Annual review of toxic contaminants discharged by MWRA: 1993

Massachusetts Water Resources Authority

Environmental Quality Department
Technical Report Series No. 93-18

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Executive Summary

The MWRA is implementing one of the most comprehensive discharge monitoring projects in the country. An important component of the MWRA monitoring plan is the periodic review and synthesis of monitoring data and information from complementary studies to evaluate projections of environmental impact and the adequacy of the monitoring design to detect meaningful changes in the environment. This report represents the first annual review of toxic contaminant monitoring data to serve this purpose.

Toxic Contaminants in the MWRA Effluent

Accurate and precise estimates of concentrations and loadings from the MWRA effluent are the foundation for assessing potential impact resulting from the effluent discharge. Results from a detailed effluent characterization study in 1993 confirmed previous reports that the concentrations of many toxic contaminants in the MWRA effluent are much lower than they were during the 1980s. Concentrations of metals (e.g., copper, cadmium, and lead) are generally about one-half the values projected by EPA in the Supplementary Environmental Impact Statement (SEIS) and concentrations of some metals are now only one-tenth the values reported in the early 1980s. These new metals data are in good agreement with recent discharge permit monitoring that is required by EPA. Concentrations of many organic contaminants, such as polycyclic aromatic hydrocarbons (PAH) and polychlorinated biphenyls (PCB), are 10 to 100 times lower than those reported in the SEIS and also are much lower than those reported in recent discharge permit monitoring.

Potential Biological Effects Resulting from Future Effluent Discharge

To help determine whether any adverse biological effects will result from effluent discharge through the new offshore outfall, the MWRA is establishing a baseline of physical, chemical, and biological characteristics throughout Massachusetts Bay and modeling how these conditions could change. The most recent effluent contaminant concentrations and dilution models indicate that water column concentrations of contaminants will remain within a factor of two of present concentrations very near the outfall diffuser. Concentrations of Cu, Pb, Ag, some lighter weight PAH (naphthalenes), dieldrin (summer only), and lindane will increase at least two-fold at the edge of the mixing zone (about 60 m from the diffuser). The increase will be greatest for naphthalenes (34-fold increase), dieldrin (six-fold), and Ag (five-fold). However, naphthalenes are not persistent in the water and dieldrin, Ag, and the other contaminants will partition into particles and be removed from the water through sedimentation.

Thus, for organisms that receive their primary exposure through the water, we do not expect exposure to contaminants to increase nor do we expect significant toxicological effects. However, elevated levels of some contaminants will exist over spatial scales that could affect aquatic organisms that spend a significant amount of time very near the diffuser. This issue deserves further consideration, and in fact, the MWRA monitoring plan addresses this issue to some extent (e.g., with caged mussels). The variability of contaminant concentrations in animal tissues will allow for detection of about a two- to three-fold change, under the present monitoring plan. It is unlikely that this level of change will take place very far from the outfall, but this level of detectable change appears to be adequate for comparisons to published biological or human-health effects levels.

For bottom-dwelling organisms (e.g., winter flounder and lobster), we are less certain about their potential body burdens because we do not have reliable estimates of the exposure concentrations in the sediment nor do we have an adequate understanding of the migration and feeding behavior of these animals, particularly after the outfall begins discharging. Further, it is quite possible that the increased organic carbon loading to the sediment will have a mitigating effect on the bioavailability of the sediment-bound contaminants. Despite these uncertainties, we believe it is unlikely that a significant population of bottom-dwelling fish or shellfish in Massachusetts Bay will inhabit the small zone surrounding the outfall that will experience increased contaminant loads. Thus, the risk of any broad scale ecological effects or of any human-health effects resulting from consumption of contaminated seafood does not appear to be any greater than present risks.

The fact that most of our projections indicate that very little change or impact is likely to take place in Massachusetts Bay should not lull us into complacency about potential ecological effects. Our future efforts should be directed toward issues that we know the least about and have the greatest potential for broader scale impact. We believe that more attention should be placed on identifying other possible toxicants in the MWRA effluent and on other possible biological effects (e.g., reproductive and developmental effects), rather than focusing only on refining conventional models and reducing uncertainty around our present predictions of impact that are generally considered acceptable.

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1.0 INTRODUCTION

The Massachusetts Water Resources Authority (MWRA) is implementing a pollution abatement program in Boston Harbor that has already resulted in a dramatically lower loading of bacteria, toxic contaminants, and sewage particles to Boston Harbor (Alber *et al.*, 1993). As a result, water quality has significantly improved throughout much of Boston Harbor and there is evidence that the health of fish, shellfish, and other benthic organisms is beginning to improve as well (Alber *et al.*, 1993; Aubrey and Connor, 1993; Blake *et al.*, 1993a; Moore and Stegeman, 1993). However, the MWRA pollution abatement program includes the diversion of the MWRA effluent offshore beginning in 1995, bringing nutrients, toxic contaminants, and organic matter directly into Massachusetts Bay and raising concerns that this effluent diversion will adversely impact the Bay. This issue was first addressed in detail during the Secondary Treatment Facilities Plan (STFP) studies (MWRA, 1988) and during the development of the Supplemental Environmental Impact Statement (SEIS; EPA, 1988). These assessments suggested that discharge through the new outfall would cause little harm to the environment but that, very near the outfall diffuser, there could be violations of human-health-based water quality criteria and that the benthic environment would likely change. A review of these assessments and projections related to toxic contaminants (Shea and Kelly, 1992) has shown that the projections were very conservative because

- The concentrations of many toxic contaminants used in previous models were too high; recent analyses of toxic contaminants in the MWRA effluent have found much lower concentrations.
- Assumptions made in the previous models were too conservative; initial dilution was underestimated, and contaminant transport and removal mechanisms were not considered.

Despite being very conservative, these projections probably provided adequate information to assess the relative merits of specific locations for the new offshore outfall; which was the primary purpose of the STFP and SEIS assessments. In addition, conservative projections offer the advantage of a higher margin of safety in the protection of the Massachusetts Bay.

The EPA recently agreed with the conclusions of Shea and Kelly (1992) in their assessment of potential impact of the MWRA outfall on endangered species. The EPA found that the new outfall is unlikely to adversely impact endangered species (EPA, 1993). Thus, available evidence strongly indicates that discharge of toxic contaminants in the MWRA effluent through the new outfall will have minimal and possibly unmeasurable impact on the environment of Massachusetts Bay. To verify these predictions, the MWRA has begun implementing one of the most comprehensive discharge monitoring projects in the country (MWRA, 1991). An important component of the MWRA monitoring plan is the periodic review and synthesis of monitoring data and information from other complementary studies to evaluate projections of impact and the adequacy of the monitoring design to detect meaningful change.

This report represents the first annual review of toxic contaminant monitoring data to serve this purpose. The format of this report is based on that of the MWRA monitoring plan (MWRA, 1991) where baseline studies related to toxic contaminants were divided into three primary areas (1) Effluent Monitoring, (2) Benthic Monitoring, and (3) Fish and Shellfish Monitoring. We consider each of these areas separately (in Sections 2-4), using data that became available in 1993 and early 1994. We summarize our review (in Section 5) by revisiting the questions that were asked in the MWRA monitoring plan to assess whether the monitoring plan is producing adequate information to answer these questions. This format follows the process recommended by the National Research Council (NRC, 1990) and adopted by the MWRA monitoring plan (MWRA, 1991). Finally, we would like to caution the reader that this report was written under the assumption that you are familiar with the MWRA effluent outfall monitoring program and general concepts regarding the fate and effects of toxic contaminants in marine ecosystems.

2.0 EFFLUENT MONITORING

Accurate and precise estimates of concentrations and loadings from the MWRA effluent are the foundation for assessing potential impact resulting from the effluent discharge. Contaminant concentrations in the MWRA effluent are routinely measured under three separate monitoring programs: (1) NPDES permit-compliance monitoring, (2) local limits monitoring, and (3) Deer and Nut Island treatment plant monitoring (Bigornia-Vitale and Sullivan, 1993). However, we have previously shown that these measurements do not provide reliable effluent concentrations or loading estimates for many organic contaminants because their concentrations in the effluent are near or below the reporting limit or method detection limit (MDL) of the standard EPA analytical methods used for these monitoring programs (Shea and Kelly, 1992). In addition, there are many contaminants that are useful for assessing impact or risk that are not measured by these standard EPA methods (e.g., individual PCB congeners, additional aromatic and heterocyclic hydrocarbons, linear alkyl benzenes). For these reasons, the MWRA has augmented the NPDES (and local limits) analyses with a Detailed Effluent Characterization Study (DECS) (Shea, 1993). In this section of the report we summarize the most recent data on toxic contaminants in the MWRA effluent and discuss the implications to contaminant exposure and monitoring.

2.1 Contaminant Concentrations

Data from the 1993 DECS analyses are summarized in Table 2-1 along with Fiscal Year (FY) 1992 NPDES monitoring data (Bigornia-Vitale and Sullivan, 1993) and the SEIS primary treatment projections (EPA, 1988). A complete description and listing of the DECS results will be reported elsewhere (Uhler and West, 1994). A list of abbreviations used in this report for organic contaminants is given in Appendix A.

The new DECS trace metal data are in good agreement with the FY 1992 NPDES metals data; the DECS data are slightly lower in Cd, Cr, Hg, and Ni; and slightly higher in Cu, Ag, and Zn. The overall agreement indicates that recent NPDES trace metal data are

Table 2-1. Summary of Contaminant Concentrations in MWRA Effluent

Contaminant	Detailed Effluent Characterization (ng/L)[1]	NPDES (Fiscal Year 1992) (ng/L)[2]	SEIS Projections Primary Treatment (ng/L)[3]
Arsenic	NA[4]	1590	1810
Cadmium	550	700	2270
Chromium	3400	4300	16890
Copper	72000	58000	82650
Lead	12000	10130	11940
Mercury	160	230	1240
Nickel	6000	8870	21380
Selenium	NA	1120	15260
Silver	4200	3200	4010
Zinc	87000	70760	165300
Acenaphthene	162	1000	NA
Benzo(a)pyrene	29	1000	NA
Fluoranthene	180	1000	NA
Fluorene	190	1000	5250
Naphthalene	1380	1000	NA
Butylbenzyl phthalate	NA	2070	20260
Bis(2-ethylhexyl) phthalate	NA	9640	24900
Di-n-octyl phthalate	NA	1130	20930
Aldrin	<0.5	20	220
Chlordane	4.0	130	NA
Dieldrin	52	20	30
4,4'DDT	6.3	150	60
Endrin	4.0	10	NA
Heptachlor	<0.5	10	230
Lindane	15	20	NA
PCB	50	100	1020

[1] Data from 1993 Detailed Effluent Characterization Study (Uhler and West, 1994).
[2] Data from Fiscal Year 1992 NPDES monitoring (Bigornia-Vitale and Sullivan, 1993).
[3] Data from Table 6.4.4a in the SEIS (EPA, 1988).
[4] NA: Data not available.

reliable. Note that our previous speculation that NPDES Hg data could be too high (Shea and Kelly, 1992) is not supported by the new DECS data. However, the concentrations of Cd, Cr, Hg, Ni, and Zn in the DECS and NPDES data sets are less than half the values used in the SEIS, with Hg about 1/6 the SEIS value.

In contrast to the trace metals data, most of the new DECS organic contaminant measurements are substantially lower than the NPDES and SEIS values. Except for naphthalenes, the polycyclic aromatic hydrocarbons (PAH) are 10 to 100 times lower than both the NPDES and SEIS values. Phthalates were not measured in the DECS, but the latest NPDES values are up to 20 times lower than the SEIS values. The NPDES phthalate values are near the MDL (1000 ng/L), so actual concentrations could be even lower. The 1991 local limits monitoring data indicate that most of the phthalates are typically below 200 ng/L, which is the reporting limit for that program. The new DECS pesticide concentrations are up to 100 times lower than the FY 1992 NPDES values, and up to 1000 times lower than those in the SEIS. The DECS total PCB concentrations are similar to the FY 1992 NPDES values (although these are simply MDLs) and are about 20 times lower than the SEIS values.

The new DECS organic contaminant data (Table 2-1) are in good agreement with those reported by Shea (1992a,b) using the same analytical methods. This is important because recent estimates of water quality criteria exceedances (Shea and Kelly, 1992) and potential impact to endangered species (EPA, 1993) were based on the data of Shea (1992a,b). The only notable differences were higher concentrations of hexachlorobenzene (HCB) and total PCBs in the DECS data and lower concentrations of dieldrin and lindane in the DECS data. The higher concentrations of total PCBs were due almost entirely to unusually high concentrations of PCB congener 180 and this is probably a phthalate interference.

2.2 Contaminant Loads: Past, Present, and Future

Data from past NPDES monitoring have been used to suggest that, during the 1980s, there was about a 75% decrease in total metal loadings from MWRA effluents, exclusive of sludge

(MWRA, 1992; Shea and Kelly, 1992). The decreases reported for individual metal loads from 1980 to 1990 ranged from a few percent to nearly 90%. Although there is some uncertainty over the quality of earlier metals data (see Shea and Kelly, 1992), these dramatic reductions in loads are supported by more recent, good-quality data for copper, lead, and zinc (Figure 2-1). Significant decreases in all three metals were observed over the period 1989 through 1991, with an indication that a plateau was reached in 1991. Effluent data from 1993 (Uhler and West, 1994) indicate that this plateau has continued through the end of 1993. Note that 10 years ago, the loadings of copper, lead, and zinc were reported to be about 10 times higher than the loads in 1991 or 1992 (MWRA, 1992; Shea and Kelly, 1992).

Any significant reduction in metal concentrations in the effluent should be reflected in lower observed concentrations in the water column and aquatic organisms in Boston Harbor, assuming the MWRA effluent is the dominant source of metals to Boston Harbor. In fact, metal concentrations (copper, cadmium, iron, nickel, lead, and zinc) in Boston Harbor water have decreased about 60 to 80% since 1983 (Wallace *et al.*, 1993), corresponding extremely well with the approximately 75% reduction in most metals in the MWRA effluent. The limited data that are available on bioaccumulation of metals in caged mussels at Deer Island are inconclusive (Bigornia-Vitale and Sullivan, 1993). There was no difference between the 1987- and 1991-concentrations of copper, lead, and zinc in caged mussels deployed at Deer Island, but in 1991 there also was no statistically significant accumulation of metals at Deer Island compared to pre-deployment mussels and compared to a "clean" control location (Gloucester, MA). There are insufficient data on organic contaminants in the MWRA effluent to assess historical trends, but caged mussel studies have shown (Table 2-2) that body burdens of lower-molecular-weight (LMW) PAHs, total PCBs, and several chlorinated pesticides have decreased significantly since 1987 (Downey *et al.*, 1993). Curiously, body burdens of high-molecular-weight (HMW) PAHs increased from 1991 to 1992, but the source of this increase is probably not the MWRA effluent because the effluent is depleted in the HMW PAH (see below). Seven years of NOAA Mussel Watch data on contaminants in mussels near Deer Island (Battelle, 1992) indicate that there is a decreasing trend in copper (drops by 7% per year) and PAHs (drops by 17% per year for total PAHs).

Table 2-2. Comparison of mussel bioaccumulation data (ng/g) at Deer Island from 1987, 1991, and 1992 (from Downey *et al.*, 1993).

Contaminant	1987 Mean±SD	1991 Mean±SD	1992 Mean±SD
Total PAH	2343±251	1207±439	1934±480
LMW PAH	1221±184	516±243	427±136
HMW PAH	1123±165	691±206	1507±366
Total PCB	630±264	199±33	133±22
Total DDT	63±34	48±14	25±6
Dieldrin	11±4	3±1	3±1
Alpha-chlordane	22±6	10±4	7±1
Trans-nonachlor	18±4	9±3	8±2

Total PAH and total PCB include fewer analytes than those listed in Appendix A (see Downey *et al.*, 1993).

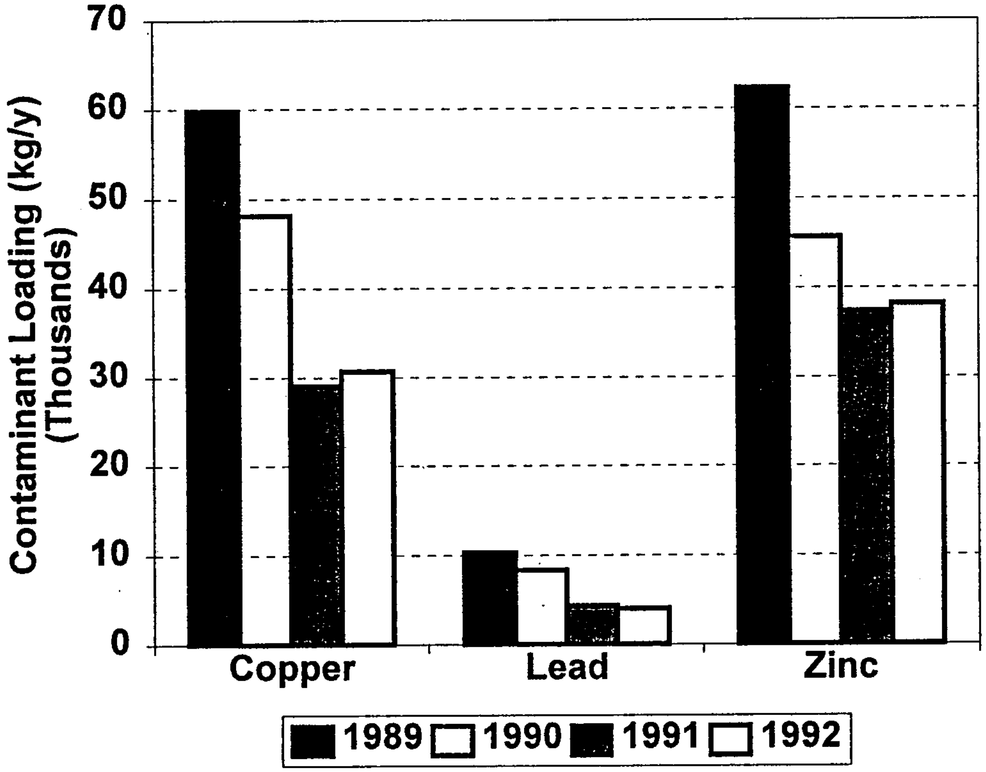


Figure 2-1 **Annual geometric mean loading of Cu, Pb, and Zn from the MWRA effluent over the last four fiscal years. Data are flow-weighted combined loads from the Deer and Nut Island treatment plants using NPDES data.**

Using the 1993 effluent contaminant concentrations (Table 2-1) and contaminant removal efficiencies from the literature we summarize estimates for past, present, and future MWRA contaminant loads in Table 2-3 (a full list will be provided in Uhler and West (1994). The decrease in trace metals concentrations from pre-1991 to 1993 is mostly a result of the MWRA source reduction program and the cessation of sludge discharge in late 1991. The removal of Cu piping in the effluent sampling stream also contributed to the observed decrease in Cu. The apparent decrease in PAH, PCB, and pesticides is due mostly to better analytical methods (Shea and Kelly, 1992). Nearly all of the future loading projections in Table 2-3 are lower than those in the SEIS (EPA, 1988). The largest differences are found for organic contaminants; smaller differences were found for cadmium, mercury, chromium, copper, nickel, and zinc. Very little difference is seen for arsenic, silver, and lead.

Table 2-3. Estimates of MWRA Toxic Contaminant Loads (kg/y)

	Pre-1991	1993	1996	1998
Arsenic	NA	881	734	587
Cadmium	2070	433	344	255
Chromium	15700	2445	1712	978
Copper	66000	30776	19649	8523
Lead	18000	5667	5090	4513
Mercury	220	138	91	44
Nickel	9500	4800	4320	3840
Silver	3780	2028	1159	290
Zinc	120000	37626	26338	15050
Total PCB	400	36	20	5
Total PAH	22200	4700	2663	625
Total phthalates	8800	9689	5489	1289
Benzene	2700	709	402	94
Flow (MGD)	390	390	395	400

Data for pre-1991 are from Menzie-Cura (1991). Total PCB and total PAH values for 1993 and beyond include those analytes listed in Appendix A. In 1996, the new primary treatment plant will be completed along with one battery of secondary treatment. In 1998, two batteries of secondary treatment will be completed. We do not expect further reductions in mean annual loads beyond 1998 because the third and fourth batteries of secondary treatment will treat only a small fraction of the total annual flow.

2.3 Variability of Contaminant Concentrations and Loads

Annual mean contaminant concentrations and loads are useful for modelling long term fate and effects, but for shorter term assessments we need to consider temporal variability and patterns in effluent quality. We have reviewed most of the MWRA effluent data over the last 10 years and several general patterns emerged. Trace metal concentrations and loads estimated from NPDES data are quite stable from month to month with relative standard deviations (RSD) about the (arithmetic) annual mean concentration about 8% for copper, 35% for lead, and 12% for zinc. The more recent DECS trace metals data (Figure 2-2) support these estimates of variability, with RSDs of 11-19% for Cu, Ni and Zn; and 37-43% for Ag, Cd, Cr, Hg, and Pb (Uhler and West, 1994). There is no seasonal pattern in any of the trace metal data sets. Variability on even shorter time scales can be higher, 2- to 3-fold changes in Zn loads were observed over just a few days (MWRA, 1988). However, weekly data on Cu and Pb in the Deer Island effluent indicate that changes of this magnitude are probably of low frequency and short duration (MWRA, unpublished data).

