The Sedimentary Environment of Massachusetts Bay: Physical, Chemical and Biological Characteristics

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THE SEDIMENTARY ENVIRONMENT OF MASSACHUSETTS BAY: PHYSICAL, CHEMICAL AND BIOLOGICAL CHARACTERISTICS

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INTRODUCTION

High contaminant loads in the sediments of Boston Harbor have been well documented scientifically and widely publicized in the media. This has focused public attention on improving water and sediment quality in the Harbor. To help to mitigate Boston Harbor pollution, the Massachusetts Water Resources Authority (MWRA) will improve sewage treatment and divert the existing Deer Island outfall 15 km into Massachusetts Bay (effluent will be discharged through a series of diffusers along the last 2 km of the outfall tunnel, shown in Figure 1). This diversion of the effluent is scheduled for 1995 and raises concerns about the overall future trends in Massachusetts Bay water and sediment quality and possible farfield effects of contaminants that may extend to Cape Cod Bay and the Stellwagen Basin/Bank. There is evidence, presented in this report and elsewhere, that the resuspension and transport of contaminated sediments from Boston Harbor into Massachusetts Bay have already impacted these areas (e.g., there are gradients of chemical and microbial contaminants).

Over the past 20 years, a variety of studies have been conducted by the local, State, and Federal governments, academic researchers, the MWRA, and several contract research organizations to define the status and trends in water and sediment quality, and gain a better understanding of the biogeochemical processes that affect the ecosystem. Data on inorganic and organic contaminants in sediments have been measured and mapped along with the benthic biology, sediment grain size, and various organism/sediment relationships. In addition, a seasonal pattern in sediment deposition/erosion has been identified in the region and many long-term depositional sites have been identified. However, a formal review and synthesis of these data has not been done. The major purpose of this review is to evaluate the completeness of existing baseline information so that a monitoring strategy can be established to determine the short- and long-term impacts of future sewage discharge into Massachusetts Bay. This monitoring program should provide the MWRA with information about improvement in currently impacted sites, and should allow early warning of potential adverse effects in Massachusetts Bay associated with the new discharge.

Massachusetts Bay is defined as being within the area enclosed by Cape Ann to the north and an eastern limit extending southeastward to Provincetown at the tip of Cape Cod (Figure 1). There are two prominent bathymetric features within the Bay: Stellwagen Bank, near its eastern boundary, and Stellwagen Basin, adjacent to and westward of the Bank. Currently, dumping of waste is limited to dredged material at the Massachusetts Bay Disposal Site (MBDS). This is an area defined as being within a 1-nmi radius centered at 42° 25.7′, 70° 34.0′ (EPA, 1989). Historically, industrial wastes were dumped ca. 1 nmi west of this site, although this practice was discontinued under the Ocean Dumping Ban Act of 1988 (ODBA). There are 13 municipal wastewater outfalls permitted to discharge into the Bay. Of the total wastewater released, the MWRA discharges about 85% (Castagna, 1987).

To sufficiently characterize Massachusetts Bay sediments, this review of historical data focuses on (1) the physical nature of the sedimentary environment, (2) the soft benthic biological fauna, and (3) the distribution of organic and metallic pollutants. This review will be useful in any future assessment of the contaminant loading into and overall health of the Massachusetts Bay sedimentary environment, and will also aid in the development of future sediment monitoring programs.

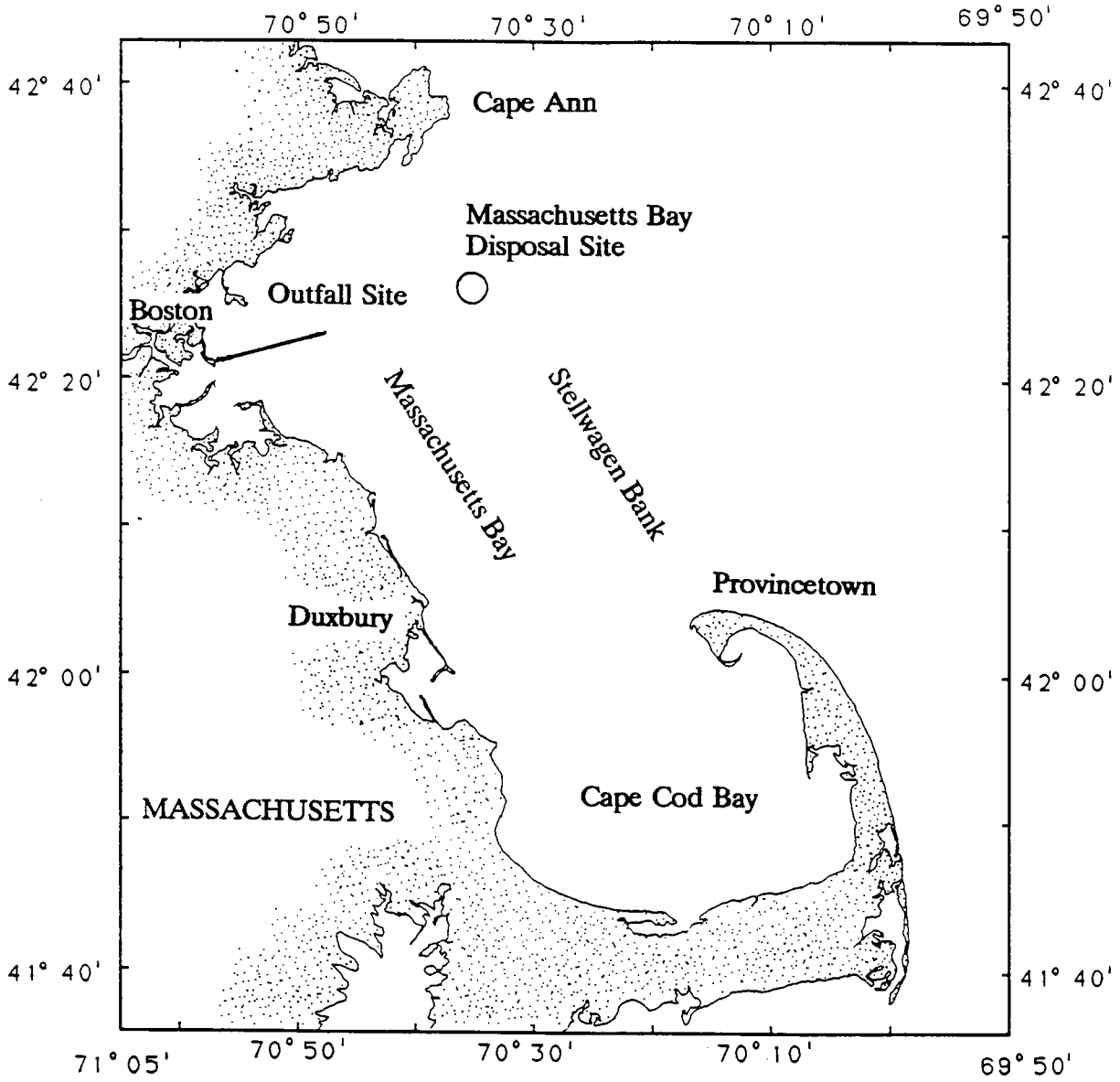

Figure 1. Massachusetts Bay.

1.0 SEDIMENTARY ENVIRONMENT

Between 1984 and 1987, three benthic surveys using the REMOTS® sediment profile imaging camera were conducted in Massachusetts Bay. A summary of these surveys is presented here as part of the critical review of benthic data for Massachusetts Bay. It is important to note that REMOTS data are used, as with any remote sensing study, to infer processes from imaged structures. To validate and calibrate remotely acquired data, one needs to establish "ground-truth" verification at a subset of the mapped stations. The REMOTS maps are graphic hypotheses about physical, chemical, and biological processes. The main utility of these maps is that they provide information about benthic gradients in time and space. The mapped patterns should be used to establish subset stations for ground-truth verification using more traditional sampling methods.

Survey 1 was carried out during July 1984 at 57 stations (Marine Surveys, 1984). This first survey was conducted as part of the 301(h) waiver application. Two additional surveys were undertaken as part of the secondary treatment facilities plan (STFP) to assist in the selection of candidate outfall sites. Survey 2 included 61 stations in late February and early March, 1987 (SAIC, 1987a). Survey 3 included 70 stations (35 reoccupied from survey 2) in late August, 1987. Three to five replicate images were taken at all stations sampled.

The locations of all stations occupied in the three REMOTS surveys are shown in Figure 2. Note that the 1984 summer survey was laid out on an orthogonal grid whereas stations of the 1987 winter and summer surveys were located to define the boundaries of sedimentological gradients. Figure 3 shows the locations of stations occupied in the summer surveys (1 and 3) and stations common to winter and summer surveys 2 and 3 ("both"). Figure 4 shows the locations of stations occupied in the winter (survey 2) as well as those stations common to winter and summer surveys 2 and 3 ("both"). Summer and winter survey data are separated to evaluate seasonal variation in benthic processes.

Navigation for all three surveys was provided by a the SAIC Integrated Navigation and Data Acquisition System (INDAS). Loran C was chosen as the positioning system. The positioning accuracy was determined by comparing Loran C theoretical coordinates with the geodetic position of several navigation aids provided by the United States Coast Guard (USCG) Light List. Loran C corrections were then calculated for the survey area using the SAIC INDAS software. These corrected values allow repositioning accuracy to within ± 20 m.

The methods for obtaining REMOTS sediment-profile images and computer image analysis of acquired images, as well as an explanation of the successional paradigm that allows recognition of successional seres from these photographs, may be found in the cited survey reports and an overview of this technique is given in Rhoads and Germano (1986).

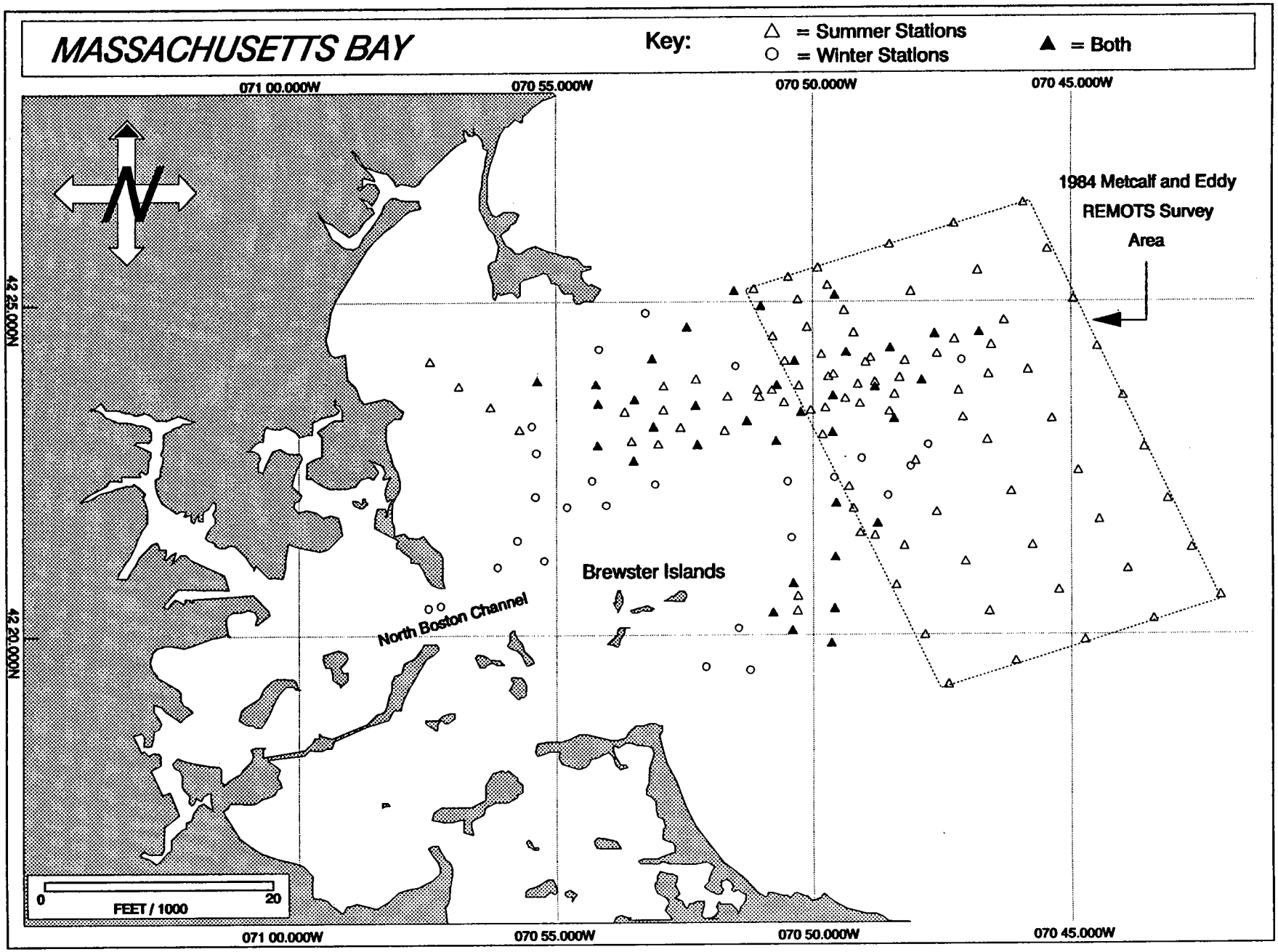

Figure 2. Location of all REMOTS® Stations (n=153) Occupied in the 1984 and 1987 Surveys.

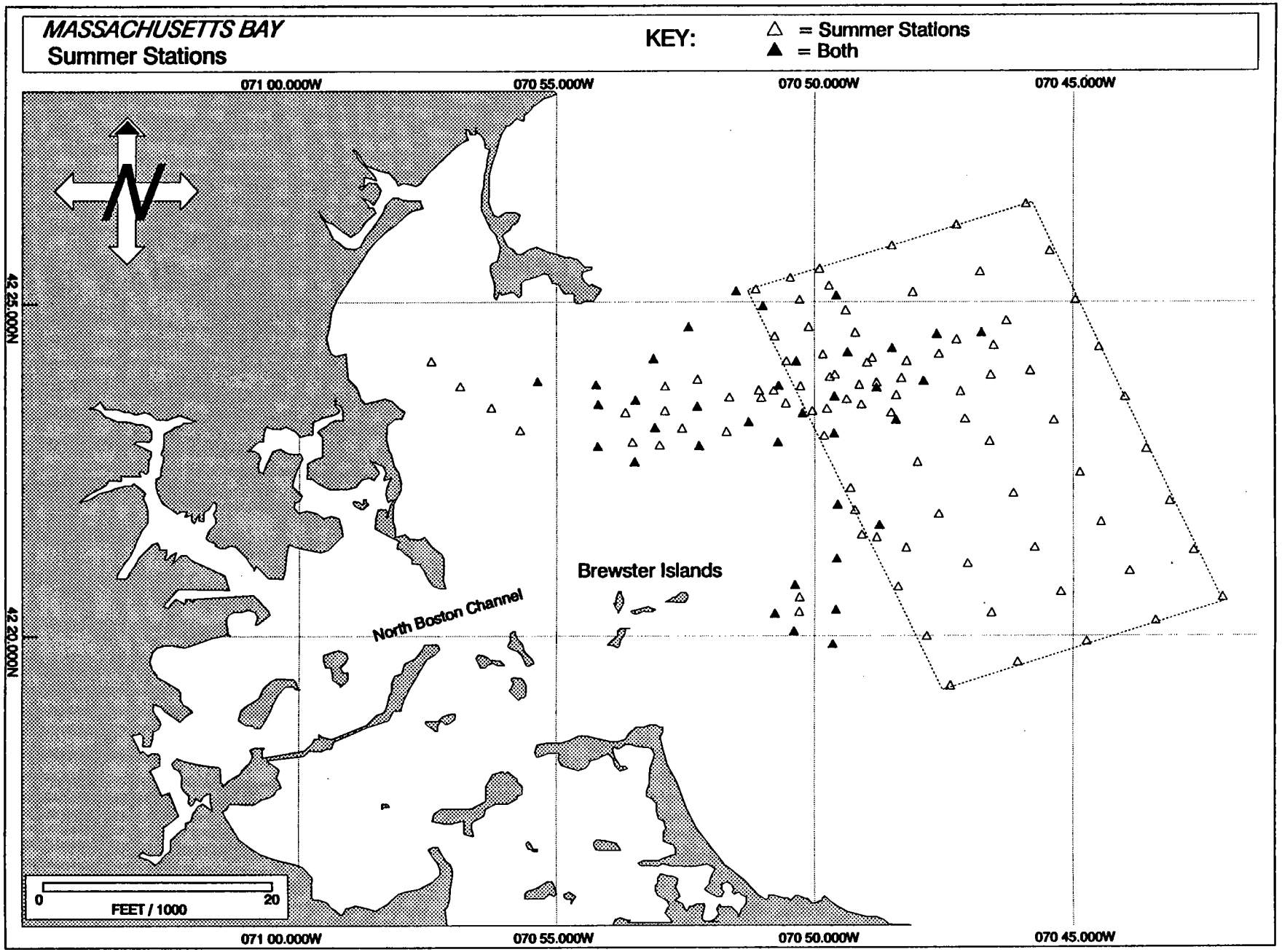

Figure 3. Location of Summer REMOTS® Stations Sampled in July, 1984 (Orthogonal Grid Pattern) and August, 1987 (Irregularly Spaced Stations). Stations that were also sampled in the winter of 1987 are shown as solid triangles ("both").

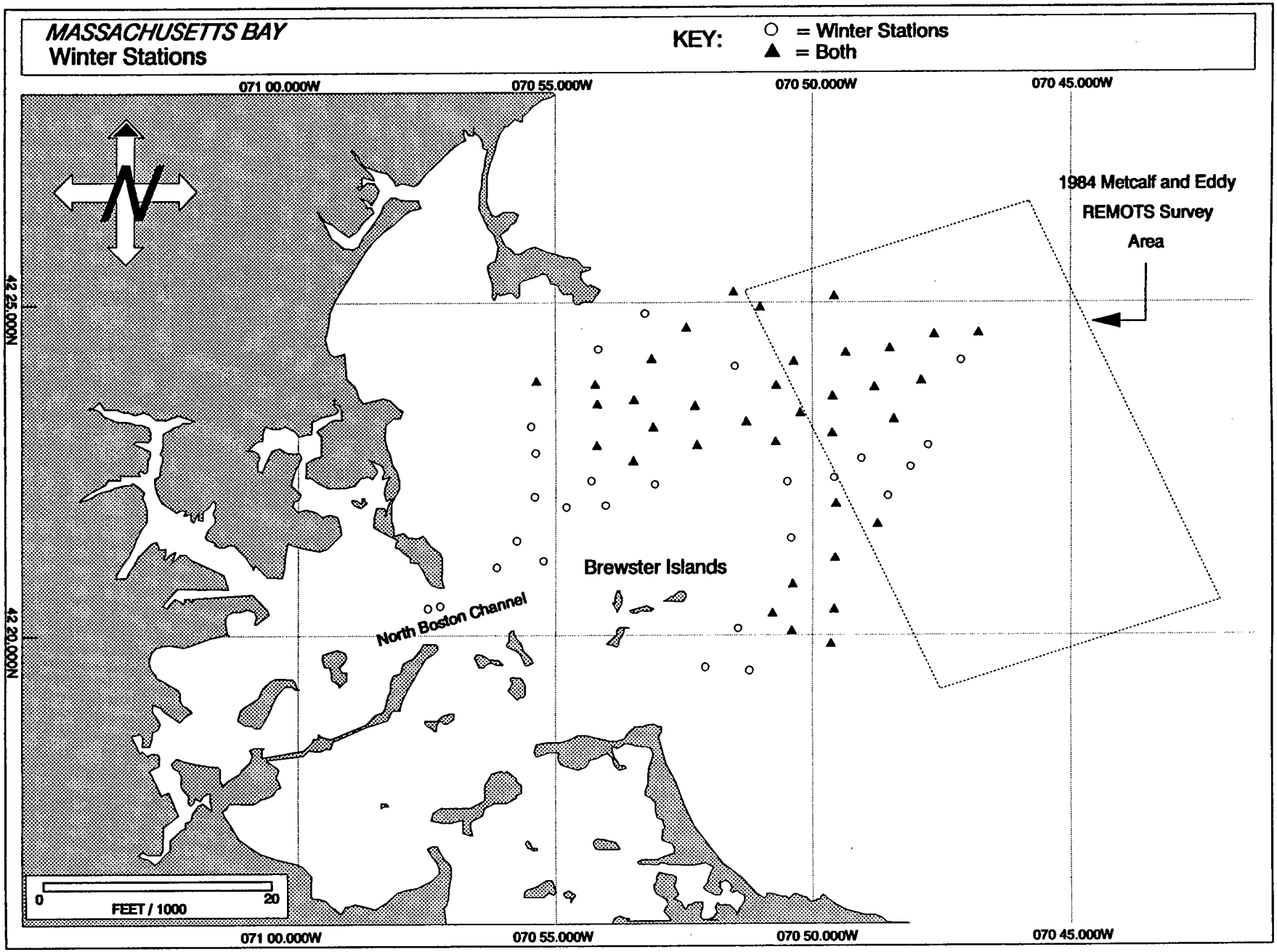

Figure 4. Location of Winter REMOTS® Stations Sampled in February and March of 1987. Stations that were also sampled in the summer of 1987 are shown as solid triangles ("both").

1.1 DISTRIBUTION OF MAJOR MODAL GRAIN SIZE

The distribution of the major modal grain size, as estimated from sediment-profile images, is shown in Figure 5. The overall pattern is a composite derived from the three separate surveys. Most of the area consists of drowned glacial moraines that are presently being reworked by wind-driven waves and tidal currents. The resulting bathymetry and grain-size patterns are therefore very complex in the surveyed area. The relationship between sediment distributions and bathymetry is shown in Figures 5 and 6. Most of the sediments consist of granule, gravel, and boulder-sized materials or very fine sand mixed with silt plus clay. In general, these sediments are very poorly sorted, consisting of admixtures of both fine and coarse materials. Another area of coarse-grained sediment is located south of Deer Island where high velocity currents scour Boston North Channel and President Roads.

In the late summer benthic biological activity (mainly tube-dwelling polychaetes and amphipods) trap and bind organic-rich silt and clay onto the otherwise sand/gravel bottom. This biologically deposited material is in the form of densely aggregated tube mats. Some of this material may also represent deposits of fecal pellets or muds mixed with fecal pellets. Several sediment-profile images indicate that this biologically bound biogenic mud often forms a thin veneer over sand, gravel, and boulder surfaces. This phenomenon was observed in both summer surveys (1 and 3), but was absent in the winter survey (2). The apparent seasonal change in the distribution of surficial "biogenic" mud leads to the hypothesis that, after the summer peak in benthic population density and metabolic activity (feeding and tube building), the tube mats break down and are washed away from these coarse-grained deposits. This inference is supported by the presence of an extensively rippled sand bottom that was mapped in the winter period (survey 2).

1.2 CANDIDATE DEPOSITIONAL AREAS

The potential receiving sites for the biogenic mud are bottoms in low kinetic energy where silt plus clay sediments accumulate on a year-round basis. Two silt plus clay areas were identified in surveys 2 and 3. These areas are identified as LKA (low kinetic area) in Figures 5 and 6. LKA No. 1 is located in a topographic depression (>25 m) southeast of Nahant. LKA No. 2 is at a similar depth in the center of the surveyed area (Figure 6). The designation of these two areas as low kinetic sites is based not only on the fine grain size of these deposits but also on the presence of biogenic topography imaged in the sediment-profile photographs. The surface of the bottom is dominated by mounds and depressions created by bioturbation. These structures are not preserved in areas of high kinetic energy where the bottom is resuspended by supercritical flow.

Results of a detailed bathymetric and side-scan-sonar survey, conducted by the USGS in the general area mapped by the REMOTS, confirm the existence of a silt or finer-grained deposit in the area of LKA 2 (Bothner *et al.*, 1990). This area has high (ca. 40 ft) relief produced by numerous drumlins and depressions. The axis of the silt/clay deposit trends in a southerly to southeasterly direction for a distance of about 15,000 ft and generally lies below a depth of over 95 ft. Several smaller areas of silty sediment were also identified in the survey, but LKA 2 appears to represent the largest area of fine-grained sediment. The USGS survey did not extend westward far enough to overlap LKA 1, and so comparable side-scan-sonar data are not available for the latter area.

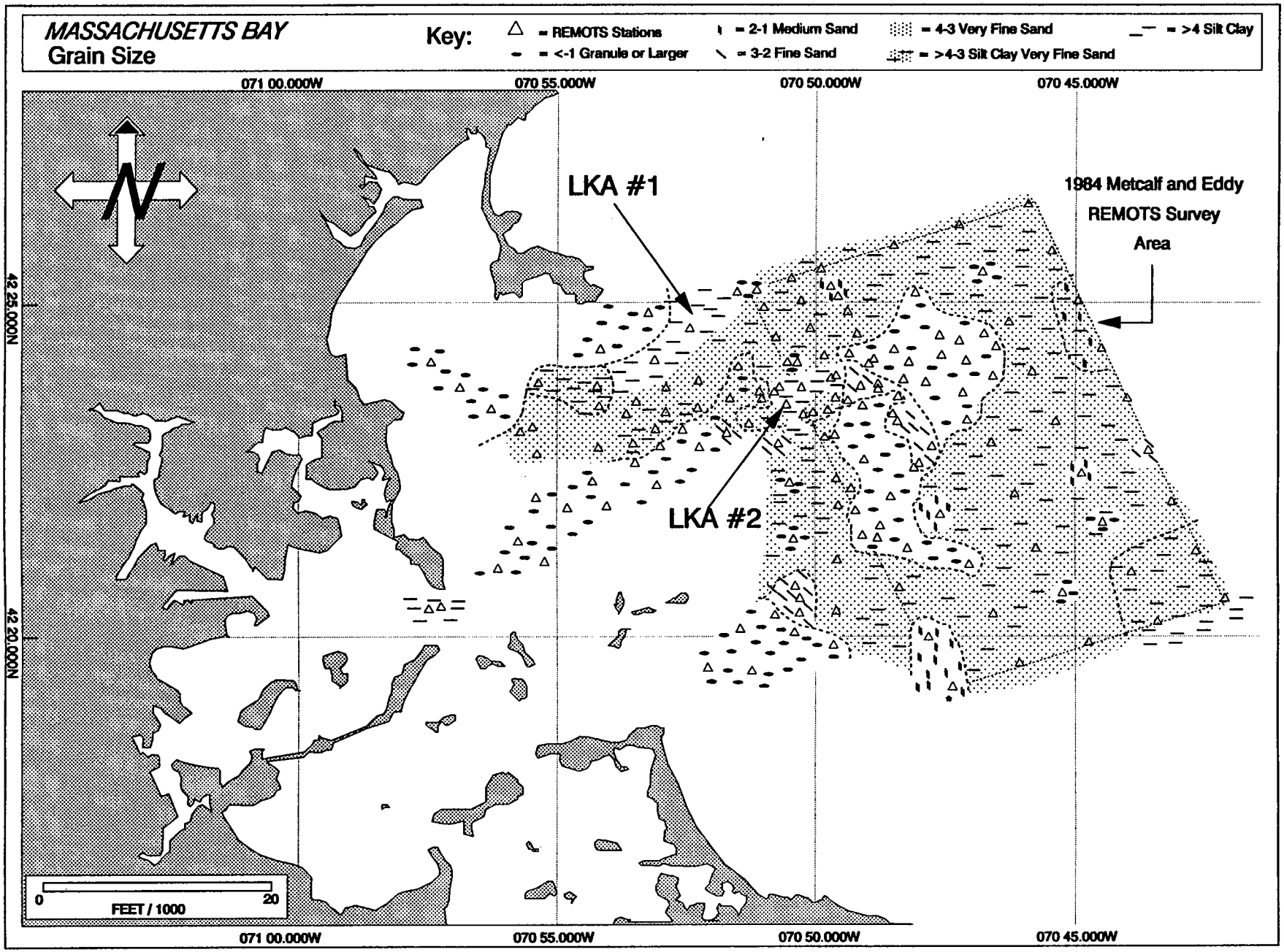

Figure 5. Distribution of Major Modal Grain-size (in Phi Classes) as Estimated from REMOTS® Sediment Profile Images. Two low kinetic energy areas are identified (LKA No. 1 and LKA No. 2) as potential long-term depositional sites for organic-rich sediment.

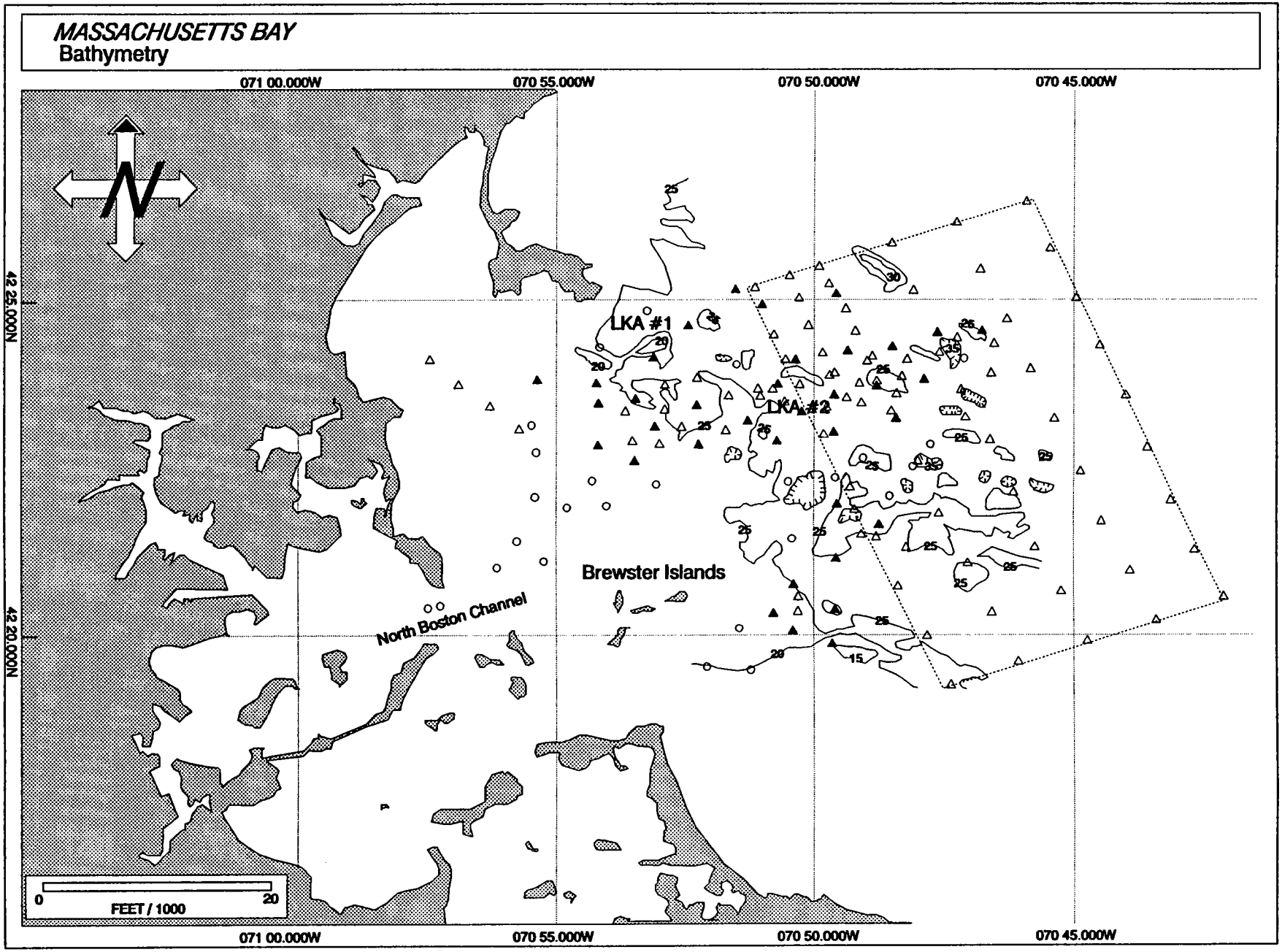

Figure 6. The Location of LKA No. 1 and LKA No. 2 with Respect to the 25 meter Isobath.

According to earlier work in Long Island Sound by Rhoads and Yingst (1978), low kinetic energy mud deposits can exist where the long-term mean flow velocity at 100 cm above the bed is less than 36 cm/s (ca. 0.7 kn). While velocities of 16 to 28 cm/s (z = 100 cm) can resuspend bioturbated muds, this surficial erosion involves only 1-2 mm of the surface sediment, and biogenic topography remains undisturbed. Massive sediment transport requires much higher velocities.

These are not the only areas in the region that are depositional areas for fine-grained sediment. Farther offshore, in the region of the Massachusetts Bay Disposal Site (MBDS), there are extensive areas of bioturbated silt and clay (SAIC, 1986; Trowbridge and Shepard, 1932; see Appendix A). To the southeast, there is a large area of silt/clay in the center of Cape Cod Bay (Young and Rhoads, 1971; see Appendix A). Extensive areas of silt and clay are also found shoreward of the survey area. However, because of the proximity of LKA No. 1 and LKA No. 2 to the proposed outfall site, these two areas are the most likely candidates for long-term deposition of organic-rich and fine-grained sediments.

1.3 BIOLOGICAL MIXING DEPTHS

The depth in the sediment affected by bioturbation can be inferred from changes in optical reflectance in sediment-profile images. Surface sediments that are intensively mixed by infaunal organisms have relatively high optical reflectance because of the presence of ferric hydroxide coatings on sediment particles. These oxidized iron phases result from the exposure of the grains to dissolved oxygen (DO) in the overlying water. In the absence of bioturbation, DO will penetrate only a few millimeters into the bottom. The actual penetration depth depends on the molecular diffusion coefficient, tortuosity, and the reducing ability of the sediment. Bioturbation serves to increase the rate of supply of DO into the bottom, and such "biodiffusion" may be orders of magnitude greater than molecular diffusion (Matisoff, 1982).

The transition from high-reflectance surface sediment (oxidized grain coatings) to low-reflectance sediment at depth (sulfitic sediments) has been termed the apparent redox potential discontinuity (RPD) depth. The RPD depths measured during the three surveys is mapped in Figures 7 and 8. The apparent RPD depth is almost always deeper than the true RPD depth as measured with an Eh electrode. This is because the ferric hydroxide coatings remain oxidized for a period of time after they are mixed downward (by organisms) into pore waters that are devoid of free molecular oxygen. This metastable condition of solid phase iron is therefore a conservative first-order indicator of particle mixing depths. Sediments that have high oxygen demand and/or low-order benthic successional stages tend to have mean apparent RPD depths of less than 3.0 cm. Alternatively, if a bottom is experiencing erosion, the thickness of the oxidized surface layer is reduced by the physical removal of this interval. This latter mechanism is usually associated either with ripples or sand waves on the bottom or with eroded mud clasts in fine-grained sediments.

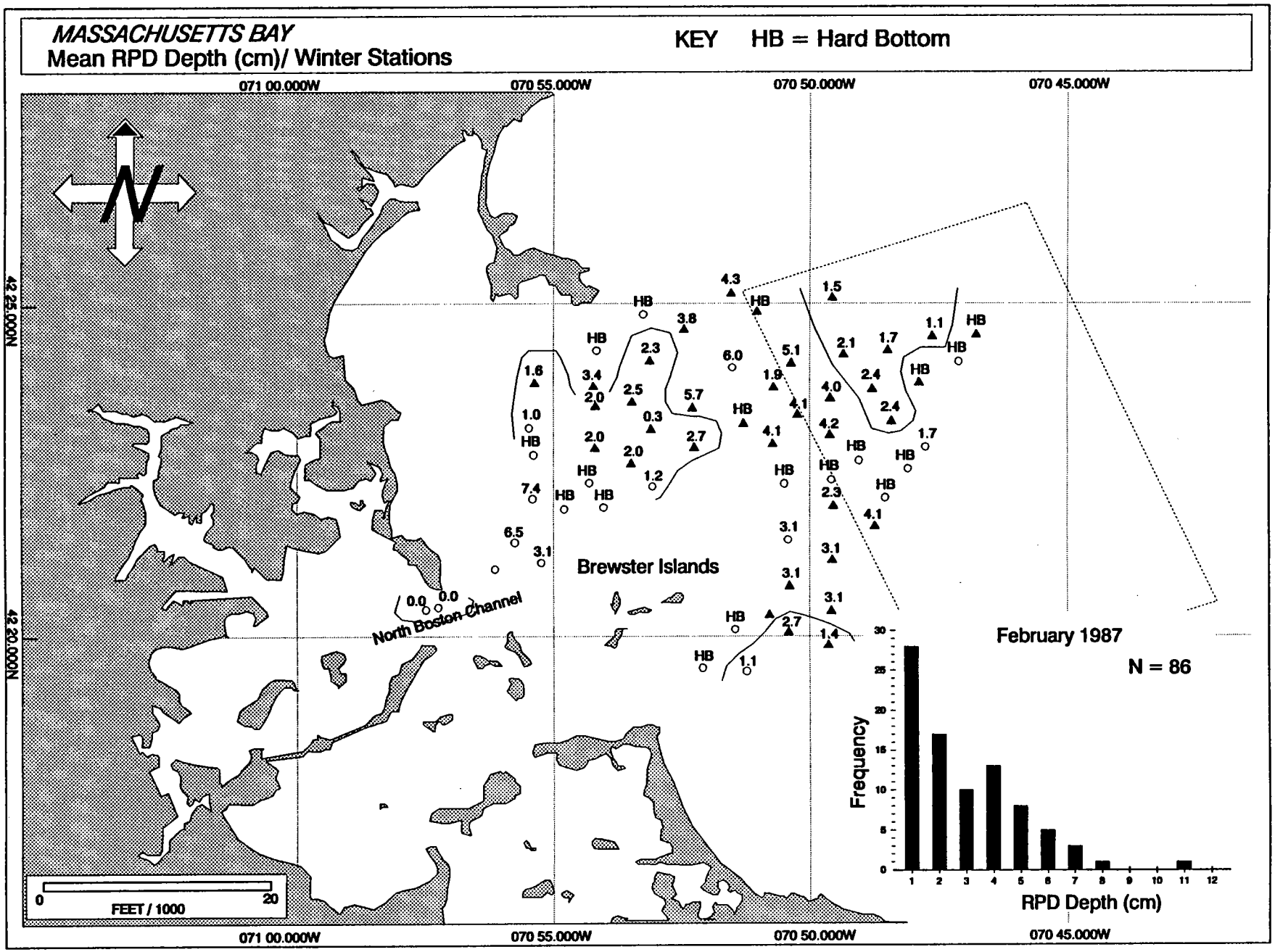

Figure 7. Mean apparent RPD depths (cm) in the Winter Survey. Areas less than 3.0 cm deep are delimited by contour boundaries. The frequency distribution for all station replicates is shown as an inset.

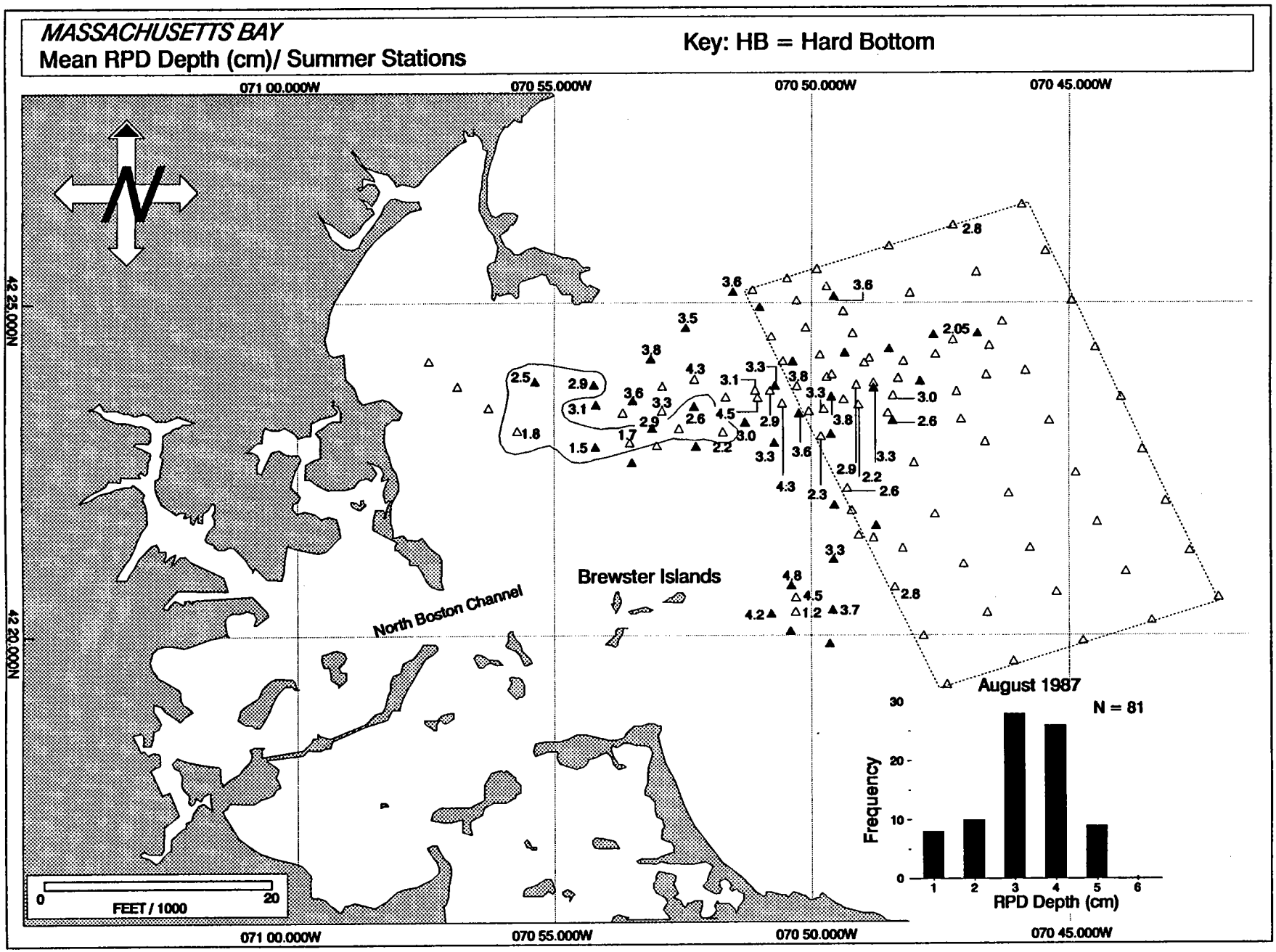

Figure 8. Mean Apparent RPD Depths (cm) in the Summer Surveys. Areas less than 3.0 cm deep are delimited by a contour line. The frequency distribution for all station replicates is shown as an inset.

The mean apparent RPD depths as mapped for the winter 1987 survey 2 are shown in Figure 7. There are three areas where values are less than 3 cm: (1) an area near the northern approach to the Boston North Channel, (2) an area immediately east of the Brewster Islands, and (3) an area located near the northeastern limits of the survey area. The overall frequency distribution for mixing depths during the winter (all replicates, $n = 86$) shows a major mode falling within the less than 1.0-cm-deep class (inset, Figure 7).

The apparent RPD depths as mapped for the summer periods (1984 and 1987 surveys 1 and 3) are shown in Figure 8. Stations located near the northeast end of Boston North Channel have mixing depths of less than 3 cm. The mean mixing depth frequency distribution shows the major mode to fall within the 3-cm-depth class (inset, Figure 8). A comparison of the apparent RPD depth distributions between stations sampled in both summer and winter reveals that the summer depths are slightly, but not significantly greater, than those measured in winter ($p = 0.07$, Mann-Whitney U-test).

1.4 BENTHOS

Sediment-profile images provide information on the vertical distribution of benthos. Based on a successional paradigm, imaged organisms (and the sedimentological structures that they produce) inferences can be made about the disturbance history of the seafloor and benthic habitat quality (Rhoads and Germano, 1982). In the following summary, Stage I seres are represented by dense aggregations of polychaetes at the sediment/water interface. Stage III seres are represented by headdown deposit feeders. Stage II seres (not an important infaunal component in these surveys) are tubicolous amphipods. Stage I seres tend to dominate habitats that are organically enriched or have recently experienced physical disturbance. Stage III seres are typical of more oligotrophic benthic environments where the substratum is stable over long periods of time (Rhoads and Germano, 1986).

Figure 9 shows the winter distribution of seres as interpreted from the REMOTS images. Because of the coarse nature of the sediments covering most of the area, the camera prism did not penetrate deep enough at many stations to assess either the presence or absence of Stage III infauna. Stations designated as having epifauna on rocks (ER) also may have been populated by Stage I or III infauna, but again limited camera penetration did not allow assessment of the infauna.

Despite these limitations, Stage III seres were found throughout the surveyed area. Three stations at the northeast edge of North Boston Channel apparently are dominated by Stage I seres. These communities may account for the shallow mixing depths measured at these same stations (Figure 7).

Summer successional distributions are given in Figure 10. Again, Stage III infauna were widespread over the surveyed area. An area apparently dominated by Stage I seres was located in the northwest quadrant of the July 1984 grid.

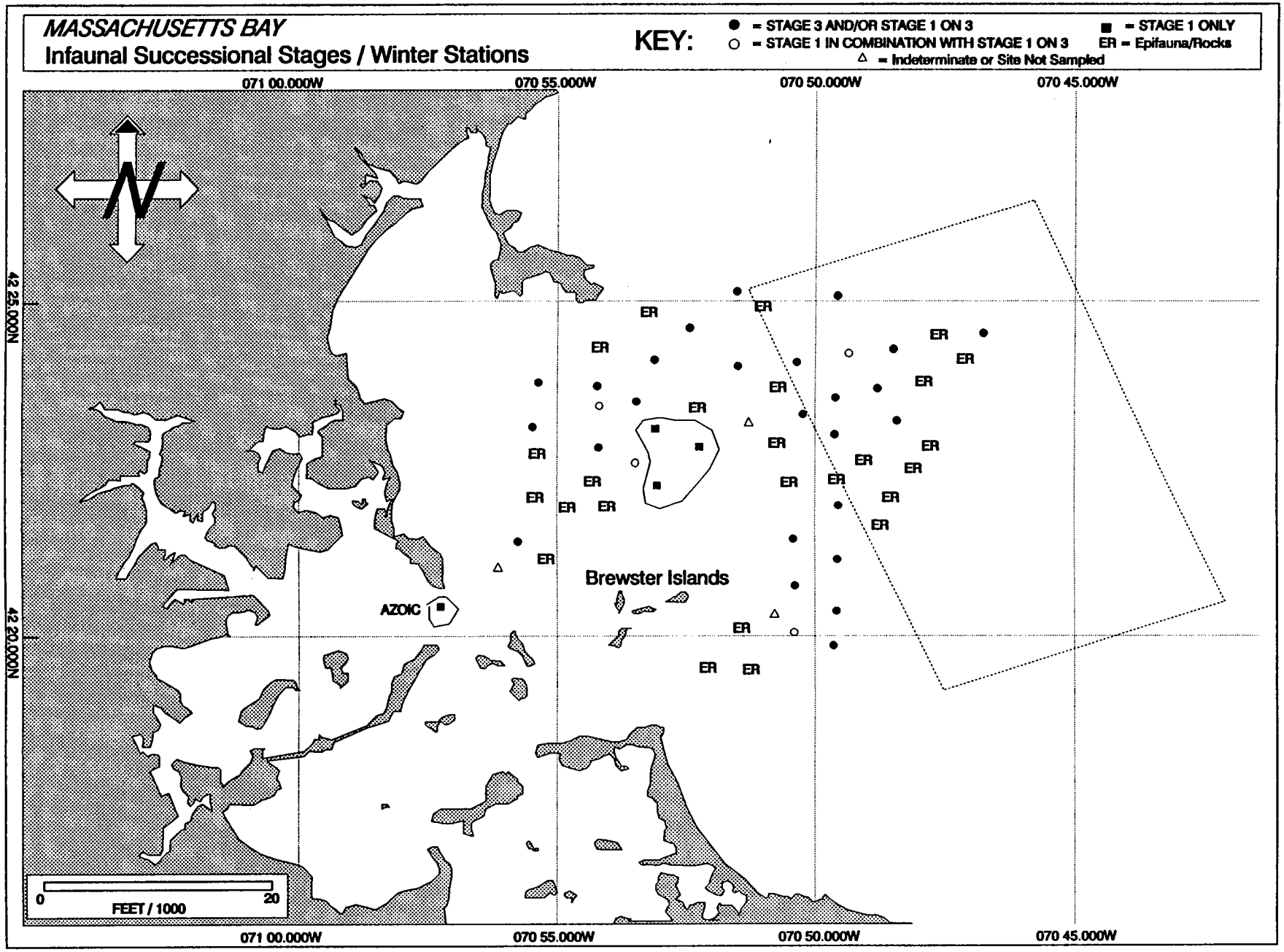

Figure 9. Infaunal Successional Stages as Mapped in the Winter Survey. Stations where only Stage I assemblages were imaged (or stations apparently devoid of macrofauna ("Azoic") are delimited by a line.

