocarbon pollution of edible shellfish by an oil spill. Marine Biology 5: 195-202.

Brown, B., T. Stenner, and J. Gale. 1987. Identification and Collection of Historical Data for Buzzards Bay, Massachusetts, Task II — Summary of Existing Literature. Draft Report to the U.S. Environmental Protection Agency, Region I, by Battelle Ocean Sciences, Duxbury, MA, 142 pp.

Brownawell, B.J. 1986. The Role of Colloidal Organic Matter in the Marine Geochemistry of PCBs. Ph.D. Dissertation, Woods Hole Oceanographic Institution-Massachusetts Institute of Technology Joint Program in Oceanography and Ocean Engineering, Woods Hole Oceanographic Institution, Woods Hole, MA.

Brownawel, B.J. and J.W. Farrington 1985. Partitioning of PCBs in marine sediments. In Marine and Estuarine Geochemistry. eds. A.C. Sigleo and A. Hattori. pp 97-120 Lewis Publishers Chelsea, MI.

Brownawell, B.J. and J.W. Farrington. 1986. Biogeochemistry of PCBs in interstitial waters of a coastal marine sediment. Geochimica et Cosmochimica Acta 50: 157-169.

Bruland, K.W., K. Bertine, M. Koide, and E. Goldberg. 1974. History of metal pollution in the southern California coastal zone. Environmental Science and Technology 8: 425-432.

Burns, K.A. 1975. Distribution of Hydrocarbons in a Saltmarsh Ecosystem after an Oil Spill and Physiological Changes in Marsh Animals from the Polluted Environment. Ph.D. Dissertation, Woods Hole Oceanographic Institution-Massachusetts Institute of Technology Joint Program in Oceanography and Ocean Engineering, Woods Hole Oceanographic Institution, Woods Hole, MA, 101 pp.

Burns, K.A. and J.M. Teal. 1971. Hydrocarbon Incorporation into the Salt Marsh Ecosystem from the West Falmouth Oil Spill. Technical Report, WHOI 71-69, Woods Hole Oceanographic Institution, Woods Hole, MA, 14 pp.

Burns, K.A. and J.M Teal. 1979. The West Falmouth oil spill: Hydrocarbons in the salt marsh ecosystem. Estuarine and Coastal Marine Science 8: 349-360.

Butler, P.A. (1973). Organochlorine residues in estuarine mollusks, 1965-72. National Pesticide Monitoring Program. Pesticide Monitoring Journal 6 (4): 238-362.

Butler, G. ed. 1978. Principles of Ecotoxicology, SCOPE 12. John Wiley and Sons, New York.

Butman, B., R. Signell, P. Shoukimass, and R.C. Beardsley. 1988. Current Observations in Buzzards Bay, 1982-1986, Data Report. U.S. Geological Survey, U.S. Department of the Interior, Woods Hole, MA, 211 pp.

Cairns, J. 1983. Are single species toxicity tests alone adequate for estimating environmental hazard? Hydrobiologia 100: 47- 57.

Camp Dresser and McKee, Inc. 1979. Report for the City of New Bedford, Massachusetts, Section 301 (h) Application for Modification of Secondary Treatment Requirements for Discharges into Marine Waters, Vol 2. Camp Dresser and McKee Inc., Boston, MA, various paginations.

Capuzzo, J.M. 1981. Predicting pollution effects in the marine environment. Oceanus 24 (1): 25-33.

Capuzzo, J.M. 1987. Biological effects of petroleum hydrocarbons: Assessments from experimental results. In: D.F. Boesch and N.N. Rabalais (eds.), Long-Term Environmental Effects of Offshore Oil and Gas Development. Elsevier Applied Science, London, pp. 343-410.

Capuzzo, J. McDowell, J.W. Farrington, P. Rantamaki, C.H. Clifford, B.A. Lancaster, D.F. Leavitt, and X. Jia. 1989. The relationship between lipid composition and seasonal differences in the distribution of PCBs in *Mytilus edulis L.* Marine Environmental Research 28: 259-264.

Capuzzo, J. McDowell, M.N. Moore, and J. Widdows. 1988. Effects of toxic chemicals in the marine environment: Predictions of impacts from laboratory studies. Aquatic Toxicology 11: 303-311.

Capuzzo, J. McDowell, A.E. McElroy, and G. Wallace. 1987. Fish and Shellfish Contamination in New England Waters: An Evaluation and Review of Available Data on the Distribution of Chemical Contaminants. Report for the Coast Alliance, Washington, DC, 59 pp plus appendices.

Capuzzo, J.M., B.A. Lancaster and G.C. Sasaki. 1984. The effects of petroleum hydrocarbons on lipid metabolism and energetics of larval development and metamorphosis in the American lobster (*Homarus americanus* Milne Edwards). Marine Environmental Research 14: 201-228.

Carson, R.L. 1962. Silent Spring. Houghton Mifflin, Boston.

CEQ. 1984. Environmental Quality 1983, 14th Annual Report of the Council on Environmental Quality, Executive Office of the President of the United States, Washington, DC.

Chiou, C.T. 1985. Partition coefficients of organic compounds in lipid-water systems and correlations with fish bioconcentration factors. Environmental Science and Technology 19: 57-62.

Christensen, N.O. 1980. Diseases and anomalies in fish and invertebrates in Danish littoral regions which might be connected with pollution. In: A.D. McIntyre and J.B. Pearce (eds.), Biological Effects of Marine Pollution and the Problems of Monitoring. Rapports et Proces-Verbaux des Reunions, Vol. 179. Conseil International pour L'Exploration de la Mer, Copenhagen, pp. 103-109.

Commonwealth of Massachusetts. 1982. Acushnet River Estuary PCB Commission, Status Report. Office of the Governor of Massachusetts, State House, Boston, MA, 35 pp. plus appendices.

Connor, M.S. 1984. Comparison of the carcinogenic risks from fish vs. groundwater contamination by organic compounds. Environmental Science and Technology 18: 628-630.

Connor, M.S. 1989. Estimating the public health risk of organic carcinogens in fish. In: M. Champ and P.K. Park (eds.), Oceanic Processes in Marine Pollution, Volume 3, Marine Waste Management: Science and Policy. Robert W. Kreiger Publishing Co., Malabar, FL, pp. 221-229.

Dethlefsen, V.D. 1980. Observations on fish diseases in the German Bight and their possible relation to pollution. In: A.D. McIntyre and J.B. Pearce (eds.), Biological Effects of Marine Pollution and the Problems of Monitoring. Rapports et Proces-Verbaux des Reunions, Vol. 179. Conseil International pour L'Exploration de la Mer, Copenhagen, pp. 110-117.

Donkin, P. and J. Widdows. 1986. Scope for growth as a measurement of environmental pollution and its interpretation using structure-activity relationships. Chemistry and Industry 21: 732-737.

Edington, D.N. and J.A. Robbins. 1976. Records of lead deposition in Lake Michigan since 1800. Environmental Science and Technology 10: 266-274.

Eganhouse, R.P. and I.R. Kaplan. 1981. Extractable organic matter in municipal wastewaters. 1. Petroleum hydrocarbons: Temporal variations and mass emission rates to the ocean. Environmental Science and Technology 16: 180-186.

Elskus, A.A., J.J. Stegeman, L.C. Susani, D. Black, R.J. Pruell, and S.J. Fluck. 1989. Polychlorinated biphenyls concentration and cytochrome P-450E expression in winter flounder from contaminated environments. Marine Environmental Research 28: 25-30.

Estrella, B.T. 1984. Black Gill and Shell Disease in American lobster (*Homarus americanus*) as Indicators of Pollution in Massachusetts Bay and Buzzards Bay, Massachusetts. Massachusetts Division of Marine Fisheries Technical Report #14049-19-125-5-85-C.R., Boston, MA.

Farrington, J.W. 1989. Bioaccumulation of hydrophobic organic pollutant compounds. In: S.A. Levin, M.A. Harwell, J.R. Kelly, and K.D. Kimball (eds.), Ecotoxicology: Problems and Approaches. Springer-Verlag, New York, pp. 279-313.

Farrington, J.W. 1983. Bivalves as sentinels of coastal chemical pollution: The mussel (and oyster) watch. Oceanus 26(2): 18-30.

Farrington, J.W. and J.G. Quinn. 1973. Petroleum Hydrocarbons and Fatty Acids in Wastewater Effluents. Journal Water Pollution Control Federation 45: 704-712.

Farrington, J.W. and S.G. Wakeham. 1980. Hydrocarbons in contemporary aquatic sediments. In: R.A. Baker (ed.), Contaminants and Sediments, Volume 1. Ann Arbor Science Publishers, Inc. Ann Arbor, MI, pp. 3-32.

Farrington, J.W., N.M. Frew, P.M Gschwend, and B.W. Tripp. 1977. Hydrocarbons in cores of northwestern Atlantic coastal and continental margin sediments. Estuarine and Coastal Marine Science 5: 793-808.

Farrington, J.W., R.W. Risebrough, P.L. Parker, A.C. Davis, B. deLappe, J.K. Winters, D. Boatwright, and N.M. Frew. 1982. Hydrocarbons, Polychlorinated biphenyls, and DDE in Mussels and Oysters from the U.S. Coast, 1976-1978. The Mussel Watch. Technical Report No. WHOI 82-42, Woods Hole Oceanographic Institution, Woods Hole, MA, 106 pp.

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, DDT, PCB, hydrocarbon and artificial radionuclide data. Environmental Science and Technology 17: 490-496.

Farrington, J.W., B.W. Tripp, A.C. Davis, and J. Sulanowski. 1985. One view of the role of scientific information in the solution of enviro-economic problems. In: N.L. Chao and W.K. Smith (eds.), Proceedings of the International Symposium on Utilization of Coastal Ecosystems: Planning, Pollution, and Productivity, 21-27 November 1982, Rio Grande, Brazil, Volume 1. Duke University Marine Laboratory, Beaufort, NC, and Editora Da Furg, Luiz Lorea, 261, Cx Postal 474,96200-Rio Grande-RS-Brazil, pp.73-102.

Farrington, J.W., A.C. Davis, B.J. Brownawell, B.W. Tripp, C.H. Clifford, and J.B. Livramento. 1986a. The biogeochemistry of polychlorinated biphenyls in the Acushnet River Estuary, Massachusetts. In: M.L. Sohn (ed.), Organic Marine Geochemistry, ACS Symposium Series 305. American Chemical Society, Washington, DC, pp. 174-197.

Farrington, J.W., X. Jia, C.H. Clifford, B.W. Tripp, J.B. Livramento, A.C. Davis, N.M. Frew, and C.G. Johnson. 1986b. No. 2 Fuel Oil Compound Retention and Release by *Mytilus edulis*: 1983 Cape Cod Canal Oil Spill. Technical Report No. WHOI-86-8, Woods Hole Oceanographic Institution, Woods Hole, MA, 73pp.

Gardner, D. 1978. Mercury in fish and waters of the Irish Sea and other United Kingdom fishing grounds. Nature 272: 49-51.

George, S.G. 1982. Subcellular accumulation and detoxication of metals in aquatic animals. In: W.B. Vernberg, A. Calabrese, F.P. Thurberg, and F.J. Vernberg (eds.), Physiological Mechanisms of Marine Pollutant Toxicity. Academic Press, New York, pp. 3-52.

Gil, L.W. 1988. Buzzards Bay Cranberry Bog Input Study. Special Water Quality Study 1988. Massachusetts Department of Environmental Quality Engineering, Division of Water Pollution Control, Technical Services Branch, Westborough, MA, 85 pp.

Gilbert, T., A. Clay, and A. Barker. 1973. Site Selection and Study of Ecological Effects of Disposal of Dredged Materials in Buzzards Bay, Massachusetts. Report to the

U.S. Army Corps of Engineers, DAWC 33-77-c-0024 by New England Aquarium, Boston, MA, 70 pp.

Goldberg, E.D. 1976. The health of the oceans. The UNESCO Press, Paris, France.

Goldberg, E., V.T. Bowen, J.W. Farrington, G.R. Harvey, J.H. Martin, P.L. Parker, R.W. Risebrough, W. Robertson IV, E. Schneider, and E. Gamble. 1978. The Mussel Watch. Environmental Conservation 5 (2): 101-125.

Hahn, M.E., B.R. Woodin, and J.J. Stegeman. 1989. Induction of cytochrome P-450E (P-450IA1) by 2,3,7,8-tetrachloro-dibenzofuran (2,3,7,8-TCDF) in the marine fish scup (*Stenotomus chrysops*). Marine Environmental Research 28: 61-65.

Hites, R.A., R.E. LaFlamme, and J.W. Farrington. 1977. Sedimentary polycyclic aromatic hydrocarbons: The historical record. Science 198: 829-831.

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. Canadian Journal of Fisheries and Aquatic Sciences 40 (Supplement No.2): 41-53.

Holden, A.V. 1973. International cooperative study of organochlorine and mercury residues in wildlife, 1969-71. Pesticide Monitoring Journal 7:37-52.

Ikalainen, A.J. and D.C. Allen. 1989. New Bedford Harbor Superfund Project. In: Contaminated Marine Sediments — Assessment and Remediation. National Research Council, National Academy Press, Washington, DC, pp. 312-350.

Jenkins, K.D., D.A. Brown, G.P. Hershelman, and W.C. Meyer. 1982. Contaminants in white croakers *Genyonemus lineatus* (Ayres, 1855) from the Southern California Bight. I. Trace metal detoxification/toxification. In: W.B. Vernberg, A. Calabrese, F.P. Thurberg, and F.J. Vernberg (eds.), Physiological Mechanisms of Marine Pollutant Toxicity. Academic Press, New York, pp. 177-196.

Kelly, B. 1978. Metal Concentrations of Some of the Macrobenthos of New Bedford Harbor and Evaluation of the Use of Slipper Limpet *Crepidula fornicata* as a Monitor of Coastal Metal Pollution. M.S. Thesis, Southeastern Massachusetts University, North Dartmouth, MA, 83 pp.

Koojiman, S.A.L.M. and J.A.J. Metz. 1984. On the dynamics of chemically stressed populations: The deduction of population consequences from effects on individuals. Ecotoxicology and Environmental Safety 8: 254-274.

Krebs, C.T. and K.A. Burns. 1977. Long term effects of an oil spill on populations of the salt-marsh crab *Uca pugnax*. Science 197: 484-487.

Krebs, C.T. and I. Valiela. 1977. Effects of experimentally applied chlorinated hydrocarbons on the biomass of the fiddler crab *Uca pugnax*. Estuarine and Coastal Marine Science 6: 375-386.

Krebs, C.T., I. Valiela, G. Harvey, and J.M. Teal. 1974. Reduction in field populations of fiddler crabs by uptake of chlorinated hydrocarbons. Marine Pollution Bulletin 5: 140-142.

Leavitt, D.F., J. McDowell Capuzzo, R.M. Smolowitz, D.L. Miosky, B.A. Lancaster, and C.L. Reinisch. 1990. Hematopoietic neoplasia in *Mya arenaria*: prevalence and indices of physiological condition. Marine Biology 105: 313-321.

Mackay, N.J., R.J. Williams, J.L. Kacprzac, M.N. Kazacos, A.J. Collins, and E.H. Auty. 1975. Heavy metals in cultivated oysters (*Crassostrea commercialis* = *Saccostrea cucullata*) from the estuaries of New South Wales. Australian Journal of Marine and Freshwater Research 26: 31-46.

Malins, D.C. and H.O. Hodgins. 1981. Petroleum and marine fishes: A review of uptake, disposition, and effects. Environmental Science and Technology 15: 1272-1280.

Malins, D.C., B.B. McCain, D.W. Brown, S.-L. Chan, M.S. Myers, J.T. Landahl, P.G. Prohaska, A.J. Friedman, L.D. Rhodes, D.G. Burrows, W.D. Gronlund, and H.O. Hodgins. 1984. Chemical pollutants in sediments and diseases of bottom dwelling fish in Puget Sound, Washington. Environmental Science and Technology 18: 705-713.

McCain, B.B., H.O. Hodgins, W.D. Gronlund, J.W. Hawkes, D.W. Brown, M.S. Myers, and J.H. Vandermeulen. 1978. Bioavailability of crude oil from experimentally oiled sediments to English sole (*Parophrys vetulus*), and pathological consequences. Journal of the Fisheries Research Board of Canada 35: 657.

McElroy, A.E., J.W. Farrington, and J.M. Teal. 1989. Bioavailability of polycyclic aromatic hydrocarbons in the aquatic environment. In: U. Varanasi (ed.), Metabolism of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment. CRC Press, Inc., Boca Raton, FL, pp. 1-39.

Mearns, A.J., M.B. Matta, D. Simecek-Beatty, M.F. Buchman, G. Shigenaka, and W.A. Wert. 1988. PCB and Chlorinated Pesticide Contamination in U.S. Fish and Shellfish: A Historical Assessment Report. NOAA Technical Memorandum NOS OMA 39. National Ocean Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, Rockville, MD, 139 pp.

Metcalf and Eddy. 1983. Acushnet Estuary PCBs Data Management Final Report. Submitted to: the U.S. Environmental Protection Agency, Region I, Boston, MA, by Metcalf and Eddy, Inc., Boston, MA, 137pp. plus appendices.

Miller, D.R. 1984. Effects of pollutants at the ecosystem level, SCOPE 22. P.I. Sheehan (ed.), John Wiley and Sons, New York.

Murchelano, R.A. and R.E. Wolke. 1985. Epizootic carcinoma in winter flounder, *Pseudopleuronectes americanus*. Science 228: 587-589.

NAS. 1979. Polychlorinated Biphenyls. A report of the National Research Council, National Academy Press, Washington, DC.

NAS. 1985. Oil in the Sea. A report of the National Research Council, National Academy Press, Washington, DC, 601 pp.

Neff, J.M. 1979. Polycyclic Aromatic Hydrocarbons in the Aquatic Environment: Sources, Fates and Biological Effects. Applied Science Publishers, Ltd., London, 262 pp.

Nisbet, I.C.T. and L.M. Reynolds. 1984. Organochlorine residues in common terns and associated estuarine organisms, Massachusetts, USA, 1971-1981. Marine Environmental Research 11: 33-66.

NOAA. 1987. A Aummary of Selected Data on Chemical Contaminants in Tissues Collected during 1984, 1985, and 1986. NOAA Technical Memorandum NOS OMA 38. Ocean Assessment Division, Office of Oceanography and Marine Assessment, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, Rockville, MD, 23 pp. plus appendices.

NOAA. 1988. A Summary of Selected Data on Chemical Contaminants in Sediments Collected during 1984, 1985, 1986 and 1987. National Status and Trends Program for Marine Environmental Quality, Progress Report. NOAA Technical Memorandum NOS OMA 44, Ocean Assessment Division, Office of Oceanography and Marine Assessment, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, Rockville, MD, 14 pp. plus appendices.

NOAA. 1989a. Agricultural Pesticide Use in Estuarine Drainage Areas: A Preliminary Summary for Selected Pesticides. A.S. Pait, D.R.G. Farrow, L.A. Lowe, and P.A. Pacheco. Strategic Assessment Branch, Ocean Assessment Division, Office of Oceanography and Marine Assessment, National Ocean Service, National Oceanic and Atmospheric Administration, Rockville, MD.

NOAA. 1989b. A Summary of Data on Tissue Contamination from the First Three Years (1986-1988) of the Mussel Watch Project. NOAA Technical Memorandum NOS OMA 49, Ocean Assessment Division, Office of Oceanography and Marine Assessment, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, Rockville, MD, 22 pp. plus appendices.

Norwood, C.B., M. Hackett, R.J. Pruell, B.C. Butterworth, K.J. Williamson, and S.M. Naumann. 1989. Polychlorinated dibenzo-p-dioxins and dibenzofurans in selected estuarine sediments. Chemosphere 18: 553-560.

O'Connor, J.M. and J.W. Rachlin. 1982. Perspectives on metals in New York Bight organisms: Factors controlling accumulation and body burdens. In: G.F. Mayer (ed.), Ecological Stress and the New York Bight: Science and Management. Estuarine Research Federation, Columbia, SC, pp. 655-673.

Perkins, E.M., D.A. Brown, and K.D. Jenkins. 1982. Contaminants in white croakers *Genyonemus lineatus* (Ayres, 1855) from the southern California Bight. III. Histopathology. In: W.B. Vernberg, A. Calabrese, F.P. Thurberg, and F.J. Vernberg (eds.), Physiological Mechanisms of Marine Pollutant Toxicity. Academic Press, New York, pp. 215-231.

Phillips, D.J.H. 1977. The use of biological indicator organisms to monitor trace metal pollution in marine and estuarine environments: A review. Environmental Pollution 13: 281-317.

Phillips, D.J.H. 1980. Quantitative Aquatic Biological Indicators. Their Use to Monitor Trace Metal and Organochlorine Pollution. Appl. Science Publishers Ltd., London.

Pruell, R.J., C.B. Norwood, R.D. Bowen, W.S. Boothman, P.F. Rogerson, M. Hackett, and B.C. Butterworth. 1990. Geochemical study of sediment contamination in New Bedford Harbor, Massachusetts. Marine Environmental Research 29: 77-101.

Reinisch, C.L., A.M. Charles, and A.M. Stone. 1984. Epizootic neoplasia in soft shell clams collected from New Bedford Harbor. Hazardous Waste 1: 73-81.

Rhoads, D.K. 1963. Rates of sediment reworking by *Yoldia limatula* in Buzzards Bay, Massachusetts and Long Island Sound. Journal of Sedimentary Petrology 33: 723-727.

Rhoads, D.K. 1967. Biogenic reworking of intertidal and subtidal sediments in Barnstable Harbor and Buzzards Bay, Massachusetts, USA. Journal of Geology 75: 461-476.

Rosenfeld, L.K., R.P. Signell, and G.P. Gawarkiewicz. 1984. Hydrographic Study of Buzzards Bay, 1982-1983. Technical Report No. 84-5, Woods Hole Oceanographic Institution, Woods Hole, MA, 134 pp.

Sanders, H.L., J.F. Grassle, G.R. Hampson, L.S. Morse, S. Garner-Price, and C.C. Jones. 1980. Anatomy of an oil spill: long-term effects from the grounding of the barge *Florida* off West Falmouth, Massachusetts. Journal of Marine Research 38: 265-380.

Schwartz, J.P. 1988. Distribution and Concentration of Polychlorinated Biphenyls in Lobster, Winter Flounder, and Quahogs from Buzzards Bay, Massachusetts. Report submitted to U.S. EPA Buzzards Bay Program from Massachusetts Division of Marine Fisheries.

Signell, R.P. 1987. Tide- and Wind Forced Currents in Buzzards Bay, Massachusetts. Masters Thesis and Technical Report No. 87-15, Woods Hole Oceanographic Institution, Woods Hole, MA, 86 pp.

Sindermann, C.J., F.B. Bang, N.O. Christensen, B. Dethlefsen, J.C. Harshbarger, J.R. Mitchell, and M.F. Mulcahy. 1980. The role and value of pathobiology in pollution effects monitoring programs. In: A.D. McIntyre and J.B. Pearce (eds.), Biological Effects of Pollution and Problems of Monitoring. Rapports et Proces-Verbaux des Reunions, Vol. 179. Conseil Internationl pour L'Exploration de la Mer, Copenhagen, pp. 135-151.

Smith, C.E., T.H. Peck, R.J. Klauda, and J.B. McLaren. 1979. Hepatomas in Atlantic tomcod *Microgadus tomcod* (Walbaum) collected in the Hudson River Estuary in New York. Journal of Fish Disease 2: 313.

Stegeman, J.J. 1981. Polynuclear aromatic hydrocarbons and their metabolism in the marine environment. In: H.V. Gelboin and P.O.P. Ts'O (eds.), Polycyclic Hydrocarbons and Cancer. Academic Press, New York, pp. 1-60.

Summerhayes, C.P., J.P. Ellis, P. Stoffers, S.R. Briggs, and M.G. Fitzgerald. 1977. Fine Grained Sediment and Industrial Waste Distribution and Dispersal in New Bedford Harbor and Western Buzzards Bay, Massachusetts. Technical Report No. 76-115, Woods Hole Oceanographic Institution, Woods Hole, MA, 110 pp.

Thomason, J., K.K. Turekian, and R.J. McCaffrey. 1975. The accumulation of metals in and release from sediments of Long Island Sound. In: L.E. Cronin (ed.), Estuarine Research 1: 28-44.

U.S. EPA. 1977. Criteria Document for PCBs. Prepared for the U.S. Environmental Protection Agency by the Massachusetts Audubon Society, July 1976. Document #PB-255-397, U.S. Department of Commerce, National Technical Information Service, Washington, DC.

U.S. EPA. 1988. Assessment of Quincy Bay Contamination, Summary Report. U.S. Environmental Protection Agency Contract No. 68-02-4357, Region I, Boston, MA.

U.S. FDA. 1982a. FY 79 Pesticides and Metals in Fish Program. Compliance Program Report of Findings (7305.007). U.S. Food and Drug Administration, Washington, DC.

U.S. FDA. 1982b. FY 79 Total Diet Studies — Adult. Compliance Program Report of Findings (7305.002). U.S. Food and Drug Administration, Washington, DC.

von Westernhagen, H., H. Rosenthal, V. Dethlefsen, W. Ernst, U. Harms, and P.D. Hansen. 1981. Bioaccumulating substances and reproductive success in Baltic flounder *Platichthys flesus*. Aquatic Toxicology 1: 85-99.

Weaver, G. 1984. PCB contamination in and around New Bedford, Mass. Environmental Science and Technology 18: 22A-27A.

Widdows, J., T. Bakke, B.L. Bayne, P. Donkin, D.R. Livingstone, D.M. Lowe, M.N. Moore, S.V. Evans, and S.L. Moore. 1982. Response of *Mytilus edulis* on exposure to the water accommodated fraction of North Sea oil. Marine Biology 67: 15-31.

Youngblood, W.W. and M. Blumer. 1975. PAH in the environment: Homologous series in soils and recent sediments. Geochimica et Cosmochimica Acta 39: 1303-1314.

Appendix

All data have been normalized to wet weight values; where wet weight:dry weight conversion factors were not given in specific reports, average literature values were used to calculate wet weight conversion of the data. For both trace metal and organic contaminant data, values which vary by factors of 2 to 4 may be attributed to natural seasonal and/or spatial variation; values which vary by factors of 5 to 10 for trace metals and by factors of 5 to several orders of magnitude for organic contaminants are considered for the purpose of this analysis to be a conservative assessment of possible chemical contamination.