The data from Shea (1992a,b) and the 1993 DECS data indicate that the variability of organic contaminant concentrations and loads is highly dependent on the contaminant and the sampling period. For example, three effluent samples collected over a one-week period (Shea, 1992a,b) exhibited concentration ranges of about a factor of two (for most contaminants, including DDT) during November 1991 and up to factor of 35 (for DDT) during June 1992. Concentration ranges were greater for influent than effluent and were also greater for Nut Island versus Deer Island. The more recent DECS data indicate relatively low variability (20 - 30% RSD) and no seasonal pattern (from late spring to late fall) for PAH, LAB, PCB, and many pesticides. However, hexachlorobenzene (HCB), dieldrin, and DDT metabolite (DDD and DDE) concentrations were highly variable and were much higher in early summer than during late summer and fall (Figure 2-2). The annual mean concentration for these pesticides is not representative of actual concentrations in most months. Note that the highest concentrations of these contaminants appear during the early summer, when water-column stratification could restrict effluent dilution to bottom waters at

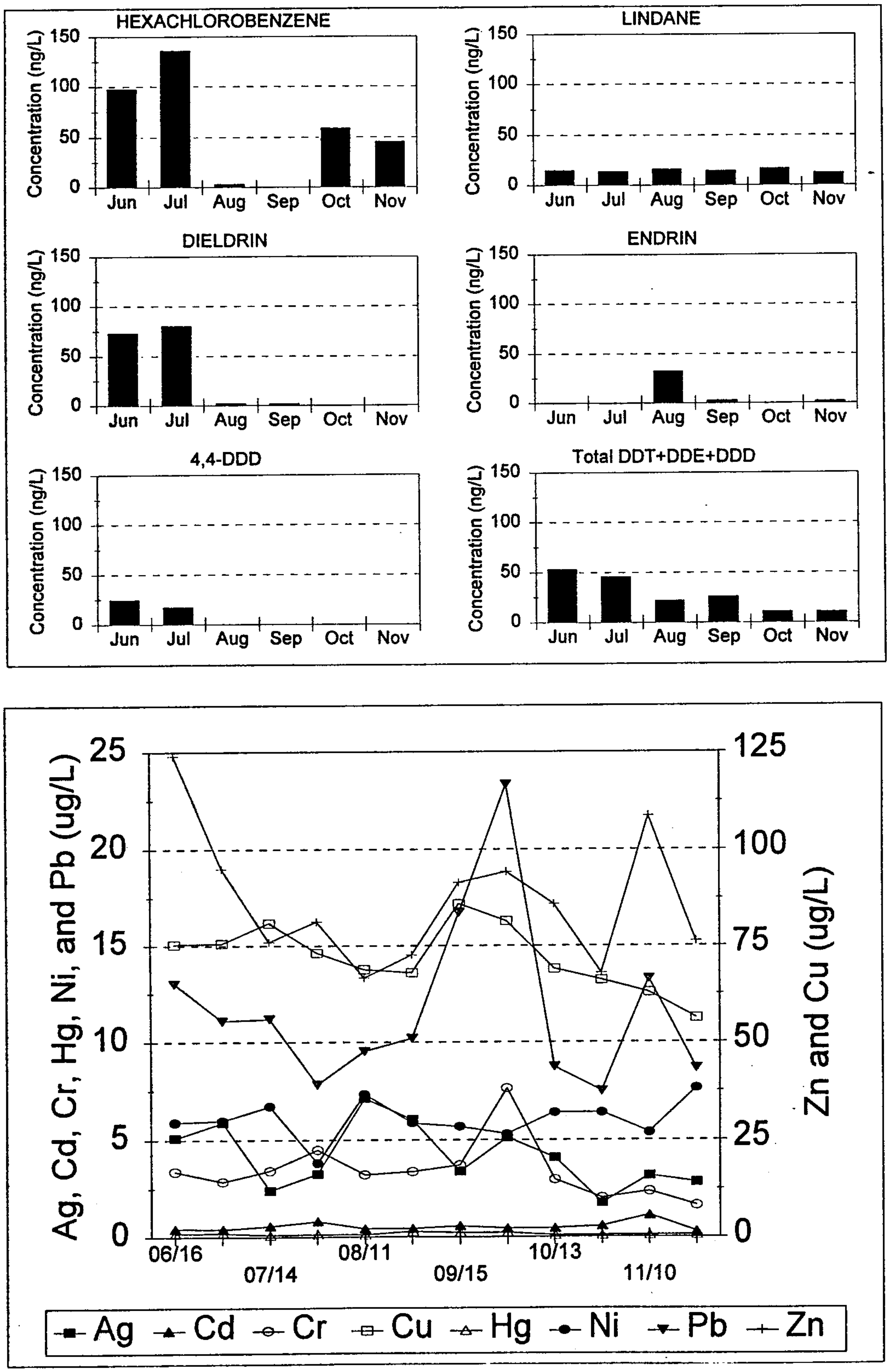


Figure 2-2 Concentrations of trace metals (bottom) and some pesticides (top) from June 1993 through November 1993 (from Uhler and West, 1994).

the future outfall. Monitoring of contaminant exposure (caged mussels) could also be affected by these fluctuations depending on the time of sample collection.

The variability in monthly contaminant loads is driven primarily by the contaminant concentrations rather than the effluent flow (Uhler and West, 1994). Day-to-day variability in flows can be much greater than with monthly flows, but we do not have daily contaminant concentrations to estimate the corresponding fluctuations in loads.

2.4 Comparison of the MWRA Contaminant Load to Other Sources

Estimating the relative contribution of the MWRA load to the total bay-wide toxic contaminant load is important when assessing potential regional impact and ability to detect change in the farfield region. Unfortunately, estimates of loads from other contaminant sources (Menzie-Cura, 1991) have much greater uncertainty than those of the MWRA loads and some important sources (e.g., oil spillage) were not included. New estimates for the Merrimack, Charles, Mystic, and Danvers Rivers, atmospheric deposition, the Lynn sewage effluent, and storm sewers/urban runoff will be available later in 1994 through studies sponsored by the Massachusetts Bays Program (MBP) and revisions are being made to some of the other loading estimates reported by Menzie-Cura (M. Alber, pers. comm., 1993). Until these new data are available, we can only estimate preliminary relative contributions. Comparing the revised MWRA loads (Table 2-2) to the bay-wide estimates of Menzie-Cura (1991) indicates that the relative contribution of MWRA to bay-wide loads is most significant for copper (Figure 2-3) and PAHs (not shown). However, the MWRA makes only a small contribution to the total loads of PCB (Figure 2-3), lead and zinc (not shown). The small increase in loads in 1995 is due to the diversion of the MWRA effluent through the new outfall with only primary treatment (100% export versus the present 75% export from Boston Harbor - see Shea and Kelly, 1992). We expect reductions in the MWRA contribution in 1996 when the first battery of secondary treatment is operational and again in 1998 after the second battery is on line. We are not aware of any evidence that the additional third and fourth batteries of secondary treatment will result in further reductions in contaminant loads.

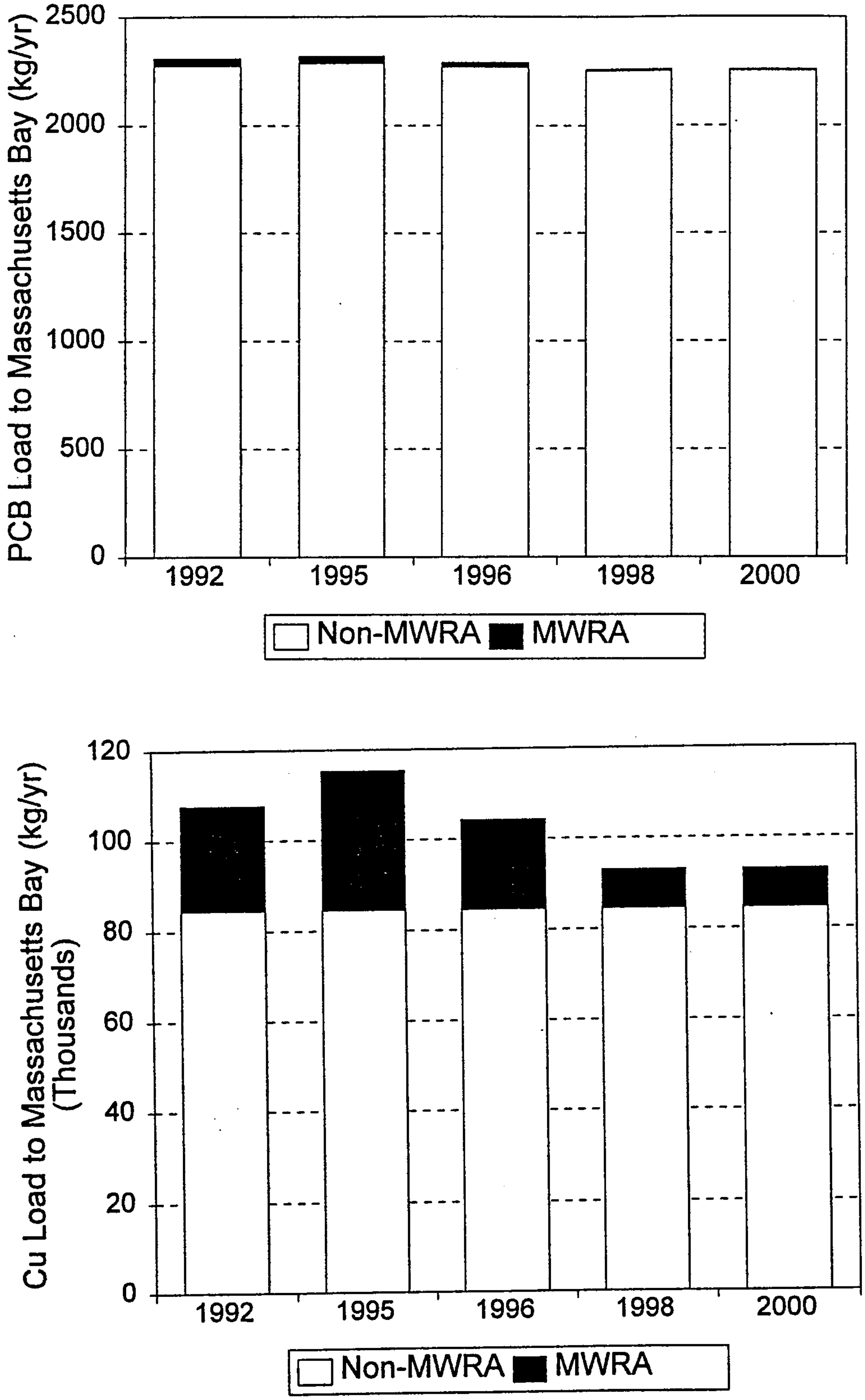


Figure 2-3 Projections of total Cu (bottom) and PCB (top) loads to Massachusetts and Cape Cod Bays and the relative contribution from MWRA and non-MWRA sources.

Note that it now appears likely that the outfall will not be online until 1996, so there will be only a very short period or even no period when primary-treated effluent is discharged and thus we would expect no increase in loading due to the outfall relocation.

2.5 Chemical Signature of the MWRA Effluent

The DECS results provide a reliable and robust data set that can be used to establish the chemical signature or fingerprint of the MWRA effluent to help track the fate of the effluent and to help verify any circumstantial evidence of increased exposure to toxicants or related biological effects. The chemical patterns for several classes of contaminants are shown in Figure 2-4. A complete description of these patterns is given in Uhler and West (1994).

The PAHs are enriched in low molecular weight compounds indicating that the dominant source of PAH to the sewerage system is relatively fresh and light petroleum products, such as lube oil (see principal components analysis in Uhler and West, 1994). The relative abundance patterns are very consistent over the course of the 6 months that were sampled. The PCB congener distributions are also enriched in the lower molecular weight compounds (lower chlorination number), though the pattern is not as consistent with time as the PAHs. The relative abundance of many pesticides changed with time (Figure 2-2). The trace metal signatures reported by Uhler and West (1994) are very similar to those reported in the NPDES data, with stable concentrations throughout the year.

The chemical signatures in Figure 2-4 are potentially a powerful tool that could be used to discriminate among possible sources of contaminants to Massachusetts Bay. Comparable data on the chemical signatures of other sources will be available later this year through studies sponsored by the MBP. The MBP is also funding a study that will use pattern recognition and statistical methods (e.g., principal components analysis) to help assess the relative contribution of various sources to sediment contaminant inventories (Shea, 1994). The chemical signature of the MWRA effluent should continue to be monitored indefinitely, as changes to the treatment are likely to alter the relative abundance of contaminants.

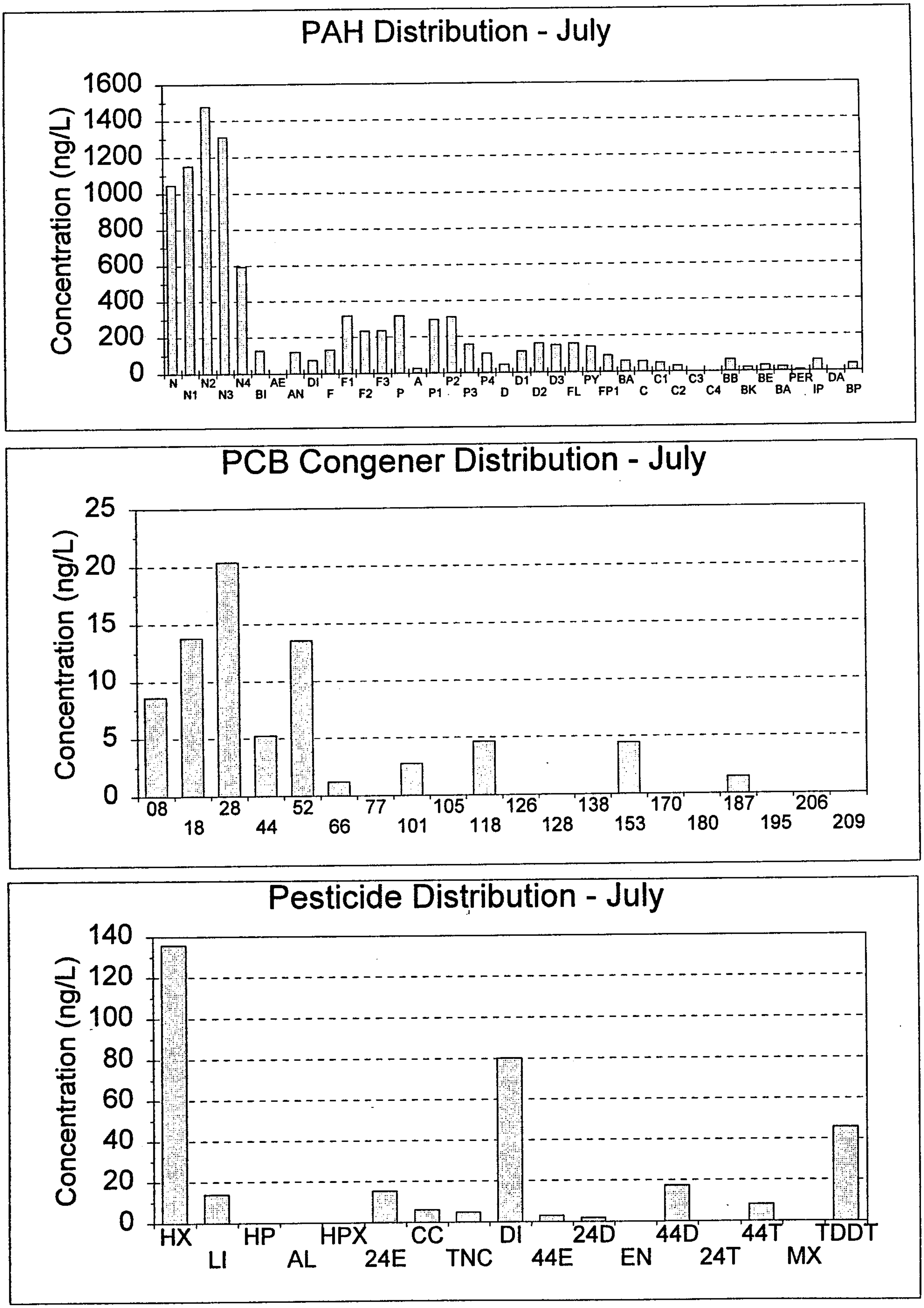


Figure 2-4 Distribution of contaminants in the MWRA Deer Island effluent.

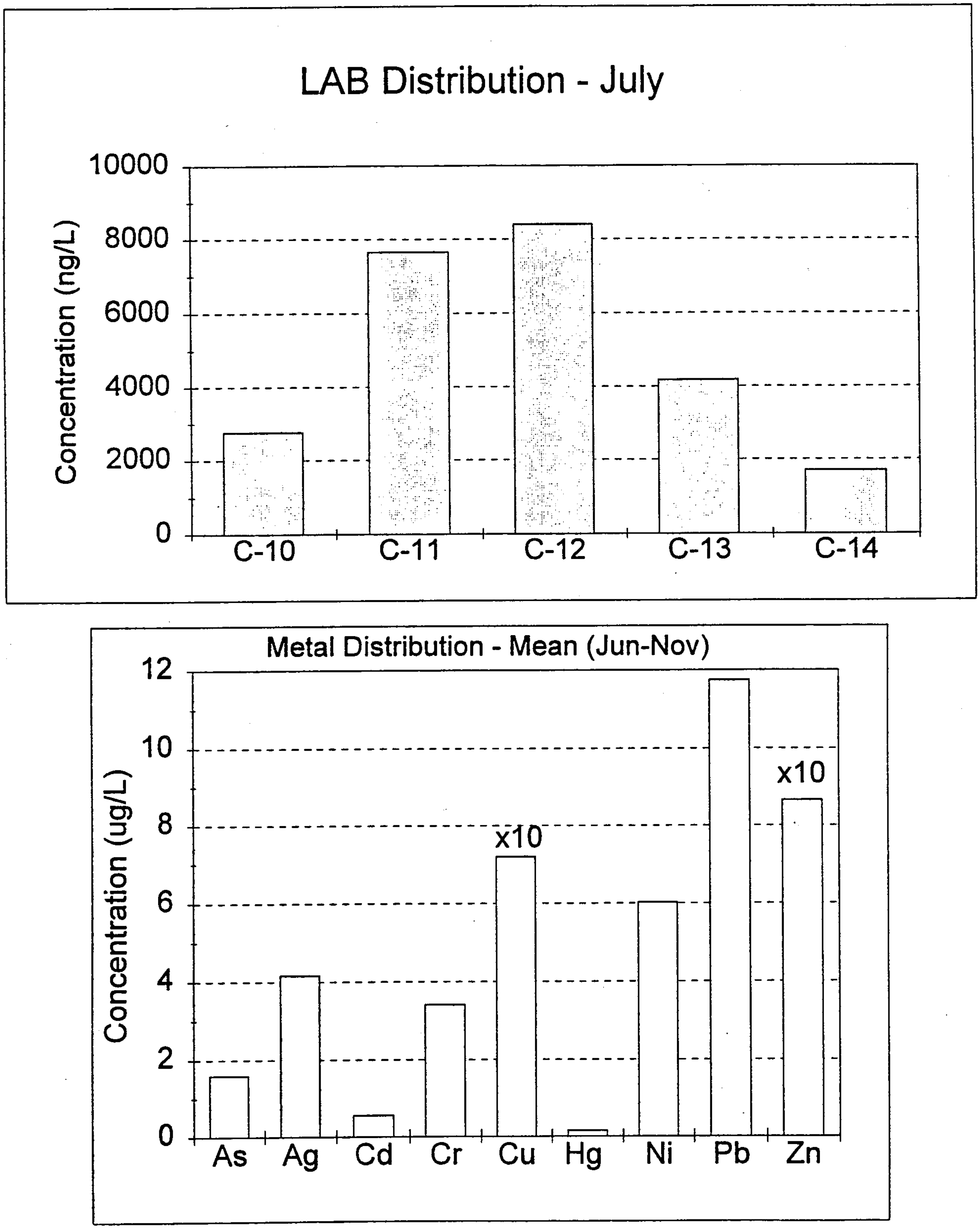


Figure 2-4 Distribution of contaminants in the MWRA Deer Island Effluent. All data are from the 1993 DECS results except As, which is from the FY 1992 NPDES data. Cu and Zn data were divided by 10 to fit on the graph (x10).

2.6 Effluent Toxicity

The MWRA has performed three types of effluent toxicity testing every month as part of the NPDES monitoring program: (1) a 96-h acute toxicity test using one- to five-day-old mysid shrimp *(Mysidopsis bahia)*, (2) a chronic survival and growth test using sheepshead minnow *(Cyprinodon variegatus)*, and (3) a chronic reproduction test using red marine algae *(Champia parvula)*. The results of these toxicity tests over the period July 1991 through June 1992 were consistent with results from previous years. The reported *no observed effects concentrations* (NOEC) are expressed as percent effluent (5% effluent is a 20:1 dilution) and as LC_{50} (the *lethal concentration* required to kill 50% of a population of test organisms). For the mysid, the annual mean NOEC was 12%, which is not in compliance with the NPDES limit of 20%. The mean LC_{50} value was 29% effluent; there is no NPDES limit for LC_{50}. For the sheepshead minnow, the mean annual NOEC for survival was 37% and for growth was 28%, both well above (in compliance with) the NPDES permit limit of 10%. For red algae, the mean annual NOEC was 1.7%, well below (not in compliance with) the NPDES permit limit of 10%. Considering data from individual months, the mysid test was in compliance with the NPDES permit three to four months out of the year, the sheepshead minnow test was always in compliance, and the red algae test was always non-compliant.