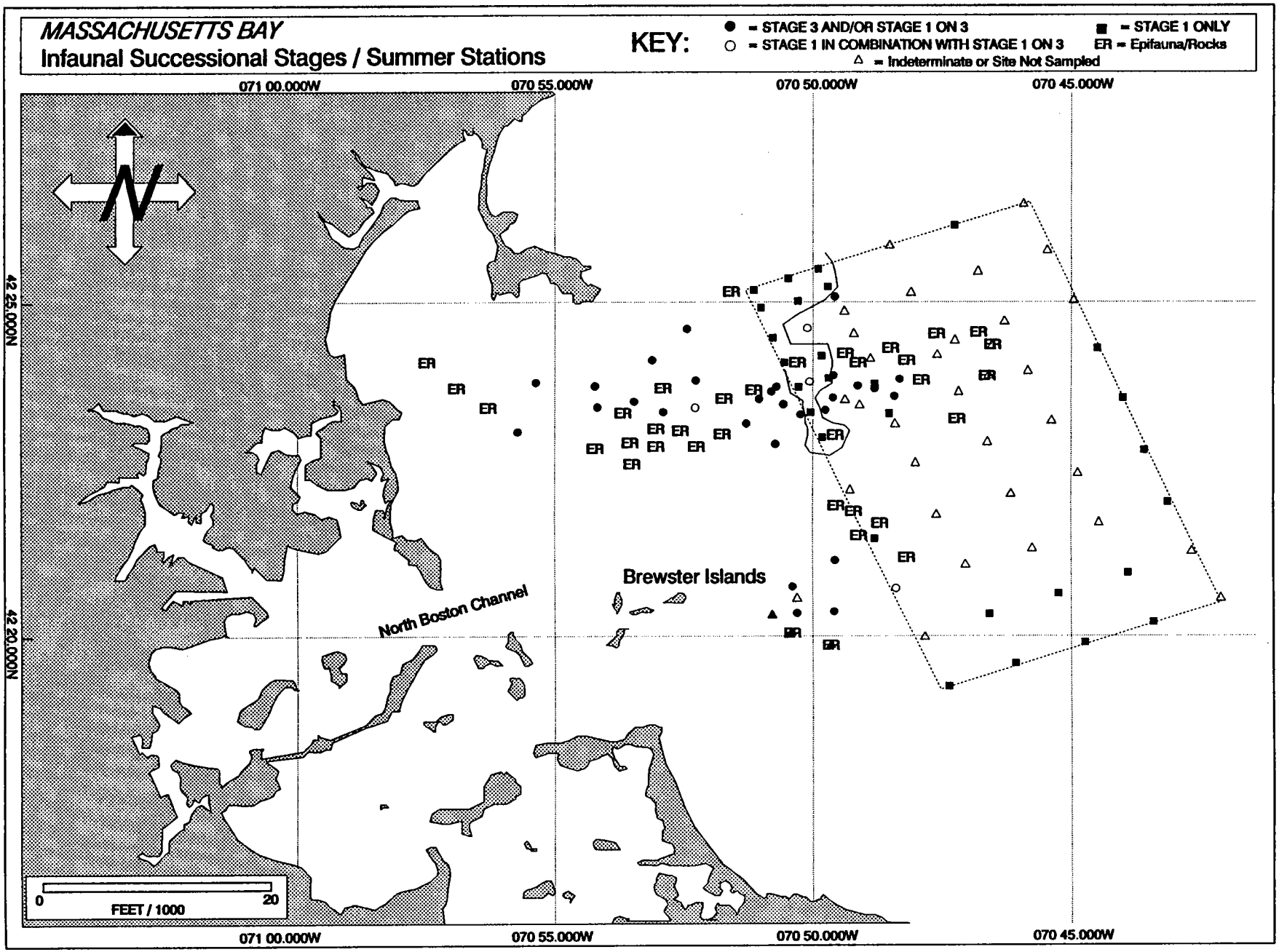

Figure 10. Infaunal Successional Stages as Mapped in the Summer Surveys. Stations where only Stage I assemblages were imaged are delimited by a line.

1.5 ORGANISM/SEDIMENT INDEX

From the parameters available from analysis of REMOTS images, an organism/sediment index (OSI) is calculated [see Rhoads and Germano (1986) for how this index is calculated]. The potential range of this index is +11 (a high-quality benthic habitat dominated by Stage III infauna and a deep bioturbation zone) to −10 (an azoic methanogenic bottom). Generally, benthic habitats of high quality have OSI values over +6.

The distribution of summer OSI values is shown in Figure 11 (surveys 1 and 3). Most stations have OSI values indicative of high habitat quality (OSI > +9) as reflected in deep bioturbation depths (a deep apparent RPD), and the presence of well developed Stage III infaunal assemblages. Only three stations fall marginally below the +6 OSI threshold. These three stations lie within the upper northern third of the 1984 sampling grid. The reason for these marginal values is unknown but may be related to physical disturbance of the bottom.

The distribution of winter OSI values (survey 2) is shown in Figure 12. The overall frequency distribution of OSI values shows the surveyed area to represent high habitat quality with the exception of five stations. Two of these stations consist of sulphidic muds located near the Deer Island mud flats and represent a bottom that is receiving high rates of sewage loading. This loading is apparent from the presence of methane gas bubbles in the sediment column, no apparent RPD, and an azoic or Stage I infaunal condition. Three stations south of Nahant in Broad Sound yield mean OSI values < +6. Subsequent study of this area in September and October 1990 showed that the bottom apparently is affected by sedimentation from the Deer Island outfall (SAIC, 1990). One station in Broad Sound had a sediment oxygen demand and ammonia flux comparable to Boston Harbor sediments.

The low kinetic areas (LKAs) have high OSI values in both summer and winter surveys. These fine-grained sediments have deep apparent RPD depths and contain Stage III infauna. If these LKA deposits experience a significant increase in organic loading in the future, such a change should be readily detected as a decline in OSI values. The early stages of organic over-enrichment is typically accompanied by a rebound in the RPD as sediment oxygen demand increases.

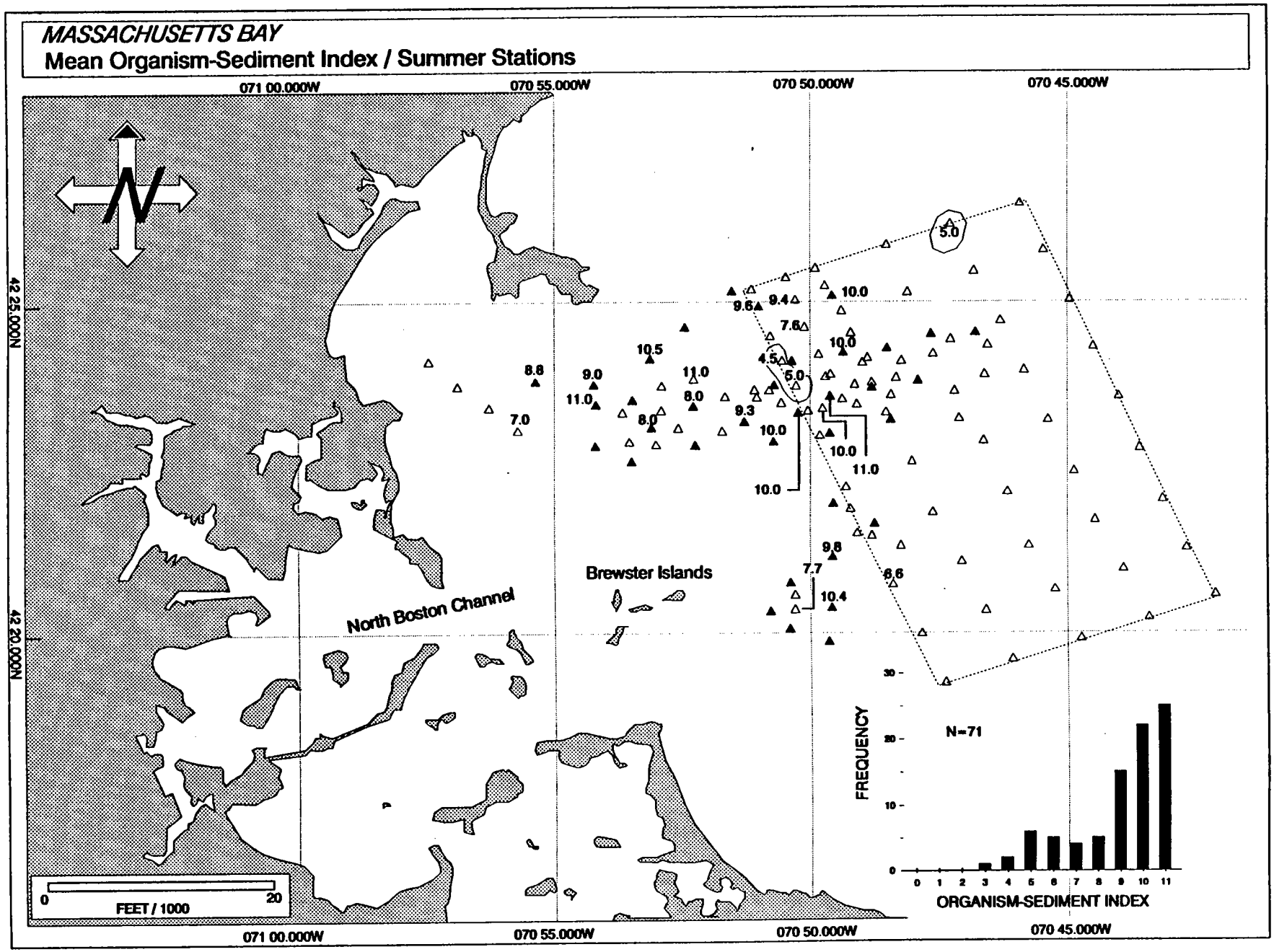

Figure 11. Mean Station Organism-Sediment Indices (OSI's) Calculated for the Summer Surveys. Areas where the mean OSI is equal to, or less than OSI = +6 are identified by a contour line. The frequency distribution for all station replicates is shown in the inset.

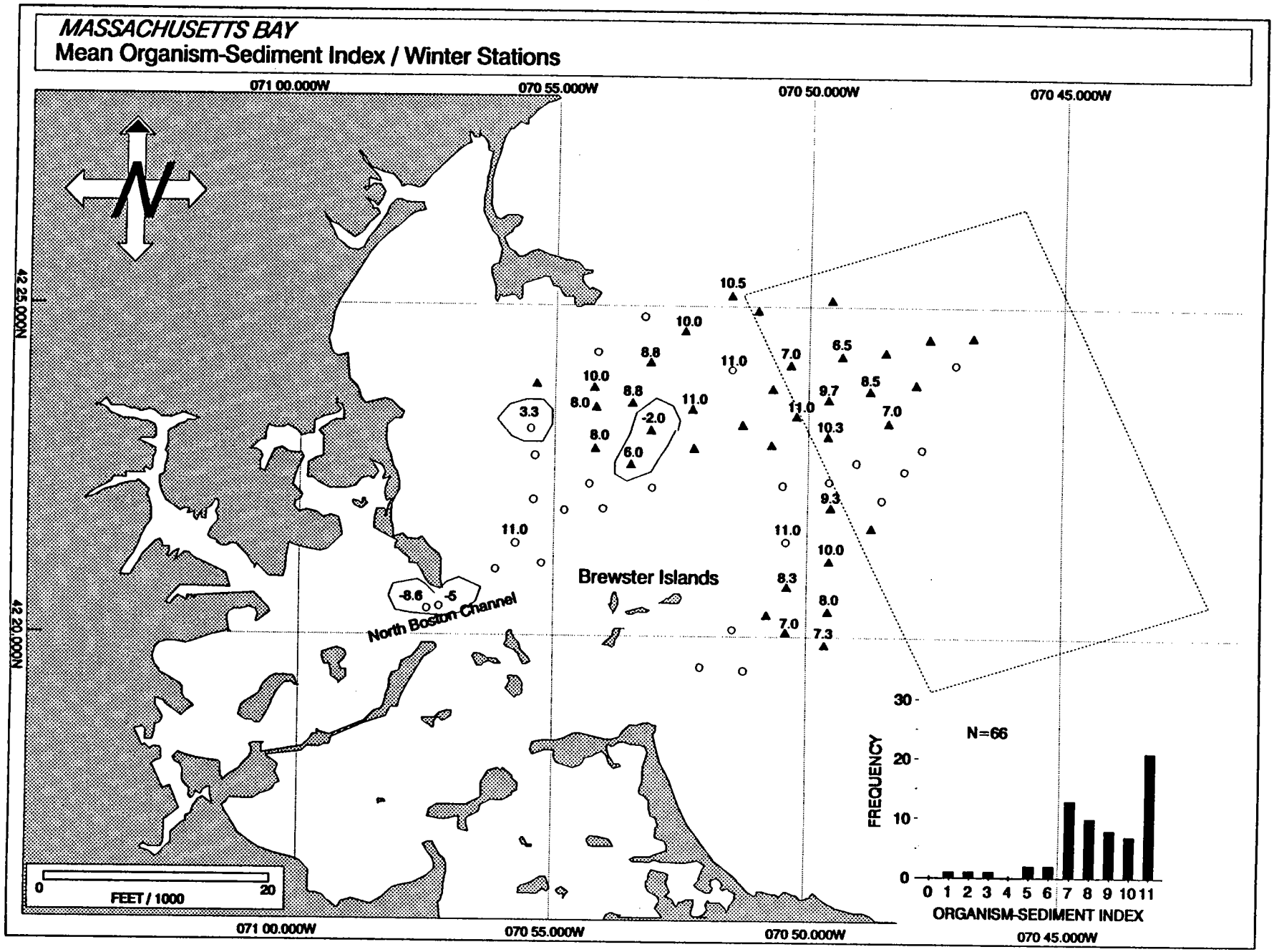

Figure 12. Mean Station Organism-Sediment Indices (OSI's) Calculated for the Winter Survey. Areas where the mean OSI is equal to, or less than, OSI = +6 are identified by a contour line. The frequency distribution for all station replicates is shown in the inset.

2.0 BIOLOGICAL ENVIRONMENT

2.1 REVIEW OF BENTHIC BIOLOGICAL STUDIES IN THE MASSACHUSETTS BAY-CAPE COD BAY ECOSYSTEM

The benthic communities of Massachusetts Bay and Cape Cod Bay are known from a series of studies performed between 1976 and 1988. To date, the results of these studies reside in unpublished reports to Federal, State, and municipal agencies. The location of the stations sampled in Massachusetts Bay and Cape Cod Bay is depicted in Figures 13 and 14.

The earliest study of infaunal benthos from the region was by Gilbert *et al.* (1976), in which 37 stations were established from Cape Ann to Cape Cod. Two replicate 0.1-m^2 Smith-McIntyre grabs were taken at each of these stations for analysis of benthic infauna. The samples were sieved on a 0.5-mm-mesh sieve. A new summary of the dominant species and densities of infauna from this database has been prepared; it is presented in Appendix B. The main results of the Gilbert *et al.* (1976) survey indicate that infaunal density, species diversity, and species richness are high. The numbers of individuals per square meter ranged from about 4000 to more than 60,000. The highest densities occurred between Boston Harbor and Cape Ann to about 20 km offshore and in an area adjacent to Cape Cod Bay (Figure 15). Relatively low densities were found in the northernmost offshore area, at Station C-3, and at some additional stations scattered throughout the study area.

Species richness varied between 40 and 125 species per station. The highest numbers of species were found at some coastal and offshore stations in an area roughly parallel to the northern coastline. Species richness was lowest in two neighboring offshore areas in the middle of the outer Massachusetts Bay. Almost the entire study area was dominated by the spionid polychaete *Spio limicola*. This species comprised 18%-80% of the total fauna. High percentages of *S. limicola* were found in the northwestern part of the Bay, whereas the lowest numbers occurred among the offshore stations in the northern Bay (Figure 16).

Five coastal stations (C-1 through C-5 in Figure 14) were dominated by different species. The most remarkable stations were C-2 with 71% of the fauna consisting of an unidentified cirratulid polychaete, and C-5 that was dominated by *Ampharete arctica,* a species that was rare or absent at the other stations. The only offshore stations not dominated by *S. limicola* were stations 2 and 3, off Cape Ann, and station 12, in the central Bay. Other very abundant and widespread species included the polychaetes *Aricidea catherinae, Prionospio steenstrupi, Mediomastus californiensis, Chaetozone setosa,* and *Myriochele ?heeri* (Appendix B).

During the 301(h) waiver application process, benthic stations were established in Boston Harbor and in Massachusetts Bay. Three Massachusetts Bay stations were sampled in 1978 (DWI, DWII, and DWIII), one in 1979 (DWI), and one in 1982 (PD). The results of these studies were recently reviewed by Blake and Maciolek (manuscript in preparation). Following denial of the application, additional work including the establishment of a grid of stations near a proposed offshore outfall site was conducted by Metcalf & Eddy (1984). Five of these stations were sampled for benthic infauna (Stations 32, 38, 40, 42, and 53). All of the 301(h) stations are shown in Figure 13 (station numbers or letters not shown).

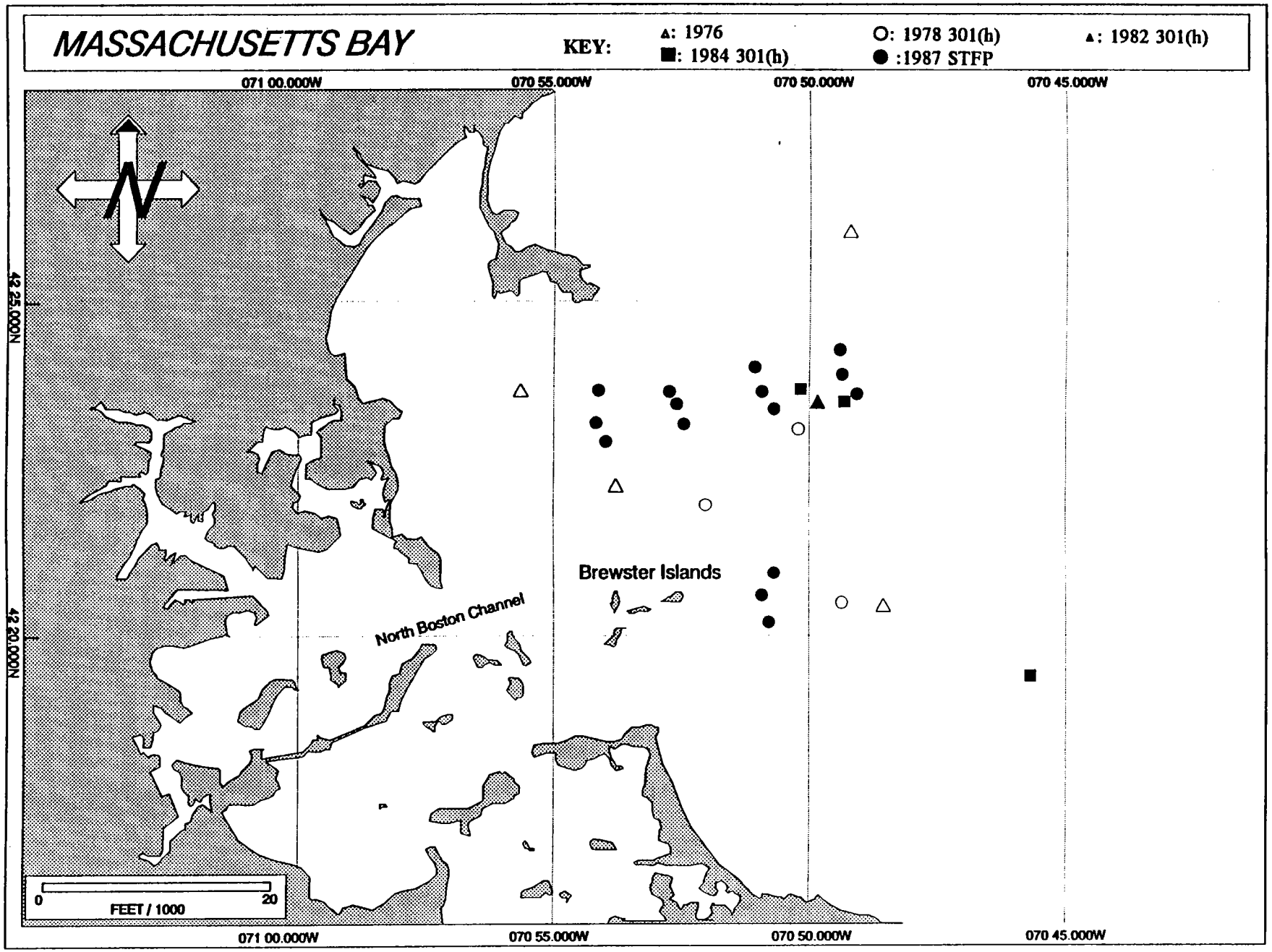

Figure 13. Map of Benthic Station Occupied in Massachusetts Bay as part of the 301(h) Waiver Application Process, Secondary Treatment Facilities Plan, and in the Massachusetts Bay Survey by Gilbert et al. (1976).

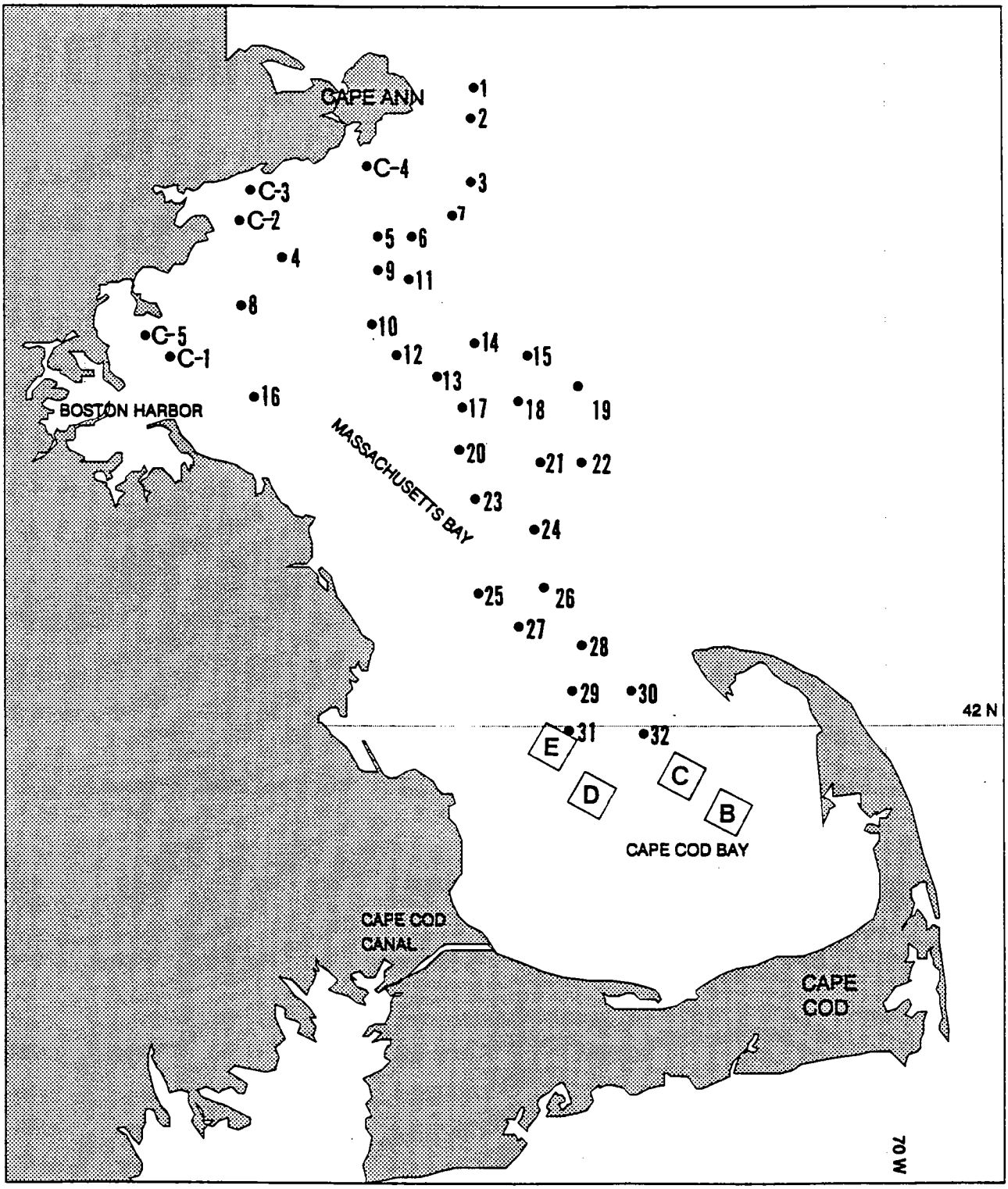

Figure 14. Distribution of Benthic Stations Occupied during the Massachusetts Bay Survey by Gilbert *et al.*, 1976 (numbers) and the Cape Cod Bay Environmental Impact Report by Battelle, 1987b (letters inside boxes).

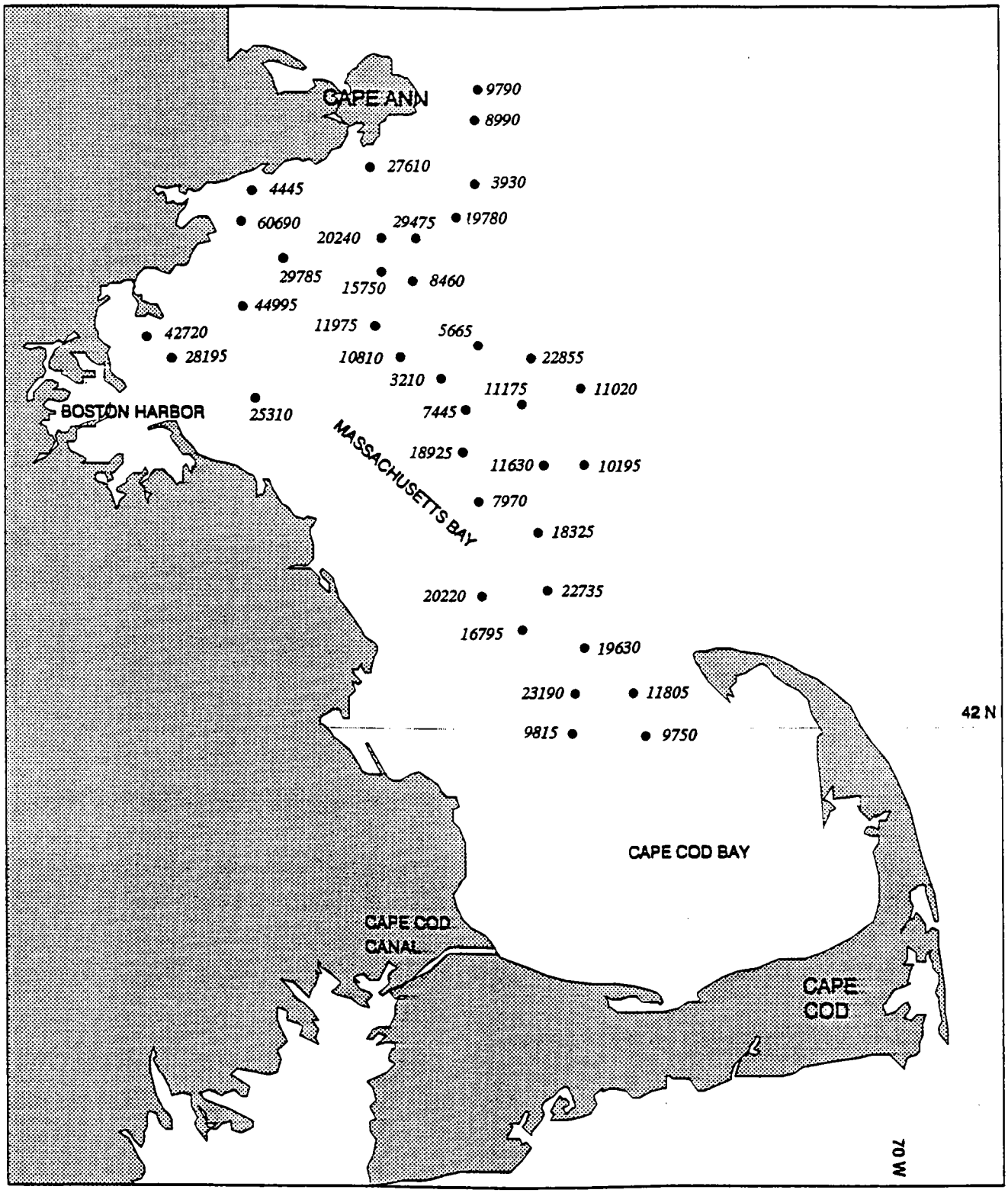

Figure 15. **Map of the Massachusetts Bay–Cape Cod Bay System Showing Total Densities (Number of Individuals per m^2) of Benthic Infauna. (after Gilbert et al., 1976.)**

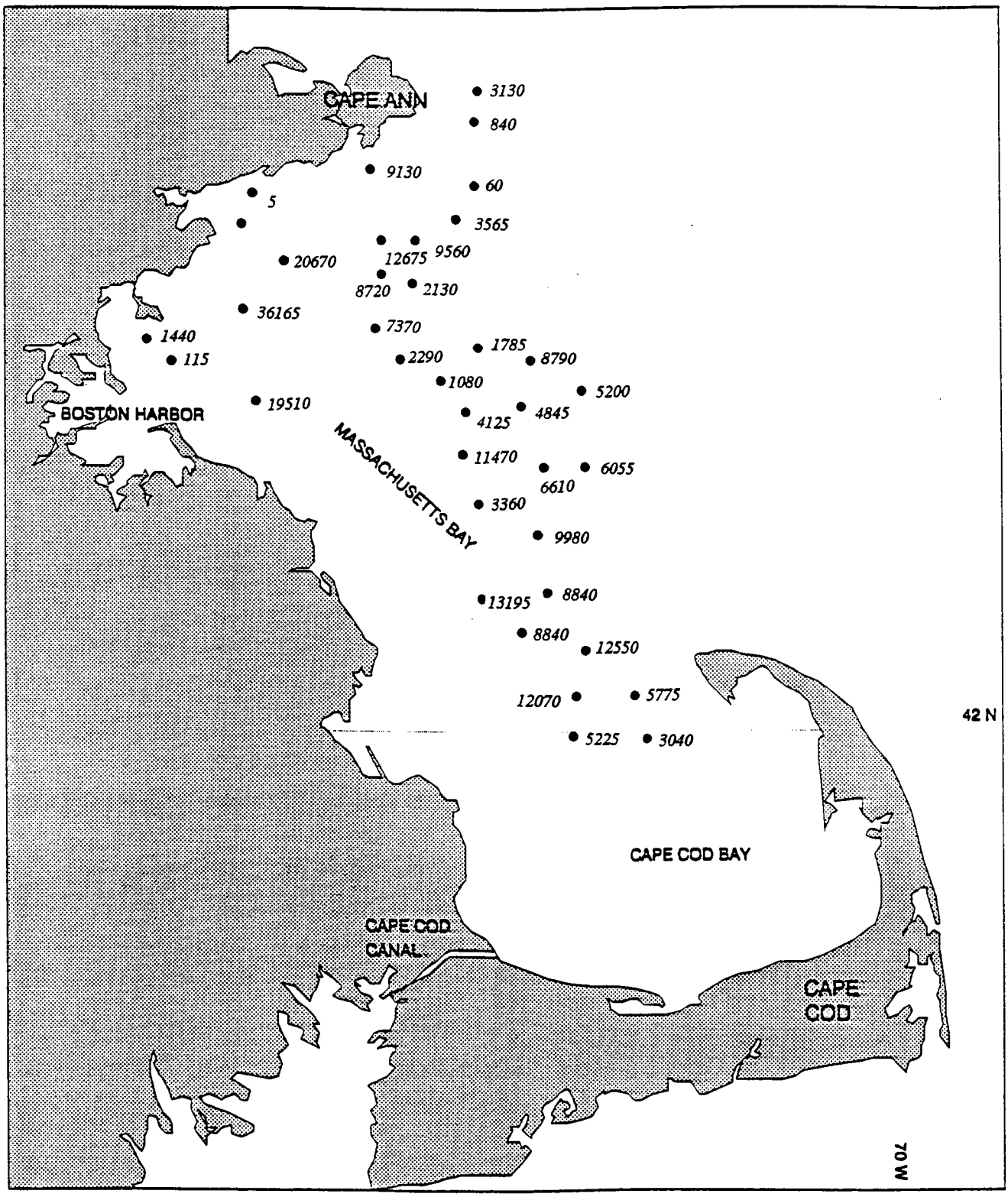

Figure 16. Map of the Massachusetts Bay–Cape Cod Bay System Showing Distribution and Total Density (Number of Individuals per m^2) of Spio limicola a Common Spionid Polychaete. (after Gilbert et al., 1976, as Spio filicornis.)

The results from Massachusetts Bay suggest considerable year-to-year variability. For example, in 1978, 32 species of benthic invertebrates were found at three stations; in 1979, 81 species were found at one station; and in 1982, there were 112 species at one station (Blake and Maciolek, manuscript in preparation). Of particular interest are the results from Station DWI, sampled in 1978 and in 1979, and Station PD sampled in 1982. At DWI, total densities were low in 1978 (5950/m^2), but increased dramatically in 1979 (23,230/m^2) with high densities of *Mediomastus ambiseta*. Four species were shared among the 10 most abundant species in 1978 and 1979, with *Polydora socialis* the highest-ranked species in 1978 and *Mediomastus ambiseta* the highest-ranked in 1979. The densities recorded from Station PD in Massachusetts Bay were the highest ever recorded in Massachusetts Bay, with a total of 100,140 individuals per square meter. The site was totally dominated by two spionids, *Spio limicola* and *Polydora quadrilobata,* that together accounted for 76% of all individuals. It is likely that the high densities of *Spio limicola* resulted from rapid population buildup and formation of dense tube mats. Similar populations of spionids forming tube mats have been recorded in the genus *Polydora* (Blake, 1971).

Following denial of the 301(h) waiver, additional investigations of benthic communities were required as part of the development of an STFP. These studies included the first temporal sampling plan in which infaunal benthos was evaluated in spring, summer, and winter. These studies were also the first in Massachusetts Bay to incorporate the 0.3-mm-mesh sieve in addition to the 0.5-mm sieve (Blake *et al.*, 1987, 1988). In addition to the infauna, the STFP studies also included hard bottom transects and permanent photographic quadrate sites that were evaluated seasonally for changes in epifauna (Sebens *et al.*, 1987). Broader-scale photographic surveys using a remotely operated vehicle (ROV) were also conducted (Etter *et al.*, 1987).

The infaunal studies conducted as part of the STFP were planned with the goal of evaluating spatial and temporal patterns in the benthic communities at seven potential outfall sites in Massachusetts Bay. Infaunal analyses were planned for three stations established on each of the seven transects. Preliminary studies conducted with an ROV and a sediment profiling camera prior to the selection of the seven transect sites indicated that the seabed in Massachusetts Bay was heterogeneous, both in terms of sedimentary and biological characteristics (Battelle, 1987a; SAIC, 1987a,b; this study).

The first survey of the seven transects was conducted in March 1987. During that survey, grab samples were collected successfully at only five transects. Two transects proved to have bottoms composed largely of rocks and cobbles interspersed with varying amounts of finer sediment, and it was not possible to take grab samples. A special study was subsequently initiated to characterize the two hard-bottom transects (Sebens *et al.*, 1987).

The infaunal program was established on the first cruise in March and included the collection of six replicate 0.05-m^2 grab samples from each of three stations on the five soft-bottom transects. These stations were resampled in May and August 1987. A subset of three of the six replicates was processed to assess infaunal community structure from all 15 stations on the first cruise. Three additional replicates were subsequently processed at five of the stations, one per transect, to assess the efficiency of replication. From these analyses, it was determined that six replicates were necessary to evaluate the temporal differences in benthic communities. Based on this information, one station (A2, B2, C2, D2, and F1) from each of the five transects was selected

for temporal analysis (see Figure C-3, Appendix C). Samples from these stations were collected and analyzed in March, June, and August 1987. Stations B2 and D2 were again sampled and the data analyzed in February 1988. Results of the temporal analysis are presented by Blake *et al.* (1987; 1988).

The results from March 1987 are the only ones where samples from all 15 of the benthic infaunal stations were processed and analyzed. These results allowed an assessment of differences in benthic assemblage among stations and transects to elucidate patterns and trends in benthic communities with increasing distance from shore and sources of contamination in Boston Harbor (see Figure 35). These results also permitted an assessment of the sedimentary and biological conditions at each of the possible outfall sites.

The most important results that were reported by Blake *et al.* (1987) indicated that the benthic infauna of Massachusetts Bay shared affinities with adjacent continental shelf habitats rather than nearshore estuarine habitats. The dominant species of benthic assemblages in Massachusetts Bay were, with few exceptions, ones that were not typical dominants in nearshore areas such as Boston Harbor. These results suggested that the benthos of Massachusetts Bay was not stressed by input of contaminants or organic enrichment. However, historical data suggest that periodic episodes of organic enrichment undoubtedly occur to account for occasional population explosions of spionid polychaetes such as *Spio limicola.*

To consider the distribution of the Massachusetts Bay benthic infauna more completely, a series of maps were prepared by Blake *et al.* (1987). These maps plotted the distributions of 17 dominant species, depicting the average number of individuals per square meter for each species. These species were selected because they were regularly listed on the dominant- species lists at several stations. Summaries of the biology and the importance of these species in the Massachusetts Bay/Cape Cod Bay ecosystem are presented in the next Section.

2.2 TEMPORAL PATTERNS

Evidence that station similarity patterns were maintained temporally was demonstrated in the results of clustering Stations A2, B2, C2, D2, and F1. These results indicated that each station tended to be more similar to itself over time than to any other station during a given period. The only exceptions were with the stations on transects C and D, which tended to cluster more closely as a group and showed some mixing of cruises and stations. These results are similar to those of the 3-year Georges Bank Benthic Infauna Monitoring Program in that stations from a highly heterogeneous region tended to remain more similar to themselves over time than to other stations (Maciolek-Blake *et al.*, 1985). This finding is significant because a station "signature" can be established, and when changes in the clustering patterns do occur they can be readily noted and assessed. In this manner, it will be possible to determine when natural or external influences come into play.

The knowledge that there can be a temporal consistency of similarity at stations in Massachusetts Bay provides a powerful framework on which a monitoring program can be developed. It is significant that the major clustering patterns of the temporally sampled stations remained consistent over time despite several statistically

significant changes in the densities of dominant species. This fact suggests that the signature of a station is maintained by the entire suite of species found at a site, rather than being determined only by the dominant species. In this regard, it will be important to continue to use NESS (normalized expected species shared) as a clustering algorithm because it is more sensitive to the less common species. Unfortunately, a full set of seasonal samples has yet to be developed and continued collection of data at the temporal stations is needed for the fall and winter seasons in order to fully establish the temporal consistency of stations in Massachusetts Bay.

An important benthic database was developed for Cape Cod Bay by Battelle (1987a) as part of an environmental impact report (EIR) to designate a dredged-material disposal site (see Figure 14: stations B, C, D, and E). The data have not been fully analyzed, but several trends and patterns in the benthos reflect similarities and differences between Cape Cod Bay and Massachusetts Bay. The spionid polychaete *Spio limicola* was the overall dominant species at most of the stations sampled for infauna. Two other polychaetes, *Euchone incolor* and *Mediomastus ambiseta* were the highest- ranked species on some sampling occasions. *E. incolor* has been shown (Rhoads and Young, 1971) to form dense populations on the fecal mounds of the holothurian *Molpadia oolitica* in Cape Cod Bay. The absence of the *Molpadia* mounds in Massachusetts Bay undoubtedly contributes to an explanation of why this species is not one of the dominants. Oligochaetes and the polychaete *Cossura longocirrata* were also high-ranked species in Cape Cod Bay. Neither *E. incolor* nor *C. longocirrata* were important components of the benthos in Massachusetts Bay as sampled during the 301(h) and STFP programs, although *C. longocirrata* was found by Gilbert *et al.* (1976) in moderate abundances at offshore stations north of Hull and in generally low abundances offshore between Hull and Cape Cod Bay. Infaunal densities in Cape Cod Bay were very high and compared favorably to those collected in Massachusetts Bay. An incorrect multiplication factor appears to have been used in calculating the densities per square meter that were presented in the EIR (Battelle, 1987a). Corrected average densities for four candidate disposal sites range from 18,063 to 27,653 individuals per square meter at Site B, 13,648 to 31,422 at Site C, 7210 to 18,290 at Site D, and 11,760 to 12,069 at Site E. Individual samples frequently exceeded 50,000 individuals per square meter. Summed data for the 10 highest-ranked species and total density at one station sampled in Site B on three occasions in 1985 are presented in Table 1.

Table 1. Ten Dominant Species from 1985 Cape Cod Bay EIR. Station B1 summarized from three surveys. (Data from Battelle, 1987.)

Species	Total Number Indviduals	Density per m^2
Spio limicola	2313	5140
Mediomastus californiensis	2213	4918
Euchone incolor	1591	3535
Oligochaeta spp.	787	1748
Cossura longocirrata	466	1036
Prionospio steenstrupi	370	822
Aricidea catherinae	327	727
Ninoe nigripes	146	324
Parougia caeca	139	309
Capitella capitata	114	253
Total Density	8509	18,909

2.3 DISTRIBUTION AND BIOLOGY OF SOME DOMINANT BENTHIC INFAUNAL SPECIES FROM MASSACHUSETTS BAY AND CAPE COD BAY

Research that focuses on individual species is often neglected when larger, community assessments are undertaken. Yet, unless we understand how individual species live, it will not be possible to fully interpret changes in faunal assemblages. Species differ in thier response to stress and knowledge of the ecology of individual species will assist in an interpretation of changes that might occur in a long-term study of benthic populations.

At present, very little is known about the bilogy of benthic invertebrates in Massachusetts Bay. Despite the high density and wider geographic distribution of some local species there is essentially no information on their mode of feeding, rate of growth, timing or mode of reproduction, development, and recruitment. As part of the analysis of infaunal data during the STFP, some information on sediment preference, seasonal density, and recruitment was compiled for 21 species. These results are summarized in this section.

Exogone hebes

During the STFP sampling effort, this small syllid polychaete was most abundant at nearshore Station B3 and at stations on both offshore transects D and F. The species was rare or infrequent at the other stations of the study area. The densities recorded for this species were similar to the highest densities of this species observed at Georges Bank (Maciolek-Blake *et al.*, 1985). The stations where *E. hebes* was most abundant generally had the highest percent sand/gravel in the sediments, except for the nearshore transect A, where sand content was high but *E. hebes* was rare. At these stations, the ripple marks on the bottom observed in the ROV reconnaissance (Etter *et al.*, 1987) suggest that strong currents were present, and, despite the high sand content, the conditions may not have been appropriate for *E. hebes*. Correlations with sediment grain size using Spearman's coefficient of rank correlation indicate that sediment is not well correlated with the distribution of this species. The species was not important in the Cape Cod Bay EIR study.

Temporal data were compiled from Stations D2 and F1. The percent contribution of the 0.3-mm fraction was high between March and May, but declined sharply between May and August. Differences between seasons indicate that at Station D2 spring (May) recruitment was followed by a growth phase in August. In contrast, at Station F1 both the 0.5- and 0.3-mm fractions declined in August following a heavy spring recruitment, suggesting that a non size-selective disturbance decreased the density of the *E. hebes* population. A predatory polychaete, *Phyllodoce mucosa,* appears in the dominance list at Station F1 for the first time in August and may be responsible for the decrease in *E. hebes* density. (See Figure 25 and Tables 18 and 22 in Blake *et al.*, 1987 for data).

Exogone verugera

This syllid was found by Blake *et al.* (1987) at the offshore transects C and D, with densities similar to the high densities of this species recorded from Georges Bank, where *E. verugera* is a major component of the infauna (Maciolek-Blake *et al.*, 1985). The very high densities recorded from Station D3 (15,093 individuals per square meter), however, were unusual for this species. The three stations with the highest abundances of *E. verugera* (C2, D2, and D3) were among the deepest in the study area and had high gravel/sand or sand content in the sediment. *Exogone verugera* has a strong correlation with stations having high gravel content (r_s = 0.707) and a moderate correlation with high sand content (r_s = 0.521). The species was also found in high abundance by Gilbert *et al.* (1976) at Station 7 (2185 individuals per square meter).

Temporal data from Blake *et al.* (1987, Figure 25) indicated that declines in the 0.3-mm fraction of this species were evident between May and August at Stations C2 and D2. Consideration of the 0.3- and 0.5-mm fractions separately indicates that changes in density varied spatially and temporally both within and between size fractions. At Station C2, both the 0.5- and 0.3-mm fractions exhibited declines between cruises, but the 0.3-mm fraction suffered a more severe drop in density between May and August. There was an increase in the silt content of Station C2 between May and August (9% to 18%), which might account for the disturbance of the 0.3-mm fraction in a species that is highly correlated with sediments having high gravel and sand. The disparity in the reduced densities in the different size fractions suggests that there was a size-selective disturbance on the juveniles that settled in May so that they did not survive until August. At Station D2, a different situation is evident. Between March and May, both the 0.5- and 0.3-mm fractions increased in density, suggesting that both growth and recruitment had occurred. Between May and August, recruitment into the 0.3-mm fraction was heavy, whereas the 0.5-mm fraction suffered a marked decline in density. Again, the disparity in changes in density of the different size fractions suggest a size-selective disturbance, but at Station D2, the adults were being affected.

Ninoe nigripes.

According to Blake *et al.* (1987), this lumbrinerid polychaete exhibited high densities at most stations of transects B and C, and moderate densities at nearshore Station A1. There was a strong correlation in the distribution of this species with sediment type. The sediments at the stations where this species was abundant were similar in having a nearly equal mixture of sand and silt. Spearman's coefficient correlated very high for silt with *N. nigripes* (r_s = 0.905). The species was rare or absent at other stations. Gilbert *et al.* (1976) reported *Ninoe nigripes* from Station 16 northeast of Hull and from a group of stations at the entrance to Cape Cod Bay. The species was a dominant in the Cape Cod EIR study.

Some seasonal data were compiled for this lumbrinerid polychaete at Station B2 (Blake *et al.*, 1987). Both the 0.5- and 0.3-mm fractions declined between March and May, but the density of the 0.5-mm fraction increased between May and August, suggesting that growth took place during this period.

Aricidea catherinae

This small paraonid polychaete was a major component of several stations sampled by Blake *et al.* (1987) (transects A and B and Station F4). The species was also moderately abundant at Stations B3, C3, and F2. Correlations of this species with sediment type were not obvious and r_s values using Spearman's coefficient of correlation were low for all sediment types. The species (as *A. quadrilobata*) was also found to be widespread and abundant (500-2600 individuals per square meter) throughout the outer Bay (Gilbert *et al.*, 1976), but was rare or absent in nearshore areas and off Cape Ann. In Cape Cod Bay, *A. catherinae* was a dominant species (Battelle, 1987a).

Some seasonal data were developed for this paraonid polychaete at Stations A2 and B2. Although the species declined in percent of juveniles between May and August at both stations, there was a marked difference at the two stations in the March to May results (Blake *et al.*, 1987). At Station A2, the decline in percent juveniles was steady over all three cruises, whereas at Station B2, the percent of juveniles increased substantially in May, followed by a steep decline in August. The data on differences and percent difference between the sampling dates for the two size fractions suggest that there was an overall growth of individuals in the 0.3-mm fraction into the population at Station A2. An early disturbance at Station B2 between March and May was followed by a pattern of growth similar to that seen at Station A2. Stations A2 and B2 have very different sediment characteristics. Station A2 is sandy, whereas Station B2 has a nearly equal distribution of sand and silt. Decreases in the numbers of individuals in the 0.3-mm fraction at Station B2 in August are correlated with an increase in the percent contribution of silt and clay during the same period.

Leitoscoloplos acutus

This orbiniid polychaete was reported by Blake *et al.* (1987) to be most abundant at both nearshore and offshore stations that had the highest silt content. *L. acutus* has a strong correlation with silt content according to tests using Spearman's coefficient (r_s = 0.883). The distribution of this species is similar to that of *N. nigripes* (see above) in that the species occurred at those stations having nearly equal amounts of sand and silt. Gilbert *et al.* (1976) found *L. acutus* south and east of Nahant and at some scattered stations in the southern half of the outer Bay. The silt plus clay content at all those stations was at least 55%, and at some offshore stations (18 and 21) the values were greater than 80%. The species was rare in Cape Cod Bay (Battelle, 1987a).

Seasonal data were compiled for this orbiniid polychaete at Station B2 where the species ranked 6th in all three cruises. Both size fractions declined in density between the three cruises, but the drop in density for the 0.3-mm fraction was much greater than for the 0.5-mm fraction (1% and 72%, respectively) between May and August.

Prionospio steenstrupi

This spionid polychaete was the most abundant infaunal species encountered in the STFP (Blake *et al.*, 1987). It had low densities at only three offshore stations (D1, F1, and F2). The species was ubiquitous in all sediment types, but the highest densities occurred at stations with high silt content (B1, B2, and C3). The species has a moderate correlation with silt content (r_s = 0.613). *P. steenstrupi* (as *P. malmgreni*) was also found in high abundances throughout the area (Gilbert *et al.*, 1976; Blake and Maciolek, manuscript in preparation; Battelle, 1987a). The highest densities were reported from coastal stations, whereas densities at the entrance to Cape Cod bay were generally low.

Seasonal data on differences and percent differences in the 0.5- and 0.3-mm fractions between cruises are presented by Blake *et al.*, 1987 (Table 22). The spatial and temporal changes in density of the 0.3- and 0.5-mm size fractions were extremely complicated and no clear pattern emerged. The numbers of individuals of this species in the 0.3-mm fraction were generally low, and it is probable that recruitment takes place either in the winter or late summer. The results follow the overall density patterns closely in that Stations A2 and B2 exhibit steady declines in total density and in the 0.5-mm fraction over the three cruises. At Station C2, the densities of the 0.5-mm fraction increased in May and declined in August. The 0.3-mm fraction declined monotonically over the three cruises. In contrast, the 0.5-mm fraction declined in May, but increased in August at Station D2. Because the 0.3-mm fraction densities of this species were so low during the periods sampled, it is difficult to make conclusive statements about recruitment and growth. Size frequency data and information on presence of eggs and sperm in adults developed from these same collections would enhance our efforts to understand the population biology of this dominant species.

Spio limicola

This spionid polychaete occurred in high densities only at Stations C1 and F4, and was either absent or had only low to moderate densities at the other stations. There did not appear to be a strong correlation with sediment texture. The results of Spearman's coefficient, however, were a moderate negative correlation with sand (r_s = -0.506) and a moderate positive correlation with silt (r_s = 0.506). The species has historically been very abundant in Massachusetts Bay (Gilbert *et al.*, 1976; Blake and Maciolek, manuscript in preparation; Metcalf and Eddy, 1984) and in Cape Cod Bay (Battelle, 1987a). *S. limicola* is believed to be the species responsible for the dense tube mats that periodically appear in the area.