PCB-CB156	Quahog	BB-6B	1978		0.004	Whole	Parrington et al. (1986)
PCB-CB156	Sand Flounder	BB-6B	1979		0.10	Edible Fillets	Parrington et al. (1986)
PCB-CB156	Winter Flounder	BB-6B	1979		0.063	Edible Fillets	Parrington et al. (1986)
PCB-CB156	Winter flounder (F)	BB-6	1987	0.000-0.018	0.006	Muscle	Pruell (unpub. data)
PCB-CB156	Winter flounder (M)	BB-6	1987		0.014	Muscle	Pruell (unpub. data)
PCB-CB180	Bay Scallop	BB-19	1978		0.0004	Whole	Parrington et al. (1986)
PCB-CB180	Lobster	BB-6A	1979	0.004-0.031	0.015	Muscle	Parrington et al. (1986)
PCB-CB180	Lobster	BB-6B	1979	0.029-1.50	0.86	Viscera	Parrington et al. (1986)
PCB-CB180	Mud Crab	BB-6B	1979		0.025	Whole	Parrington et al. (1986)
PCB-CB180	Mussel	BB-19	1985	0.00014-0.0012	0.0004	Whole	Capuzzo & Parrington (unpub)
PCB-CB180	Mussel	BB-6A	1985	0.003-0.005	0.004	Whole	Capuzzo & Parrington (unpub)
PCB-CB180	Mussel	BB-6A	1981		0.006	Whole	Parrington et al. (1986)
PCB-CB180	Quahog	BB-6B	1978		0.004	Whole	Parrington et al. (1986)
PCB-CB180	Sand Flounder	BB-6B	1979		0.044	Edible Fillets	Parrington et al. (1986)
PCB-CB180	Winter Flounder	BB-6B	1979		0.062	Edible Fillets	Parrington et al. (1986)
PCB-CB180	Winter flounder (F)	BB-6	1987	0.001-0.034	0.012	Muscle	Pruell (unpub. data)
PCB-CB180	Winter flounder (M)	BB-6	1987		0.033	Muscle	Pruell (unpub. data)
PCB-CB183	Mussel	BB-19	1985	0.0002-0.0009	0.0005	Whole	Capuzzo & Parrington (unpub)
PCB-CB183	Mussel	BB-6A	1985	0.002-0.006	0.004	Whole	Capuzzo & Parrington (unpub)
PCB-CB194	Lobster	BB-6B	1979	0.0004-0.0025	0.001	Muscle	Parrington et al. (1986)
PCB-CB194	Lobster	BB-6B	1979	0.034-0.110	0.068	Viscera	Parrington et al. (1986)
PCB-CB194	Mud Crab	BB-6B	1979		0.0015	Whole	Parrington et al. (1986)
PCB-CB194	Quahog	BB-6B	1978		0.0004	Whole	Parrington et al. (1986)
PCB-Total	Bay Scallop	BB-19	1978		0.051	Whole	Parrington et al. (1986)
PCB-Total	Crab	BB-20		0.023-0.104	0.070		Battelle-Superfund
PCB-Total	Crab	BB-6A		0.014-3.43	0.93		Battelle-Superfund
PCB-Total	Crab	BB-6B		0.076-0.457	0.27		Battelle-Superfund
PCB-Total	Lobster	BB-6B	1979		0.645	Muscle	Parrington et al. (1986)
PCB-Total	Lobster	BB-6B	1979		25.76	Viscera	Parrington et al. (1986)
PCB-Total	Mud Crab	BB-6B	1979		0.58	Whole	Parrington et al. (1986)
PCB-Total	Mussel	BB-19	1985		0.251	Whole	Capuzzo & Parrington (unpub)
PCB-Total	Mussel	BB-20		0.008-0.051	0.03	Whole	Battelle-Superfund
PCB-Total	Mussel	BB-6A	1985		1.87	Whole	Capuzzo & Parrington (unpub)
PCB-Total	Mussel	BB-6A	1981		1.32	Whole	Parrington et al. (1986)
PCB-Total	Mussel	BB-6A		1.04-3.18	2.32	Whole	Battelle-Superfund
PCB-Total	Mussel	BB-6B		1.14-7.27	3.96	Whole	Battelle-Superfund
PCB-Total	Mussel	BB-6C		0.24-0.30	0.28	Whole	Battelle-Superfund
PCB-Total	Polychaetes	BB-20		0.058-1.41	0.53	Whole	Battelle-Superfund
PCB-Total	Polychaetes	BB-6A			13.20	Whole	Battelle-Superfund
PCB-Total	Polychaetes	BB-6C		0.045-0.255	0.134	Whole	Battelle-Superfund
PCB-Total	Quahog	BB-20		0.003-0.14	0.037	Whole	Battelle-Superfund
PCB-Total	Quahog	BB-6A		0.14-2.82	0.74	Whole	Battelle-Superfund
PCB-Total	Quahog	BB-6B	1978		0.216	Whole	Parrington et al. (1986)
PCB-Total	Quahog	BB-6B		0.002-0.150	0.048	Whole	Battelle-Superfund
PCB-Total	Quahog	BB-6C		0.004-0.23	0.076	Whole	Battelle-Superfund
PCB-Total	Sand Flounder	BB-6B	1979		5.32	Edible Fillets	Parrington et al. (1986)
PCB-Total	Winter Flounder	BB-6B	1979		1.55	Edible Fillets	Parrington et al. (1986)
PCB-Total	Winter flounder (F)	BB-6	1987		0.58	Muscle	Pruell (unpub. data)
PCB-Total	Winter flounder (M)	BB-6	1987		1.52	Muscle	Pruell (unpub. data)
PIC	Ocean quahog	BB	?		3.7	Whole	EMCO (1983)
PIC	Soft-shelled clam	BB-6	1980?	34-362	153	Whole	Cantillo (1985)
Phenanthrene	Mussel	BB-19	1985	0.0002-0.011	0.003	Whole	Capuzzo & Parrington (unpub)
Phenanthrene	Mussel	BB-6A	1985	0.002-0.127	0.019	Whole	Capuzzo & Parrington (unpub)
Pyrene	Mussel	BB-19	1985	0.0004-0.0015	0.0011	Whole	Capuzzo & Parrington (unpub)
Pyrene	Mussel	BB-6A	1985	0.001-0.135	0.055	Whole	Capuzzo & Parrington (unpub)

PCB-CB114	Winter flounder (M)	BB-6	1987		0.255	Muscle	Pruell (unpub. data)
PCB-CB128	Bay Scallop	BB-19	1978		0.0007	Whole	Farrington et al. (1986)
PCB-CB128	Lobster	BB-6B	1979	0.004-0.048	0.024	Muscle	Farrington et al. (1986)
PCB-CB128	Lobster	BB-6B	1979	0.44-2.00	1.15	Viscera	Farrington et al. (1986)
PCB-CB128	Mud Crab	BB-6B	1979		0.015	Whole	Farrington et al. (1986)
PCB-CB128	Mussel	BB-19	1985	0.0003-0.003	0.001	Whole	Capuzzo & Farrington (unpub)
PCB-CB128	Mussel	BB-6A	1985	0.007-0.021	0.017	Whole	Capuzzo & Farrington (unpub)
PCB-CB128	Mussel	BB-6A	1981		0.021	Whole	Farrington et al. (1986)
PCB-CB128	Quahog	BB-6B	1978		0.004	Whole	Farrington et al. (1986)
PCB-CB128	Sand Flounder	BB-6B	1979		0.12	Edible Fillets	Farrington et al. (1986)
PCB-CB128	Winter Flounder	BB-6B	1979		0.077	Edible Fillets	Farrington et al. (1986)
PCB-CB129	Mussel	BB-19	1985	0.0001-0.0007	0.0003	Whole	Capuzzo & Farrington (unpub)
PCB-CB129	Mussel	BB-6A	1985	0.0008-0.003	0.002	Whole	Capuzzo & Farrington (unpub)
PCB-CB137	Mussel	BB-19	1985	0.0001-0.0004	0.0002	Whole	Capuzzo & Farrington (unpub)
PCB-CB137	Mussel	BB-6A	1985	0.0016-0.0030	0.002	Whole	Capuzzo & Farrington (unpub)
PCB-CB138	Bay Scallop	BB-19	1978		0.006	Whole	Farrington et al. (1986)
PCB-CB138	Lobster	BB-6B	1979	0.026-0.250	0.122	Muscle	Farrington et al. (1986)
PCB-CB138	Lobster	BB-6B	1979	2.29-11.00	6.00	Viscera	Farrington et al. (1986)
PCB-CB138	Mud Crab	BB-6B	1979		0.054	Whole	Farrington et al. (1986)
PCB-CB138	Mussel	BB-19	1985	0.001-0.011	0.008	Whole	Capuzzo & Farrington (unpub)
PCB-CB138	Mussel	BB-6A	1985	0.047-0.133	0.109	Whole	Capuzzo & Farrington (unpub)
PCB-CB138	Mussel	BB-6A	1981		0.13	Whole	Farrington et al. (1986)
PCB-CB138	Quahog	BB-6B	1978		0.026	Whole	Farrington et al. (1986)
PCB-CB138	Sand Flounder	BB-6B	1979		0.75	Edible Fillets	Farrington et al. (1986)
PCB-CB138	Winter Flounder	BB-6B	1979		0.33	Edible Fillets	Farrington et al. (1986)
PCB-CB138	Winter flounder (F)	BB-6	1987	0.002-0.175	0.065	Muscle	Pruell (unpub. data)
PCB-CB138	Winter flounder (M)	BB-6	1987		0.213	Muscle	Pruell (unpub. data)
PCB-CB141	Bay Scallop	BB-19	1978		0.0004	Whole	Farrington et al. (1986)
PCB-CB141	Lobster	BB-6B	1979	0.0003-0.006	0.003	Muscle	Farrington et al. (1986)
PCB-CB141	Lobster	BB-6B	1979	0.022-0.27	0.12	Viscera	Farrington et al. (1986)
PCB-CB141	Mud Crab	BB-6B	1979		0.006	Whole	Farrington et al. (1986)
PCB-CB141	Mussel	BB-19	1985	0.0001-0.0003	0.0002	Whole	Capuzzo & Farrington (unpub)
PCB-CB141	Mussel	BB-6A	1985	0.0014-0.0031	0.002	Whole	Capuzzo & Farrington (unpub)
PCB-CB141	Mussel	BB-6A	1981		0.004	Whole	Farrington et al. (1986)
PCB-CB141	Quahog	BB-6B	1978		0.003	Whole	Farrington et al. (1986)
PCB-CB141	Sand Flounder	BB-6B	1979		0.072	Edible Fillets	Farrington et al. (1986)
PCB-CB141	Winter Flounder	BB-6B	1979		0.034	Edible Fillets	Farrington et al. (1986)
PCB-CB153	Bay Scallop	BB-19	1978		0.005	Whole	Farrington et al. (1986)
PCB-CB153	Lobster	BB-6B	1979	0.029-0.26	0.133	Muscle	Farrington et al. (1986)
PCB-CB153	Lobster	BB-6B	1979	2.30-13.00	6.93	Viscera	Farrington et al. (1986)
PCB-CB153	Mud Crab	BB-6B	1979		0.220	Whole	Farrington et al. (1986)
PCB-CB153	Mussel	BB-19	1985	0.002-0.014	0.008	Whole	Capuzzo & Farrington (unpub)
PCB-CB153	Mussel	BB-6A	1985	0.045-0.134	0.107	Whole	Capuzzo & Farrington (unpub)
PCB-CB153	Mussel	BB-6A	1981		0.14	Whole	Farrington et al. (1986)
PCB-CB153	Quahog	BB-6B	1978		0.021	Whole	Farrington et al. (1986)
PCB-CB153	Sand Flounder	BB-6B	1979		0.77	Edible Fillets	Farrington et al. (1986)
PCB-CB153	Winter Flounder	BB-6B	1979		0.37	Edible Fillets	Farrington et al. (1986)
PCB-CB153	Winter flounder (F)	BB-6	1987	0.003-0.224	0.082	Muscle	Pruell (unpub. data)
PCB-CB153	Winter flounder (M)	BB-6	1987		0.242	Muscle	Pruell (unpub. data)
PCB-CB154	Winter flounder (F)	BB-6	1987	0.002-0.125	0.059	Muscle	Pruell (unpub. data)
PCB-CB154	Winter flounder (M)	BB-6	1987		0.163	Muscle	Pruell (unpub. data)
PCB-CB156	Bay Scallop	BB-19	1978		0.0002	Whole	Farrington et al. (1986)
PCB-CB156	Lobster	BB-6B	1979	0.004-0.044	0.022	Muscle	Farrington et al. (1986)
PCB-CB156	Lobster	BB-6B	1979	0.31-1.80	1.00	Viscera	Farrington et al. (1986)
PCB-CB156	Mud Crab	BB-6B	1979		0.025	Whole	Farrington et al. (1986)
PCB-CB156	Mussel	BB-19	1985	0.0001-0.001	0.0007	Whole	Capuzzo & Farrington (unpub)
PCB-CB156	Mussel	BB-6A	1985	0.0042-0.012	0.010	Whole	Capuzzo & Farrington (unpub)
PCB-CB156	Mussel	BB-6A	1981		0.014	Whole	Farrington et al. (1986)

PCB-CB060	Mussel	BB-6A	1985	0.044-0.084	0.063	Whole	Capuzzo & Parrington (unpub)	PCB-CB087	Sand Flounder	BB-6B	1979		0.25	Edible Fillets	Parrington et al. (1986)
PCB-CB060	Mussel	BB-6A	1981		0.052	Whole	Parrington et al. (1986)	PCB-CB087	Winter Flounder	BB-6B	1979		0.084	Edible Fillets	Parrington et al. (1986)
PCB-CB060	Quahog	BB-6B	1978		0.007	Whole	Parrington et al. (1986)	PCB-CB095	Bay Scallop	BB-19	1978		0.006	Whole	Parrington et al. (1986)
PCB-CB060	Sand Flounder	BB-6B	1979		0.15	Edible Fillets	Parrington et al. (1986)	PCB-CB095	Lobster	BB-6B	1979	0.01-0.32	0.163	Muscle	Parrington et al. (1986)
PCB-CB060	Winter Flounder	BB-6B	1979		0.029	Edible Fillets	Parrington et al. (1986)	PCB-CB095	Lobster	BB-6B	1979	1.6-11.0	5.40	Viscera	Parrington et al. (1986)
PCB-CB060	Winter flounder (F)	BB-6	1987	0.001-0.026	0.012	Muscle	Pruell (unpub. data)	PCB-CB095	Mud Crab	BB-6B	1979		0.099	Whole	Parrington et al. (1986)
PCB-CB060	Winter flounder (M)	BB-6	1987		0.039	Muscle	Pruell (unpub. data)	PCB-CB095	Mussel	BB-19	1985	0.003-0.012	0.008	Whole	Capuzzo & Parrington (unpub)
PCB-CB070	Bay Scallop	BB-19	1978		0.002	Whole	Parrington et al. (1986)	PCB-CB095	Mussel	BB-6A	1985	0.257-0.498	0.398	Whole	Capuzzo & Parrington (unpub)
PCB-CB070	Lobster	BB-6B	1979	0-0.029	0.013	Muscle	Parrington et al. (1986)	PCB-CB095	Mussel	BB-6A	1981		0.34	Whole	Parrington et al. (1986)
PCB-CB070	Lobster	BB-6B	1979	0.058-0.78	0.35	Viscera	Parrington et al. (1986)	PCB-CB095	Quahog	BB-6B	1978		0.036	Whole	Parrington et al. (1986)
PCB-CB070	Mud Crab	BB-6B	1979		0.013	Whole	Parrington et al. (1986)	PCB-CB095	Sand Flounder	BB-6B	1979		1.10	Edible Fillets	Parrington et al. (1986)
PCB-CB070	Mussel	BB-19	1985	0.0005-0.004	0.003	Whole	Capuzzo & Parrington (unpub)	PCB-CB095	Winter Flounder	BB-6B	1979		0.20	Edible Fillets	Parrington et al. (1986)
PCB-CB070	Mussel	BB-6A	1985	0.094-0.191	0.142	Whole	Capuzzo & Parrington (unpub)	PCB-CB101	Bay Scallop	BB-19	1978		0.005	Whole	Parrington et al. (1986)
PCB-CB070	Mussel	BB-6A	1981		0.13	Whole	Parrington et al. (1986)	PCB-CB101	Lobster	BB-6B	1979	0.002-0.072	0.031	Muscle	Parrington et al. (1986)
PCB-CB070	Quahog	BB-6B	1978		0.015	Whole	Parrington et al. (1986)	PCB-CB101	Lobster	BB-6B	1979	0.22-2.80	1.26	Viscera	Parrington et al. (1986)
PCB-CB070	Sand Flounder	BB-6B	1979		0.30	Edible Fillets	Parrington et al. (1986)	PCB-CB101	Mud Crab	BB-6B	1979		0.036	Whole	Parrington et al. (1986)
PCB-CB070	Winter Flounder	BB-6B	1979		0.093	Edible Fillets	Parrington et al. (1986)	PCB-CB101	Mussel	BB-19	1985	0.002-0.011	0.008	Whole	Capuzzo & Parrington (unpub)
PCB-CB077	Winter flounder (F)	BB-6	1987	0.000-0.018	0.007	Muscle	Pruell (unpub. data)	PCB-CB101	Mussel	BB-6A	1985	0.080-0.244	0.183	Whole	Capuzzo & Parrington (unpub)
PCB-CB077	Winter flounder (M)	BB-6	1987		0.011	Muscle	Pruell (unpub. data)	PCB-CB101	Mussel	BB-6A	1981		0.18	Whole	Parrington et al. (1986)
PCB-CB086	Mussel	BB-19	1985	0.0002-0.001	0.001	Whole	Capuzzo & Parrington (unpub)	PCB-CB101	Quahog	BB-6B	1978		0.022	Whole	Parrington et al. (1986)
PCB-CB086	Mussel	BB-6A	1985	0.008-0.022	0.017	Whole	Capuzzo & Parrington (unpub)	PCB-CB101	Sand Flounder	BB-6B	1979		0.64	Edible Fillets	Parrington et al. (1986)
PCB-CB087	Bay Scallop	BB-19	1978		0.0003	Whole	Parrington et al. (1986)	PCB-CB101	Winter Flounder	BB-6B	1979		0.15	Edible Fillets	Parrington et al. (1986)
PCB-CB087	Lobster	BB-6B	1979	0.001-0.016	0.008	Muscle	Parrington et al. (1986)	PCB-CB101	Winter flounder (F)	BB-6	1987	0.002-0.059	0.028	Muscle	Pruell (unpub. data)
PCB-CB087	Lobster	BB-6B	1979	0.052-0.54	0.26	Viscera	Parrington et al. (1986)	PCB-CB101	Winter flounder (M)	BB-6	1987		0.064	Muscle	Pruell (unpub. data)
PCB-CB087	Mud Crab	BB-6B	1979		0.010	Whole	Parrington et al. (1986)	PCB-CB105	Mussel	BB-19	1985	0.001-0.002	0.002	Whole	Capuzzo & Parrington (unpub)
PCB-CB087	Mussel	BB-19	1985	0.0005-0.002	0.001	Whole	Capuzzo & Parrington (unpub)	PCB-CB105	Mussel	BB-6A	1985	0.003-0.060	0.038	Whole	Capuzzo & Parrington (unpub)
PCB-CB087	Mussel	BB-6A	1985	0.021-0.062	0.047	Whole	Capuzzo & Parrington (unpub)	PCB-CB105	Winter flounder (F)	BB-6	1987	0.001-0.118	0.044	Muscle	Pruell (unpub. data)
PCB-CB087	Mussel	BB-6A	1981		0.057	Whole	Parrington et al. (1986)	PCB-CB105	Winter flounder (M)	BB-6	1987		0.118	Muscle	Pruell (unpub. data)
PCB-CB087	Quahog	BB-6B	1978		0.009	Whole	Parrington et al. (1986)	PCB-CB118	Winter flounder (F)	BB-6	1987	0.002-0.284	0.059	Muscle	Pruell (unpub. data)

PCB	Winter flounder	Bb-6	1979	0.2-8.1	2.36	Edible	NA BKF	PCB-C3049	Bay Scallop	BB-19	1978		0.012	Whole	Farrington et al. (1986)
PCB	Winter flounder	Bb-6	1980	0-22	6.4	Edible	Weaver (1984)	PCB-C3049	Lobster	BB-6B	1979	0-0.018	0.007	Muscle	Farrington et al. (1986)
PCB	Winter flounder	BB-6A-6C	1979	0.2-8.1	1.89	Edible	Kolek & Courvels (1981)	PCB-C3049	Lobster	BB-6B	1979	0.036-0.52	0.21	Viscera	Farrington et al. (1986)
PCB	Winter flounder	BB-6A-6C-20	1976	2.6-13.0	7.77	Edible	Kolek & Courvels (1981)	PCB-C3049	Mud Crab	BB-6B	1979		0.004	Whole	Farrington et al. (1986)
PCB	Winter flounder	BB-6B/6C	1980	ND-5.9	2.95	Edible	Kolek & Courvels (1981)	PCB-C3049	Mussel	BB-19	1985	0.00035-0.003	0.002	Whole	Capuzzo & Farrington (unpub)
PCB-C3028	Bay Scallop	BB-19	1978		0.003	Whole	Farrington et al. (1986)	PCB-C3049	Mussel	BB-6A	1985	0.152-0.305	0.202	Whole	Capuzzo & Farrington (unpub)
PCB-C3028	Lobster	BB-6B	1979	0.001-0.13	0.067	Muscle	Farrington et al. (1986)	PCB-C3049	Mussel	BB-6A	1981		0.07	Whole	Farrington et al. (1986)
PCB-C3028	Lobster	BB-6B	1979	0.36-2.2	1.13	Viscera	Farrington et al. (1986)	PCB-C3049	Quahog	BB-6B	1978		0.015	Whole	Farrington et al. (1986)
PCB-C3028	Mud Crab	BB-6B	1979		0.043	Whole	Farrington et al. (1986)	PCB-C3049	Sand Flounder	BB-6B	1979		0.27	Edible Fillets	Farrington et al. (1986)
PCB-C3028	Mussel	BB-6A	1981		0.052	Whole	Farrington et al. (1986)	PCB-C3049	Winter Flounder	BB-6B	1979		0.021	Edible Fillets	Farrington et al. (1986)
PCB-C3028	Quahog	BB-6B	1978		0.022	Whole	Farrington et al. (1986)	PCB-C3049	Winter Flounder (F)	BB-6	1987	0.001-0.038	0.026	Muscle	Prwell (unpub. data)
PCB-C3028	Sand Flounder	BB-6B	1979		0.18	Edible Fillets	Farrington et al. (1986)	PCB-C3049	Winter Flounder (M)	BB-6	1987		0.055	Muscle	Prwell (unpub. data)
PCB-C3028	Winter Flounder	BB-6B	1979		0.021	Edible Fillets	Farrington et al. (1986)	PCB-C3052	Bay Scallop	BB-19	1978		0.001	Whole	Farrington et al. (1986)
PCB-C3028/031	Mussel	BB-19	1985	0.0004-0.003	0.002	Whole	Capuzzo & Farrington (unpub)	PCB-C3052	Lobster	BB-6B	1979	0-0.019	0.008	Muscle	Farrington et al. (1986)
PCB-C3028/031	Mussel	BB-6A	1985	0.154-0.394	0.28	Whole	Capuzzo & Farrington (unpub)	PCB-C3052	Lobster	BB-6B	1979	0.036-0.59	0.26	Viscera	Farrington et al. (1986)
PCB-C3040	Winter flounder (F)	BB-6	1987	0.009-0.193	0.102	Muscle	Prwell (unpub. data)	PCB-C3052	Mud Crab	BB-6B	1979		0.010	Whole	Farrington et al. (1986)
PCB-C3040	Winter flounder (M)	BB-6	1987		0.215	Muscle	Prwell (unpub. data)	PCB-C3052	Mussel	BB-19	1985	0.0004-0.003	0.002	Whole	Capuzzo & Farrington (unpub)
PCB-C3044	Bay Scallop	BB-19	1978		0.009	Whole	Farrington et al. (1986)	PCB-C3052	Mussel	BB-6A	1985	0.094-0.183	0.14	Whole	Capuzzo & Farrington (unpub)
PCB-C3044	Lobster	BB-6B	1979	0-0.013	0.005	Muscle	Farrington et al. (1986)	PCB-C3052	Mussel	BB-6A	1981		0.056	Whole	Farrington et al. (1986)
PCB-C3044	Lobster	BB-6B	1979	0.041-0.26	0.12	Viscera	Farrington et al. (1986)	PCB-C3052	Quahog	BB-6B	1978		0.013	Whole	Farrington et al. (1986)
PCB-C3044	Mud Crab	BB-6B	1979		0.003	Whole	Farrington et al. (1986)	PCB-C3052	Sand Flounder	BB-6B	1979		0.29	Edible Fillets	Farrington et al. (1986)
PCB-C3044	Mussel	BB-19	1985	0.0004-0.002	0.001	Whole	Capuzzo & Farrington (unpub)	PCB-C3052	Winter Flounder	BB-6B	1979		0.01	Edible Fillets	Farrington et al. (1986)
PCB-C3044	Mussel	BB-6A	1985	0.070-0.173	0.109	Whole	Capuzzo & Farrington (unpub)	PCB-C3052	Winter flounder (F)	BB-6	1987	0.002-0.054	0.023	Muscle	Prwell (unpub. data)
PCB-C3044	Mussel	BB-6A	1981		0.044	Whole	Farrington et al. (1986)	PCB-C3052	Winter flounder (M)	BB-6	1987		0.029	Muscle	Prwell (unpub. data)
PCB-C3044	Quahog	BB-6B	1978		0.015	Whole	Farrington et al. (1986)	PCB-C3060	Bay Scallop	BB-19	1978		0.0007	Whole	Farrington et al. (1986)
PCB-C3044	Sand Flounder	BB-6B	1979		0.24	Edible Fillets	Farrington et al. (1986)	PCB-C3060	Lobster	BB-6B	1979	0.001-0.05	0.023	Muscle	Farrington et al. (1986)
PCB-C3044	Winter Flounder	BB-6B	1979		0.011	Edible Fillets	Farrington et al. (1986)	PCB-C3060	Lobster	BB-6B	1979	0.15-1.30	0.64	Viscera	Farrington et al. (1986)
PCB-C3047	Winter flounder (F)	BB-6	1987	0.000-0.036	0.016	Muscle	Prwell (unpub. data)	PCB-C3060	Mud Crab	BB-6B	1979		0.022	Whole	Farrington et al. (1986)
PCB-C3047	Winter flounder (M)	BB-6	1987		0.049	Muscle	Prwell (unpub. data)	PCB-C3060	Mussel	BB-19	1985	0.00030.002	0.001	Whole	Capuzzo & Farrington (unpub)

PCB	Quahog	BB-6C	1986		0.08	Whole	Schwartz (1988)
PCB	Quahog	BB-7	1986		0.02	Whole	Schwartz (1988)
PCB	Quahog	BB-7	1986		0.06	Whole	Schwartz (1988)
PCB	Quahog	BB-7	1986		0.10	Whole	Schwartz (1988)
PCB	Quahog	BB-8	1986		0.03	Whole	Schwartz (1988)
PCB	Quahog	BB-8	1986		0.03	Whole	Schwartz (1988)
PCB	Quahog	BB-9	1986		0.01	Whole	Schwartz (1988)
PCB	Quahog	BB-9	1986		0.02	Whole	Schwartz (1988)
PCB	Quahog	BB-9	1986		0.04	Whole	Schwartz (1988)
PCB	Red hake	BB-6C	1979		0.1	Edible	Kolek & Cenrels (1981)
PCB	Red hake	BB-6C	1980		0	Edible	Weaver (1984)
PCB	Sand lance	BB-19	1974	.26-1.25	0.775	Whole	Nisbet & Reynolds (1984)
PCB	Scup	BB-6	1980	0-11.4	2.1	Edible	Weaver (1984)
PCB	Scup	BB-6A/6C	1979	1.3-2.3	1.8	Edible	Kolek & Cenrels (1981)
PCB	Scup	BB-6B	1976	7.9-10	8.95	Edible	Kolek & Cenrels (1981)
PCB	Scup	BB-6C	1980		0	Edible	Kolek & Cenrels (1981)
PCB	Shellfish	BB-6C	1979	.001-1.0	0.157	Edible	New Bedford 301(h)
PCB	Silver hake	BB-6	1980	0.7-6.4	3.5	Edible	Weaver (1984)
PCB	Silver hake	BB-6B	1976		0.7	Edible	Kolek & Cenrels (1981)
PCB	Soft-shelled clam	BB-6	1980?	0.75-14.0	4.7	Edible	Cantillo (1985)
PCB	Soft-shelled clam	BB-6A	1976	21-53	32	Edible	Kolek & Cenrels (1981)
PCB	Soft-shelled clam	BB-6A	1979		14.6	Edible	Kolek & Cenrels (1981)
PCB	Striped bass	BB-18	1986	0.36-7.5	1.5	Edible	MA DMF
PCB	Striped bass	BB-19/20	1977	0.1-0.5	0.3	Edible	MA DMF
PCB	Striped bass	BB-19/20	1979	0.2-0.5	0.283	Edible	MA DMF
PCB	Striped bass	BB-6	1977	0.4-3	1.46	Edible	MA DMF
PCB	Striped bass	BB-6	1980	0.1-3.0	1.2	Edible	Weaver (1984)
PCB	Striped bass	BB-6B	1977	.4-2.7	1.65	Edible	Kolek & Cenrels (1981)
PCB	Striped bass	MA-CCB	1977	0.5-1.3	0.7	Edible	MA DMF
PCB	Striped bass	NY	1977	0.2-0.8	0.375	Edible	MA DMF
PCB	Summer flounder	BB-6	1980	0.2-22	9.3	Edible	Weaver (1984)
PCB	Summer flounder	BB-6A/6B	1976	7.1-10	8.33	Edible	Kolek & Cenrels (1981)
PCB	Summer flounder	BB-6A/6B	1979	.2-2.1	1.15	Edible	Kolek & Cenrels (1981)
PCB	Summer flounder	BB-6C	1980	.3-4	2.15	Edible	Kolek & Cenrels (1981)
PCB	Tautog	BB-6	1980	0.1-11.0	1.7	Edible	Weaver (1984)
PCB	Tautog	BB-6B	1976		1.2	Edible	Kolek & Cenrels (1981)
PCB	Tautog	BB-6B/6C	1979	.3-1.3	0.76	Edible	Kolek & Cenrels (1981)
PCB	Tautog	BB-6B/6C	1980	.2-4.6	1.13	Edible	Kolek & Cenrels (1981)
PCB	Windowpane flounder	BB-6	1979	3.1-14.3	7.08	Edible	MA DMF
PCB	Windowpane flounder	BB-6	1981	2.5-5.1	3.37	Edible	MA DMF
PCB	Windowpane flounder	BB-6	1980	3.1-14.3	8.8	Edible	Weaver (1984)
PCB	Windowpane flounder	BB-6A/6C	1979	3.1-14.3	6.96	Edible	Kolek & Cenrels (1981)
PCB	Winter flounder	BB-19	1986	0.18-1.18	0.52	Edible	Schwartz (1988)
PCB	Winter flounder	BB-19	1986	0.04-0.91	0.39	Edible	Schwartz (1988)
PCB	Winter flounder	BB-20	1986		0.69	Liver	FDAA (1987)
PCB	Winter flounder	BB-20	1986	0.13-0.75	0.41	Edible	Schwartz (1988)
PCB	Winter flounder	BB-20	1985	0.3-26.8	5.43	Whole	Hillman et al. (1989)
PCB	Winter flounder	BB-20	1985	0.09-0.81	1.3	Edible	Hillman et al. (1989)
PCB	Winter flounder	BB-6A	1985	4.6-13.5	7.99	Whole	Hillman et al. (1989)
PCB	Winter flounder	BB-6A	1985	0.6-1.2	0.91	Edible	Hillman et al. (1989)
PCB	Winter flounder	BB-6B	1985	1.2-16.0	6.79	Whole	Hillman et al. (1989)
PCB	Winter flounder	BB-6B	1985	0.5-1.8	1.07	Edible	Hillman et al. (1989)
PCB	Winter flounder	BB-6C	1985	0.8-5.0	2.35	Whole	Hillman et al. (1989)
PCB	Winter flounder	BB-6C	1985	0.13-0.84	0.38	Edible	Hillman et al. (1989)
PCB	Winter flounder	BB-6	1980		0.08	Edible	Farrington (unpub)
PCB	Winter flounder	BB-6	1980		5.2	Edible	Farrington (unpub)
PCB	Winter flounder	BB-6	1980	0.2-5.9	2.4	Edible	MA DMF
PCB	Winter flounder	BB-6	1981	0.4-5.1	1.45	Edible	MA DMF