Considering that water-quality criteria (WQC) do not account for cumulative or synergistic effects and they may not protect against toxicants that have no WQC, effluent toxicity testing is a reasonable approach to protecting against toxicity. However, the choice of indicator organisms and toxicity endpoints is critical to making these data useful indicators of potential effluent toxicity and is being given further consideration by EPA. The red algae test *(Champia)* is being withdrawn from the NPDES testing, but no replacement has been specified.

It is interesting to note that a toxicity identification evaluation performed by MWRA and EPA on the Deer Island effluent indicated that most of the effluent toxicity was due to surfactants, with some toxicity also due to residual chlorine. None of the contaminants of concern identified by EPA in the SEIS and none of the priority pollutant contaminants measured in the NPDES monitoring program contributed to the toxicity. Residual chlorine should not pose a problem at the new treatment plant because it will include dechlorination prior to discharge. Surfactants are likely to be much less of a problem because removal efficiencies during activated-sludge secondary treatment can exceed 99%. However, there is recent evidence that metabolites of common non-ionic surfactants (ethoxylates) exhibit significant toxicity (Dorn *et al.,* 1993) following secondary treatment and that alkylphenols, in particular, exhibit significant estrogenic activity and have been implicated in reproductive problems in fish (Colburn *et al.,* 1993). These types of sublethal effects (i.e. developmental and reproductive) are not presently considered in effluent toxicity testing or in the derivation of WQC and thus Massachusetts Bay may not be adequately protected from these sublethal effects even if conventional toxicity testing indicates that we should expect no problems. The potential risk associated with these sublethal effects are unknown and have not been studied in the MWRA effluent previously because they have only recently been identified as a potential problem (e.g., Colburn *et al.,* 1993).

2.7 Implications to Contaminant Exposure

To help assess the environmental and monitoring implications of the revised effluent concentrations (Table 2-1) we estimated the concentrations of contaminants in the water of Massachusetts Bay that would result from the new discharge by using simple, conservative models of effluent dilution (EPA, 1993). Note that these models will lead to overpredictions of ambient concentrations (Shea and Kelly, 1992). We then compared these expected concentrations to present ambient concentrations and to marine WQC. Two examples are illustrated for Cu and dieldrin in Figure 2-5. The expected total Cu concentration (increment due to MWRA plus ambient) is plotted as a function of water surface area for both primary and secondary treatment. The primary treatment data are from FY92 NPDES results; the secondary treatment data are predictions after two batteries are on line (i.e, 1998). For primary treated effluent, 100:1 dilution (which corresponds to the hypothetical edge of the

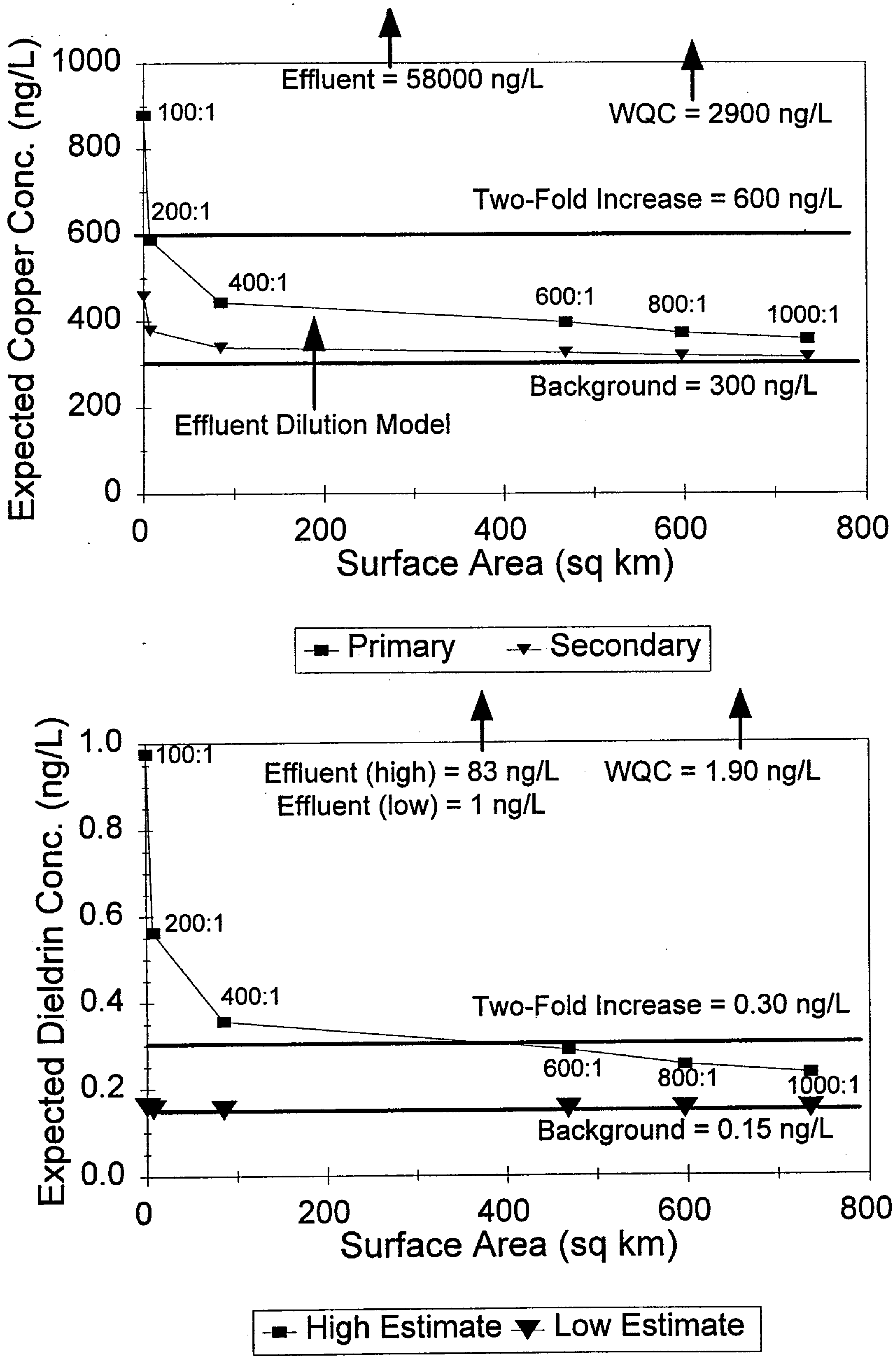


Figure 2-5 <u>Top</u>: Expected total copper concentrations as a function of surface area in Massachusetts Bay for primary and secondary treatment.
<u>Bottom</u>: Expected dieldrin concentrations as a function of surface area in Massachusetts Bay for the highest and lowest effluent concentrations.

zone of initial dilution and is about 60 m from the diffuser) would result in total Cu concentrations approximately three times lower than the WQC and three times higher than present ambient concentrations. A 200:1 dilution would nearly double the Cu concentration within the 7 km^2 of this dilution zone. A 400:1 dilution would increase Cu about 50% over 85 km^2. For reference, the nearfield monitoring box is approximately 120 km^2. Dilutions of 600:1, 800:1, and 1000:1 correspond to areas of 468 km^2, 597 km^2, and 735 km^2, respectively. For secondary treated effluent the increases would be much smaller, only a 50% increase at 100:1 dilution.

The range in contaminant concentrations in the effluent can have a large effect on our estimates of water column exposure as illustrated in Figure 2-5 for dieldrin. The high estimate (measured in July 1993) indicates that dieldrin concentrations will be about one-half the WQC value after the 100:1 initial dilution. This will be about six times above ambient concentrations, and a two-fold increase in dieldrin will take place over nearly 400 km^2. However, using the low estimate (October and November 1993) indicates that dieldrin concentrations will remain essentially unchanged even within the zone of initial dilution.

Projections for water-column exposure for other contaminants are summarized in Table 2-4. Most of the effluent:ambient ratios are below 100, so a doubling of these contaminant concentrations is not expected beyond about a few ten's of meters away from the diffuser. The following contaminants, in order of increasing effluent:ambient ratios, would double in concentration beyond the 60-m zone of initial dilution: Pb, Cu, lindane, acenaphthene, fluorene, Ag, dieldrin, naphthalene, total PAH and total LAB. The LAB clearly offer great potential to track the effluent discharge and verify dilution models given the very high effluent:ambient ratio, the low variability in effluent concentrations (Uhler and West, 1994), and their moderate resistance to degradation. The total PAH and naphthalene might appear to offer only slightly less potential as a tracer, but naphthalene is relatively volatile and easily degraded and most of the total PAH is composed of naphthalenes.

Table 2-4. Summary of Contaminant Exposure Projections

Contaminant	Effluent Conc. (ng/L)[1]	Ambient Conc. (ng/L)[2]	Effluent Ambient (ratio)	WQC (ng/L)[3]	Effluent WQC (ratio)
Arsenic	1590	500	3	36000	0.044
Cadmium	700	27	26	9300	0.075
Chromium	4300	250	17	50000	0.086
Copper	58000	300	193	2900	**20**
Lead	10130	100	101	8500	**1.19**
Mercury	230	4	58	25	**9.2**
Nickel	8870	500	18	8300	**1.07**
Silver	3200	7	457	2300	**1.39**
Zinc	70760	750	94	86000	0.82
Acenaphthene	162	0.5	324	500000	0.0003
Benzo(a)pyrene	29	0.5	58	300000	0.0001
Fluoranthene	180	5	36	16000	0.0113
Fluorene	190	0.5	380	300000	0.0006
Naphthalene	1380	0.4	3450	2350000	0.0006
Total PAH	19840	12	1653	NA	NA
Butylbenzyl phthalate	2070	NA	NA	3400	0.61
Bis(2-ethylhexyl) phthalate	9640	NA	NA	3400	**2.8**
Di-n-octyl phthalate	1130	NA	NA	3400	0.33
Aldrin	0.5	0.03	17	1300	0.0004
Chlordane	4.0	0.1	40	4	**1**
Dieldrin	52	0.09	578	1.9	**27**
4,4'DDT	6.3	0.2	29	1	**6.3**
Endrin	4.0	0.05	80	2.3	**1.7**
Heptachlor	0.5	0.4	1	3.6	0.14
Lindane	15	0.05	300	340	0.044
Total PCB	50	7	7	30	**1.7**
Total LAB	29874	1	29874	NA	NA

[1] Metals and phthalate data are from Fiscal Year 1992 NPDES monitoring (Bigornia-Vitale and Sullivan, 1993); all other data are from 1993 Detailed Effluent Characterization Study (Uhler and West, 1994).

[2] Ambient data are a composite from several sources (see Shea and Kelly, 1992).

[3] WQC are the lower of the acute and chronic marine WQC.

None of the effluent:WQC ratios in Table 2-4 exceed 100, indicating that any WQC violations will be confined to within 60 m of the diffuser. The highest value (27) is for dieldrin and as we have seen in Figure 2-5, the higher concentrations in early summer might require a 40-fold dilution to meet WQC, while throughout much of the year no dilution will be required. Based on this analysis, WQC would be exceeded only within a very small area, less than 0.26 km^2. This has significant implications for potential biological effects and for design of the monitoring plan. We are not aware of any study that predicts meaningful changes would take place in Massachusetts Bay following a doubling of contaminant concentrations over an area of 0.26 km^2, or even a four-fold increase in concentrations as might occur for dieldrin in July.

One concern related to monitoring the MWRA effluent discharge is to maximize the chances of statistically detecting meaningful change with a reasonable monitoring approach, such as the present monitoring plan. The coefficient of variation (cv) in measured concentrations of trace metals and organic contaminants in Massachusetts Bay water is on the order of a factor of two (Hunt *et al.*, 1988). The cv is similar for caged mussels (MWRA, 1988; Downey *et al.*, 1993). Therefore, to detect changes in water column contamination (either directly or via accumulation in mussels) with a reasonable monitoring approach (i.e. not requiring an unreasonably large number of samples) will require that measurements be taken within about 60 m (100:1 dilution) of the outfall. We illustrate this point in Figure 2-6 where the range of observed trace metal concentrations is plotted (the vertical line) along with the expected concentration increment after 100:1 dilution (the tick mark). Only Cu and Ag are expected to exceed their highest observed concentrations after 100:1 dilution.

Thus, unless there is a substantial increase in the number of measurements taken, there is very little point to taking measurements of contamination in the water column beyond the initial mixing zone. Also note that because the mixing zone is a highly variable environment (in both space and time), measurements taken in this area should be integrative over time and space. The use of multiple caged mussel deployments or semi-permeable membrane devices (Huckins *et al.*, 1993) should be considered.

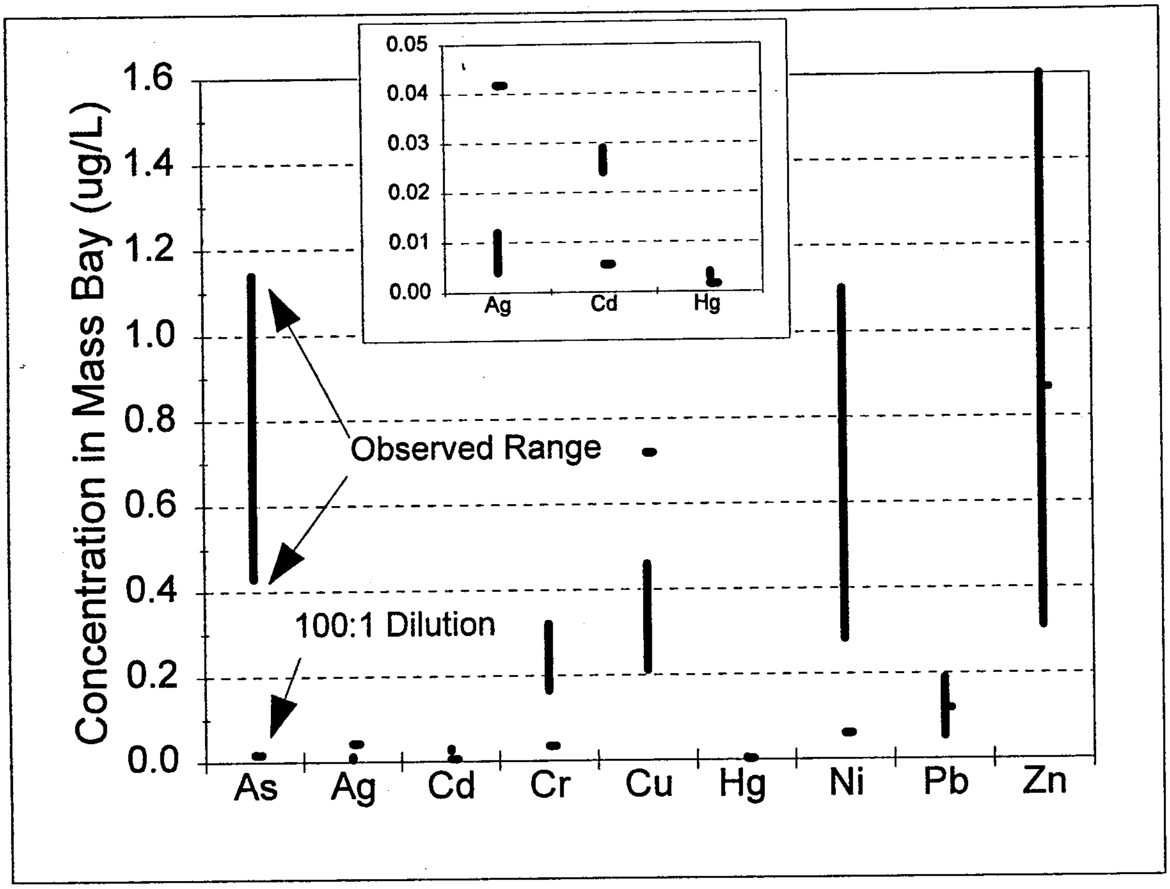


Figure 2-6 The range of observed metal concentrations in water near the future outfall site (vertical line) is compared to the expected increase in concentration (tick mark) after 100:1 dilution is attained.

3.0 BENTHIC MONITORING

The EPA predicted that the benthic environment near the future outfall would change as a result of enrichment of organic carbon and possibly toxic contaminants in the sediment (EPA, 1988). To help determine whether any adverse effects take place, the MWRA is establishing a baseline of physical, chemical, and biological characteristics of sediment throughout Massachusetts Bay. In this section, we summarize the 1992 data on sediment chemistry. The 1992 soft-bottom biology results are discussed by Blake *et al.* (1993b).

3.1 Spatial Distribution of Contaminant Concentrations

Contaminant concentrations were measured in surface sediments at 20 stations within the nearfield (NF) monitoring box (Figure 3-1), at 12 stations throughout the farfield (FF) region of Massachusetts Bays (Figure 3-1), and at 8 additional stations sampled by the USGS (see map in Appendix A). The chemistry data are compiled in Appendix A. In general, the concentrations and relative contaminant distributions (Appendix A and Figures 3-2 and 3-3) are similar to those measured previously (see Shea *et al.*, 1991). Concentrations are highest near Boston Harbor and the Massachusetts Bay Disposal Site (MBDS) and there is a general increase in concentrations with the amount of fine-grained sediment and total organic carbon.

Stations NF2, NF8, NF16, NF20, FF12 and FF13 have the greatest enrichment of metals relative to Al (Figure 3-2). Silver, which is enriched in the MWRA effluent, has the highest sediment enrichment at FF12 and FF13. These two sites also have the highest counts of *Clostridium perfringens* (Figure 3-4). Silver is also enriched in Cape Cod Bay (FF6 and FF7), but not *C. perfringens*. Sites NF2, NF8, and NF16 are highly enriched in both Ag and *C. perfringens*. The organic contaminants exhibit a somewhat different spatial enrichment pattern with PAH, chlordanes, and DDT enriched at stations NF5, NF6, NF7, NF12, NF20, FF1, FF10, and FF12 (Figure 3-2). Most of the other organic contaminants follow different and less consistent enrichment patterns probably because they have lower concentrations and greater contributions from other sources.

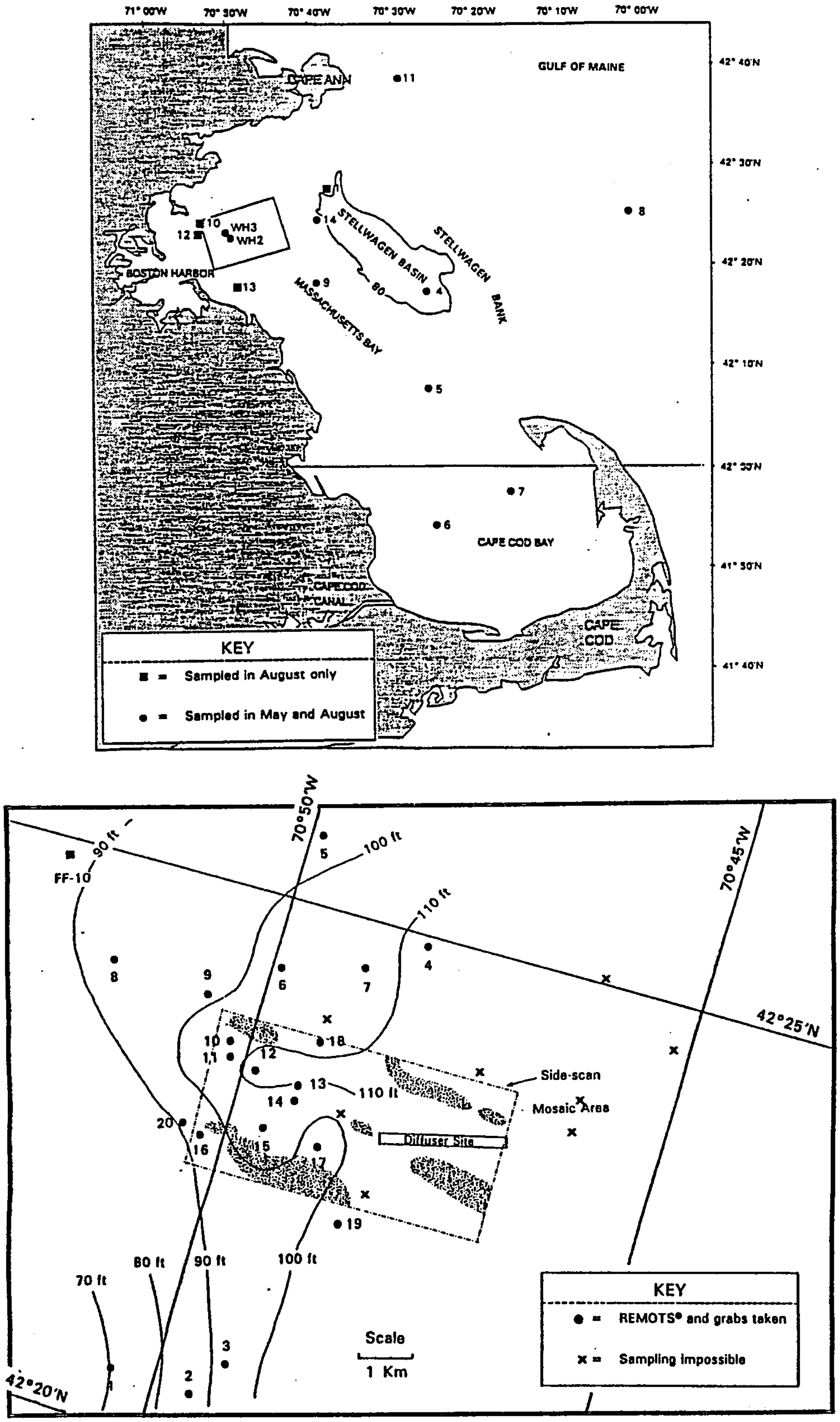


Figure 3-1. Location of sediment stations. The top map shows the farfield stations; the rectangular box represents the nearfield monitoring area shown in detail in the map at the bottom. See Blake *et al.* (1993b) for details on the locations.