Although the dense tube mats were not evident in 1987, the species was among the dominants at Stations B2, C2, and D2, and data have been developed on the seasonal density and contributions of the juvenile and adult populations. The results for percent juveniles at Stations C2 and D2 are presented in Blake *et al.* (1987, Figure 25). The results are dramatic in that the juvenile fraction nearly disappeared between May and August, indicating an end to the larval recruitment evident during March and May. Data on the differences in the 0.5- and 0.3-mm fractions between cruises are presented in Blake *et al.* (1987, Table 22). For Stations B2 and D2, the results indicate increases in the 0.5-mm fraction during the recruitment phase and declines in the 0.3-mm fraction, suggesting a pattern of growth from the 0.3-mm fraction to the 0.5-mm fraction. In contrast, the

dramatic declines in both size fractions between May and August at Station C2 indicate a powerful disturbance on *S. limicola.* There was a doubling of silt content at Station C2 between May and August (9% to 18%), which might account for the disturbance on the 0.5-mm fraction.

Polydora quadrilobata

Seasonal data for *P. quadrilobata* were developed at Stations A2, C2, and D2. The results are similar to those for *S. limicola* in that the juveniles declined to near zero between May and August following what appeared to be recruitment during the earlier cruises. Increases in the 0.5-mm fraction between May and August may reflect growth of new recruits into the 0.5-mm fraction. The decline in juveniles between May and August agrees with earlier results on the reproduction and larval development of this species, indicating that it produces larvae in late winter and early spring (Blake, 1969).

Polydora socialis

This spionid also exhibited steep declines in the percent juveniles between May and August (see Figure 25 in Blake *et al.*, 1987), indicating an end to larval recruitment. Data on differences in the 0.5- and 0.3-mm fractions for Stations C2 and D2 indicate that both size fractions declined between May and August. The disturbance appears to be non size-selective because both fractions declined, but the nature of the disturbance cannot be determined at this time. Data on the reproductive biology and larval development of *P. socialis* would suggest predation, or some other barrier to recruitment, because the summer months are periods of greatest reproductive activity for this species (Blake, 1969).

Tharyx acutus

Tharyx acutus is widespread in Boston Harbor and Massachusetts Bay. As part of the STFP, this cirratulid polychaete was found by Blake *et al.* (1987) in highest abundances at Station F2 and at stations on transect A, but was also very abundant elsewhere in the study area, except Station C3 and at stations on transect D. There are no apparent correlations with sediment type that could account for the distribution of this species, and tests using Spearman's coefficient were low for all sediment types. No data are available on this species from other studies, although Gilbert *et al.* (1976) recorded two unidentified species of *Tharyx* that may include *T. acutus.* Three other cirratulid polychaetes are among the 10 highest-ranked species at Station F2, thus representing an unusual assemblage.

This cirratulid is one of the dominant species in Massachusetts Bay. Seasonal data on the percent juveniles at Stations A2, B2, and F1 indicate that the highest numbers of juveniles were present in March, and declined between May and August. Data developed on the differences in the 0.5- and 0.3-mm fractions between sampling occasions at Stations A2 and B2, however, indicate that both fractions declined monotonically across all three cruises, probably reflecting a non size-selective disturbance.

Monticellina baptisteae

This cirratulid polychaete was found by Blake *et al.* (1987) in high abundances at stations with high silt content except for Station F2 that had no silt. Results of the Spearman's coefficient indicated that the species had a strong correlation with silt (r_s = 0.855) and a moderate correlation with clay (r_s = 0.617). In other studies, this species was not reported, but it may have been misidentified. The taxonomy of this species is being reviewed in a forthcoming paper (Blake, 1991).

Chaetozone setosa.

Seasonal data on this cirratulid polychaete were developed at Station F1. The 0.3-mm fraction declined steeply between March and May and again between May and August, reducing the juvenile component to near zero by late summer (see Figure 26 in Blake *et al.*, 1987). At the same time, the 0.5-mm fraction increased between May and August, indicating growth (see Table 22 in Blake *et al.*, 1987). These results suggest that the species reproduces in the winter and early spring.

Mediomastus californiensis

This species is very common in Massachusetts Bay. During the STFP study (Blake *et al.*, 1987), *M. californiensis* was present at all stations in the study area, but although it was abundant at 9 or 10 stations, it was never the top-ranked species and never achieved the extreme densities recorded for Cape Cod Bay (Battelle, 1987a) where it is the second highest ranked species. This species has a strong correlation with silt content (r_s = 0.718). Gilbert *et al.* (1976) found *M. californiensis* (as *Heteromastus filiformis*) in fairly high abundances throughout the Bay, with the highest numbers of individuals occurring at stations with about 80% silt plus clay (Figure 17).

This capitellid polychaete is an important component of Stations A2, B2, C2, and D2. Data on the percent contribution of the 0.3-mm fraction or juveniles over three seasons are presented by Blake *et al.* (1987, Figure 26). There was an overall decrease in the contribution of the juveniles to the total density from March to August, suggesting that recruitment was heaviest in late winter and early spring. Differences between the cruises for the 0.5- and 0.3-mm fractions are presented in Blake *et al.*, 1987, Table 22. These differences vary from station to station. At Station A2, early increases in the 0.5-mm fraction coupled with decreases in the 0.3-mm fraction between March and May suggest that growth was taking place. Declines in these values between May and August, however, indicate a non size-selective disturbance on both fractions. At Stations B2 and C2, both fractions declined in both periods, indicating continuous disturbance on the population regardless of size. For Station D2, the results are similar to those for Stations B2 and C2, except that there is a slight increase in the 0.5-mm fraction between May and August, indicating that the disturbance was more restricted to the 0.3-mm fraction.

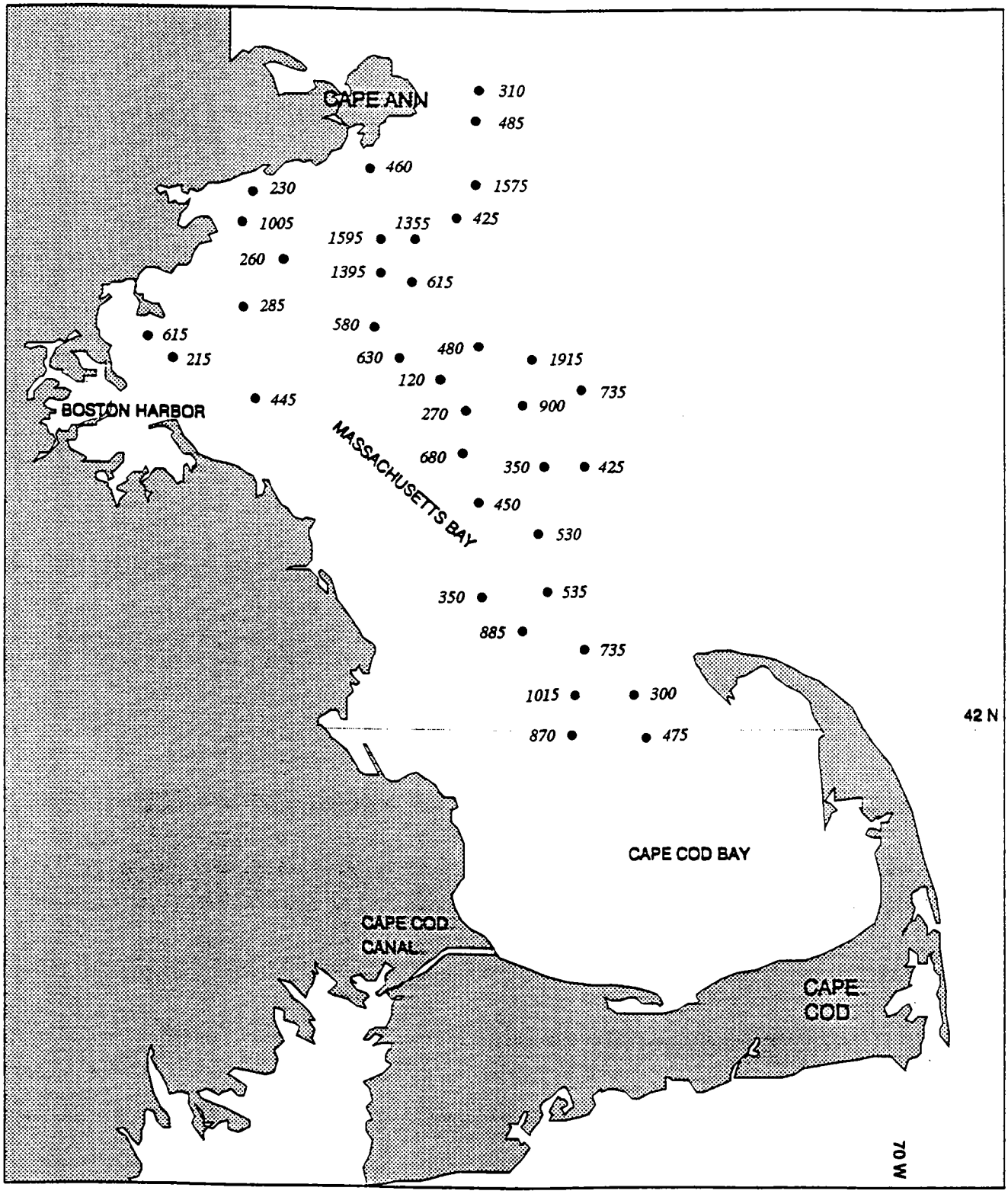

Figure 17. Map of the Massachusetts Bay–Cape Cod Bay System Showing the Distribution and Total Density (Number of Individuals per m^2) of Mediomastus californiensis, a Common Capitellid Polychaete. (after Gilbert et al., 1976, as Hetermastus filiformis.)

Owenia fusiformis

This oweniid polychaete has a very patchy distribution in Massachusetts Bay. During the STFP study, it was the number 1-ranked species at Station A1. The species was also important at STFP Stations B3 and F4, but was rare or absent elsewhere in the study area. The species exhibited a moderate correlation with silt with Spearman's coefficient (r_s = 0.643). Gilbert *et al.* (1976) found this species in low densities at a few scattered offshore stations.

Euclymene sp A

This maldanid polychaete was most abundant at offshore STFP stations sampled by Blake *et al.* (1987) with high sand content but little or no silt. Correlations between sediment type and density using Spearman's coefficient were not strong. These densities and sediment types are comparable to densities recorded for this species at stations on Georges Bank (Maciolek-Blake *et al.*, 1985). The species has not been identified during other studies.

Enchytraeidae sp. 1

Oligochaetes were only identified to species during the STFP study (Blake *et al.*, 1987), and may therefore be more widespread than is apparent from the available historical data. Enchytraeidae sp. 1 was characteristic of offshore Stations D1, D3, F1, and F2, which had little or no silt. Correlations using Spearman's coefficient, however, were not significant. At Station D1, the densities were sufficient to make it the number 1-ranked species. The highest densities recorded were at Station F2.

Tubificoides apectinatus

Blake *et al.* (1987) found this oligochaete to be most abundant at nearshore transect A (Stations A2 and A3) and either rare or absent elsewhere. Both Stations A2 and A3 were sandy with obvious ripple marks (Etter *et al.*, 1987), suggesting a high-energy environment. Sediment correlations using Spearman's coefficient were all low, but highest with percent sand (r_s = 0.364).

This oligochaete was very abundant at Station A2. Data on differences between three seasons for the 0.5- and 0.3-mm fractions indicate that growth occurred between March and May, but the declines in both fractions between May and August were indicative of a disturbance to the population.

Phoronis architecta

This species is the common phoronid of the Atlantic coast. During the STFP, it was found on inner transects A and B and at offshore Station F4. *Phoronis architecta* is a suspension feeder, and its distribution may be correlated more with water movement than with sediment type, because correlations with sediment type were not significant. Gilbert *et al.* (1976) reported *P. architecta* from their coastal stations (except Cape Ann, Stations C-4) and a few scattered offshore stations.

Corophium crassicorne

This amphipod was characteristic of the deeper STFP locations such as Station C2 and stations on transect D. There does not appear to be any obvious sediment correlation with sediment texture, and correlations with Spearman's coefficient were not significant. The species was not reported in other studies.

This amphipod exhibited declines of percent juveniles in the 0.3-mm fraction between March and May at Station D2 while the 0.5-mm fraction was increasing, suggesting growth of a previously settled juveniles. The continued decrease in the 0.3-mm fraction between May and August accompanied by a comparable decrease in the 0.5-mm fraction suggests that a size-indifferent disturbance occurred between May and August.

Unciola irrorata

This amphipod occurred at offshore STFP Stations C1, C2, and D2, but was rare to absent elsewhere. Correlations between density and sediment grain size were not significant with Spearman's coefficient. *U. irrorata* was found in moderate abundances at a few stations around Cape Ann and at one coastal station (C-1) by Gilbert *et al.* (1976), but was not common in the 301(h) and Cape Cod Bay surveys.

The percent contribution of juveniles increased substantially between March and May and then declined sharply in August (Blake *et al.*, 1987, Figure 26). Data on the difference between cruises for the 0.5- and 0.3-mm fractions indicate that large increases in the 0.5-mm fraction occurred, at Station C2, suggesting growth in the population. At Station D2, on the other hand, both fractions exhibited significant declines between May and August, indicating a disturbance was operating on the population at that site.

Astarte undata

This small bivalve retained high numbers of juveniles in the total population throughout the three sampling periods (Blake *et al.*, 1987, Figure 26). Differences between cruises in terms of the 0.5- and 0.3-mm fractions indicate that some disturbance of the entire population occurred between May and August at Station D2, but that mortality appeared to be heaviest in the 0.3-mm fraction.

3.0 CHEMICAL CONTAMINANTS IN SURFACE SEDIMENTS

Chemical contamination of surface sediments in Massachusetts Bay has not been studied systematically, but there have been several studies within the last 20 years that, when combined, provide baywide data for several trace metals, PCB, PAH, and ancillary parameters (grain size and total organic carbon). In this review, chemistry data from many of these studies were compiled and reviewed to evaluate their comparability and utility in establishing the status of chemical contamination in Massachusetts Bay sediments. Data were reviewed for the geographic region in Massachusetts and Cape Cod Bays extending from Cape Ann to Cape Cod and including Stellwagen Bank, but excluding Boston Harbor (see Figure 1). This review was intended to answer the following questions.

Are the available data on sediment-bound contaminants comparable?
What are the general patterns of contaminant distribution?
Are the levels of contaminants elevated above background?
What is the variability in the observed contaminant spatial distribution?
What are the implications of this review to future monitoring?

The most extensive survey of the distribution of contaminants in sediments of Massachusetts Bay was reported by Gilbert *et al.* (1976). During this study, sediment grab and core samples were collected at 37 stations located throughout Massachusetts Bay (see Figure C-1, Appendix C) in areas containing mostly fine-grained sediments. Analyses included metals, PCBs, hydrocarbons, and Pb-210 activity in core sections, metal analysis in the interstitial porewater, and sediment grain size. In addition, benthic fauna were characterized in the sediment grab samples (these data are reviewed in Section 2.0) and side-scan sonar and underwater photography were used to characterize the bottom topography.

Results from a National Oceanic and Atmospheric Administration/North East Monitoring Program (NOAA/ NEMP) survey of organic contaminants in sediments of Massachusetts Bay were presented by Boehm *et al.* (1984). They reported PAH, PCB, and coprostanol (a fecal sterol that can be used as a sewage tracer) concentrations in sediments at 16 locations in Massachusetts Bay, two in Cape Cod Bay, and six sites in Boston Harbor (see Figure C-2, Appendix C). Complementary trace-metal data were provided by NOAA (NOAA, unpublished data).

The most intensive sediment sampling program conducted in the vicinity of the proposed sewage outfall was for the Deer Island Secondary Treatment Facilities Plan (STFP) (Battelle, 1987b). In this study, sediment samples were collected along four transects near the proposed outfall tunnel and an additional transect approximately 2 nmi south of this area (see Figure C-3, Appendix C). Samples were collected on three separate cruises from early spring to late summer, and analyzed for PAH, metals, TOC, and grain size, along with the benthic biological parameters reviewed in Section 2.0. Sediment cores were collected at four additional sites to assess historical trends (Battelle, 1989).

These three studies represent the bulk of the data on the contaminants in surface sediments of Massachusetts Bay. In addition to these studies, there have been several surveys with more narrow scopes, focusing on either

a more restricted spatial scale such as Boston Harbor, the Massachusetts Bay Disposal Site (MBDS), or the Cape Cod Bay Disposal Site (CCBDS), or a specific environmental process (e.g., contaminant flux measurements). All of the data sources that were considered for this report are listed in Table 2.

3.1 COMPARABILITY OF AVAILABLE DATA

The evaluation of data comparability presented here is a simple qualitative assessment of methods used for sample collection and analysis. The following questions were used to evaluate data comparability.

Are the field and analytical methods comparable to current, established procedures?
Is there adequate documentation of quality control results and quality assurance?
Do the reported quality control results meet typical acceptance criteria?

A brief evaluation of the data sets is presented below and some comments on data comparability are given in Table 2.

3.1.1 Trace-Metal Data

Studies of metal concentrations in sediments from Massachusetts Bay include the bay-wide survey by Gilbert *et al.* (1976), the NOAA/NEMP survey conducted in 1983 (NOAA, unpublished data), the STFP survey along the proposed outfall tunnel (Battelle, 1987b, 1989), the Salem Harbor/Salem Sound survey by Gardner *et al.* (1986), and several smaller studies of the MBDS (listed in EPA, 1989), the CCBDS (listed in Battelle, 1990a), and the NOAA Mussel Watch Program (Battelle, 1990b).

Differences in the sediment depth collected (top 2 cm versus top 5 cm) in these studies were not considered to be significant based on sediment core data from Gilbert *et al.* (1976), Battelle (1989), and studies from other estuarine and coastal regions. Documentation of field equipment and methods was generally incomplete; therefore, the comparability of sample-collection methods is unknown. However, assuming that representative samples were collected, the most significant process affecting trace-metal data comparability is the method of sediment digestion.

TABLE 2. SOURCES OF DATA CONSIDERED FOR THIS REVIEW.

Reference	Parameters	Comments
Battelle, 1987b	Metals, PAH, TOC, grain size	MWRA-STFP study. Transects near proposed outfall tunnel. Partial digestion for metals. Methods/QC results are well documented.
Battelle, 1989	Metals, radionuclides, PAH, PCB	Depth profiles (cores) at four stations in Massachusetts Bay. Partial digestion for metals. Methods/QC results are well documented.
Battelle, 1990a	Metals, PAH, PCB	Four stations in Cape Cod Bay. EPA CLP procedures were used for PAH and PCB. Total digestion was used for metals. QC results are not well documented.
Battelle, 1990b	Metals, PAH, PCB, pesticides	NOAA Mussel Watch Program. Several sites in Boston Harbor and Massachusetts Bay. Methods/QC results are well documented. Total metals digestion.
Boehm *et al.*, 1984	PAH, PCB, coprostanol, TOC	16 stations in Massachusetts Bay, 10 in the Gulf of Maine and 22 in New York Bight. Methods/QC are well documented.
EPA, 1989	Metals, PAH, PCB, others	Compilation of data from the Massachusetts Bay Disposal Site (MBDS). The methods/QC results for most of the individual studies were well documented. Mostly total metals digestion. Individual studies are discussed in the text.
Fitzgerald, 1980	Metals	Restricted to Boston Harbor. Partial sediment digestion for metals. Methods/QC are not well documented.
Gardner *et al.*, 1986	Metals, LAB, TOC	Salem Harbor area. Metals - 2 N nitric acid digestion. QC results are not well documented.
Gilbert *et al.*, 1972	Metals	Restricted to Boston Harbor. Rigorous partial digestion. QC results are not well documented.

TABLE 2. SOURCES OF DATA CONSIDERED FOR THIS REVIEW (continued)

Reference	Parameters	Comments
Gilbert, 1975	Metals	Massachusetts Bay Disposal Site (no station positions were given). Rigorous partial digestion. QC results are not well documented.
Gilbert *et al.*, 1976	Metals Organics	37 stations in Massachusetts and Cape Cod Bays. Rigorous partial digestion for metals. Poor chromatographic resolution for organics. QC results are not well documented.
Gschwend and Hites, 1981	PAH	PAH flux measurements in Boston Harbor and other sites in northeastern US.
JRB Associates, 1984	Metals, PAH	Four sites in Cape Cod Bay (for EIR). Methods/QC were not available.
Larsen *et al.*, 1986	PAH	Primarily in the Gulf of Maine. QC results are not well documented.
Metcalf and Eddy, 1984	Metals, PCB, pesticide, GS, TOC	301(h) Waiver Application, mainly restricted to Boston Harbor. EPA-600 methods, QC results were not available.
NOAA, unpublished	Metals	Unpublished metals data. Sediment digestion method was very similar to Battelle, 1987b.
SAIC, 1986	Metals	Four stations at MBDS. Total digestion.
Windsor and Hites, 1979	PAH	13 stations in the Charles River, Massachusetts Bay, and Gulf of Maine. QC results are not well documented.

Total sediment digestion using HF acid yields the highest recovery of metals and generally the best precision. This method was used for the NOAA Mussel Watch samples and most of the studies at the two disposal sites. Gilbert *et al.* (1972, 1976) and Gilbert (1975) used a partial, but rigorous sediment digestion by refluxing with aqua regia. An intermediate (aqua regia) partial sediment digestion was performed by Battelle (1987b, 1989) and NOAA (unpublished data); these two studies had comparable recoveries of similar reference materials, indicating that the sample data are also comparable. A few studies at the MBDS site used either undocumented methods or other partial digestion methods that probably yielded lower recoveries for some metals. Gardner *et al.* (1986) used a mild 2 N nitric acid leach that would be expected to yield the lowest recoveries, but also would better reflect the bioavailable fraction of metal in the sediment. (This is the method currently being tested by the EPA to evaluate sediment quality.) Unfortunately, the differences in sediment digestion methods make direct comparison of these various data sets inappropriate. Significant differences in the completeness of different partial digestion procedures (recovery of metals) have been reported by Loring (1986). A factor of 2 or more difference between the weak acid leach and the more rigorous aqua regia method was reported. Although the differences in recovery can have relatively good precision (CV $<$ 10%) when analyzing standard reference material (thus providing a normalizing factor), the reproducibility of this difference with natural samples (of varying mineralogy and organic content) is often poor (CV $>$ 30%). In addition, most of the studies did not report results from standard reference material analysis. Therefore, the only trace-metal data that appear to be directly comparable are those from Battelle (1987b, 1989) and NOAA (unpublished data), although formal documentation of the methods and quality control results for the NOAA study were not available. The remaining data are still useful for evaluating general trace-metal levels and distribution patterns, but direct, quantitative comparison of the data should not be made. The data acquired by using a total sediment digestion method should be comparable to any future monitoring data obtained by using this method.

Gilbert *et al.* (1976) also extracted a subset of the sediment samples with hydroxylamine hydrochloride, which leaches metals sorbed to the oxides of Fe and Mn that coat sediment particles. The results indicated that most of the Cd and Pb, and about half of the Cr were reversibly sorbed to the Fe and Mn oxide coatings. Smaller portions of Zn and Cu were extracted, indicating that these metals were associated with organic coatings or tightly bound in aluminosilicate or sulfide minerals. Metals sorbed to Fe and Mn oxides are believed to be more bioavailable than if bound to other sediment phases (see review by Shea, 1988). These data could be useful when interpreting the metals data in terms of potential effects on benthic organisms, but are not discussed further in this document.

3.1.2 Organic Contaminant Data

Studies of organic contaminants in sediments from Massachusetts Bay include the bay-wide survey by Gilbert *et al.* (1976), a NOAA/NEMP survey conducted by Battelle in 1983 (Boehm *et al.*, 1984), the STFP survey along the proposed outfall tunnel (Battelle, 1987b), the Salem Harbor/Salem Sound survey by Gardner *et al.* (1986), and several smaller studies of the MBDS (listed in EPA, 1989), the CCBDS (listed in Battelle, 1990a), the NOAA Mussel Watch Program (Battelle, 1990b), and studies on contaminant fluxes (e.g., Windsor and Hites, 1979).

The organic contaminant data reported by Gilbert *et al.* (1976) were acquired by using analytical methods that are not comparable to those used today (lower extraction recoveries and poor resolution, packed-column gas chromatography), and, therefore, they are not discussed further in this review. The organic contaminant data from the NOAA/NEMP and STFP studies are all comparable because the data were obtained by similar methods (although the individual analytes were not identical), and the quality control results were acceptable and well documented. These two data sets represent the bulk of available data and are combined below to evaluate the spatial distribution of organic contaminants. The NOAA Mussel Watch organic contaminant data are also comparable to these two studies, but the number of sampling sites in Massachusetts Bay is very small. In general, organic contaminant data from the studies at the MBDS and CCBDS were not comparable to the NOAA/NEMP and STFP studies because of differences in analytical methods, poor field documentation (e.g., missing station locations), or poor quality-control results and documentation. Therefore, these other studies are discussed below only with respect to their general conclusions.

3.2 GENERAL PATTERNS OF CONTAMINANT DISTRIBUTION

3.2.1 Spatial Distribution of Metals

Within the constraints on data comparability given above, a general distribution of trace metals in surface sediments can be described. Trace-metal concentrations are about 10 to 100 times lower in the sediments of Massachusetts Bay than in the sediments of Boston and Salem Harbors. There is also a general trend of decreasing concentrations with distance from these two Harbors (often logarithmical). There is general agreement among the studies that concentrations of metals are highest in the fine-grained sediments in the northern part of Broad Sound, near the midsection of the proposed outfall tunnel, and near the MBDS. Concentrations of Pb, Cu, and Cr were elevated in the fine-grained sediments 10-20 km to the southeast of the MBDS in the deeper regions of Stellwagen Basin. All other areas had uniformly low concentrations of metals.

In general, TOC was positively correlated with the silt-plus-clay fraction (see Figure C-4, Appendix C) and metal concentrations were highest in the finer-grained (and high TOC) sediments. However, correlations of metal concentrations with grain size (or TOC) were not high for all metals. Pearson-Neuman correlation coefficients were calculated for the STFP data (Battelle, 1987b) for each of the three cruises. Moderately high, positive correlations (n=45) to the silt-plus-clay fraction were found for Cd (r=0.600), Cr (r=0.917), Cu (r=0.744), and Hg (r=0.634) in April; Cr (r=0.826) and Hg (r=0.779) in June; and Cr (r=0.773) and Hg (r=0.627) in August. All other correlations had an $r<0.6$.

Similarly, using the data from Gilbert *et al.* (1976), only Cd, Cr, and Ni had moderately high, positive correlations (>0.6, n=8) to silt plus clay in the four near-coastal stations. The 30 offshore stations had poor correlations ($r<0.3$, n=60) to silt plus clay for all metals. For TOC, only Cr had a consistently high, positive correlation (>0.6). Mercury, Cu, and Pb had a high correlation on only a single STFP cruise. Thus, neither the grain size nor the TOC strongly influences the metal concentrations in Massachusetts Bay sediments. In addition, between-element correlations were consistently low, with the exception of correlations to Ni. Plots of metal concentrations versus Ni are shown in Figures C-5 through C-21 for the Battelle (1987b), NOAA

(unpublished), and Gilbert *et al.* (1976) data. Data points that lie above the general trend in these plots indicate possible enrichment (accumulation) of the metal relative to the rest of Massachusetts Bay. Normalization of the metal concentrations to Ni significantly reduced the amount of scatter (and number of outliers) in these data. Normalization of metal concentrations to Al or Fe is much more common than Ni; however, Al and Fe were measured in only a small fraction of the overall data set and there is excellent correlation between Fe and Ni in samples where they were both measured (see Figure C-15, Appendix C).

Some general trends in metal distributions that were consistent throughout the various data sets are listed below. Chromium, Cu, Hg, and Zn exhibit decreased concentrations outside Boston and Salem Harbors, a small maximum near the midsection of the proposed outfall tunnel, and a second maximum near the MBDS. Chromium exhibits its highest concentrations in muddy areas. Lead does not show the first maximum at the proposed outfall, but is elevated near the MBDS and to the southeast in Stellwagen Basin. Mercury had the greatest variability (10-fold) among these metals, probably because the levels are near the common detection limit at many sites. These metals also were elevated in Salem Harbor, decreasing about an order of magnitude from the South Essex Sewage District (SESD) outfall to Massachusetts Bay.

Cadmium concentrations decrease about an order of magnitude (0.30 to 0.03 μg/g) along the proposed outfall tunnel and increase again at the MBDS. Cadmium is elevated in the Salem Harbor area, decreasing with distance from the SESD. Throughout the rest of Massachusetts Bay, Cd is uniformly low, with a possible increase in the depositional areas near the Massachusetts Bay/Cape Cod Bay boundary. Within the proposed outfall area, Cd exhibited the highest variability (30-fold) between cruises and stations. This variability is likely due to Cd levels being near the detection limit, particularly for the NOAA/NEMP data set (NOAA, unpublished data).

Nickel concentrations were fairly uniform (within a factor of 1 to 2) along the proposed outfall tunnel and throughout Massachusetts Bay, which is consistent with its typical weak anthropogenic signal in marine sediments. Arsenic concentrations were uniformly low (<5 μg/g) throughout western Massachusetts Bay, with one station (F2, Figure C-3) having elevated levels (>30 μg/g) on all three cruises. This station was south of the proposed outfall tunnel and about 6 km east of Hull. Lead concentrations were highest near the Harbor areas and the MBDS. The rest of the Bay had a uniform distribution of slightly elevated concentrations.

Within the proposed outfall area (see stations in Figure C-3, Appendix C), three patterns emerge in the sediment metal distributions. (1) Cadmium, Cr, Cu, Hg, Zn, and probably V show a similar distribution pattern with highest absolute concentrations near the middle of the proposed outfall tunnel (transect B in Figure C-3) and also north in Broad Sound. (2) Arsenic had a similar pattern, except south of the proposed outfall where the concentrations were much higher. (3) Nickel and Pb were evenly distributed throughout this region.

To help to establish the baseline spatial distribution (and variability in Section 3.4) of metals in sediments throughout the entire Massachusetts and Cape Cod Bays region, the Battelle (1987b) and NOAA (unpublished) data were combined and concentration contours were mapped. Contour maps of Cu, Cr, and Pb concentrations are shown in Figures 18, 19, and 20, respectively, along with similar contour plots developed by Gilbert *et al.* (1976). Although the absolute values differ between the two data sets, the spatial distributions are quite similar.

The most striking feature apparent in both sets of maps is the elevated metal levels at the MBDS. Metals generally decline with distance from this dump site to concentrations that are near background (or regional) levels. This is consistent with the studies at the MBDS, which show a logarithmic decrease in metal concentrations from the site (EPA, 1989). The sedimentary metals increase again near some of the Harbor areas where anthropogenic inputs are highest (these data were excluded from the contour maps to minimize contouring artifacts) and at depositional areas in Stellwagen Basin.

Sediment depth profiles (up to 60 cm) for the metals showed that Cu, Pb, Hg, and Zn all decreased with depth (and age), indicating recent input for these metals (Gilbert *et al.*, 1976). However, Cd and Cr exhibited only a minor enrichment in the surface sediments. Ni was uniformly distributed with depth, giving further evidence that anthropogenic sources are insignificant for this metal. Vertical distribution data from Battelle (1989) indicate there has been recent accumulation of Pb, Hg, and probably Cr in the recently deposited fine-grained sediments of Massachusetts Bay, particularly in the Stellwagen Basin. There was also evidence of rapid mixing of the upper 10-20 cm of fine-grained sediment, indicating that dilution of recent (or future) input must be considered when interpreting data from surface sediments.

Although most of the trace-metal data cannot be compared directly, they do provide a consistent picture of metal distributions in the sediments of Massachusetts Bay. Areas of metal accumulation appear to be near the midsection of the proposed outfall tunnel, the depositional areas of Broad Sound (north of the proposed tunnel), the MBDS, the deeper regions of Stellwagen Basin, and possibly the depositional area in northern Cape Cod Bay. These regions are good sites to monitor long-term accumulation of contaminants in the sediments of Massachusetts Bay.

3.2.2 Spatial Distribution of Organic Contaminants

Windsor and Hites (1979) conducted a study of PAH in the sediments of Boston Harbor, Massachusetts Bay, and the Gulf of Maine. The concentrations of total PAH were 120 μg/g in the Charles River and 8.5 μg/g in Boston Harbor. Concentrations in the Massachusetts Bay ranged from 0.160 to 3.4 μg/g, with a predominance of combustion-derived PAH (high molecular weight PAH). Concentrations of PAH decreased logarithmically with distance from Boston Harbor. There was also a corresponding decrease in the contribution of petroleum-derived PAH (low molecular weight PAH) with distance from Boston. In the Gulf of Maine sediments, total PAH concentrations ranged from 0.20 to 0.87 μg/g, and were entirely from combustion sources.

Gschwend and Hites (1981) estimated the fluxes of individual PAH to Boston Harbor to be in the range of 14 to 39 ng/cm^2/year, with an average of about 20 ng/cm^2/year. They concluded that urban runoff was the major source of PAH to the Harbor, with atmospheric deposition accounting for only a 10th of the total flux. Vertical profiles from sediment cores showed a uniform distribution of PAH, indicating that very little microbial degradation had occurred in these sediments. The authors postulated that the PAH were tightly bound by soot particles and, therefore, were not readily available for utilization as an energy source.

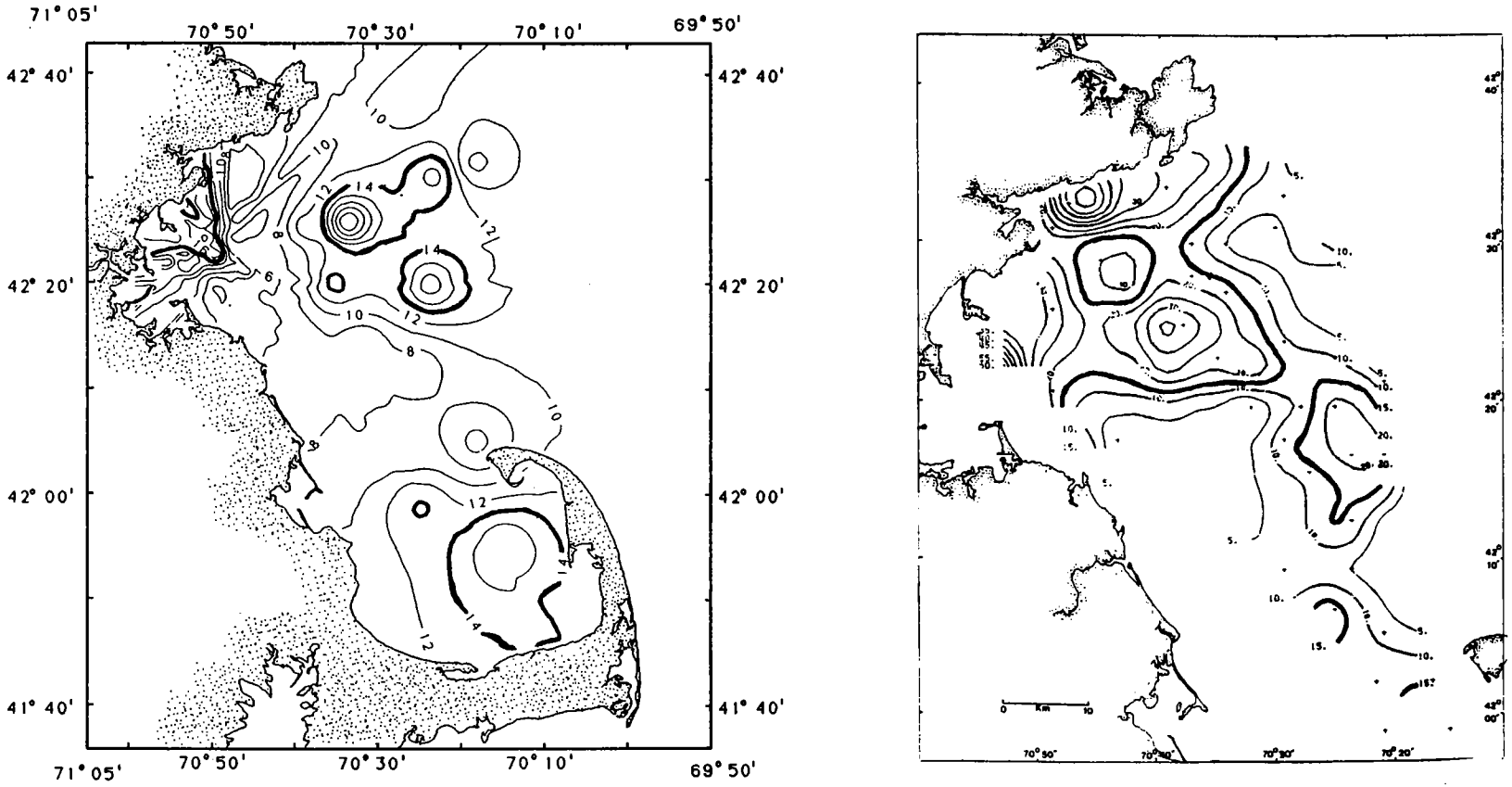

Figure 18. **Contour Map of Copper Concentration (ppm, Dry Weight). Left Map is Data From Battelle (1987b) and NOAA (Unpublished), Contour Interval is 2 ppm, Bold Line is 14 ppm. Right Map is From Gilbert *et al.* (1976), Contour Interval is 5 ppm, Bold Line is 15 ppm.**

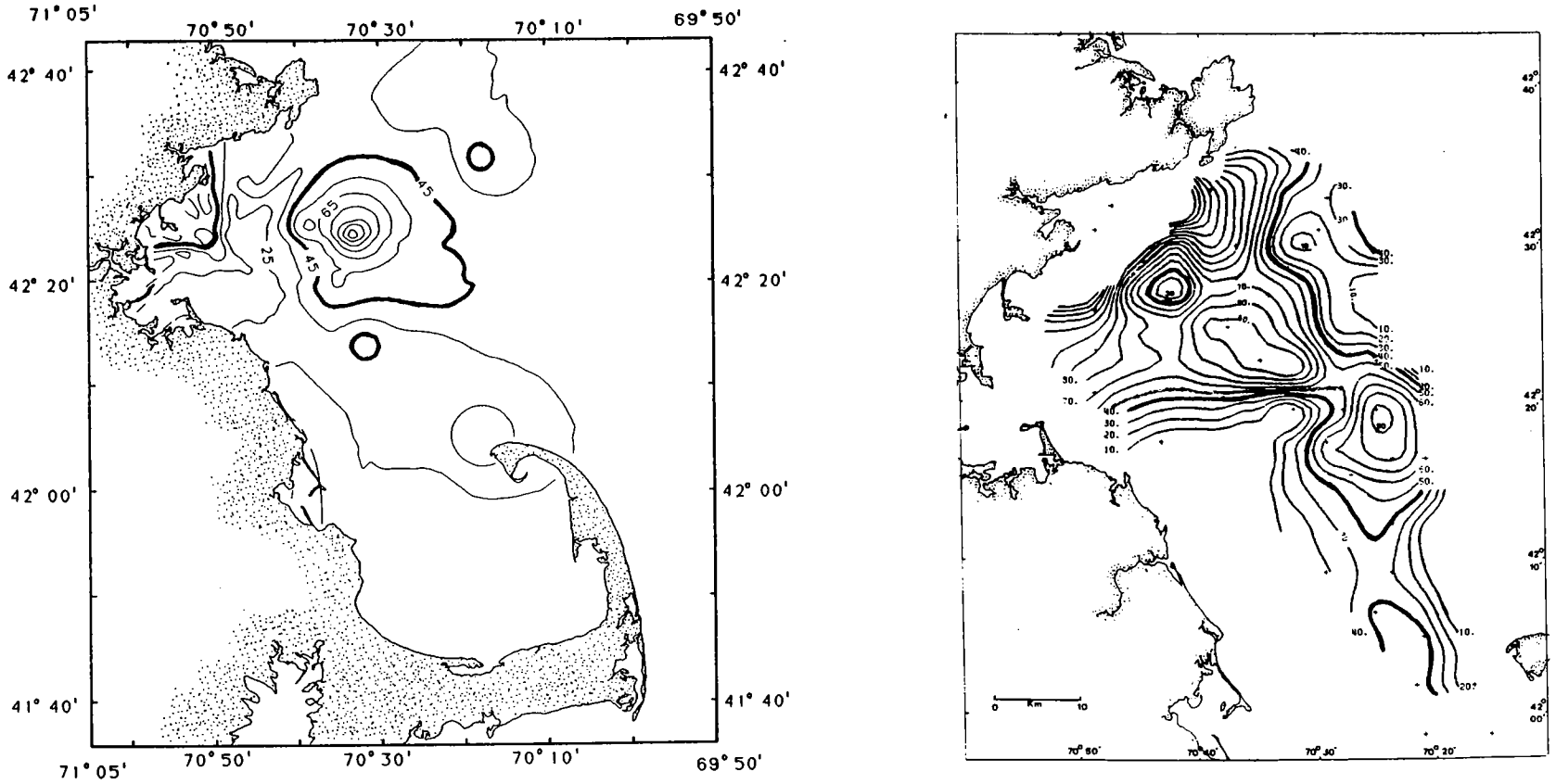

Figure 19. Contour Map of Chromium Concentration (ppm, Dry Weight). Left Map is Data From Battelle (1987b) and NOAA (Unpublished), Contour Interval is 10 ppm, Bold Line is 45 ppm. Right Map is From Gilbert *et al.* (1976), Contour Interval is 10 ppm, Bold Line is 40 ppm.

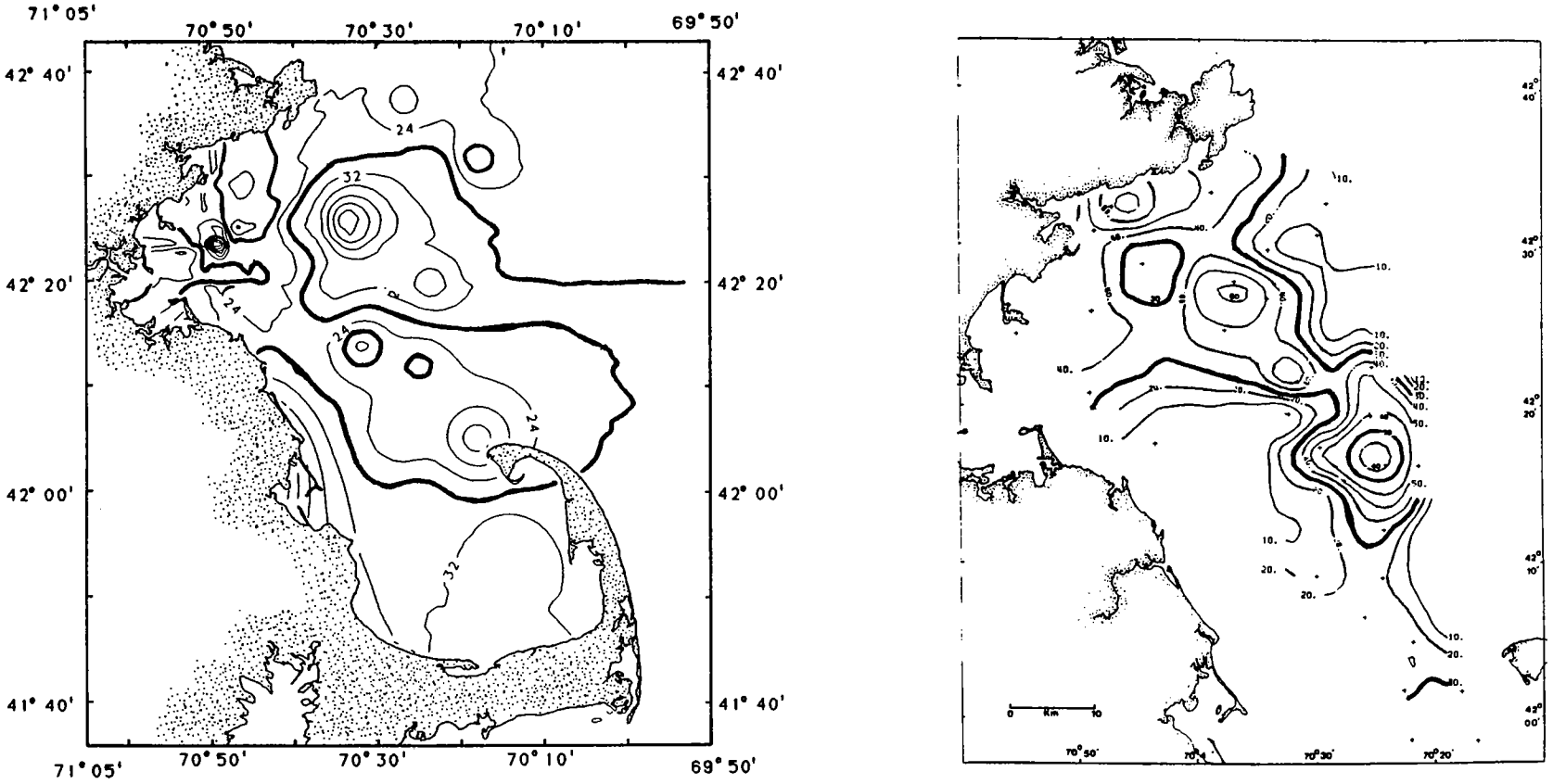

Figure 20. **Contour Map of Lead Concentration (ppm, Dry Weight). Left Map is Data From Battelle (1987b) and NOAA (Unpublished), Contour Interval is 4 ppm, Bold Line is 28 ppm. Right Map is From Gilbert *et al.* (1976), Contour Interval is 10 ppm, Bold Line is 30 ppm.**

Boehm *et al.* (1984) reported on the spatial distribution of PAH, PCB, and coprostanol throughout Massachusetts Bay with mean concentrations of 1.2 μg/g, 20.9 ng/g, and 0.13 μg/g, respectively. The PAH concentrations were significantly lower than those in Boston Harbor and uniformly distributed throughout the Bay, with a slight elevation near the MBDS. PCB concentrations were more variable, with an elevated concentration (83 ng/g) near the MBDS and slightly elevated levels (>20 ng/g) near the proposed outfall area, in Stellwagen Basin, and in the depositional area in northern Cape Cod Bay. A more recent study of PAH contamination was performed along the proposed sewage outfall tunnel as part of the Deer Island Secondary Treatment Facilities Plan (Battelle, 1987b). The sediments outside Boston Harbor had variable, but low concentrations of PAH.

To help to establish the baseline spatial distribution (and variability in Section 3.4) of organic contaminants in sediments throughout the entire Massachusetts and Cape Cod Bays region, the data from the STFP studies (Battelle, 1987b), the NOAA/NEMP survey (Boehm *et al.*, 1984), and comparable data from the MBDS studies (EPA, 1989) were combined. Contour maps of PAH, PCB, and coprostanol concentrations are shown in Figures 21, 22, and 23, respectively. In general, concentrations of PAH and PCB were highest in the finer-grained sediments (see Figure C-22), and sediments with high TOC (TOC was positively correlated with the silt-plus-clay fraction, Figure C-4). Plots of PAH and PCB concentrations versus TOC are shown in Figures 24 and 25, respectively. Data points that lie above the general trend in these plots indicate possible enrichment (accumulation) of PAH or PCB relative to the rest of Massachusetts Bay. Normalization of the PAH and PCB concentrations to TOC significantly reduced the amount of scatter (and number of outliers) in these data.

Contaminant concentrations (from Boehm *et al.*, 1984) were plotted as a function of the distance from Deer Island, Boston Harbor (see Figures C-23 and C-24). These same data are plotted normalized to coprostanol (Figure 26) and TOC (Figure 27) to visualize the influence of these parameters on the contaminant distributions. Although the purpose of this review is not to identify possible sources of or transport mechanisms for contaminants, PCBs (and to a lesser extent PAH) appear to be enriched relative to coprostanol with distance from Boston Harbor, indicating that PCB and PAH are not transported in the same manner as coprostanol or that other (nonsewage) sources of PCB and PAH become more important offshore. It is also interesting to note that, contrary to the conclusions of Windsor and Hites (1979), there appears to be a significant source of petroleum-related PAH to sites offshore in Massachusetts Bay as estimated by the fossil fuel pollution index, FFPI (Figure 28). The FFPI is a ratio of petrogenic PAH to total PAH, and can vary from about 100 (for fresh petroleum) to about zero (for combustion PAH only). Values for the FFPI are generally within 20 for sites near the proposed outfall, but sites offshore have FFPI values approaching 50.

The wide distribution of low concentrations of coprostanol (see Figure 23) in the sediments of Massachusetts Bay (Boehm *et al.*, 1984) indicates that sewage and septic inputs are found throughout the Bay. Coprostanol is a potential tracer for sewage effluent from the proposed outfall diffuser. However, there are insufficient baseline data to make future comparisons. Another potential sewage tracer, linear alkyl benzene (LAB), has not been measured adequately to establish a baseline in the sediments of Massachusetts Bay. The only data are from Gardner *et al.* (1986), who measured LAB in Salem Harbor. These data showed excellent correlation with Cr and decreased with distance from the South Essex Sewage Discharge in Salem Harbor. If either of these sewage tracers is to be monitored to assess transport and potential impacts of future sewage discharges to Massachusetts Bay, a more complete baseline would be required, along with a quantitative characterization of the sewage effluent.