PCB	Lobster	BB-6A	1985		1.13	Whole	Hillman et al. (1989)	PCB	Quahog	B-18	1986		0.01	Whole	Schwartz (1988)
PCB	Lobster	BB-6A	1985	0.68-0.98	0.83	Muscle	Hillman et al. (1989)	PCB	Quahog	BB-10	1986		0.01	Whole	Schwartz (1988)
PCB	Lobster	BB-6A	1985	67.3-93.3	80.3	Hepatopancreas	Hillman et al. (1989)	PCB	Quahog	BB-10	1986		0.02	Whole	Schwartz (1988)
PCB	Lobster	BB-6A-6C	1980	0.1-84	8.7	Edible	Weaver (1984)	PCB	Quahog	BB-10	1986		0.01	Whole	Schwartz (1988)
PCB	Lobster	BB-6B	1985	0.13-0.69	0.37	Whole	Hillman et al. (1989)	PCB	Quahog	BB-10	1986		0.01	Whole	Schwartz (1988)
PCB	Lobster	BB-6B	1985	0.60-0.52	0.66	Muscle	Hillman et al. (1989)	PCB	Quahog	BB-11	1986		0.01	Whole	Schwartz (1988)
PCB	Lobster	BB-6B	1985	19.6-24.6	22.2	Hepatopancreas	Hillman et al. (1989)	PCB	Quahog	BB-12	1986		0.01	Whole	Schwartz (1988)
PCB	Lobster	BB-6C	1982	0.7-8.8	4.4	Edible	Weaver (1984)	PCB	Quahog	BB-12	1986		0.01	Whole	Schwartz (1988)
PCB	Lobster	BB-6C	1985	0.024-0.29	0.11	Whole	Hillman et al. (1989)	PCB	Quahog	BB-14	1986		0.01	Whole	Schwartz (1988)
PCB	Lobster	BB-6C	1985	0.12-0.34	0.23	Muscle	Hillman et al. (1989)	PCB	Quahog	BB-15	1986		ND	Whole	Schwartz (1988)
PCB	Lobster	BB-6C	1985	14.0-14.8	14.4	Hepatopancreas	Hillman et al. (1989)	PCB	Quahog	BB-15	1986		0.03	Whole	Schwartz (1988)
PCB	Lobster	H	1980		0.02	Muscle	Farrington (unpub)	PCB	Quahog	Bb-15	1986		ND	Whole	Schwartz (1988)
PCB	Lobster	H	1980		0.06	Viscera	Farrington (unpub)	PCB	Quahog	BB-16	1986		0.02	Whole	Schwartz (1988)
PCB	Mussel	BB	1974		0.06	Whole	Hisbet & Reynolds (1984)	PCB	Quahog	BB-16	1986		<0.01	Whole	Schwartz (1988)
PCB	Mussel	BB-13	1977		0.026	Whole	Farrington et al. (1982)	PCB	Quahog	BB-16	1986		<0.01	Whole	Schwartz (1988)
PCB	Mussel	BB-13	1976		0.03	Whole	Goldberg et al. (1978)	PCB	Quahog	BB-16	1986		0.01	Whole	Schwartz (1988)
PCB	Mussel	BB-18	1980		0.042	Whole	Farrington et al. (1985)	PCB	Quahog	BB-18	1986		0.01	Whole	Schwartz (1988)
PCB	Mussel	BB-5	1981		0.7	Whole	Farrington et al. (1982)	PCB	Quahog	BB-18	1986		0.01	Whole	Schwartz (1988)
PCB	Mussel	BB-5	1986		0.226	Whole	NOAA (1987)	PCB	Quahog	BB-18	1986		0.01	Whole	Schwartz (1988)
PCB	Mussel	BB-6A	1978		4.34	Whole	Farrington et al. (1982)	PCB	Quahog	BB-4	1986		0.05	Whole	Schwartz (1988)
PCB	Mussel	BB-6A	1981		3.08	Whole	Farrington et al. (1982)	PCB	Quahog	BB-5	1986		0.08	Whole	Schwartz (1988)
PCB	Mussel	BB-6A	1980		6.86	Whole	Farrington et al. (1982)	PCB	Quahog	BB-5	1986		0.07	Whole	Schwartz (1988)
PCB	Mussel	BB-6A	1974		3.72	Whole	Hisbet & Reynolds (1984)	PCB	Quahog	BB-6A-6C	1979	.3-1.6	1.73	Whole	Kolek & Ceurvels (1981)
PCB	Mussel	BB-6C	1986		0.95	Whole	NOAA (1987)	PCB	Quahog	BB-6B	1979	0.35-6.93	2.86	Whole	Deubert et al. (1981)
PCB	Mussel	BB-6C	1986		0.207	Whole	NOAA (1987)	PCB	Quahog	BB-6B	1980		0.37	Whole	Farrington (unpub)
PCB	Mussel	NY	1979		0.02	Whole	Farrington et al. (1982)	PCB	Quahog	BB-6B/6C	1978	0.029-.524	0.149	Whole	Batch et al. (1981)
PCB	Ocean quahog	BB	?		0.0037	Whole	EXCO (1983)	PCB	Quahog	BB-6B/6C-20	1976	.1-3.3	1.45	Whole	Kolek & Ceurvels (1981)
PCB	Oyster	BB-5	1978	0.07-0.321	0.214	Whole	Batch et al. (1981)	PCB	Quahog	BB-6C	1980		0.05	Whole	Farrington (unpub)
PCB	Oyster	BB-6A	1979		15.8	Whole	Kolek & Ceurvels (1981)	PCB	Quahog	BB-6C	1986		0.20	Whole	Schwartz (1988)

Heptachlor	Winter flounder	BB-10	1967	ND-1.42	0.465	Muscle	Smith & Cole (1970)	PCB	Four spot flounder	BB-6	1980		0.8		KA DKF
Naphthalene	Mussel	BB-19	1985	0.00004-0.0008	0.00015	Whole	Capuzzo & Farrington (unpub)	PCB	Four spot flounder	BB-6	1980		0.8	Edible	Weaver (1984)
Naphthalene	Mussel	BB-6A	1985	0.00003-0.001	0.00029	Whole	Capuzzo & Farrington (unpub)	PCB	Four spot flounder	BB-6C	1980		0.8	Edible	Kolek & Cearvels (1981)
PAH	Mussel	BB-13	1976		0.006	Whole	Farrington et al. (1982)	PCB	Lobster	BB	1980		0.8	Viscera	Farrington (unpub)
PAH	Mussel	BB-5	1986		0.039	Whole	NOAA (1987)	PCB	Lobster	BB	1980		0.01	Muscle	Farrington (unpub)
PAH	Mussel	BB-6A	1978		0.061	Whole	Farrington et al. (1982)	PCB	Lobster	BB-10	1985-1986	0.30-1.00	0.52		Schwartz (1988)
PAH	Mussel	BB-6C	1986		0.148	Whole	NOAA (1987)	PCB	Lobster	BB-14	1985-1986	0.10-0.50	0.26		Schwartz (1988)
PAH	Mussel	BB-6C	1986		0.230	Whole	NOAA (1987)	PCB	Lobster	BB-18	1985-1986	0.90-2.80	1.66		Schwartz (1988)
PAH	Soft-shelled clam	BB-6B	1980?	0.0053-0.073	0.036	Whole	Cantillo (1985)	PCB	Lobster	BB-18	1985-1986	0.4-2.38	1.04		Schwartz (1988)
PAH-Total	Mussel	BB-19	1985		0.014	Whole	Capuzzo & Farrington (unpub)	PCB	Lobster	BB-18	1985-1986	0.53-1.57	0.99		Schwartz (1988)
PAH-Total	Mussel	BB-6A	1985		0.386	Whole	Capuzzo & Farrington (unpub)	PCB	Lobster	BB-19	1985-1986	0.90-3.30	1.54		Schwartz (1988)
PCB	American eel	BB-6A/6B	1979	11-730	218	Edible	Kolek & Cearvels (1981)	PCB	Lobster	BB-19	1985-1986	0.75-1.50	1.08		Schwartz (1988)
PCB	American eel	BB-6A/6B	1976		92	Edible	Kolek & Cearvels (1981)	PCB	Lobster	BB-20	1985	0.019-0.15	0.068	Whole	Gillman et al. (1989)
PCB	Atlantic silverside	BB-19	1974	0.19-1.77	0.715	Whole	Hisbet & Reynolds (1984)	PCB	Lobster	BB-20	1985	0.033-0.12	0.064	Muscle	Gillman et al. (1989)
PCB	Bay scallop	BB-19	1978		0.0231	Edible	Natch et al. (1981)	PCB	Lobster	BB-20	1985	1.8-8.1	4.2	Hepatopancreas	Gillman et al. (1989)
PCB	Black sea bass	BB-6C	1979		0.4	Edible	Kolek & Cearvels (1981)	PCB	Lobster	BB-4	1977	ND-0.7	0.4		KA DKF
PCB	Blue crab	BB-6A	1979		4.2	Edible	Kolek & Cearvels (1981)	PCB	Lobster	BB-4	1985-1986	0.32-0.79	0.55		Schwartz (1988)
PCB	Blue crab	BB-6A/6B	1976	1-5.6	3.3	Edible	Kolek & Cearvels (1981)	PCB	Lobster	BB-6	1980		4.2	Viscera	Farrington (unpub)
PCB	Bluefish	BB	1977	0.4-3.9	1.74		KA DKF	PCB	Lobster	BB-6	1980		0.1	Muscle	Farrington (unpub)
PCB	Bluefish	BB-6	1977	1-1.6	1.3		KA DKF	PCB	Lobster	BB-6	1977	1-11.7	5.84	Edible	Kolek & Cearvels (1981)
PCB	Bluefish	BB-6	1979	1.4-16.5	8.95		KA DKF	PCB	Lobster	BB-6	1978	1-9.0	2.92		Kolek & Cearvels (1981)
PCB	Bluefish	BB-6	1980	0.2-18.5	2.1	Edible	Weaver (1984)	PCB	Lobster	BB-6	1979	.8-68.2	12.5		Kolek & Cearvels (1981)
PCB	Bluefish	BB-6B	1977	1.0-1.6	1.3	Edible	Kolek & Cearvels (1981)	PCB	Lobster	BB-6	1980	2.0-11	4.25	Edible	Kolek & Cearvels (1981)
PCB	Bluefish	BB-6B	1979	1.4-16.5	8.9	Edible	Kolek & Cearvels (1981)	PCB	Lobster	BB-6	1979	3-9.3	5.77	Edible	KA DKF
PCB	Bluefish	NY	1977	0.5-1.2	0.74		KA DKF	PCB	Lobster	BB-6	1980	0.5-11	3.56	Edible	KA DKF
PCB	Butterfish	BB-6	1980	ND-0.9	0.5	Edible	Weaver (1984)	PCB	Lobster	BB-6	1981	0.1-20.3	2.24	Edible	KA DKF
PCB	Cunner	BB-6	1980	20-57	38	Edible	Weaver (1984)	PCB	Lobster	BB-6	1982	0.3-23.8	3.87	Edible	KA DKF
PCB	Cunner	BB-6B	1976		20	Edible	Kolek & Cearvels (1981)	PCB	Lobster	BB-6	1983	1-9.3	3.7	Edible	KA DKF
PCB	Dogfish	BB-6	1980		0.2	Edible	Weaver (1984)	PCB	Lobster	BB-6	1985	1.1-11.4	4.35	Edible	KA DKF

Fish and Shellfish Contaminant Data from Buzzards Bay: Organic Contaminants

Chemical	Organism	Site	Date	Range (ww)	ppm, Wet Wt.	Tissue Analyzed	References
1-ethylnaphthal	Mussel	BB-19	1985	0.00007-0.00013	0.00006	Whole	Capuzzo & Farrington (unpub)
1-ethylnaphthal	Mussel	BB-6A	1985	0.0001-0.003	0.00061	Whole	Capuzzo & Farrington (unpub)
1-methylnaphtha	Mussel	BB-19	1985	0.000012-0.0002	0.00006	Whole	Capuzzo & Farrington (unpub)
1-methylnaphtha	Mussel	BB-6A	1985	0.00003-0.005	0.00012	Whole	Capuzzo & Farrington (unpub)
1-methylphenant	Mussel	BB-19	1985	0.0001-0.002	0.00052	Whole	Capuzzo & Farrington (unpub)
1-methylphenant	Mussel	BB-6A	1985	0.002-0.074	0.016	Whole	Capuzzo & Farrington (unpub)
1-methylpyrene	Mussel	BB-19	1985	0.00002-0.0001	0.00006	Whole	Capuzzo & Farrington (unpub)
1-methylpyrene	Mussel	BB-6A	1985	0.002-0.018	0.008	Whole	Capuzzo & Farrington (unpub)
2,3,6-trimethyl	Mussel	BB-19	1985	0.0002-0.0004	0.00029	Whole	Capuzzo & Farrington (unpub)
2,3,6-trimethyl	Mussel	BB-6A	1985	0.001-0.095	0.029	Whole	Capuzzo & Farrington (unpub)
2,6-dimethylnap	Mussel	BB-19	1985	0.00002-0.0002	0.00006	Whole	Capuzzo & Farrington (unpub)
2,6-dimethylnap	Mussel	BB-6A	1985	0.0002-0.029	0.007	Whole	Capuzzo & Farrington (unpub)
2-methylphenant	Mussel	BB-19	1985	0.0001-0.002	0.00047	Whole	Capuzzo & Farrington (unpub)
2-methylphenant	Mussel	BB-6A	1985	0.0004-0.139	0.029	Whole	Capuzzo & Farrington (unpub)
3,6-dimethylphe	Mussel	BB-19	1985	0.0001-0.001	0.00038	Whole	Capuzzo & Farrington (unpub)
3,6-dimethylphe	Mussel	BB-6A	1985	0.003-0.089	0.022	Whole	Capuzzo & Farrington (unpub)
Acenaphthene	Mussel	BB-6A	1985	0.0001-0.003	0.002	Whole	Capuzzo & Farrington (unpub)
Anthracene	Mussel	BB-6A	1985	0.0006-0.008	0.002	Whole	Capuzzo & Farrington (unpub)
Benzo[a]anthrac	Mussel	BB-19	1985	0.0003-0.001	0.00077	Whole	Capuzzo & Farrington (unpub)
Benzo[a]anthrac	Mussel	BB-6A	1985	0.008-0.127	0.057	Whole	Capuzzo & Farrington (unpub)
Benzo[a]pyrene	Mussel	BB-19	1985	0.00004-0.0002	0.00008	Whole	Capuzzo & Farrington (unpub)
Benzo[a]pyrene	Mussel	BB-6A	1985	0.0002-0.006	0.002	Whole	Capuzzo & Farrington (unpub)
Benzo[b]fluoran	Mussel	BB-19	1985	0.0001-0.0009	0.00035	Whole	Capuzzo & Farrington (unpub)
Benzo[b]fluoran	Mussel	BB-6A	1985	0.0002-0.034	0.014	Whole	Capuzzo & Farrington (unpub)
Benzo[c]phenant	Mussel	BB-19	1985	0.00006-0.0003	0.00016	Whole	Capuzzo & Farrington (unpub)
Benzo[c]phenant	Mussel	BB-6A	1985	0.0019-0.015	0.005	Whole	Capuzzo & Farrington (unpub)
Benzo[e]pyrene	Mussel	BB-19	1985	0.0002-0.0008	0.00044	Whole	Capuzzo & Farrington (unpub)
Benzo[e]pyrene	Mussel	BB-6A	1985	0.0062-0.021	0.012	Whole	Capuzzo & Farrington (unpub)
Benzo[g,h,i]per	Mussel	BB-19	1985	0.00004-0.0003	0.00018	Whole	Capuzzo & Farrington (unpub)
Benzo[g,h,i]per	Mussel	BB-6A	1985	0.0003-0.004	0.002	Whole	Capuzzo & Farrington (unpub)
Benzo[k]fluoran	Mussel	BB-19	1985	0.00008-0.0004	0.00017	Whole	Capuzzo & Farrington (unpub)
Benzo[k]fluoran	Mussel	BB-6A	1985	0.003-0.012	0.006	Whole	Capuzzo & Farrington (unpub)
Biphenyl	Mussel	BB-19	1985	0.00005-0.00017	0.00011	Whole	Capuzzo & Farrington (unpub)
Biphenyl	Mussel	BB-6A	1985	0.0002-0.0022	0.00085	Whole	Capuzzo & Farrington (unpub)
Chl.Pest.	Mussel	BB-5	1986		0.006	Whole	NOAA (1987)
Chl.Pest.	Mussel	BB-6C	1986		0.02	Whole	NOAA (1987)
Chl.Pest.	Mussel	BB-6C	1986		0.01	Whole	NOAA (1987)
Chl.Pest.	Winter Flounder	BB-20	1986		0.01	Liver	NOAA (1987)
Chrysene	Mussel	BB-19	1985		0.001	Whole	Capuzzo & Farrington (unpub)
Chrysene	Mussel	BB-6A	1985		0.029	Whole	Capuzzo & Farrington (unpub)
DDE	Mussel	BB-13	1976		0.00015	Whole	Goldberg et al. (1978)
DDT	Atlantic silverside	BB	1974	.04-.06	0.0525	Whole	Nisbet & Reynolds (1984)
DDT	Mussel	BB	1974		0	Whole	Nisbet & Reynolds (1984)
DDT	Mussel	BB-5	1986		0.013	Whole	NOAA (1987)
DDT	Mussel	BB-6C	1986		0.038	Whole	NOAA (1987)
DDT	Mussel	BB-6C	1986		0.006	Whole	NOAA (1987)
DDT	Sand lance	BB	1974	.03-.05	0.04	Whole	Nisbet & Reynolds (1984)
DDT	Winter Flounder	BB-20	1986		0.014	Liver	NOAA (1987)
DDT	Winter flounder	BB-10	1967	.08-1.07	0.293	Muscle	Smith & Cole (1970)
Dibenzothiophen	Mussel	BB-19	1985	0.00002-0.0002	0.00009	Whole	Capuzzo & Farrington (unpub)
Dibenzothiophen	Mussel	BB-6A	1985	0.0003-0.017	0.003	Whole	Capuzzo & Farrington (unpub)
Dieldrin	Winter flounder	BB-10	1967	ND-.05	0.0375	Muscle	Smith & Cole (1970)
Fluoranthene	Mussel	BB-19	1985	0.0012-0.015	0.0046	Whole	Capuzzo & Farrington (unpub)
Fluoranthene	Mussel	BB-6A	1985	0.003-0.194	0.065	Whole	Capuzzo & Farrington (unpub)
Fluorene	Mussel	BB-19	1985	0.00004-0.001	0.00039	Whole	Capuzzo & Farrington (unpub)
Fluorene	Mussel	BB-6A	1985	0.0002-0.016	0.003	Whole	Capuzzo & Farrington (unpub)

Pb	Lobster	BB-4	1986	0.06-0.12	0.10		DMPC (1988)	Se	Mussel	BB-6C	1986		0.32	Whole	NOAA (1987)
Pb	Mussel	BB-5	1986		0.13	Whole	NOAA (1987)	Se	Shellfish	BB-6	1979		2	Soft Parts	New Bedford 301(h)
Pb	Mussel	BB-6C	1986		0.64	Whole	NOAA (1987)	Se	Winter Flounder	BB-20	1986		1.11	Liver	NOAA (1987)
Pb	Mussel	BB-6C	1986		0.48	Whole	NOAA (1987)	Sn	Mussel	BB-5	1986		0.004	Whole	NOAA (1987)
Pb	Mussel	BB-13	1976		0.49	Whole	Goldberg et al. (1983)	Sn	Mussel	BB-6C	1986		0.02	Whole	NOAA (1987)
Pb	Mussel	BB-13	1977		0.504	Whole	Goldberg et al. (1983)	Sn	Mussel	BB-6C	1986		0.01	Whole	NOAA (1987)
Pb	Mussel	BB-13	1978		0.574	Whole	Goldberg et al. (1983)	Sn	Winter Flounder	BB-20	1986		0.06	Liver	NOAA (1987)
Pb	Quahog	BB-	1978-1979	nd-1.03	0.223	Edible	Gemest & Batch (1981)	Tl	Shellfish	BB-6	1979		0.07	Soft Parts	New Bedford 301(h)
Pb	Quahog	BB-10	1986	0.13-0.28	0.17	Soft Parts	DMPC (1988)	Sn	Anadara (bivalve)	BB-6			54	Soft Parts	Kelly (1978)
Pb	Quahog	BB-11	1986	0.12-0.28	0.18	Soft Parts	DMPC (1988)	Sn	Cancer crab	BB-6			20	Soft Parts	Kelly (1978)
Pb	Quahog	BB-12	1986	0.12-0.28	0.21	Soft Parts	DMPC (1988)	Sn	Channeled whelk	BB-6			228	Soft Parts	Kelly (1978)
Pb	Quahog	BB-14	1986		0.25	Soft Parts	DMPC (1988)	Sn	Crepidula (slipper	BB-6			19.6	Soft Parts	Kelly (1978)
Pb	Quahog	BB-15	1986	0.13-0.20	0.14	Soft Parts	DMPC (1988)	Sn	Lady crab	BB-6			23.4	Soft Parts	Kelly (1978)
Pb	Quahog	BB-16	1986	0.13-0.23	0.15	Soft Parts	DMPC (1988)	Sn	Mud crab	BB-6			14.2	Soft Parts	Kelly (1978)
Pb	Quahog	BB-18	1986	0.13-0.25	0.16	Soft Parts	DMPC (1988)	Sn	Mussel	BB-19	1985	8.0-15.0	11	Whole	Capuzzo & Farrington (unpub. data)
Pb	Quahog	BB-4	1986		0.15	Soft Parts	DMPC (1988)	Sn	Mussel	BB-5	1986		7.4	Whole	NOAA (1987)
Pb	Quahog	BB-5	1986		0.15	Soft Parts	DMPC (1988)	Sn	Mussel	BB-6A	1985	10.0-23.0	16	Whole	Capuzzo & Farrington (unpub. data)
Pb	Quahog	BB-6C	1986	0.13-0.33	0.23	Soft Parts	DMPC (1988)	Sn	Mussel	BB-6C	1986		14.6	Whole	NOAA (1987)
Pb	Quahog	BB-7	1986	0.13-0.28	0.22	Soft Parts	DMPC (1988)	Sn	Mussel	BB-6C	1986		15.0	Whole	NOAA (1987)
Pb	Quahog	BB-8	1986	0.13-0.25	0.17	Soft Parts	DMPC (1988)	Sn	Mussel	BB-13	1976		11.48	Whole	Goldberg et al. (1983)
Pb	Quahog	BB-9	1986	0.12-0.33	0.17	Soft Parts	DMPC (1988)	Sn	Mussel	BB-13	1977		11.2	Whole	Goldberg et al. (1983)
Pb	Shellfish	BB-6	1979	0.09-0.16	0.11	Soft Parts	New Bedford 301(h)	Sn	Mussel	BB-13	1978		11.48	Whole	Goldberg et al. (1983)
Pb	Winter Flounder	BB-19	1986		<0.13	Edible	DMPC (1988)	Sn	Quahog	BB-	1978-1979	0.23-37.08	15.12	Edible	Gemest & Batch (1981)
Pb	Winter Flounder	BB-20	1986		<0.13	Edible	DMPC (1988)	Sn	Quahog	BB-6			20.57	Soft parts	Kelly (1978)
Pb	Winter Flounder	BB-20	1986		0.29	Liver	NOAA (1987)	Sn	Shellfish	BB-6	1979	1.1-20.5	7.9	Soft parts	New Bedford 301(h)
Sb	Shellfish	BB-6	1979		0.7	Soft Parts	New Bedford 301(h)	Sn	Spider crab	BB-6			16.4	Soft parts	Kelly (1978)
Sb	Winter Flounder	BB-20	1986		0.003	Liver	NOAA (1987)	Sn	Winter Flounder	BB-20	1986		33.3	Liver	NOAA (1987)
Se	Mussel	BB-5	1986		0.30	Whole	NOAA (1987)								
Se	Mussel	BB-6C	1986		0.34	Whole	NOAA (1987)								

Hg	Lobster	NH-14	1986		0.04		DMFC (1988)	Ni	Mussel	NH-13	1985	1.0-1.8	4.1	Whole	Capuzzo & Ferriguelo (unpub. data)
Hg	Lobster	NH-13	1986		0.001-0.04		DMFC (1988)	Ni	Mussel	NH-5	1986		0.69	Whole	NOAA (1987)
Hg	Lobster	NH-20	1985		0.002		DMFC (1988)	Ni	Mussel	NH-14	1985	1.0-16.0	7.1	Whole	Capuzzo & Ferriguelo (unpub. data)
Hg	Lobster	NH-4	1986		0.01-0.04	0.023		Ni	Mussel	NH-9C	1986		0.14	Whole	NOAA (1987)
Hg	Mussel	NH-5	1986			0.01	Whole	Ni	Mussel	NH-9C	1986		0.27	Whole	NOAA (1987)
Hg	Mussel	NH-9C	1986			0.01	Whole	Ni	Mussel	NH-13	1976		0.112	Whole	Goldberg et al. (1983)
Hg	Mussel	NH-9C	1986			0.02	Whole	Ni	Mussel	NH-13	1977		0.056	Whole	Goldberg et al. (1983)
Hg	Quahog	NH-10	1986		0.005-0.010	0.009	Soft Parts	Ni	Mussel	NH-13	1979		0.252	Whole	Goldberg et al. (1983)
Hg	Quahog	NH-11	1986		0.005-0.008	0.006	Soft Parts	Ni	Quahog	NH-10	1986	0.015-0.33	0.10	Soft Parts	DNFC (1988)
Hg	Quahog	NH-12	1986		0.004-0.013	0.01	Soft Parts	Ni	Quahog	NH-11	1986	0.14-0.40	0.29	Soft Parts	DNFC (1988)
Hg	Quahog	NH-14	1986			0.005	Soft Parts	Ni	Quahog	NH-12	1986	0.13-0.21	0.14	Soft Parts	DNFC (1988)
Hg	Quahog	NH-15	1986		0.004-0.03	0.015	Soft Parts	Ni	Quahog	NH-14	1986		0.15	Soft Parts	DNFC (1988)
Hg	Quahog	NH-16	1986		0.003-0.093	0.028	Soft Parts	Ni	Quahog	NH-15	1986	0.13-0.43	0.23	Soft Parts	DNFC (1988)
Hg	Quahog	NH-18	1986		0.003-0.013	0.006	Soft Parts	Ni	Quahog	NH-16	1986	0.13-0.50	0.23	Soft Parts	DNFC (1988)
Hg	Quahog	NH-4	1986			0.004	Soft Parts	Ni	Quahog	NH-18	1986	0.13-0.15	0.14	Soft Parts	DNFC (1988)
Hg	Quahog	NH-5	1986		0.002-0.004	0.003	Soft Parts	Ni	Quahog	NH-4	1986		0.15	Soft Parts	DNFC (1988)
Hg	Quahog	NH-9C	1986		0.005-0.007	0.006	Soft Parts	Ni	Quahog	NH-5	1986	0.13-0.18	0.15	Soft Parts	DNFC (1988)
Hg	Quahog	NH-7	1986		0.004-0.050	0.013	Soft Parts	Ni	Quahog	NH-9C	1986		0.13	Soft Parts	DNFC (1988)
Hg	Quahog	NH-8	1986		0.004-0.015	0.004	Soft Parts	Ni	Quahog	NH-7	1986	0.13-0.23	0.16	Soft Parts	DNFC (1988)
Hg	Quahog	NH-9	1986		0.004-0.30	0.02	Soft Parts	Ni	Quahog	NH-8	1986	0.13-0.18	0.16	Soft Parts	DNFC (1988)
Hg	Shellfish	NH-6	1979		.01-0.102	0.031	soft parts	Ni	Quahog	NH-9	1986	0.12-0.28	0.17	Soft Parts	DNFC (1988)
Hg	Winter Flounder	NH-19	1986			0.01	Edible	Ni	Shellfish	NH-6	1979	0.75-0.6	2.6	soft parts	Fox Bedford 301(h)
Hg	Winter Flounder	NH-20	1986			0.005	Edible	Ni	Winter Flounder	NH-19	1986		<0.13	Edible	DNFC (1988)
Hg	Winter Flounder	NH-20	1986			0.05	Liver	Ni	Winter Flounder	NH-20	1986		<0.13	Edible	DNFC (1988)
Ni	Lobster	NH-10	1986		0.04-0.14	0.09		Ni	Winter Flounder	NH-20	1986		0.07	Liver	NOAA (1987)
Ni	Lobster	NH-14	1986			<0.04		Pb	Lobster	NH-10	1986	0.05-0.09	0.04		DNFC (1988)
Ni	Lobster	NH-19	1986		0.04-0.25	<0.15		Pb	Lobster	NH-14	1986		0.04		DNFC (1988)
Ni	Lobster	NH-20	1986			<0.25		Pb	Lobster	NH-19	1986	0.05-0.25	<0.15		DNFC (1988)
Ni	Lobster	NH-4	1986		0.04-0.55	0.15		Pb	Lobster	NH-20	1986		<0.25		DNFC (1988)