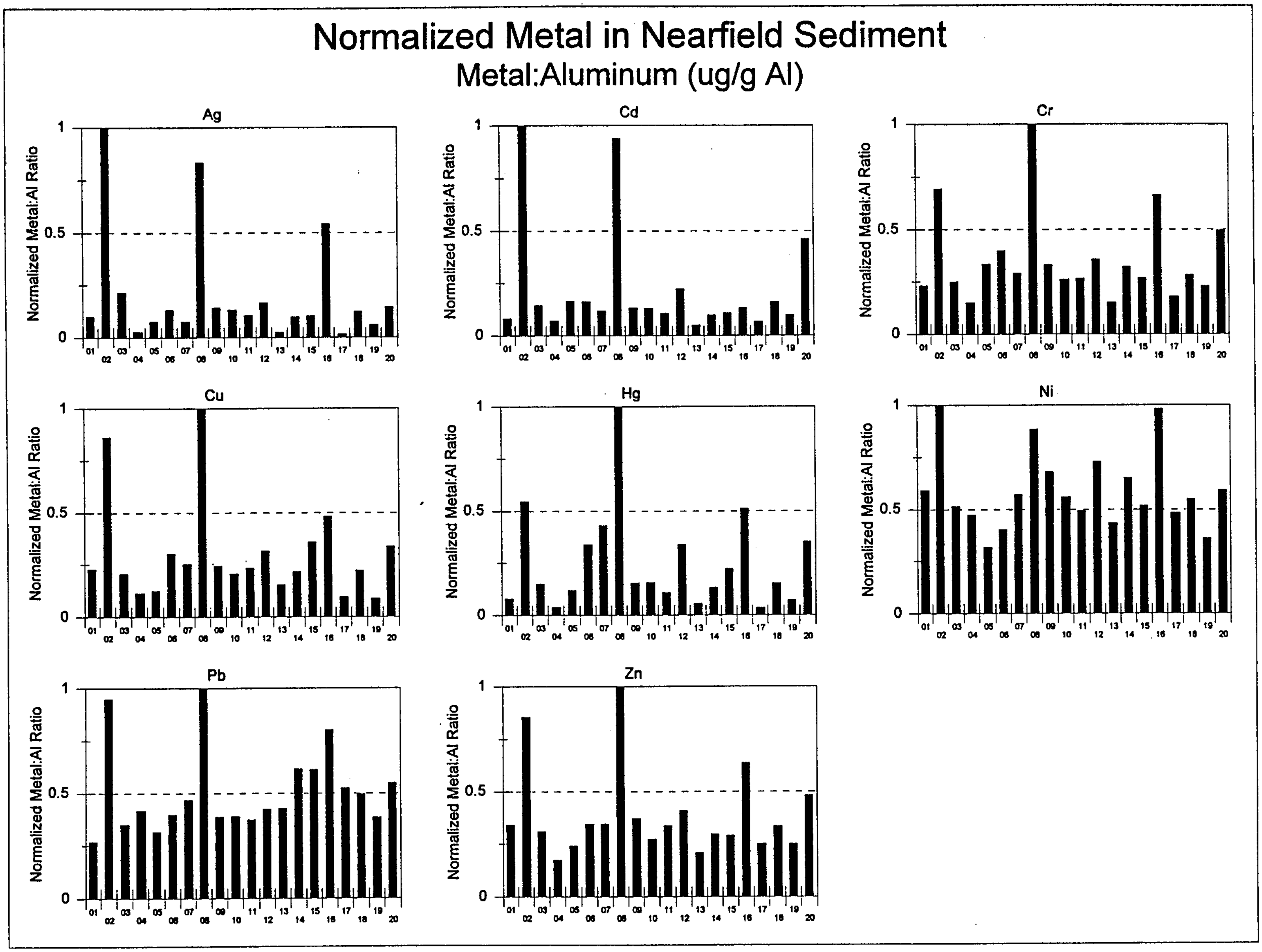


Figure 3-2 Spatial distribution of contaminants in sediment of Massachusetts Bay.
a. Metal:aluminum ratio normalized to the highest value.
Nearfield: station locations (x-axis) are shown in Figure 3-1.

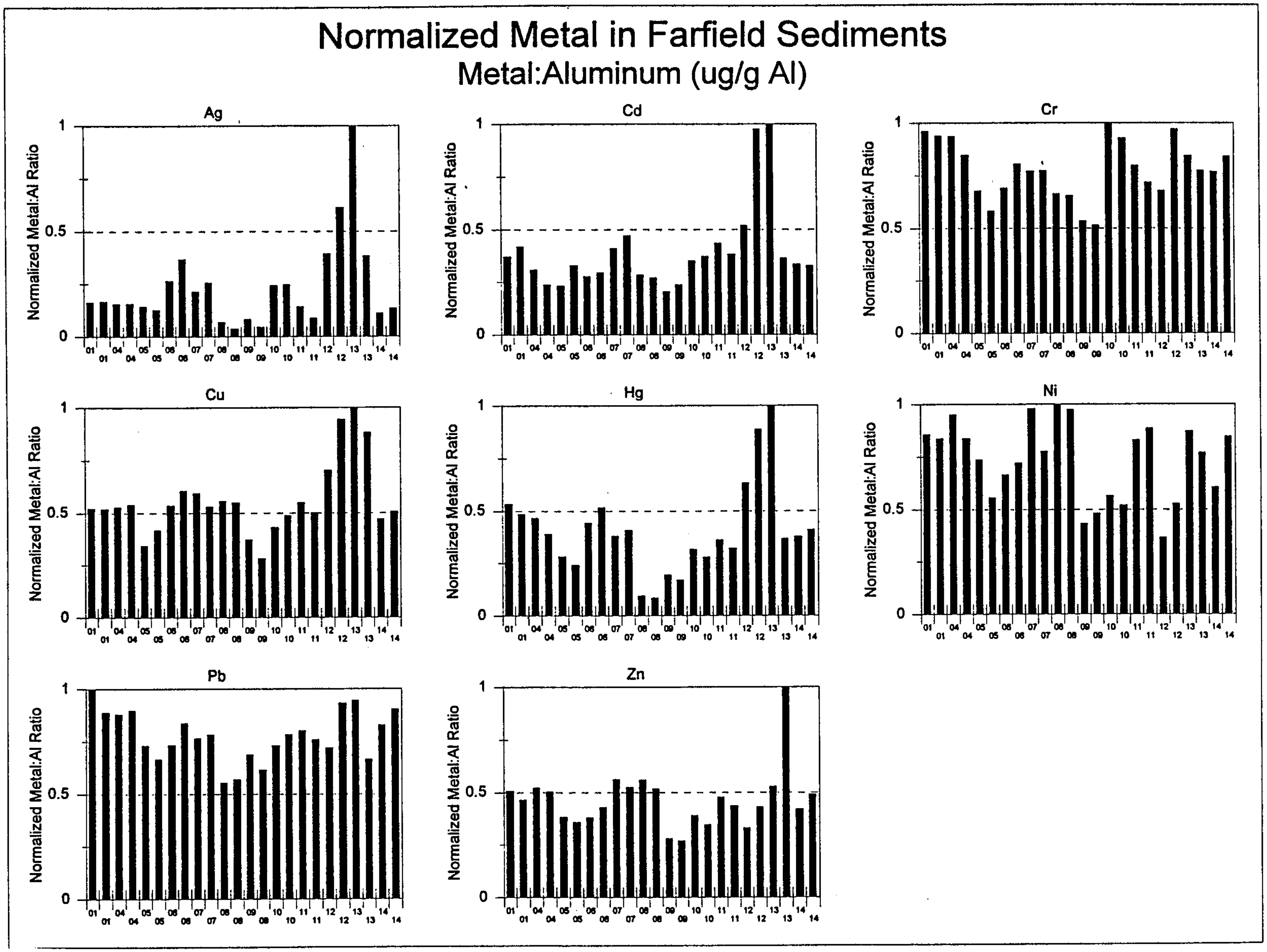


Figure 3-2 Spatial distribution of contaminants in sediment of Massachusetts Bay.
b. Metal:aluminum ratio normalized to the highest value.
Farfield: station locations (x-axis) are shown in Figure 3-1.

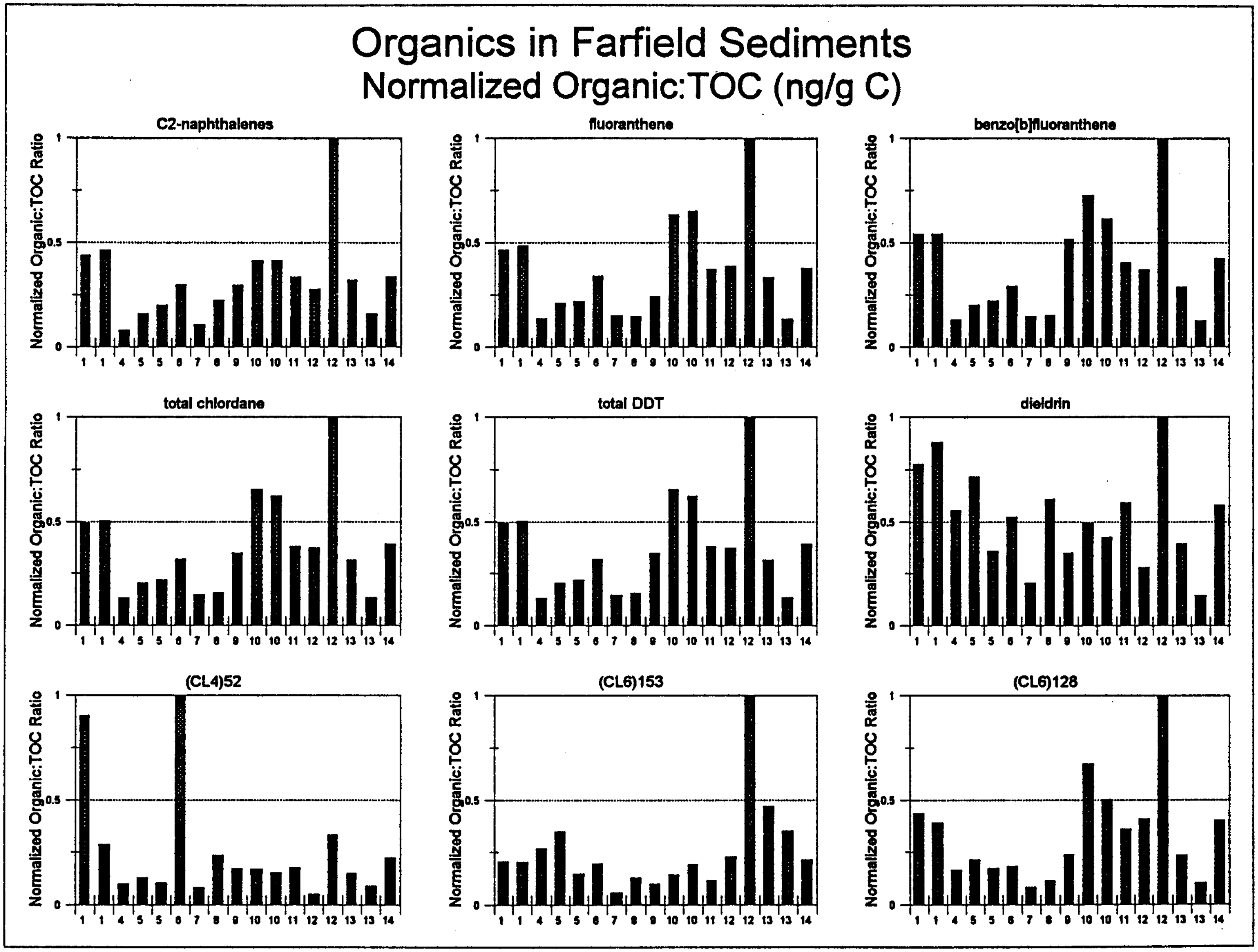


Figure 3-2 Spatial distribution of contaminants in sediment of Massachusetts Bay.
c. Organic contaminant:TOC ratio normalized to the highest value.
Nearfield: station locations (x-axis) are shown in Figure 3-1.

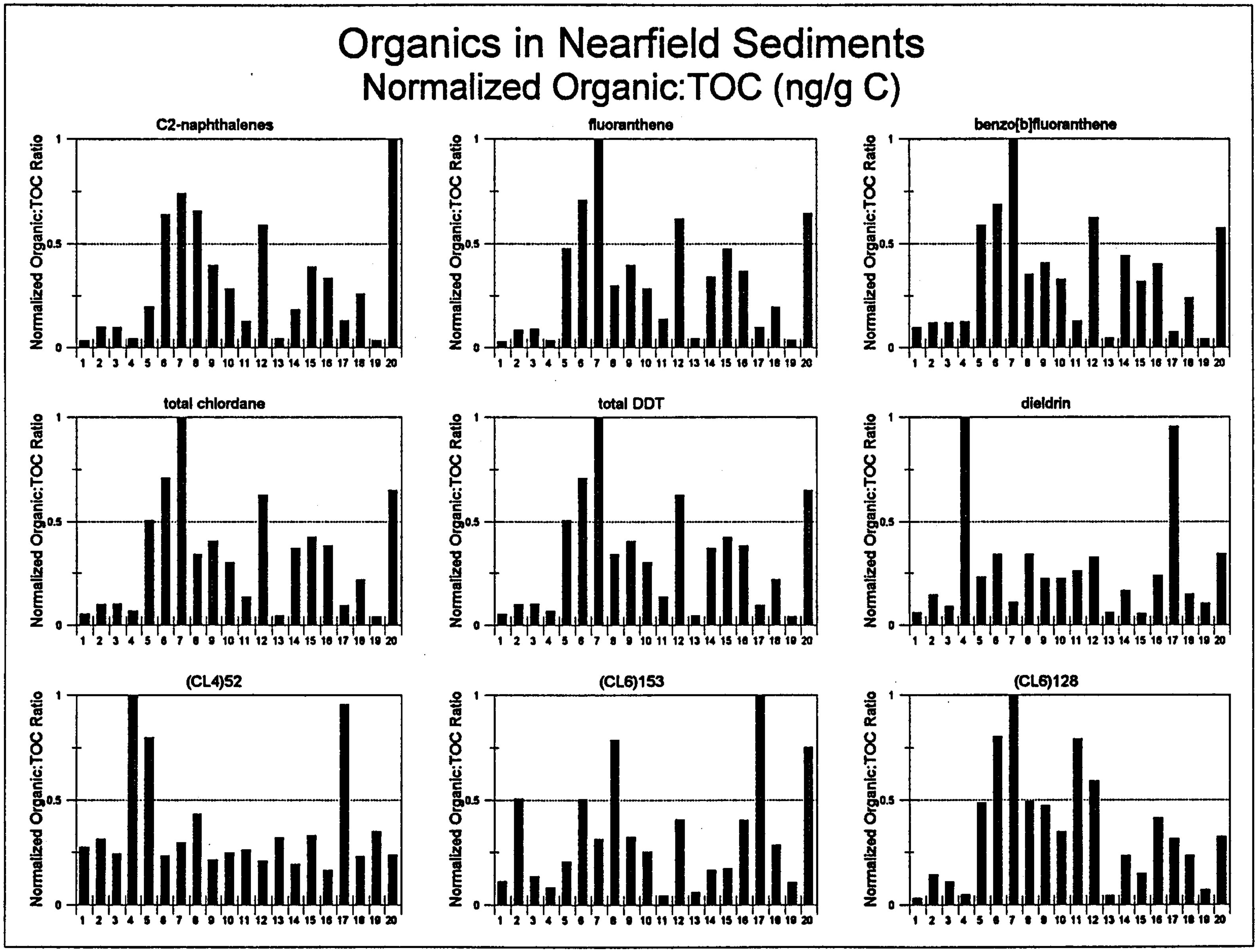


Figure 3-2 Spatial distribution of contaminant is sediment of Massachusetts Bay.
d. Organic contaminant:TOC ratio normalized to the highest value.
Farfield: station locations (x-axis) are shown in Figure 3-1.

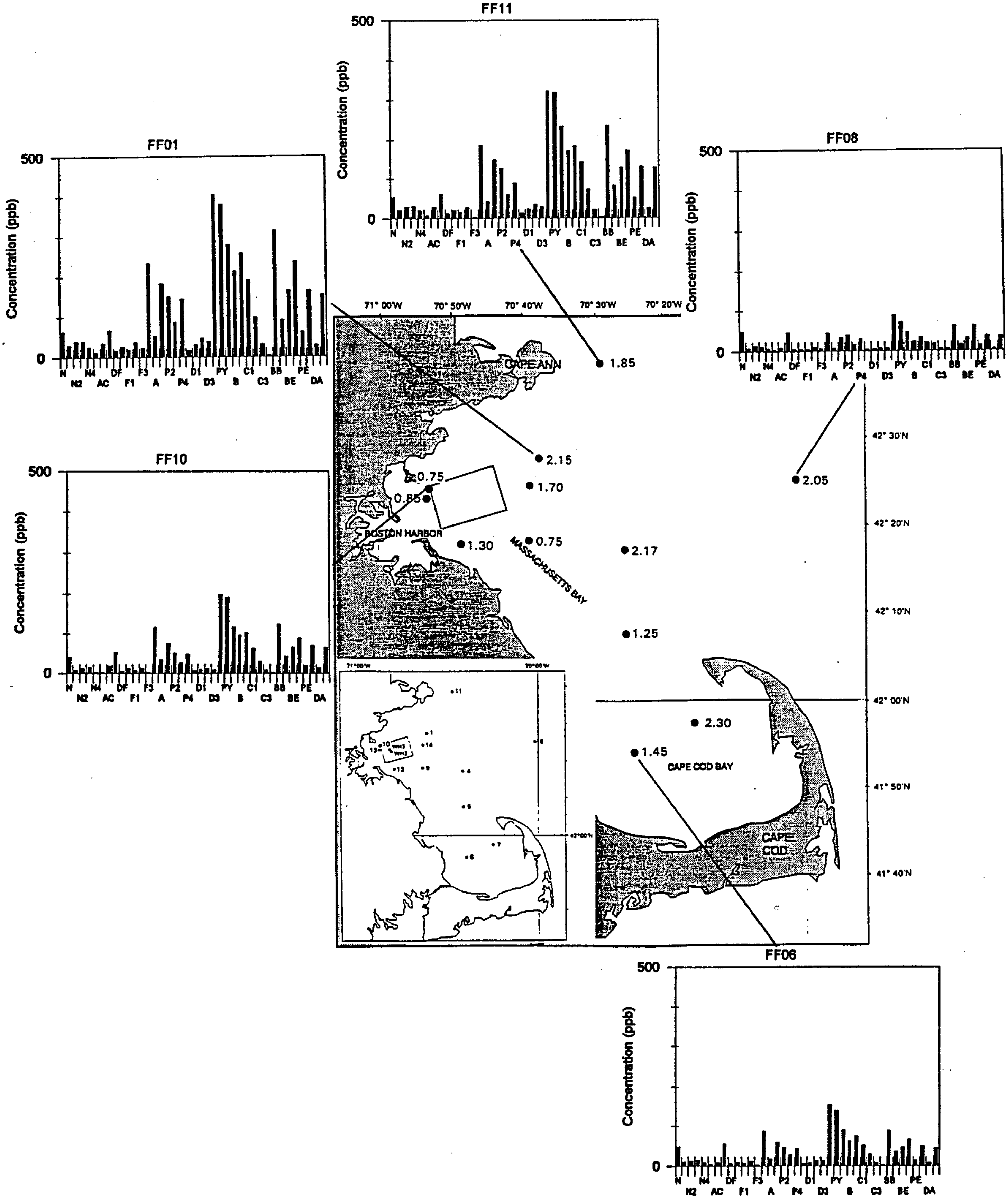


Figure 3-3 Organic contaminant distributions at several farfield stations. Absolute concentrations are shown. TOC values are listed adjacent to the station marker.