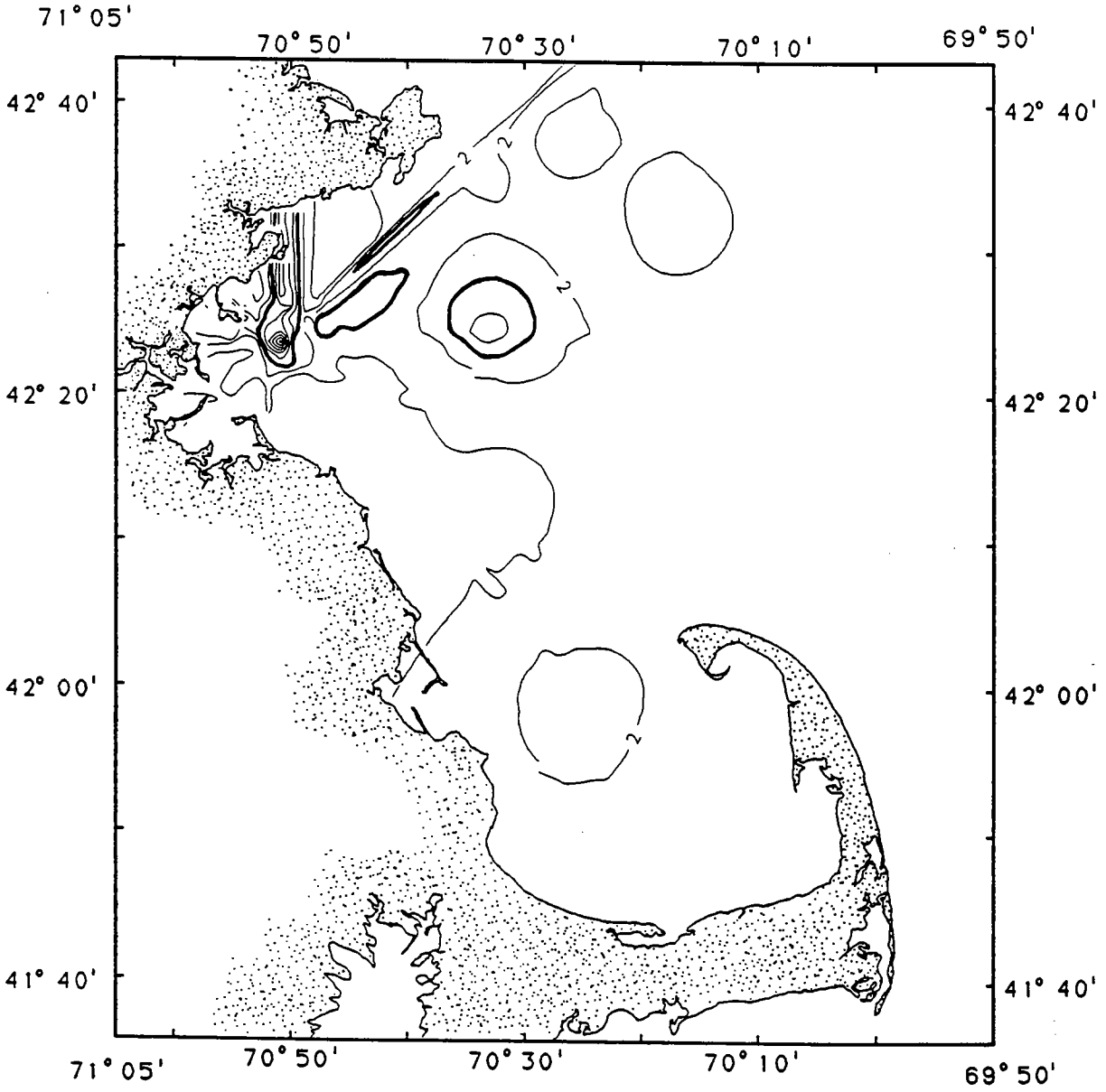

Figure 21. Contour Map of PAH Concentration (ppm, Dry Weight) in Massachusetts Bay, Contour Interval is 1 ppm, Bold Line is 3 ppm. Data are From Battelle (1987b), Boehm *et al.*, (1984), and EPA (1989).

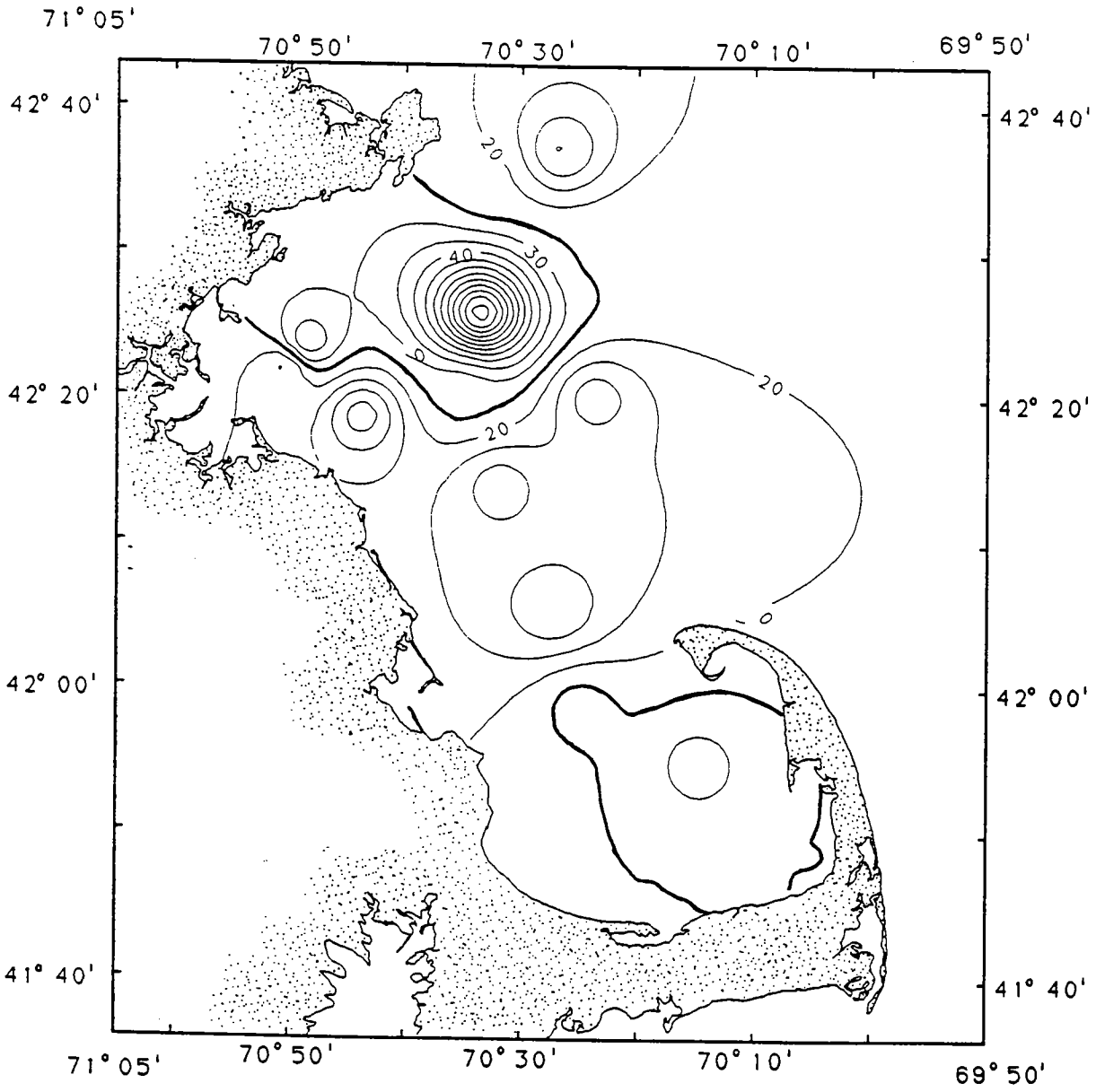

Figure 22. Contour Map of PCB Concentration (ppb, Dry Weight) in Massachusetts Bay, Contour Interval is 5 ppb, Bold Line is 25 ppb. Data are From Boehm *et al.* (1984) and EPA (1989).

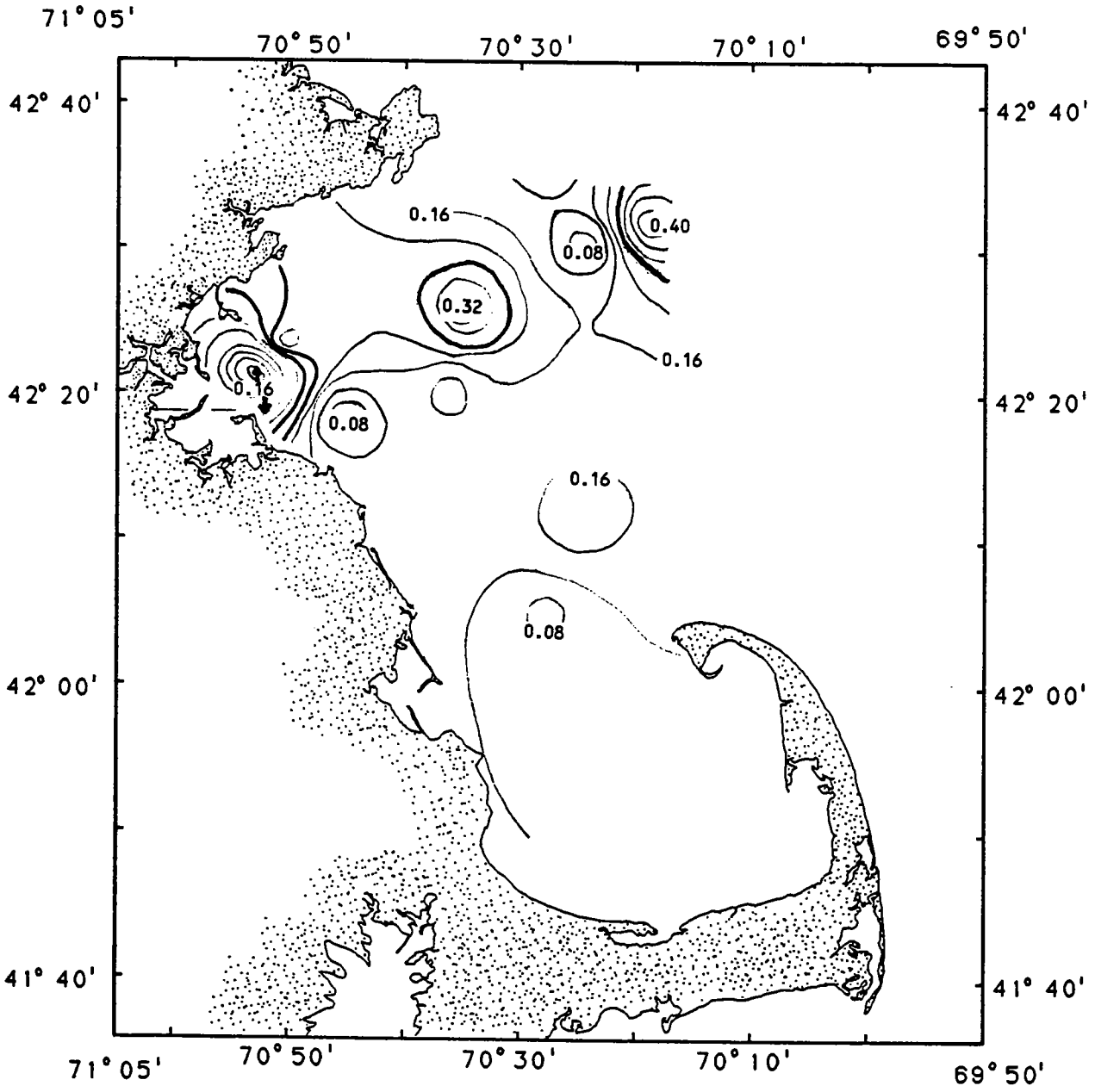

Figure 23. Contour Map of Coprostanol Concentration (ppm, Dry Weight) in Massachusetts Bay. Contour Interval is 0.04 ppm, Bold Line is 0.24 ppm. Data are From Boehm *et al.*, (1984).

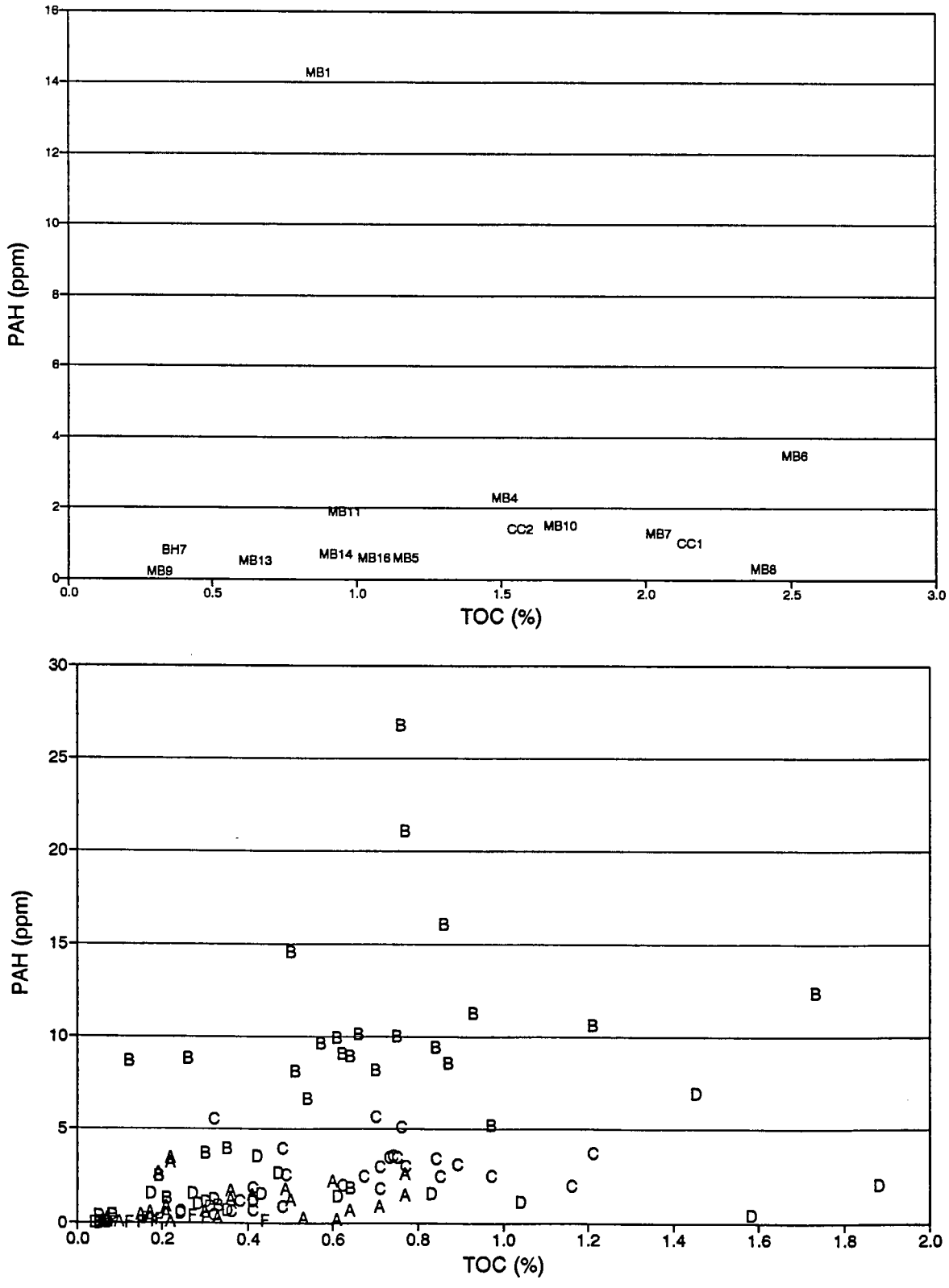

Figure 24. **Concentration of PAH vs. TOC. Top: Data From Boehm *et al.*, 1984. Bottom: Data From Battelle, 1987b; Letters Refer to Transect (See Figure C-3, Appendix C, for Location).**

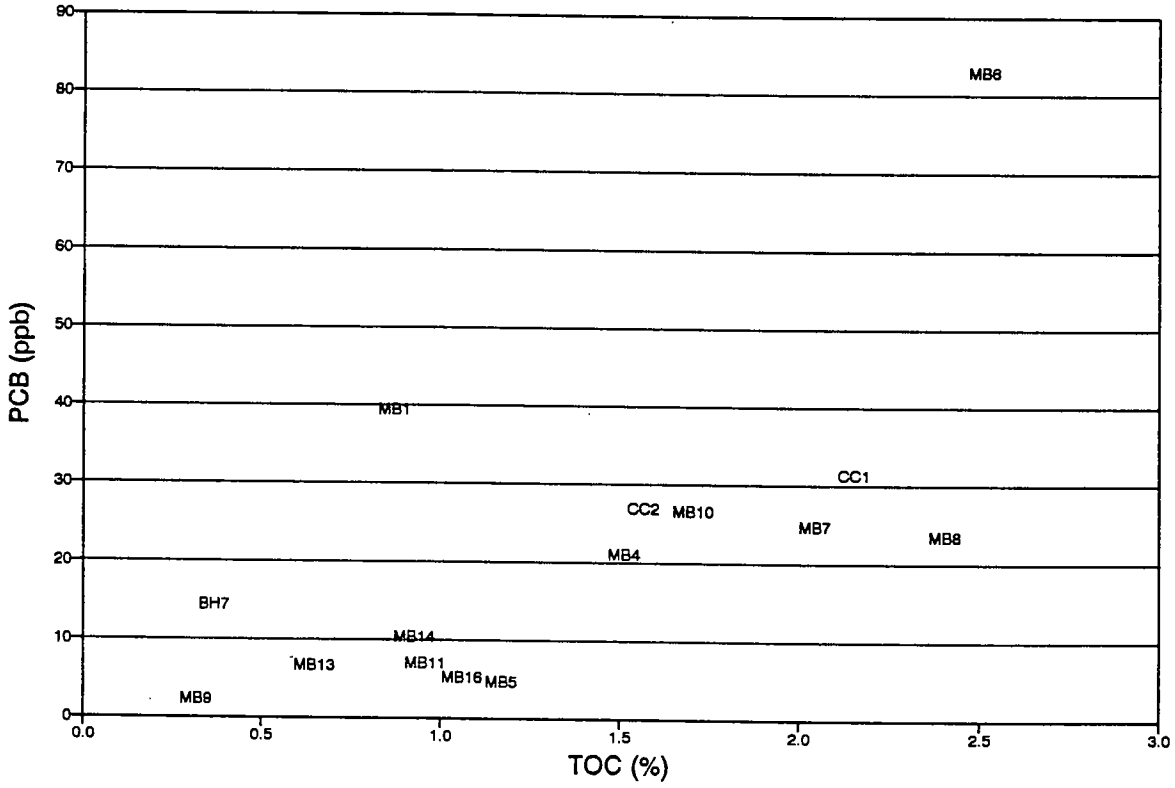

Figure 25. **Concentration of PCB vs. TOC (Data From Boehm *et al.*, 1984).**

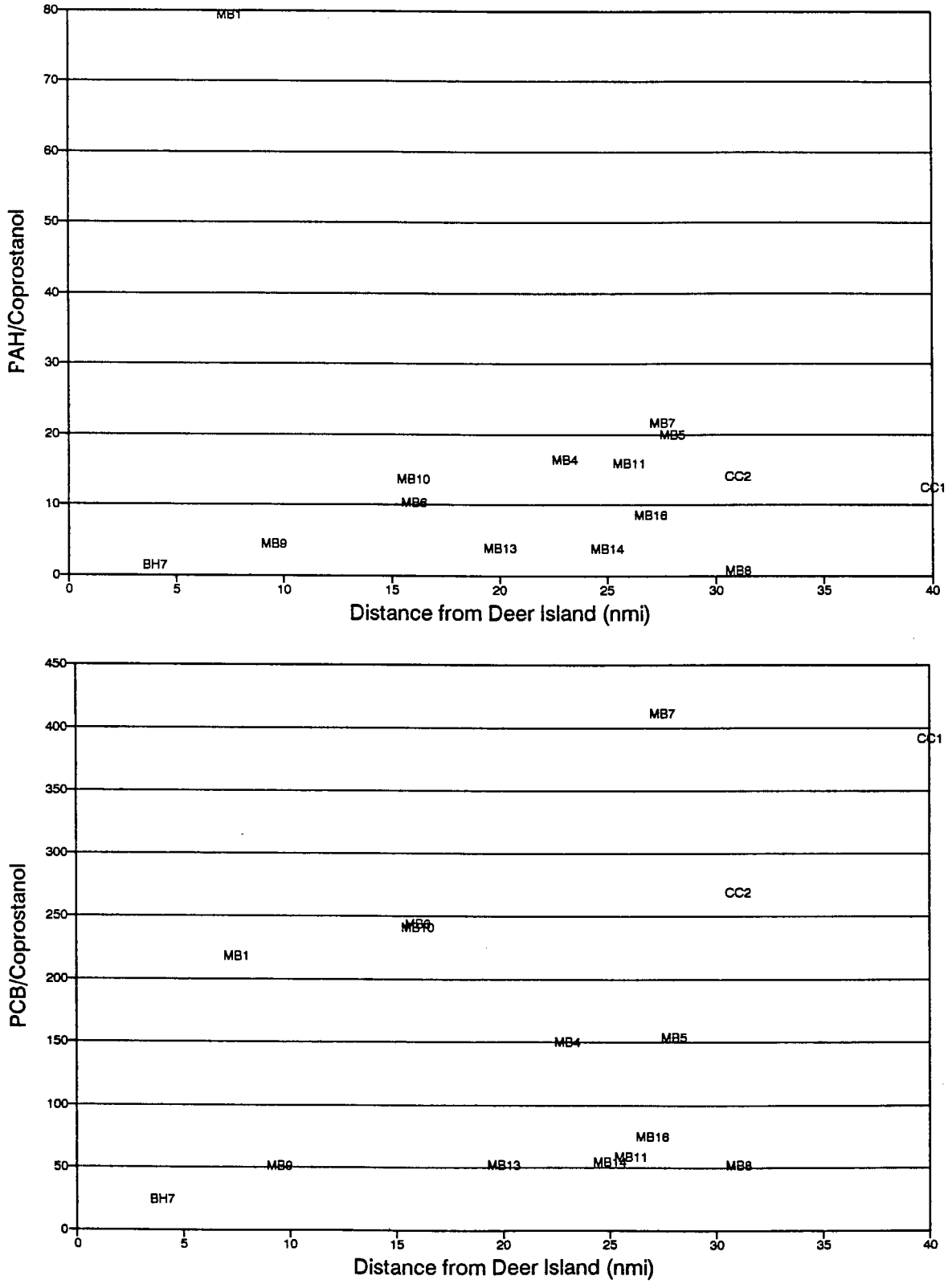

Figure 26. **Top: PAH/Coprostanol Plotted as a Function of the Distance From Deer Island, Boston Harbor. Bottom: PCB/Coprostanol Plotted as a Function of the Distance From Deer Island, Boston Harbor. All Data are From Boehm *et al.*, 1984.**

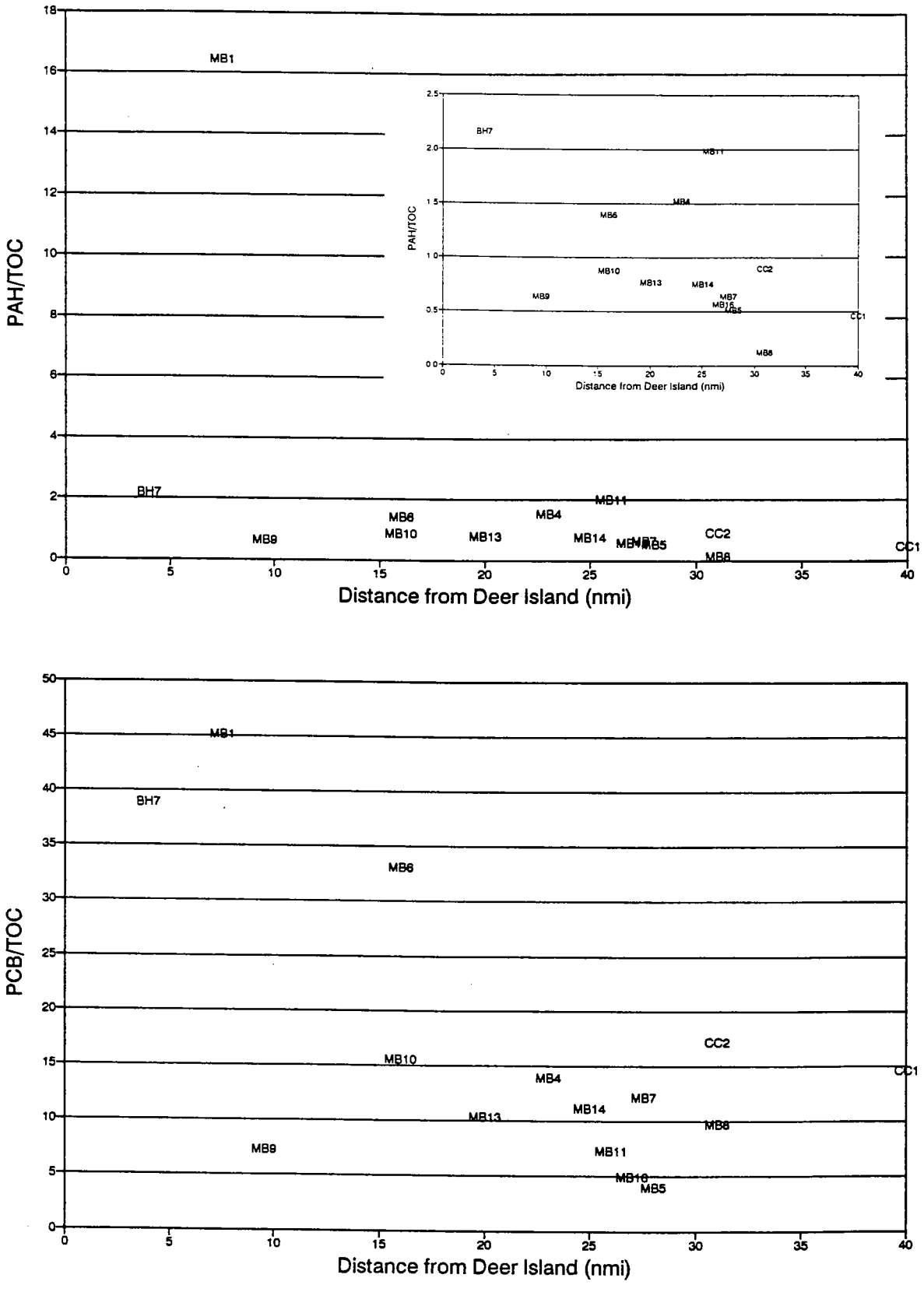

Figure 27. **Top: PAH/TOC Plotted as a Function of the Distance From Deer Island, Boston Harbor (Insert Shows Detail at Smaller Scale). Bottom: PCB/TOC Plotted as a Function of the Distance From Deer Island, Boston Harbor.**

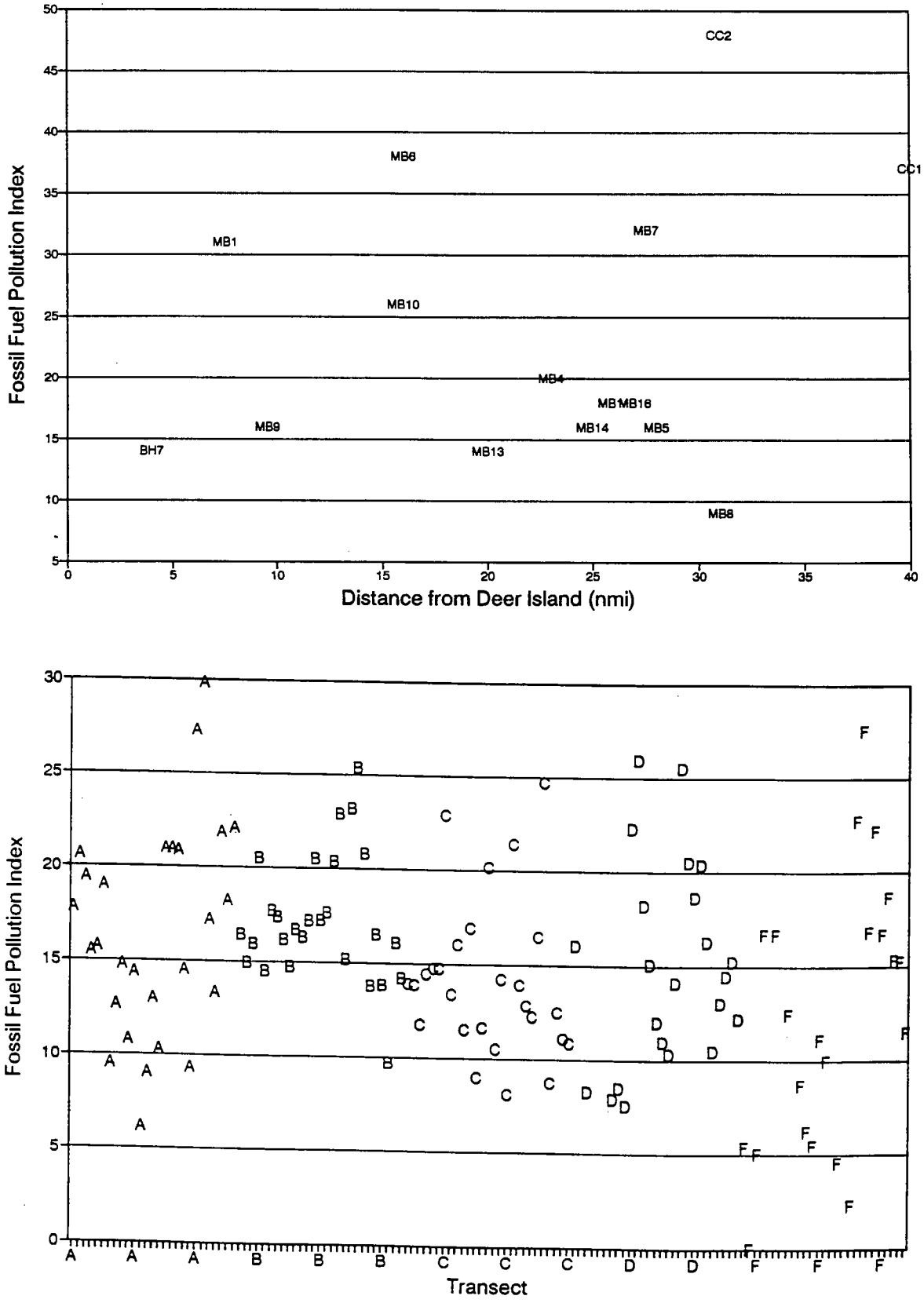

Figure 28. Top: Fossil Fuel Pollution Index Plotted as a Function of Distance From Deer Island, Boston Harbor (Data From Boehm *et al.*, 1984). Bottom: Fossil Fuel Pollution Index Plotted as a Function of Transect During Secondary Treatment Facilities Plan Surveys (Battelle, 1987b).

3.3 COMPARISON OF CONTAMINANT LEVELS TO BACKGROUND AND OTHER REGIONS

Direct comparisons of the trace-metal data presented here to background values (e.g., average crustal abundance) are not appropriate because of the different digestion methods used in each study and the variability in metal levels in natural sediments (i.e., it is difficult to establish background levels). Areas of metal enrichment can be found from the contour plots (Figures 18 through 20) and plots of metal concentrations normalized to Ni (Figures C-5 through C-21). Comparison of the observed metals levels to effects levels is given below.

Comparison of PAH and PCB data with other areas is more straightforward because of the relative comparability of the data, but these comparisons can still result in unsupported conclusions about the severity of contamination. For example, the NOAA/NEMP report on organic contaminants (Boehm *et al.*, 1984) has been widely referenced as documentation that Massachusetts Bay is a contaminated coastal area (e.g., relative to the New York Bight, an area of well documented environmental stress). However, data reported by Boehm *et al.* (1984) reveal that the New York Bight actually has a higher PAH reservoir than Massachusetts Bay (58.5 vs. 52.8 kg PAH/km^2) and both areas have concentrations of PAH similar to many other urban coastal environments (Battelle, 1990b). The estimated PCB reservoir in Massachusetts Bay is only twice that of the New York Bight (0.87 vs. 0.37 kg PCB/km^2), which is well within the uncertainty of these calculations. In addition, data from these coastal areas are from several different sources and no statistical analyses have been performed to include analytical and sampling errors (and bias) or a test for significance in differences among these areas. Considering all of this, there are no data that conclusively support the conclusion that the sediments of Massachusetts Bay are heavily contaminated or even more contaminated than those of most other urban coastal environments.

To put the concentrations of organic contaminants in Massachusetts Bay sediments in perspective, the levels of PAH and PCB are shown in Figures 29 and 30, respectively, for several locations on the east coast. The data shown are the mean plus-or-minus range for the NOAA Mussel Watch Program, 1986-1989 (Battelle, 1990b), except for the Massachusetts Bay data, which are from Boehm *et al.* (1984). If data from other studies were included, the range of values would be much larger for each location and the mean might also change significantly. Restricting the presentation to the NOAA Mussel Watch data and those of Boehm *et al.* (1984) yields an internally consistent data set, because all of the field and analytical methods were the same and the analyses even were performed by the same laboratory. Using this comparison, Massachusetts Bay sediments are no more contaminated than those of other urban estuarine and coastal regions on the east coast and appear to be much less contaminated than those of the New York Bight.

Although EPA and several States are in the process of developing sediment quality criteria (Shea, 1988), there are currently no regulatory criteria to compare against the data reviewed above. Possible effects levels have been estimated by NOAA (1990) from a compilation of six different approaches used to assess sediment toxicity. (Differences in sediment digestion/extraction or analysis methods were not incorporated into these effects values.) These "consensus" effects values have two ranges — (1) Effects Range-Low (ER-L) and (2) Effects Range-Median (ER-M) — which represent the lower 10 and 50 percentiles in the data, respectively.

Thus, if contaminant concentrations equaled the ER-L, only 10% of data reviewed by NOAA (1990) would exhibit adverse biological impact. If the concentration equaled the ER-M, 50% of the data would show adverse biological impact. Using these values, and excluding data from Boston Harbor and directly in the MBDS, very few metal concentrations observed in Massachusetts Bay sediments exceed the ER-L (the more protective range). For the metals Ag, Cd, Cr, Cu, Ni, and Zn, less than three values (out of over 100) exceed the ER-L and no values exceed the ER-M. For Hg, about 20 values exceed the ER-L, although most of these values are near the common detection limit for Hg and, therefore, are probably not accurate. No Hg values exceed the ER-M. For Pb, about 10 values exceed the ER-L, a few exceed the ER-M. Several metals exceed the ER-L (and to a lesser extent the ER-M) at the MBDS and within Boston and Salem Harbors. For PCB, the ER-L is exceeded only at the MBDS and in Boston Harbor; the ER-M is exceeded only within Boston Harbor. For PAH, the ER-L is exceeded only at sites within or very near Boston Harbor; the ER-M is exceeded at a few sites within the Harbor. In general, comparison of observed data to the NOAA effects levels indicates that the sediments in Massachusetts Bay should be relatively healthy.

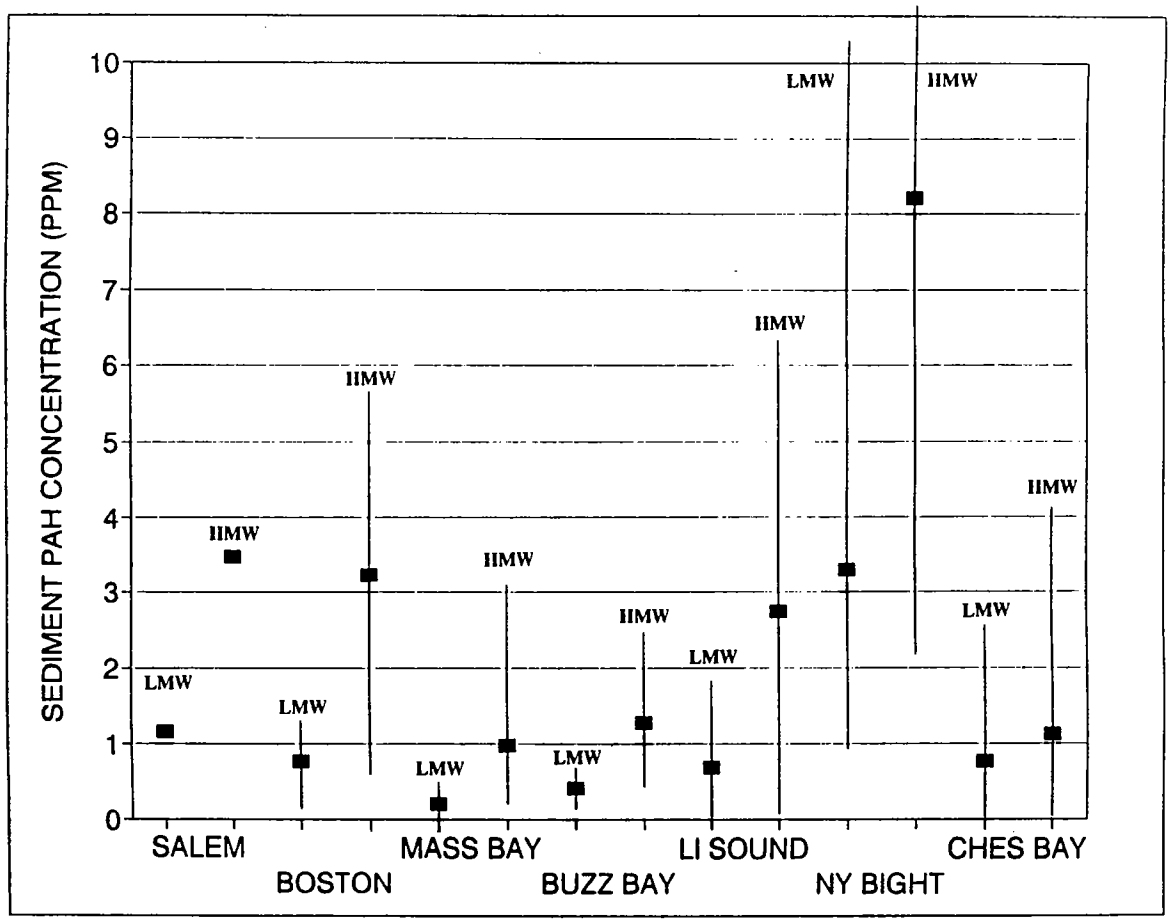

Figure 29. **Comparison of Low Molecular Weight (LMW) PAH (Petroleum Sources) and High Molecular Weight (HMW) PAH (Combustion Sources) in the Sediments of Several Coastal Areas in the Eastern U.S. The Symbol Represents the Mean Concentration for the Entire Region and the Vertical Line Represents the Range of Mean Values for Individual Stations Within the Region.**

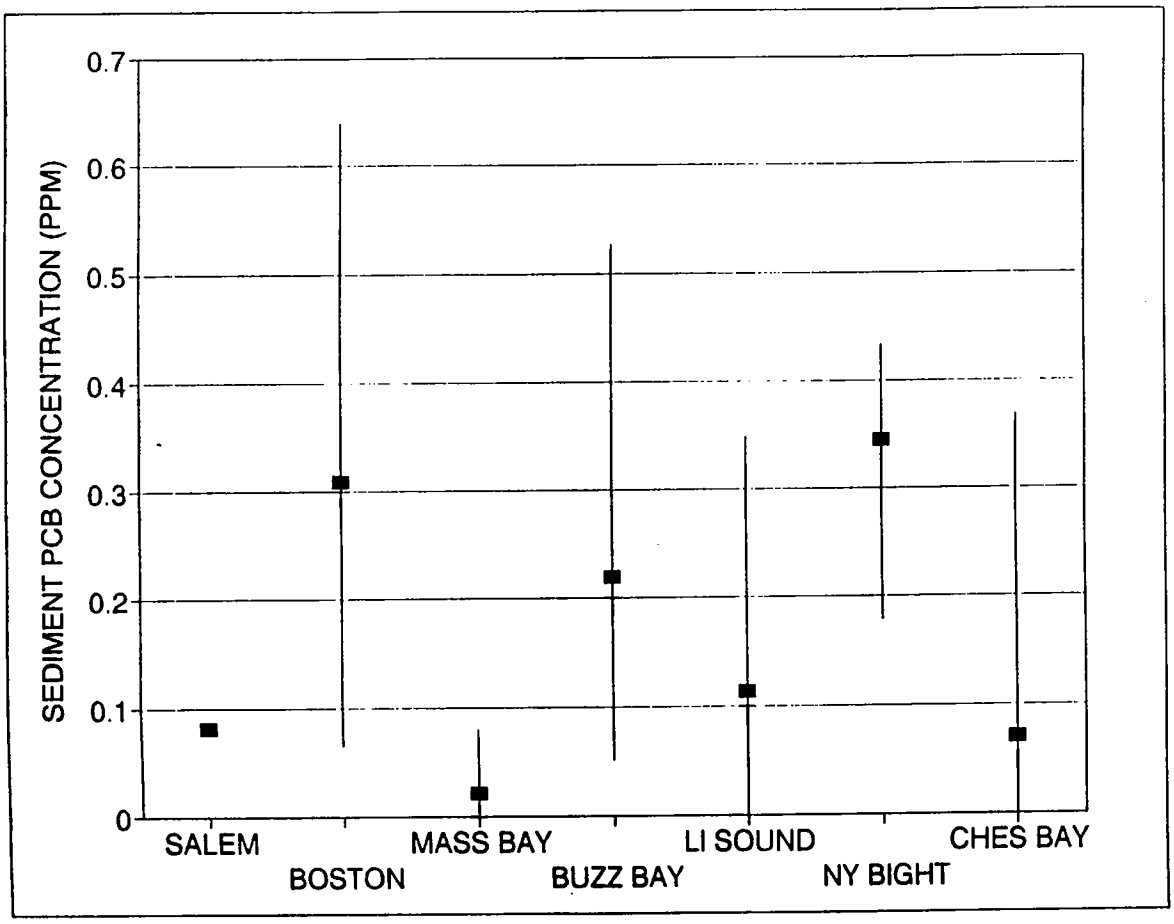

Figure 30. Comparison of PCB Concentrations in the Sediment of Several Coastal Areas in the Eastern U.S. The Symbol Represents the Mean Concentration for the Entire Region and the Vertical Line Represents the Range of Mean Values for Individual Stations Within the Region.

3.4 VARIABILITY OF SELECTED CONTAMINANT SPATIAL DISTRIBUTIONS

In this Section, the measured concentrations of chromium and total PAH have been statistically analyzed with the method of ordinary kriging (Journel and Huijbregts, 1981). This analysis addresses the following four questions concerning the spatial distribution of each of these contaminants in the sediments of Massachusetts Bay.

1. What is the statistical correlation structure of the contaminant distribution, in terms of both short-scale (e.g., within sampling stations) and large-scale (e.g., between sampling stations) variations?
2. How do the contaminant concentrations vary across Massachusetts Bay; where are the regions with highest observed concentrations?
3. How adequate is the existing network of sampling stations for providing reliable estimates of contaminant concentrations across Massachusetts Bay?
4. What recommendations can be made, based on the answers to the first three questions, for future monitoring strategies in Massachusetts Bay?

The statistical interpretation includes two data analysis steps. In the first step, the spatial correlation structure for each contaminant is assessed from the available data, using a semivariogram analysis. In the second step, the estimated contaminant concentration and associated estimation precision are calculated for each node of a dense grid of points covering Massachusetts Bay, using the kriging procedure with the available data and semivariogram model. The kriging results supply the basis for assessing the regions of high contamination and the adequacy of the existing network of sampling stations.

3.4.1 Statistical Correlation Structure

Data from the STFP (Battelle, 1987b, 1989) and NOAA/NEMP (Boehm *et al.*, 1984; NOAA, unpublished) studies were combined to provide information about the spatial variations of contaminant concentrations both at very short scales of a few hundred meters within individual sampling stations, and at larger scales of several kilometers between sampling stations. Close to Boston Harbor, we have a large number of measurements taken at a series of nearby sampling stations (see Figure C-3, Appendix C). The remainder of the data were collected from more widely dispersed sampling stations across Massachusetts Bay. It should be noted that approximately 80% of the samples were collected from nine closely spaced sampling locations within each of 15 sampling stations outside Boston Harbor (STFP study). Therefore, the assessment of spatial variations, particularly at short scale, may be biased toward the variations characteristic of this region of Massachusetts Bay.

The spatial correlation structure for each contaminant is assessed with a semivariogram analysis. The semivariogram for a particular contaminant assesses the degree to which measured concentrations taken at two different locations tend be different, as a function of the separation distance between the locations. Generally, measurements taken at two locations close together are less variable than measurements taken at two locations farther apart. That is, the semivariogram increases as the distance separating two sampling stations increases. Because the data cover both short and long distances, we were able to evaluate spatial variations at three

increasing scales, a short scale less than 0.4 km (i.e., within sampling stations), an intermediate scale 0.4 to 4 km (between nearby sampling stations), and a large scale 4 to 40 km (between more distant stations).

Chromium. The results of the semivariogram analysis for chromium are presented in Figure 31; the following points should be noted about this Figure.

- Because chromium concentrations are measured in micrograms per gram ($\mu g/g$), the semivariogram is measured in units of micrograms per gram, squared [$(\mu g/g)^2$]. All frames in this Figure show both the experimental semivariogram values calculated from the data and the semivariogram model that was fitted to those values.
- Because the data for Salem Harbor were generally more than an order of magnitude higher than those for the rest of Massachusetts Bay, they were excluded from the semivariogram analysis. Three additional high concentrations (i.e., 52.58, 121.5, and 116.8 $\mu g/g$) measured at stations MB01, D2, and 33 were also excluded because they were measured in samples having unusually high silt content. We decided that including these data in the semivariogram would unfairly bias the results toward very high levels of variability.
- Figure 31(a) shows, for all three spatial scales, the increasing level of variability as a function of separation distance. The overall variability in the chromium concentrations appears to stabilize at a level of about 630 $(\mu g/g)^2$. This overall variability appears to be attributable to variations at each of the three spatial scales.
- Less than 15% [i.e., 100 $(\mu g/g)^2$] of the total variability appears to be attributable to sources at short spatial scales less than 0.4 km within sampling stations. Figure 31(b) shows the short-scale experimental and modeled semivariograms.
- As shown in Figure 31(c), approximately 50% [i.e., 330 $(\mu g/g)^2$] of the total variability is associated with sources at intermediate scales between 0.4 and 4 km. On a percentage basis, this scale of variability between nearby sampling stations contributes the largest source of spatial variations.
- Figure 31(d) shows that approximately 30% [i.e., 200 $(\mu g/g)^2$] of the spatial variability is associated with large-scale sources of variation between distant sampling stations from 4 to 40 km apart.

Total PAH. The results of the semivariogram analysis for total PAH are presented in Figure 32. The following points regarding this figure should be noted.

- Two high concentrations (i.e., 21.09 and 26.77 $\mu g/g$) measured at stations B2B31 and B2B32 were excluded from the semivariogram analysis because they were not deemed to be characteristic of the general variability in Massachusetts Bay.
- Figure 32(a) shows the levels of variability found in the total PAH data at all three spatial scales. Note that the highest levels of variability [approximately 9 $(\mu g/g)^2$] are reached for relatively short spatial scales less than 4 km. This finding for total PAH can be contrasted with the case of chromium (Figure 31) where the highest levels of variability were reached for relatively large spatial scales of 30 km. The implications of these findings for monitoring design are discussed in a later section of this report.
- Figures 32(b) and (c) show that approximately 45% [i.e., 4 $(\mu g/g)^2$] of the total variability is attributable to sources at short spatial scales less than 0.4 km within sampling stations, whereas the remainder of the variability [i.e., 5 $(\mu g/g)^2$] can be attributed to sources at intermediate scales from 0.4 to 4 km between nearby sampling stations.

Figure 31-A. All Spatial Scales From 0.04 km to 40 km.

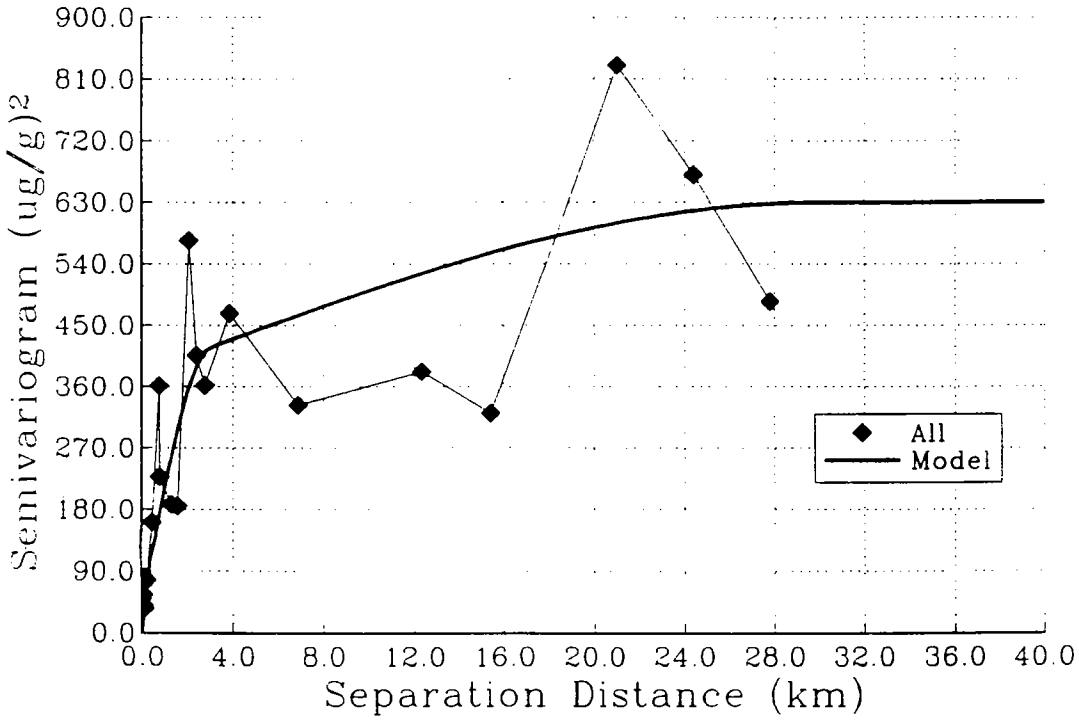

Figure 31-B. Small Spatial Scale From 0.04 km to 0.4 km.