Cr	Mussel	BB-19	1985	1.0-16.0	4.7	Whole	Capuzzo & Farrington (unpub. data)	Cu	Mussel	BB-5	1984		0.78	Whole	NOAA (1987)
Cr	Mussel	BB-5	1986		0.04	Whole	NOAA (1987)	Cu	Mussel	BB-6A	1985	1.0-3.0	1.9	Whole	Capuzzo & Farrington (unpub. data)
Cr	Mussel	BB-6A	1985	2.0-28.0	3.7	Whole	Capuzzo & Farrington (unpub. data)	Cu	Mussel	BB-6C	1986		1.41	Whole	NOAA (1987)
Cr	Mussel	BB-6C	1986		0.19	Whole	NOAA (1987)	Cu	Mussel	BB-6C	1986		1.40	Whole	NOAA (1987)
Cr	Mussel	BB-6C	1986		0.20	Whole	NOAA (1987)	Cu	Mussel	BB-13	1976		0.602	Whole	Goldberg et al. (1983)
Cr	Quahog	BB-10	1986	0.008-0.15	0.09	Soft Parts	DMPC (1988)	Cu	Mussel	BB-13	1977		0.954	Whole	Goldberg et al. (1983)
Cr	Quahog	BB-11	1986	0.05-0.16	0.09	Soft Parts	DMPC (1988)	Cu	Mussel	BB-13	1978		4.9	Whole	Goldberg et al. (1983)
Cr	Quahog	BB-12	1986	0.058-0.075	0.07	Soft Parts	DMPC (1988)	Cu	Quahog	BB-10	1986	0.33-0.65	0.50	Soft Parts	DMPC (1988)
Cr	Quahog	BB-14	1986		<0.08	Soft Parts	DMPC (1988)	Cu	Quahog	BB-11	1986	0.43-0.83	0.63	Soft Parts	DMPC (1988)
Cr	Quahog	BB-15	1986	0.05-0.08	0.07	Soft Parts	DMPC (1988)	Cu	Quahog	BB-12	1986	0.30-1.08	0.73	Soft Parts	DMPC (1988)
Cr	Quahog	BB-16	1986	0.05-0.23	0.095	Soft Parts	DMPC (1988)	Cu	Quahog	BB-14	1986		0.30	Soft Parts	DMPC (1988)
Cr	Quahog	BB-18	1986		<0.08	Soft Parts	DMPC (1988)	Cu	Quahog	BB-15	1986	0.35-0.73	0.54	Soft Parts	DMPC (1988)
Cr	Quahog	BB-4	1986		<0.08	Soft Parts	DMPC (1988)	Cu	Quahog	BB-16	1986	0.25-2.50	0.87	Soft Parts	DMPC (1988)
Cr	Quahog	BB-5	1986		<0.08	Soft Parts	DMPC (1988)	Cu	Quahog	BB-18	1986	0.13-0.33	0.22	Soft Parts	DMPC (1988)
Cr	Quahog	BB-6C	1986		<0.08	Soft Parts	DMPC (1988)	Cu	Quahog	BB-4	1986		0.28	Soft Parts	DMPC (1988)
Cr	Quahog	BB-7	1986		<0.08	Soft Parts	DMPC (1988)	Cu	Quahog	BB-5	1986	0.23-0.28	0.25	Soft Parts	DMPC (1988)
Cr	Quahog	BB-8	1986	0.05-0.08	0.07	Soft Parts	DMPC (1988)	Cu	Quahog	BB-6C	1986	0.48-0.53	0.50	Soft Parts	DMPC (1988)
Cr	Quahog	BB-9	1986	0.073-0.185	0.10	Soft Parts	DMPC (1988)	Cu	Quahog	BB-7	1986	0.25-0.85	0.45	Soft Parts	DMPC (1988)
Cr	Shellfish	BB-6	1979	1.27-4.2	2.9	soft parts	New Bedford 301(h)	Cu	Quahog	BB-8	1986	0.23-0.78	0.43	Soft Parts	DMPC (1988)
Cr	Winter Flounder	BB-19	1986	0.08-0.15	0.09	Edible	DMPC (1988)	Cu	Quahog	BB-9	1986	0.48-1.15	0.75	Soft Parts	DMPC (1988)
Cr	Winter Flounder	BB-20	1986		<0.08	Edible	DMPC (1988)	Cu	Quahog	BB-	1978-1979	0.78-4.02	2.088	Edible	Gennet & Batch (1981)
Cr	Winter Flounder	BB-20	1986		0.04	Liver	NOAA (1987)	Cu	Quahog	BB-6			5.64	Soft Parts	Kelly (1978)
Cu	Anadora (bivalve)	BB-6			8.7	Soft parts	Kelly (1978)	Cu	Shellfish	BB-6	1979	0.23-1.80	0.9	Soft Parts	New Bedford 301(h)
Cu	Cancer crab	BB-6			14.6	Soft parts	Kelly (1978)	Cu	Spider crab	BB-6			10.4	Soft Parts	Kelly (1978)
Cu	Channeled whelk	BB-6			86	Soft parts	Kelly (1978)	Cu	Winter Flounder	BB-19	1986		0.18	Edible	DMPC (1988)
Cu	Crepidula (slipper	BB-6			30	Soft parts	Kelly (1978)	Cu	Winter Flounder	BB-20	1986		0.10	Edible	DMPC (1988)
Cu	Lady crab	BB-6			12.2	Soft parts	Kelly (1978)	Cu	Winter Flounder	BB-20	1986		3.92	Liver	NOAA (1987)
Cu	Mud crab	BB-6			9.8	Soft parts	Kelly (1978)	Fe	Quahog	BB-	1978-1979	7.09-46.8	18.12	Edible	Gennet & Batch (1981)
Cu	Mussel	BB-19	1985	1.0-2.0	1.5	Whole	Capuzzo & Farrington (unpub. data)	Hg	Lobster	WB-10	1986	0.02-0.04	0.03		DMPC (1988)

Fish and Shellfish Contaminant Data from Buzzards Bay: Trace Metal Contaminants

Trace Metal	Organism	Site	Date	Range (ww)	ppm, Wet Wt.	Tissue Analyzed	References
Ag	Mussel	BB-5	1986		0.01	Whole	NOAA (1987)
Ag	Mussel	BB-6C	1986		0.03	Whole	NOAA (1987)
Ag	Mussel	BB-6C	1986		0.05	Whole	NOAA (1987)
Ag	Mussel	BB-13	1976		0.0084	Whole	Goldberg et al. (1983)
Ag	Mussel	BB-13	1977		0.007	Whole	Goldberg et al. (1983)
Ag	Mussel	BB-13	1978		0.007	Whole	Goldberg et al. (1983)
Ag	Shellfish	BB-6	1979	0.07-0.43	0.19	Soft Parts	New Bedford 301(h)
Ag	Shellfish	BB-6	1979	1.6-3.9	3.1	Soft Parts	New Bedford 301(h)
Ag	Winter Flounder	BB-20	1986		0.09	Liver	NOAA (1987)
As	Mussel	BB-5	1986		1.17	Whole	NOAA (1987)
As	Mussel	BB-6C	1986		1.91	Whole	NOAA (1987)
As	Mussel	BB-6C	1986		1.82	Whole	NOAA (1987)
As	Winter Flounder	BB-20	1986		5.11	Liver	NOAA (1987)
Be	Shellfish	BB-6	1979	.05-0.170	0.125	Soft Parts	New Bedford 301(h)
Cd	Anadora (bivalve)	BB-6			3.45	Soft Parts	Kelly (1978)
Cd	Cancer crab	BB-6			0.4	Soft Parts	Kelly (1978)
Cd	Channeled whelk	BB-6			2.6	Soft Parts	Kelly (1978)
Cd	Crepidula (slipper	BB-6			0.56	Soft Parts	Kelly (1978)
Cd	Lady crab	BB-6			0.5	Soft Parts	Kelly (1978)
Cd	Lobster	BB-10	1986	0.16-0.25	0.21		DMPC (1988)
Cd	Lobster	BB-14	1986	0.11-0.15	0.14		DMPC (1988)
Cd	Lobster	BB-19	1986	0.16-0.43	0.19		DMPC (1988)
Cd	Lobster	BB-20	1986		<0.18		DMPC (1988)
Cd	Lobster	BB-4	1986	0.11-0.25	<0.05		DMPC (1988)
Cd	Mud crab	BB-6			0.106	Soft Parts	Kelly (1978)
Cd	Mussel	BB-5	1986		0.05	Whole	NOAA (1987)
Cd	Mussel	BB-6C	1986		0.18	Whole	NOAA (1987)
Cd	Mussel	BB-6C	1986		0.17	Whole	NOAA (1987)
Cd	Mussel	BB-13	1976		0.266	Whole	Goldberg et al. (1983)
Cd	Mussel	BB-13	1977		0.238	Whole	Goldberg et al. (1983)
Cd	Mussel	BB-13	1978		0.336	Whole	Goldberg et al. (1983)
Cd	Quahog	BB-10	1986	0.005-0.10	0.05	Soft Parts	DMPC (1988)
Cd	Quahog	BB-11	1986	0.04-0.07	0.05	Soft Parts	DMPC (1988)
Cd	Quahog	BB-12	1986	0.03-0.05	0.04	Soft Parts	DMPC (1988)
Cd	Quahog	BB-14	1986		<0.05	Soft Parts	DMPC (1988)
Cd	Quahog	BB-15	1986	0.05-0.08	0.06	Soft Parts	DMPC (1988)
Cd	Quahog	BB-16	1986	0.05-0.08	0.063	Soft Parts	DMPC (1988)
Cd	Quahog	BB-18	1986		<0.05	Soft Parts	DMPC (1988)
Cd	Quahog	BB-4	1986		<0.05	Soft Parts	DMPC (1988)
Cd	Quahog	BB-5	1986		0.05	Soft Parts	DMPC (1988)
Cd	Quahog	BB-6C	1986		0.05	Soft Parts	DMPC (1988)
Cd	Quahog	BB-7	1986		<0.05	Soft Parts	DMPC (1988)
Cd	Quahog	BB-8	1986	0.05-0.10	0.07	Soft Parts	DMPC (1988)
Cd	Quahog	BB-9	1986	0.03-0.05	0.04	Soft Parts	DMPC (1988)
Cd	Quahog	BB-	1978-1979	nd-0.338	0.139	Edible	Gemest & Batch (1981)
Cd	Quahog	BB-6			0.144	Soft parts	Kelly (1978)
Cd	Shellfish	BB-6	1979	0.04-0.52	0.29	soft parts	New Bedford 301(h)
Cd	Spider crab	BB-6			0.134	Soft parts	Kelly (1978)
Cd	Winter Flounder	BB-19	1986		<0.05	Edible	DMPC (1988)
Cd	Winter Flounder	BB-20	1986		<0.05	Edible	DMPC (1988)
Cd	Winter Flounder	BB-20	1986		0.10	Liver	NOAA (1987)
Cr	Lobster	BB-10	1986	0.04-0.06	0.05		DMPC (1988)
Cr	Lobster	BB-14	1986		0.03		DMPC (1988)
Cr	Lobster	BB-19	1986	0.03-0.15	<0.08		DMPC (1988)
Cr	Lobster	BB-20	1986		<0.15		DMPC (1988)
Cr	Lobster	BB-4	1986	0.04-0.08	<0.06		DMPC (1988)

Appendix References

Camp Dresser and McKee, Inc. 1979. Report for the City of New Bedford, Massachusetts, Section 301 (h) Application for Modification of Secondary Treatment Requirements for Discharges into Marine Waters, Vol 2. Camp Dresser and McKee Inc., Boston, MA, various paginations.

Cantillo, A.Y. 1985. Trace Contaminant Levels in Tissues. Draft report of results of Northeast Monitoring Program Rockville, Md.: Ocean Assessments Division, National Ocean Service, National Oceanic and Atmospheric Administration.

Deubert, K.H., P. Rule, and I. Corte-Real. 1981. PCB residues in *Mercenaria mercenaria* from New Bedford Harbor, 1978. Bulletin of Environmental Contamination and Toxicology 27: 683-688.

DWPC. 1988. Report to the Buzzards Bay Project, U.S. Environmental Protection Agency, from the Division of Water Pollution Control, Westborough, MA.

ERCO (Energy Resource Company, Inc.). 1983. Organic Pollutant Levels in the Ocean Quahog (*Arctica islandica*) from the Northeastern United States. Task 1, Modification No. 3, Contract NA-81-FA-C-0013, National Oceanic and Atmospheric Administration.

Farrington, J.W., R.W. Risebrough, P.L. Parker, A.C. Davis, B. deLappe, J.K. Winters, D.Boatwright, and N.M. Frew. 1982. Hydrocarbons, Polychlorinated Biphenyls, and DDE in Mussels and Oysters from the U.S. Coast, 1976-1978. The Mussel Watch. Technical Report No. WHOI 82-42. Woods Hole Oceanographic Institution, Woods Hole, MA, 106 pp.

Farrington, J.W., A.C. Davis, B.J. Brownawell, B.W. Tripp, C.H. Clifford, and J.B. Livramento. 1986. The biogeochemistry of polychlorinated biphenyls in the Acushnet River Estuary, Massachusetts. In: M.L. Sohn (ed.), Organic Marine Geochemistry, ACS Symposium Series 305. American Chemical Society, Washington, DC, pp. 174-197.

Genest, P.E. and W.I. Hatch. 1981. Heavy metals in *Mercenaria mercenaria* and sediments from the New Bedford Harbor region of Buzzard's Bay, Massachusetts. Bulletin of Environmental Contamination and Toxicology 26: 124-130.

Goldberg, E.D., M. Koide, V. Hodge, A.R. Flegal, and J. Martin. 1983. U.S. Mussel Watch: 1977-78 results on trace metals and radionuclides. Estuarine, Coastal and Shelf Science 16: 69-93.

Goldberg, E., V.T. Bowen, J.W. Farrington, G.R. Harvey, J.H. Martin, P.L. Parker, R.W. Risebrough, W.Robertson IV, E.Schneider, and E. Gamble. 1978. The Mussel Watch. Environmental Conservation 5 (2): 101-125.

Hatch, W.I., D.W. Allen, P.D. Brady, A.C. Davis, and J.W. Farrington. 1981. Polychlorinated biphenyls in clams and oysters from New Bedford Harbor, Massachusetts, March 1978. Pesticides Monitoring Journal 15: 123-127.

Hillman, R.E., M.S. Connor, and R.A. McGrath. 1989. Ratios of Edible Tissue to Whole Body PCB Concentrations in Flounder and Lobster from New Bedford Harbor. Final report submitted to the Buzzards Bay Project, U.S. Environmental Protection Agency, from Battelle Ocean Sciences, Duxbury, MA.

Kelly, B. 1978. Metal Concentrations of Some of the Macrobenthos of New Bedford Harbor and Evaluation of the Use of Slipper Limpet *Crepidula fornicata* as a Monitor of Coastal Metal Pollution. M.S. Thesis, Southeastern Massachusetts University, North Dartmouth, MA, 83 pp.

Kolek, A. and R. Ceurvels. 1981. PCB in Marine Organisms in New Bedford, 1976-1980. Massachusetts Division of Marine Fisheries, Boston, MA.

Nisbet, I.C.T. and L.M. Reynolds. 1984. Organochlorine residues in common terns and associated estuarine organisms, Massachusetts, USA, 1971-1981. Marine Environmental Research 11: 33-66.

NOAA. 1987. A Summary of Selected Data on Chemical Contaminants in Tissues Collected during 1984, 1985, and 1986. NOAA Technical Memorandum NOS OMA 38. Ocean Assessment Division, Office of Oceanography and Marine Assess-

ment,National Oceanic and Atmospheric Administration, U.S. Department of Commerce, Rockville, MD, 23pp. plus appendices.

Schwartz, J.P. 1988. Distribution and Concentration of Polychlorinated Biphenyls in Lobster, Winter Flounder, and Quahogs from Buzzards Bay, Massachusetts. Report submitted to U.S. EPA Buzzards Bay Program from Massachusetts Division of Marine Fisheries.

Smith, R.M. and C.F. Cole. 1970. Chlorinated hydrocarbon insecticide residues in winter flounder, *Pseudopleuronectes americanus*, from the Weweantic River estuary, Massachusetts. Journal of the Fisheries Research Board of Canada 27: 2374-2380.

Weaver, G. 1984. PCB contamination in and around New Bedford, Mass. Environmental Science and Technology 18: 22A-27A.

Chapter 4

Characterization of pollutant inputs to Buzzards Bay[1]

Christine Werme[2], Brigitte Hilbig[3], Anne Giblin[4]

Introduction

Buzzards Bay, a coastal embayment located between Cape Cod and southeastern Massachusetts, and its watershed serve as a transportation corridor for ships and barges, a vacationland for tourists, a site for businesses and industries, and a home for the many residents of the 18 towns within the watershed.

Residential, commercial, and industrial use of the Bay and its watershed have put pressure upon the region"pressures that left unchecked may threaten the marine environment and public health. Consequently, the Buzzards Bay Project (BBP), under joint management of the U.S. Environmental Protection Agency and the Massachusetts Executive Office of Environmental Affairs, is identifying and researching priority water quality problems in Buzzards Bay and developing a management plan for the future protect of resources and human health.

The BBP has identified three priority pollution problems:

- Closure of shellfish beds due to pathogen contamination.
- Eutrophication due to nutrient enrichment.
- Toxic contamination of fish lobster, and shellfish and the effects of this contamination on humans.

Various action plans are being developed to mitigate these problems. One action is to monitor these problems to assess and document the success of the other action plans.

Developing a monitoring program requires characterization of the pollutant inputs of interest: pathogens, nutrients, and toxic compounds (Table 1). These pollutants enter Buzzards Bay from point and nonpoint sources. Point sources include all sources for which National Pollutant Discharge Elimination System (NPDES) permits have been issued. Although these discharges can be identified for the Buzzards Bay watershed, little information is available on their contributions of contaminants to the Bay (Table 2). Nonpoint sources of contaminants to the Bay include atmospheric deposition, urban and non-urban runoff, and groundwater flow. This chapter summarizes information about these inputs.

[1] *This report was prepared by Science Applications International Corporation, Woods Hole Massachusetts, 02543 for the Buzzards Bay Project.* [2] *439 Grove St., Norwell, MA 02061,* [3] *Science Applications International Corporation, Woods Hole Massachusetts, 02543,* [4] *Ecosystems Center, Marine Biological Laboratory, Woods Hole, MA 02543*

Table 1. Contaminants in Buzzards Bay

- Pathogen Contamination
  - Coliforms
  - Other Indicators
- Nutrient Enrichment
  - Nitrogen
  - Phosphorus
- Toxic Contamination
  - Petroleum and Fossil Fuels
  - Pesticides
  - PCBs
  - Metals
    - Cadmium, Lead, Mercury, Copper, Arsenic, Chromium, Silver, Nickel

Pathogen Contamination

Human pathogens comprise two groups of organisms, bacteria and viruses. While bacteria have been monitored for many years to assess risks for human health, there are no routine methods available to address contamination with viruses. The number of virus species occurring in human wastes is very high, and little is known about their behavior in the environment outside their hosts.

In the soil, viruses are more viable under lower temperature and higher moisture. Other important parameters are pH and concentration of metallic cations. These parameters influence the ability of a virus to adsorb to sediment particles and prolong survival. In water, the important environmental parameters are the same as in soil. Exposure to sunlight also decreases the viability of viruses.

The most common bacterial group used as indicators for human pathogens are the fecal coliforms. These organisms are not pathogens but are present in high numbers in human wastes and may indicate the presence of human pathogens.

Methods used for fecal coliform counts are the membrane-filter technique (MF) and the multiple-tube-fermentation technique (MPN). The MF technique consists of passing a defined volume of water through a filter membrane, transferring the filter onto culturing medium, and counting the resulting bacterial colonies after a time of incubation. The MPN technique includes incubation of a series of dilutions of the water sample in tubes and statistical treatment of the resulting numbers of bacterial cultures; this method is required by the Commonwealth of Massachusetts for closure of shellfish beds, but both methods are used in current monitoring studies.

Lack of comparability of the two methods presents some difficulties in the evaluation of the data, including different detection limits (MF: fecal coliforms/100 ml, MPN: fecal coliforms/100 ml). In addition, data from MPN tests are often presented as "<x" or ">x" and may therefore be difficult to summarize and compare with other studies. Moreover, analyses at a given location conducted by different groups often produce different results that may affect the practical aspect of bacterial monitoring by causing conflicting decisions in terms of shellfish bed closures (Figure 1).

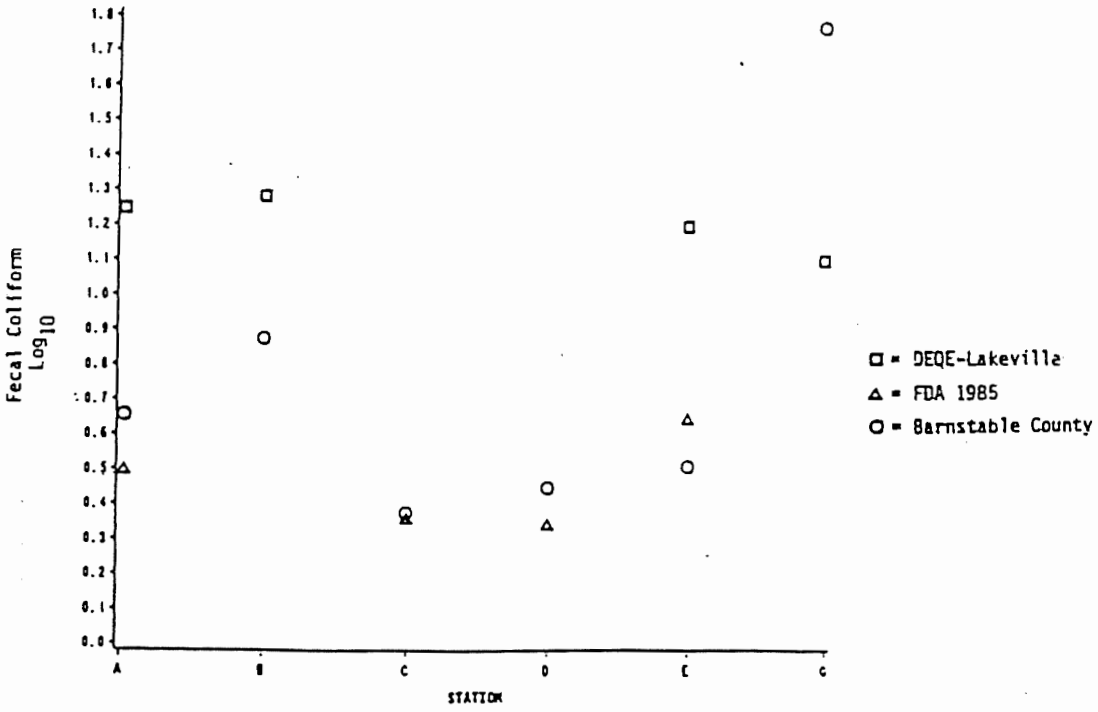

Figure 1. Comparison of three data sets originating from different sampling efforts at six stations in Buttermilk Bay (from Stenner et al., 1988).

Evaluation of Bacterial Indicators

In recent years, it has become obvious that fecal coliforms are not only non-specific for human wastes, but also occur in non-fecal material such as decaying plants (Heufelder, 1988). Fecal coliforms are therefore not a reliable indicator for human pathogens. Consequently, attempts have been made to identify alternatives.

Escherichia coli

E. coli is a species of fecal coliforms, thought to be more specific to fecal material than, for example, the genus *Klebsiella*. However, *E. coli* has been found in non-fecal sources and therefore may not always be related to human waste. Tests for *E. coli* produce definite results only if they are negative, i.e., if the source is non-fecal.

Clostridium perfringens

C. perfringens is being considered as an indicator for human pathogens because it is always present in human wastes and because the spores can be recovered by certain culturing methods, so that contamination sources can be detected after fecal coliform die-off. *C. perfringens* was thought to be highly specific to human wastes, but it has been found in high concentrations in dog wastes and in beach wrack, supposedly originating from waterfowl droppings. This bacterium is therefore only useful for monitoring if the source is known, e.g., a septic system.

Fecal Streptococci

A subgroup of the fecal streptococci, the enterococci, are positively related to incidence of gastrointestinal diseases among swimmers and are used as indicators for human pathogens by a number of states. Heufelder (1988) reported that in Buttermilk Bay the results were inconclusive: it was not clear whether or not enterococci numbers correlated with the number of human sources for enteric bacteria.

Although an alternative to fecal coliforms is desirable for monitoring health hazards introduced by human waste, none of the organisms tested to date has proven to be specific to humans. Moreover, there are no conclusive or inexpensive and simple tests for any of these organisms. Fecal coliform counts are still the best indicator for monitoring human pathogens. Until more of the biology of enteric bacteria is known, conservative measures will be the only way to ensure that human health is not threatened by consumption of contaminated seafood or by direct contact with contaminated water during recreational activities.

Sources of pathogens to Buzzards Bay include (1) sewage outfalls; (2) onsite septic systems; (3) stormwater runoff; (4) marinas; (5) freshwater streams; (6) marsh sediments and beach wrack; (7) wildlife, waterfowl, and domestic animals; (8) agricultural runoff; and (9) CSOs. Case studies conducted in Wareham, Westport, Buttermilk Bay, and Bourne can be used to demonstrate the magnitude of these sources.

Sewage Outfalls

Publicly owned treatment works (POTWs), or sewage treatment facilities, are required to disinfect all wastes before discharging into the water. The efficiency of the disinfection is monitored under the National Pollutant Discharge Elimination System (NPDES) permit system, so that theoretically no pathogens should be transferred with the wastes. However, occasional failures and malfunctions make POTWs still a potential source for pathogens in the bay. Approximately 37 million gallons of wastes are discharged daily into Buzzards Bay; by far the greatest portion (about 30 million gallons) comes from New Bedford (Grimes and Heufelder, BBP, in press).

The New Bedford plant receives wastewater from the City of New Bedford and parts of the Towns of Dartmouth and Acushnet. The average flow is 29-30 million gallons/day (MGD).

Case study: Wareham

In June 1986, samples were taken upstream and downstream of the Wareham POTW. Fecal coliform loadings were generally moderately elevated; none of the samples contained more than 500 fecal coliforms per 100 ml. No difference between the upstream and downstream stations was apparent. Fecal streptococci loadings were only slightly elevated as well; the upstream station and about half of the downstream stations contained less than 200 colonies/100 ml. The remaining downstream samples contained between 200 and 500 fecal streptococci/100 ml. While these loadings make the area sur-

rounding the Wareham POTW unsuitable for swimming and shellfishing, it not known how much of the contamination originates from the treatment plant. Both the relatively high numbers of fecal streptococci and the similarity of upstream and downstream loadings may point to a nonhuman source of bacterial contamination.

Septic Systems

The siting of residential septic systems is regulated by the Commonwealth so that bacteria are retained by the soil before the wastes come into contact with either groundwater or surface water. Enforcement of the State regulations is the responsibility of local Boards of Health. Although only occasional failures of the septic systems release bacteria into the water, viruses appear to withstand soil retention and may enter the groundwater more often than previously thought (Yates, 1987 in Grimes and Heufelder, BBP, in press). Severe failures of septic systems are usually very short-termed because of the accompanying odor problems (Heufelder, 1988).

Table 2. Major and minor NPDES dischargers in the Buzzards Bay watershed.

Facility	Maj/Min	Receiving Water	Parameters, Comments
Acushnet Capacitor	Minor	Acushnet R. Estuary	
Acushnet Co. Ribber Div.	Minor	Acushnet R. Estuary	
Acushnet Co. Tileist Golf Div.	Minor	Acushnet R. Estuary	
Acushnet Nursing, Inc.	Minor	Acushnet R. Estuary	
Aerovox Corporation	Major	Acushnet R. Estuary	Permit expired June 1980
Commonwealth Electric Cannon	Major	New Bedford Harbor	Flow, oil & grease
Cornell-Dublier Electric Corp.	Major	Acushnet River	Flow, oil & grease, PCBs
Dartmouth WPCF	Major	Buzzards Bay	Flow, total coliforms, permit exp. December 1989
Don Adams Oil Co.	Minor	Slocums River	
Fairhaven WPCF	Major	Acushnet R. Estuary	Flow, total coliforms
Franconia Fuel Co., Inc.	Minor	Wareham River	Permit expired April 1981
Glen Petroleum Corp.	Minor	Acushnet R. Estuary	
Goodyear Tire & Rubber Co.	Minor	Clarks Cove	
Isotronics, Inc.	Minor	Buzzards Bay	
John Dugan Buick	Minor		
Lincoln Park Inc.	Minor	Westport River	Permit expired September 1981
Lobster Trap Co.	Minor	Back River	
Marion, Town of	Major	Aucoot Bay	Flow, fecal coliforms
Maritime Terminal, Inc.	Minor	Acushnet R. Estuary	Permit expired November 1984
New Bedford WTP	Major	New Bedford Harbor	Flow, oil & grease, PCBs, fecal coliforms, VOCs
Revere Copper Products, Inc.	Major	Acushnet R. Estuary	Cr (total and hexavalent), Cu, Pb, Ni, Zn
Shawmut Avenue Landfill	Minor	Appogansett Swamp	
Skipper Motor Inn, Inc.	Minor	Acushnet R. Estuary	
Teledyne-Rodney Metals	Minor	New Bedford Harbor	
Tilcon Massachusetts, Inc.	Minor	Acushnet River	
Tremont Nail Co., Inc	Minor	Wankino River	
Wareham WTF	Major	Agawam River	No current permit

Case study: Westport

In the Westport River area, two sampling locations for surface water were identified characterized by residences in and out of compliance with Title V. (Title V is a state regulation governing the placement of cesspools and septic tanks in relation to surface waters.) Samples were taken in July and August 1986. The amounts of fecal coliforms found at the within-compliance stations were as high as 10 colonies/100 ml in July and 100 colonies/100 ml in August. Loadings at the out-of-compliance stations were as high as 100 colonies/100 ml in July and more than 500 colonies/100 ml in August. The concentrations of fecal streptococci were not as high, but similar in the overall pattern.