a. PAH

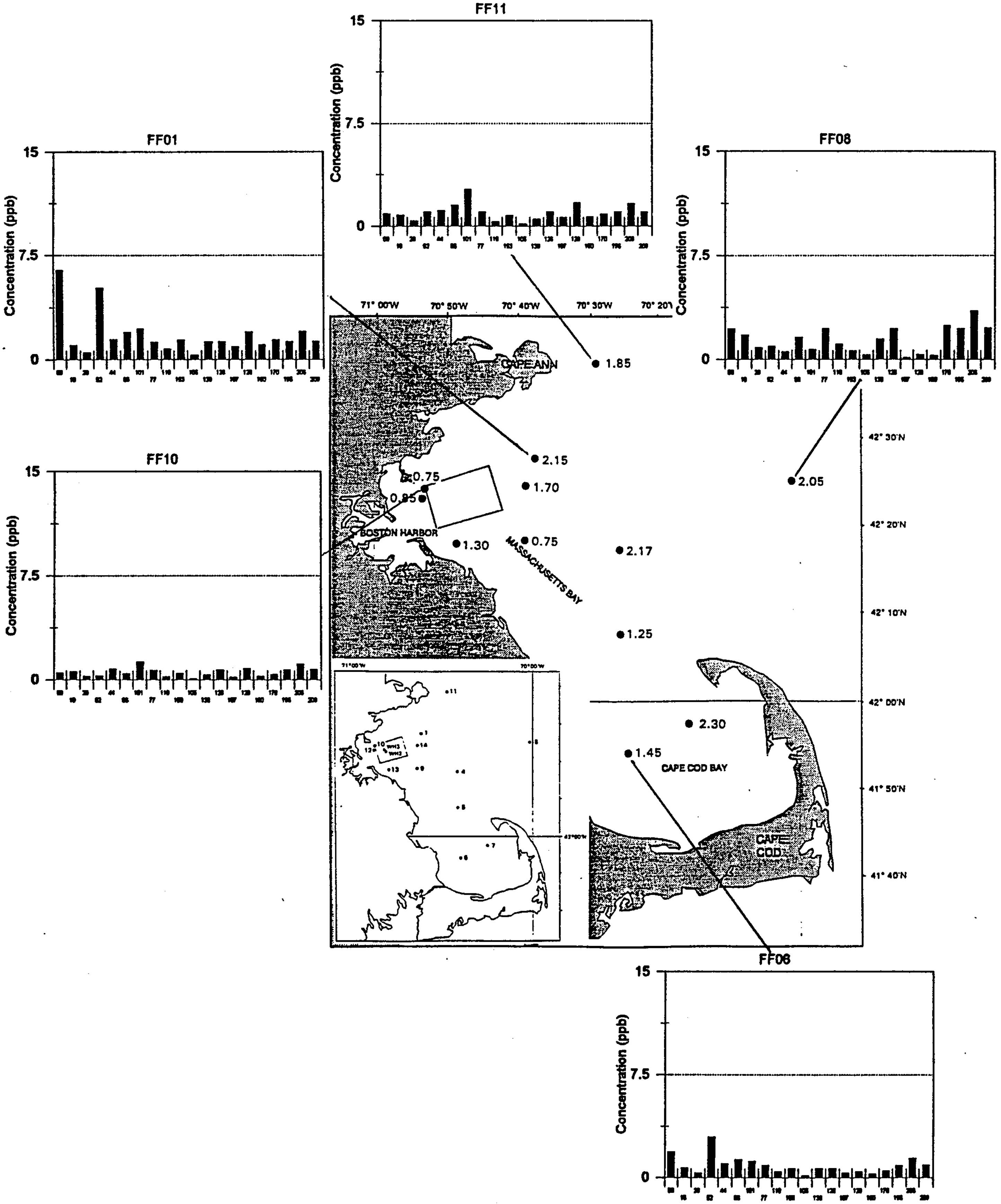


Figure 3-3 Organic contaminant distributions at several farfield stations. Absolute concentrations are shown. TOC values are listed adjacent to the station marker.

b. PCB

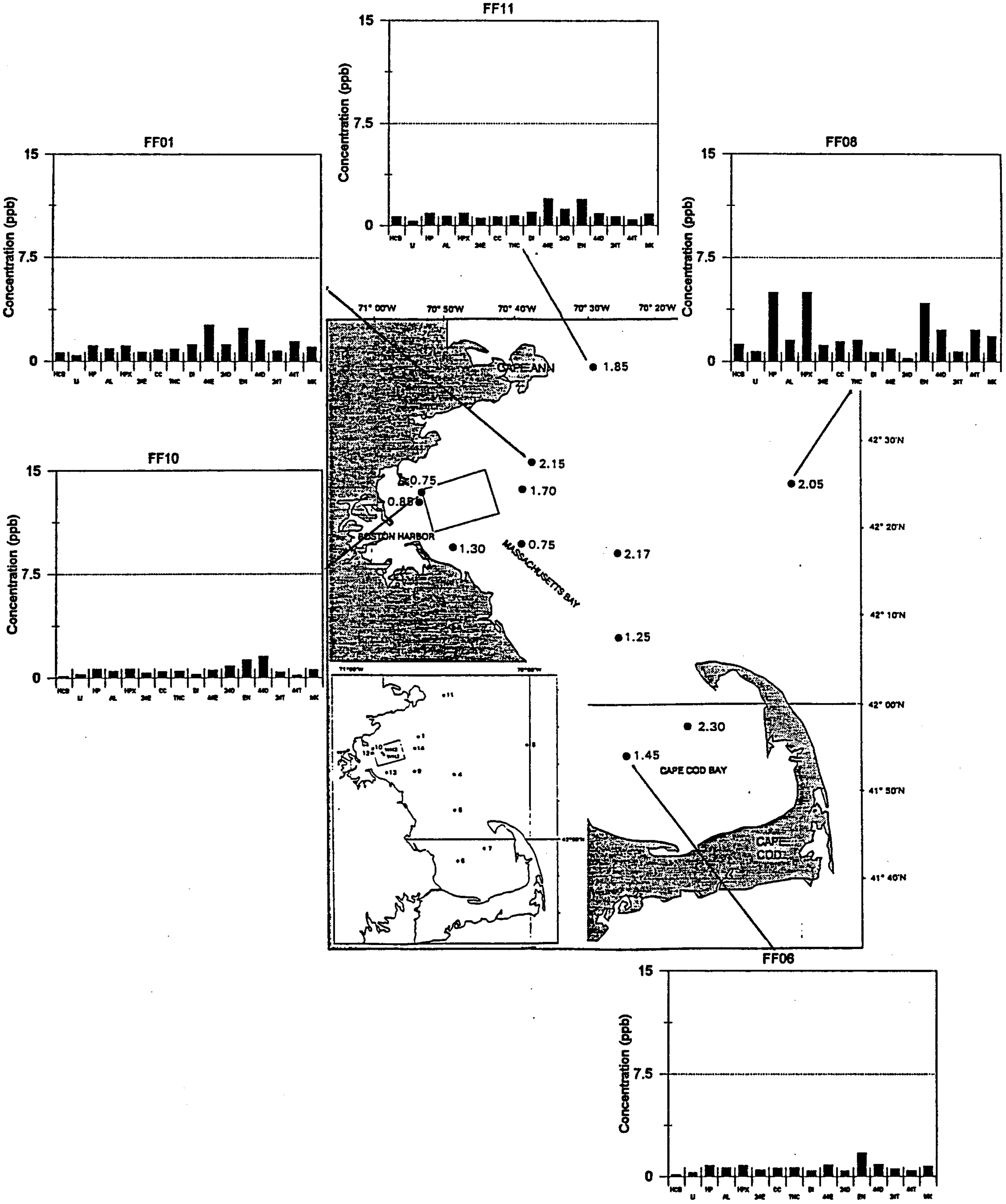


Figure 3-3 Organic contaminant distributions at several farfield stations. Absolute concentrations are shown. TOC values are listed adjacent to the station marker.

c. Pesticides

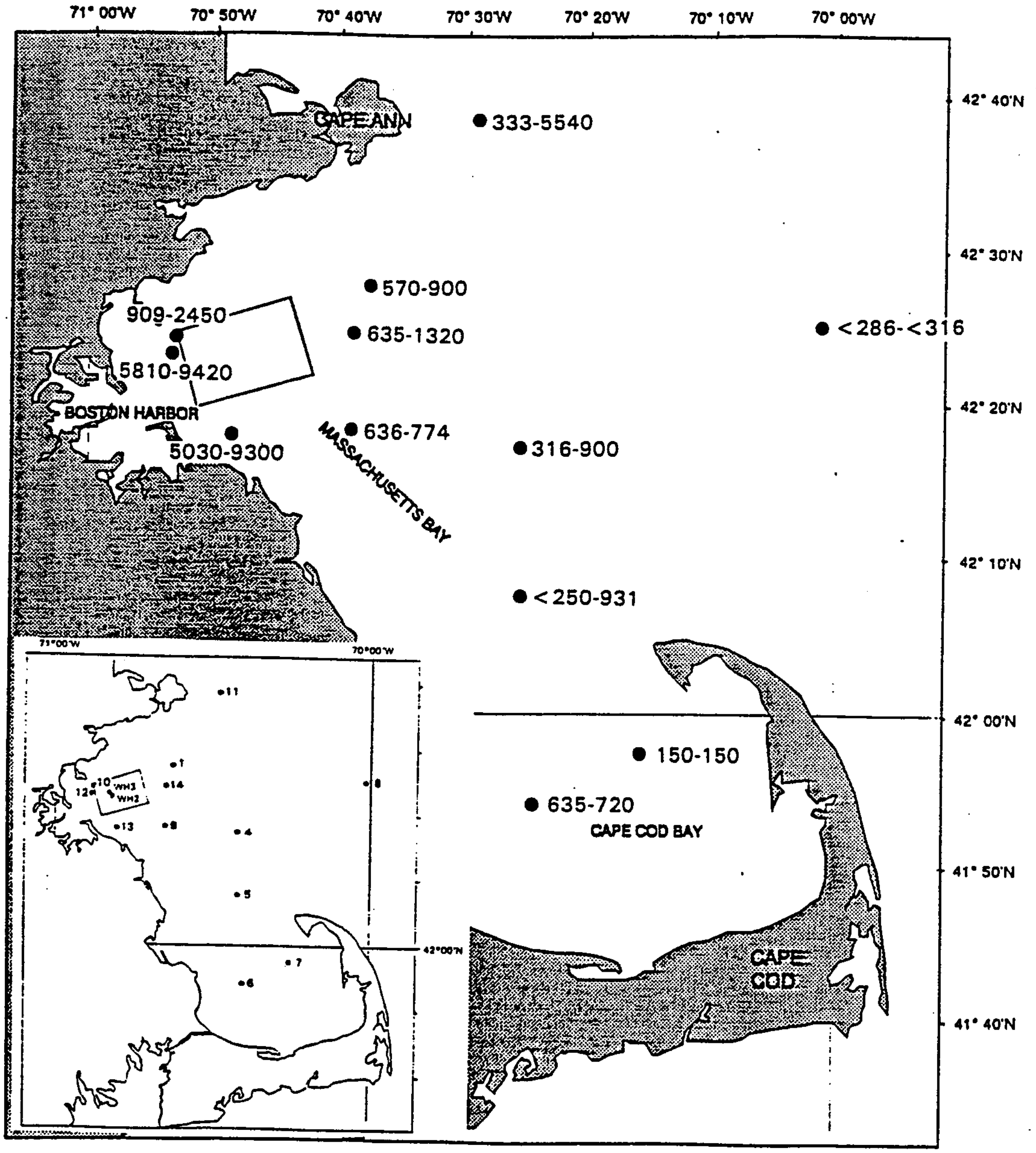


Figure 3-4 Spatial distribution of *Clostridium perfringens* in farfield sediments (from Blake *et al.* (1993b). Range of spore counts are shown.

Note that stations NF2, NF8, and NF12 are three of the more stable nearfield sites with respect to benthic biology and sedimentology (Blake *et al.,* 1993b). In the nearfield there is some overlap between stations enriched in contaminants and stations depleted in species richness (NF2, NF4, NF8, and NF20) and diversity (NF6, NF8, and NF20). This overlap is less apparent for the farfield, where species richness was lowest at FF8 and FF11 and species diversity was lowest at FF5, FF7, FF8, and FF11. The lack of overlap is not surprising given the low contaminant concentrations relative to published threshold values (draft sediment quality criteria and NOAA effects levels).

3.2 Relative Abundance of Contaminants

The relative concentrations of the individual PAHs in the sediment are quite different from those in the MWRA effluent (compare Figure 2-4 to Figure 3-3). The predominance of higher-molecular-weight PAHs in the sediment sample is typical of sediments far removed from sources of fresh petroleum. Weathering of the lighter PAHs during transport offshore effectively enriches the sediment in the higher-molecular-weight PAHs. Note that this type of contaminant distribution comparison (or more sophisticated pattern recognition analysis) could be very useful in providing a sensitive indicator of change and in estimating the amount of change due to the new MWRA outfall. Because the MWRA effluent is enriched in lighter PAHs, a change in the sediment PAH distribution is likely to be observed much more easily than a change in total PAHs. This type of data analysis is not explicitly considered in the present MWRA monitoring plan.

3.3 Variability of Contaminant Concentrations

The within-station variability (for farfield samples) is similar to that reported previously (Shea *et al.,* 1991). Using a one-tailed (we are really only concerned with increases in concentration) power curve (alpha = 0.05, beta = 0.20, cv = 0.50, n=3), the detectable change generally ranges from about a factor of two to five depending on both the contaminant and the station. EPA predicted changes of this order or larger near the outfall for many contaminants (EPA, 1988), so the present monitoring design is probably adequate to meet the objective of testing whether the impact is within the bounds projected in the SEIS (MWRA, 1991). Of course, actual contaminant increases will probably be much lower than predicted in the SEIS (Shea and Kelly, 1992) so it is quite likely that no increase will be detected for most contaminants at most sampling sites. This does not mean that the monitoring plan is inadequate; rather it indicates that exposure to sediment-bound contaminants will not change much from the present conditions.

There are insufficient data from these sampling sites to rigorously assess interannual variability. However, we can compare data from nearfield stations 9, 10, and 11 with the "C" transect that was sampled in 1987 for the STFP because they are essentially the same sites. There are no obvious or remarkable trends in the data; the TOC-normalized fluoranthene data (Figure 3-5) are typical of the data sets. A given station (e.g., NF10) can exhibit a wide range of concentrations within and between years, but we could still detect a level of change that is relevant to "meaningful change" because contaminant concentrations are well below draft sediment quality criteria (SQC). Therefore, a priority of the monitoring design must be to locate the most stable depositional sites within range of expected impact.

3.4 Implications for Assessing Changes and Effects

The sediment (and contaminant) deposition modeling used in the STFP and SEIS was reviewed by Shea and Kelly (1992) and EPA (1993). The model clearly overestimates the rate of deposition to nearfield sediments. This is compounded by an overestimate of the concentration of contaminants on the settling particles. However, it is not a trivial exercise to improve upon the accuracy of this simple model. Based on recent models of particle transport in Boston Harbor (Stolzenbach *et al.*, 1993) and on new estimates of effluent particle settling (Wang, 1988) from holography measurements, it is possible that 75 to 90% of the effluent particles will not settle within the nearfield monitoring box. These smaller, non-settling particles are generally enriched in contaminants (by weight) and, thus, could represent nearly all of the initial particle-bound contaminant load. Therefore, 10% of the

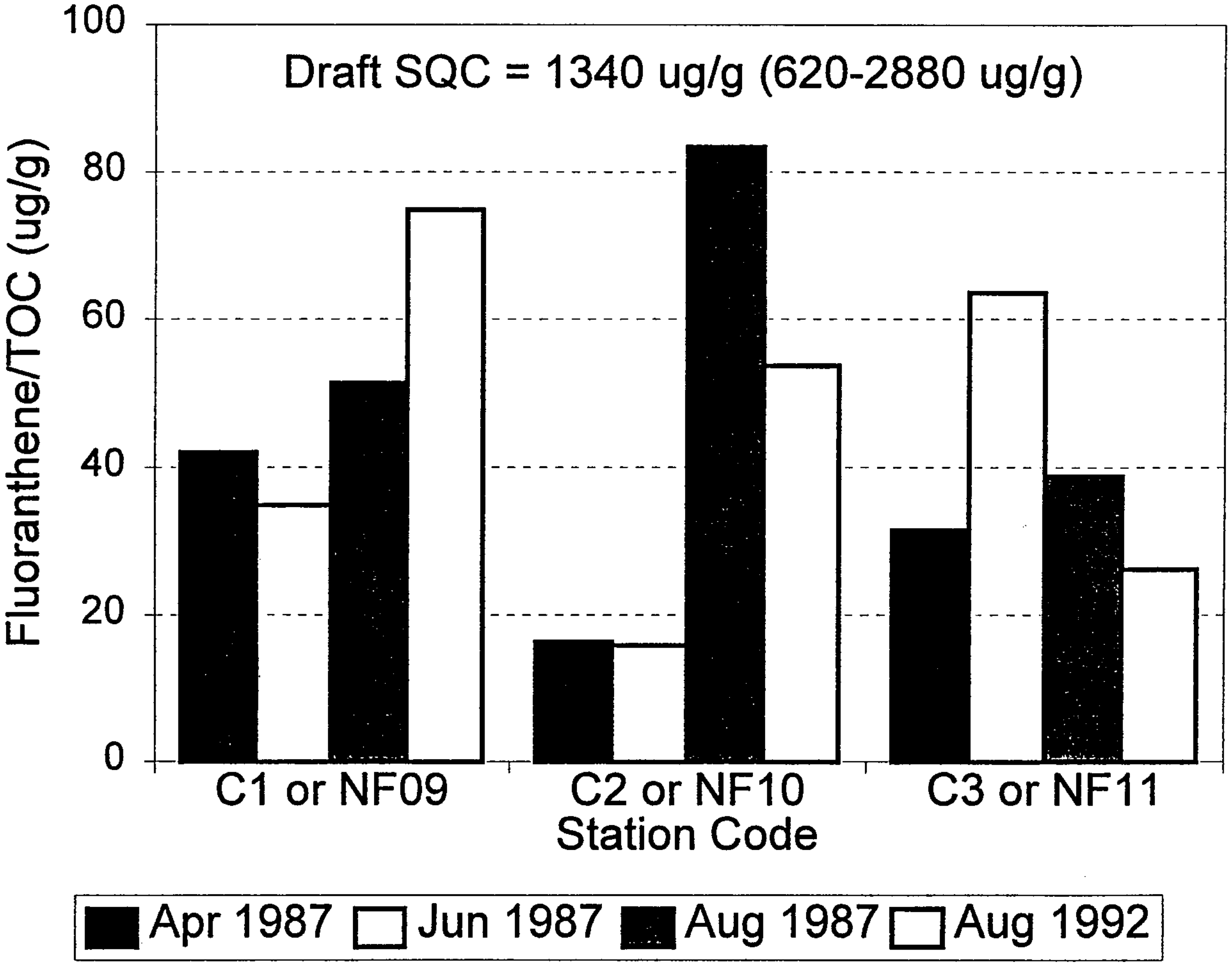


Figure 3-5 Time series of fluoranthene/TOC values at three nearfield sediment stations.

total contaminant load is probably a more reasonable upper bound for deposition in the nearfield. Of course much of this will be transported to other areas during the fall/winter storms, but significant accumulations could still take place in depositional areas within the nearfield. Unlike the water column (discussed above), where there have been significant improvements in modeling, there is still a high degree of uncertainty over the deposition of effluent contaminants to the bottom sediments of Massachusetts Bay. This places a high priority on developing a refined nearfield sediment (and contaminant) deposition model and monitoring in stable depositional areas within the nearfield. Farfield modeling and monitoring are of less importance because the total load to Massachusetts Bay will not be changing much over the next decade (see Section 2).

Our uncertainty in modeling contaminant fate does not prevent us from assessing potential meaningful changes in the sediment. Despite some controversy over the applicability of organic carbon-normalized SQC, there is little doubt that organic carbon reduces the availability of hydrophobic contaminants (PAH, PCB, chlorinated pesticides) to organisms (Shea, 1988). It also has been shown that organic carbon from sewage sludge reduces the availability of some metals more than typical sedimentary organic matter (Swartz, 1986). Thus, the issue is not simply future enrichment of contaminants in sediments, rather it is enrichment relative to the organic carbon. A statistically significant increase in contaminant concentrations could take place in nearfield sediments while the effective exposure (a function of bioavailability) remains the same or even decreases because of corresponding increases in sedimentary organic carbon. This possibility is illustrated in Figure 3-6 and 3-7 for PCB and fluoranthene, respectively.