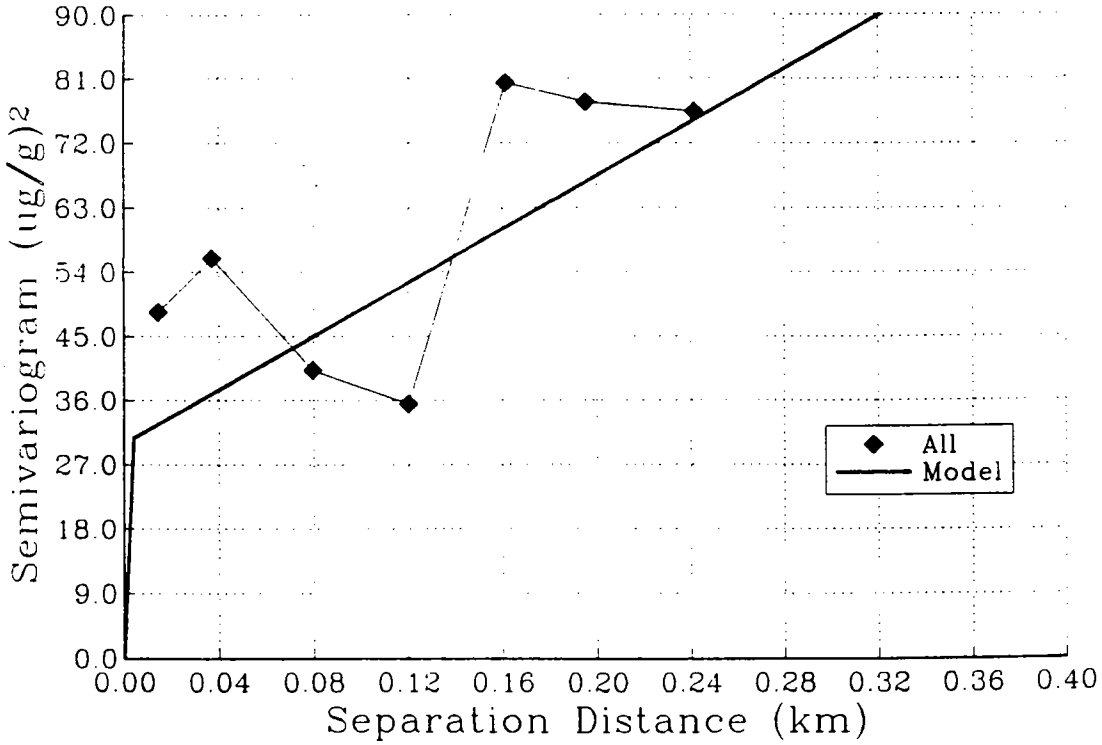

Figure 31. Semivariograms for Chromium. Points with Boxes Denote Experimental Values Calculated From the Data; Smooth Line Denotes the Model that was Fit to the Experimental Values.

Figure 31-C. Intermediate Spatial Scales From 0.4 km to 4 km.

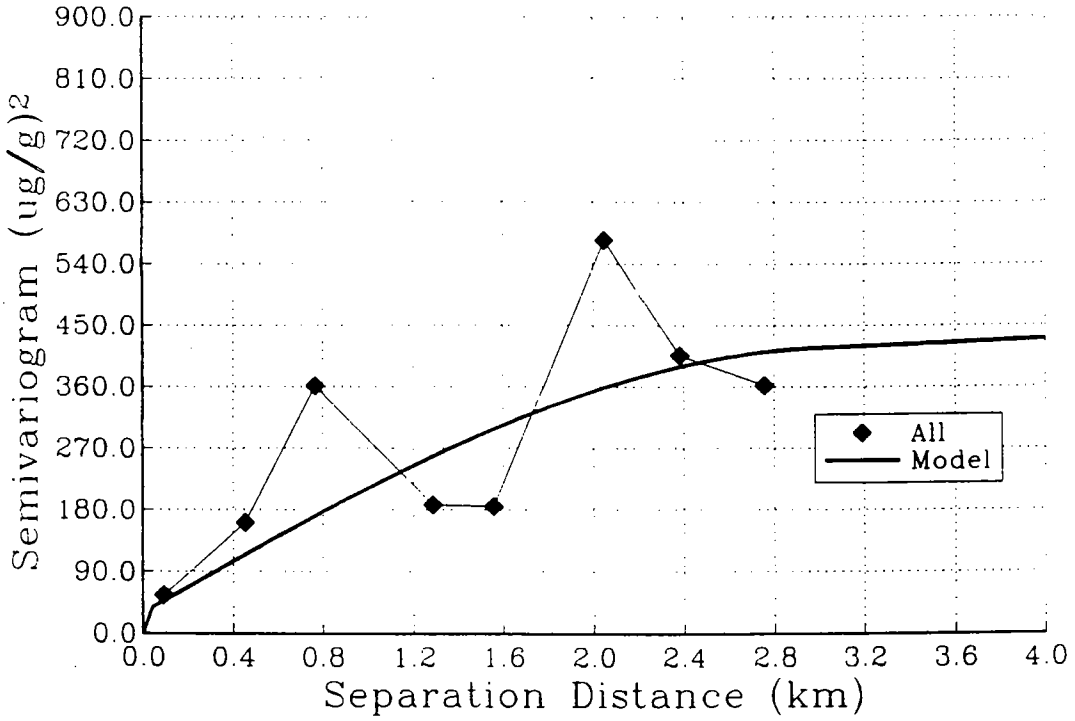

Figure 31-D. Large Spatial Scale From 4 km to 40 km.

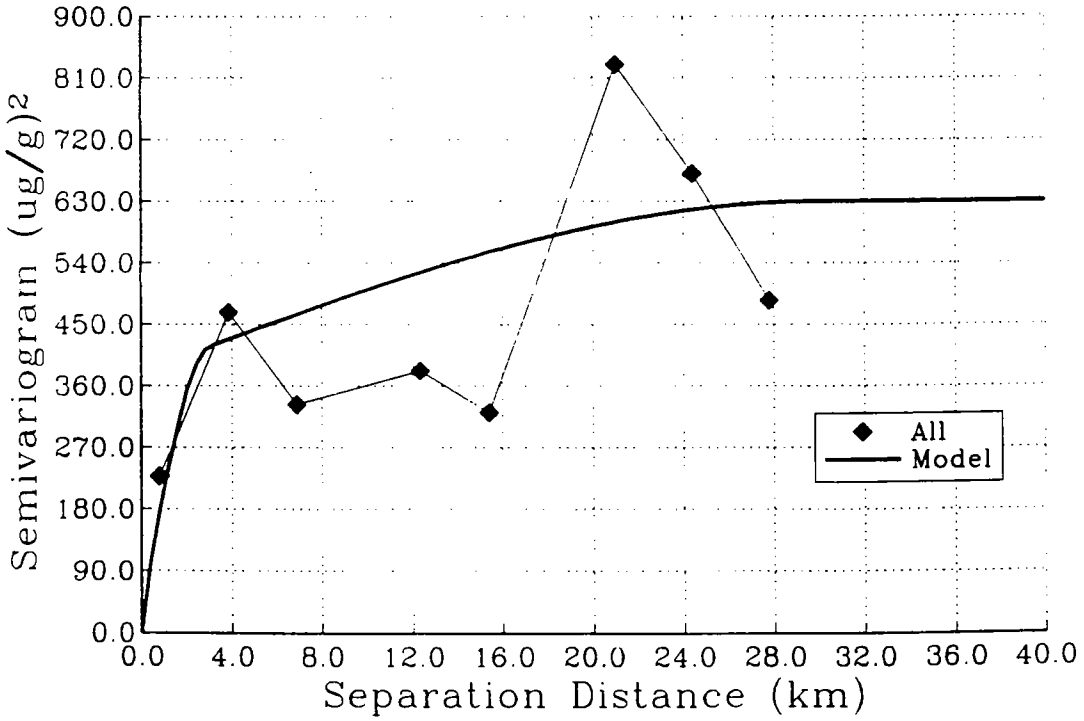

Figure 31. Semivariograms for Chromium. Points with Boxes Denote Experimental Values Calculated From the Data; Smooth Line Denotes the Model that was Fit to the Experimental Values.

Figure 32-A. All Spatial Scales From 0.04 km to 40 km.

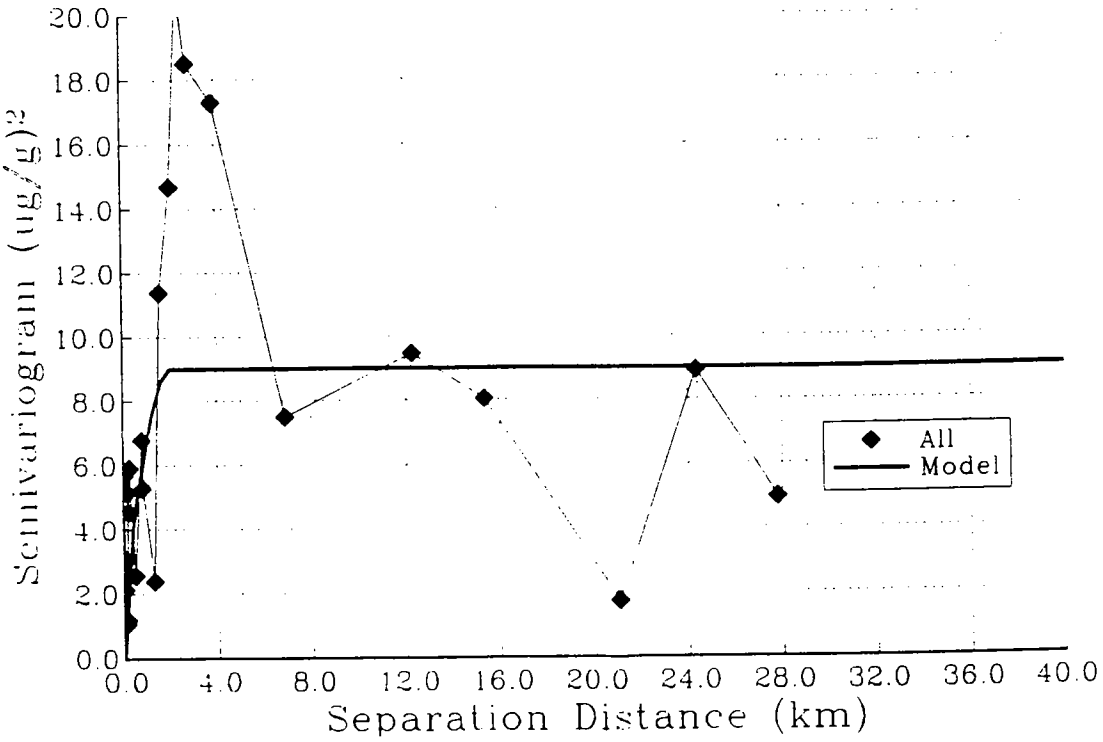

Figure 32-B. Small Spatial Scale From 0.04 to 0.4 km.

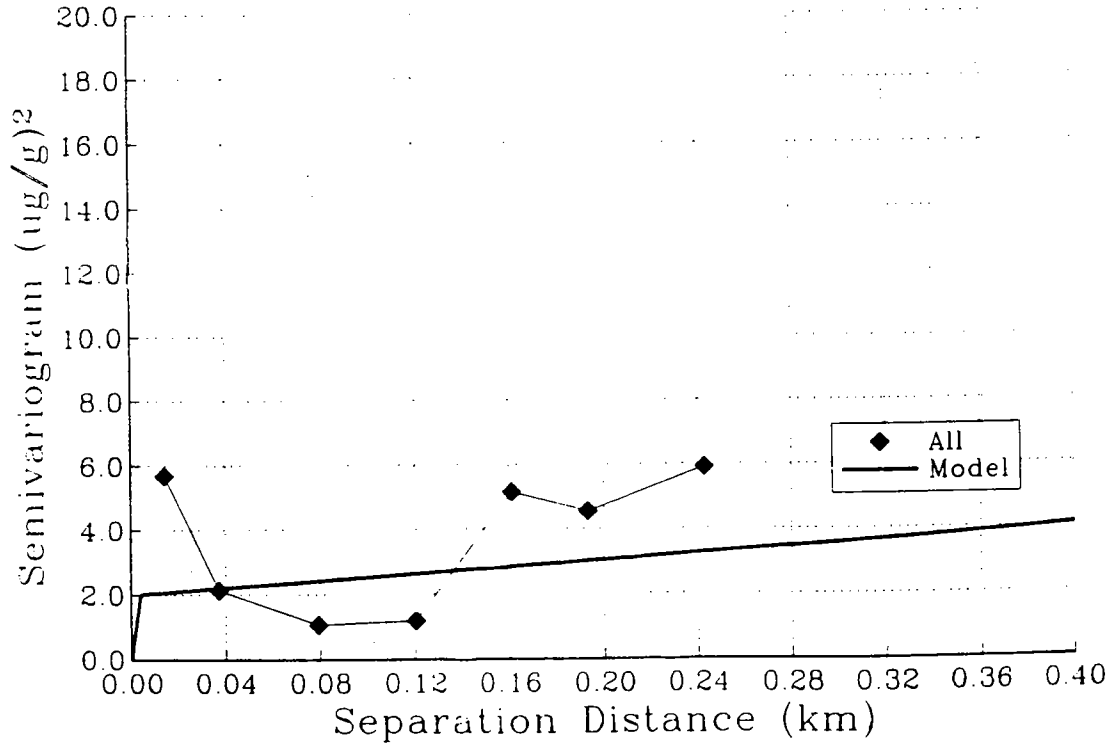

Figure 32. **Semivariograms for Total PAH. Points with Boxes Denote Experimental Values Calculated From the Data; Smooth Line Denotes the Model that was Fit to the Experimental Values.**

Figure 32-C. Intermediate Spatial Scales From 0.4 km to 4 km.

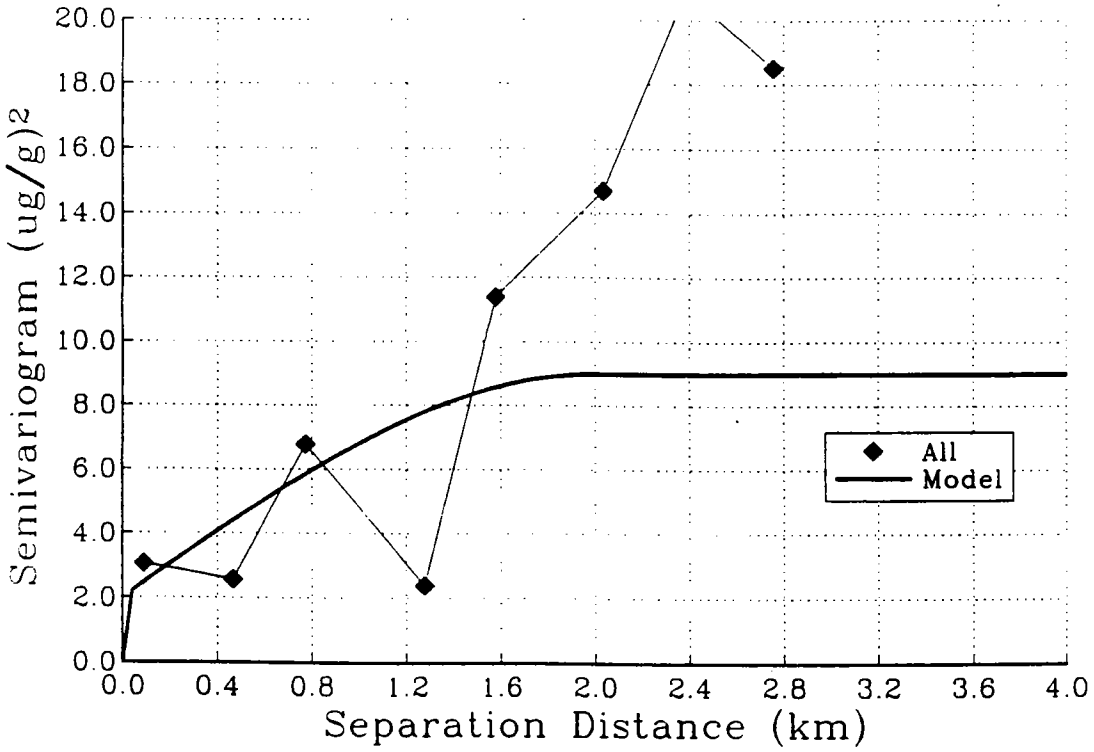

Figure 32-D. Large Spatial Scales From 4 km to 40 km.

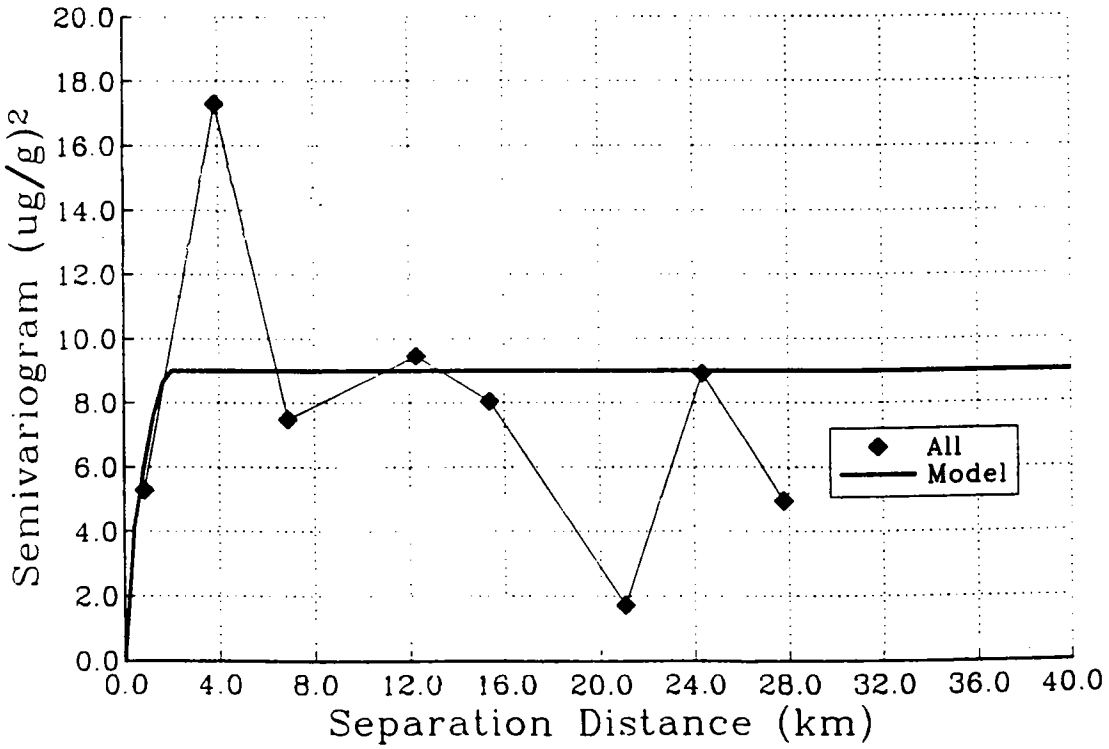

Figure 32. Semivariograms for Total PAH. Points with Boxes Denote Experimental Values Calculated From the Data; Smooth Line Denotes the Model that was Fit to the Experimental Values.

- Figures 32(c) and (d) indicate that no significant additional variability is introduced to the total PAH measurements at scales larger than 4 km. This means that total PAH levels are no more different between sampling stations located more than 4 km apart than they are between sampling stations located about 2-4 km apart.
- The rapid increase in experimental semivariogram points for separation distances between 1.6 and 2.8 km [see Figure 32(c)] is felt to be an artifact of the nonuniform sampling locations (i.e., preponderance of sampling within a small region close to Boston Harbor) rather than an accurate reflection of the entire Massachusetts Bay. Therefore, the semivariogram model fitted for total PAH more closely follows the experimental points for distances greater than 3 km.

3.4.2 Kriging Analysis

Ordinary kriging is a statistical spatial interpolation method that provides the best possible (i.e., unbiased and minimum-variance) contaminant concentration estimates for a given set of data. Kriging forms its estimates as linear combinations of the available data, where the data weights are determined by using the semivariogram model. The kriging estimates are, by design, more precise than any other linear interpolator, such as inverse-distance or nearest-neighbor estimators. Also, in contrast with all other classical spatial interpolators, the kriging method uses the semivariogram to calculate an estimation standard deviation for each of its estimates, so that the statistical reliability of the interpolation can be assessed.

In this analysis, an area was defined for kriging that extends from Boston Harbor to the tip of Cape Cod (approximately 75 km East—West) and from Cape Anne to the base of Cape Cod (approximately 80 km North—South). This area was overlain by a square 1.5-km grid of 2750 points. The kriging estimates and associated estimation standard deviations were then calculated for each point on the grid, using the available data and the semivariogram models depicted in Figures 31 and 32.

Chromium. The results of the kriging analysis for chromium are presented in Figure 33 in the form of a contour map of the estimation standard deviation, which quantifies the uncertainty associated with the estimation of chromium concentrations in the sediments of Massachusetts Bay. A Figure showing the kriging estimates of chromium concentration is not given in this section because these estimates were generally similar to those depicted earlier in Figure 19. The following points should be noted with Figures 19 and 33.

- Although several data were excluded from the spatial correlation analysis so that they would not bias the semivariogram models, all data have been included in the kriging analysis, except the Salem Harbor data, which may not provide an accurate reflection of typical chromium concentration variations for the rest of Massachusetts Bay.
- Figure 19 shows that chromium concentrations in the sediments of Massachusetts Bay range from about 25 to 100 μg/g. Typical concentrations are generally found below 35 μg/g, particularly south in Cape Cod Bay and east toward the ocean. However, the typical concentrations increase substantially to 75 μg/g or more in northwestern regions close to Boston Harbor.
- The estimation standard deviation presented in Figure 33 can be used to make statistical confidence statements about the contamination levels. For example, Figures 19 and 33 indicate that a two-sided 95% confidence interval for the chromium concentration just west of the tip of Cape Cod is approximately 50 μg/g $\pm$ 48 μg/g.

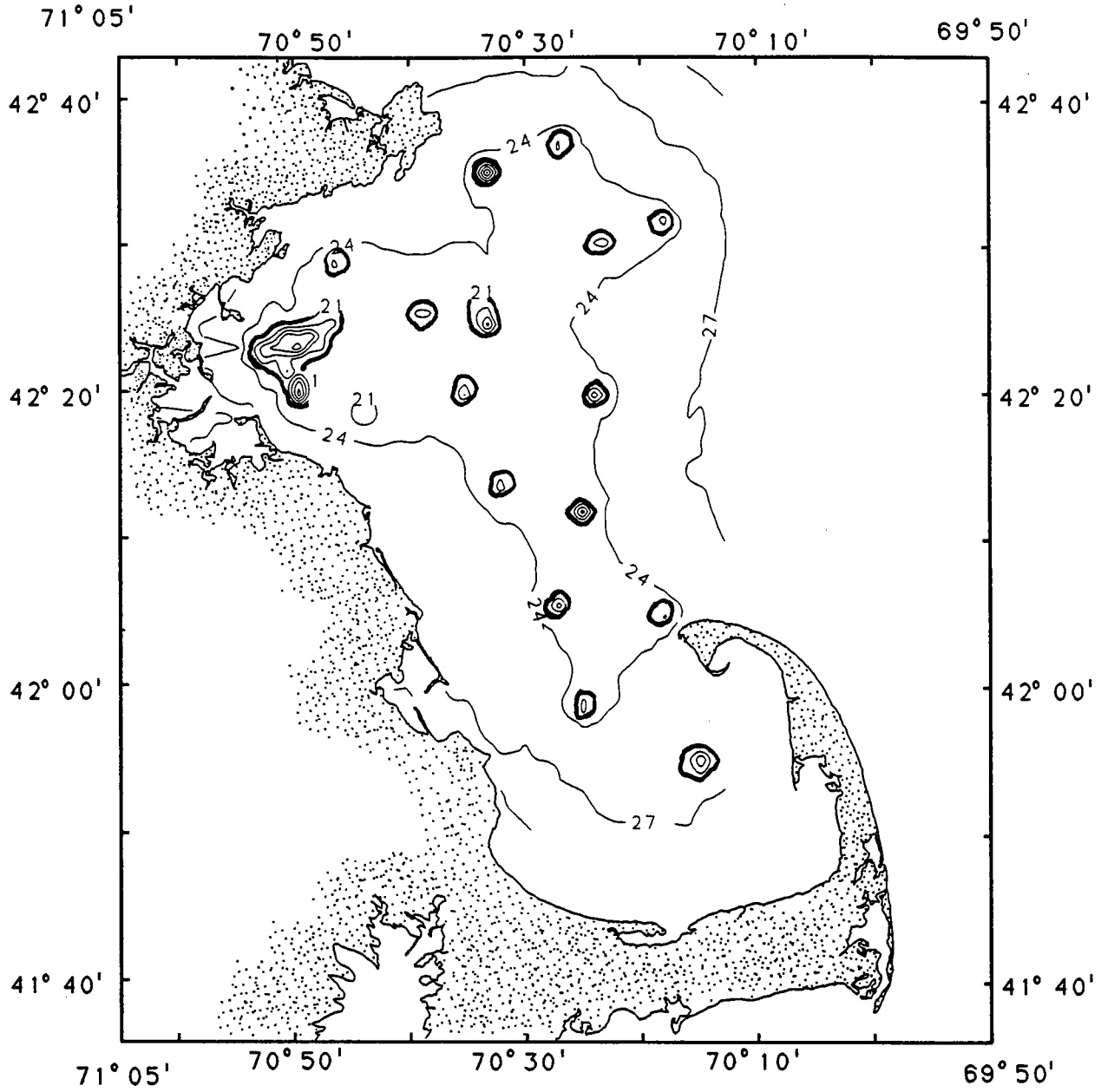

Figure 33. **Standard Deviation (ppm) for Chromium Concentration Contours (Figure 19) Estimated by Kriging. Contour Interval is 3 ppm, Bold Line is 21 ppm.**

- The magnitude of the estimation standard deviation at a particular point on the kriging grid is closely tied to the number of available data located in the immediate neighborhood of that point. Therefore, in areas, such as southern Cape Cod Bay, that are farthest from the greatest density of available data, the estimation standard deviation (i.e., estimation uncertainty) is greatest.

Total PAH. The results of the kriging analysis for total PAH are presented in Figure 34, which summarizes the estimation standard deviations associated with the estimates of total PAH concentration shown in Figure 21. The following points should be noted with these two Figures.

- As in the case of chromium discussed previously (Figures 19 and 33), all data have been included in the total PAH kriging analysis, except the Salem Harbor data.
- Figure 21 shows that total PAH concentrations in Massachusetts Bay range from 1 to 4 μg/g. Typical concentrations are generally around 1 μg/g in the eastern regions of the Bay, whereas higher concentrations around 4 μg/g are found close to Boston Harbor.
- The estimation standard deviations for total PAH, shown in Figure 34, range from about 2 to 4 μg/g. These deviations imply that the true total PAH concentrations could vary widely from 0 to about 10 μg/g. That is, the relative uncertainty associated with the total PAH estimates may be as large as 100% or more. The implications to monitoring design of this large degree of uncertainty are discussed in the following section.

3.4.3 Comments on Within-Station, Seasonal, and Year-to-Year Variability

The best data set to evaluate the observed variability within a station and among seasons is the data from the STFP surveys (Battelle, 1987b). These data indicate that within-station variability (among three replicate grabs) can be substantial, with CVs approaching 100%. However, typical within station CVs for most metals are closer to 20% and for PAH about 50%. This variability was somewhat lower at stations with high contaminant concentrations. The data from Gilbert *et al.* (1976), where only duplicate grabs were taken, exhibited even higher variability. In general, the differences in the mean contaminant concentrations among seasons was within one standard deviation of the three replicates for each season. A Student-Neuman-Keuls test for seasonal differences in each metal concentration (at each station) was performed on the STFP data (Battelle, 1987b). Out of 405 comparisons, only 35 (8.6%) showed any statistical difference at the 95% confidence level. Vanadium accounted for one-third of these, and no pattern could be found with these 35 statistically different sets of data (i.e., these differences were randomly distributed throughout the entire data set). There are insufficient data to properly evaluate year-to-year differences or trends in contaminant concentrations, although trace-metals data from 1976 (Gilbert *et al.)*, 1983 (NOAA, unpublished data), and 1987 (Battelle, 1987b) showed no apparent trend. This lack of change is supported by data from the NOAA National Status and Trends Program (e.g., Battelle, 1990) that has shown no changes or trends in contaminant concentrations in sediments over the last 6 years at sites throughout most of the United States.

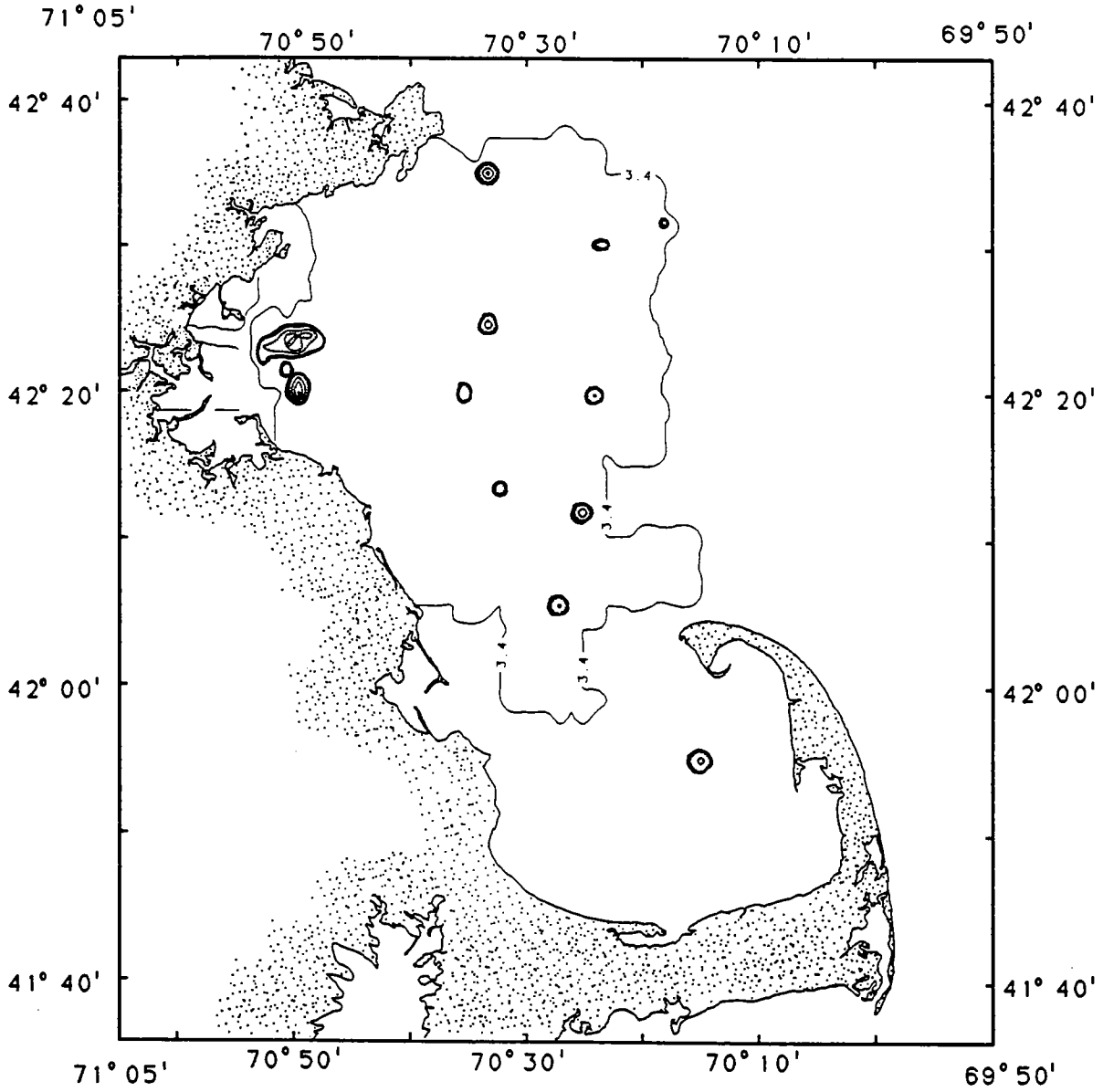

Figure 34. Standard Deviation (ppm) for Total PAH Concentration Contours (Figure 21) Estimated by Kriging. Contour Interval is 0.4 ppm, Bold Line is 3.0 ppm.

3.5 IMPLICATIONS FOR FUTURE MONITORING OF SEDIMENT CONTAMINATION

The spatial correlation and kriging analyses provide statistical assessments that can be used to help to suggest modifications to the existing monitoring network design for Massachusetts Bay. One primary objective of monitoring is to clearly identify regions with high contamination levels and track changes in those levels over time. An important statistical concern in these assessments is whether or not high estimated contamination levels are in fact accurate reflections of the true contaminant concentrations. In regions where the estimated levels are high, but the associated estimation uncertainty is also high, additional samples may be needed to help to reduce this uncertainty so that the level of contamination in these regions can be more accurately defined.

The estimated concentration maps of chromium and total PAH depicted in Figures 19 and 21 indicate potential higher levels of contamination in the region of Massachusetts Bay neighboring Boston Harbor. However, the associated estimation standard deviation maps for these two contaminants, shown in Figures 33 and 34, also indicate that the uncertainty in these estimated maps is as large as 100% or more. Therefore, the true contaminant concentrations may be significantly lower or higher than the estimated levels shown in Figures 19 and 21. Additional samples would be required to reduce the estimation uncertainty and provide a more even coverage across this region.

If additional sampling stations are to be included in the monitoring network, the obvious next question to be considered is How many additional stations and samples are needed and how should they be located in the region? A complete answer to this question is not be provided in this report because this answer must consider several factors for which we do not yet have detailed information, such as the specific pollutant levels that are of regulatory or environmental concern and sampling and analysis costs. In addition, the appropriate monitoring network design must consider the amount of variability present in the contaminant concentrations at short spatial scales within sampling stations and at large spatial scales between stations. The spatial correlation analysis discussed earlier does provide information on these different levels of variability. For chromium (see Figure 31), measurements taken close together within sampling stations show relatively less variability than measurements taken far apart between sampling stations. In contrast, measurements of total PAH taken close together show relatively high levels of variability that are similar to those for measurements taken far apart between sampling stations. These results suggest that replicate samples taken within sampling stations are more valuable for reducing estimation uncertainty in the case of total PAH than they are in the case of chromium. Therefore, the modified monitoring design for total PAH should emphasize replicate sampling within stations more than the modified design for chromium. Of course, any modified design must consider the statistical tradeoffs and environmental importance of all contaminants that are to be monitored.

Using the data from the STFP study (Battelle, 1987b), it appears that seasonal variations in contaminant levels are small as compared to the variability within a station at a single point in time. In addition, data from the NOAA Mussel Watch Program indicate that, in general, trends in sediment contaminant levels cannot be detected over a few years at most sites around the United States, including Boston Harbor and Massachusetts Bay. Therefore, future monitoring of contaminants in sediments does not require frequent sampling unless there

is reason to expect rapid change in the flux of contaminants to a particular area, and sampling more than once per year is certainly not warranted.

The within-station (and within-season) variability in the data from the STFP study (Battelle, 1987b) indicates that, in general, triplicate sediment grabs will allow detection of about a 100% change in metal concentrations at the 95% confidence level. For PAH at all concentration levels, only a 500% change will be detected. For PAH, at levels significantly above the detection limit, a 300% change will be detected. These estimates should be used as a rough guideline only. More precise estimates of the number of samples required to detect certain levels of change would require a more rigorous statistical analysis (see above).

The two common sediment digestion methods for metals analysis currently being used in National and Regional monitoring programs (and being considered for the MWRA monitoring program) are the total digestion and the 2 N HCl cold leach. The total digestion yields the most reproducible data, allows normalization to aluminum to facilitate comparison among sites and sampling events, and is the method used in the NOAA National Status and Trends Program. The 2 N nitric acid leach has shown high correlation to toxic effects and is currently being tested by the EPA as the measurement to use in comparison with proposed sediment quality criteria. The baseline spatial distribution and variability for metal concentrations presented above was based on data obtained by using a partial sediment digestion procedure that provided an intermediate recovery. Comparison of this baseline to total and/or mild acid leach digestion methods is not appropriate. This is only one example of the problems encountered when different field and analytical methods are used in monitoring programs. Future monitoring programs in Massachusetts Bay should evaluate the utility and purpose of various methods and implement the most appropriate methods consistently.

4.0 SUMMARY AND CONCLUSIONS

Based on this review of the sedimentary, biological, and chemical conditions in Massachusetts and Cape Cod Bays, several conclusions can be made about the status of this environment and the implications for future monitoring.

Bottom Topography

The sediment facies map reflects an extremely heterogeneous bottom dominated by reworked glacial morainal deposits. These reworked sands, cobbles, and gravel deposits probably represent areas of erosion or no net sedimentation. However, in the summer, the surface of these coarse-grained sediments is covered with a thin veneer of biologically deposited organic mud. In winter (or late fall), this biogenic mud is dislodged and carried away from rocks and sandy areas. The sandy sediment becomes mobile, manifested by ripples formed by bed-load transport. The biologically deposited muds appear to result from the development of dense populations of polychaetes and/or amphipods.

Benthic Biology

Spionid polychaetes are known to establish very large populations, sometimes forming tube mats (Blake, 1971). *Spio limicola* is undoubtedly one of the species responsible for the tube mats reported from Massachusetts Bay. The species is widely distributed in Massachusetts and Cape Cod Bays and has been reported to sometimes occur in very dense populations [e.g., 72,000/m^2 at Station PD in the 1982 301(h) survey]. Sebens *et al.* (1987), on a survey of hard-bottom communities for the STFP, identified a "complex" of amphipod and polychaete tubes covering rocky substrates in Massachusetts Bay. This complex corresponds to the tube mats observed by REMOTS and as part of the ROV surveys (Etter *et al.*, 1987). The species comprising this tube complex has not been identified, but consisted mostly of a mixture of amphipods and polychaetes. The episodic appearance of dense populations of benthic invertebrates that form mats of tubes is now well documented, but the processes that govern it are poorly known.

Three major benthic assemblages were identified by cluster analysis of the March 1987 STFP samples. These assemblages can be shown to be correlated with the sediment facies that were identified as part of the REMOTS survey conducted one month earlier. The results of the benthic analysis and the REMOTS sediment data are shown in Figure 35. The largest cluster includes transects A and B, Stations C1 and C3, and Station F4. These stations correspond more closely with high mud (silt plus clay) content according to the REMOTS data. In contrast, the cluster that includes Station C2, Transect D, and Station F1 encompasses sediments with high sand content. The fauna associated with these two assemblages clearly reflects their preferences for mud, sand, or mixed sediments. For example, Stations on the D transect are characterized by having a high number of syllid polychaetes of the genus *Exogone*. These polychaetes are the dominant species on Georges Bank, an environment where the sand content of the sediments is high (Maciolek-Blake *et al.*, 1985). In contrast, high numbers of paraonid and spionid polychaetes in the first assemblage reflect organisms with preference for fine sediments. The third assemblage includes only Station F2, a site that has unusual chemistry and infaunal components (see below).

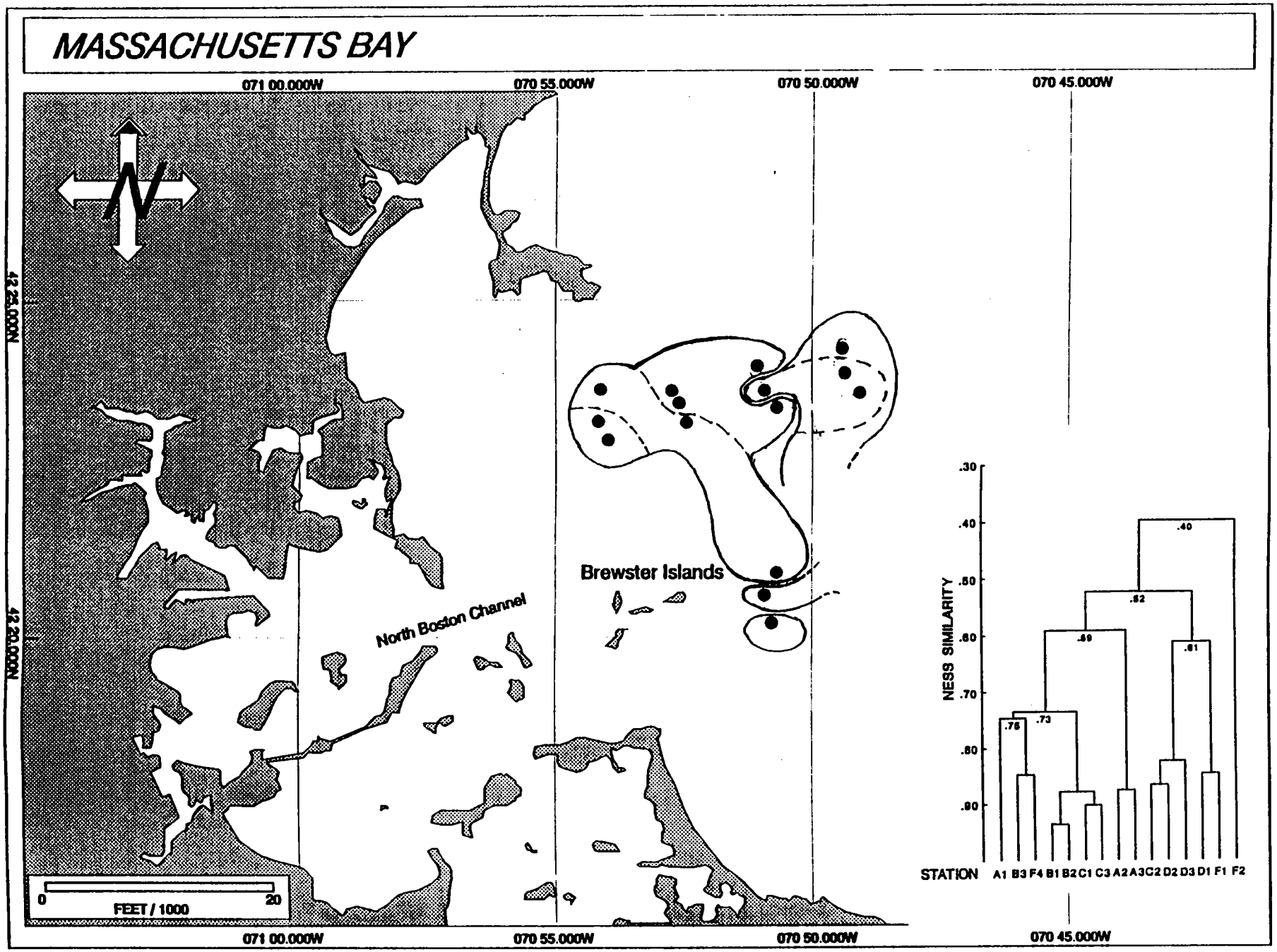

Figure 35. **Map of Benthic Stations Sampled as Part of the STFP in March 1987. Stations are grouped according to major clusters defined by NESS dendrogram in lower right.**

The sediments at Station F2 are high in silt plus clay, but also have been shown to have elevated levels of arsenic (30.3 μg/g). The dominant species at Station F2 include four species of cirratulid polychaetes: *Tharyx acutus, Chaetozone setosa, Monticellina baptisteae,* and *Caulleriella* sp. B. High positive correlation with arsenic was found with the first (r_s = 0.908) and fourth (r_s = 0.864) species. This result is noteworthy because *Tharyx marioni* has been determined by Gibbs *et al.* (1983) in the United Kingdom to be able to concentrate high levels of arsenic (>2000 μg/g dry weight) in environments having low ambient arsenic concentrations. At present, we do not know if the high arsenic levels recorded from Station F2 are due to metal binding with the sediments, or whether the residues of the polychaete tissues contained the arsenic. There are currently no data available on body-burden concentrations of arsenic or any other metal in the local cirratulid species.

In an effort to determine if some of the dominant infauna were correlated with metals and other contaminants that are bound with fine sediments, six polychaete species known from the results of Blake *et al.* (1987) to have a high silt correlation were tested against concentrations of chromium, copper, lead, and total PAH using Spearman's rank correlation coefficient. The species that were tested and their r_s correlations with silt were *Ninoe nigripes* (0.905), *Leitoscoloplos acutus* (0.883), *Prionospio steenstrupi* (0.613), *Spio limicola* (0.506), *Owenia fusiformis* (0.643), and *Mediomastus californiensis* (0.718). These results (presented in Table 3) indicate some positive correlations with chromium and copper, but none with lead and total PAHs. For chromium, relatively high r_s was found for *N. nigripes* (0.834), *L. acutus* (0.770), and *M. californiensis* (0.731). *Ninoe nigripes* exhibited a relatively high correlation with copper (0.707). These results do not indicate any modification of the distribution of individual species of benthic organisms associated with contaminant concentrations because each species is widely distributed in areas encompassing a wide range of contaminant levels.

Oligochaetes may also play an important role in the uptake of contaminants. An intriguing study by Klerks and Levinton (1989) suggests that oligochaetes are able to adapt genetically to elevated levels of metals in their environment. Klerks and Levinton studied a freshwater species that inhabited sites contaminated by metals originally derived from mills. They found that these populations had evolved a tolerance to the metals that control populations lacked. At present, there is no information of genetic adaptation to pollution in oligochaetes or other infaunal species that might inhabit dredged-material disposal sites.

The infaunal density data developed by Gilbert *et al.* (1976) using 0.5-m^2 screens, the 301(h) studies, and the STFP studies compare favorably, which suggests that densities of 20-40,000 individuals per square meter are common in the summer months. The most important differences that have been noted among these different studies has been with dominant species. Most stations are invariably dominated by a spionid or capitellid polychaete. While the capitellid is usually the same species (*Mediomastus californiensis*), the spionids change. *Spio limicola* was the highest-ranked species in Gilbert *et al.* (1976) and the 301(h) programs. In the STFP program, however, *S. limicola* was not abundant, and was replaced as a top dominant by *Prionospio steenstrupi.* The dominance of *P. steenstrupi* and reduced importance of *S. limicola* persisted through four seasonal samplings in 1987 and 1988. These year-to-year differences in spionid polychaete dominance cannot be explained with existing data because there is little known about the reproduction, larval development, larval

Table 3. **Correlations Between Contaminants and Benthic Infauna Using Spearman's Rank Coefficient.**

Species	Chromium	Copper	Lead	Total PAH
Ninoe nigripes	0.834	0.707	0.009	0.557
Leitoscoloplos acutus	0.770	0.526	0.064	0.625
Prionospio steenstrupi	0.646	0.624	0.126	0.386
Spio limicola	0.430	0.347	0.225	0.263
Owenia fusiformis	0.185	0.043	-0.040	-0.049
Mediomastus californiensis	0.731	0.615	0.024	0.410

settlement requirements, and tube building requirements of these species. In the 1987 STFP studies, recruitment data suggested that *P. steenstrupi* might recruit in the winter, whereas *S. limicola* exhibited heavy recruitment in March and May. Differences in the timing of reproduction, production of larvae, and larval settlement may be linked to seasonal differences in climate, rainfall, nutrient availability, phytoplankton production, or a combination of these or other factors. A precise determination of conditions that would dictate whether *S. limicola* or *P. steenstrupi* were the dominant spionid has yet to be determined.

Areas of Sediment Deposition and Contaminant Accumulation

Two major areas near the proposed outfall have been identified as long-term depositional sites for fine-grained sediment. These areas have been labeled as Low Kinetic Area (LKA) No. 1 southeast of Nahant and LKA No. 2 located near the center of the surveyed area. Both of these depositional sites are located in local topographic depressions (>25 m). LKA No. 1 and LKA No. 2 are nearfield candidate sites for receiving biogenically deposited mud once the tube mats or tube complexes are eroded from the coarse-grained facies or rocks in the winter. There are no contaminant data corresponding to LKA No.1, but stations surrounding this area have elevated levels of metals and PCB. There are insufficient PAH data in this area to determine if they are accumulating in LKA No.1. LKA No. 2 is just beyond the midsection of the proposed outfall tunnel, an area that exhibited a small maxima for several metals and PAH, and fine-grained sediment.

An area located to the northeast of North Boston Channel has been identified by REMOTS as a potential site for benthic enrichment and has elevated levels of metals and possibly PAH. Stations in this area have shallow biological mixing depths, and three winter stations show that only Stage I seres are present. Organism/sediment indices (OSI) in this region were also found to be low during the winter survey.

The reason that the area at the northeastern end of North Boston Channel appears to be experiencing high rates of organic loading may be related to a hydraulic drop in this region. On the ebb flow, sewage entrained in the channel is carried out of the Harbor by high-velocity flow. The bottom of North Boston channel is scoured by this flow and no fine-grained material is deposited in this channel. As the water moves out of the northeastern end of the channel, it is no longer confined and the mean ebb flow velocity drops in this area. This hydraulic drop may be the key mechanism for depositing sewage particulates in this area.

The distribution of PAH and metals in the vicinity of the outfall area were similar. The highest concentrations of PAH and metals were found between 4 and 6 nmi from Deer Island, near the midsection of the proposed outfall tunnel. The data indicated that enrichment of contaminants was occurring in this region, and sediment grain-size data provided strong evidence that this is a depositional area. Possible areas of contaminant accumulation that are farther away from the proposed outfall include the MBDS, the deeper regions of Stellwagen Basin, and a depositional area in Cape Cod Bay. Identification of the ultimate areas of deposition would require additional information on water circulation within the Bays, rates of sedimentation and sediment resuspension, and suspended-particle transport mechanisms.

General Health of Massachusetts Bay Sediments

Aside from the area to the northeast of North Boston Channel and the immediate area of the Deer Island Outfall, the surveyed region is generally of high benthic habitat quality as defined by the REMOTS OSI and traditional benthic taxonomy. The sediment contaminant data do not have sufficient resolution in these regions to make direct comparison to the biological data. However, comparison of the levels of contaminants with data from other regions in the northeast and to NOAA biological effects levels indicates that the only areas where contaminant levels might be of concern are the MBDS and areas very near Boston and Salem Harbors. Overall, the sedimentary environment in Massachusetts Bay is quite healthy.

Monitoring Implications

Future monitoring stations should include the area near the northeast end of North Boston Channel; with the placement of the new offshore diffuser, the sediment quality of this area is expected to improve. Additional future monitoring stations should be located in LKA No.1 and No.2. If the area of the new diffuser experiences organic enrichment, enhanced benthic productivity, and deposition of sewage-associated contaminants, these two low kinetic energy sites are prime candidate areas for the long-term accumulation of detritus and seston associated with these phenomena. Farfield monitoring stations should include the depositional areas in Stellwagen Basin and northern Cape Cod. Monitoring of the MBDS should continue. All monitoring should include detailed source characterization to help to discriminate among the multiple sources of contaminants to Massachusetts Bay.

The chemistry data reviewed in this report came from fragmented studies of narrow scope. The sampling and analytical methodology was not consistent among most of the studies. Thus, direct comparison of the data is difficult and, in some cases, inappropriate. Proper management of Massachusetts Bay can be achieved only through the implementation of a comprehensive and integrated monitoring effort.