These results show that improperly installed septic systems present an environmental threat in the Buzzards Bay area. The study was conducted during days with very little rainfall (0.1 to 0.6 in). Under heavy precipitation, these septic systems may cause much higher bacterial contamination through stormwater runoff.

Stormwater Runoff

Stormwater, i.e., precipitation, is generally free of pathogens while falling. However, while flowing into rivers and streams and ultimately into the Bay, stormwater runoff receives bacteria and viruses from a variety of sources, such as pet and livestock wastes and wastes of wildlife and waterfowl. Currently, contamination from stormwater runoff is the major reason for shellfish bed closures in Massachusetts (Heufelder, 1988).

In response to the environmental threat posed to many coastal areas by stormwater, numerous investigations have focused on the assessment and monitoring of stormwater runoff during the last ten to 20 years. Aside from reporting bacterial loadings in soil and water, many attempts have been made to track back the source of contamination from these findings. However, results are often inconclusive or difficult to generalize because several factors influencing the bacterial flora in stormwater runoff must be considered before the results can be interpreted.

Case study: Bourne

Gale Associates (1989) characterized sources of fecal coliform contamination of stormwater in the watersheds of Bourne. Bourne consists of three watersheds with a combined area of approximately 950 acres. About 50 percent of this area is urban (484 acres); the remaining 50 percent is forest (34 percent, 317 acres), open wetland (15 percent, 139 acres), and agricultural land (1 percent, 11 acres). The majority of the developed land is residential, with 0.5 to 2 houses per acre (often summer cottages turned into year-round homes). Analyses of stormwater collected during or after storm events revealed high contents of fecal coliforms in almost all samples. The highest amounts of fecal coliforms were found in stormwater samples collected in August; subsequent samples taken in October and November contained generally less fecal coliforms (Tables 3-5).

Fecal coliform/fecal streptococci ratios were generally low, indicating that only in very few cases, human wastes, i.e., failing septic systems, were the source of bacterial contamination.

Table 3. Fecal coliform bacteria--Barlows Landing stormwater

	Fecal Coliform Bacteria (Colonies/100 ml)		
Station	August 1988	October 1988	November 1988
BL-5	3,900	-	<100
BL-5-O	-	-	100
BL-5-CB	300	500	100
BL-6	12,000	4,000	<100
BL-6-CB	34,000	6,300	300
BL-6-SWALE	-	-	100
BL-9	19,000	1,000	300
BL-9-CB	28,000	2,100	100
BL-12	1,300	-	100
BL-21	1,000	-	<100
BL-21-CB	-	4,800	9,100
BL-24	100	-	<100
BL-24-CB	5,200	-	<100
BL-29	27,000	3,200	400
BL-32	500	-	-
BL-32-CB	12,000	-	-
BL-35-S	-	12,000	500
BL-36-S	-	13,000	<100
BL-42	4,300	-	200
BL-43	2,400	-	-
BL-46	100	-	-
BL-53	-	-	6,400
BL-54	4,700	-	800
BL-54-O	31,000	-	1,100
BL-54-CB	-	-	5,200
BL-54-S	-	140,000	5,500
BL-57-O	-	-	100
BL-63	-	-	300
BL-63-CB	-	-	100

The fecal coliform/fecal streptococci ratios declined from August to November, generally suggesting that nonhuman wastes became more important as pollutant sources with the progressing season. The trend was clearest in Barlows Landing, a watershed with about 63 percent urban land; it was less obvious in both the 90 percent urbanized watershed of Hen Cove and the mostly forested watershed of Pocasset River with only 30 percent urban land. There was no clear relationship between the density of houses per acre and the degree of contamination.

Non-human wastes, such as pet wastes or droppings of wild mammals and birds, may be the major source for bacterial contamination in stormwater, even in the mostly urban watershed of Hen Cove. Heufelder (1988) found the fecal coliform concentration in dog waste to be approximately 10^6 cells/g feces and estimated the average daily amount of feces deposited by one dog at about 450 g. The resulting bacterial loadings in stormwater may therefore be substantial during heavy rainfall. However, the decline in bacterial contamination may in part be the result of seasonal effects described above, particularly the short

Table 4. Fecal coliform bacteria--Hen Cove stormwater station.

	Fecal Coliform Bacteria (Colonies/100 ml)		
Station	August 1988	October 1988	November 1988
HC-1	4,700	-	100
HC-1-O	3,500	700	-
HC-3	-	-	2,700
HC-3-O	-	100	-
HC-3-CB	-	-	1,300
HC-4	1,100	-	<100
HC-4-O	12,000	-	<100
HC-5	-	-	200
HC-6	6,700	1,900	<100
HC-6-O	1,200	300	-
HC-6-CB	800	1,200	-
HC-7	700	1,600	1,200
HC-22	-	-	100
HC-24	19,000	-	<100
HC-24-O	-	-	800
HC-24-CB	54,000	-	-
HC-25	200	-	<100
HC-26	<100	-	500
HC-45	<100	3,300	1,500
HC-47	600	-	-
HC-49	-	400	<100
HC-54	<100	700	200
HC-57	1,800	-	-
HC-57-CB	1,200	-	-
HC-62	1,700	-	100
HC-62-O	3,600	1,100	200
HC-62-CB	4,100	-	-
HC-62-SWALE	-	-	200
HC-64	21,000	-	<100
HC-65	50,000	-	2,800

survival time of fecal coliforms in lower temperatures. National Urban Runoff Program (NURP) studies have shown that the fecal coliform loadings of stormwater may differ by a factor of 20 between summer and winter even if the land use is constant during all seasons (Heufelder, 1988).

Moreover, the weather pattern around the days of sampling differed slightly and may have influenced the results. The dry periods preceding the storm events were five days in August, three days in October and five days in November. High numbers of bacteria in the stormwater runoff sampled in August may thus in part result from a long accumulation period, and the decline in October may be due in part to a shorter dry period.

The time of sampling during the rain event was also slightly different among the field efforts. The samples in August were taken from the first flush, while sampling in October started a few hours after it had begun to rain, and the November samples were taken on the second day of a two-day storm event. It is therefore likely that the general decrease of bacterial loading from August to November is not quite as pronounced in the soil as suggested by the data, but rather reflects the different times of sampling.

Table 5. Fecal coliform bacteria--Pocasset River stormwater station.

Station	Fecal Coliform Bacteria (Colonies/100 ml)		
	August 1988	October 1988	November 1988
PR-1	5,900	-	-
PR-3	-	-	<100
PR-4	1,800	-	-
PR-4-O	-	-	<100
PR-4-CB	36,000	-	-
PR-11	-	300	100
PR-13-CB	-	1,600	-
PR-14	3,300	8,100	600
PR-14-O	2,900	4,100	100
PR-14-CB	5,600	-	-
PR-16	1,100	-	100
PR-16-O	-	100	<100
PR-19	300	-	3,400
PR-19-CB	<100	100	<100
PR-20	200	-	<100
PR-22	15,000	1,500	<100
PR-22-O	28,000	1,200	-
PR-22-CB	-	-	200
PR-26	100	-	-
PR-26-O	100	-	300
PR-30	1,800	-	200
PR-32	300	-	-
PR-33	9,700	3,400	1,400
PR-33-S	-	100	300
PR-35	100	-	-
PR-35-S	-	-	300
PR-35-CB	-	200	-
PR-36	2,100	-	-
PR-37-O	9,200	-	-
PR-39-O	6,800	-	-
PR-40	100	-	-
PR-41	300	2,800	200
PR-41-O	41,000	900	400
PR-43	100	-	-
PR-50	100	600	d
PR-57	100	-	-
PR-60	<100	1,400	400
PR-70	100	-	-
PR-78	1,000	-	100
PR-78-S	100	-	100
PR-79	1,900	-	-
PR-79-O	-	-	<100
PR-79-S	100	-	-
GC-S	-	-	100
SP-S	-	100	<100
MP-S	-	6,200	100
HF-S	-	-	1,000,000

Overall, it proved to be difficult to define the sources of contamination in stormwater runoff, although assumptions could be made that human wastes were not an important factor. Most of the loading could be accounted for by estimating the number of bacteria contained in dog wastes; high numbers of fecal strep in the fall may also be a result of migrating water fowl, such as Canada geese, concentrating in the area for a short time to feed in the open wetlands. However, the failure of septic systems may remain undetected because of a high die-off rate of fecal coliforms in colder weather in the late fall and winter.

Case Study: Buttermilk Bay

Seven surface drains were sampled during eight rain events between May and December 1986. Two of these drains were in densely populated residential areas (about 8 units per acre), one drain was in an industrial area, and the remaining four were in moderately populated residential areas (less than 8 units/acre). Fecal coliform loadings were highest in the densely populated areas, intermediate in less densely populated residential areas, and lowest in the commercial area (Figure 2).

This relation between bacterial contamination and population density suggests that sources in Buttermilk Bay may be human, although the bacterial loadings could be fully accounted for if one would assume the resident dog population in Buttermilk Bay as the source (Heufelder, 1988).

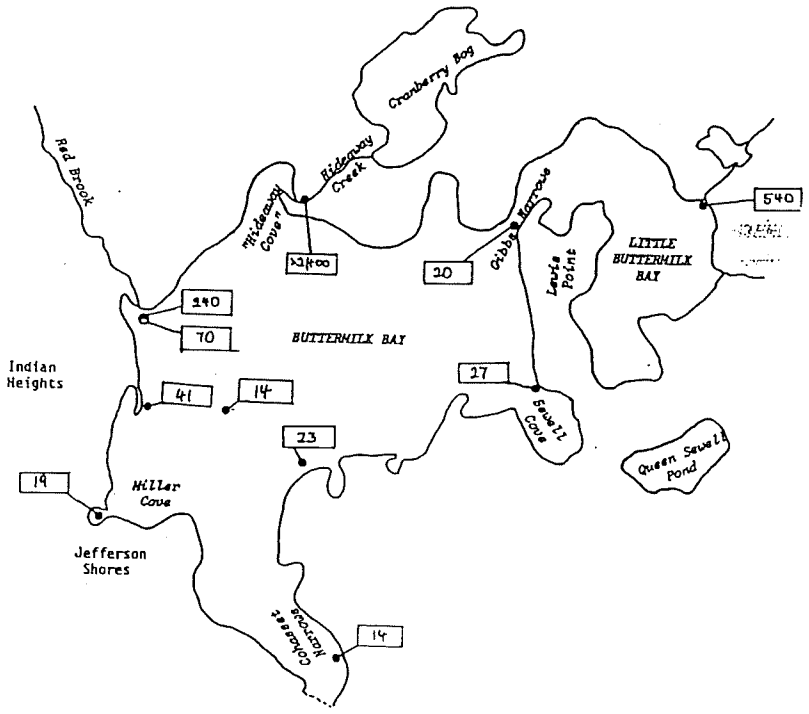

Figure 2. Highest fecal coliform concentrations (colonies/100 ml) found at routine monitoring stations in Buttermilk Bay during 14 sampling events between September 1985 and October 1986. Red Brook station stampled at two depths (from Heufelder, 1988).

The amounts of stormwater entering the bay after a 1-in rainfall is estimated at 2050 m^3, containing about 10^4 fecal coliforms/100 ml. To dilute the runoff to concentrations acceptable for shellfishing, between 18 and 440 percent of the tidal prism of Buttermilk Bay are necessary.

Marinas

Pathogens from marinas and boats may originate from direct discharge of human waste or from sediments resuspended by boat propellers (Heufelder, 1988). Because discharge of untreated human waste is not allowed in Massachusetts coastal waters, contamination of the seawater from this source probably occurs only occasionally, although the effects may be considerable at times. Because of the irregular occurrence, bacterial loadings originating from boats and marinas are very difficult to determine. There are some rough estimates (Grimes and Heufelder, BBP, in press) based on the number of residential boats, derived from the number of slips and moorings; the percentage of boats longer than 26 ft, i.e., boats usually equipped with marine sanitation devices; the estimated discharge of fecal coliforms per person and day; and the assumption that all boats are occupied by two persons. The fecal coliform loading from boats in Buzzards Bay can be estimated at 4.3×10^{12} per day. To dilute the boat wastes to an acceptable fecal coliform concentration of 14 fecal coliforms/100 ml water, 10^6 ft^3 are required per boat. That volume of water translates into an area of about 2.3 acres in 3-ft deep water.

According to Heufelder (1988), resuspension of coliform-laden sediments by boat propellers is unlikely in most marinas. Marinas are located in sufficiently deep water to prevent such resuspension. Similar to discharges from boats, this pathogen source is almost impossible to assess.

Case Study: Wareham

One marina with a pump-out facility and two marinas without such facilities were sampled on two days in August 1986. The results were inconclusive, mostly due to the relatively cool and rainy weather of that summer. It was therefore difficult to sample the worst-case scenario which would occur just after a hot, sunny weekend with most of the boats being out. In all samples taken at the marinas, bacterial loadings were generally low and roughly the same, except for one sample that was taken on 3 August near the marina with a pump-out facility. This contamination was most likely caused by a different source.

Freshwater Streams

High loads of fecal coliforms are introduced into rivers and other bodies of fresh water along the western and northwestern part of the bay, mainly around Westport, Fairhaven, Mattapoisett, and especially the Acushnet River. These areas represent the most densely populated part of the Bay. The rivers and brooks at the eastern and southeastern parts of the Bay have a generally low load of fecal coliforms (Technical Services Branch, in preparation). Among the sources for pathogenic contamination of freshwater streams are failing septic systems and stormwater runoff. The latter depends on the soil and geological profile of the area. However, elevated bacterial loadings are not necessarily an

indication of manmade contamination. Freshwater wetlands are known to show highly variable natural background levels because the environment can at times offer optimal conditions for survival of enteric bacteria outside their hosts (Kadlec and Tilton, 1979).

Case Studies: Wareham and Westport

In the Wareham watershed, three rivers were sampled: the Agawam River, Wareham River, and East River. Both the Agawam and East River were practically free of fecal coliforms and streptococci; all samples but one contained less than 10 colonies/100 ml of either group. The loadings found in the Wareham River increased for both fecal coliforms and fecal streptococci as the season progressed; for example, on 7 July all samples contained less than 10 fecal coliforms and fecal streptococci/100 ml, whereas on 21 July, over 60 percent of all samples contained up to 500 fecal coliforms/100 ml, and half of the samples contained between 100 and 200 fecal streptococci/100 ml. Although these levels are slightly elevated, the contamination is not severe.

In comparison, the Westport River was more contaminated. Out of 10 to 12 stations sampled, at least 40 percent were highly contaminated, with fecal coliforms ranging from 900 to 2900 colonies/100 ml in September and from 550 to 25,000 colonies/100 ml in October. Fecal streptococci counts were lower except for one station in September, but the loadings showed the same tendency as the fecal coliforms.

Marsh Sediments and Beach Wrack

Marshes and decaying matter in the intertidal zone along the beaches may function as a nonpoint source for pathogens at times, because these environments may provide sheltered habitats for enteric bacteria suitable for prolonged survival and even multiplication. The primary source for fecal coliforms is probably wildlife, including birds, rodents, and rabbits, but the decaying plant matter may also be a source of bacteria.

Case Study: Wareham

Salt marshes and bog drainages were sampled four times between June and September 1986. At all sampling events, 90 to 100 percent of the stations had very low bacterial loadings (less than 10 colonies/100 ml for fecal coliforms; less than 100 colonies/100 ml for fecal streptococci), with the exception of the samples taken on 11 August after approximately 2 in of rainfall during 3 preceding days. The number of fecal streptococci was generally about ten times higher than the number of fecal coliforms, so the bacteria were probably of nonhuman origin.

Case Study: Buttermilk Bay

Six locations along the beaches of Buttermilk Bay were sampled for beach wrack, consisting mostly of dead and decaying eelgrass (*Zostera marina*). The fecal coliform loadings of the wrack itself were

highly variable, ranging from numbers below the detection limit to more than 24,000 colonies/g beach wrack during the summer. The rainwater retained in the wrack was examined during a storm event in November 1986; results suggest that rain may cause local contamination of the seawater during the next incoming tide. Loadings were between 2300 and more than 16,000 fecal coliforms/100 ml. Suspected sources for fecal coliforms were dog wastes and bird droppings, especially from Canada geese.

Wildlife, Waterfowl, and Domestic Animals

Case Study: Buttermilk Bay

The significance of waterfowl as a source for pathogens was investigated with focus on two different routes: direct deposition into the water and deposition on land followed by transport through stormwater. Although amounts of fecal coliforms from directly deposited feces can be high at times (estimated daily input up to 3.1×10^{11} organisms in the winter), the impact on overall water quality is minimal when uniform mixing of the water body is assumed, i.e., when open-water localities are considered.

Nearshore waters are mostly contaminated by droppings deposited on land during low tide and washed into the sea either by the next incoming tide or rain. However, the study showed no clear evidence that high numbers of waterfowl observed on the beach correlated with elevated loadings of fecal coliforms in the water. In fact, the highest loadings were found during the summer months when usage of the beaches by waterfowl was lowest. As in stormwater runoff, many factors connected with the biology of the enteric bacteria may have influenced the pathogen contamination in the study area in a complex way, so that interpretation of these data is difficult.

Agricultural Runoff

Case Study: Westport

An agricultural area in the Westport River watershed characterized by dairy and beef cattle farms was investigated in July 1986. Most of the land consisted of cow pastures, but a small area was used for corn fields. All of the ten stations sampled showed highly elevated bacterial loadings; the numbers of fecal coliforms ranged from 1800 to 80,000 colonies/100 ml, and the number of fecal streptococci was between 2000 and 6400 colonies/100 ml. The samples were taken from several creeks, brooks and coves after about 1.7 inches of rainfall during three preceding days.

CSOs

The annual discharge volume from CSOs in New Bedford is over 1.5 billion gallons, loaded with a total coliform concentration of about 4×10^6 colonies/100 ml (CDM, 1983). Dry weather flows have been estimated as 4.7 MGD, resulting in a daily load of approximately 4×10^{14} fecal coliforms to the Bay.

Nutrient Enrichment

Nutrients, particularly nitrogen, enter Buzzards Bay from a variety of sources. Nitrogen stimulates algal productivity in coastal waters such as Buzzards Bay and in some instances may lead to eutrophication. Excessive inputs of nitrogen may be especially damaging to small, sensitive embayments.

Although nitrogen is the nutrient of most concern in marine waters, increased levels of phosphorus may be of concern in the freshwater lakes, ponds, and streams of the Buzzards Bay watershed. In freshwater systems, phosphorus rather than nitrogen stimulates algal growth and leads to eutrophic conditions.

Sources of nitrogen to the bay include direct precipitation onto the Bay; groundwater flow from septic systems; precipitation onto undeveloped land; lawn fertilizer; fertilizers applied to cranberry bogs and other agricultural crops; export from salt marshes; runoff from developed land; and direct inputs from sewage treatment plants located in New Bedford, Wareham, Dartmouth, Fairhaven, Falmouth, and Marion (Table 6).

Table 6. Nitrogen Inputs To Buzzards Bay.

Source	Load (metric tons/yr)	Percent of total
Direct Precipitation	217	11
Septic Systems	276	14
Precipitation on Undeveloped Land	29.8	2
Lawn Fertilizer	69.8	4
Cranberry Bogs	57.8	3
Upland Crops	37.5	2
Runoff from Developed Land	36.8	2
Sewage Treatment Plants	1210	62
TOTAL	1934.70	100

Direct Precipitation onto Buzzards Bay

Concentrations of dissolved inorganic nitrogen (DIN) in precipitation are well studied in Massachusetts. Data are available from Valiela and Teal (1979) and from National Acid Deposition Program (NADP) stations in South Truro, Quabbin Reservoir, and Middlesex. Kelly and Valiela (BBP, in press) used 1987 data from the NADP stations at Middlesex and Quabbin Reservoir to calculate DIN concentrations of 27 M in precipitation. The annual precipitation at these stations averaged 105 cm/year. The annual atmospheric deposition was 25.7 mmol DIN/m^2.

Annual DIN deposition is constant even when average DIN concentrations in the precipitation vary. For example, during an unusually wet year, 155 cm of rain fell on the NADP station at South Truro. The average DIN concentration of the rain was only 15.5 M. However, DIN deposition for the year was 24.9 mmol/m^2, nearly identical to the deposition calculated by Kelly and Valiela.

Values calculated from the NADP data are also nearly identical to those calculated using information from Great Sippewissett Marsh, a salt marsh bordering Buzzards Bay in Falmouth (Valiela and Teal, 1979). They calculated that 26.6 mmol DIN/m^2 was deposited on the marsh each year. These values are also consistent with regional estimates of deposition of nitrates and ammonia that have been reported by the National Academy of Sciences. Wet and bulk precipitation values underestimate dry deposition, but dry deposition of DIN to water surfaces is low.

Unfortunately, few data are available from which to calculate deposition of dissolved organic nitrogen (DON). Data from Valiela and Teal, suggest that deposition of DON may equal that of DIN.

Using the concentration of DIN from Valiela and Teal and information from the BBP on the area of the Bay (600 km^2), total input of DIN to Buzzards Bay from precipitation is approximately 210 metric tons/year:

$$26.6\ mmol/m^2 \times 600\ KM^2 \times 14\ g/m^2 = 220\ metric\ tons\ N/year$$

This value could be doubled if deposition of DON equals that of DIN.

Septic Systems

Approximately 120,000 people live within areas of the Buzzards Bay watershed that are served by individual, onsite septic systems. Approximately 2.3 kg N/person/year is released from such systems to the groundwater according to studies conducted on Cape Cod and Long Island (CCPEDC, 1979). (Input to the septic system is about twice the amount released from the system.) Assuming that all the nitrogen released from septic systems eventually enters Buzzards Bay, approximately 276 metric tons of N/year enters the Bay from this source:

$$20{,}000\ person \times 2.3\ kg\ N/person/year = 276\ metric\ tons\ N/year$$

Precipitation Onto Undeveloped Land

Undeveloped land (areas of pasture, forest, non-forest wetland, open space, urban open space, and water) comprises approximately 800 km^2 of the Buzzards Bay area (using data developed by the Massachusetts GIS Project, administered through the Massachusetts Executive Office of Environmental Affairs, Table 7). If 26.6 mmolN/m^2 are deposited onto the land surface (Teal and Valiela, 1979) and 90 percent of the nitrogen that reaches the surface is taken up by plants rather than being transported to the Bay, then approximately 30 metric tons N/year enter the Bay from precipitation onto undeveloped land surfaces:

$$26.6\ mmol/m^2 \times 800\ km^2 \times 14\ g/mmol \times .1 = 29.8\ metric\ tons\ N/year$$

The estimate of 90 percent loss may be low"groundwater concentrations in undeveloped levels characteristically have low concentrations of nitrates, and work by Valiela and Costa (1988) in Buttermilk Bay suggested that almost all the nitrogen in precipitation deposited onto land is retained.

Lawn Fertilizers

Application rates of lawn fertilizers vary widely, and the amounts of DIN that leach into groundwater following application is not well known. The Long Island 208 Plan (1978) estimated that approximately 3 lb N/year was applied

Table 7. Land use in the Buzzards Bay watershed.

Land Use	Acres	Sq Mi	Sq Km	Percent
Cropland	9,256	14.46	37.45	3.5
Pasture	6,161	9.63	24.94	2.4
Forest	161,153	251.80	652.16	61.5
Nonforest Wetland	4,766	7.45	19.30	1.8
Mining	1,585	2.48	6.42	0.6
Open Land	12,675	19.80	51.28	4.8
Participatory Recreation	778	1.22	3.16	0.3
Spectator Recreation	520	0.81	2.10	0.2
Waterbased Recreation	2,045	3.19	8.26	0.8
Multifamily Residential	834	1.30	3.37	0.3
< 1/4 Acre Lot Residential	6,850	10.70	27.71	2.6
1/4-1/2 Acre Lot Residential	14,045	21.95	56.85	5.4
>1/2 Acre Lot Residential	12,572	19.64	50.87	4.8
Salt Marsh	4,907	7.67	19.87	1.9
Commercial	2,415	3.77	9.76	0.9
Industrial	1,380	2.16	5.59	0.5
Urban Open Land	4,568	7.14	18.49	1.7
Transportation	3,515	5.49	14.22	1.3
Waste Disposal	822	1.28	3.32	0.3
Water	6,980	10.91	28.26	—
Woody Perennial	10,993	17.18	44.50	4.2
TOTAL ACRES	272,909			
TOTAL SQ MI		426.18		
TOTAL SQ KM			1,103.81	

to 1,000 ft^2 of lawn area and that typical lawns were 5,000 ft^2. The 208 Plan assumed that 60 percent of the amount of DIN applied to lawns is leached to the groundwater. Therefore, approximately 9 lb DIN/lawn enters the groundwater each year. Other data on fertilizer application and leach rates vary. Giblin and Gaines (1990) used a survey of hardware stores in the Town of Orleans to estimate that fertilizer use was only 5 lb/lawn/year.

Horsely and Witten (BBP, in preparation) used 15 lb/lawn/year as an application rate and assumed that 30 percent leached, so that 5 lb/lawn/year reached the groundwater. Their review of the literature indicated that leaching rates were closer to 30 than 60 percent. A CCPEDC study calculates the average lawn size in Falmouth as 3,000 ft^2, much smaller that the size estimated for Long Island.

The BBP has information has information on the total land area composed of residential lots (Table 7). The data are compiled by lot size (<1/4, 1/4-1/2, and >1/2 acre). If 4 lots/acre are present in the <1/4 acre category, 3 lots/acre are in the 1/4-1/2 acre category, and 1 lot/acre is in the >1/2 acre category, then

there are approximately 82,000 single-family dwellings within the watershed (Table 8). This estimate agrees with other estimates made by the BBP which calculated that if there were 250,000 people in the watershed and approximately 3 persons/household, then there would be approximately 83,000 lots.

Other data must be assumed. For example, lawn size is probably proportional to lot size. Approximately 280, 460, and 1,400 m^2 (3,000, 5,000, and 15,000 ft^2) of lawn for <1/4, 1/4-1/2, and >1/2 acre lots, respectively, may be estimated for the region. Using these values, there are approximately 45 km^2 of lawn in the watershed (Table 8).

Table 8. Lawn area in the Buzzards Bay watershed.

Land Use	Area (acres)	Units per acre	Total Units	Lawn Size per unit (m^2)
<1/4 Acre Lots	6,850	4	27,400	280
1/4-1/2 Acre Lots	14,045	3	42,135	460
>1/2 Acre Lots	12,545	1	12,545	1,400
TOTAL	33,440		82,080	

Approximately 5 g/m^2 DIN/m^2/year (1 lb DIN/1,000 ft^2/year) is applied to the lawns. If 30 percent leaches to the groundwater, then approximately 70 metric tons of nitrogen reach the Bay from lawn fertilizers each year:

44.61 km^2 x 5 g/m^2/year x 30 percent = 67.5 metric tons/year

Cranberry Bogs

Teal and Howes (BBP, in preparation) estimated that 13 kg N/ha/year enter the Bay from cranberry bogs. There are 44.55 km^2 of cranberry bogs in the region. Therefore, approximately 58 metric tons N/year enter the Bay from cranberry bogs:

1300 kg N/km^2/year x 44.6 km^2 = 58 metric tons N/year

Other Agriculture

Application rates of fertilizers to crops other than cranberries range from less than 10 to more than 100 kg N/ha. Leaching rates generally increase with increased application rates. Assuming that most of the agricultural land in the Buzzards Bay watershed is planted with strawberries, vegetables, and other crops that require low application rates of fertilizers, 10 kg/ha is probably a realistic estimate of application. Using data on total area of cropland (Table 6), there are approximately 37.6 km^2 of cropland in the region. Therefore, approximately 37.6 metric tons N/year enters the Bay from this source:

1000 kg N/km^2/year x 37.6 km^2 = 37.6 metric tons N/year

Salt Marshes

Salt marshes are known to intercept nitrogen from groundwater and export it to coastal waters. Valiela and Teal (1979) estimated that Great Sippewissett Marsh in Falmouth received 6,500 kg N/year from groundwater and exported

5,350 kg N/year to Buzzards Bay. Because these values are approximately equal, salt marshes have not been included in Table 6 as a source of nitrogen to the bay.