There is a simple, linear relationship between PCB and total organic carbon (TOC) in nearfield sediments (Figure 3-6). When effluent is discharged through the new outfall we should observe one of three plausible scenarios: (1) a depletion of contaminants relative to TOC, (2) no detectable change in the ratio of contaminants to TOC, or (3) an enrichment of contaminants relative to TOC. In the first scenario, we would not expect any direct changes in toxicological effects but we could see problems related to enrichment of TOC. The

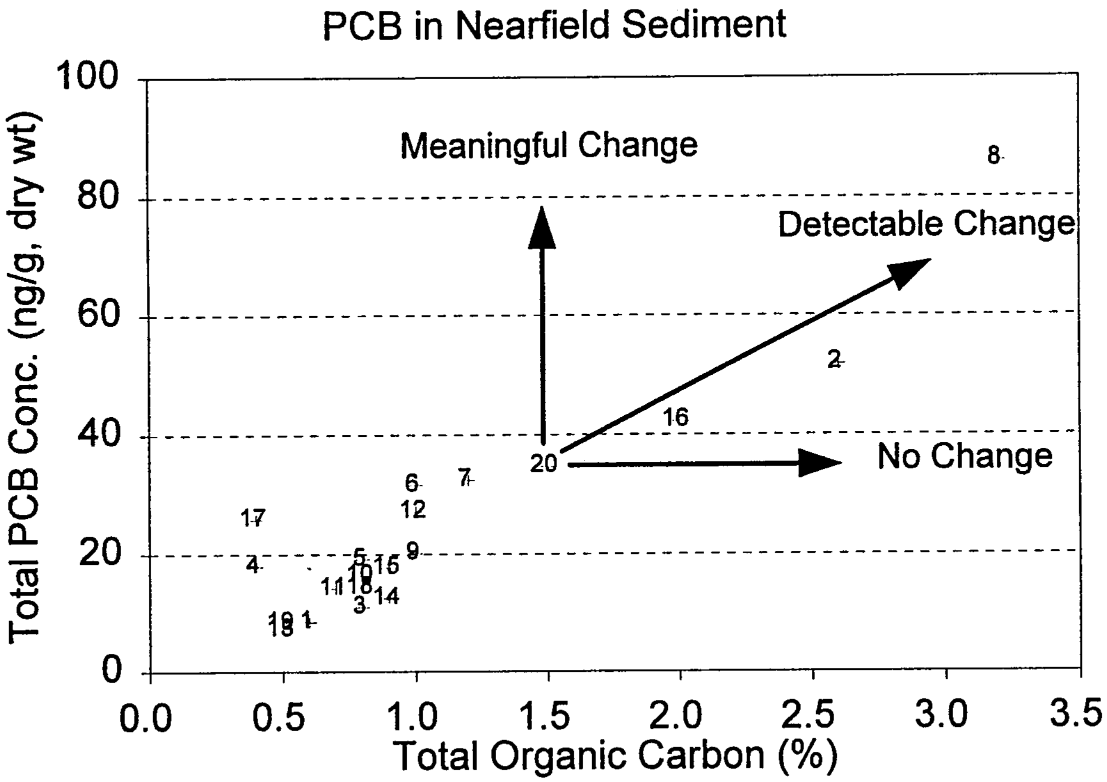


Figure 3-6 **Relationship between PCB and TOC in nearfield sediments and three plausible scenarios of change after effluent is discharged through the new outfall.**

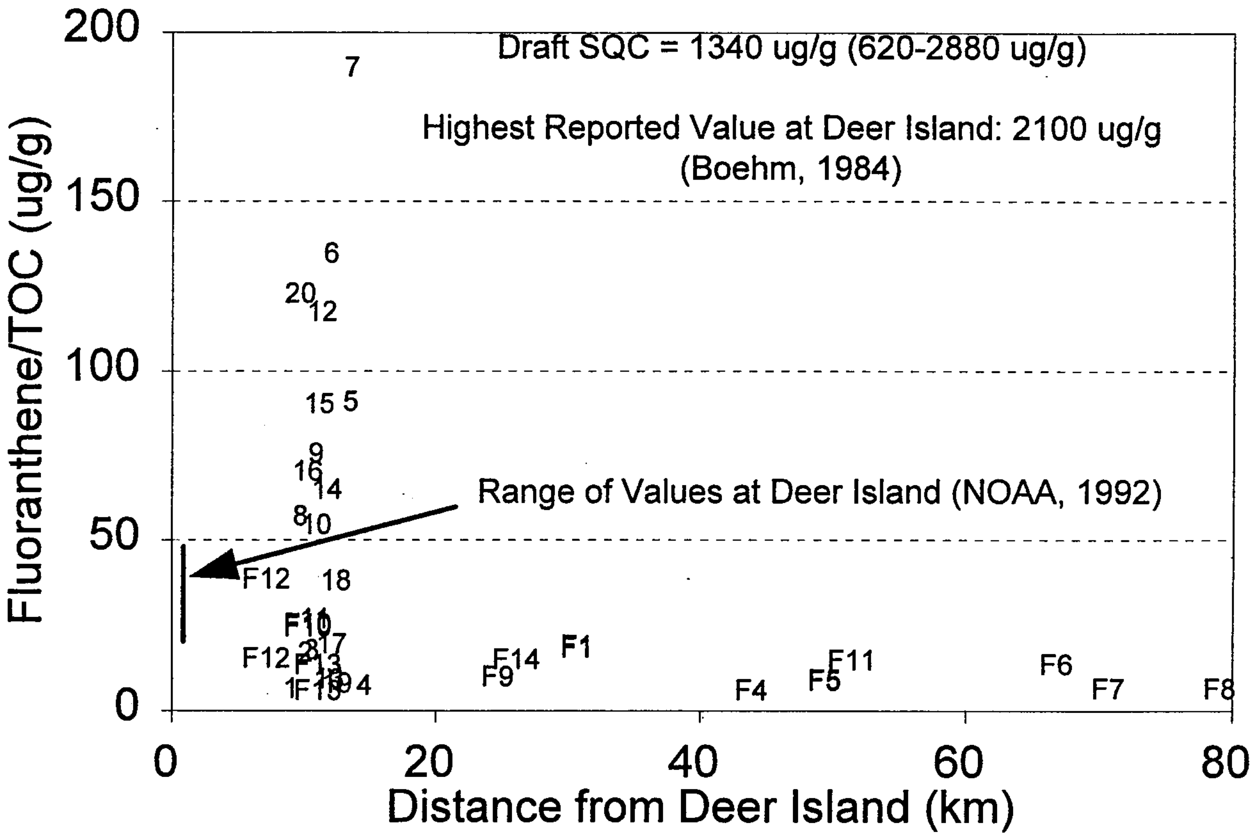


Figure 3-7 TOC-normalized fluoranthene concentrations as a function of distance from Deer Island. Station numbers are used to designate data points (see Figure 3-1 for station locations).

second scenario could result from either no change in concentrations of both contaminants and TOC or equal relative changes. Thus, contaminant concentrations could change significantly (statistically) yet the availability of the contaminant remains low due to the increased TOC. The third scenario is one that we must monitor. Enrichment of contaminants relative to TOC would indicate the potential for adverse biological effects — raising the question — how large must the increase be before we decide that it is meaningful? For contaminants that have draft sediment quality criteria, we could simply make comparisons of the TOC normalized concentrations to the SQC (Di Toro *et al.,* 1991). There are no SQC for PCBs. For fluoranthene (Figure 3-7), we would need about a 10- to 100-fold increase in present concentrations (relative to TOC) to exceed the draft SQC. Note that recent measurements of fluoranthene in sediments near the present Deer Island outfall are in the same range as that found in the nearfield sediments (Figure 3-7) indicating that changes in fluoranthene concentrations in Massachusetts Bays sediment will not be large.

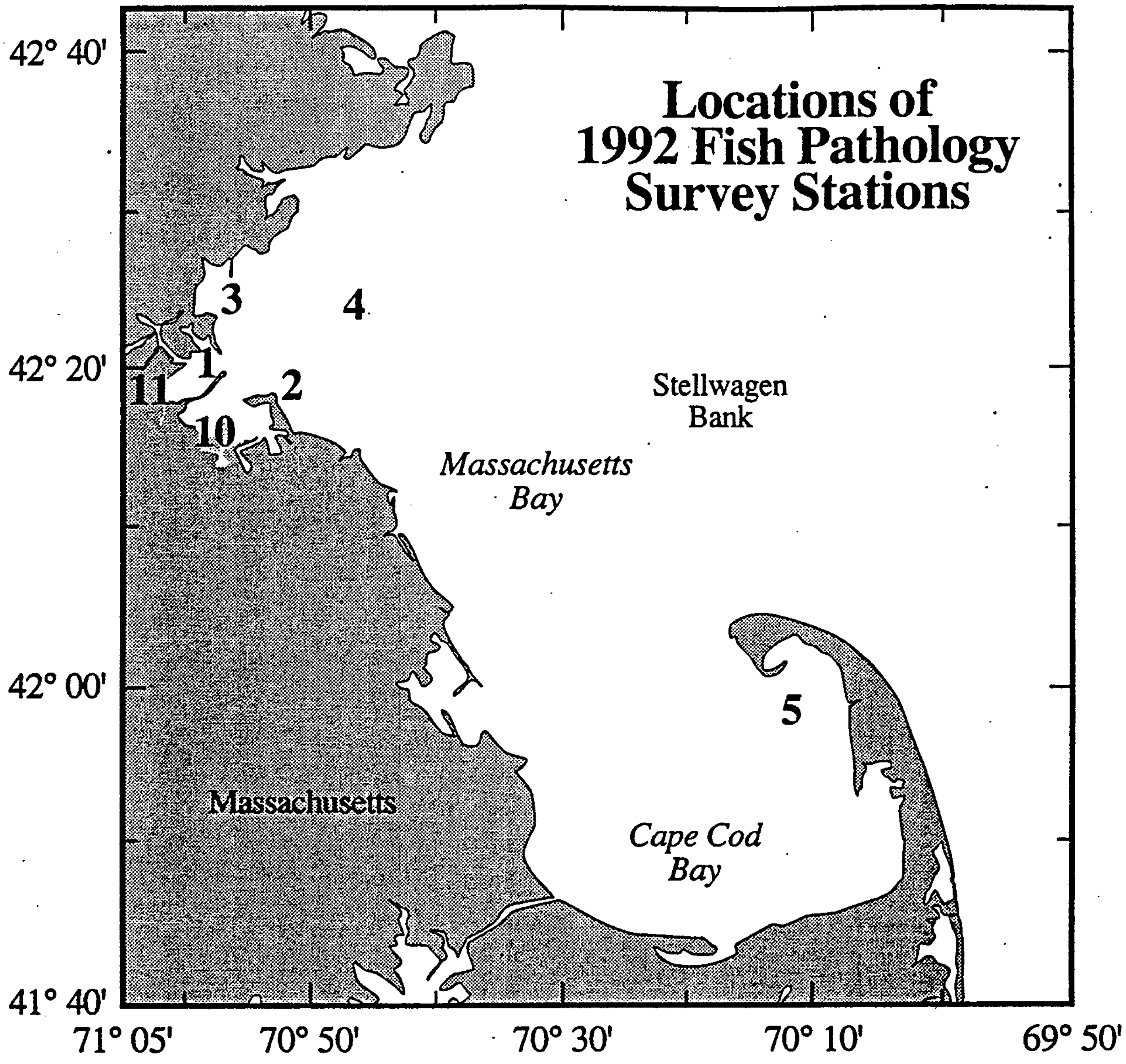
Finally, with respect to future NPDES permit limits, if we could estimate the fraction of sedimentary organic carbon that comes from the MWRA effluent, we could calculate an effluent permit limit based on the draft SQC. Alternatively, we could use the conservative assumption that all the sedimentary organic carbon comes from the MWRA effluent, which may not be a bad assumption for the low organic carbon sediments near the future outfall. Given the importance of the ratio of contaminants to organic carbon, as well as the effect of organic enrichment on benthic respiration and infuana, it is imperative that we understand the fate of organic carbon discharged through the MWRA outfall.

4.0 FISH AND SHELLFISH MONITORING

To monitor for potential toxicological effects on fish and shellfish (or on humans that eat the fish), the MWRA is establishing a baseline of metals and organic contaminants in tissues of these organisms, along with an assessment of their physiological condition (MWRA, 1991). In 1992, contaminant concentrations were measured separately in winter flounder fillet and liver, and lobster meat and hepatopancreas and histopathological observations were taken of the flounder. At each station (Figure 4-1), one composite flounder sample (from seven individuals) was analyzed along with three individual flounder that were selected to span the range of observed histological scores to help establish the relationships among chemical and physiological monitoring parameters. Three individual lobsters were analyzed; no lobster composites were made.

Contaminant concentrations varied considerably among individual animals at a given site (see Appendix A for full data set) and the mean values were generally much higher than the corresponding value for the composite sample as shown in Figures 4-2 and 4-3 for PCB. Some stations have both the lowest and highest observed body burdens for a given contaminant (e.g., PCB in lobster muscle at Cape Cod Bay) and some have the lowest value of one contaminant and the highest value of another contaminant (Figures 4-4 and 4-5).

Histological indices also varied considerably among individuals at a given site (Moore and Stegeman, 1993) and there was a clear association between contaminant body burdens and various histological indices (Figures 4-2, 4-4, and 4-6). The prevalence of hydropic vacuolation (HV) in liver cells appeared to be a particularly good indicator of contaminant-induced physiological effects and is shown in Figure 4-7. In particular, the prevalence of centrotubular HV (CHV) has a strong association with contaminant concentrations in the tissue of the fish (Figure 4-2). PCB body burdens provide the best correlation and sensitivity to CHV. We have no data on exposure concentrations in the sediment that coincide with collection of these fish, but data from previous studies (Shea *et al.*, 1991) indicate that there is a much weaker association between observed effects in flounder and concentrations of



Key: Site 1 - Deer Island Flats (DI)
Site 2 - Off Nantasket Beach (NB)
Site 3 - Broad Sound (BS)
Site 4 - Future Outfall Site (FOS)
Site 5 - Eastern Cape Cod Bay (CCB)

Figure 4-1 Location of fish and lobster sampling sites in 1992. Lobster were sampled at sites 1, 4, and 5 only. Sites 10 and 11 were flounder pathology only.

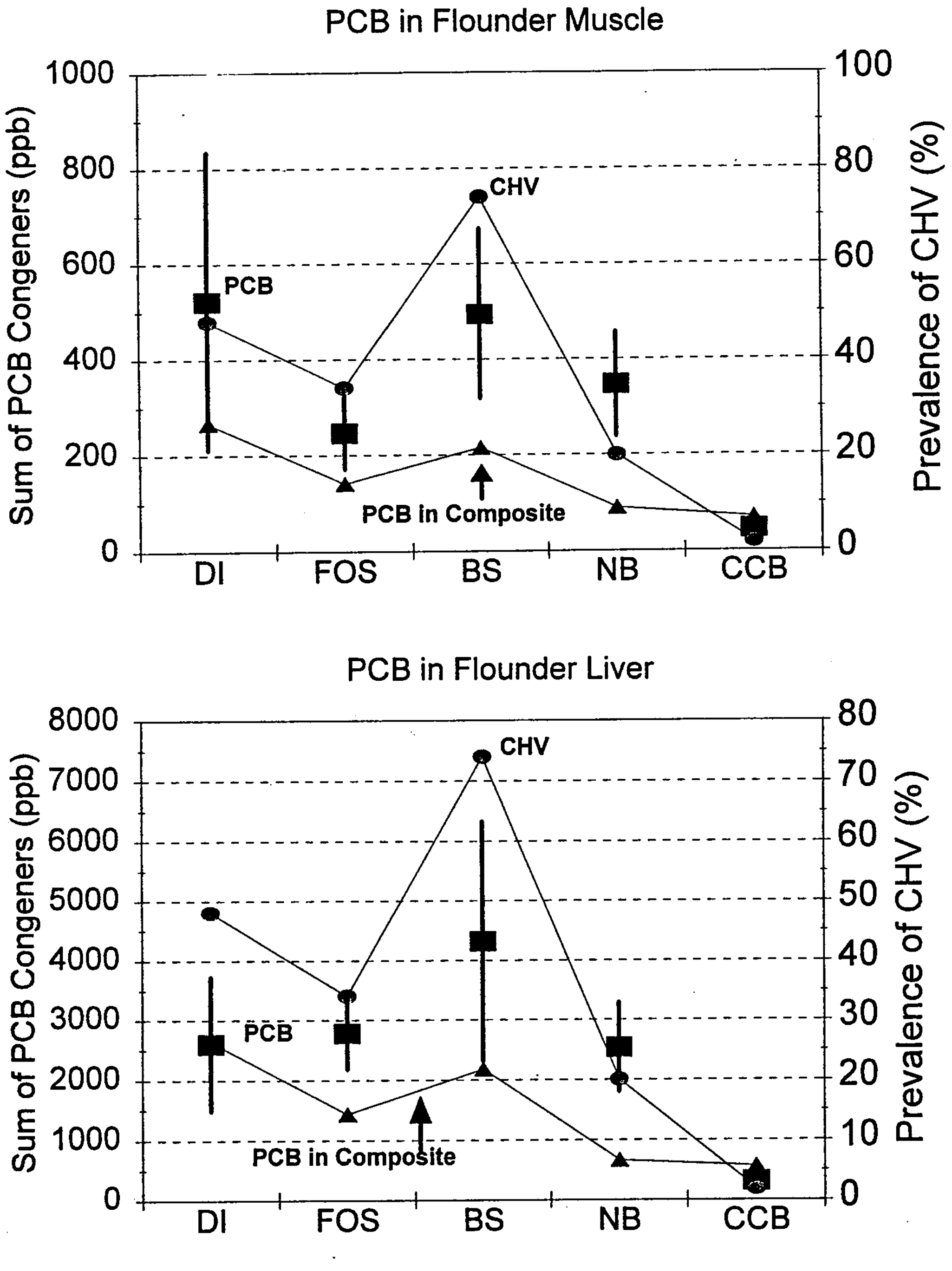


Figure 4-2 Sum of PCB congeners in flounder tissue and prevalence of centrotubular hydropic vacuolation (CHV) of liver cells at flounder sampling sites (see Figure 4-1 for locations). The mean (square) and standard deviation (vertical line, n = 3) are shown for individual animals; results of composite samples are represented by a triangle.

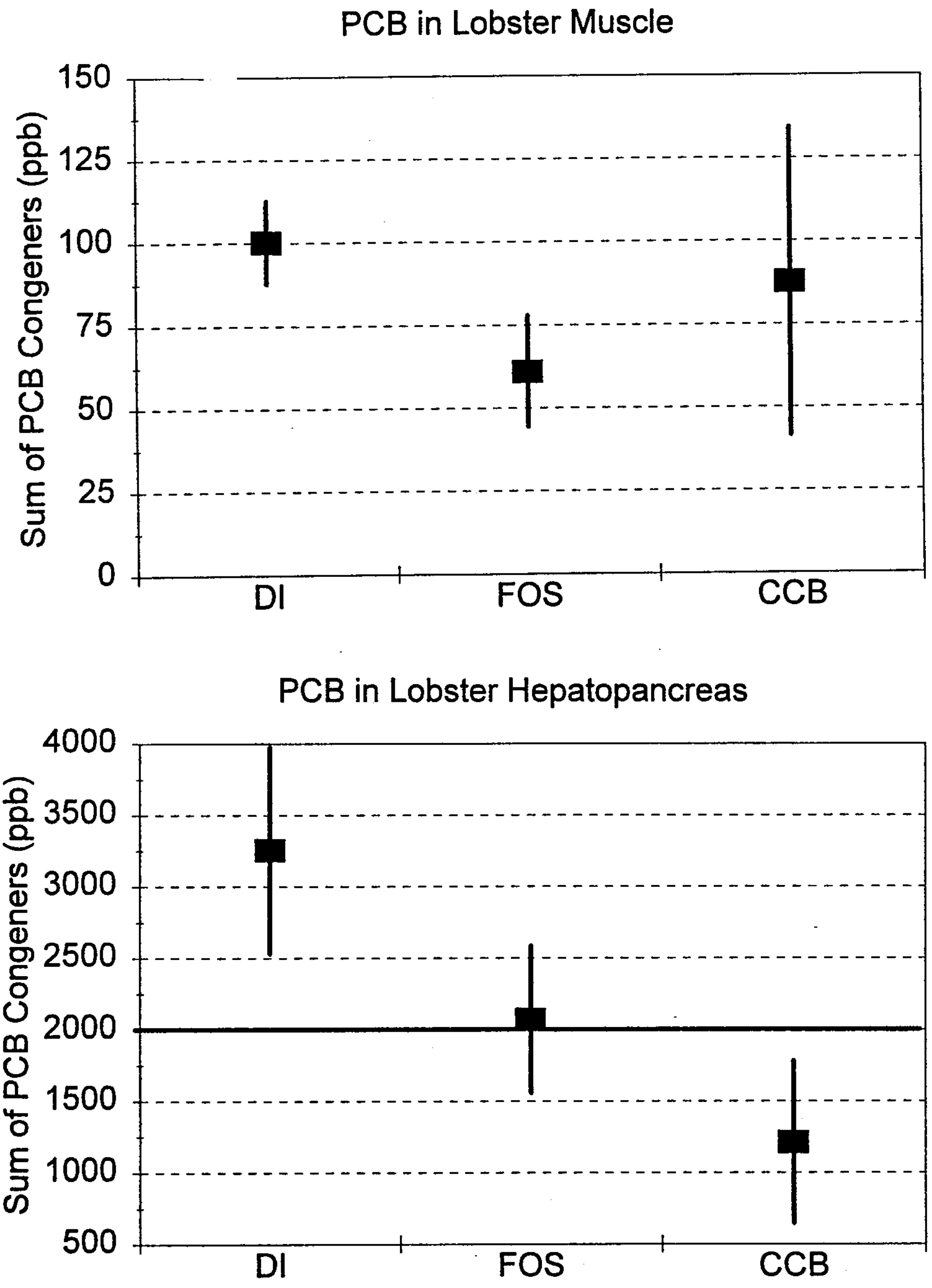
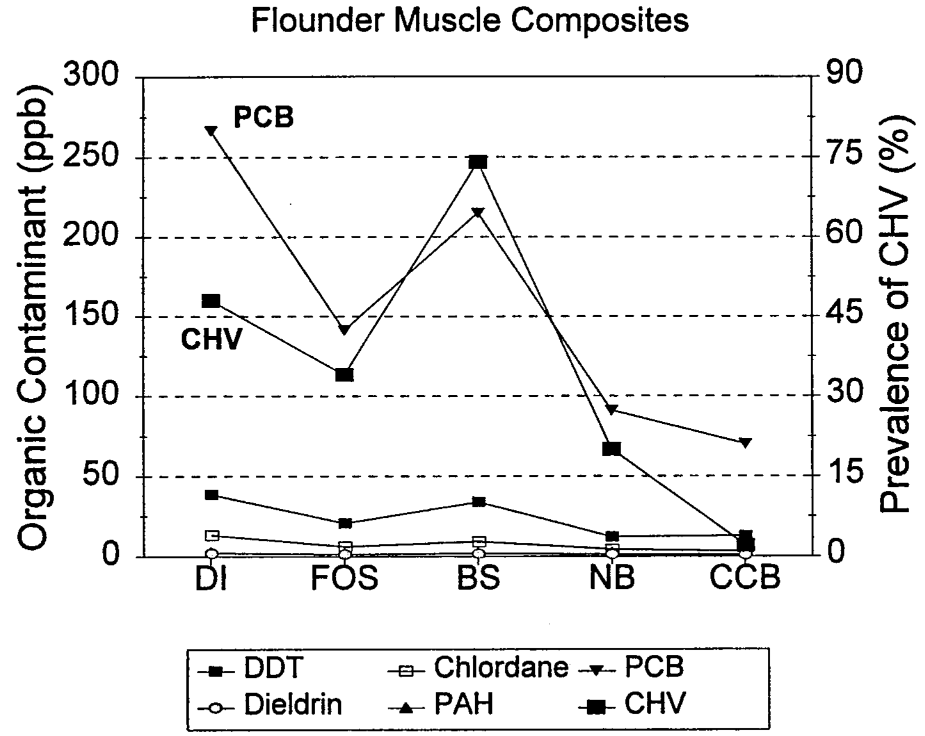


Figure 4-3 Sum of PCB congeners in individual lobster tissue at the lobster sampling sites (see Figure 4-1 for locations). The mean (square) and standard deviation (vertical line, n=3) are shown.