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Appendix A

CHARACTERIZATION OF SEDIMENTS IN MASSACHUSETTS AND CAPE COD BAYS

CHARACTERIZATION OF SEDIMENTS IN MASSACHUSETTS AND CAPE COD BAYS

Massachusetts Bay

There is a large area of silt/clay sediment between Stellwagen Bank and the Massachusetts coast (Figure A-1). The deposit ranges from about 8 to 14 nmi off the coast and trends NW—SE over a distance of 24 nmi. This silt/clay basin is surrounded on all sides by coarser-grained deposits. Figure A-2 shows the relationship of the silt/clay basin to regional bathymetry. The silt/clay deposit is confined to water depths of greater than 200 ft (61 m). The most organic-rich sediments (2%-4% organic carbon) lie within the axis of the silt/clay basin (Figure A-2). The northwestern end of the silt/clay basin contains the Massachusetts Bay dredged-material disposal site.

The Massachusetts Bay silt/clay basin, especially the area of 2%-4% organic carbon, is the most likely far-field area for long-term accumulation of fine particulates that may move seaward (net easterly drift) from the diffuser site. The distribution maps of metals such as Cr, Cu, Pb [Figures 14(a)-)c)] and PCBs, PAHs, and coprostanol [Figures 15(a)-(c)] show "hot spots" within the northern end of this basin near the dredged-material disposal site. From the mapped gradients in these contaminants around the disposal site, it appears that particles and adsorbed contaminants that are deposited within the silt/clay basin tend to remain within this basin. If this inference is valid, the basin may serve to focus and concentrate contaminants over the long term.

Cape Cod Bay

Cape Cod Bay is a potential area of impact for effluent or enhanced primary productivity that might be transported out of Massachusetts Bay. For this reason, depositional sites within Cape Cod Bay require identification.

There is an extensive data set, based on 435 benthic stations, for both sediment and fauna for Cape Cod Bay. These data were collected in the late 1960s by the Systematic-Ecology Program (SEP) of the Marine Biological Laboratory (MBL) at Woods Hole, Massachusetts. Although most of these data are unpublished, subsets have been used to prepare two summary papers (Young and Rhoads, 1971; Rhoads and Young, 1971). A synopsis of these two papers is given below.

Figure A-3 shows the distribution of major textural classes in Cape Cod Bay. The sediment map was prepared from data acquired from all 435 stations. The results presented by Young and Rhoads (1971) and Rhoads and Young (1971) involved detailed sampling at seven stations, which were located on a transect extending from the mouth of Barnstable Harbor to deep water off Race Point. These seven stations were chosen because they represented a wide range of water depths, temperature, sediment types, and faunal facies.

The central part of the Bay is composed of silt and silt plus clay. This sediment facies can be delimited by a 20% silt-plus-clay content isopleth and generally lies below a depth of 20 m (Figure A-4). A summer thermocline exists in the Bay from mid-April to mid-October. The intersection of the thermocline with the seafloor is defined by the 10 °C isotherm (Figure A-4).

There appears to be a close spatial correspondence between the shallow-water limit of silt plus clay (ca. 20 m) and the intersection of the pycnocline with the seafloor. Young and Rhoads (1971) suggest a cause-and-effect relationship between the pycnocline and silt plus clay content of the bottom. The water column below the pycnocline contains higher sediment loads than does the water above the pycnocline. This high turbidity is related to tidal resuspension of the upper few millimeters of the sediment surface. These suspended organ-

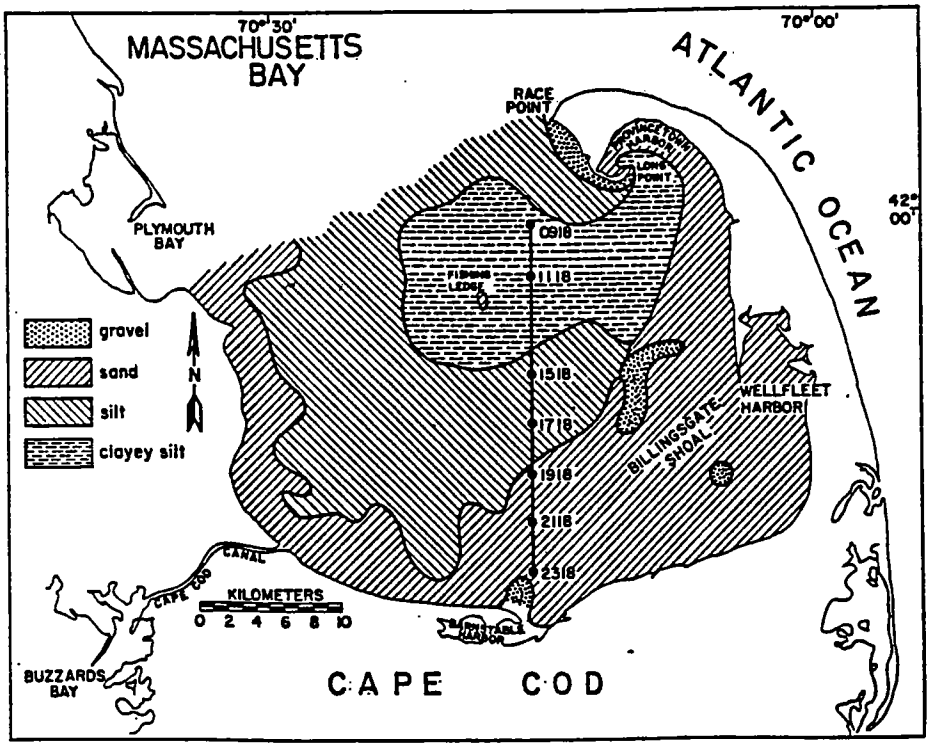

Figure A-1. Distribution of Major Textural Types in Cape Cod Bay (from Young and Rhoads, 1971).

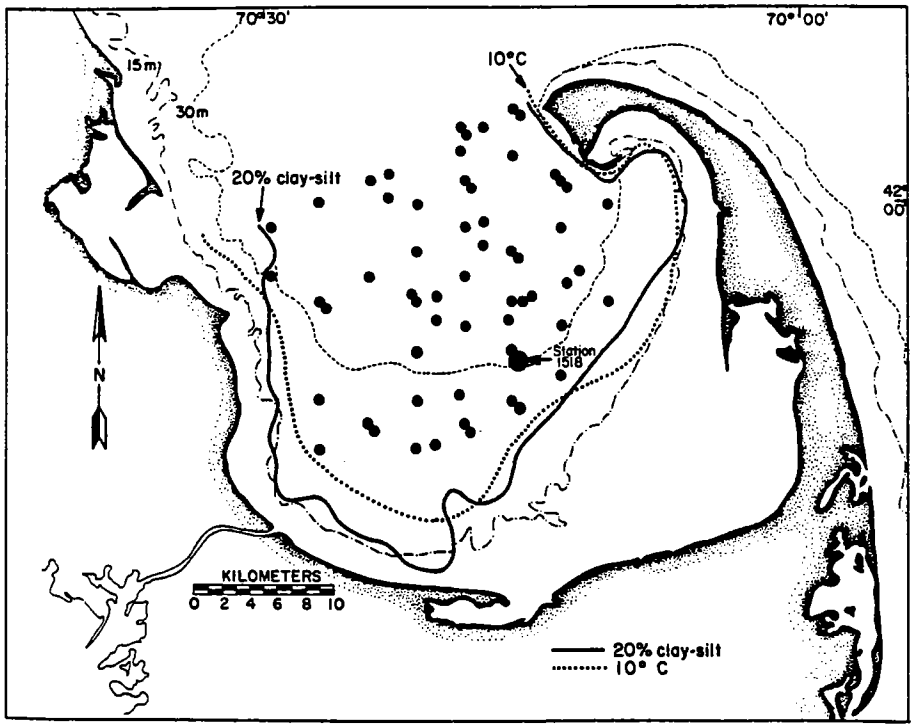

Figure A-2. Distribution of the Stage III Biomass Dominant in the Silt + Clay Facies of Cape Cod Bay. Note that the distribution of the 20% silt-clay boundary closely coincides with the 10°C summer isotherm. This marks the intersection of the thermocline with the seafloor (from Young and Rhoads, 1971).

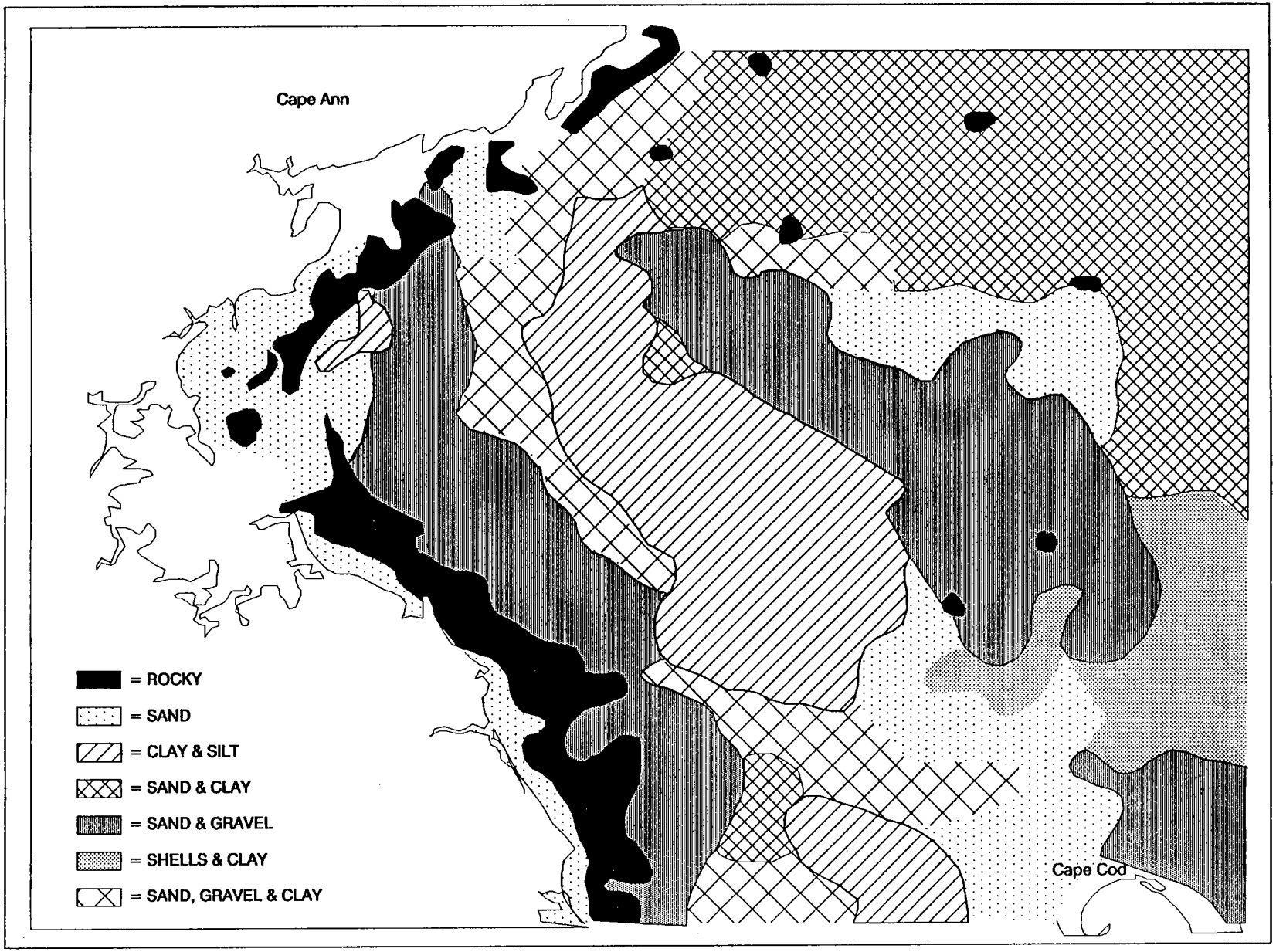

Figure A-3. Distribution of Major Sediment Types in Massachusetts Bay. Redrawn from Trowbridge and Shepherd (1932).

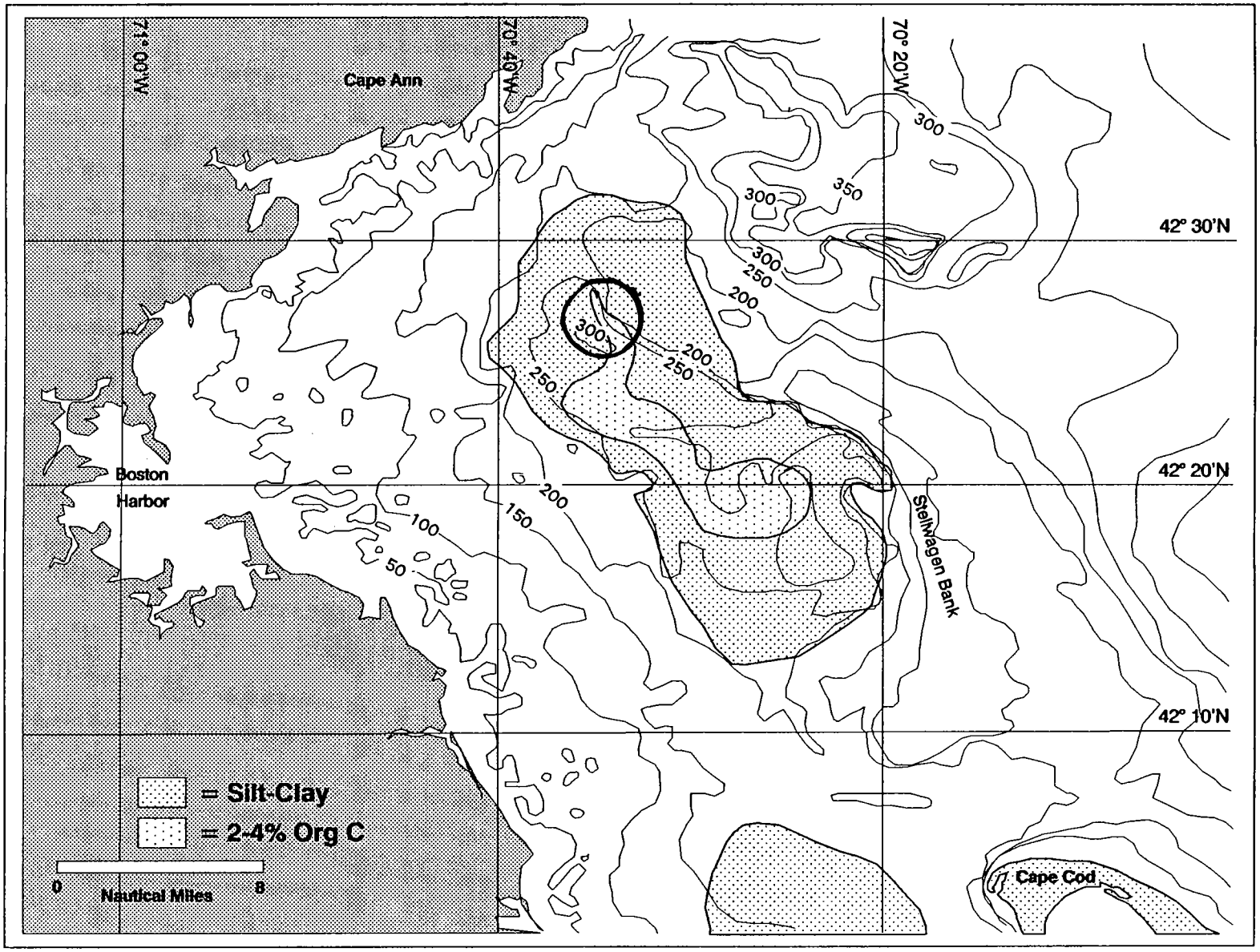

Figure A-4. **The Distribution of Fine-grained and Organic-rich Sediment in Massachusetts Bay Relative to Bottom Topography. Redrawn from Trowbridge and Shepard (1932). The circle denotes the location of the Massachusetts Bay Disposal Site.**

ic/mineral aggregates are transported both vertically and laterally but are isolated from waters shallower than 20m by the strongly developed pycnocline. Although winter wind-mixing may destroy the pycnocline and spread fine-grained sediment over a larger area, the reestablishment of the spring/summer/fall pycnocline is proposed as a mechanism to explain the sharp and faunal gradients across this sediment/thermal boundary.

Organism/sediment relationships in the silt/clay facies are largely structured by the deposit-feeding (Stage III) caudate holothurian species *Molpadia oolitica.* This vertically oriented holothurian mixes sediments to a depth of approximately 20 cm. The microtopography of the bottom of the Bay is dominated by the fecal mounds of *M. oolitica* (Rhoads and Young, 1971). These fecal mounds are densely populated with the sabellid polychaete *Euchone incolor.*

The likely long-term fate of effluent or phytoplankton detritus transported into Cape Cod Bay from Massachusetts Bay is that it will ultimately be deposited (and resuspended) within the silt-plus-clay basin. Tidal resuspension of recently deposited organic matter and deep vertical bioturbation of the bottom by *M. oolitica* would likely stimulate efficient aerobic decomposition of organic matter. The natural system can be expected to accommodate additional organic loading up to some critical loading rate (Rice and Rhoads, 1989). Beyond this critical rate, there will be net accumulation of labile organic matter. Accumulation of this organic matter can increase sediment oxygen demand, water-column oxygen demand, and loss of deeply bioturbating species.

The critical organic loading rate for causing these changes in Cape Code Bay is not known. It is also not known for Massachusetts Bay. A recent review of the relationship between organic loading rates and benthic responses suggests that this critical rate is on the order of >500 mg C/m^2/year (Valenti, 1990).

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Appendix B

BENTHIC BIOLOGY DATA

Table B-1. Top Ten Dominant Species at Station 1 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	3,130	32.0
2	*Myriochele ?heeri*	1,365	13.9
3	*Prionospio steenstrupi*	950	9.7
4	*Maldane sarsi*	455	4.6
5	*Mediomastus californiensis*	310	3.2
6	*Thyasira ?gouldi*	295	3.0
7	*Praxillella praetermissa*	255	2.6
8	*Ampharete acutifrons*	250	2.6
9	*Aegininia longicornis*	250	2.6
10	*Hippomedon propinquus*	175	1.8
	TOTAL FAUNA	9,790	

Table B-2. Top Ten Dominant Species at Station 2 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Myriochele ?heeri*	3,265	36.3
2	*Prionospio steenstrupi*	1,380	15.4
3	*Spio limicola*	840	9.3
4	*Mediomastus californiensis*	485	5.4
5	*Praxillella praetermissa*	455	5.1
6	*Levinsenia gracilis*	220	2.4
7	*Maldane sarsi*	210	2.3
8	*Ampharete arctica*	165	1.8
9	*Chaetozone setosa*	165	1.8
10	*Hippomedon propinquus*	155	1.7
	TOTAL FAUNA	8,990	

Table B-3. Top Ten Dominant Species at Station 3 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Mediomastus californiensis*	1,575	40.1
2	*Levinsenia gracilis*	355	9.0
3	*Myriochele ?heeri*	340	8.7
4	*Prionospio steenstrupi*	285	7.3
5	*Hippomedon propinquus*	245	6.2
6	*Eudorella emarginata*	165	4.2
7	Megalopa larvae	120	3.1
8	*Chaetozone setosa*	90	2.3
9	*Cossura longocirrata*	70	1.8
10	*Spio limicola*	60	1.5
	TOTAL FAUNA	3,930	

Table B-4. Top Ten Dominant Species at Station 4 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	20,670	69.4
2	*Myriochele ?heeri*	960	3.2
3	*Maldane sarsi*	605	2.0
4	*Prionospio steenstrupi*	535	1.8
5	*Thyasira ?gouldi*	520	1.7
6	*Praxillella praetermissa*	400	1.3
7	*Polycirrus ?medusa*	395	1.3
8	*Phyllodoce mucosa*	335	1.1
9	*Pholoe minuta*	290	1.0
10	*Harmothoe imbricata*	285	1.0
	TOTAL FAUNA	29,785	

Table B-5. Top Ten Dominant Species at Station 5 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	12,675	62.6
2	*Mediomastus californiensis*	1,595	7.9
3	*Prionospio steenstrupi*	1,585	7.8
4	*Myriochele ?heeri*	640	3.2
5	*Aricidea catherinae*	595	2.9
6	*Praxillella praetermissa*	570	2.8
7	*Chaetozone setosa*	325	1.6
8	*Sternaspis scutata*	285	1.4
9	*Hippomedon propinquus*	245	1.2
10	*Levinsenia gracilis*	210	1.0
	TOTAL FAUNA	20,240	

Table B-6. Top Ten dominant Species at Station 6 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	9,560	32.4
2	*Prionospio steenstrupi*	3,520	11.9
3	*Myriochele ?heeri*	2,685	9.1
4	*Maldane sarsi*	2,480	8.4
5	*Mediomastus californiensis*	1,355	4.6
6	*Aricidea catherinae*	1,240	4.2
7	*Ampharete arctica*	980	3.3
8	*Leucon* sp.	775	2.6
9	*Hippomedon propinquus*	595	2.0
10	*Sternaspis scutata*	505	1.7
	TOTAL FAUNA	29,475	

Table B-7. Top Ten Dominant Species at Station 7 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	3,565	18.0
2	*Myriochele ?heeri*	3,140	15.9
3	*Exogone verugera*	2,185	11.0
4	*Unciola irrorata*	1,910	9.7
5	*Prionospio steenstrupi*	1,420	7.2
6	*Praxillella praetermissa*	925	4.7
7	*Macrosetella* sp.	685	3.5
8	*Mediomastus californiensis*	425	2.1
9	*Polydora ?caeca*	305	1.5
10	*Ampharete arctica*	285	1.4
	TOTAL FAUNA	19,780	

Table B-8. Top Ten Dominant Species at Station 8 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	36,165	80.4
2	*Prionospio steenstrupi*	1,575	3.5
3	*Tharyx* sp. A	685	1.5
4	*Pholoe minuta*	540	1.2
5	*Myriochele ?heeri*	475	1.1
6	*Maldane sarsi*	345	0.8
7	*Harmothoe imbricata*	325	0.7
8	*Phyllodoce mucosa*	300	0.7
9	*Mediomastus californiensis*	285	0.6
10	*Leitoscoloplos acutus*	285	0.6
	TOTAL FAUNA	44,995	

Table B-9. Top Ten Dominant Species at Station 9 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	8,720	55.4
2	*Prionospio steenstrupi*	1,420	9.0
3	*Mediomastus californiensis*	1,395	8.9
4	*Myriochele ?heeri*	570	3.6
5	*Aricidea catherinae*	540	3.4
6	*Hippomedon propinquus*	475	3.0
7	*Chaetozone setosa*	295	1.9
8	*Maldane sarsi*	285	1.8
9	*Praxillella praetermissa*	240	1.5
10	*Leucon* sp.	190	1.2
	TOTAL FAUNA	15,750	

Table B-10. Top Ten Dominant Species at Station 10 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	7,370	61.5
2	*Prionospio steenstrupi*	755	6.3
3	*Aricidea catherinae*	605	5.1
4	*Mediomastus californiensis*	580	4.8
5	*Ampharete arctica*	430	3.6
6	*Leucon* sp.	320	2.7
7	*Sternaspis scutata*	310	2.6
8	*Myriochele ?heeri*	265	2.2
9	*Cossura longocirrata*	145	1.2
10	*Levinsenia gracilis*	135	1.1
	TOTAL FAUNA	11,975	

Table B-11. Top Ten Dominant Species at Station 11 (from Gilbert et al., (1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	2,130	25.2
2	*Prionospio steenstrupi*	2,035	24.1
3	*Myriochele ?heeri*	665	7.9
4	*Mediomastus californiensis*	615	7.3
5	*Aricidea catherinae*	510	6.0
6	*Cossura longocirrata*	400	4.7
7	*Nephtys bucera*	365	4.3
8	*Levinsenia gracilis*	250	3.0
9	*Leucon* sp.	205	2.4
10	*Hippomedon propinquus*	180	2.1
	TOTAL FAUNA	8,460	

Table B-12. Top Ten Dominant Species at Station 12 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	Prionospio steenstrupi	2,775	25.7
2	Spio limicola	2,290	21.2
3	Aricidea catherinae	1,810	16.7
4	Mediomastus californiensis	630	5.8
5	Myriochele ?heeri	340	3.1
6	Cossura longocirrata	330	3.1
7	Chaetozone setosa	315	2.9
8	Paramphinome pulchella	305	2.8
9	Ampharete arctica	205	1.9
10	Levinsenia gracilis	185	1.7
	TOTAL FAUNA	10,810	

Table B-13. Top Ten dominant Species at Station 13 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	1,080	33.6
2	*Prionospio steenstrupi*	550	17.1
3	*Aricidea catherinae*	400	12.5
4	*Myriochele ?heeri*	200	6.2
5	*Chaetozone setosa*	140	4.4
6	*Cossura longocirrata*	130	4.0
7	*Mediomastus californiensis*	120	3.7
8	*Micrura* sp.	35	1.1
9	*Paramphinome pulchella*	35	1.1
10	*Pholoe minuta*	30	0.9
	TOTAL FAUNA	3,210	

Table B-14. Top Ten Dominant Species at Station 14 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	1,785	31.5
2	*Prionospio steenstrupi*	1,185	20.9
3	*Mediomastus californiensis*	480	8.5
4	*Myriochele ?heeri*	410	7.2
5	*Aricidea catherinae*	225	4.0
6	*Centropages* sp. A	190	3.4
7	*Thyasira ?gouldi*	105	1.9
8	*Euchone rubrocincta*	95	1.7
9	*Nucula tenuis*	85	1.5
10	*Ampharete arctica*	75	1.3
	TOTAL FAUNA	5,665	

Table B-15. Top Ten Dominant Species at Station 15 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	8,790	38.5
2	*Prionospio steenstrupi*	4,140	18.1
3	*Aricidea catherinae*	2,660	11.6
4	*Mediomastus californiensis*	1,915	8.4
5	*Thyasira ?gouldi*	820	3.6
6	*Myriochele ?heeri*	700	3.1
7	*Chaetozone setosa*	295	1.3
8	*Praxillella praetermissa*	240	1.1
9	*Nucula tenuis*	230	1.0
10	*Levinsenia gracilis*	195	0.9
	TOTAL FAUNA	22,855	

Table B-16. Top Ten Dominant Species at Station 16 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	19,510	77.1
2	*Prionospio steenstrupi*	1,360	5.4
3	*Mediomastus californiensis*	445	1.8
4	*Tharyx* sp. A	385	1.5
5	*Levinsenia gracilis*	355	1.4
6	*Ninoe nigripes*	350	1.4
7	*Pholoe minuta*	280	1.1
8	*Leitoscoloplos acutus*	270	1.1
9	*Goniada maculata*	210	0.8
10	*Harmothoe imbricata*	190	0.8
	TOTAL FAUNA	25,310	

Table B-17. Top Ten Dominant Species at Station 17 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	4,125	55.4
2	*Prionospio steenstrupi*	975	13.1
3	*Aricidea catherinae*	365	4.9
4	*Mediomastus californiensis*	270	3.6
5	*Levinsenia gracilis*	155	2.1
6	*Nucula tenuis*	125	1.7
7	*Paramphinome pulchella*	120	1.6
8	*Cossura longocirrata*	120	1.6
9	*Chaetozone setosa*	120	1.6
10	*Myriochele ?heeri*	115	1.5
	TOTAL FAUNA	7,445	

Table B-18. Top Ten dominant Species at Station 18 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	4,845	43.4
2	*Prionospio steenstrupi*	1,195	10.7
3	*Aricidea catherinae*	1,180	10.6
4	*Mediomastus californiensis*	900	8.1
5	*Chaetozone setosa*	355	3.2
6	*Myriochele ?heeri*	350	3.1
7	*Thyasira ?gouldi*	335	3.0
8	*Sternaspis scutata*	300	2.7
9	*Leitoscoloplos acutus*	180	1.6
10	*Cossura longocirrata*	145	1.3
	TOTAL FAUNA	11,175	

Table B-19. Top Ten Dominant Species at Station 19 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	5,200	47.2
2	*Prionospio steenstrupi*	1,330	12.1
3	*Aricidea catherinae*	1,195	10.8
4	*Mediomastus californiensis*	735	6.7
5	*Myriochele ?heeri*	470	4.3
6	*Chaetozone setosa*	195	1.8
7	*Levinsenia gracilis*	185	1.7
8	*Sternaspis scutata*	150	1.4
9	*Cossura longocirrata*	120	1.1
10	tubificid oligochaete	120	1.1
	TOTAL FAUNA	11,020	

Table B-20. Top Ten Dominant Species at Station 20 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	11,470	60.1
2	*Prionospio steenstrupi*	2,225	11.8
3	*Aricidea catherinae*	1,400	7.4
4	*Mediomastus californiensis*	680	3.6
5	*Chaetozone setosa*	655	3.5
6	*Cossura longocirrata*	325	1.7
7	*Myriochele ?heeri*	240	1.3
8	*Praxillella praetermissa*	165	0.9
9	*Scalibregma inflatum*	160	0.8
10	*Centropages* sp. A	160	0.8
	TOTAL FAUNA	18,925	

Table B-21. Top Ten Dominant Species at Station 21 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	6,610	56.8
2	*Aricidea catherinae*	1,355	11.7
3	*Mediomastus californiensis*	350	3.0
4	*Chaetozone setosa*	260	2.2
5	*Sternaspis scutata*	215	1.8
6	*Myriochele ?heeri*	150	1.3
7	*Praxillella praetermissa*	145	1.2
8	*Cossura longocirrata*	130	1.1
9	*Pholoe minuta*	125	1.1
10	*Leitoscoloplos acutus*	125	1.1
	TOTAL FAUNA	11,630	

Table B-22. Top Ten Dominant Species at Station 22 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	6,055	59.4
2	*Prionospio steenstrupi*	1,040	10.2
3	*Aricidea catherinae*	905	8.8
4	*Mediomastus californiensis*	425	4.2
5	*Chaetozone setosa*	230	2.3
6	*Myriochele ?heeri*	190	1.9
7	*Cossura longocirrata*	130	1.3
8	*Centropages* sp. A	125	1.2
9	*Scalibregma inflatum*	85	0.8
10	*Praxillella praetermissa*	75	0.7
	TOTAL FAUNA	10,195	

Table B-23. Top Ten Dominant Species at Station 23 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	3,360	42.2
2	*Aricidea catherinae*	740	9.3
3	*Mediomastus californiensis*	450	5.6
4	*Prionospio steenstrupi*	285	3.6
5	*Chaetozone setosa*	265	3.3
6	tubificid oligochaete	240	3.0
7	*Maldane sarsi*	235	2.9
8	*Pholoe minuta*	170	2.1
9	*Sternaspis scutata*	135	1.7
10	*Myriochele ?heeri*	130	1.6
	TOTAL FAUNA	7,970	

Table B-24. Top Ten Dominant Species at Station 24 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	9,980	54.5
2	*Prionospio steenstrupi*	1,340	7.3
3	*Aricidea catherinae*	1,055	5.8
4	*Myriochele ?heeri*	695	3.8
5	*Mediomastus californiensis*	530	2.9
6	*Sternaspis scutata*	500	2.7
7	*Leitoscoloplos acutus*	450	2.5
8	*Chaetozone setosa*	400	2.2
9	*Pholoe minuta*	280	1.5
10	*Thyasira ?gouldi*	250	1.4
	TOTAL FAUNA	18,325	

Table B-25. Top Ten Dominant Species at Station 25 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	13,195	65.3
2	*Prionospio steenstrupi*	1,300	6.4
3	*Aricidea catherinae*	720	3.6
4	*Thyasira ?gouldi*	450	2.2
5	*Chaetozone setosa*	365	1.8
6	*Mediomastus californiensis*	350	1.7
7	*Leitoscoloplos acutus*	340	1.7
8	*Sternaspis scutata*	245	1.2
9	*Maldane sarsi*	235	1.2
10	*Eteone longa*	225	1.1
	TOTAL FAUNA	20,220	

Table B-26. Top Ten Dominant Species at Station 26 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	8,840	38.9
2	*Thyasira ?gouldi*	1,195	5.3
3	*Prionospio steenstrupi*	1,105	4.9
4	*Mediomastus californiensis*	535	2.4
5	*Aeginina longicornis*	500	2.2
6	*Aricidea catherinae*	495	2.2
7	*Maldane sarsi*	415	1.8
8	*Myriochele ?heeri*	400	1.8
9	*Pholoe minuta*	320	1.4
10	*Nucula tenuis*	310	1.4
	TOTAL FAUNA	22,735	

Table B-27. Top Ten Dominant species at Station 27 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	8,840	52.4
2	*Aricidea catherinae*	1,275	7.6
3	*Prionospio steenstrupi*	1,035	6.2
4	*Mediomastus californiensis*	885	5.3
5	*Myriochele ?heeri*	680	4.0
6	*Chaetozone setosa*	665	4.0
7	*Thyasira ?gouldi*	375	2.2
8	*Leitoscoloplos acutus*	345	2.1
9	*Sternaspis scutata*	315	1.9
10	*Tharyx* sp. B	175	1.0
	TOTAL FAUNA	16,795	

Table B-28. Top Ten Dominant Species at Station 28 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	12,550	63.9
2	*Aricidea catherinae*	1,055	5.4
3	*Mediomastus californiensis*	735	3.7
4	*Chaetozone setosa*	545	2.8
5	*Thyasira ?gouldi*	485	2.5
6	*Prionospio steenstrupi*	410	2.1
7	*Aegininia longicornis*	380	1.9
8	*Pholoe minuta*	345	1.8
9	*Leitoscoloplos acutus*	310	1.6
10	*Ninoe nigripes*	285	1.5
	TOTAL FAUNA	19,630	

Table B-29. Top Ten Dominant Species at Station 29 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	12,070	52.0
2	*Aricidea catherinae*	1,945	8.4
3	*Mediomastus californiensis*	1,015	4.8
4	*Chaetozone setosa*	970	4.2
5	*Alvania carinata*	670	2.9
6	*Prionospio steenstrupi*	650	2.8
7	tubificid oligochaete	620	2.7
8	*Tharyx* sp. B	575	2.5
9	*Pholoe minuta*	275	1.2
10	*Leitoscoloplos acutus*	260	1.1
	TOTAL FAUNA	23,190	

Table B-30. Top Ten Dominant Species at Station 30 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	5,775	48.9
2	*Aricidea catherinae*	780	6.6
3	*Tharyx* sp. B	585	5.0
4	*Prionospio steenstrupi*	370	3.1
5	*Ninoe nigripes*	370	3.1
6	*Chaetozone setosa*	305	2.6
7	*Mediomastus californiensis*	300	2.5
8	*Stenopleustes inermis*	250	2.1
9	*Nephtys incisa*	215	1.8
10	*Myriochele ?heeri*	170	1.4
	TOTAL FAUNA	11,805	

Table B-31. Top Ten Dominant Species at Station 31 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	5,225	53.2
2	*Mediomastus californiensis*	870	8.9
3	*Chaetozone setosa*	480	4.9
4	*Aricidea catherinae*	405	4.1
5	*Thyasira ?gouldi*	235	2.4
6	*Ninoe nigripes*	225	2.3
7	*Pholoe minuta*	170	1.7
8	*Alvania carinata*	155	1.6
9	tubificid oligochaete	140	1.4
10	*Tharyx* sp. A	135	1.4
	TOTAL FAUNA	9,815	

Table B-32. Top Ten Dominant Species at Station 32 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	3,040	31.2
2	*Aricidea catherinae*	915	9.4
3	*Centropages* sp. A	730	7.5
4	*Tharyx* sp. A	480	4.9
5	*Mediomastus californiensis*	475	4.9
6	*Calanus finmarchicus*	415	4.3
7	*Levinsenia gracilis*	390	4.0
8	*Prionospio steenstrupi*	330	3.4
9	tubificid oligochaete	325	3.3
10	*Ninoe nigripes*	300	3.1
	TOTAL FAUNA	9,750	

Table B-33. Top Ten Dominant Species at Station C-1 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Prionospio steenstrupi*	16,040	56.9
2	*Ampharete arctica*	1,910	6.8
3	*Nephtys incisa*	1,780	6.3
4	cirratulid	1,310	4.6
5	*Phyllodoce mucosa*	1,310	4.6
6	tubificid oligochaete	1,055	3.7
7	*Aricidea jeffreysii*	920	3.3
8	*Haploops setosa*	485	1.7
9	*Photis macrocoxa*	405	1.4
10	*Myriochele ?heeri*	305	1.1
	TOTAL FAUNA	28,195	

Table B-34. Top Ten Dominant Species at Station C-2 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	cirratulid	42,885	70.7
2	*Prionospio steenstrupi*	5,230	8.7
3	*Echinarachnius parma*	2,155	3.6
4	*Phoronis architecta*	1,030	1.7
5	*Mediomastus californiensis*	1,005	1.7
6	tubificid oligochaete	775	1.3
7	*Nephtys incisa*	765	1.3
8	*Photis macrocoxa*	715	1.2
9	*Pectinaria gouldi*	480	0.7
10	*Phyllodoce mucosa*	480	0.7
	TOTAL FAUNA	60,690	

Table B-35. Top Ten Dominant Species at Station C-3 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Nephtys incisa*	725	16.1
2	cirratulid	675	15.2
3	*Aricidea jeffreysii*	520	11.7
4	*Prionospio steenstrupi*	500	11.2
5	tubificid oligochaete	375	8.4
6	*Mediomastus californiensis*	230	5.2
7	*Spirorbis spirillum*	190	4.3
8	*Phyllodoce mucosa*	165	3.7
9	*Diastylis sculpta*	165	3.7
10	*Phoronis architecta*	105	2.4
	TOTAL FAUNA	4,445	

Table B-36. Top Ten Dominant Species at Station C-4 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Spio limicola*	9,130	33.1
2	*Prionospio steenstrupi*	3,580	13.1
3	*Myriochele ?heeri*	1,690	6.1
4	*Polydora ?caeca*	990	3.6
5	*Pholoe minuta*	730	2.6
6	*Centropages* sp. A	670	2.4
7	*Euchone rubrocincta*	660	2.4
8	*Harmothoe fragilis*	600	2.2
9	*Leucothoe spinicarpa*	590	2.1
10	*Mediomastus californiensis*	460	1.7
	TOTAL FAUNA	27,610	

Table B-37. Top Ten Dominant Species at Station C-5 (from Gilbert et al., 1976)

Rank	Species	Density per m^2	Percent of Total Fauna
1	*Ampharete arctica*	16,460	38.5
2	*Prionospio steenstrupi*	7,845	18.4
3	*Myriochele ?heeri*	2,265	5.3
4	*Phoronis architecta*	1,560	3.7
5	cirratulid	1,440	3.4
6	*Spio limicola*	1,440	3.4
7	*Phyllodoce mucosa*	1,290	3.0
8	*Polydora ?caeca*	1,070	2.5
9	*Leitoscoloplos acutus*	980	2.3
10	*Polydora* sp. A	755	1.8
	TOTAL FAUNA	42,720	

Appendix C

SEDIMENT CHEMISTRY DATA

The symbols used in Figures C-4 through C-26 represent the station codes shown in Figures C-1 through C-3 and Tables C-1 through C-6. Station codes for the STFP data (Battelle, 1987b) were abbreviated to include only the first letter (i.e., only the transect letter is given). Triplicate STFP data (Battelle, 1987b) are plotted in Figures C-4 through C-9 and C-22; duplicate data from Gilbert *et al.* (1976) are plotted in Figures C-16 through C-21 (e.g., 4A and 4B in Figure C-16). Mean concentration values are plotted for both NOAA (unpublished) data, Figures C-10 through C-15, and data from Boehm *et al.* (1984), Figures C-23 through C-26.

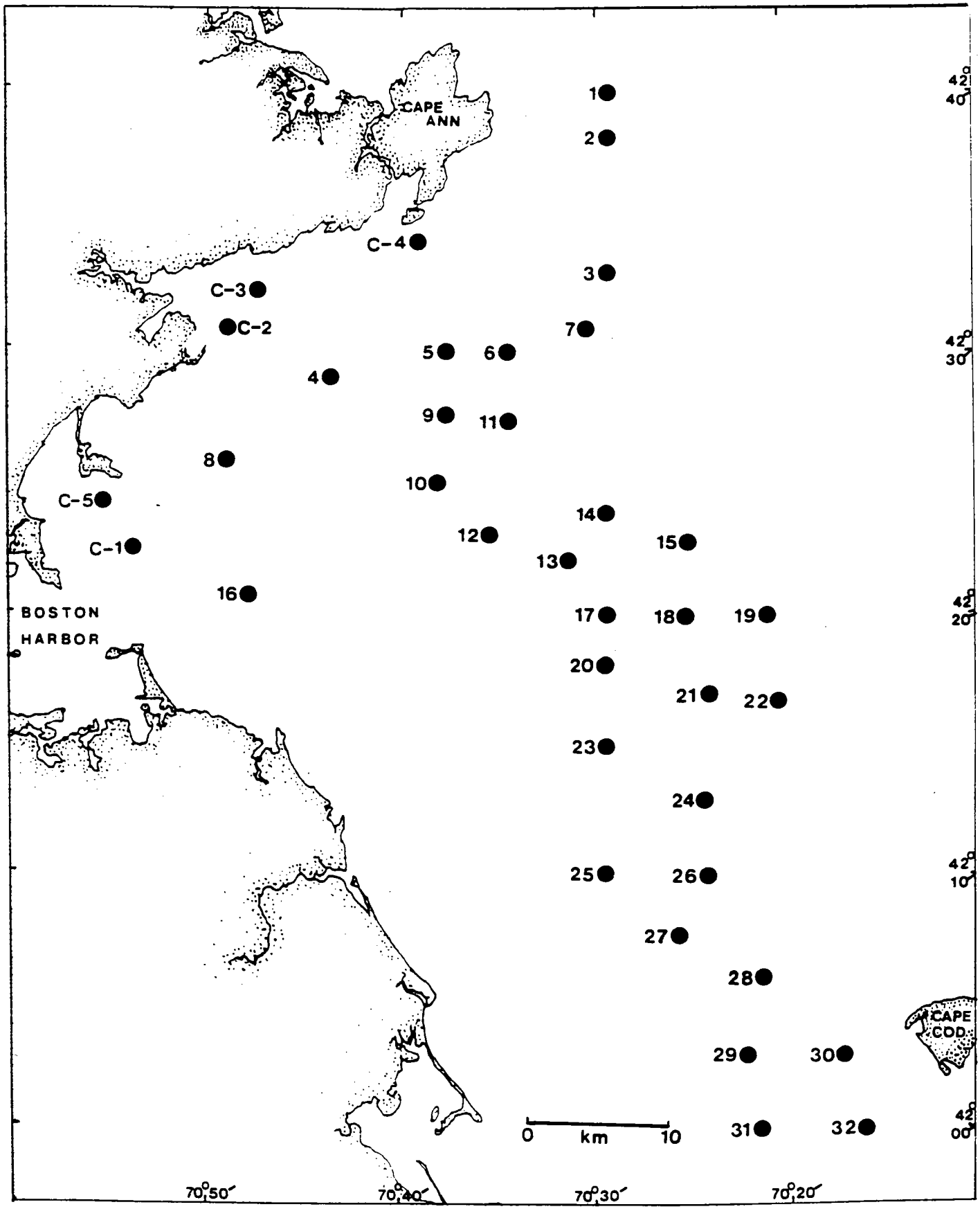


Figure C-1. Location of Sampling Stations on Survey by Gilbert *et al.*, 1976.

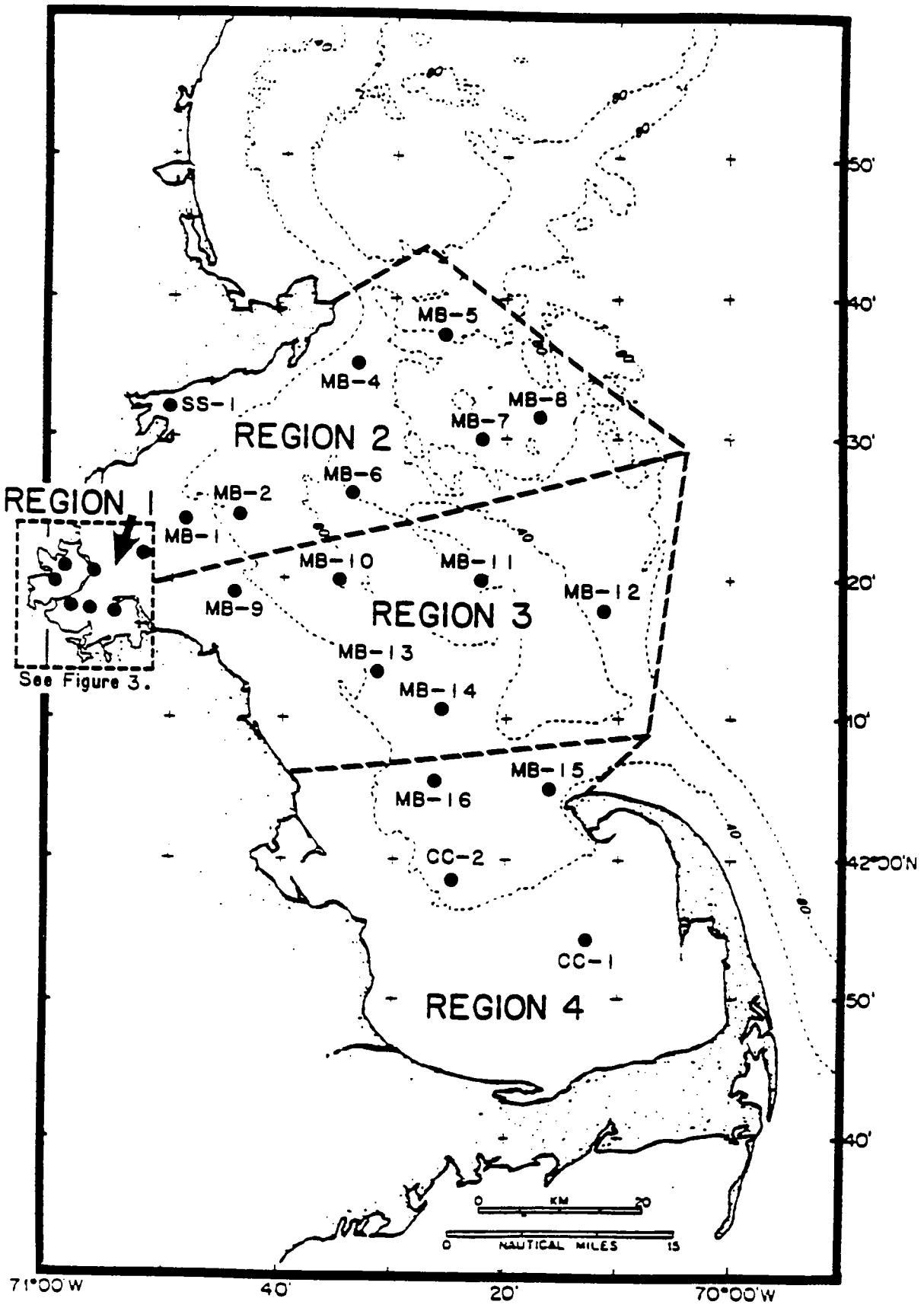


Figure C-2. **Location of Sampling Stations on NOAA-NEMP Survey (from Boehm *et al.*, 1984).**

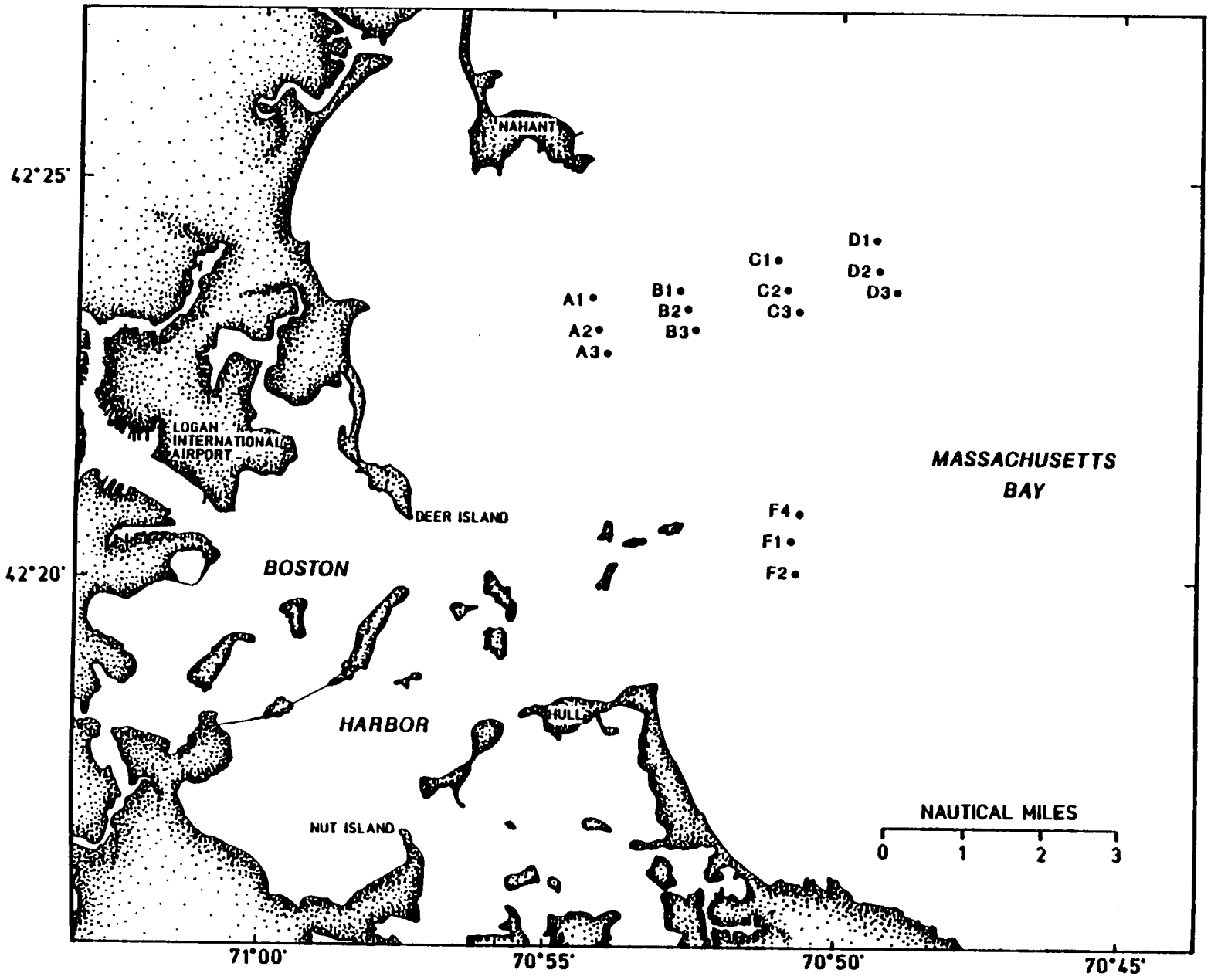


Figure C-3. Station Locations within the Outfall Siting Area Sampled During the MWRA-STFP Surveys (from Battelle, 1987b).