Runoff from Developed Surfaces

Nitrogen in runoff from developed surfaces originates from precipitation, animal wastes, and other sources. Runoff may enter the Bay directly or leach into the groundwater and be transported to the Bay. Koppelman (1982) estimated that nitrogen concentrations in road runoff were 1.5 mg/L (107 μM), four times the concentrations of DIN in precipitation. Using data on the areas of paved surfaces (mining, recreation, commercial, industrial, transportation, and waste disposal) (Table 7), there are approximately 44.6 km^2 of paved surface in the area. Assuming that 55 cm of rain falls each year, an estimated 36.8 metric tons N/year enter the Bay.

$$1.5\ mg/L \cdot 44.6\ km^2 \times 55\ cm = 36.8\ metric\ tons\ N/year$$

Sewage Treatment Plants

Sewage treatment plants in New Bedford, Wareham, Dartmouth, Fairhaven, Falmouth, and Marian release nitrogen into Buzzards Bay. Extrapolating from data on flows from these plants (Smith, 1988) and the average dry weather discharge value from the New Bedford facility, an estimated 1220 metric tons

Table 9. Nitrogen inputs from sewage treatment plants.

Sewage Treatment Plant	Load (metric tons/year)
New Bedford	960
Wareham	28.3
Dartmouth	56.6
Fairhaven	139
Falmouth	14.7
Marion	7.03
TOTAL	1210

N/year enter the Bay in sewage effluent (Table 9). (The input from Falmouth was adjusted, because approximately one half of the effluent from the Falmouth facility is disposed of in rapid infiltration beds.)

Toxic Contamination

Toxic chemicals, including petroleum products, pesticides, polychlorinated biphenyls (PCBs), other organic compounds, and metals, enter Buzzards Bay through a variety of point and nonpoint sources. These sources include atmospheric deposition, industrial and municipal outfalls, and runoff from the land into rivers or the Bay. Chemical contaminants resulting from industrial activities enter Buzzards Bay primarily in the western portion near the New Bedford, Fairhaven, and Dartmouth urban areas. Chemicals associated with agricultural activities are more likely to enter the Bay from agricultural and road

runoff, creeks, and small rivers in Westport, Dartmouth, Fairhaven, Mattapoisett, Marion, Wareham, Bourne, and Falmouth.

Petroleum and Fossil Fuel Hydrocarbons

Hydrocarbon inputs to Buzzards Bay result from accidental spills, industrial and municipal wastes, stormwater runoff, small boats and other marine craft, and creosote-treated wood pilings (Table 10). No oil exploration or production takes place in Buzzards Bay, and there are no refineries within the Buzzards Bay watershed. However, small tankers and barges move through the bay, traveling within sealanes and through the Cape Cod Canal, transporting heating and industrial oil and gasoline to the greater Boston market and to New Bedford.

Table 10. Petroleum inputs to Buzzards Bay

Source	Load (metric tons/yr)	Percent of Total
Accidental Spills	60	20-40
Discharges and Runoff	90-250	60-80
Small Boats	Not calculated	
Creosoted Pilings	Minimal	
TOTAL	150-310	100

Accidental spills from these tankers and barges have been a major but highly variable source of petroleum compounds to the bay.

Farrington and Capuzzo (1990) estimated that almost 1,000 metric tons of petroleum and petroleum products were spilled into the bay during 1973-August 1989, approximately 60 metric tons/year averaging over the entire period. However, approximately 650 metric tons of No. 2 fuel oil were spilled in one incident in 1974, when the barge *Bouchard 65* struck a submerged object. Approximately 300 metric tons were spilled when the same barge ran aground in 1978. Numerous small spills accounted for less than 5 metric tons during 1973-August 1989 (approximately 0.25 metric tons/year). Consequently, effects of oil spills in Buzzards Bay are expected to be locally devastating but not as serious on a bay-wide scale. Mitigation and monitoring of the effects of future oil spills will rely on rapid and effective responses to emergencies.

Farrington and Capuzzo (1990) suggested that inputs of petroleum products from sewage effluent, industrial discharges, and stormwater runoff equal inputs from accidental spills. Extrapolating from data for Providence, Rhode Island and Los Angeles, California, Farrington and Capuzzo calculated inputs from sewage effluent, industrial runoff, and stormwater runoff to be 90-250 metric tons/year.

Hoffman (1985) examined hydrocarbons in runoff as a function of land use in the Narragansett Bay watershed. She measured runoff from residential (single family suburb), commercial (shopping mall), industrial (heavy industry), and highway areas during 21 storm events. Using land-use statistics from the BBP and Hoffman's figures for loading, approximately 220 metric tons of hydrocarbons enter the Bay from runoff from residential, industrial, commercial, and highway areas each year. This figure is within the range calculated by Far-

Table 11. Land use within one half mile of shore.

Land Use	Acres	Sq Km	Percent
Cropland	2,500	10.1	4.6
Pasture	1,151	4.7	2.1
Forest	23,618	95.6	44.0
Nonforest Wetland	588	2.9	1.1
Mining	314	1.3	0.6
Open Land	2,980	12.1	5.5
Participatory Recreation	183	0.7	0.3
Spectator Recreation	176	0.7	0.3
Waterbased Recreation	1,357	5.5	2.5
Multifamily Residential	45	0.2	0.1
<1/4 acre lot residential	3,804	15.4	7.0
1/4-1/2 acre lot residential	5,038	20.4	9.3
> 1/2 acre lot residential	4,139	16.8	7.7
Salt Marsh	4,514	18.3	8.3
Commercial	1,094	4.4	2.0
Industrial	594	2.4	1.1
Urban Open Land	1,000	4.1	1.8
Transportation	432	1.8	0.8
Waste Disposal	49	0.2	0.8
Woody Perennial	504	2.0	0.1
TOTAL ACRES	54,080		
TOTAL SQ KM		219	

rington and Capuzzo for all discharges and runoff. It assumes, however, that runoff from the entire drainage basin reaches the Bay. A similar calculation using data on land use within one half mile from the shore (Table 11) suggests that only 60 metric would enter Buzzards Bay from runoff from commercial, industrial, and road surfaces (Table 12).

(The estimate used the BBP figures for "transportation" as an equivalent to Hoffman's "highways." The BBP figure includes airports, docks, divided highways, freight storage, and railroads. Hoffman's highways included only major, eight-lane roads.) Although Farrington and Capuzzo did not calculate inputs from small boats with inboard diesels and outboard engines, they noted that there are approximately 4,300 slips and moorings in the Buzzards Bay area and that approximately 20,000 vessels move through the Cape Cod Canal each year. Two-stroke outboard motors, the most popular motors used on recreational boats, have been shown to discharge raw gasoline and oil, nonvolatile and volatile oil compounds, and phenol into the water (Kuzminski and Jackivicz, 1972). However, discharges of these compounds is highly variable.

Waste oil from commercial fishing vessels may be a greater problem than oil from recreational boats. The New Bedford fishing fleet changes 380-450 liters

Table 12. Runoff of petroleum hydrocarbons to Buzzards Bay.

Land Use Type[1]	Area (km^2)	Loading Factor[2] ($kg/km^2/yr$)	Load (kg/yr)
Residential	53	180	9500
Commercial	4.4	580	2600
Industrial	2.4	14000	34000
Road	1.8	7800	14000
TOTAL			60000

[1] From BBP (Table 11)
[2] From Hoffman, 1985

(100-120 gallons) of oil after every trip and is suspected of occasionally dumping it into the Bay. There are 200 vessels in the fleet, and each makes 1-4 trips each month. If the vessels make an average of 2 trips/month, more than 1 million gallons of oil are used each year. Some of this oil is taken in by the New Bedford Seafood Coop and by private boat yards, but much of it is not accounted for. The fishing fleets in Dartmouth, Westport, Mattapoisett, and Wareham may also dump waste oil into the Bay.

Farrington and Capuzzo also noted that creosote-treated pilings could be a source of hydrocarbons to Buzzards Bay. However, creosote is no longer used for marine structures within the Bay. Leaching from old structures is probably minimal (B. Tripp, Woods Hole Oceanographic Institution, personal communication, 1990).

Pesticides

Chlorinated pesticides were widely used during the 1950s and 1960s to kill mosquitos in the Buzzards Bay watershed. Organochlorine pesticides such as DDT and dieldrin were among the early pesticides used. Less persistent organophosphate and carbamate compounds have now replaced those.

Spraying for mosquitos may have contributed to the pesticide load within the bay in the past. Currently, only *Bacillus thuringiensis* is sprayed over wetlands routinely (Table 13). During 1990, the State ordered emergency spraying of malathion to kill mosquitos in a portion of the Buzzards Bay watershed. Although malathion is short-lived and the spray program was designed to avoid introduction of the chemical to the Bay, improper application may have been responsible for fish kills that were reported at the time.

Farrington and Capuzzo (1990) estimated that approximately 10,000 kg (20,000 lb) of pesticides are applied to cranberry bogs within the Buzzards Bay watershed each year. Approximately 6,000 kg (13,000 lb) of pesticides are applied to other crops. Pesticides commonly used on cranberry bogs include parathion, lorsban, diazinon, and carbaryl, all organophosphate insecticides with half lives of a week or less. Because those pesticides are relatively nonpersistent, Farrington and Capuzzo suggested that transport to the bay is limited.

Table 13. Pesticide inputs to Buzzards Bay

Source	Load (kg/yr)	Percent of Total
New Bedford Sewage Outfall	2-3	100
New Bedford CSOs	Not detected	0
Cranberry Bogs	-[1]	-
Other Agriculture	-[1]	-
Lawns	-[1]	-
Mosquito Spraying	-[2]	-
TOTAL	2-3	100

[1] Organophosphates with short half lives

[2] Except in emergencies, only *Bacillus thuringus* is used

However, pesticides could affect localized embayments if improper application or an accidental spill occurred. Fish kills downstream from cranberry bogs during the 1970s and early 1980s were suspected of resulting from improper application of pesticides (J. Fiske, personal communication, 1990). During this period, a task force made up of personnel from the Massachusetts Division of Marine Fisheries, the University of Massachusetts Cranberry Experiment Station, Ocean Spray Cranberries, Inc., and Chemapeo Corp. was formed to educate growers about improved spraying practices.

Some pesticides may continue to be introduced to Buzzards Bay in sewage effluent or stormwater runoff. Data for the draft New Bedford Harbor Facilities Plan indicate that 2-3 kg gamma-BHC, DDT, and DDD may enter the harbor in dry-weather discharges each year (CDM, 1989a; EPA, 1989). No pesticides were detected in the studies conducted for the New Bedford CSO Facilities Plan (CDM, 1989b).

PCBs

Manufacture of electrical components by several New Bedford firms caused major inputs of PCBs into New Bedford Harbor and Buzzards Bay. Between the 1940s and 1970s, manufacturing operations discharged PCBs directly into the Acushnet River and indirectly into the municipal sewage treatment plant (Table 14). Although PCBs are no longer used in manufacturing, remobilization from New Bedford Harbor sediments allows for continuing bioavailability of PCBs.

Assuming a 1-percent rate of loss, Farrington and Capuzzo (1990) estimated that approximately 145 metric tons of PCBs were lost by Cornell-Dublier, Aerovox, and Monsanto between 1958 and 1977. The exact pathways and amounts of PCBs that entered the harbor during this period are unknown, although one estimate quoted by Farrington and Capuzzo was that 100 metric tons were introduced into bay prior to the 1970s. Extensive studies of PCBs in New Bedford Harbor and Buzzards Bay have been conducted and continue as part of EPA's Superfund program.

Estimates of remobilization of PCBs from New Bedford Harbor sediments are still being made. A reasonable estimate of release of PCBs from north of the

Table 14. PCB inputs to Buzzards Bay.

Source	Load (kg/yr)	Percent of Total
Atmospheric Deposition	48-340	3-17
Remobilization from Upstream Sediments[1]	1600	81-96
New Bedford Sewage Outfall	25	1
TOTAL	1673-1965	

[1]Most of these PCBs are expected to remain within the New Bedford Harbor hurricane barrier.

Coggeshall Street Bridge in the upper estuary is 4.4 kg PCB/day or 1,600 kg/year. Most of these PCBs would be expected to remain inside the hurricane barriers that stand at the mouth of New Bedford Harbor.

Inputs of PCBs to Buzzards Bay from sources other than New Bedford Harbor sediments are probably minor. EPA and the City of New Bedford have estimated that just 10 kg/year Aroclor 1242 and 13 kg/year Aroclor 1254 enter Buzzards Bay through dry water effluent discharge of the New Bedford sewage treatment plant (CDM, 1989a; EPA, 1989).

The decline in PCB use has made estimating inputs from atmospheric deposition difficult. Mayer (1982) cited a mean rate of fallout of 80 g/m^2/year, which would result in fallout of 48 kg PCB/year. Menzie et al. (Massachusetts Bay Project, in preparation) estimated deposition as 0.12-0.34 mg/m^2/year, which would result in loads as high as 340 kg/year. Because PCB release into the environment has steadily declined, these estimates are high. Farrington and Capuzzo (1990) noted that boat paints and dredged material disposal may also be sources of PCBs to the bay, but that these sources are probably minor.

Other Organic Pollutants

Organic pollutants other than petroleum products, pesticides, and PCBs may also enter Buzzards Bay, primarily through municipal discharges from urban areas. Compounds which may be expected are volatile organic compounds associated with solvent degreasers and cleaning fluids. Because the New Bedford area is the only significantly urbanized area along Buzzards Bay, most of these organic compounds may be expected to enter the Bay in New Bedford's municipal effluent. Tables 15 and 16 present average annual dry weather discharges, data developed by the City of New Bedford (CDM, 1989a) and EPA (1989).

Recent research has indicated that tributyl tin (TBT), which is sometimes added to marine paint as an antifoulant, is toxic to marine organisms at very low levels. TBT was never extensively used on boats resident within Buzzards Bay and it is now banned for use on vessels shorter than 25 m. Leaching rates on the vessels still permitted to use TBT may not exceed 4 g TBT/cm^2/day. Unfortunately, sufficient data to calculate inputs of TBT to Buzzards Bay are not available.

Table 15. Volatile organic compound loadings to Buzzards Bay (dry weather average for 12 samples)[1]

Constituent	Load (kg/yr)	Standard Deviation
Methylene chloride	200	189
1,2-Dichloroethane	190	108
Chloroform	378	134
1,2-Dichloroethane	190	151
1,1,1-Trichloroethane	398	451
Trichloroethane	473	237
Tetrachloroethane	310	392
1,1,2,2-Tetrachloroethane	211	192
Toluene	1,508	1,555
Ethylbenzene	347	400
Total Xylenes	1,691	2,049
2-Butanone	1,173	3,295
Acetone	5,329	4,587
Benzene	161	151
4-Methyl,2-pentanone	262	315

[1] from CDM, 1989a

Table 16. Acid/base/neutral loadings to Buzzards Bay (dry weather average for 12 samples)[1].

Constituent	Load (kg/y)	Standard Deviation
Phenol	484	549
Benzyl Alcohol	285	128
2-Methylphenol	247	103
4-Methylphenol	1,274	889
Benzoic Acid	1,815	853
4-Chloro,3-Methylphenol	244	105
Isophorone	252	105
1,2,4-Trichlobenzene	256	100
2-Methylnaphthalene	254	118
N-Nitrosodiphenylamine	325	267
Di-N-Butyl Phthalate	265	144
Butyl Benzyl Phthalate	249	108
Bis(2-Ethylhexyl) Phthalate	2,392	6,173
Napthalene	345	100
Diethyl Phthalate	275	90

[1] from CDM, 1989a

Trace Metals

Trace metals, including cadmium, lead, mercury, copper, arsenic, chromium, silver, and nickel enter Buzzards Bay through atmospheric deposition, through industrial activity, from boat paint, in sewage effluent, and in dredged material. There are also natural sources of trace metals, such as the weathering of rocks. The New Bedford Harbor area, especially the Inner Harbor, has received inputs of trace metals in the past and continues to be a major source to Buzzards Bay.

Cadmium

Table 17. Cadmium inputs to Buzzards Bay

Source	Load (kg/yr)	Percent Total
Atmospheric Deposition	0-400	0-82
New Bedford Sewage Outfall	70	14-79
New Bedford CSO	Not Detected	0
Runoff	19	4-21
TOTAL		89-489

No direct measurements of atmospheric deposition of cadmium exist for Buzzards Bay (Table 17). Data from other sources are variable. Extrapolating from data in Menzie et al. (Massachusetts Bay Project, in preparation), atmospheric deposition of cadmium ranges from 0-400 kg/year.

The City of New Bedford (CDM, 1989a) and EPA (1989) estimated that approximately 70 kg Cd/year enter Buzzards Bay in the effluent from the New Bedford sewage treatment plant. Data on flows from other sewage treatment plants in the area (Smith, 1988) and average concentrations of cadmium in the Falmouth plant indicate that inputs from other treatment plants in the watershed are relatively minor.

Most concentrations of cadmium were below the detection limit of 0.005 mg/L for the New Bedford CSO Facilities Plan (CDM, 1989b). Consequently, there are no data on cadmium inputs from stormwater runoff.

Table 18. Runoff of cadmium to Buzzards Bay

Land Use Type [1]	Area (km^2)	Loading factor ($kg/km^2/yr$)	Load (kg/yr)
Residential	53	0.18	9.5
Commercial	4.4	0.69	3.0
Industrial	2.4	0.85	2.0
Road	1.8	2.5	4.5
TOTAL			19.0

[1] From BBP (Table 11)

Hoffman (1985) estimated that for Narragansett Bay, urban runoff accounted for little input of cadmium. Using her loading factors and assuming that only runoff from an area within 0.5 mile of the coast reaches the bay, approximately 21 kg Cd enter the Bay from runoff (Table 18). If all runoff from the watershed reached the Bay, the value would be 71 kg Cd/year.

Table 19. Lead inputs to Buzzards Bay

Source	Load (kg/yr)	Percent Total
Atmospheric Deposition	2340	23
New Bedford Sewage Outfall	1150	11
Other Sewage Treatment Plants	Minimal	0
New Bedford CSOs	1290	13
Runoff	5430	53
Total		100

Lead

Estimates of atmospheric deposition of lead onto surface waters are also variable. Groet (1976) estimated atmospheric deposition rates of 39 mg Pb/m^2/year for southern New England. Using 10 percent of that rate to correct for declines in lead emissions, 2340 kg Pb/year are deposited on the Bay (Table 19).

Approximately 1,150 kg Pb/year enter Buzzards Bay in dry weather discharge from the New Bedford outfall (CDM, 1989a; EPA, 1989). An additional 1,286 kg Pb/year has been estimated to enter the bay from CSOs and storm drains in the New Bedford area (CDM, 1989b).

Table 20. Runoff of lead to Buzzards Bay

Land Use Type[1]	Area (km^2)	Loading factor2 (kg/km^2/yr)	Load (kg/yr)
Residential	53	22.4	1190
Commercial	4.4	43.6	192
Industrial	2.4	1.66	3.98
Road	1.8	2250	4050
TOTAL			5430

[1] From BBP (Table 11)

Hoffman (1985) estimated that most inputs of lead to Narragansett Bay resulted from urban runoff. Using her loading rates and assuming that only runoff from the land area within 0.5 miles of the shore reaches the Bay, approximately 6,000 kg Pb/year enter Buzzards Bay (Table 20). This estimate may be too high, however, because use of leaded gasoline, the major source of lead to the environment in 1985, has declined.

Table 21. Mercury inputs to Buzzards Bay

Source	Load (kg/yr)	Percent Total
Atmospheric Inputs	3.9-12	19-41
New Bedford Sewage Outfall	17	59-81
New Bedford CSOs	Not Detected	0
TOTAL	20.9-29	100

Mercury

Data from Menzie et al. (Massachusetts Bay Project, in preparation) suggest that 3.9-12 kg Hg/year enter Buzzards Bay from atmospheric deposition. Approximately 17 kg Hg/year enter Buzzards Bay in municipal effluent from New Bedford (CDM, 1989a; EPA, 1989) (Table 21). Concentrations of mercury in CSOs were less than detection limits of 0.0003 mg/L (CDM, 1989b).

Table 22. Copper inputs to Buzzards Bay

Source	Load (kg/yr)	Percent Total
Atmospheric Depostion	3000	27
New Bedford Sewage Outfall	5300	47
Other Sewage Treatment Plants	6	1
New Bedford CSOs	570	5
Runoff	530	5
Boat Paints	1667	15
TOTAL	11073	100

Copper

In the past, metal-plating industries located in New Bedford were major sources of copper. Copper was a common antifouling material in boat paints in the 1950s and 1960s, and it continues to be used today. Copper pipes used in water delivery systems also remain a major source of copper (Table 22), as indicated by the high value for the New Bedford sewage outfall.

Extrapolating from data from Boston, approximately 3,000 kg Cu/year are deposited on Buzzards Bay from the atmosphere (Menzie et al., Massachusetts Bay Project, in preparation). Currently, boat paints that contain copper may leach at a rate of 10 g/cm^2/day. Although reliable data on the number and size of boats in the bay and the extent of use of copper-containing antifouling paints do not exist, a crude estimate of copper inputs can be calculated. Assuming an average boat length of 20 feet, the maximum area coated with antifouling paint per boat would be 300,000 cm^2 (calculated as the area of a half a sphere, 20 feet

in diameter). There are approximately 4,300 boat slips in Buzzards Bay, and if each of these represents a boat, 13 kg Cu/day leach into Buzzards Bay.

Assuming that boats are in the water for approximately 4 months of the year, 1,667 kg Cu/year leach into the bay.

Municipal effluent from New Bedford contributes approximately 5,314 kg Cu/year (CDM, 1989a; EPA, 1989). An additional 568 kg/year Cu enter from New Bedford CSOs and storm drains (CDM, 1989b).

Table 23. Runoff of copper to Buzzards Bay

Land Use Type[1]	Area (km^2)	Loading factor ($kg/km^2/yr$)	Load (kg/yr)
Residential	53	3.0	159
Commercial	4.4	3.0	13
Industrial	2.4	35	84
Road	1.8	150	270
TOTAL			530

[1] From BBP (Table 11)

Using the loading factors included in Hoffman (1985) and assuming that only runoff from land areas within 0.5 miles of the shore enter the Bay, 590 kg Cu/year enter the Bay from runoff (Table 23). If runoff occurred from the entire watershed, it would total 2,800 kg Cu/year.

Arsenic

Little information is available on inputs of arsenic to Buzzards Bay (Table 24). Extrapolating from data in Menzie et al. (Massachusetts Bay Project, in prepara-

Table 24. Arsenic inputs to Buzzards Bay

Source	Load (kg/yr)	Percent Total
Atmospheric Deposition	23-150	20-63
New Bedford Sewage Outfall	90	38-80
New Bedford CSOs	Not detected	0
TOTAL	113-240	100

tion), 23-150 kg As/year enter the Bay from the atmosphere. Approximately 88 kg As/year enter Buzzards Bay in municipal effluent from New Bedford (CDM, 1989a; EPA, 1989). Arsenic was not detected at detection limits of 0.01 mg/L in studies conducted for the New Bedford CSO Facilities Plan (CDM, 1989b).

Table 25. Chromium inputs to Buzzards Bay

Source	Load (kg/yr)	Percent Total
Atmospheric Deposition	23-829	1-17
New Bedford Sewage Outfall	3500	74-89
New Bedford CSOs	430	9-11
Boat Paint	Not Calculated	-
TOTAL	3953-4759	100

Chromium

Chromium enters Buzzards Bay from the atmosphere, from municipal and industrial discharges, and from runoff (Table 25). Atmospheric inputs of chromium to Buzzards Bay equal 23-829 kg Cr/year, extrapolating from data from Boston (Menzie et al., Massachusetts Bay Project, in preparation).

Average dry weather inputs of chromium from New Bedford effluent are approximately 3514 kg/year (CDM, 1989a; EPA, 1989). Inputs from CSOs and storm drains are lower, approximately 432 kg Cr/year (CDM, 1989b).

Chromium is also used in some boat paints, but insufficient data are available to estimate inputs of chromium to the Bay from this source.

Table 26. Silver inputs to Buzzards Bay

Source	Load (kg/yr)	Percent Total
New Bedford Sewage Outfall	330	100
New Bedford CSOs	Not Detected	0
TOTAL	330	100

Silver

Municipal effluent from New Bedford contributes approximately 330 kg Ag/year to Buzzards Bay (CDM, 1989a; EPA, 1989) (Table 26). Concentrations of silver were less than the detection limit of 0.01 mg/L in studies conducted for the New Bedford CSO Facilities Plan (CDM, 1989b).

Table 27. Nickel inputs to Buzzards Bay

Source	Load (kg/yr)	Percent Total
Atmospheric Deposition	1800	39
New Bedford Sewage Outfall	2800	61
New Bedford CSOs	Not Detected	0
TOTAL	4600	100

Nickel

Inputs of nickel from atmospheric deposition onto Buzzards Bay total approximately 1800 kg/year, extrapolating from data for Boston (Menzie et al., Massachusetts Bay Project, in preparation) (Table 27).

Inputs of nickel to Buzzards Bay include approximately 2,792 kg/year from dry weather effluent discharge from New Bedford (CDM, 1989a; EPA, 1989). Nickel was not detected at detection limit of 0.03 mg/L in most CSO and stormwater samples in New Bedford (CDM, 1989b).

Conclusions

Concentrations of fecal coliforms vary by time and within and among receiving waters (Figures 3-13, Appendix A). Except for major sources, such as sewage treatment plants and the New Bedford CSOs, inputs of pathogens to Buzzards Bay are difficult to quantify. Runoff is the most prominent source of pathogens to the Bay. Future monitoring should refine analysis methods. Because localized effects of pathogens threaten beach use and shellfish harvest, future monitoring should local sources and fate of pathogens.

The New Bedford sewage treatment plant is the major source of nitrogen to the Bay. Inputs from precipitation and from septic systems are also important sources of nitrogen to the Bay as a whole. Locally, agriculture and runoff, as well as septic systems, may be significant sources of nitrogen. Future monitoring should focus on sources, fates, and effects of nitrogen inputs in the Bay as a whole and within local embayments.

Toxic compounds enter Buzzards Bay through many sources"atmospheric deposition, oil spills, sewage treatment plants, and runoff. The industrial area around New Bedford has been the major source of many toxic contaminants to Buzzards Bay, and future monitoring should appraise the pollution remediation projects being undertaken in that area. Monitoring should also address effectiveness of actions to prevent oil-spill pollution and to mitigate stormwater discharges.

References

CCPEDC. 1979. Cape Cod Planning and Development Commission. Draft EIS and proposed 208 water quality management plan for Cape Cod. 1st District Court House, Barnstable, Massachusetts.

CDM. 1989a. Draft Environmental Impact Report for New Bedford Facilities Plan. Prepared for the City of New Bedford.

CDM. 1989b. Draft New Bedford CSO Facilities Plan. Prepared for the City of New Bedford.

EPA. 1989. Draft Environmental Impact Statement for New Bedford Facilities Plan.

Farrington, J.W. and J. McDowell Capuzzo. 1990. Toxic chemicals in Buzzards Bay: sources, fates, and effects. Prepared for the Buzzards Bay Project.

Giblin, A.E. and A. Gaines. 1990. Nitrogen inputs to a marine embayment: the importance of groundwater. Biogeochemistry 10: 309-328.

Grimes, D.J. and G.R. Heufelder. In press. Synthesis report on pathogens. Buzzards Bay Project.

Groet. 1976. Regional and local variations in heavy metal concentrations of bryophytes in the northeastern United States. Oikos 27:445-456.

Heufelder, G.R. 1988. Bacteriological monitoring in Buttermilk Bay. BBP-88-03. EPA 503/4-88-001.

Hoffman, E. 1985. Urban runoff pollutant inputs to Narragansett Bay: comparison to point sources. Estuarine Quality: Proc. Conf. Perspectives Nonpoint Source Poll. 159-164.

Horsely and Witten. In preparation. Buzzards Bay Project.

Kadlec, R.H. and D.L. Tilton. 1979. The use of freshwater wetlands as a tertiary wastewater treatment alternative. CRC Crit. Rev. Environ. Control 185-212.

Kelly, J. and I. Valiela. In press. Nutrients and trophic status of Buzzards Bay, Massachusetts. Buzzards Bay Project.

Koppelman, J. 1982. The Long Island Segment of the Nationwide Urban Runoff Program. The Log Island Region Planning Board.

Kuzminski and Jackivicz. 1972. Interaction of outboard motors with the aquatic environment: causative factors and effects. Prepared for the Massachusetts Water Resources Commission under Contract No. 15-51451.

Mayer, G. 1982. Ecological Stress and the New York Bight: Science and Management. Estuarine Research Foundation. Columbia, SC.

Smith, J.P. 1988. The status of municipal wastewater systems with discharges to coastal waters. Prepared for Buzzards Bay Project.

Stenner, T.W., T.C. Gulbransen, and E.M. Baptiste. 1988. Water Quality Data Assessment for Buzzards Bay, Massachusetts. Prepared for the Buzzards Bay Project.

Teal, J.M. and B. Howes. In preparation. Nitrogen balance in a Massachusetts cranberry bog. Buzzards Bay Project.

Valiela, I. and J.E. Costa. 1988. Eutrophication of Buttermilk Bay, a Cape Cod coastal embayment: concentrations of nutrients and watershed nutrient budgets. Environ. Manage. 12(4): 539-553.

Valiela, I. and J.M. Teal. 1979. Nitrogen budget of a salt marsh ecosystem. *Nature* 280: 652-656.