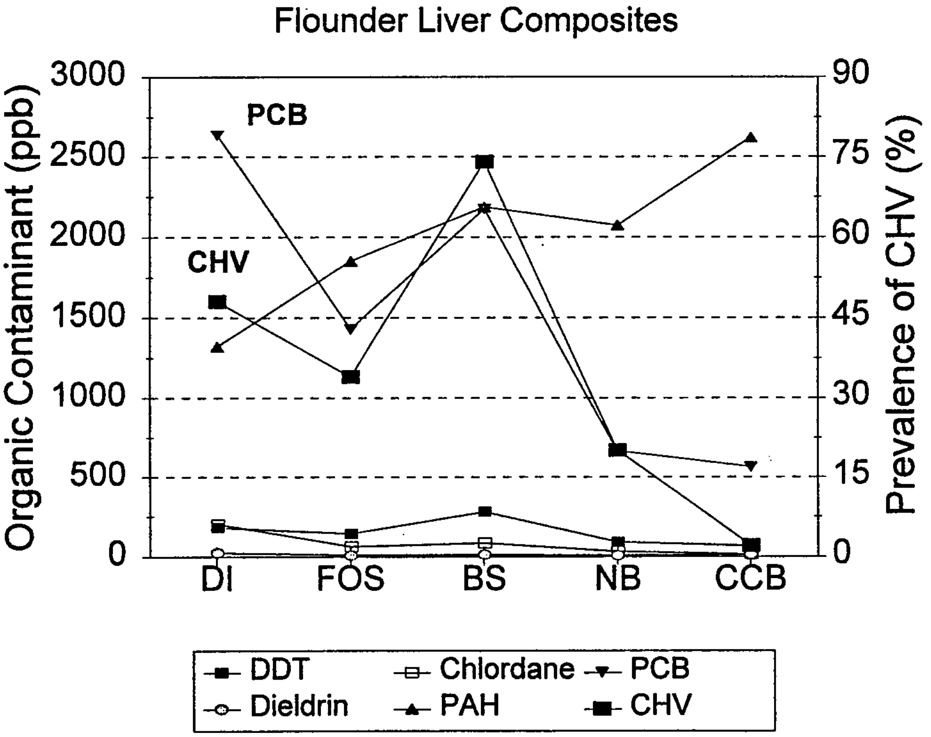


Figure 4-4 Contaminant concentrations in flounder tissue and prevalence of centrotubular hydropic vacuolation (CHV) of liver cells at flounder sampling sites (see Figure 4-1 for locations). Results of composite samples (n=7) are shown.

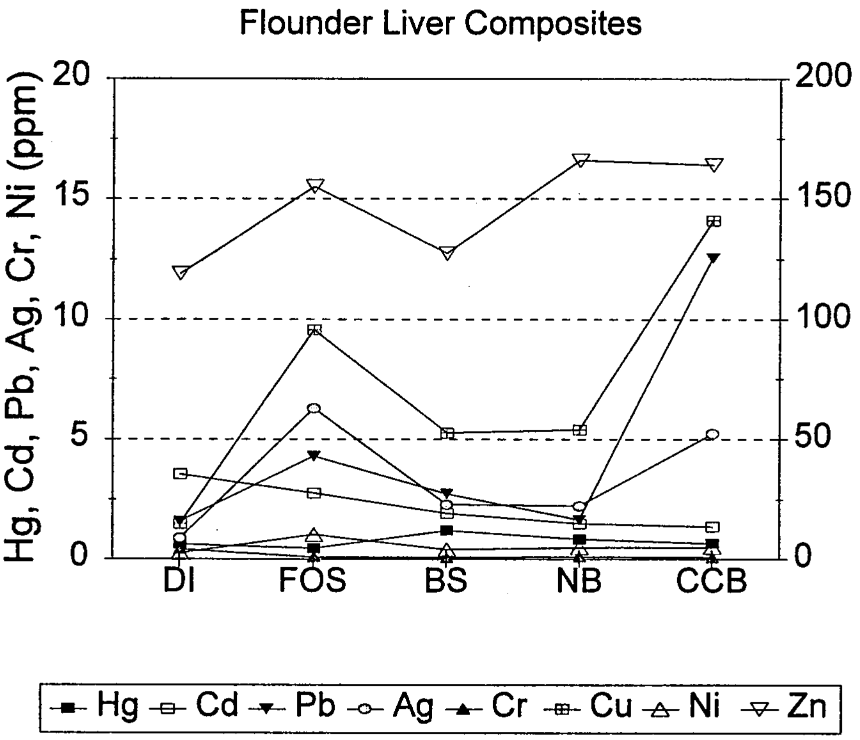


Figure 4-5 **Metal concentrations in flounder liver at flounder sampling sites (see Figure 4-1 for locations). Results of composite samples are shown.**

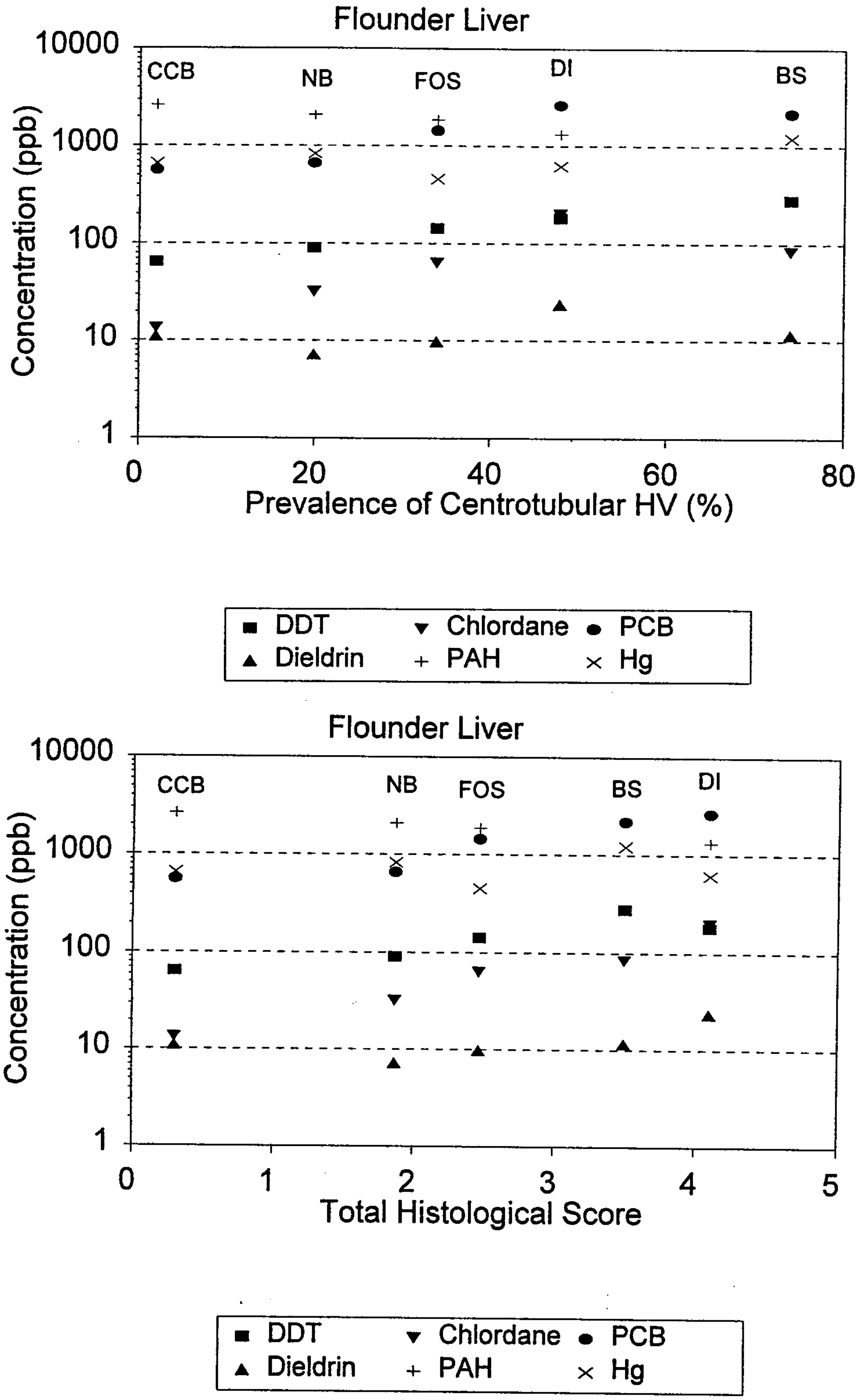


Figure 4-6 Association between contaminant body burdens and observed histological indices.

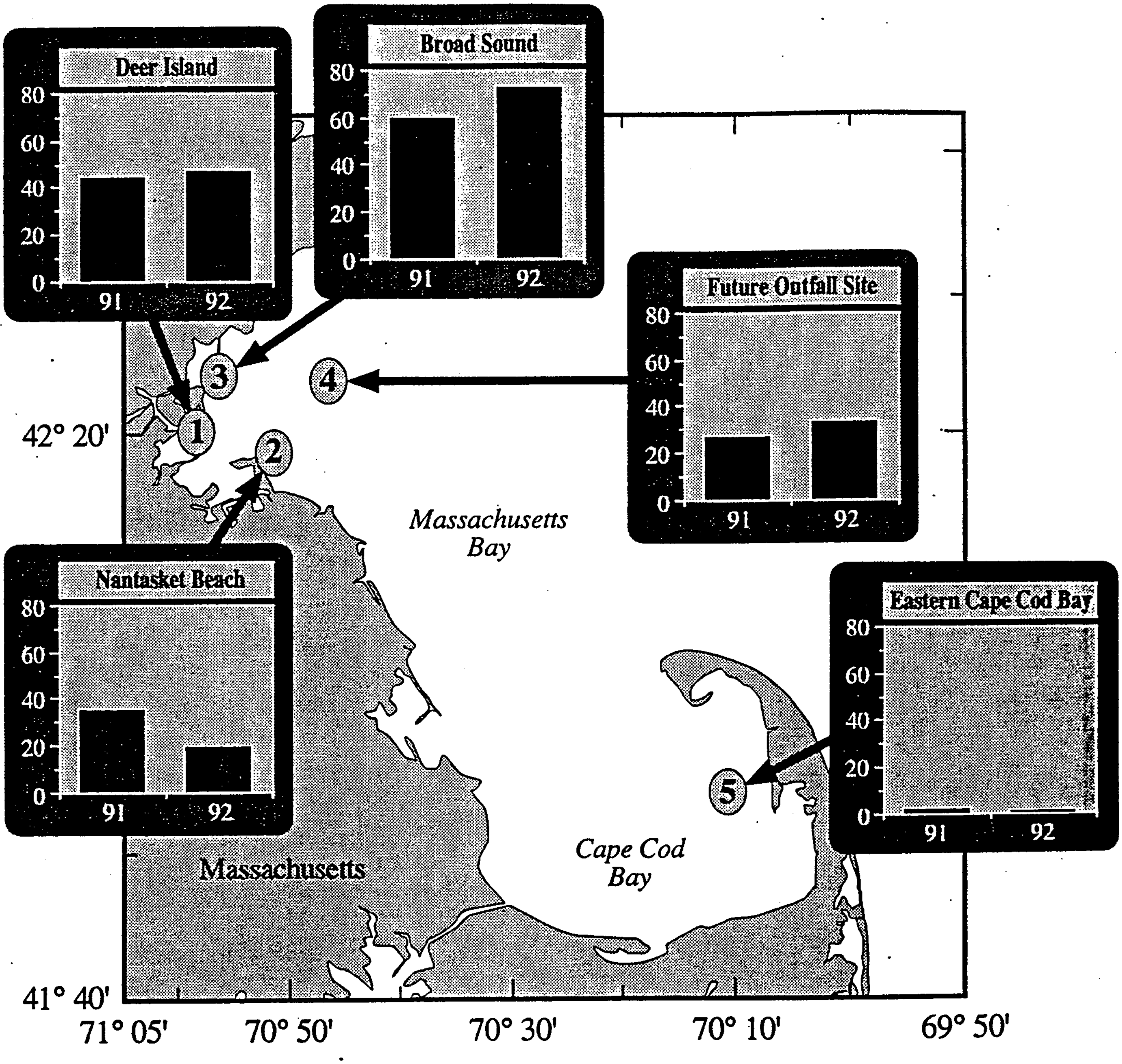


Figure 4-7 **Prevalence of hydropic vacuolation in winter flounder liver in 1991 and 1992 from five stations in Massachusetts and Cape Cod Bays (from Moore and Stegeman, 1993).**

contaminants in the sediments. Equilibrium between the contaminants in the sediment and the fish apparently has not been established at these sites. This could be due to other routes of exposure, migration of the fish from areas of different sediment quality, or both. The large range in contaminant body burdens (and pathological scores) within a population at a given site indicates that we may be sampling mixed populations that have migrated from different areas.

Our uncertainty over animal migration is significant, and it becomes greater as we consider how the new outfall could affect migration. This needs to be addressed because an assumption in the monitoring design is that each year the same population is being sampled. But how do we know whether animals that are typically inhabitants of Boston Harbor or elsewhere won't bias the measured effect by migrating toward the outfall when discharge begins? It is entirely possible that contaminant body burdens and/or histological indices could increase in animals collected at the new outfall site not because of increases to historically indigenous populations, but rather due to increased migration of animals that normally reside in Boston Harbor or other more contaminated areas (Lynn Harbor) to the new outfall site once it becomes operational. These new immigrant animals will bring with them the contaminant body burdens and physiological conditions that they acquired in the more contaminated areas. There presently is no means of distinguishing between these two causes in the monitoring plan. This problem could be minimized by analyzing only the young of the year, but then the measure will be limited to shorter-term exposure that is unlikely to reveal significant pathological changes because of the long induction time. The use of caged mussels complements the winter flounder and lobster measurements by removing the variables of animal migration and contaminant metabolism (flounder have a much greater ability to metabolize organic contaminants than do mussels). However, as discussed in Section 2 we expect no change in the body burdens of caged mussels beyond about 60 m from the diffuser.

With respect to human health, there are no exceedances of any FDA action limits at any of the sites. We are near (within about a factor of four) the FDA limit for total PCBs (2 ppm, wet weight based on total Aroclors) in lobster hepatopancreas at Deer Island, the future outfall site, and Cape Cod Bay. We are near the FDA action limit for mercury (1 ppm, wet weight) in both lobster hepatopancreas and lobster muscle at the same three sites. Thus, consumption of lobster hepatopancreas represents the greatest risk to human health.

With respect to detecting change, the greatest power is obtained by analyzing composite samples rather than individual samples. This is clearly seen in a comparison of composite data versus individual data (Figure 4-2) and is well supported by statistical theory (A. Solow, pers. comm., 1993). A detailed discussion of this variability and the implications to detecting changes is given in Appendix B. The power analyses presented in Appendix B indicate that, for most contaminants, a two- or three-fold change in mean concentration could be detected. Given the concentrations reported here, this should be adequate to detect whether concentrations are approaching or have exceeded FDA action limits.

5.0 SUMMARY

The primary goal of this report is to provide a synthesis of recent data related to toxic contaminants to help in the review of the MWRA effluent outfall monitoring plan. With this goal in mind, we summarize this report using a format that follows the monitoring questions related to toxic contaminants (MWRA, 1991).

Public Concern: Is it safe to eat fish and shellfish?

Will toxic chemicals accumulate in the edible tissues of fish and shellfish, and thereby contribute to human health problems?

The most recent effluent contaminant concentrations and dilution models indicate that water column concentrations of contaminants will remain within a factor of two of present concentrations very near the outfall diffuser. Thus, for any fish that receives it's primary exposure through the water, or through feeding on organisms whose primary exposure is through the water, we do not expect contaminant body burdens to increase. For demersal and benthic organisms (winter flounder and lobster), we are less certain about their potential body burdens because we do not have reliable estimates of the exposure concentrations in the sediment nor do we have an adequate understanding of their migration and feeding behavior once the outfall begins discharging. Further, we do not know what the effect of increased organic carbon loading will have on bioavailability of the contaminants. Despite these uncertainties, it is unlikely that a significant portion of the fish or shellfish eaten by any one person would be taken from the most contaminated waters (or sediment) within the zone of initial dilution. Thus, the risk of any human health effects related to long term exposure (e.g., cancer) does not appear to be any greater than present risks. We recommend that further consideration be given to modeling the transport of contaminants to nearfield sediments and understanding the migratory behavior of flounder and lobster so that more reliable projections of future contamination can be developed.

Will the water column near the diffuser mixing zone have elevated levels of some contaminants?

Concentrations of Cu, Pb, Ag, some lighter weight PAH (naphthalenes), dieldrin (summer only), and lindane will increase at least two-fold at the edge of the mixing zone (about 60 m from the diffuser). The increase will be greatest for naphthalenes (34-fold increase), dieldrin (six-fold), and Ag (five-fold). However, naphthalenes are not persistent in the water and dieldrin, Ag, and the other contaminants will partition into particles and be removed from the water through sedimentation. Thus, the conservative dilution models used in this report (and by EPA, 1993) overestimate the spatial extent of increased contaminant concentrations. A simple fate model (see Shea and Kelly, 1992) could be linked to the dilution model to refine these conservative estimates. However, these refinements will only result in projections of lower impact or risk to aquatic organisms and thus we do not place a high priority on this model refinement.

Will contaminants affect some size classes or species of plankton and thereby contribute to changes in community structure and/or the marine food web?

The monitoring plan only indirectly address this question. We compared contaminant concentrations predicted from the conservative dilution model to marine water quality criteria and it appears that if any effects take place, they will be confined to within a few 10's of meters from the diffuser. We believe it is unlikely that broad scale toxic effects will take place given the very small volume of water that will be affected, the lifetime and movement of plankton, and the generally low contaminant concentrations. This question could be answered more directly by conducting toxicity tests on indigenous (or cultured) plankton using diluted effluent, but available data do not provide a strong justification for these additional tests.

Will finfish and shellfish that live near or migrate by the diffuser be exposed to elevated levels of some contaminants, potentially contributing to adverse health in some populations?

Elevated levels of some contaminants will exist over spatial scales that could affect populations spending a significant amount of time very near the diffuser. Populations that reside more than a few hundred meters away from the diffuser do not appear to be at any greater risk than at the present. The variability of contaminant concentrations in animal tissues will allow for detection of about a two- to three-fold change. It is unlikely that this level of change will take place very far from the outfall. However we have uncertainty over the migratory behavior of the test organisms (flounder and lobster). These migration patterns must be established or alternative endpoints (e.g., caged mussels) should be used.

Will the benthos near the outfall mixing zone and in depositional areas farther away accumulate some contaminants?

Depositional areas near the outfall will undoubtably accumulate contaminants from the new discharge, but we are uncertain as to the magnitude and spatial extent. It appears unlikely that more than about a two-fold change in contaminant concentrations will take place beyond a few kilometers from the outfall. The variability of contaminant concentrations in sediment will allow for detection of about a two- to five-fold change with the present triplicate sediment analysis approach. A refined nearfield contaminant deposition model is needed (see above).

Will benthic macrofauna near the outfall mixing zone be exposed to some contaminants, potentially contributing to changes in community structure?

Exposure to contaminants will increase in the benthic environment near the outfall. However, this increase may be mitigated by a corresponding increase in organic carbon. In fact, organic enrichment and benthic respiration would appear to be more likely to affect benthic populations.

The fact that most of our projections and those of others (e.g., EPA, 1993) indicate that very little change or impact is likely to take place in Massachusetts Bay should not lull us into complacency about potential ecological effects. Our future efforts should be directed toward issues that we know the least about and have the greatest potential for broader scale impact. For example, even though the water quality modeling presented in this report is very simple, it is also very conservative. Further refinement in the model will lead to essentially the same conclusion, only with lower expected effects and perhaps better certainty. On the other hand, there is still cause for concern about what impacts may result in sediment near the outfall. Refinement in existing models of the transport of contaminants and organic carbon to sediments could be very beneficial (and would also be useful for refining the water column modeling).