Figure C-4. TOC vs. Silt + Clay
Data from Battelle, 1987b

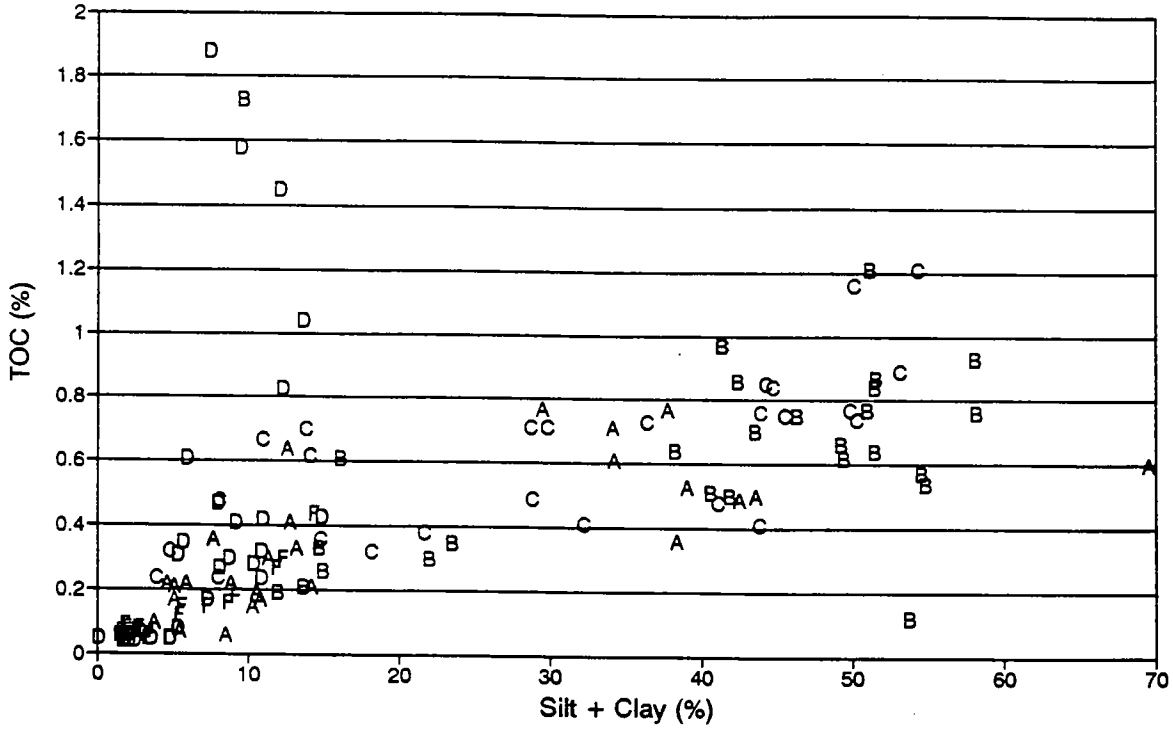


Figure C-5. Cd vs. Ni
Data from Battelle, 1987b

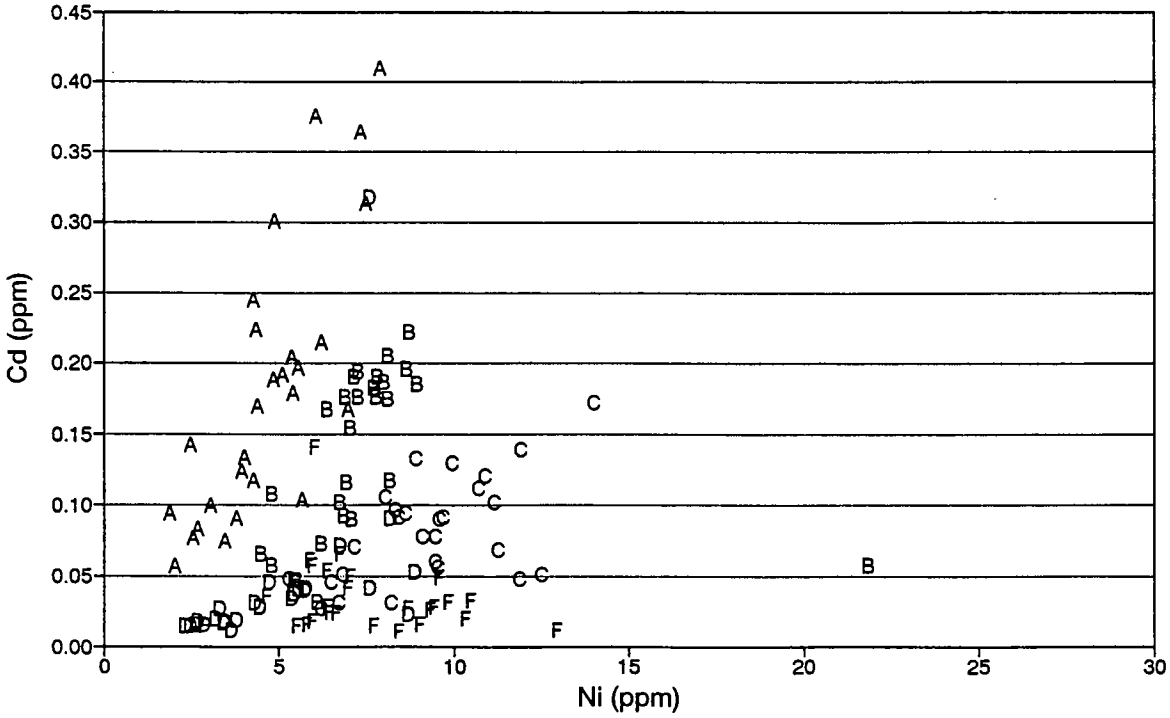


Figure C-6. Cr vs. Ni
Data from Battelle, 1987b

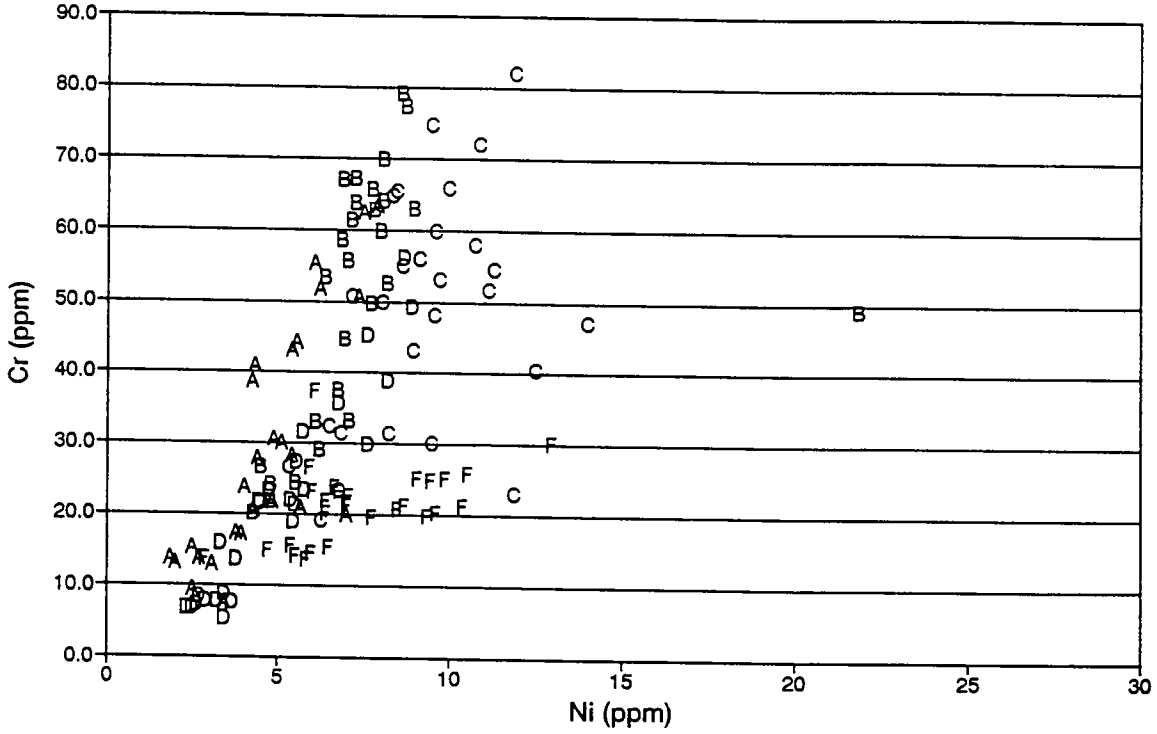


Figure C-7. Cu vs. Ni
Data from Battelle, 1987b

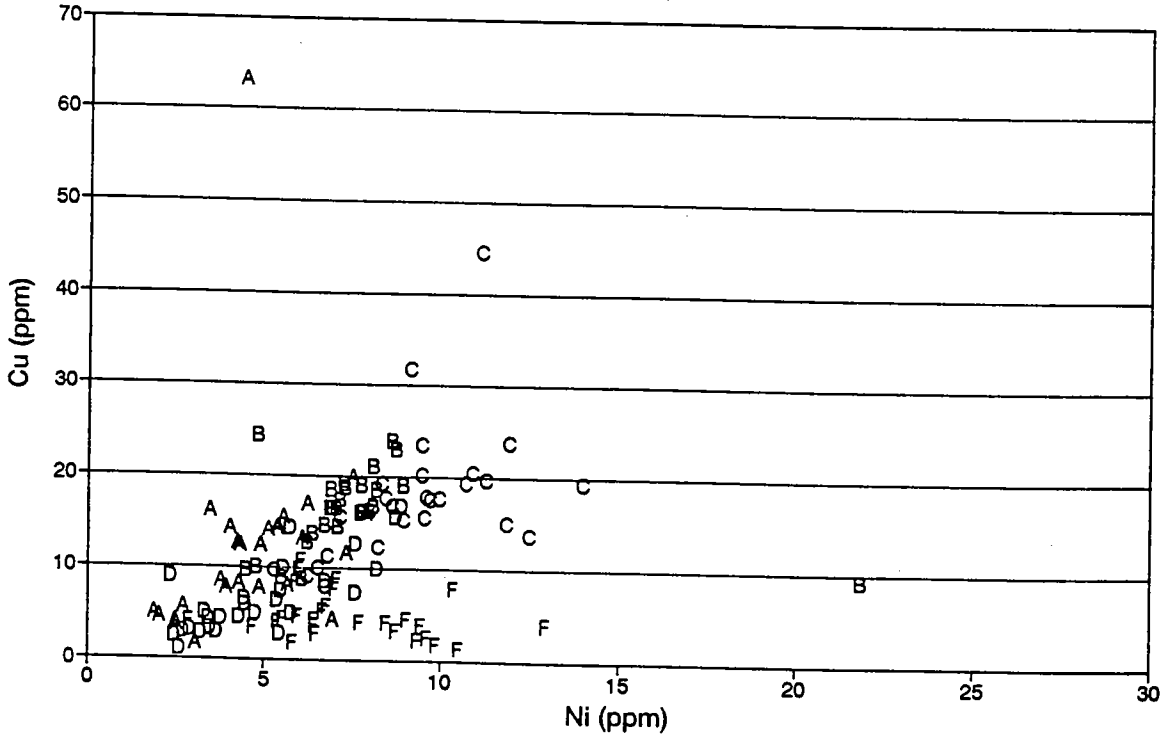


Figure C-8. Pb vs. Ni
Data from Battelle, 1987b

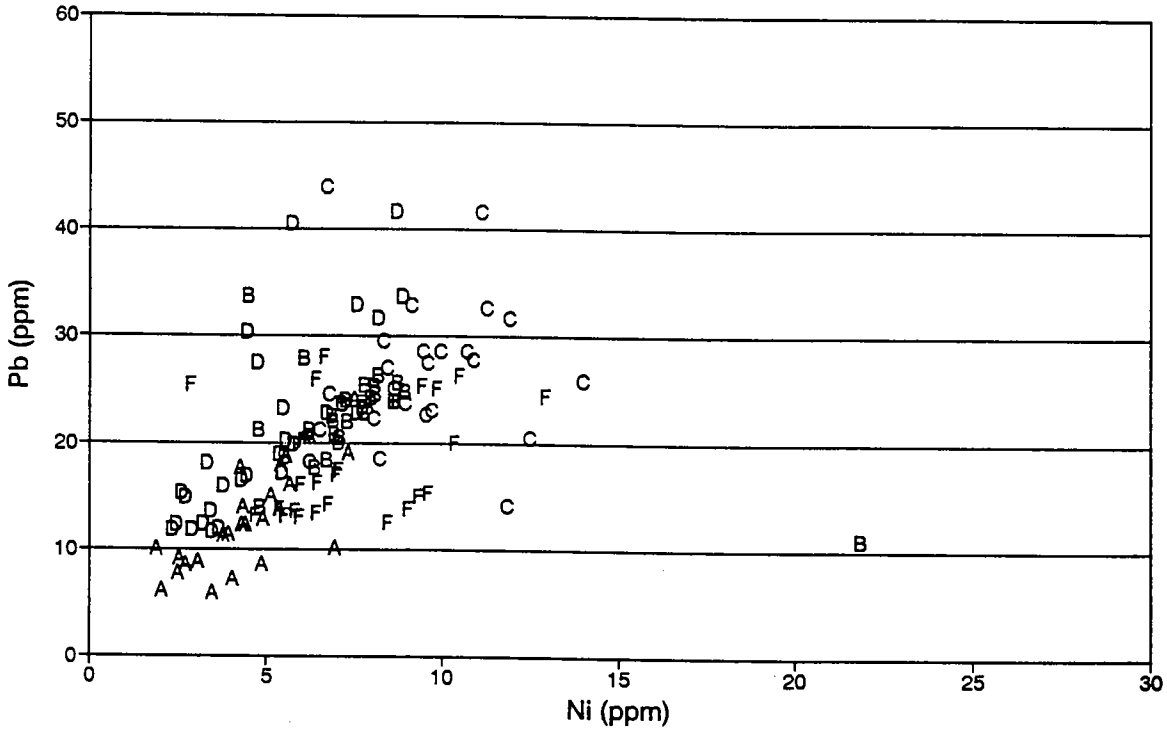


Figure C-9. Zn vs. Ni
Data from Battelle, 1987b

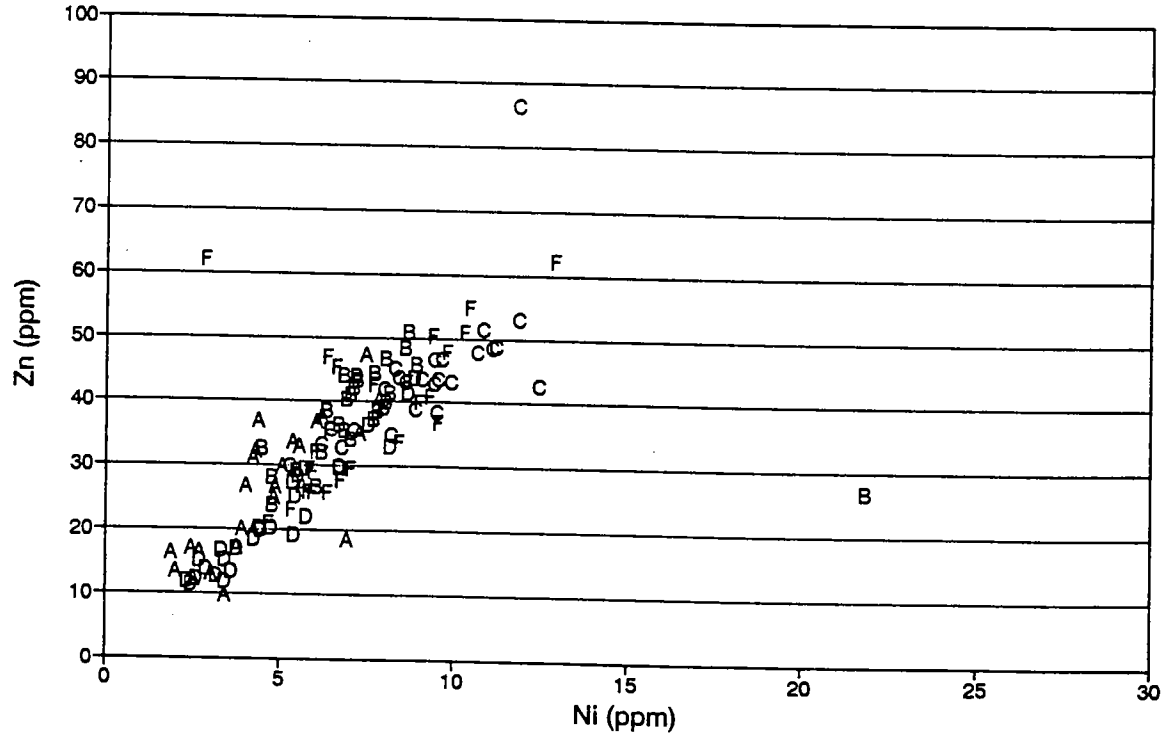


Figure C-10. Cd vs. Ni
Data from NOAA, unpublished

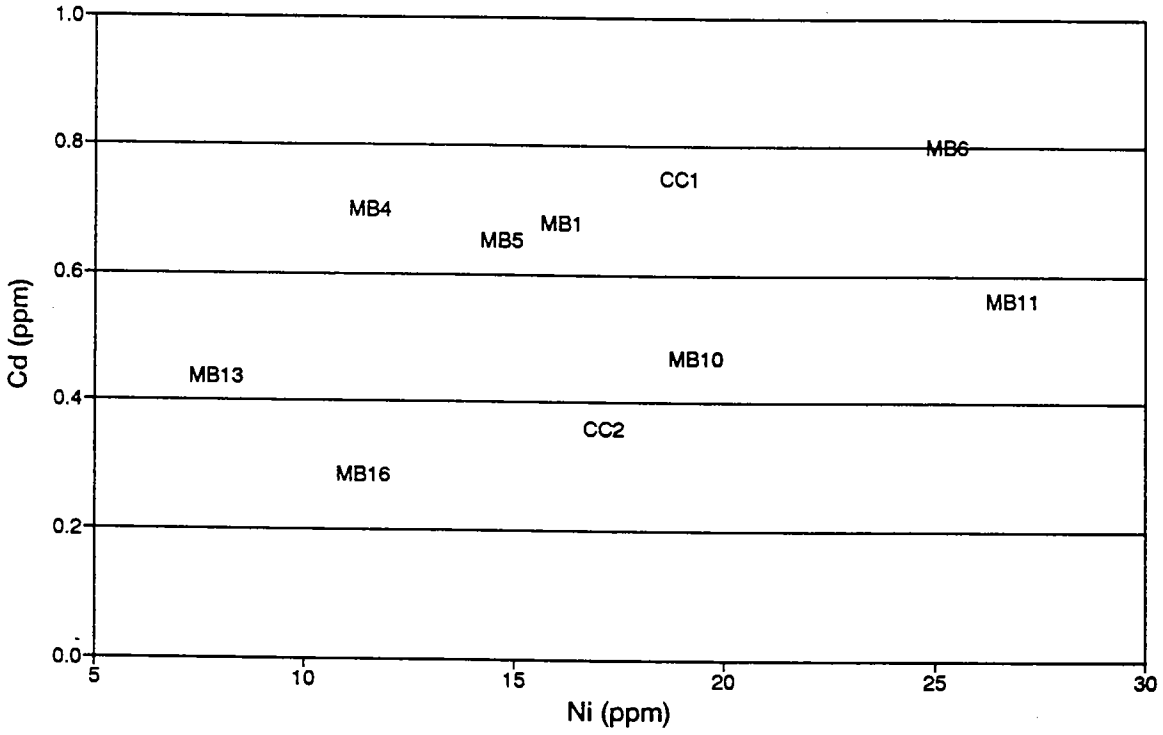


Figure C-11. Cr vs. Ni
Data from NOAA, unpublished

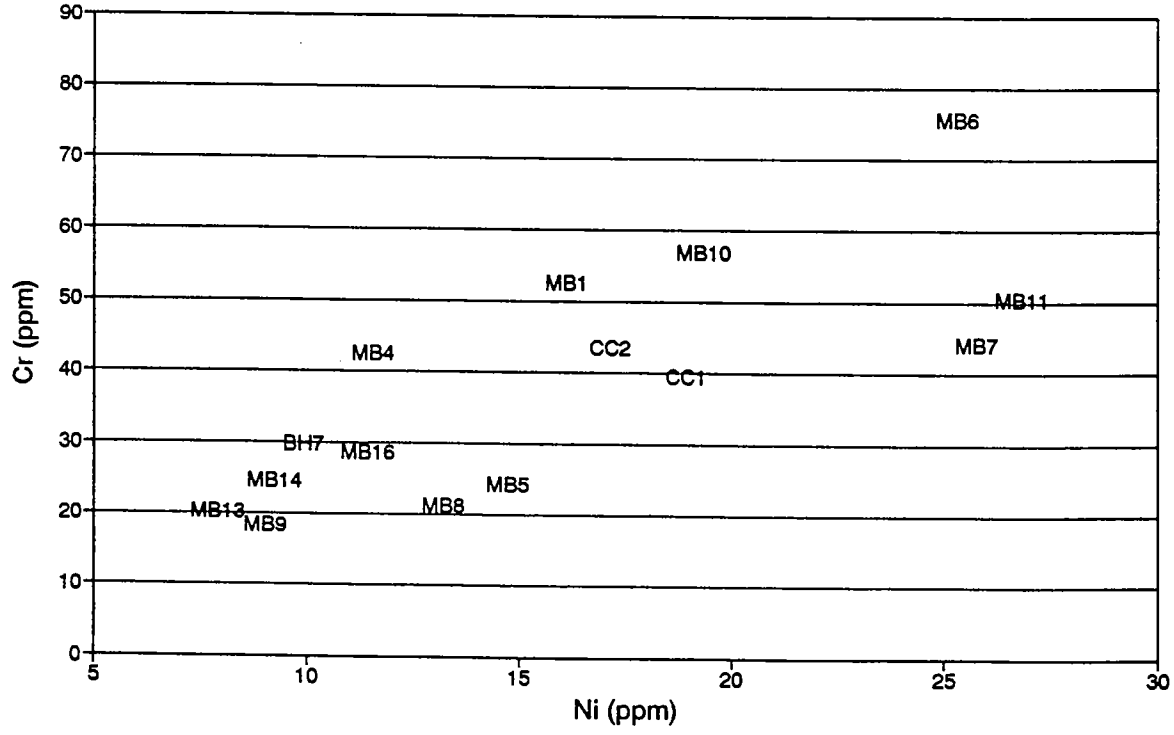


Figure C-12. Cu vs. Ni
Data from NOAA, unpublished

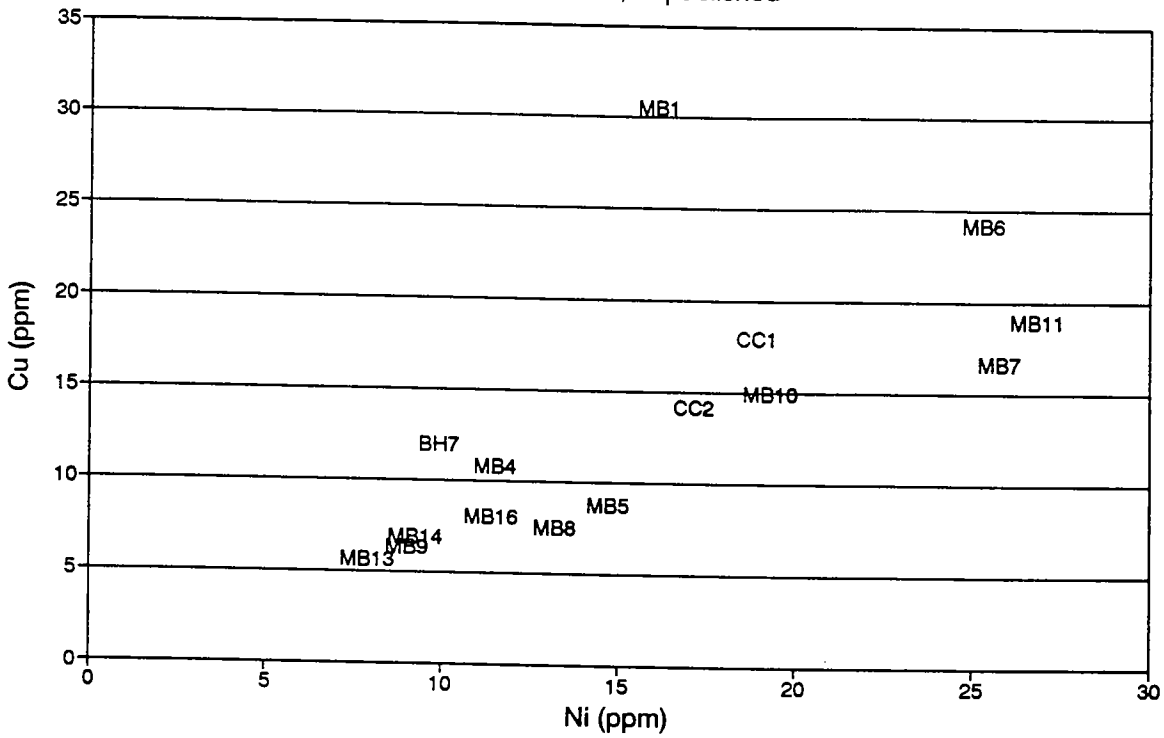


Figure C-13. Pb vs. Ni
Data from NOAA, unpublished

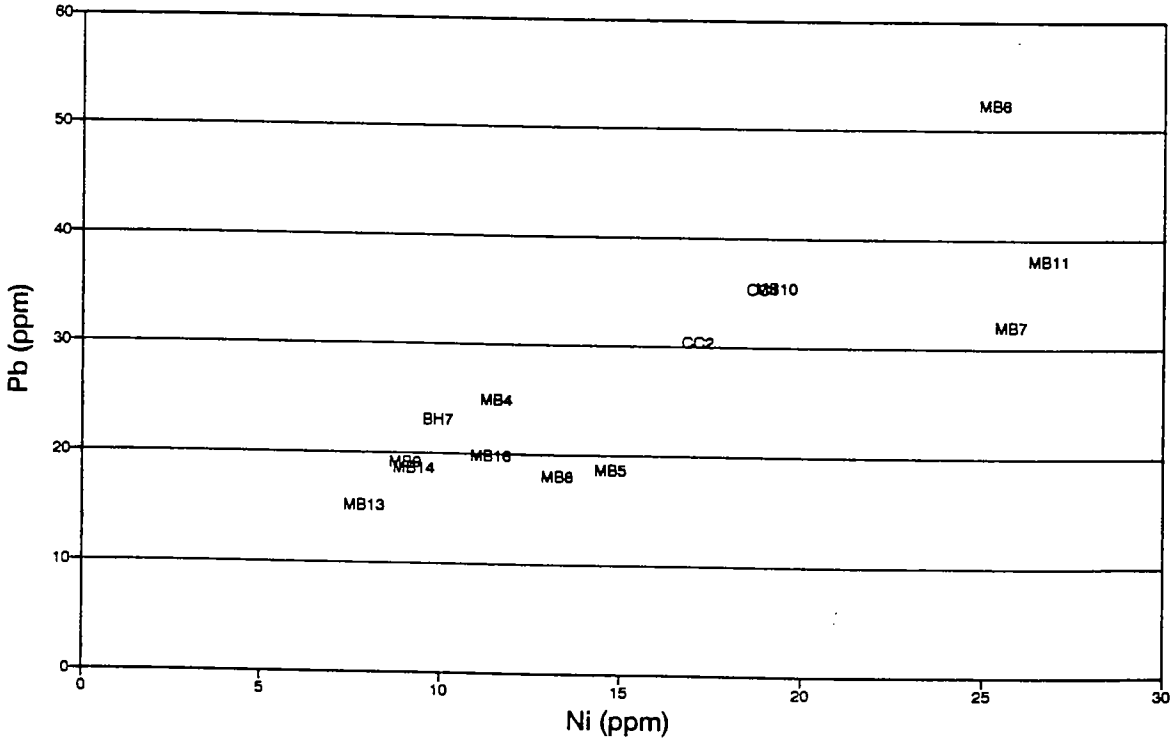


Figure C-14. Zn vs. Ni
Data from NOAA, unpublished

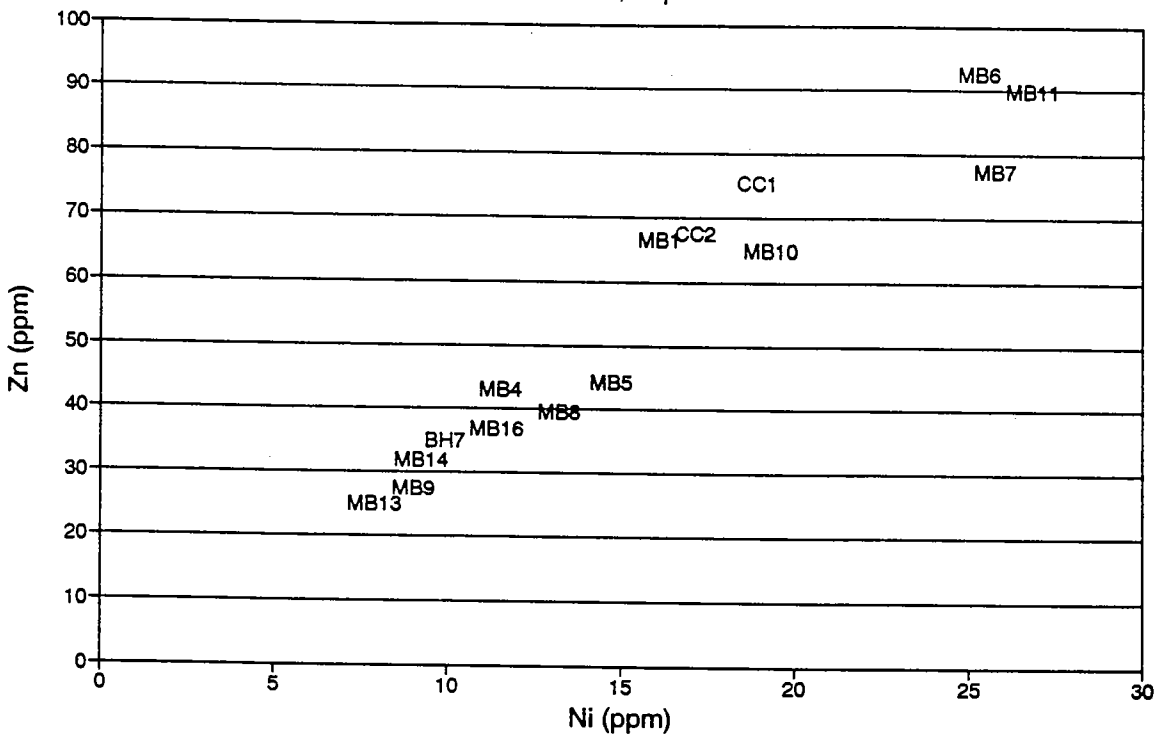


Figure C-15. Fe vs. Ni
Data from NOAA, unpublished

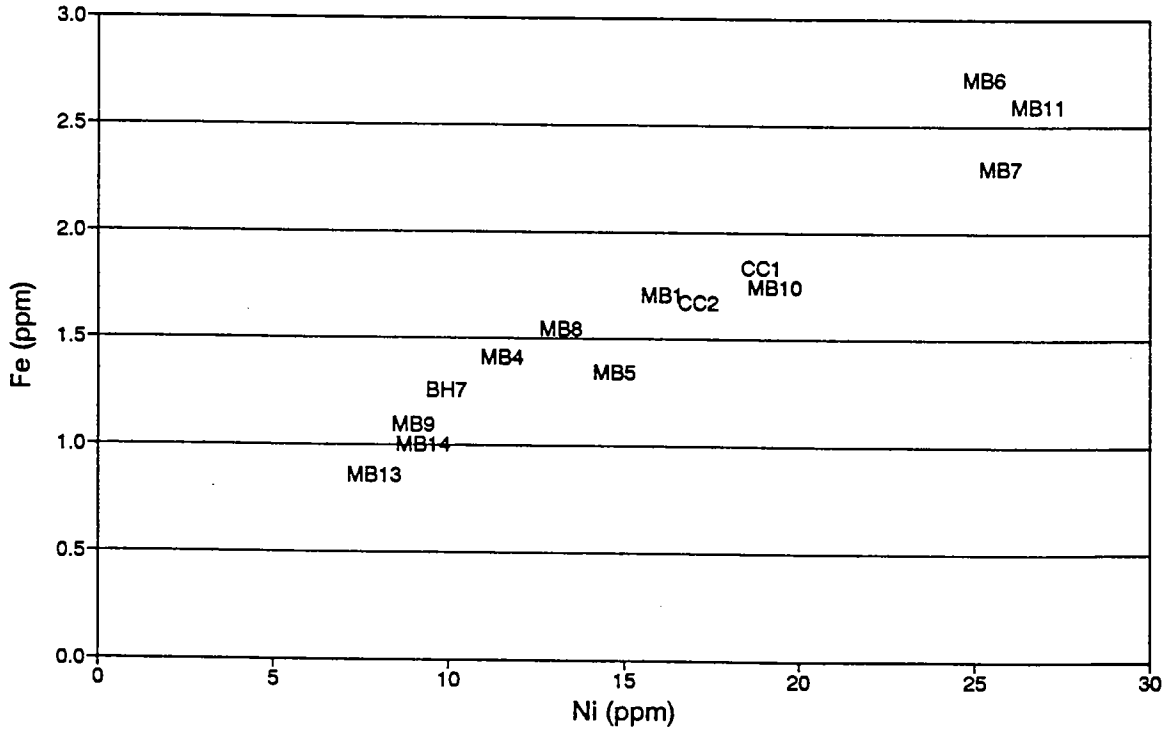


Figure C-16. Cd vs. Ni
Data from Gilbert et al., 1976

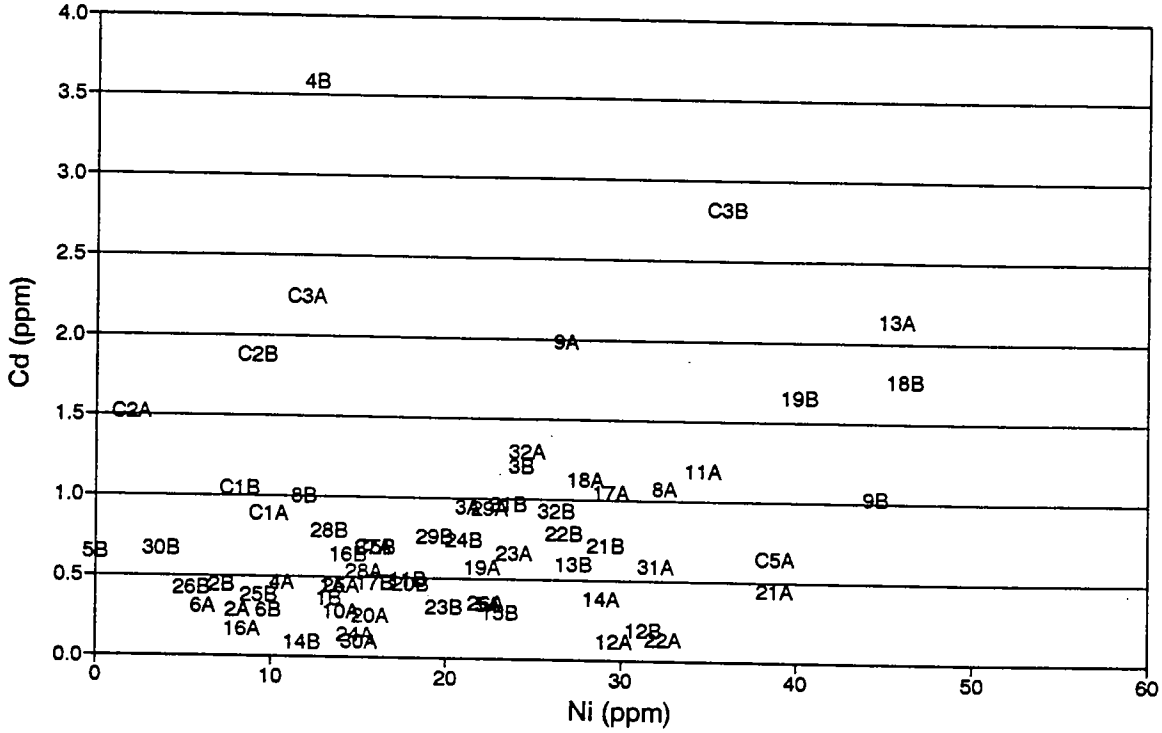


Figure C-17. Cr vs. Ni
Data from Gilbert et al., 1976

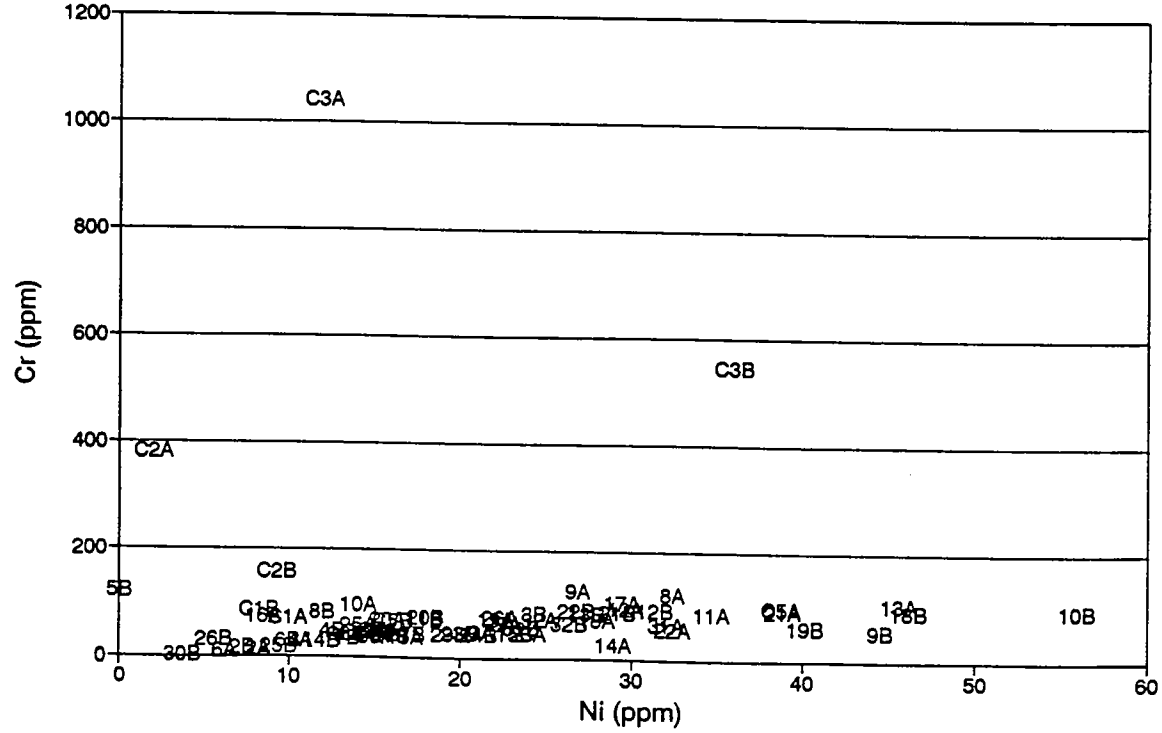


Figure C-18. Cu vs. Ni
Data from Gilbert et al., 1976

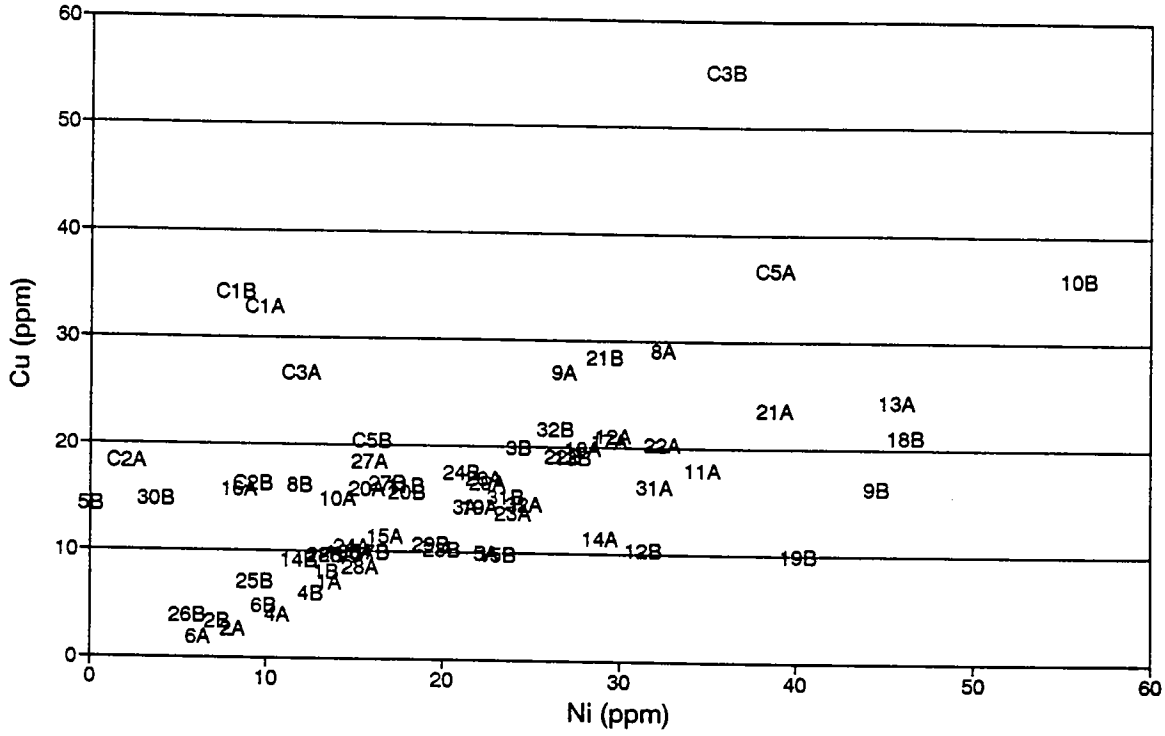


Figure C-19. Pb vs. Ni
Data from Gilbert et al., 1976

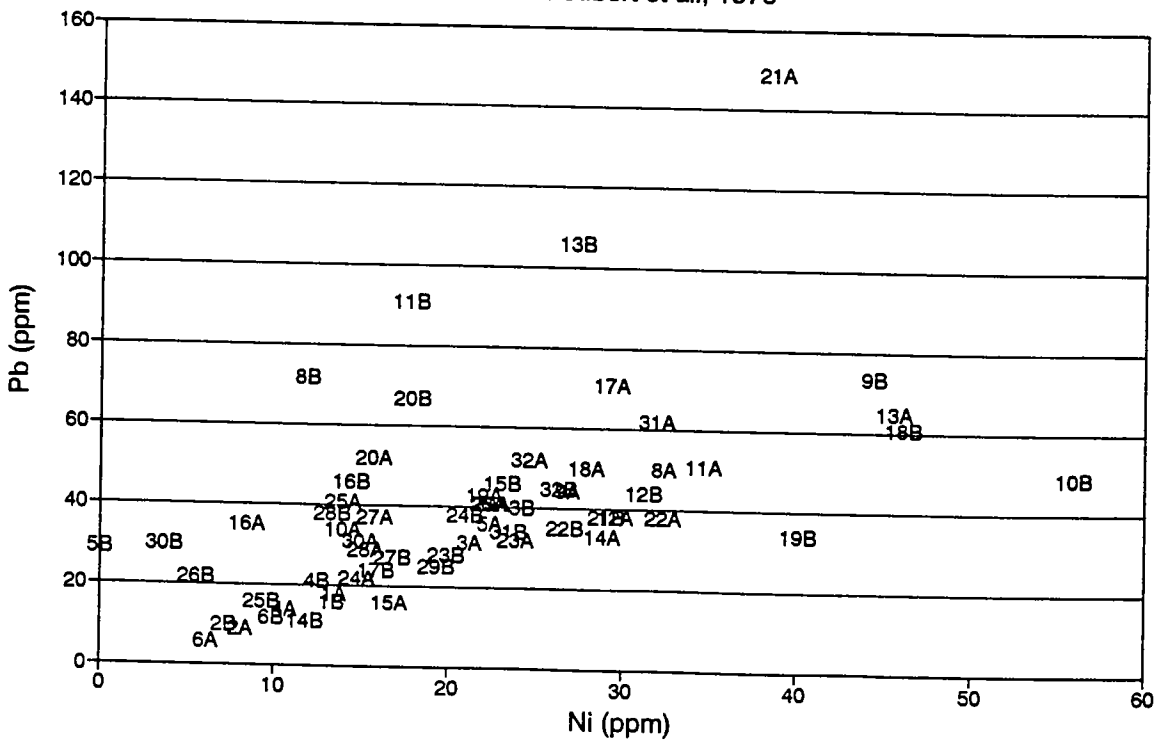


Figure C-20. Hg vs. Ni
Data from Gilbert et al., 1976

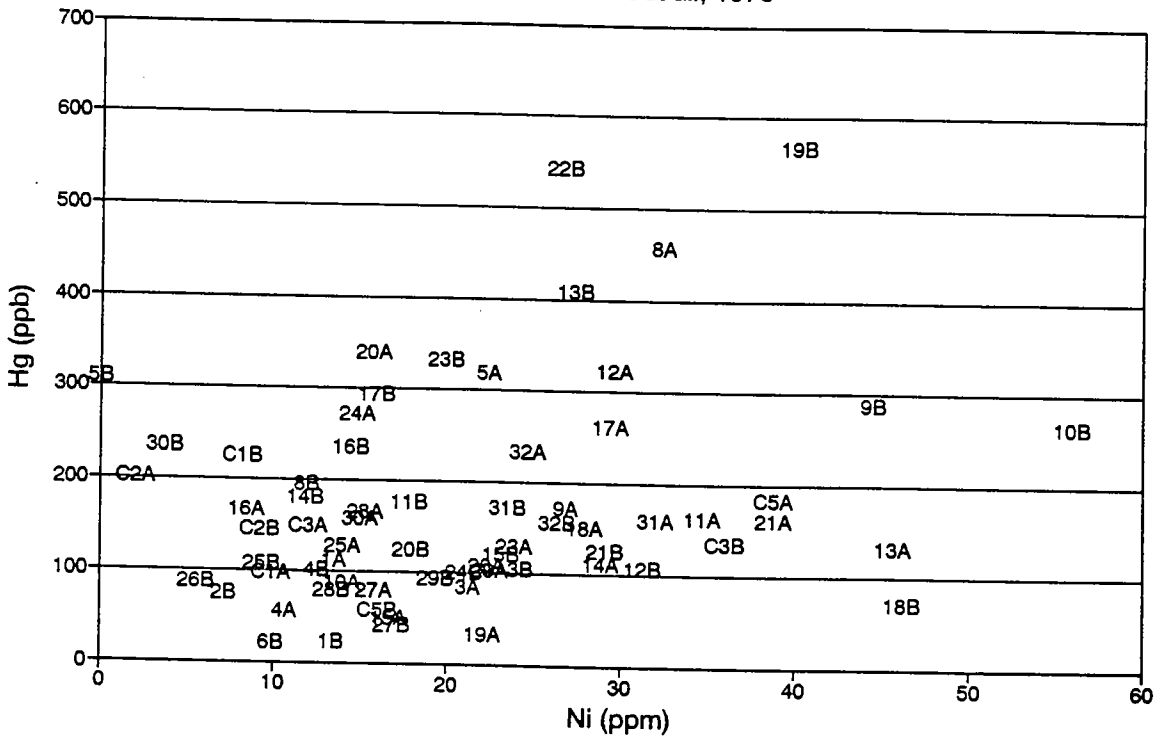


Figure C-21. Zn vs. Ni
Data from Gilbert et al., 1976

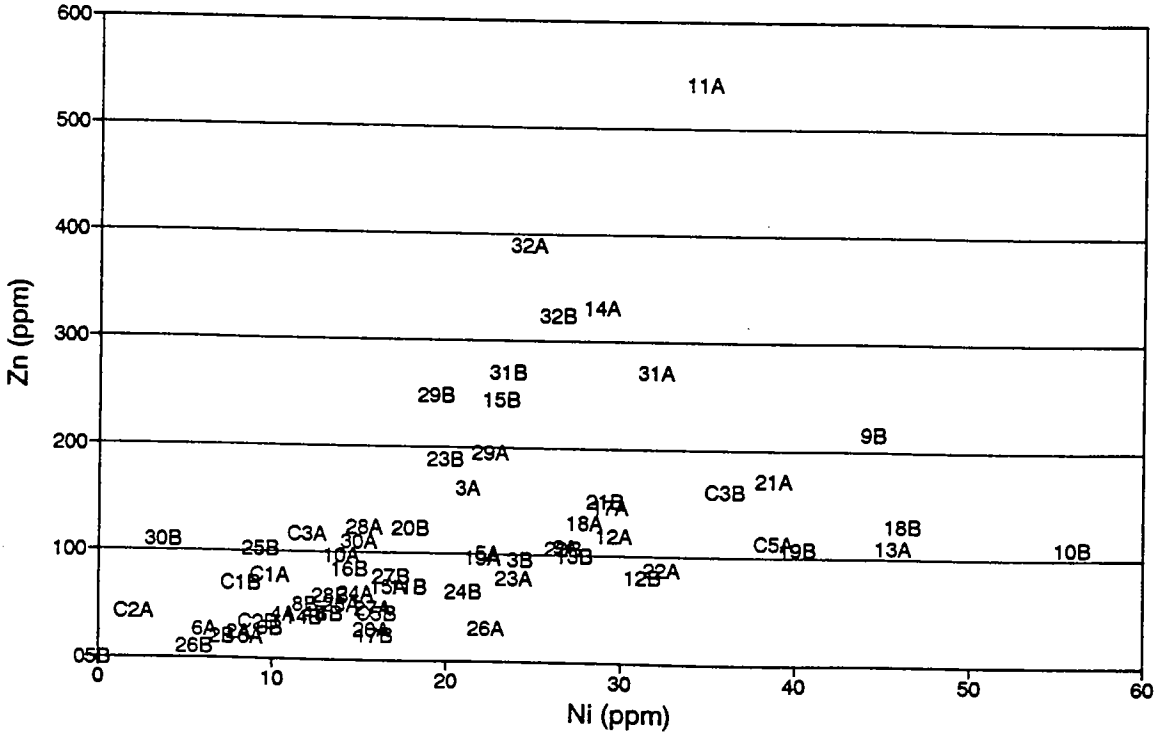
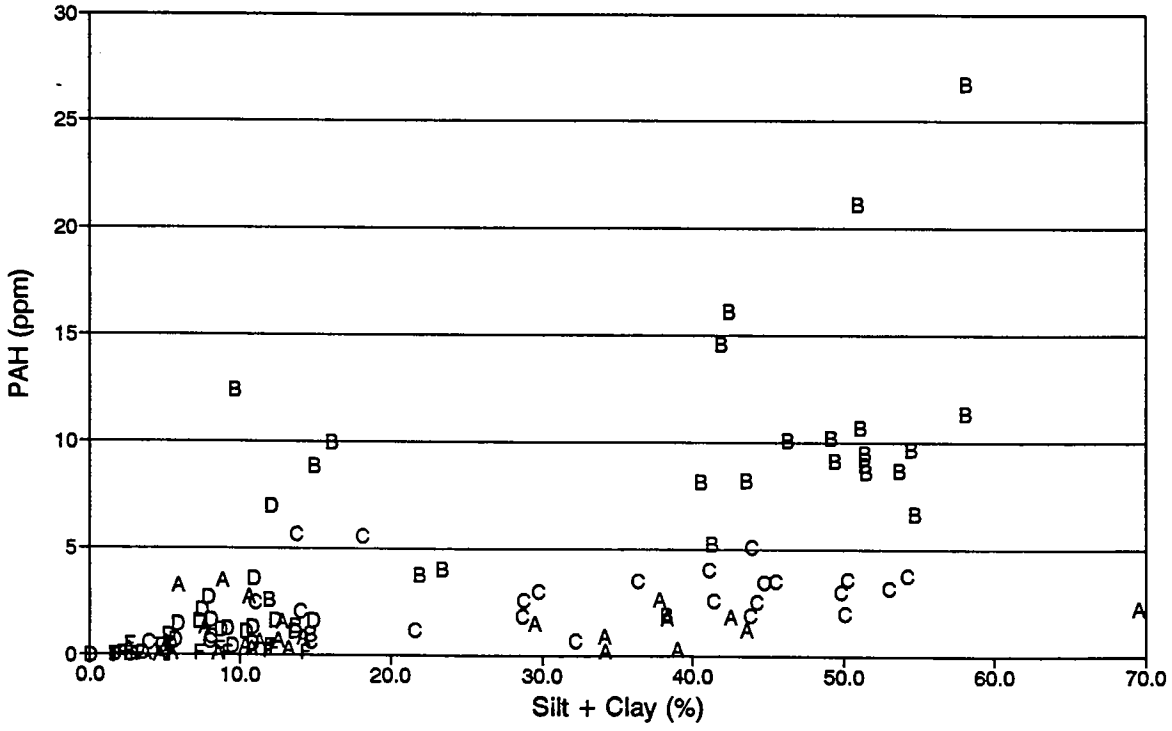


Figure C-22. PAH vs. Silt + Clay
Data are from Battelle, 1987b



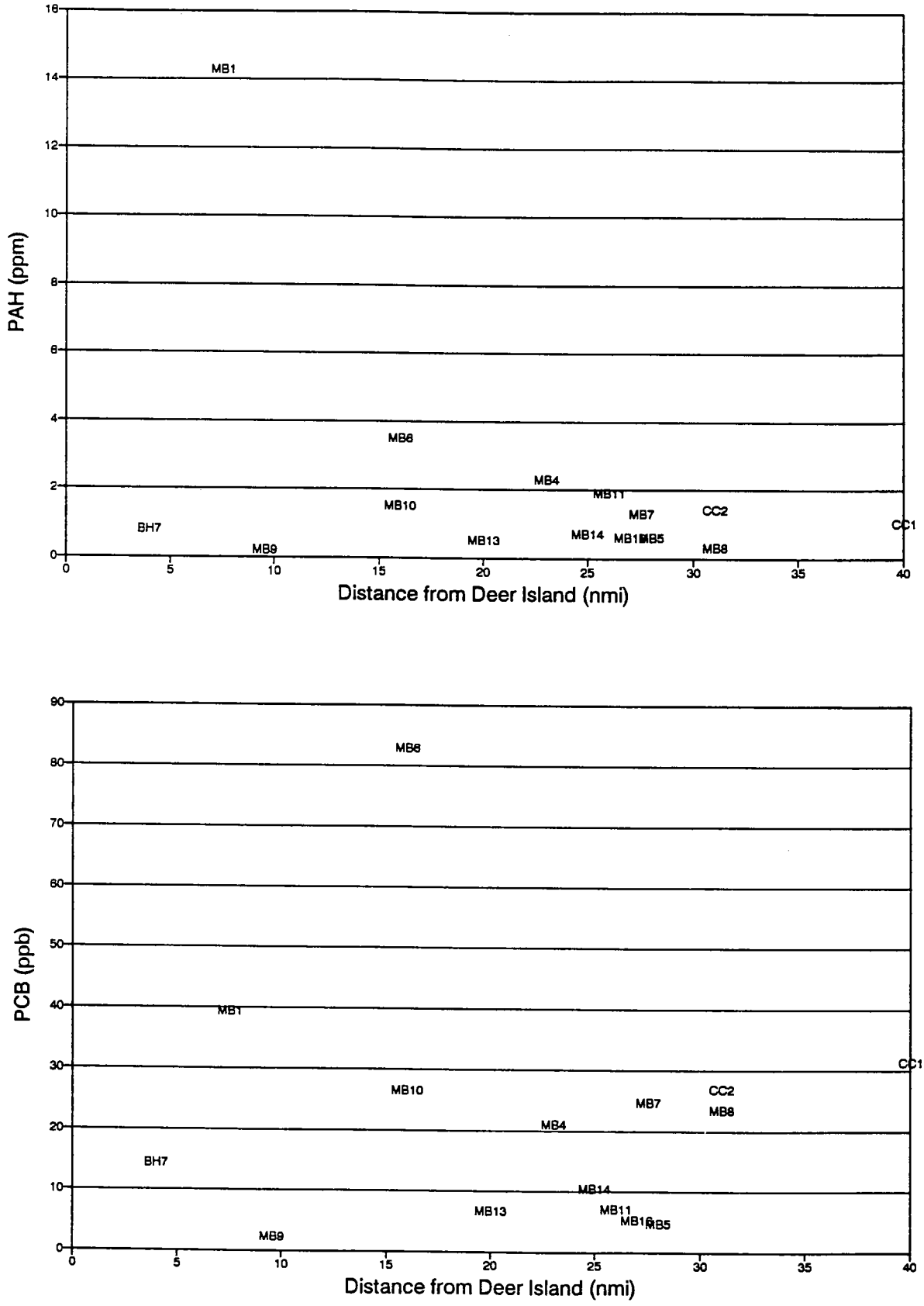


Figure C-23. PAH (Top) and PCB (Bottom) vs. Distance From Deer Island, Boston Harbor. Data are From Boehm *et al.*, 1984.