Appendix A
Coliforms in Buzzards Bay

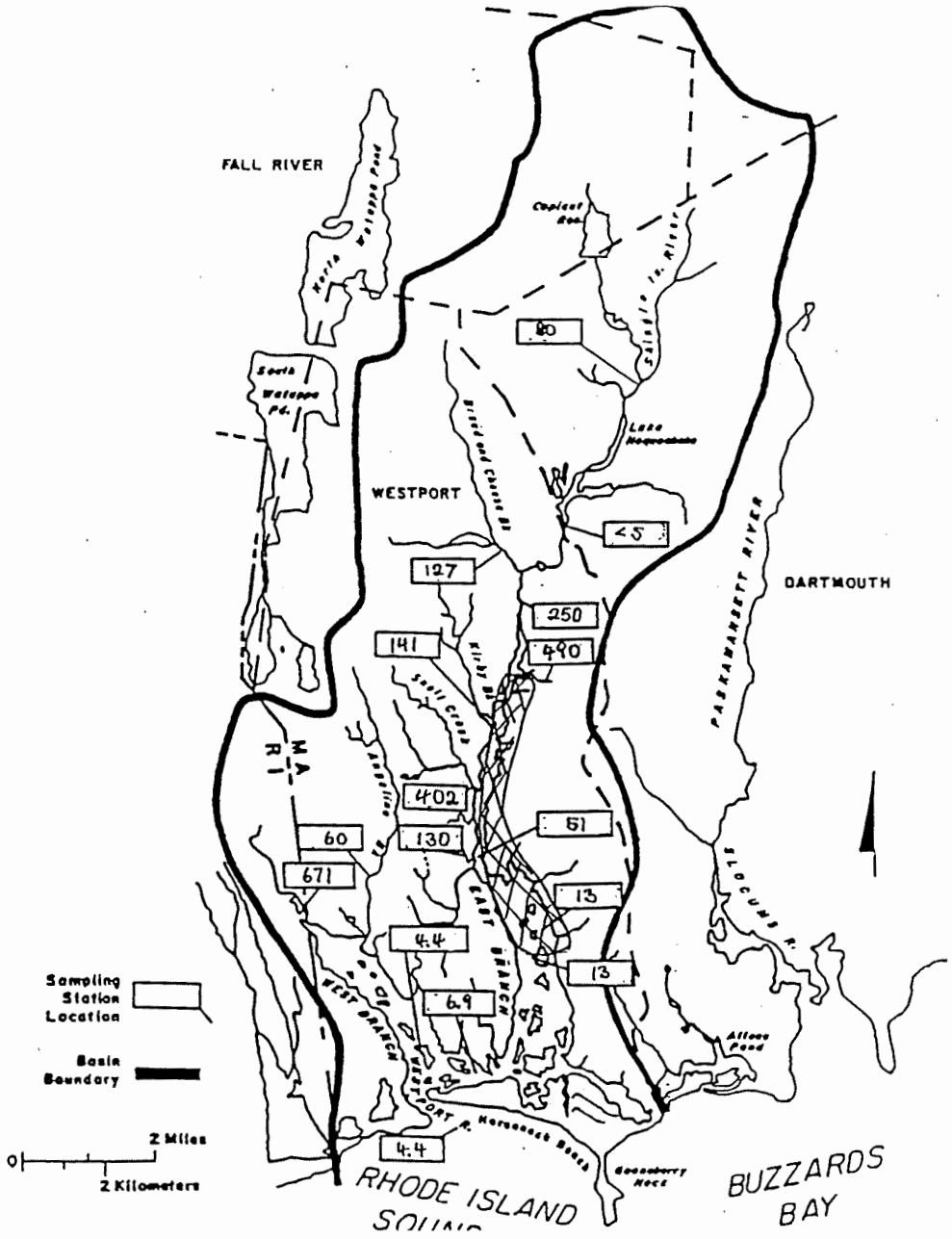

Figure 3. Mean fecal coliforms (colonies/100 ml) in the Westport River drainage basin. Shaded area: stations sampled for the Bacteriological Data Report (BBP-89-19). See table A-1 for locations and additional coliform counts

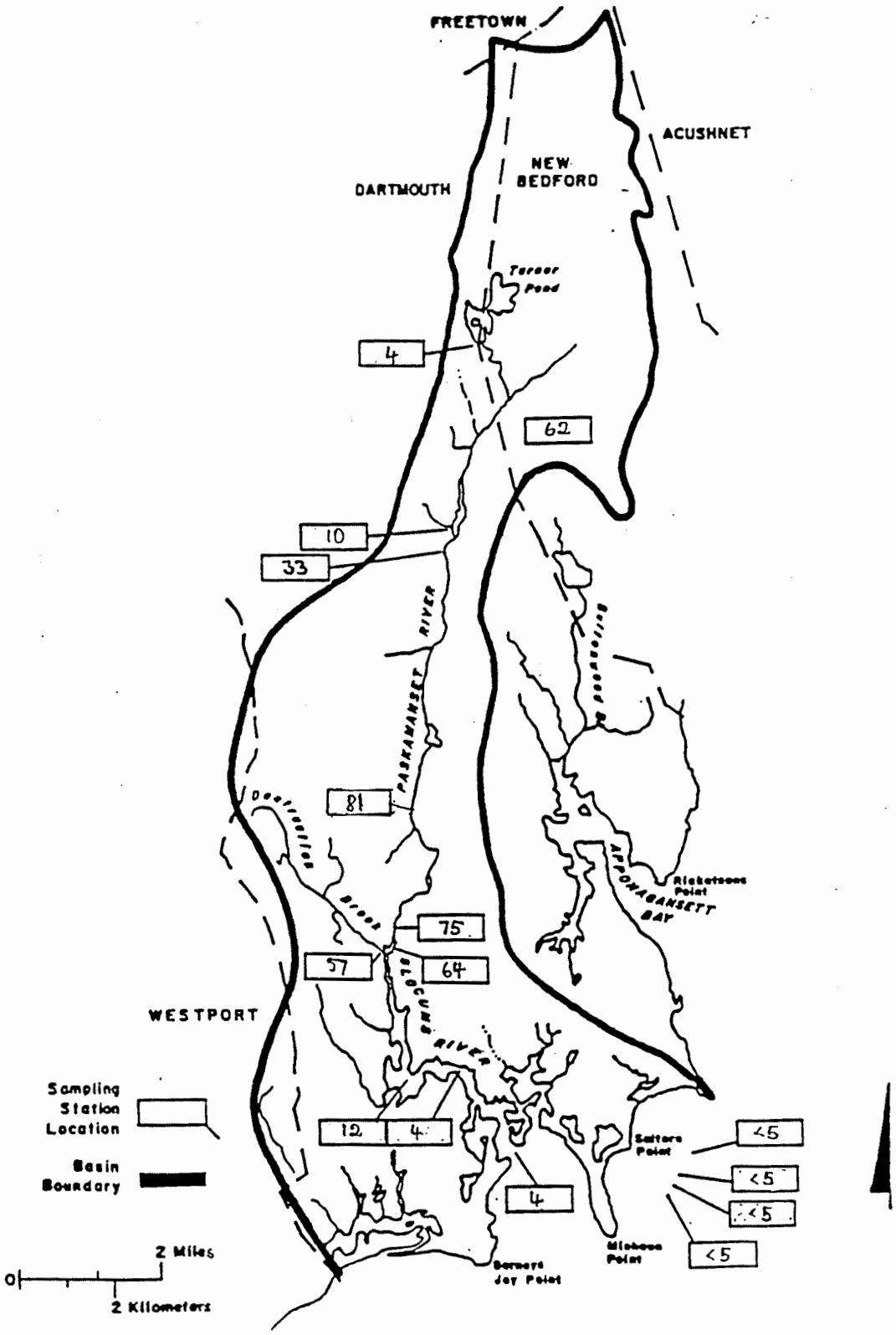

Figure 4. Mean fecal coliforms (colonies/100 ml) in the Paskamansett/Slocums River drainage basin. See Table A-2 for locations.

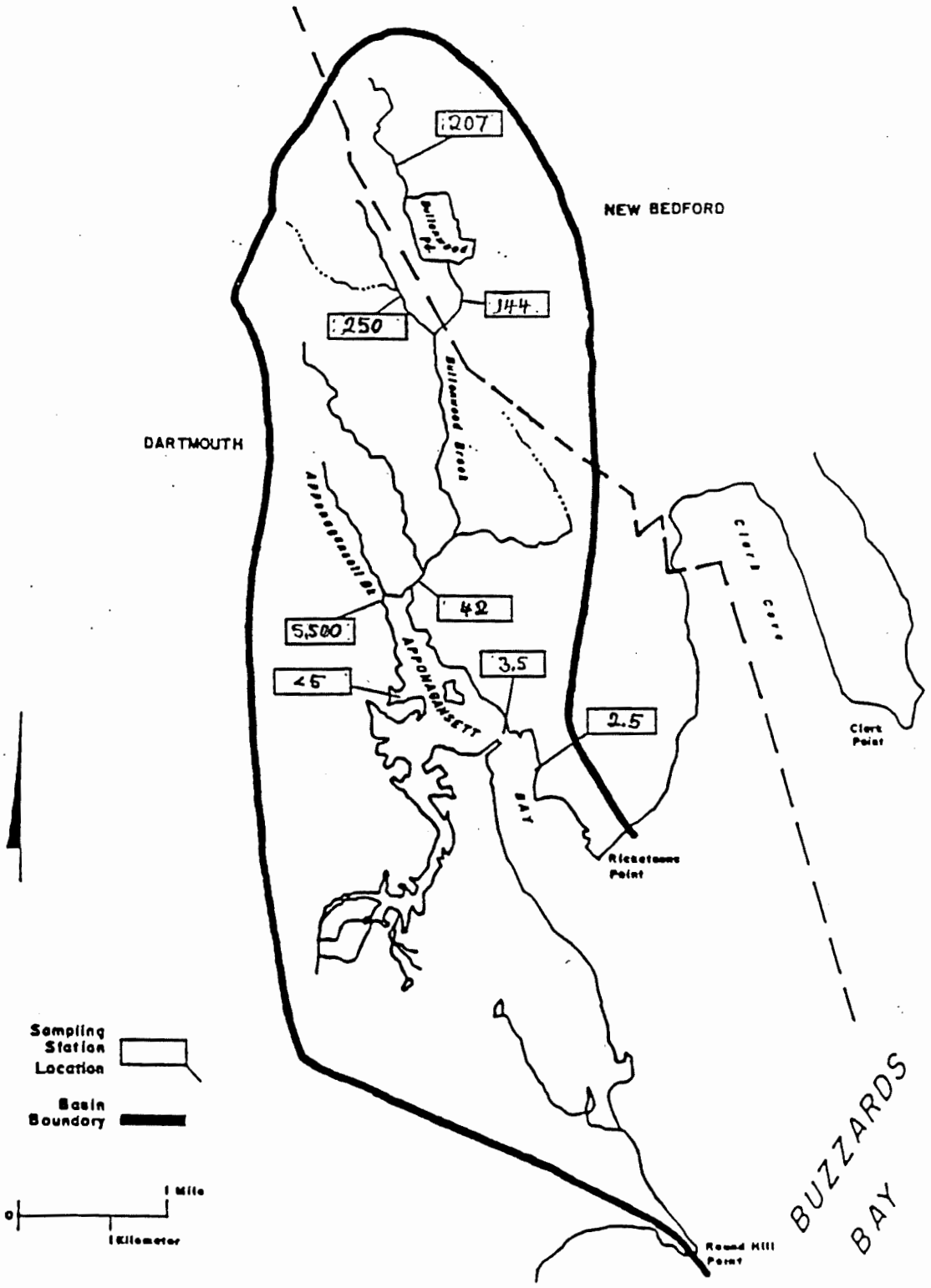

Figure 5. Mean fecal coliforms (colonies/100 ml in the Buttonwood Brook/Apponagansett Bay drainage Basin. See Table A-3 for locations.

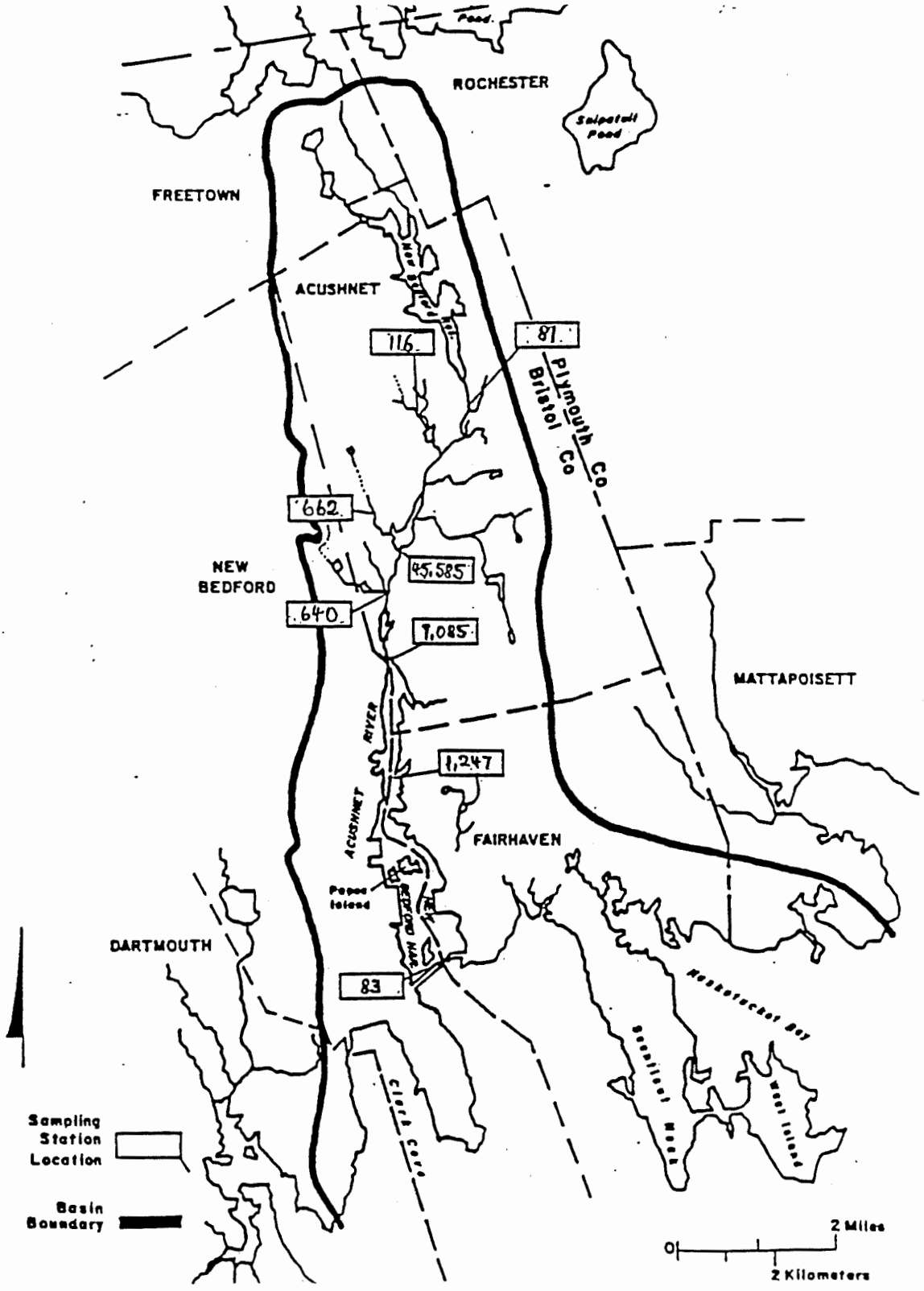

Figure 6. Mean fecal coliforms (colonies/100 ml) in the Acushnet River/New Bedford Harbor drainage basin. See Table A-4 for location.

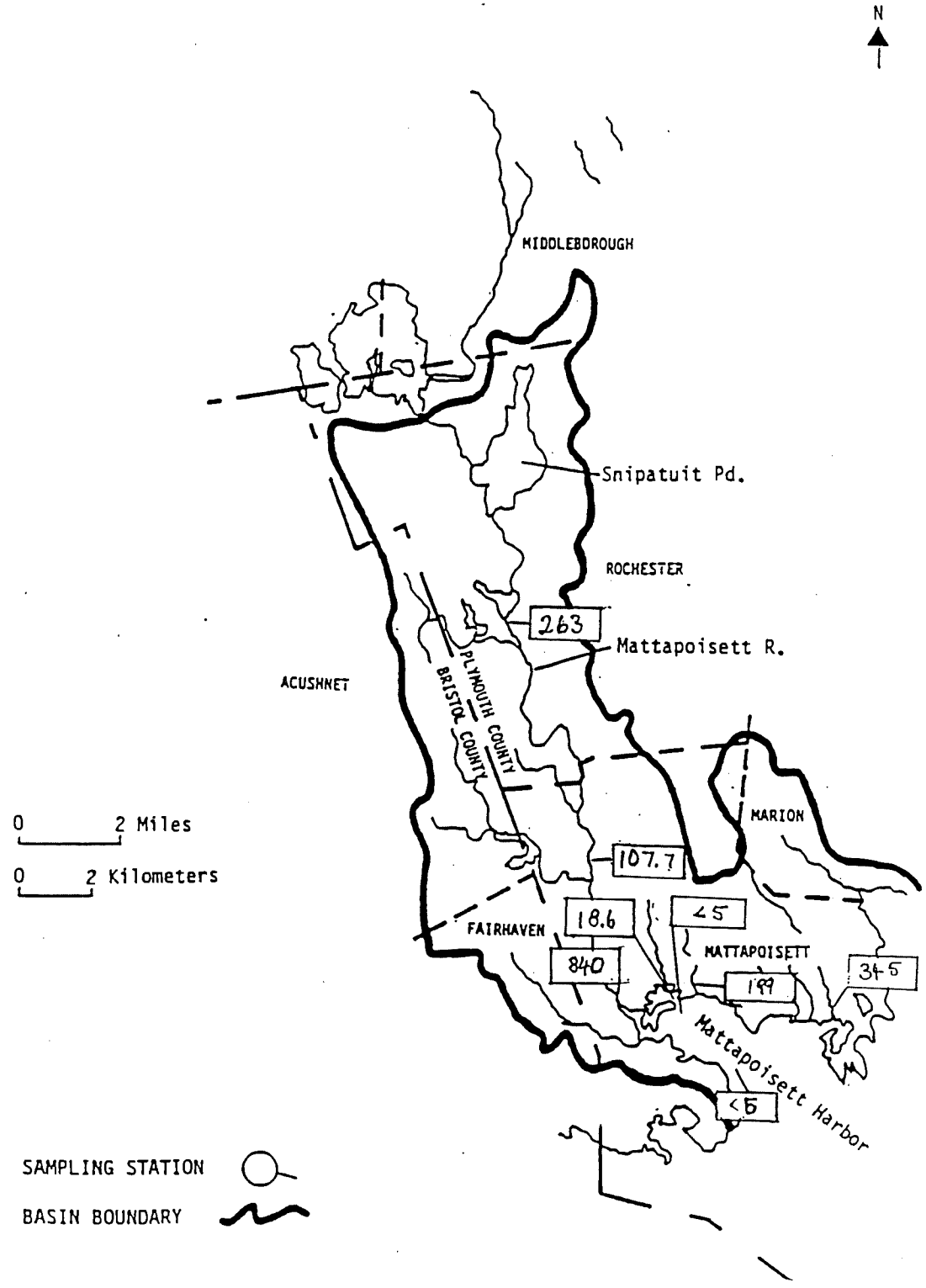

Figure 7. Mean fecal coliforms (colonies/100 ml) in the Mattapoisett River/Harbor drainage basin. See Table A-5 for locations.

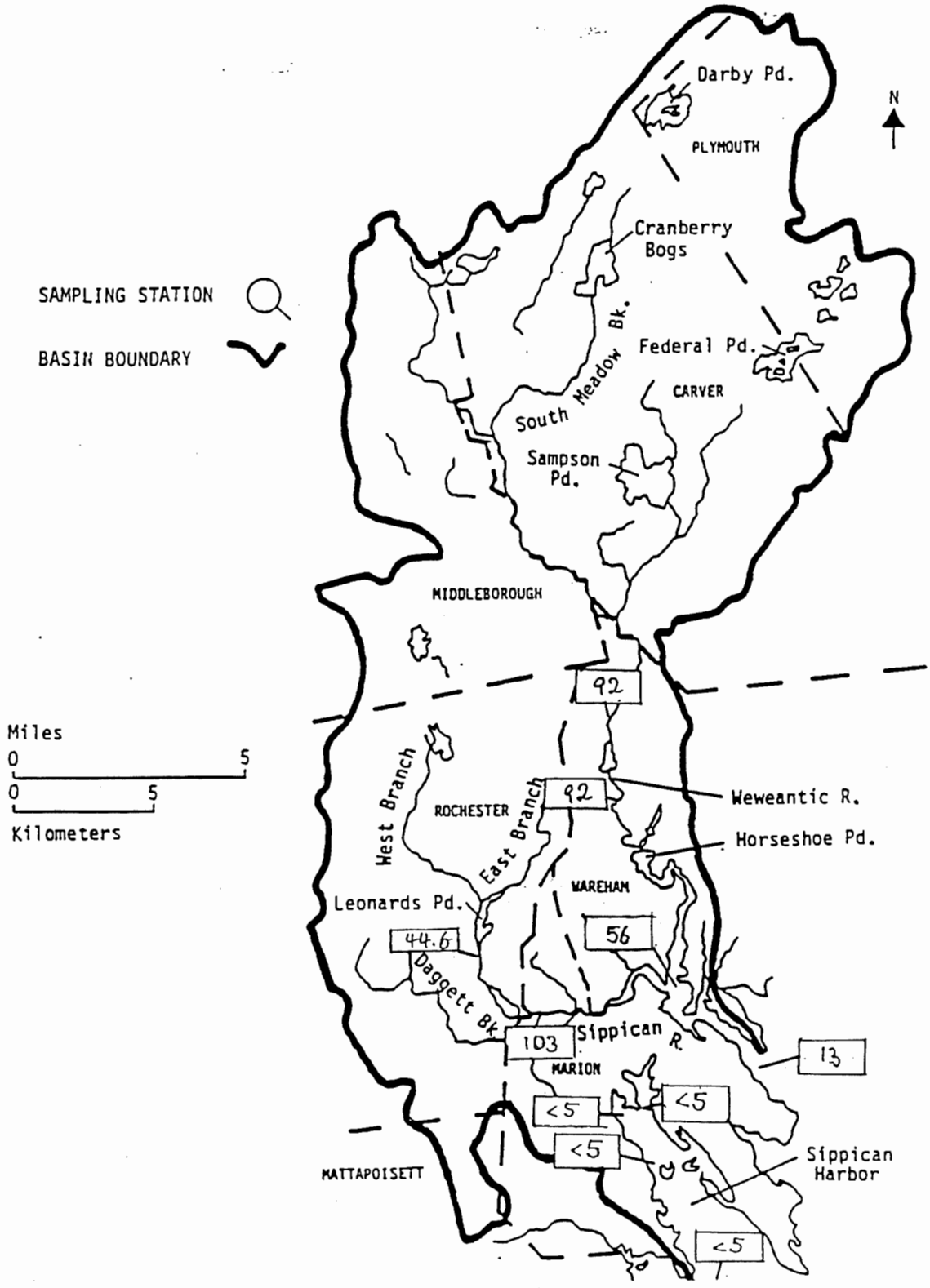

Figure 8. Mean fecal coliforms (colonies/100 ml) in the Weweantic-Sippican River/Sippican Harbor drainage basin. See Table A-6 for locations.

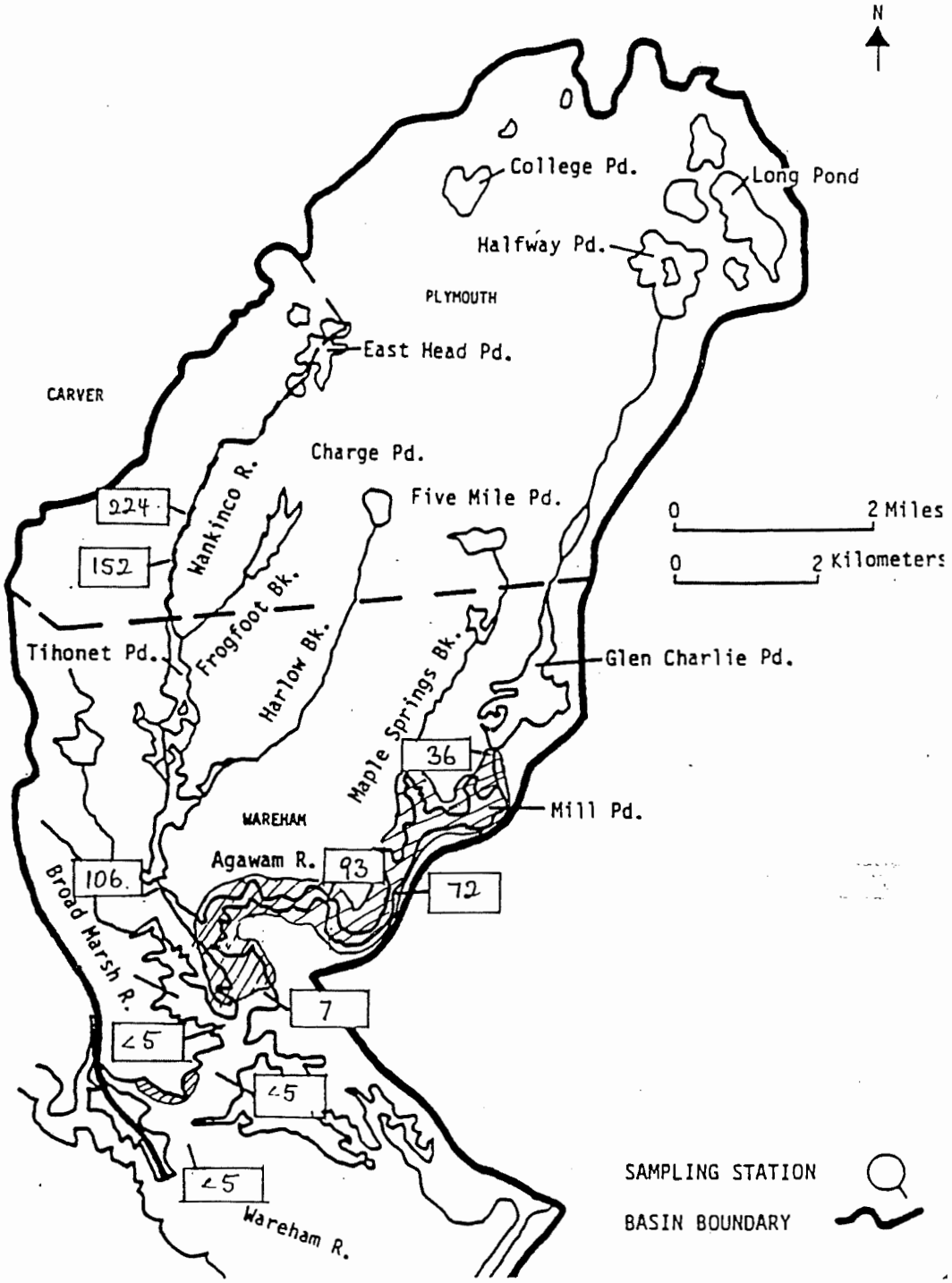

Figure 9. Mean fecal coliforms (colonies/100 ml) in the Agawam-Wankinco-Wareham River drainage basin. Shaded areas: stations sampled for the Bacteriological Data Report (BBP-89-10). See Table A-7 for locations and additional coliform counts.

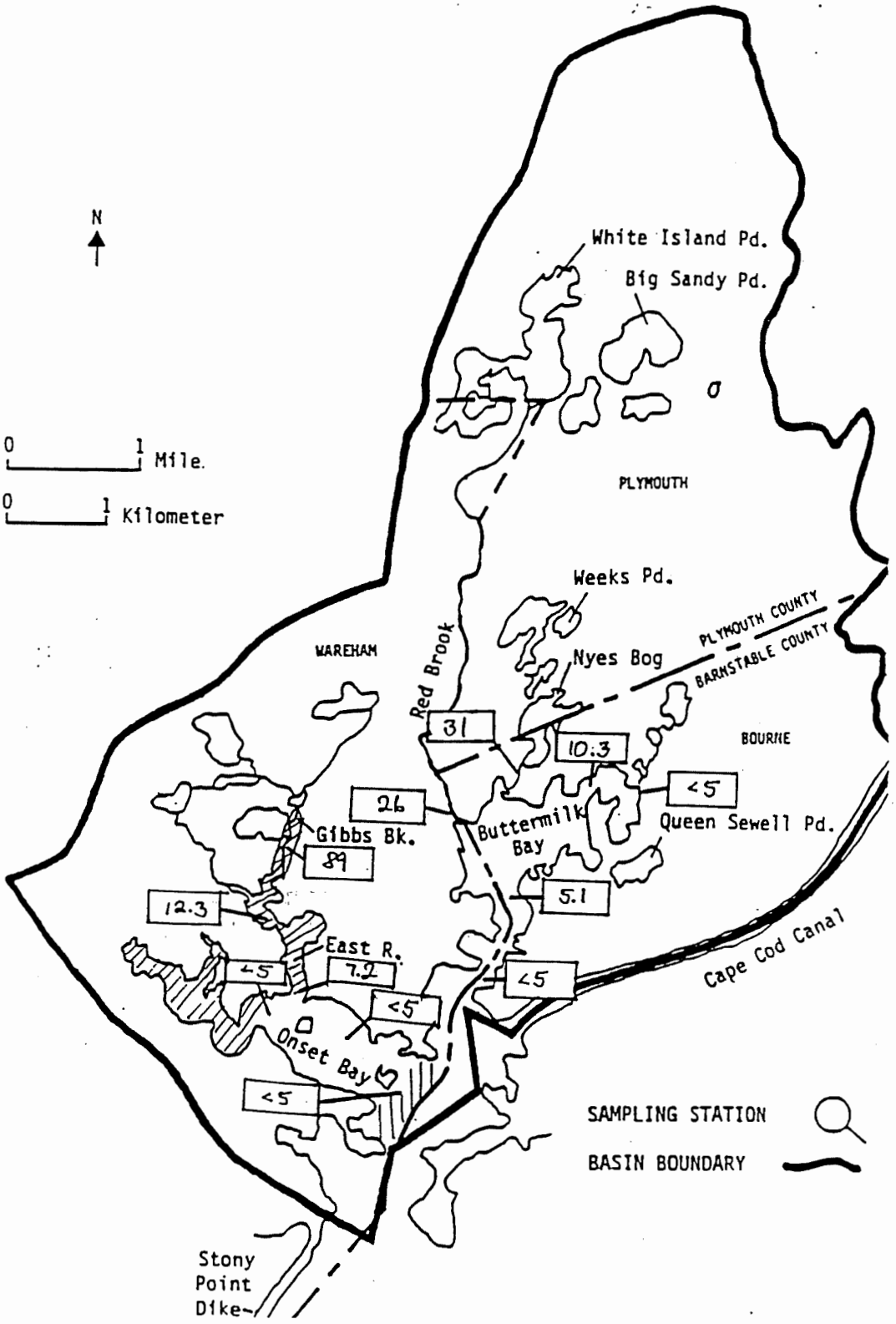

Figure 10. Mean fecal coliforms (colonies/100 ml) in the Buttermilk Bay/Onset Bay drainage basin. Shaded areas: stations sampled for the Bacteriological Data Report (BBP-89-19). See Table A-8 for locations and additional coliform counts.

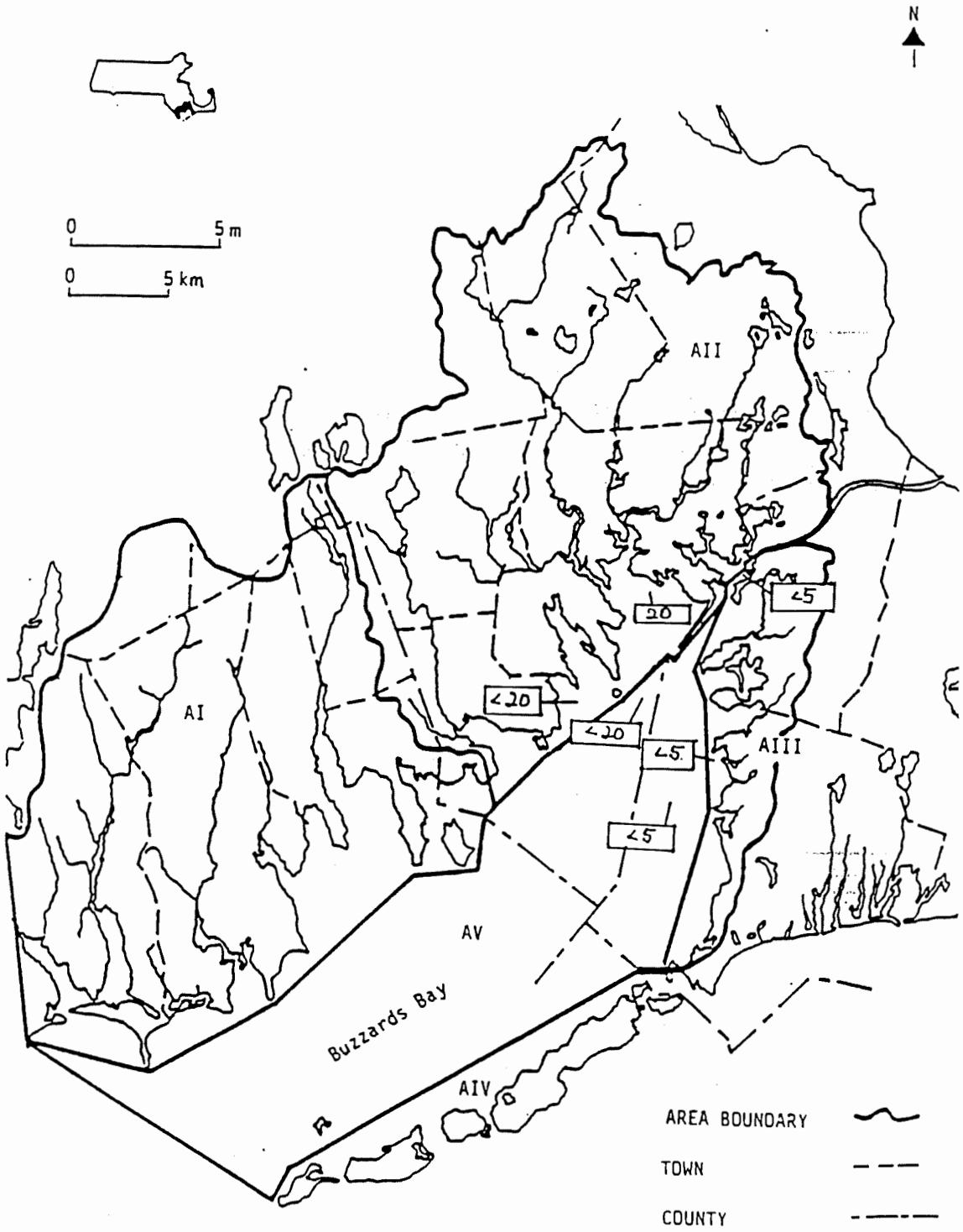

Figure 13. Mean fecal coliforms (colonies/100 ml) at selected open water stations in Buzzards Bay. See Table A-11 for locations.

Table A-1. Fecal coliforms in Buzzards Bay-Westport River Drainage Basin.

Station	Latitude °N	Longitude °W	Mean Fecal Coliforms	Source	Sampling Date
8WPW	41°32'50"	71°06'20"	671	2	June 1986
14WPW	41°31'10"	71°05'30"	4.4	2	June 1986
15RIS	41°24'57"	71°05'00"	4.4	2	June 1986
9AB	41°36'20"	71°03'12"	*60	2	June 1986
13WPH	41°30'51"	71°04'14"	6.9	2	June 1986
6SNC	41°34'54"	71°04'38"	402	2	June 1986
7LFS	41°34'14"	71°02'13"	*130	2	June 1986
5KB	41°36'01"	71°04'56"	141	2	June 1986
2BCB	41°38'02"	71°03'48"	127	2	June 1986
1SIR	41°40'13"	71°01'33"	*80	2	June 1986
3WPE	41°38'18"	71°02'55"	*<5	2	June 1986
4WPE	41°37'16"	71°03'38"	250	2	June 1986
10MF	?	?	*490	2	June 1986
609	41°36'35"	71°03'50"	80,000	3	July 1986
608	41°36'30"	71°03'45"	50,000	3	July 1986
1508	41°36'15"	71°03'45"	2,900	3	September 1986
1001	41°35'50"	71°04'00"	9,800	3	August 1986
1002	41°35'50"	71°04'03"	34,000	3	August 1986
607	41°35'45"	71°04'15"	2,200	3	July 1986
1507	41°35'40"	71°04'12"	*170	3	September 1986
605	41°35'35"	71°04'10"	1,800	3	July 1986
604	41°35'35"	71°04'13"	2,200	3	July 1986
1710	41°35'30"	71°04'40"	*11,000	3	October 1986
1701	41°35'10"	71°05'00"	*12,000	3	October 1986
1702	41°35'15"	71°04'58"	*24000	3	October 1986
1711	41°35'14"	71°04'40"	*19,000	3	October 1986
1712	41°35'20"	71°04'45"	*25,000	3	October 1986
606	41°35'30"	71°04'13"	1,800	3	October 1986
603	41°35'30"	71°04'15"	1,940	3	October 1986
1708	41°35'30"	71°04'00"	*20,000	3	October 1986
1709	41°35'29"	71°04'00"	550	3	October 1986
1506	41°35'22"	71°04'10"	*1,000	3	September 1986
1505	41°35'20"	71°04'10"	*180	3	September 1986

Table A-1 continued

Station	Latitude °N	Longitude °W	Mean Fecal Coliforms	Source	Sampling Date
BWPW	41°32'50"	71°06'20"	671	2	June 1986
1509	41°35'20"	71°03'50"	2,100	3	September 1986
1504	41°35'17"	71°04'18"	*70	3	Septembeur 1986
1503	41°35'15"	71°04'20"	*90	3	September 1986
1510	41°35'15"	71°04'10"	*60	3	September 1986
1501	41°35'05"	71°04'30"	*900	3	September 1986
1703	41°35'10"	71°04'35"	*20,000	3	October 1986
1704	41°35'10"	71°04'36"	*18,000	3	October 1986
1706	41°35'02"	71°04'35"	*14,000	3	October 1986
1003	41°35'00"	71°04'15"	*4,200	3	August 1986
801	41°34'51"	71°04'30"	113	3	July 1986
802	41°34'50"	71°04'29"	98	3	July 1986
803	41°34'49"	71°04'28"	53	3	July 1986
804	41°34'48"	71°04'27"	13	3	July 1986
805	41°34'47"	71°04'27"	13	3	July 1986
1705	41°34'45"	71°04'35"	2,800	3	October 1986
1502	41°34'45"	71°04'27"	*90	3	September 1986
806	41°34'45"	71°04'26"	25	3	July 1986
602	41°34'45"	71°04'37"	5,900	3	July 1986
601	41°34'40"	71°00'35"	4,300	3	July 1986
1004	41°34'40"	71°04'15"	*2,400	3	August 1986
1707	41°34'40"	71°04'33"	3,900	3	October 1986
1005	41°34'35"	71°04'20"	*2,100	3	August 1986
1006	41°34'30"	71°04'17"	*4,000	3	August 1986
610	41°34'15"	71°04'30"	2,300	3	July 1986
807	41°34'15"	71°04'30"	10	3	July 1986
10WPE	41°34'13"	71°04'19"	51	2	June 1986
1301	41°33'52"	71°04'28"	*1,000	3	August 1986
1302	41°33'48"	71°04'29"	*500	3	August 1986
1303	41°33'40"	71°04'26"	*1,100	3	August 1986
1304	41°33'38"	71°04'26"	*1,100	3	August 1986
1305	41°33'36"	71°04'23"	*800	3	August 1986
809	41°33'36"	71°03'39"	5	3	July 1986
808	41°33'36"	71°04'18"	5	3	July 1986
1306	41°33'29"	71°04'18"	*600	3	August 1986
1007	41°33'18"	71°03'26"	*<20	3	August 1986

Table A-1 continued

Station	Latitude °N	Longitude °W	Mean Fecal Coliforms	Source	Sampling Date
8WPW	41°32'50"	71°06'20"	671	2	June 1986
1307	41°33'14"	71°03'21"	*75	3	August 1986
1008	41°33'11"	71°03'22"	*20	3	August 1986
1010	41°33'08"	71°03'17"	*20	3	August 1986
1308	41°33'08"	71°03'15"	45	3	August 1986
1009	41°33'07"	71°03'20"	20	3	August 1986
1309	41°33'06"	71°03'13"	*60	3	August 1986
1310	41°33'04"	71°03'12"	50	3	August 1986
1011	41°33'02"	71°03'14"	*40	3	August 1986
1311	41°33'02"	71°03'11"	*65	3	August 1986
810	41°33'00"	71°03'13"	5	3	July 1986
1312	41°32'58"	71°03'10"	*90	3	August 1986
11WPE	41°32'58"	71°02'52"	13	2	June 1986
1012	41°32'56"	71°03'11"	*<20	3	August 1986
812	41°32'43"	71°03'40"	<5	3	July 1986
1013	41°32'42"	71°03'25"	*<20	3	August 1986
811	41°32'39"	71°03'09"	5	3	July 1986
12WPE	41°32'35"	71°03'38"	13	2	June 1986

Table A-2. Fecal Coliforms in Buzzards Bay--Pasmakansett/Slocums River Drainage Basin.

Station	Latitude °N	Longitude °W	Mean Fecal Coliforms	Source	Sampling Date
1PR	41°40'43"	70°58'39"	4	2	July 1986
2PR	41°39'18"	70°58'53"	62	2	July 1986
3PR	41°38'25"	70°59'11"	10	2	July 1986
4PR	41°38'18"	70°59'11"	33	2	July 1986
5PR	41°35'06"	70°59'27"	81	2	July 1986
6PR	41°34'17"	71°00'18"	75	2	July 1986
7DB	41°34'21"	71°00'47"	57	2	July 1986
14SR	41°34'20"	71°00'48"	64	2	July 1986
15SR	41°32'45"	71°00'03"	12	2	July 1986
16SR	41°32'38"	70°59'10"	4	2	July 1986
17SR	41°31'42"	70°58'40"	4	2	July 1986
18BB	41°31'33"	70°56'08"	*<5	2	July 1986
19BB	41°31'15"	70°56'35"	*<5	2	July 1986
20BB	41°31'10"	70°56'37"	*<5	2	July 1986
21BB	41°31'08"	70°56'44"	*<5	2	July 1986

Table A-3. Fecal coliforms in Buzzards Bay -- Buttonwood Brook/Aponagansett Bay Drainage Basin.

Station	Latitude °N	Longitude °W	Mean Fecal Coliforms	Source	Sampling Date
8BWB	41°38'10"	70°57'24"	207	2	July 1986
9BWB	41°37'38"	70°57'14"	144	2	July 1986
20T8W	41°36'05"	70°37'49"	*250	2	July 1986
10BWB	41°36'05"	70°57'35"	42	2	July 1986
21ABK	41°36'05"	70°57'47"	*5,500	2	July 1986
22AB	41°35'24"	70°57'42"	*<5	2	July 1986
12AB	41°35'10"	70°35'10"	3.5	2	July 1986
13AB	41°35'05"	70°56'50"	2.5	2	July 1986

Table A-4. Fecal coliforms in Buzzards Bay -- Acushnet River/New Bedford Harbor Drainage Basin

Station	Latitude °N	Longitude °W	Mean Fecal Coliforms	Source	Sampling Date
1ACR	41°43'27"	70°53'53"	81	2	October 1986
2DB	41°42'43"	70°54'46"	116	2	October 1986
3UNB	41°42'08"	70°55'16"	662	2	October 1986
4ACR	41°41'51"	70°54'56"	45,585	2	October 1986
5UNB	41°41'32"	70°55'15"	640	2	October 1986
6ACR	41°40'55"	70°55'12"	7,085	2	October 1986
8ACR	41°39'27"	70°55'07"	1,247	2	October 1986
9NBH	41°37'27"	70°54'23"	83	2	October 1986

Table A-5. Fecal coliforms in Buzzards Bay -- Mattapoisett River/Harbor Drainage Basin.

Station	Latitude °N	Longitude °W	Mean Fecal Coliforms	Source	Sampling Date
34MR	41°44'10"	70°51'45"	236	1	August 1985
35MR	41°41'05"	70°50'20"	107.7	1	August 1985
36MR	41°39'45"	70°50'20"	840	1	August 1985
38MH	41°39'24"	70°46'42"	18.6	1	August 1985
40MH	41°39'28"	70°48'52"	*<5	1	August 1985
41MH	41°38'15"	70°47'25"	*<5	1	August 1985
39MH	41°39'28"	70°47'05"	199	1	August 1985
37PI	41°38'56"	70°46'42"	345	1	August 1985

Table A-6. Fecal coliforms in Buzzards Bay -- Weweantic/Sippican River/Sippican Harbor Drainage Basin.

Station	Latitude °N	Longitude °W	Mean Fecal Coliforms	Source	Sampling Date
30SH	41°45'54"	70°42'42"	*<5	1	August 1985
31SH	41°45'26"	70°42'47"	*<5	1	August 1985
32SH	41°45'12"	70°41'52"	*<5	1	August 1985
33SH	41°40'39"	70°44'23"	*<5	1	August 1985
28SR	41°44'49"	70°48'14"	44.6	1	August 1985
29SR	41°44'05"	70°46'30"	103	1	May/August 1985
19WE	41°47'54"	70°45'50"	92	1	May/August 1985
20WE	41°46'13"	70°45'17"	92	1	May/August 1985
21WE	41°44'15"	70°44'52"	56	1	May/August 1985
22WE	41°43'10"	70°43'14"	13	1	May/August 1985

Table A-7. Fecal coliforms in Buzzards Bay -- Agawam-Wankinco-Wareham River Drainage Basin.

Station	Latitude °N	Longitude °W	Mean Fecal Coliforms	Source	Sampling Date
16WR	41°49'31"	70°42'30"	224	1	August 1985
17WR	41°48'30"	70°43'02"	152	1	August 1985
18WR	41°45'58"	70°43'20"	106	1	May/August 1985
304	41°44'07"	70°43'27"	<5	3	June 1986
305	41°44'06"	70°43'19"	<5	3	June 1986
27WA	41°43'46"	70°43'10"	<5	1	May/August 1985
201	42°35'00"	70°40'40"	342	3	June 1986
202	42°34'58"	70°40'41"	352	3	June 1986
203	42°34'57"	70°40'42"	323	3	June 1986
204	42°34'56"	70°40'39"	365	3	June 1986
205	42°34'56"	70°40'43"	328	3	June 1986
206	42°34'55"	70°40'42"	310	3	June 1986
207	42°34'55"	70°40'43"	367	3	June 1986
208	42°34'54"	70°40'43"	338	3	June 1986
303	41°47'00"	70°39'25"	5,300	3	June 1986
13AR	41°46'57"	70°39'20"	36	1	May/August 1985
302	41°46'45"	70°40'20"	10	3	June 1986
209	41°46'25"	70°41'15"	458	3	June 1986
301	41°45'45"	70°40'35"	15	3	June 1986
14AR	41°45'40"	70°40'30"	72	1	May/August 1985
210	41°45'40"	70°41'50"	497	3	June 1986
15AR	41°45'48"	70°41'22"	93	1	May/August 1985
211	41°45'34"	70°42'37"	233	3	June 1986
701	41°45'25"	70°42'46"	*90	3	July 1986
702	41°45'22"	70°42'43"	*125	3	July 1986
706	41°45'21"	70°42'32"	140	3	July 1986
901	41°45'19"	70°42'41"	*80	3	August 1986
703	41°45'19"	70°42'39"	105	3	July 1986
503	41°45'19"	70°42'25"	10	3	July 1986
902	41°45'18"	70°42'39"	*20	3	August 1986
504	41°45'18"	70°42'19"	*<5	3	July 1986
501	41°45'16"	70°42'36"	53	3	July 1986
903	41°45'15"	70°42'37"	*<100	3	August 1986

Table A-7 continued

904	41°45'14"	70°42'37"	*100	3	August 1986
502	41°45'13"	70°42'35"	*30	3	July 1986
704	41°45'13"	70°42'37"	*250	3	July 1986
705	41°45'12"	70°42'35"	*320	3	July 1986
905	41°45'11"	70°42'32"	*240	3	August 1986
707	41°45'10"	70°42'33"	*230	3	July 1986
407	41°45'09"	70°42'32"	*20	3	July 1986
408	41°45'09"	70°42'30"	*<10	3	July 1986
409	41°45'09"	70°42'28"	*<10	3	July 1986
908	41°45'07"	70°42'31"	*40	3	August 1986
906	41°45'06"	70°42'31"	*12,000	3	August 1986
907	41°45'06"	70°42'29"	40	3	August 1986
708	41°45'05"	70°42'33"	*170	3	July 1986
709	41°45'04"	70°42'31"	*160	3	July 1986
401	41°45'03"	70°42'09"	*<10	3	July 1986
402	41°45'03"	70°42'09"	*<10	3	July 1986
403	41°45'03"	70°42'04"	*<10	3	July 1986
24WA	41°44'59"	70°42'11"	7	1	May/August 1985
404	41°44'57"	70°42'04"	*<10	3	June 1986
405	41°44'57"	70°42'06"	*20	3	June 1986
506	41°44'57"	70°42'08"	*10	3	July 1986
406	41°44'57"	70°42'09"	*<10	3	June 1986
505	41°44'55"	70°42'03"	*15	3	July 1986
411	41°44'55"	70°42'33"	*10	3	June 1986
412	41°44'54"	70°42'29"	*<10	3	June 1986
212	41°44'54"	70°42'21"	53	3	June 1986
507	41°44'54"	70°42'13"	7	3	July 1986
508	41°44'54"	70°42'18"	*65	3	July 1986
711	41°55'54"	70°42'06"	*85	3	July 1986
510	41°44'53"	70°42'29"	*135	3	July 1986
712	41°44'53"	70°42'08"	*80	3	July 1986
711	41°44'51"	70°42'22"	*85	3	July 1986
509	41°45'50"	70°42'22"	*90	3	July 1986
25WA	41°44'49"	70°42'27"	<5	1	May/August 1985
26WA	41°44'09"	70°42'42"	<5	1	May/August 1985

Table A-8. Fecal coliforms in Buzzards Bay -- Buttermilk Bay/Onset Bay Drainage Basin.

Station	Latitude °N	Longitude °W	Mean Fecal Coliforms	Source	Sampling Date
6GB	41°45'26"	70°39'15"	89	1	August 1985
1212	41°45'25"	70°39'16"	*200	3	August 1986
110	41°45'25"	70°39'13"	25	3	June 1986
1205	41°45'22"	70°39'06"	*250	3	August 1986
1206	41°45'19"	70°39'07"	*240	3	August 1986
1207	41°45'18"	70°39'06"	*195	3	August 1986
1208	41°45'15"	70°39'06"	*95	3	August 1986
1209	41°45'14"	70°39'04"	*90	3	August 1986
101	41°45'13"	70°39'27"	100	3	June 1986
102	41°45'13"	70°39'29"	65	3	June 1986
1204	41°45'12"	70°39'38"	*85	3	August 1986
1804	41°45'04"	70°39'07"	*<5	3	October 1986
103	41°45'03"	70°39'25"	60	3	June 1986
104	41°45'03"	70°39'29"	65	3	June 1986
1803	41°45'02"	70°39'10"	*<5	3	October 1986
1805	41°44'58"	70°39'05"	10	3	October 1986
1802	41°44'58"	70°39'13"	*<5	3	October 1986
1210	41°44'56"	70°39'33"	*10	3	August 1986
1202	41°44'56"	70°39'41"	*365	3	August 1986
1201	41°44'55"	70°39'38"	*335	3	August 1986
1203	41°44'55"	70°39'44"	*400	3	August 1986
105	41°44'54"	70°39'28"	60	3	June 1986
106	41°44'54"	70°39'25"	80	3	June 1986
107	41°44'54"	70°39'18"	115	3	June 1986
1806	41°44'54"	70°39'05"	*10	3	October 1986
1801	41°44'53"	70°39'19"	<5	3	October 1986
1211	41°44'50"	70°39'09"	*<5	3	August 1986
108	41°44'49"	70°39'21"	25	3	June 1986
109	41°44'49"	70°39'17"	<10	3	June 1986
1807	41°44'41"	70°39'18"	*5	3	October 1986
1808	41°44'40"	70°39'20"	*10	3	October 1986
1107-13	41°44'38"	70°38'09"	73	3	August 1986
1114	41°44'37"	70°37'58"	*10	3	August 1986

Table A-8 continued

1115	41°44'34"	70°37'37"	*<10	3	August 1986
1106	41°44'34"	70°38'13"	*<10	3	August 1986
8MC	41°44'32"	70°39'14"	12.3	1	August 1985
9ER	41°44'33"	70°39'15"	7.2	1	August 1985
1102-05	41°44'30"	70°38'16"	47.5	3	August 1986
1101	41°44'24"	70°38'16"	*20	3	August 1986
1405	41°45'27"	70°39'16"	<2	3	August 1986
1404	41°45'17"	70°39'49"	<2	3	August 1986
1409	41°45'17"	70°39'15"	<2	3	August 1986
1408	41°45'16"	70°39'17"	<2	3	August 1986
1403	41°45'05"	70°39'44"	3	3	August 1986
1402	41°45'04"	70°39'41"	2	3	August 1986
1401	41°44'56"	70°39'39"	3	3	August 1986
1406	41°45'16"	70°39'17"	<2	3	August 1986
1601	41°44'51"	70°40'57"	*8	3	September 1986
1602	41°44'46"	70°40'46"	*<4	3	September 1986
1606	41°44'43"	70°39'58"	*4	3	September 1986
1603	41°44'39"	70°40'47"	*4	3	September 1986
1407	41°44'39"	70°40'00"	<2	3	August 1986
1605	41°44'39"	70°39'55"	*4	3	September 1986
1607	41°44'38"	70°40'00"	*<4	3	September 1986
1604	41°44'34"	70°40'28"	*4	3	September 1986
1609	41°44'32"	70°39'57"	*<4	3	September 1986
1608	41°44'30"	70°40'08"	*<4	3	September 1986
1610	41°44'27"	70°40'03"	*<4	3	September 1986
10OB	41°44'20"	70°39'26"	*<5	1	August 1985
11OB	41°44'10"	70°38'55"	*<5	1	August 1985
12OB	41°43'46"	70°38'34"	*<5	1	August 1985
1RB	41°45'48"	70°37'59"	26	1	May/August 1985
2BB	41°44'55"	70°37'15"	31	1	May/August 1985
3BB	41°45'59"	70°36'43"	10.3	1	May 1985
BB	41°45'59"	70°36'43"	*<5	1	May 1985
4BB	41°44'47"	70°37'18"	5.1	1	May/August 1985
5BB	41°44'19"	70°37'54"	*<5	1	May/August 1985

Table A-39. Fecal coliforms in Buzzards Bay -- Area III North.

Station	Latitude °N	Longitude °W	Mean Fecal Coliforms	Source	Sampling Date
1CC	41°44'17"	70°37'27"	*40	1	August 1985
2BR	41°43'36"	70°36'05"	105	1	August 1985
3BR	41°43'43"	70°36'50"	97	1	August 1985
4BR	41°43'19"	70°37'27"	*350	1	August 1985
5PH	41°43'03"	70°37'27"	*330	1	August 1985
7TI	41°42'50"	70°37'05"	3	1	August 1985
6PH	41°42'41"	70°38'02"	*300	1	August 1985
8PR	41°41'49"	70°35'20"	145	1	August 1985
9PR	41°41'45"	70°37'09"	130	1	August 1985
10PH	41°41'27"	70°37'32"	118	1	August 1985
13PP	41°41'08"	70°37'49"	7.1	1	August 1985
11POH	41°41'04"	70°38'17"	*220	1	August 1985
12POH	41°40'47"	70°38'44"	*600	1	August 1985
14POH	41°40'50"	70°37'27"	*480	1	August 1985
17RH	41°40'43"	70°36'21"	<5	1	August 1985
16RH	41°40'40"	70°36'43"	61	1	August 1985
15RB	41°40'30"	70°37'24"	*30	1	August 1985
19MH	41°40'31"	70°36'08"	51.8	1	August 1985
20MH	41°39'52"	70°36'30"	333	1	August 1985
21MH	41°39'34"	70°36'58"	328	1	August 1985
18HC	41°40'18"	70°38'14"	*<5	1	August 1985

Table A-10. Fecal coliforms in Buzzards Bay -- Area III South

Station	Latitude °N	Longitude °W	Mean Fecal Coliforms	Source	Sampling Date
22MH	41°39'27"	70°37'35"	6.1	1	August 1985
23MR	41°39'00"	70°37'30"	96	1	August 1985
24MR	41°39'04"	70°37'51"	193	1	August 1985
25MF	41°38'53"	70°38'11"	12.6	1	August 1985
27WH	41°38'02"	70°38'00"	290	1	August 1985
28WH	41°38'07"	70°38'53"	87.8	1	August 1985
29WH	41°38'12"	70°39'08"	*100	1	August 1985
30HB	41°37'26"	70°38'22"	24.5	1	August 1985
31WSH	41°36'34"	70°38'17"	207	1	August 1985
32WFH	41°35'50"	70°38'36"	10.8	1	August 1985
33WFH	41°36'15"	70°38'43"	12.3	1	August 1985
34WH	41°36'18"	70°39'11"	*<5	1	August 1985
35GSC	41°35'00"	70°38'35"	12.6	1	August 1985
36LSC	41°35'35"	70°38'30"	199	1	August 1985
37QH	41°32'24"	70°39'38"	*20	1	August 1985
38QH	41°32'20"	70°39'52"	*<5	1	August 1985

Table A-11. Fecal coliforms in Buzzards Bay -- Outer Bay

Station	Latitude °N	Longitude °W	Mean Fecal Coliforms	Source	Sampling Date
43SH	41°40'	70°44'	<20	1	August 1985
42WA	41°42'	70°42'	20	1	August 1985
45CC	41°43'49"	70°37'49"	<5	1	August 1985
46WH	41°38'06"	70°39'12"	<5	1	August 1985
44BU	41°40'	70°41'	<20	1	August 1985
47CL	41°35'23"	70°41'36"	<5	1	August 1985