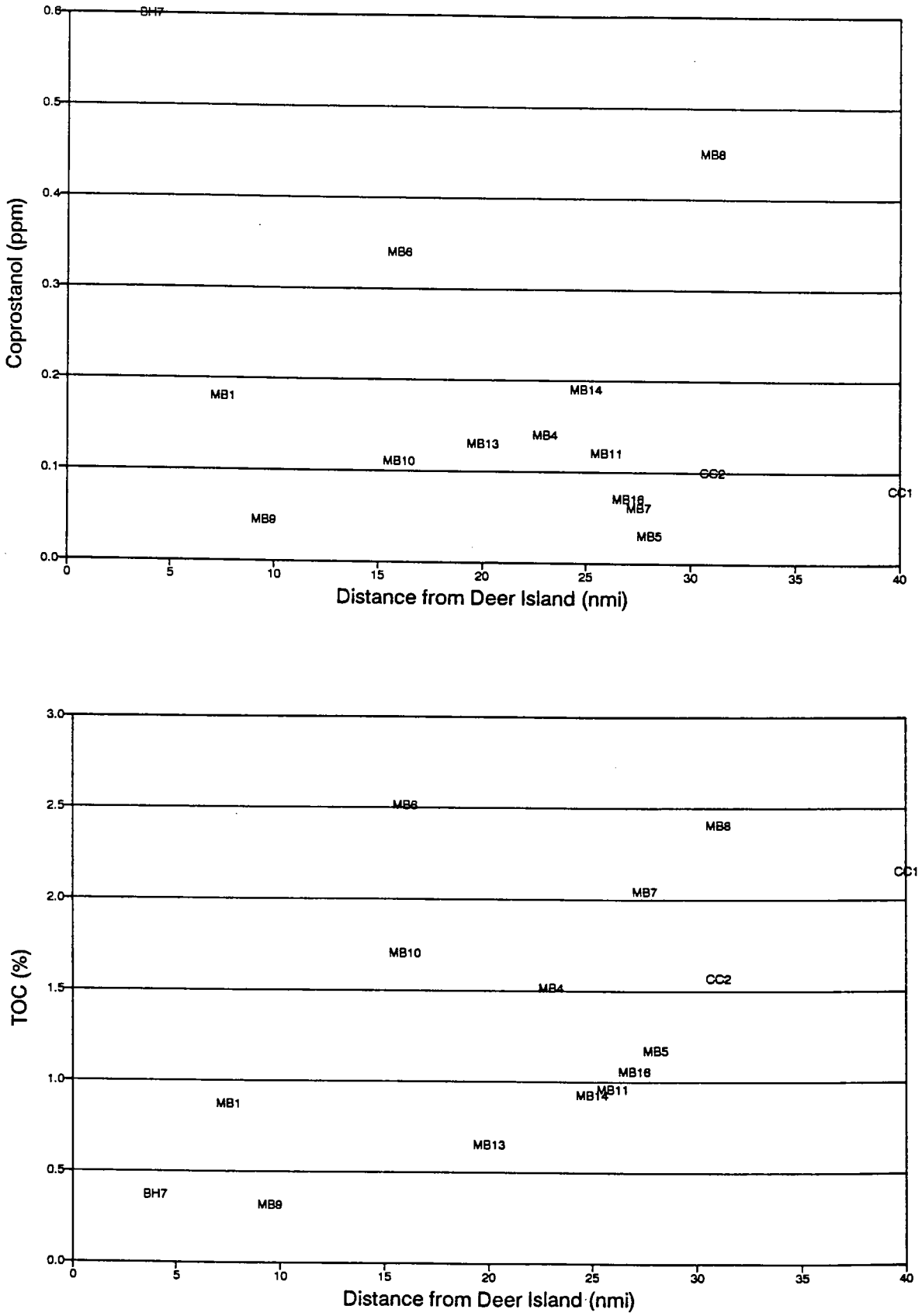


Figure C-24. Coprostanol (Top) and TOC (Bottom) vs. Distance From Deer Island, Boston Harbor. Data are From Boehm *et al.*, 1984.

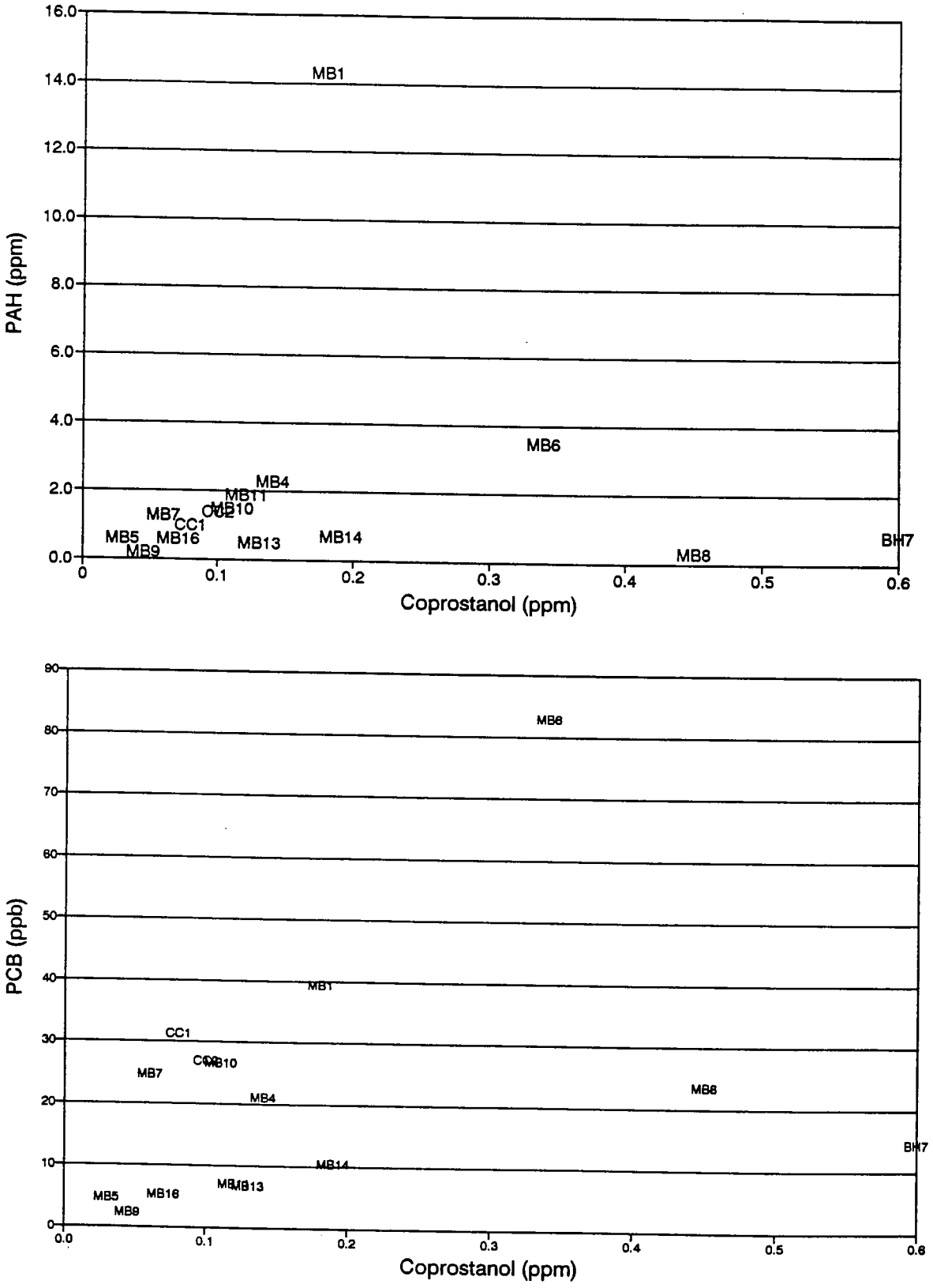


Figure C-25. PAH (Top) and PCB (Bottom) vs. Coprostanol. Data are From Boehm *et al.*, 1984.

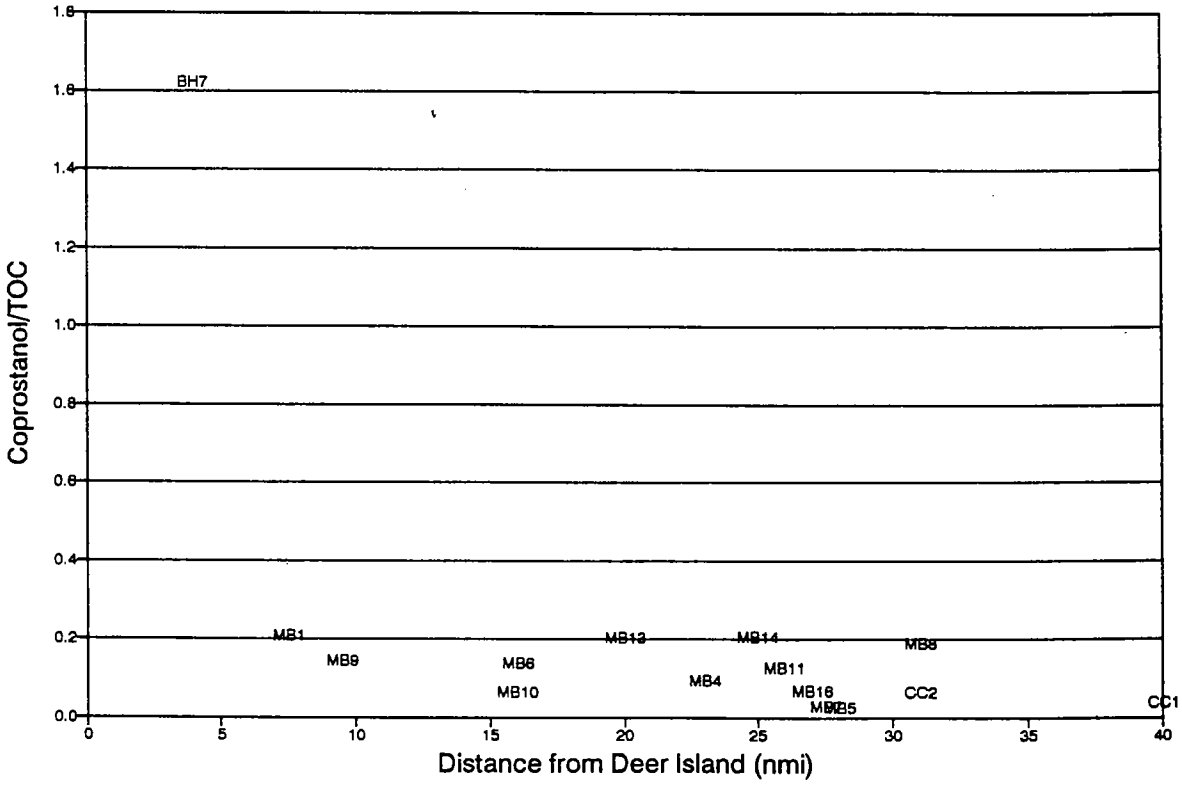


Figure C-26. Coprostanol/TOC vs. Distance From Deer Island, Boston Harbor. Data are From Boehm *et al.*, 1984.

Table C-1. Trace metal data from STFP Study (Battelle, 1987b), ppm- dry weight

Station	As	Cd	Cr	Cu	Hg	Ni	Pb	V	Zn	TOC* (%)	SILT+CLAY (%)
A1B11	3.02	0.376	55.4	13.4	0.282	6.07	20.6	14.26	37	0.36	38.3
A1B12	3.45	0.41	63.42	16.15	0.206	7.88	23.6	13.1	40.4	0.6	69.55
A1B13	3.14	0.365	50.71	11.73	0.113	7.33	19.1	11.42	35.2	0.49	42.48
A2B11	2.28	0.075	7.69	16.24	0.02	3.45	6	3.97	9.7	0.07	2.62
A2B12	2.23	0.168	19.99	4.48	0.08	6.96	10.2	6.96	18.5	0.15	10.3
A2B13	2.03	0.1	12.94	1.73	0.138	3.06	8.9	4.69	13.1	0.06	8.5
A3B11	3.03	0.189	21.71	7.88	0.068	4.85	8.7	7.28	25.1	0.3	11.26
A3B12	2.84	0.143	15.22	4.05	0.136	2.48	7.8	4.88	17.2	0.17	10.8
A3B13	2.73	0.301	30.54	12.43	0.21	4.89	12.9	7.05	26.8	0.21	14.16
B1B11	4.17	0.177	66.98	18.6	0.258	6.86	22.1	21.3	44.1	0.75	46.27
B1B12	4.19	0.177	63.71	18.82	0.298	7.25	22	23.47	43.4	0.66	49.14
B1B13	3.58	0.177	65.67	19.08	0.22	7.73	22.8	24.66	44.5	0.51	40.5
B2B11	3.93	0.222	77.4	23.05	0.427	8.7	25.6	25.59	51	0.5	41.79
B2B12	4.43	0.196	79.08	23.86	0.327	8.61	23.9	26.09	48.4	0.93	57.99
B2B13	4.13	0.175	70	21.14	0.258	8.06	24.6	23.44	46.8	1.21	51.01
B3B11	4.7	0.066	26.69	9.67	0.105	4.46	33.7	11.29	32.7	0.26	14.83
B3B12	3.59	0.116	44.74	16.65	0.178	6.92	20.8	11.6	40.5	1.73	9.57
B3B13	3.63	0.102	37.5	14.68	0.163	6.71	18.4	19.35	36.5	0.61	16.04
C1B11	5.92	0.121	71.99	20.52	0.257	10.85	27.8	29.03	51.6	1.21	54.12
C1B12	4.88	0.092	65.42	17.7	0.228	8.4	27	23.84	43.9	1.16	50.02
C1B13	4.95	0.06	74.75	20.22	0.237	9.43	28.5	26.37	46.6	0.75	45.44
C2B11	4.98	0.027	19.17	9.15	0.036	6.19	18.2	17.95	33.4	0.24	7.91
C2B12	4.7	0.031	23.32	7.93	0.09	6.7	43.9	17.68	30	0.24	3.87
C2B13	4.79	0.048	22.94	15.09	0.049	11.84	14.1	20.64	86.5	0.32	4.72
C3B11	6.2	0.092	53.12	17.68	0.181	9.67	23.1	3.3	46.8	0.71	28.61
C3B12	4.3	0.106	49.84	16.72	0.228	8.02	22.3	6.67	42.1	0.71	29.68
C3B13	5.47	0.102	51.75	44.62	0.443	11.1	41.6	2.48	48.7	0.85	44.18
D1B11	3.23	0.02	7.88	2.85	0.029	3.17	12.4	5.73	12.8	0.07	2.76
D1B12	4.13	0.012	7.67	2.98	0.029	3.61	12	5.32	13.5	0.05	3.33
D1B13	3.79	0.018	5.36	3.3	0.03	3.42	11.7	2.74	11.9	0.04	1.65
D2B11	3.51	0.318	29.92	7.36	0.117	7.55	22.8	20.39	107.6	0.83	12.23
D2B12	5.92	0.091	38.78	10.01	0.257	8.15	31.7	23.76	33	0.61	5.85
D2B13	5.32	0.072	35.49	8.54	0.097	6.71	22.9	23.12	29.9	0.43	14.77
D3B11	3.57	0.046	23.23	4.94	0.12	4.71	27.5	15.62	20.3	0.17	7.23
D3B12	3.9	0.042	23.44	5.07	0.081	5.74	20	14.74	22.2	0.28	10.3
D3B13	3.46	0.047	18.89	2.83	0.577	5.4	17.3	12.01	19.3	0.32	10.75
F1B11	7.76	0.026	19.79	2.14	0.033	9.29	15	27.87	40.9	0.05	2.95
F1B12	12.71	0.025	15.41	2.81	0.018	6.42	16.3	41.73	35.6	0.05	1.83
F1B13	6.73	0.049	20.29	2.39	0.024	9.55	15.3	39.78	36.8	0.06	2.59
F2B11	26.32	0.032	25.05	1.69	0.004	9.81	25.1	30.14	48	0.06	3.34
F2B12	27.19	0.028	24.79	3.85	0.003	9.39	25.3	36.44	50.5	0.05	3.28
F2B13	25.63	0.033	25.78	1.33	0.01	10.46	26.3	36.72	55	0.06	2.14
F4B11	2.83	0.066	23.14	5.81	0.145	6.73	14.2	16.49	27.8	0.44	14.24
F4B12	3.89	0.141	37.41	10.8	0.124	6	20.3	16.81	32.1	0.27	11.82
F4B13	2.46	0.054	20.14	4.34	0.166	6.36	13.4	16.43	25.9	0.18	9.1
A1B21	3.89	0.224	40.95	12.44	0.218	4.34	14.03	19.39	32.27	0.77	29.49
A1B22	3.43	0.245	38.66	12.76	0.258	4.25	17.68	14.35	31.19	0.5	43.59
A1B23	3.69	0.314	62.47	20.05	0.155	7.5	24.09	28.22	47.32	0.77	37.77
A2B21	2.22	0.057	13.13	4.55	0.034	2.02	6.23	10.88	13.47	0.07	5.45
A2B22	2.64	0.094	13.86	4.93	0.079	1.87	10.04	7.6	16.37	0.22	4.63
A2B23	1.95	0.077	9.47	3.71	0.174	2.53	9.15	6.19	12.31	0.1	3.74
A3B21	3.57	0.17	27.82	63.28	0.127	4.39	12.28	20.07	37.09	0.33	13.15
A3B22	3.41	0.124	17.1	7.82	0.1	3.93	11.49	9.38	20.18	0.64	12.53
A3B23	3.43	0.134	23.8	14.37	0.103	4.02	7.26	16.6	27.05	0.17	5.15
B1B21	5.87	0.186	62.97	19.09	0.238	8.92	24.79	30.68	46.02	0.12	53.64
B1B22	5.12	0.168	53.52	13.8	0.178	6.34	17.71	27.22	38.6	0.97	41.27
B1B23	4.82	0.195	67.25	19.28	0.379	7.22	24.03	23.67	43.98	0.57	54.36
B2B21	4.25	0.183	49.77	16.06	0.208	7.69	23.41	19.48	37.44	0.86	42.31
B2B22	4.93	0.155	55.74	16.6	0.203	7.01	20.1	29.42	41.12	0.54	54.62
B2B23	4.39	0.191	61.45	17.56	0.288	7.13	23.68	21.96	42.25	0.7	43.5
B3B21	3.88	0.047	24.39	9.01	0.091	5.47	18.62	18.46	29.35	0.33	14.61
B3B22	2.99	0.058	21.92	24.37	0.068	4.78	13.88	19.68	28.22	0.19	11.85
B3B23	3.29	0.058	49.11	9.24	0.318	21.81	10.98	15.11	27.01	0.21	13.61
C1B21	6.66	0.14	82.05	23.75	0.302	11.88	31.67	32.75	53.08	0.77	49.76
C1B22	6.76	0.094	55.02	16.84	0.299	8.59	25.15	32.44	43.08	0.89	52.94
C1B23	6.32	0.097	64.79	19.31	0.284	8.3	29.47	26	45.31	0.84	44.66
C2B21	6.08	0.078	30.1	23.58	0.093	9.43	251.07	28.3	42.86	0.41	32.11

*TOC: Total Organic Carbon

Table C-1 (continued). Trace metal data from STFP Study (Battelle, 1987b), ppm- dry weight

Station	As	Cd	Cr	Cu	Hg	Ni	Pb	V	Zn	TOC* (%)	SILT+CLAY (%)
C2B22	5.57	0.046	32.28	9.92	0.053	6.46	21.21	27.67	35.74	0.48	7.97
C2B23	5.48	0.048	26.74	9.68	0.078	5.27	261.62	17.12	30.09	0.36	14.65
C3B21	5.5	0.068	54.57	19.72	0.166	11.24	32.63	26.14	48.76	0.97	41.3
C3B22	5.66	0.173	47.24	19.41	0.1	13.97	25.81	29.75	152.51	0.48	41.01
C3B23	5.23	0.112	57.91	19.41	0.192	10.69	28.54	27.19	47.95	0.67	10.9
D1B21	3.57	0.015	6.9	9	0.019	2.3	11.88	10.04	11.88	0.04	2.39
D1B22	3.17	0.015	6.79	2.43	0.023	2.43	12.26	9.77	11.45	0.04	1.83
D1B23	3.86	0.016	7.34	1.09	0.031	2.58	15.27	10.59	12.19	0.04	2.09
D2B21	7.24	0.023	56.22	15.71	0.168	8.65	41.65	25.54	41.48	1.04	13.56
D2B22	5.74	0.04	31.55	14.44	0.112	5.7	40.48	17.92	29.71	1.58	9.44
D2B23	5.77	0.042	45.32	12.78	0.189	7.55	32.93	21.93	36.41	0.35	5.62
D3B21	3.99	0.027	15.96	5.05	0.143	3.29	18.08	11.08	16.79	0.24	10.8
D3B22	6.92	0.034	22.07	6.47	0.099	5.33	18.93	14.42	27.49	1.88	7.38
D3B23	5.65	0.028	21.51	5.91	0.065	4.43	30.37	13.02	20.04	0.3	8.67
F1B21	2.93	0.011	20.77	4.02	0.03	8.42	12.54	23.03	33.77	0.07	1.54
F1B22	4.02	0.016	13.64	1.84	0.029	5.77	13.55	12.03	26.05	0.06	1.42
F1B23	4.67	0.016	25.09	4.43	0.031	8.99	13.77	22.41	40.03	0.09	1.81
F2B21	29.03	0.012	30.02	3.9	0.014	12.91	24.45	46.17	62.19	0.07	1.85
F2B22	32.52	0.029	21.86	3.76	0.017	6.36	26.01	34.44	47	0.06	1.67
F2B23	29.37	0.015	13.56	4.12	0.032	2.83	25.4	39.89	62.11	0.06	1.89
F4B21	3.29	0.039	15.63	4.06	0.042	5.32	13.77	11.17	23.15	0.12	5.41
F4B22	3.66	0.036	14.89	3.47	0.066	4.65	13.12	10.17	21.07	0.15	5.52
F4B23	3.43	0.061	26.61	8.46	0.093	5.86	19.86	14.11	29.71	0.3	12.21
A1B31	3.42	0.197	44.24	15.6	0.184	5.55	18.9	13.85	33.06	0.61	34.17
A1B32	2.96	0.179	43.14	14.62	0.184	5.4	17.98	14.27	33.86	0.53	39
A1B33	2.99	0.215	51.82	17.15	0.088	6.22	20.56	17.22	37.86	0.71	34.08
A2B31	2.65	0.118	20.64	8.37	0.382	4.28	12.27	9.72	20.05	0.22	5.9
A2B32	2.46	0.091	17.29	8.65	0.239	3.79	11.37	8.76	17.29	0.22	8.82
A2B33	2.53	0.084	13.85	5.72	0.043	2.71	8.51	7.91	16.63	0.19	10.56
A3B31	0.41	0.104	20.88	8.26	0.074	5.66	16.13	10.91	27.04	0.41	12.75
A3B32	3.76	0.204	28.35	14.5	0.203	5.38	13.85	10.5	28.9	0.21	5.08
A3B33	3.4	0.192	30.14	14.24	0.116	5.1	15.07	13.44	30.04	0.36	7.66
B1B31	4.17	0.191	62.79	16.21	0.196	7.78	25.38	29.93	38.62	0.87	51.45
B1B32	3.78	0.093	58.58	16.56	0.172	6.84	22.5	29.41	35.54	0.62	49.34
B1B33	4.38	0.205	64.07	16.84	0.182	8.06	25.32	32.16	40.1	0.84	51.34
B2B31	5.23	0.032	33.06	8.86	0.133	6.05	27.89	27.7	26.84	0.77	50.84
B2B32	4.5	0.187	59.84	16.32	0.194	7.96	24.28	26.98	39.09	0.76	58.05
B2B33	4.45	0.118	52.69	18.66	0.279	8.15	26.25	17.81	41.54	0.64	51.35
B3B31	2.73	0.108	24.13	10.05	0.087	4.77	21.21	14.81	23.98	0.64	38.19
B3B32	4.21	0.073	29.17	12.85	0.15	6.18	21.19	14.67	32.13	0.3	21.84
B3B33	5.46	0.09	33.19	14.56	0.11	7.04	20.45	17.13	34.15	0.35	23.34
C1B31	6.28	0.09	59.89	17.92	0.203	9.55	27.48	33.94	43.57	0.7	13.72
C1B32	5.15	0.071	50.72	15.64	0.151	7.14	23.6	29.04	35.53	0.71	50.19
C1B33	5.22	0.13	65.86	17.69	0.229	9.91	28.56	36.55	43.27	0.76	43.85
C2B31	5.98	0.051	31.39	11.33	0.077	6.78	24.52	29.91	32.91	0.41	43.72
C2B32	5.51	0.051	40.36	13.79	0.07	12.47	20.48	38.88	42.79	0.38	21.48
C2B33	5.72	0.031	31.45	12.4	0.077	8.2	18.51	28.79	34.82	0.32	18.04
C3B31	5.68	0.078	56.05	31.8	0.241	9.08	32.89	37.41	43.7	0.49	28.66
C3B32	5.05	0.133	43.11	15.4	0.154	8.9	23.7	31.81	39.01	0.73	36.29
C3B33	5.38	0.056	48.14	15.62	0.143	9.5	22.59	32.09	38.53	0.62	13.96
D1B31	3.96	0.018	8.41	2.96	0.026	2.67	14.91	10.99	15.2	0.05	3.49
D1B32	3.93	0.016	7.88	3.17	0.024	2.85	11.91	9.89	13.83	0.08	5.29
D1B33	4.64	0.017	8.72	4.36	0.033	3.4	13.55	8.62	15.25	0.05	4.75
D2B31	5.86	0.041	27.37	10	0.086	5.5	20.29	16.8	28.6	0.27	8.03
D2B32	4.87	0.038	21.35	7.41	1.043	5.42	23.23	18.09	25.4	0.42	10.88
D2B33	6.29	0.053	49.22	16.84	0.187	8.85	33.67	23.07	43.77	1.45	12.03
D3B31	3.59	0.019	13.62	4.51	0.042	3.75	15.96	10.32	17.08	0.31	5.25
D3B32	4.3	0.031	20.16	4.53	0.055	4.25	16.46	18.79	18.59	0.47	7.87
D3B33	4.59	0.029	21.84	6.6	0.087	4.4	16.94	19.74	20.33	0.41	9.05
F1B31	16.02	0.02	21.26	7.78	0.018	10.31	20.07	30.7	50.97	0.07	2
F1B32	3.84	0.018	14.38	4.76	0.036	5.92	13.06	13.06	26.02	0.08	2.65
F1B33	4.5	0.015	14.14	4.46	0.051	5.47	13.19	12.24	28.37	0.06	1.72
F2B31	29.56	0.015	19.6	4.09	0.02	7.67	23.86	44.26	42.69	0.07	1.83
F2B32	32.96	0.027	21.27	3.16	0.017	8.65	23.89	48.07	43.18	0.07	2.55
F2B33	39.54	0.024	23.83	5.29	0.031	6.62	27.99	47.58	45.38	0.08	1.9
F4B31	3.43	0.058	23.16	9.81	0.165	5.93	16.07	12	29.11	0.16	8.57
F4B32	3.55	0.05	22.55	8.9	0.08	7.02	17.5	11.87	29.57	0.19	11.77
F4B33	4.99	0.042	21.28	7.72	0.063	6.93	17.02	14.15	29.69	0.15	7.27

*TOC: Total Organic Carbon

Table C-2. Trace metal data from NOAA-NEMP Study (NOAA, unpublished data), ppm- dry weight

Station	Ag	Cd	Cr	Cu	Fe	Ni	Pb	Zn	TOC* (%)
BH07	0.58	^	29.833	11.99	1.2567	9.9333	23.067	35.033	3.7
CC01	0.8	0.75	39.42	17.98	1.834	18.92	35.34	75.42	2.152
CC02	0.558	0.36	43.44	14.18	1.67	17.12	30.44	67.52	1.564
MB01	0.87	0.68	52.58	30.52	1.706	16.12	120.56	66.62	0.874
MB04	^	0.7	42.55	10.843	1.4125	11.553	24.9	43.175	1.51
MB05	^	0.65333	24.28	8.746	1.34	14.72	18.66	44.34	1.166
MB06	0.594	0.8	75.5	24.18	2.72	25.28	52.32	92.6	2.51
MB07	^	^	44.18	16.64	2.302	25.74	31.98	77.36	2.042
MB08	^	^	21.26	7.472	1.546	13.22	17.94	39.66	2.404
MB09	^	^	18.26	6.336	1.092	9.018	19.18	27.46	0.314
MB10	^	0.47	56.8	14.95	1.745	19.3	35.5	64.95	1.704
MB11	2.71	0.562	50.474	18.98	2.596	26.8	38.16	89.92	0.956
MB13	0.355	0.4375	20.26	5.658	0.86	7.886	15.08	24.98	0.648
MB14	1.02	^	24.6	6.922	1.004	9.242	18.66	31.94	0.926
MB16	0.276	0.285	28.7	8.076	^	11.4	19.8	37.06	1.058

*TOC: Total Organic Carbon

^: No Data

Table C-3. Trace metal data from Salem Harbor (Gardner et al., 1986), ppm- dry weight

Station	Al (mg/g)	Cd	Cr	Cu	Fe (mg/g)	Hg	Mn	Ni	Pb	ZN	TOC* (%)
1A	3.435	3.345	619	599	4.95	0.873	59.8	36.15	497.5	363	^
1B	7.79	15.75	5895	189.5	3.1	1.882	86.45	24.55	326	493.5	9.9
1E	8.76	10.15	3045	100	4.3	1.507	105.8	20.35	175.5	326.5	6.6
1D	5.35	3.44	1048	37.4	3.35	0.54	78.05	10.75	60	114.5	3.3
1C	3.025	1.195	1588.5	12.35	2.55	0.245	46.25	6.525	31.05	59.5	1.1
6	2.83	0.07045	40.3	3.7	3.2	0.056	92.65	4.995	15.15	24.9	0.54
8	9.16	0.119	67.85	13	8.5	0.216	172	15	32.2	76.9	2.2

*TOC: Total Organic Carbon

^: No Data

Table C-4. Trace metal data from New England Aquarium Survey (Gilbert et al., 1976), ppm- dry weight

Sample	Cd	Cr	Cu	Pb	Hg^	Ni	Zn
1A	0.45	41	7.2	18	112	13.5	*
1B	0.37	35	8.1	16	23	13.4	43
2A	0.29	15	2.6	9	*	8.1	24
2B	0.45	17	3.4	10	76	7.2	20
3A	0.95	48	14.2	31	85	21.2	162
3B	1.21	82	19.8	40	105	24.3	95
4A	0.46	29	4	14	57	10.6	41
4B	3.59	49	6.1	21	102	12.5	42
5A	0.34	64	10	36	319	22.4	101
5B	0.65	122	14.3	29	310	ND	ND
6A	0.31	10	1.9	6	*	6.1	26
6B	0.29	28	4.9	12	22	9.9	28
7A	No Data						
7B	No Data						
8A (C)	1.08	121	29.1	50.03	457	32.4	2495
8B	1.01	82	16.2	71.92	196	11.9	51
9A	1.99	126	27	44.27	172	26.8	108
9B	1.03	55	16.3	73.49	288	44.5	216
10A	0.28	97	14.9	34	89	14	97
10B	*	93	36	49	264	55.9	110
11A (C)	1.19	85	17.9	51	162	34.7	541
11B	0.49	69	16.2	91	177	17.8	68
12A (C)	0.12	94	21	38	323	29.6	119
12B	0.19	91	10.4	44	107	31.3	80
13A	2.14	104	24.5	65	132	45.6	110
13B	0.6	83	19	106	409	27.3	100
14A (C)	0.38	26	11.4	33	111	28.9	332
14B	0.09	28	9.2	11	181	11.8	39
15A	*	37	11.5	16	50	16.7	68
15B	0.29	46	9.8	46	121	23.1	245
16A	0.17	74	15.7	35	168	8.4	20
16B	0.64	51	9.9	46	236	14.4	84
17A	1.05	105	20.6	71	262	29.4	146
17B	0.47	39	9.9	24	294	15.9	23
18A	1.13	73	19.8	50	151	27.9	131
18B	1.77	92	21.1	61	71	46.1	131
19A	0.57	71	14.2	43	33	22.1	96
19B	1.66	61	9.9	34	568	40.1	108
20A	0.26	70	15.9	52	341	15.7	29
20B	0.46	74	15.5	67	125	17.9	123
21A	0.45	91	23.6	149	161	38.8	172
21B	0.72	87	28.4	38	126	29.1	151
22A (C)	0.13	57	20.3	38	4240	32.4	88
22B	0.79	90	19.1	35	544	26.7	106
23A	0.66	47	13.7	32	130	23.9	78
23B (C)	0.32	46	10.3	28	333	19.9	189
24A	0.14	45	10.5	22	273	14.8	62
24B	0.74	45	17.5	38	101	21	65
25A	0.45	59	9.6	41	129	14	51
25B	0.38	18	7.2	16	109	9.3	103
26A	0.35	76	16.9	41	108	22.3	31
26B	0.43	32	3.9	22	88	5.5	10
27A	0.69	42	18.4	37	80	15.8	48
27B	*	41	16.4	27	42	16.8	78
28A	0.54	48	8.5	29	166	15.3	124
28B	0.79	41	9.6	38	80	13.3	60
29A	0.94	66	16.5	41	103	22.5	195
29B (C)	0.76	44	10.7	25	94	19.3	248
30A	0.1	37	10	31	159	15	110
30B	0.67	3	14.8	30	237	3.7	110
31A	0.59	66	16.3	62	160	32	271
31B	0.97	58	15.2	34	172	23.5	270
32A (C)	1.3	73	14.6	52	234	24.6	390
32B (C)	0.93	66	21.7	45	156	26.3	324
C1A	0.89	73	32.8	55	98	9.8	78
C1B	1.05	87	34.3	55	227	8.2	70
C2A	1.52	382	18.3	45	203	2	42
C2B	1.88	157	16.3	23	146	9.2	34
C3A	2.25	1042	26.7	71	150	11.9	117
C3B	2.82	545	55.2	76	135	36	161
C4A	No data						
C4B	No data						
C5A	0.65	96	36.7	35	184	38.7	113
C5B	0.69	68	20.4	21	58	16	44

^: ppb- dry weight.
*: Below detection limit (<0.10 ppm Cd, <10 ppb Hg, and <9 ppm Zn).
ND: Not determined.
(C): Contamination suspected.

Table C-5. Organic contaminant data from STFP Study (Battelle, 1987b), dry weight

Station	Total PAH (ppm)	FFPI*	TOC** (%)	Silt+Clay (%)
A1B11	1.73	17.85	0.36	38.3
A1B12	2.23	20.66	0.6	69.55
A1B13	1.82	19.5	0.49	42.48
A1B21	1.49	15.51	0.77	29.49
A1B22	1.2	15.79	0.5	43.59
A1B23	2.63	19.05	0.77	37.77
A1B31	0.19	9.57	0.61	34.17
A1B32	0.28	12.68	0.53	39
A1B33	0.87	14.82	0.71	34.08
A2B11	0.25	10.82	0.07	2.62
A2B12	0.41	14.41	0.15	10.3
A2B13	0.1	6.19	0.06	8.5
A2B21	0.06	9.09	0.07	5.45
A2B22	0.05	13.04	0.22	4.63
A2B23	0.06	10.34	0.1	3.74
A2B31	3.23	21	0.22	5.9
A2B32	3.47	21.01	0.22	8.82
A2B33	2.69	20.92	0.19	10.56
A3B11	0.63	14.56	0.3	11.26
A3B12	0.28	9.36	0.17	10.8
A3B13	0.86	24.4	0.21	14.16
A3B21	0.278	29.856	0.33	13.15
A3B22	0.67	17.24	0.64	12.53
A3B23	0.56	13.35	0.17	5.15
A3B31	1.52	21.93	0.41	12.75
A3B32	0.46	18.29	0.21	5.08
A3B33	1.26	22.16	0.36	7.66
B1B11	10.06	12.8	0.75	46.27
B1B12	10.17	12.2	0.66	49.14
B1B13	8.16	15.95	0.51	40.5
B1B21	8.69	20.54	0.12	53.64
B1B22	5.22	11.6	0.97	41.27
B1B23	9.65	17.74	0.57	54.36
B1B31	8.61	17.4	0.87	51.45
B1B32	9.13	14.3	0.62	49.34
B1B33	9.47	12.9	0.84	51.34
B2B11	14.6	16.73	0.5	41.79
B2B12	11.31	16.33	0.93	57.99
B2B13	10.67	17.22	1.21	51.01
B2B21	16.11	20.54	0.86	42.31
B2B22	6.65	17.24	0.54	54.62
B2B23	8.21	17.63	0.7	43.5
B2B31	21.09	20.41	0.77	50.84
B2B32	26.77	22.93	0.76	58.05
B2B33	8.97	15.18	0.64	51.35
B3B11	8.85	23.22	0.26	14.83
B3B12	12.38	25.42	1.73	9.57
B3B13	9.97	20.81	0.61	16.04
B3B21	0.94	13.82	0.33	14.61
B3B22	2.54	16.54	0.19	11.85
B3B23	1.34	13.86	0.21	13.61
B3B31	1.87	9.69	0.64	38.19
B3B32	3.73	16.13	0.3	21.84
B3B33	3.98	14.23	0.35	23.34
C1B11	3.73	13.96	1.21	54.12
C1B12	1.96	13.91	1.16	50.02
C1B13	3.48	11.78	0.75	45.44
C1B21	3	14.47	0.77	49.76
C1B22	3.12	14.74	0.89	52.94
C1B23	3.44	14.76	0.84	44.66
C1B31	5.66	22.93	0.7	13.72
C1B32	3.57	13.39	0.74	50.19
C1B33	5.1	16.06	0.76	43.85
C2B11	0.6	11.52	0.24	7.91
C2B12	0.56	10.4	0.24	3.87
C2B13	0.41	8.99	0.32	4.72
C2B21	0.68	11.63	0.41	32.11
C2B22	0.86	20.21	0.48	7.97
C2B23	0.6	10.52	0.36	14.65
C2B31	1.88	14.23	0.41	43.72
C2B32	1.18	8.09	0.38	21.48
C2B33	5.57	21.43	0.32	18.04
C3B11	1.82	12.8	0.71	28.61
C3B12	2.99	12.84	0.71	29.68
C3B13	2.49	12.25	0.85	44.18
C3B21	2.53	16.51	0.97	41.3

*FFPI: Fossil Fuel Pollution Index
**TOC: Total Organic Carbon
^: No Data

Table C-5 (continued). Organic contaminant data from STFP Study (Battelle, 1987b), dry weight

Station	Total PAH (ppm)	FFPI*	TOC** (%)	Silt+Clay (%)
C3B22	3.98	24.74	0.48	41.01
C3B23	2.47	8.74	0.67	10.9
C3B31	2.55	12.5	0.49	28.66
C3B32	3.48	11.11	0.73	36.29
C3B33	2	10.86	0.62	13.96
D1B11	0.01	16.07	0.07	2.76
D1B12	^	^	0.05	^
D1B13	0.01	8.33	0.04	1.65
D1B21	^	^	0.04	2.39
D1B22	^	^	0.04	1.83
D1B23	^	^	0.04	2.09
D1B31	0.06	7.94	0.05	3.49
D1B32	0.44	8.5	0.08	5.29
D1B33	0.38	7.58	0.05	4.75
D2B11	1.56	22.31	0.83	12.23
D2B12	1.41	25.99	0.61	5.85
D2B13	1.57	18.24	0.43	14.77
D2B21	1.1	15.1	1.04	13.56
D2B22	0.4	12.03	1.58	9.44
D2B23	0.65	10.97	0.35	5.62
D2B31	1.57	10.35	0.27	8.03
D2B32	3.59	14.14	0.42	10.88
D2B33	6.98	25.54	1.45	12.03
D3B11	1.55	20.54	0.17	7.23
D3B12	1.04	18.67	0.28	10.3
D3B13	1.27	20.41	0.32	10.75
D3B21	0.52	16.28	0.24	10.8
D3B22	2.07	10.46	1.88	7.38
D3B23	1.13	13.01	0.3	8.67
D3B31	0.89	14.49	0.31	5.25
D3B32	2.66	15.26	0.47	7.87
D3B33	1.19	12.23	0.41	9.05
F1B11	0.03	5.36	0.05	2.95
F1B12	^	^	0.05	1.83
F1B13	0.01	5.05	0.06	2.59
F1B21	^	16.67	0.07	1.54
F1B22	^	^	0.06	1.42
F1B23	^	16.67	0.09	1.81
F1B31	^	^	0.07	2
F1B32	0.44	12.44	0.08	2.65
F1B33	^	^	0.06	1.72
F2B11	0.02	8.7	0.06	3.34
F2B12	0.01	6.25	0.05	3.28
F2B13	0.06	5.45	0.06	2.14
F2B21	0.02	11.1	0.07	1.85
F2B22	0.01	10	0.06	1.67
F2B23	0.01	^	0.06	1.89
F2B31	0.01	4.55	0.07	1.83
F2B32	^	^	0.07	2.55
F2B33	0.02	2.27	0.08	1.9
F4B11	0.145	22.759	0.44	14.24
F4B12	0.392	27.551	0.27	11.82
F4B13	0.095	16.842	0.18	9.1
F4B21	0.018	22.222	0.12	5.41
F4B22	0.024	16.667	0.15	5.52
F4B23	0.279	18.638	0.3	12.21
F4B31	0.235	15.319	0.16	8.57
F4B32	0.197	15.228	0.19	11.77
F4B33	0.087	11.494	0.15	7.27

*FFPI: Fossil Fuel Pollution Index
**TOC: Total Organic Carbon
^: No Data

Table C-6. Organic contaminant data from NOAA-NEMP Study (Boehm et al., 1984), dry weight

Station	Total PAH (ppm)	Coprostanol (ppm)	Total PCB (ppb)	FFPI*	TOC** (%)	Distance From Deer Island (nmi)
BH07	0.8	0.6	14.4	14	0.37	4
CC01	1	0.08	31.3	37	2.152	40
CC02	1.4	0.1	26.9	48	1.564	31
MB01	14.3	0.18	39.3	31	0.87	7.5
MB04	2.3	0.14	21	20	1.51	23
MB05	0.6	0.03	4.6	16	1.17	28
MB06	3.5	0.34	82.9	38	2.51	16
MB07	1.3	0.06	24.7	32	2.042	28
MB08	0.3	0.45	23.4	9	2.404	31
MB09	0.2	0.045	2.3	16	0.314	9.5
MB10	1.5	0.11	26.5	26	1.704	16
MB11	1.9	0.12	7	18	0.956	26
MB13	0.5	0.13	6.7	14	0.648	20
MB14	0.7	0.19	10.3	16	0.926	25
MB16	0.6	0.07	5.2	18	1.058	27

*FFPI: Fossil Fuel Pollution Index
**TOC: Total Organic Carbon

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