While we have the technical ability to refine these transport models, we believe there is another issue that should receive a higher priority. The biological effects models presented in this report (and used by EPA to set NPDES permit limits) are based primarily on conventional toxicity endpoints (death and growth). Evidence is surfacing that more subtle biological effects (developmental and reproductive) can be caused at much lower concentrations of some contaminants (e.g., some chlorinated pesticides) or caused by contaminants that are not being measured but are probably present in the MWRA effluent (e.g., nonylphenols). We believe that more attention should be placed on these contaminants and their potential biological effects, rather than focusing only on refining conventional models and reducing uncertainty around predictions of impact that are generally considered acceptable.

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Appendix A

Chemistry Data from Fish and Shellfish and Sediment Monitoring

This 60 page appendix of raw data has been bound separately in order to reduce printing costs. If you are interested in receiving these data, they are available as Environmental Quality Department Miscellaneous Publication no. 23, titled "*Appendix A to Annual review of contaminants discharged by MWRA: 1993.*

It can be requested from Bernadette McCarthy, ENQUAD, at the address or phone number on the back of this report.

Appendix B

Evaluation of the Detectable Changes in Chemical Contaminants in Flounder and Lobster Tissue

EVALUATION OF THE DETECTABLE CHANGES IN CHEMICAL CONTAMINANTS IN FLOUNDER AND LOBSTER TISSUE

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INTRODUCTION

One of the key questions in the MWRA Harbor and Outfall Monitoring project is detection of change in Boston Harbor and Massachusetts Bay as the various cleanup steps are implemented. Determining the impact of the various cleanup steps involves two functions: (1) detecting and quantifying change and (2) determining the levels of change (i.e., meaningful change) that have environmental (ecological) or human health significance. These two functions are interlinked in the monitoring program because the ability to establish an effective measurement program requires (1) a determination of what changes are significant and (2) establishing an appropriate sampling and analysis plan for detecting whether that change occurs. D. Parkhurst (1985), in a commentary on interpreting failure to reject a null hypothesis, states "First, as is recognized in almost all statistical texts, there is little or no relationship between biological importance and the statistical significance of a given result." He further states that "deciding on the degree of biological importance requires subjective scientific judgement, which some workers would rather not face." Thus, the key to determining the impact of the cleanup steps and the relocation of the Deer Island sewage outfall into Massachusetts Bay, and ultimately the final monitoring plan design, depends on agreement on the type and amount of change that will have substantive impact on the ecology (health) of the system. Once the level of acceptable (or unacceptable) change is determined, appropriate hypotheses can be prepared and monitoring schemes established to address these hypotheses.

Because the data used to test hypotheses result from field collections which have inherent variability and are also prone to measurement errors, the possibility of error in making an incorrect final conclusion regarding whether or not to reject an hypothesis has nonzero probability. There are two potential errors that can be made, Type I and Type II. The probability of committing a Type I error is denoted as α and is the probability of rejecting the hypothesis when, in reality, it is true (i.e., mistakenly rejecting a true hypothesis). The probability of committing a Type II error is denoted as β

and is the probability of failing to reject the hypothesis when in reality it is false. The probability of not committing a Type II error, that is the probability of correctly detecting an effect, is 1-β and is referred to as the power of the test. Note 1-α is often referred to as *confidence level* and is frequently set at 0.95 ($\alpha = 0.05$).

In many cases, the effects to be detected in a monitoring program can be related to characteristics like the changes in the overall mean contaminant concentration in a specified medium, such as sediments or organisms. Often, an adequate statistical model to describe the measurements includes only two terms, the mean contaminant concentration and an error term to account for differences in the measurements from the overall mean. However, in other situations the statistical model must include additional terms to account for other factors affecting the measurements, for example seasonal trends which systematically change the expected concentrations to levels above and below the overall mean concentration depending on the time of year. In either case, the power analysis must account for the uncertainty in the measurements as quantified by the variance of the errors, that is, the variance of the deviations from the overall mean and trend.

In hypothesis testing, scientists have traditionally evaluated null hypotheses based on Type I error testing. However, many scientists have come to realize that both Type I and Type II errors must be considered when evaluating environmental data if sound management decisions are to be achieved. Parkhurst states it this way: "*...failure to disprove a null hypothesis does not prove that null hypothesis.* Put another way, one should not think of accepting the null hypothesis, but rather of failure to reject it." As environmental management and decision making has progressed, the use of statistical power has become more important in the design of monitoring programs (EPA, 1992).

Determination of statistical power (1-β) involves the parameters α, the sample size (n), the standard error of the estimate (i.e., natural variability), and effect size (i.e., the magnitude of the true effect). Power analysis also requires that the estimate of error variation is stable and independent of the mean of the variable. (The standard deviation of a sample set can be used if it is shown to be independent of the mean). The larger the effect size, the more likely a given experiment with given sample size will correctly reject the null hypothesis at the stated α level (Peterman, 1989).

Another concept discussed by Peterman (1989) is reverse power analysis. This concept is relevant to environmental management, particularly if the maximum probability of a Type II error that can be (will be) tolerated can be stated. Peterman (1989) suggests that a power of 0.80 is the minimum power that should be achieved for credible results. Under this concept, designating the maximum power allowable permits calculation of the effect size that must exist to give the desired power. For the reverse power analysis, four entities are known and are used to calculate the effect size that must exist for the designated (tolerable) power. These are α, the sample size, standard error of the estimate, and desired statistical power. The calculation provides managers with an estimate of change that must occur (be measurable) under the experimental monitoring design.

In April 1992, lobster and flounder specimens were collected from several monitoring stations within Massachusetts and Cape Cod Bays. Lobsters were collected from stations located in Cape Cod Bay (CCB), Deer Island (DI) Flats in Boston Harbor, and the future Outfall Site (FOS) in Massachusetts Bay. Flounder were collected from these stations plus stations in Broad Sound (BS) and Nantasket Beach (NBB). The station locations are shown in Figure 3-5 in the main body of this report. Edible tissues (tail of the lobster and fillets of the flounder) from three individual animals of both species were each analyzed for selected contaminants (Table B-1). In addition, the hepatopancreas of each of the lobsters and the liver of the flounder were also analyzed for selected contaminants. Total PCB concentrations were calculated using the sum of the individual PCB congeners. Total PAH was determined from the sum of the individual PAH compounds. The concentrations of selected pesticides and eight trace metals were also measured. The edible tissues of these organisms were analyzed for a reduced set of chemical contaminants (chlordane, DDT, dieldrin, total PCB and mercury). In addition to analysis of the individual flounder, composite flounder tissue and liver samples (7 animals per composite) were analyzed for a subset of the analytes. These data were used to evaluate the ability to detect changes in chemical contaminant concentrations in flounder and lobster tissue from Massachusetts and Cape Cod Bays under this sampling and analytical design.

METHODS

The lobster and flounder data collected in Massachusetts Bay in 1992 were used to perform a reverse power analysis. The goal of this analysis was to evaluate the changes (effect size) that can be measured under the sampling and analytical design used for these organisms in 1992 ($n = 3$). All sample stations were kept separate. Because the primary interest in the program is detecting increases in the contaminant concentration, the analysis was run at $\alpha = 0.05$ (one tailed test = 1.96) and at β levels of 0.05 and 0.2. The analysis was performed on untransformed data using the standard deviation, thus assuming the variance was independent of the mean.

RESULTS

The results of this evaluation are tabulated by compound class, compound, or element for each of the four tissue types analyzed (Tables B-2 through B-5). The effect level that could be detected using the three replicates ranged from 1% for Zn in flounder liver from Natasket Beach to 1,029% for Cr in flounder liver from the same station. More generally, the effect level that this sample design could detect at a power of 0.80 is two to three times the mean of the three replicates (i.e., the percent detectable change generally fell between 100 and 200 percent). Thus, a doubling or tripling of the contaminant concentrations in the tissues of these species must occur before the change can be detected. Detectable change was better than about 50% for several compounds and tissues. These stations and compounds are shaded in Tables B2 - B5 for each of the tissue types. For the flounder 14 of 25 meat analysis (56%) and 32 of 65 liver analysis (49%) had detectable change that was better than 100%. For the lobster, the number of analysis that resulted in detectable change that was better than 100% was 87% for the meat (13 of 15) and 44% for the hepatopancreas (17 of 39).

The results of this analyses generally show that the number of sample replicates that must be collected and analyzed must be greater than three to reliably and consistently detect the 50 to 100% change presently included in the Outfall Monitoring Plan (MWRA, 1991). However, before defining the number of samples that should be taken to reach this goal, the acceptable level of change (meaningful change) and the power level to be used for these analyses should be defined. Once these are defined, additional statistical analysis can be conducted to determine the number of samples required to detect the required level of change.

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Table B-1. Analytes Included in Tissue Chemistry Analyses.

Trace Metals[a]
Ag Silver
Cd Cadmium
Cr Chromium
Cu Copper
Hg Mercury[b]
Ni Nickel
Pb Lead
Zn Zinc

Polychlorinated biphenyls (PCBs)[c]
2,4,-Cl_2(8)
2,2',5-Cl_3(18)
2,4,4'-Cl_3(18)
2,2',3,5'-Cl_4(44)
2,2',5,5'-Cl_4(52)
2,3',4,4'-Cl_4(66)
3,3',4,4'-Cl_4(77)
2,2'4,5,5'-Cl_5(101)
2,3,3',4,4'-Cl_5(105)
2,3',4,4'5-Cl_5(118)
3,3',4,4',5-Cl_5(126)
2,2',3,3,4,4'-Cl_6(128)'
2,2',3,4,4',5-Cl_6(138)
2,2'4,4',5,5'-Cl_6(153)
2,2'3,3,4,4',5-Cl_7(170)
2,2',3,4,4',5,5'-Cl_7(180)
2,2',3,4,5,5',6-Cl_7(187)
2,2',3,3',4,4',5,6-Cl_8(195)
2,2',3,3'4,4',5,5',6-Cl_9(206)
Decachlorobiphenyl-Cl_{10}(209)

Polynuclear Aromatic Hydrocarbons (PAHs)[a]
naphthalene
C_1-naphthalenes
C_2-naphthalenes
C_3-naphthalenes
acenaphthylene
acenaphthene
C_1-fluorenes
C_2-fluorenes
C_3-fluorenes
phenanthrene
C_1-Phenanthrenes/anthracene
C_2-Phenanthrenes/anthracene

Polynuclear Aromatic Hydrocarbons (PAHs) (continued)
C_3-Phenanthrenes/anthracene
C_4-Phenanthrenes/anthracene
dibenzothiophene
C_1-dibenzothiophenes
C_2-dibenzothiophenes
C_3-dibenzothiophenes
fluoranthene
pyrene
C_1-fluoranthenes/pyrene
benzo[*a*]anthracene
chrysene
C_1-chrysene
C_2-chrysene
C_3-chrysene
C_4-chrysene
benzo[*b*]fluoranthene
benzo[*k*]fluoranthene
benzo[*a*]pyrene
dibenzo[*a,h*]perylene
indeno[1,2,3-*c,d*]pyrene
Perylene
Biphenyl
Benzo[*e*]pyrene
Dibenzofuran

Pesticides[c]
Hexachlorobenzene
Lindane
Heptachlor
Aldrin
Heptachlorepoxide
alpha-chlordane
trans-Nonachlor
Dieldrin
Mirex
2,4'-DDD
4,4'-DDD
2,4'-DDE
4,4'-DDE
2,4'-DDT
4,4'-DDT

Lipids[c]

[a] Flounder liver; lobster hepatopancreas
[b] Also analyzed in flounder and lobster edible tissue
[c] Flounder edible tissue and liver; lobster edible tissue and hepatopancreas

Table B-2. Detectability of changes in chemical concentrations in flounder meat.

		Composite	Average	STD	P95 detectable Change		P80 Detectable Change	
STATION	COMPOUND	ng/g dry	ng/g dry	ng/g dry	ng/g	%	ng/g	%
BS	Chlordane	9.3	36.46	23.08	67.84	186.07	52.77	144.72
CCB	Chlordane	3.1	2.53	1.32	3.88	153.36	3.02	119.28
DI	Chlordane	13.2	42.91	39.93	117.37	273.53	91.29	212.74
FOS	Chlordane	5.9	9.33	1.07	3.15	33.71	2.45	26.22
NB	Chlordane	4.2	14.9	9.02	26.51	177.94	20.62	138.40
BS	DDT	33.7	58.4	24.27	71.34	122.16	55.49	95.01
CCB	DDT	12.7	10.99	4.54	13.34	121.43	10.38	94.44
DI	DDT	38.5	57.01	13.5	39.68	69.61	30.86	54.14
FOS	DDT	20.4	29.83	3.38	9.94	33.31	7.73	25.90
NB	DDT	12.1	37.03	17.74	52.14	140.82	40.56	109.53
BS	Dieldrin	1.61	2.17	0.93	2.73	125.97	2.13	97.98
CCB	Dieldrin	1.03	1.05	0.59	1.73	165.17	1.35	128.46
DI	Dieldrin	2.23	2.81	1.32	3.88	138.08	3.02	107.39
FOS	Dieldrin	1.22	0.97	0.23	0.68	69.70	0.53	54.21
NB	Dieldrin	1.19	3.49	2.04	6.00	171.82	4.66	133.63
BS	Hg	565	566	177	0.52	91.92	0.40	71.49
CCB	Hg	118	44	26	0.08	173.69	0.06	135.09
DI	Hg	170	534	126	0.37	69.36	0.29	53.94
FOS	Hg	384	681	434	1.28	187.33	0.99	145.70
NB	Hg	557	496	83	0.24	49.19	0.19	38.26
BS	PCB	215.0	495.1	177.5	521.74	105.30	405.80	81.90
CCB	PCB	70.1	45.5	18.89	55.53	122.11	43.19	94.98
DI	PCB	266.4	522.54	310.73	913.36	174.79	710.39	135.95
FOS	PCB	141.1	246.99	75.88	223.04	90.30	173.48	70.24
NB	PCB	90.8	346.68	108.89	320.07	92.32	248.94	71.81

Table B-3. Detectability of changes in chemical concentrations in flounder liver.

		Composite	Average	STD	P95 Detectable Change		P80 Detectable Change	
STATION	COMPOUND	ng/g dry	ng/g dry	ng/g dry	ng/g dry	%	ng/g dry	%
BS	Ag		2,253	1,266	9,229	250	7,178	195
CCB	Ag		4,213	2,799	6,131	146	4,769	113
DI	Ag		3,245	3,285	9,655	298	7,509	231
FOS	Ag		4,270	1,883	5,535	130	4,305	101
NB	Ag		3,134	1,656	4,869	155	3,787	121
BS	Cd		2,253	1,266	3,721	165	2,894	128
CCB	Cd		220	36	105	48	81	37
DI	Cd		1,860	906	2,663	143	2,071	111
FOS	Cd		1,507	972	2,857	190	2,222	148
NB	Cd		1,260	539	1,585	126	1,232	98
BS	Chlordane	84	324	200	588	181	457	141
CCB	Chlordane	13	16	3	10	63	8	49
DI	Chlordane	201	210	89	422	201	329	156
FOS	Chlordane	64	119	36	105	89	82	69
NB	Chlordane	32	108	39	115	106	89	83
BS	Cr		170	104	306	180	238	140
CCB	Cr		37	52	152	416	119	323
DI	Cr		177	124	363	206	283	160
FOS	Cr		47	37	108	232	84	180
NB	Cr		<0.1		0	1,323	0	1,029
BS	Cu		32,167	15,944	46,866	146	36,451	113
CCB	Cu		49,833	26,135	76,820	154	59,749	120
DI	Cu		83,433	90,294	265,408	318	206,429	247
FOS	Cu		90,433	39,870	117,193	130	91,150	101
NB	Cu		62,100	46,604	136,990	221	106,547	172
BS	DDT	279	426	155	456	107	355	83
CCB	DDT	64	49	7	21	43	16	33
DI	DDT	182	245	89	260	106	202	83
FOS	DDT	142	244	50	147	60	115	47

Table B-3. Detectability of changes in chemical concentrations in flounder liver.

		Composite	Average	STD	P95 Detectable Change		P80 Detectable Change	
STATION	COMPOUND	ng/g dry	ng/g dry	ng/g dry	ng/g dry	%	ng/g dry	%
NB	DDT	90	244	39	114	47	89	36
BS	Dieldrin	12	20	6	18	92	14	71
CCB	Dieldrin	11	6	2	5	83	4	65
DI	Dieldrin	24	16	3	8	48	6	37
FOS	Dieldrin	10	13	4	11	85	9	66
NB	Dieldrin	7	18	7	21	116	16	91
BS	Hg	1,211	1,002	80	234	23	182	18
CCB	Hg	659	152	60	176	116	137	90
DI	Hg	622	1,126	490	1,440	128	1,120	99
FOS	Hg	454	595	200	588	99	458	77
NB	Hg	832	704	292	859	122	668	95
BS	Ni		1,484	1,080	3,174	214	2,469	166
CCB	Ni		316	110	323	102	252	80
DI	Ni		1,035	677	1,991	192	1,548	150
FOS	Ni		722	566	1,663	230	1,293	179
NB	Ni		459	34	99	22	77	17
BS	PAH	2,185	14,430	9,191	27,015	187	21,012	146
CCB	PAH	2,619	9,752	3,034	8,919	91	6,937	71
DI	PAH	1,320	16,747	13,503	39,691	237	30,871	184
FOS	PAH	1,851	4,229	3,641	10,703	253	8,324	197
NB	PAH	2,072	15,116	8,081	23,754	157	18,475	122
BS	Pb		6,453	1,266	18,886	291	14,689	227
CCB	Pb		4,150	2,799	8,226	198	6,398	154
DI	Pb		3,140	2,822	8,294	264	6,451	205
FOS	Pb		2,950	1,605	4,718	160	3,670	124
NB	Pb		3,967	3,707	10,897	275	8,475	214
BS	PCB	2,174	4,314	2,003	5,887	136	4,579	106
CCB	PCB	565	319	74	219	68	170	53
DI	PCB	2,636	2,604	1,123	3,302	127	2,568	99

Table B-3. Detectability of changes in chemical concentrations in flounder liver.

		Composite	Average	STD	P95 Detectable Change		P80 Detectable Change	
STATION	COMPOUND	ng/g dry	ng/g dry	ng/g dry	ng/g dry	%	ng/g dry	%
FOS	PCB	1,429	2,777	604	1,774	64	1,380	50
NB	PCB	666	2,537	746	2,193	86	1,705	67
BS	Zn		146,667	1,080	40,824	28	31,752	22
CCB	Zn		156,333	110	33,969	22	26,421	17
DI	Zn		117,333	8,994	26,436	23	20,562	18
FOS	Zn		166,000	6,976	20,506	12	15,949	10
NB	Zn		119,333	471	1,386	1	1,078	1

Table B-4. Detectability of changes in chemical concentrations in lobster meat.							
		Average	STD	P95 Detectable change		P80 Detectable change	
SAMPLE	COMPOUND	ng/g dry[1]	ng/g dry[1]	ng/g dry[1]	%	ng/g dry[1]	%
DI	Chlordane	3.73	0.56	1.65	44.13	1.28	34.32
CCB	Chlordane	1.57	0.07	0.21	13.11	0.16	10.19
FOS	Chlordane	1.49	0.22	0.65	43.40	0.50	33.76
DI	DDT	15.72	1.95	5.73	36.46	4.46	28.36
CCB	DDT	19.26	10.10	29.69	154.14	23.09	119.89
FOS	DDT	9.96	2.01	5.91	59.32	4.60	46.14
DI	Dieldrin	5.51	0.05	1.38	25.07	1.07	19.50
CCB	Dieldrin	3.52	0.63	1.85	52.61	1.44	40.92
FOS	Dieldrin	3.95	0.29	0.85	21.58	0.66	16.78
CCB	Hg (μg/g)	0.92	0.39	1.14	123.51	0.88	96.07
FOS	Hg (μg/g)	0.85	0.23	0.68	79.51	0.53	61.84
DI	Hg (μg/g)	1.23	0.43	1.26	102.93	0.98	80.05
CCB	PCB	87.71	46.48	136.62	155.77	106.26	121.15
DI	PCB	100.10	12.38	36.39	36.36	28.30	28.28
FOS	PCB	60.99	16.76	49.26	80.77	38.32	62.82

[1] Except as noted in the table.

Table B-5. Detectability of changes in chemical concentrations in lobster hepatopancreas.

		Average	STD	P95 Detectable Change		P80 Detectable Change	
STATION	COMPOUND	ng/g dry[1]	ng/g dry[1]	ng/g dry[1]	%	ng/g dry[1]	%
CCB	Ag μg/g	3.53	1.53	4.50	127.40	3.50	99.09
DI	Ag μg/g	5.07	3.45	10.14	200.02	7.89	155.57
FOS	Ag μg/g	3.52	0.26	0.82	23.38	0.64	18.19
CCB	Cd μg/g	27.1	15.9	46.65	172.01	36.28	133.78
DI	Cd μg/g	34.8	37.6	110.52	317.41	85.96	246.87
FOS	Cd μg/g	13.0	3.7	10.82	83.40	8.41	64.87
CCB	Chlordane	18.6	12.6	37.10	199.12	28.85	154.87
DI	Chlordane	196.7	150.9	443.64	225.54	345.06	175.42
FOS	Chlordane	51.5	33.1	97.24	188.77	75.63	146.82
CCB	Cr μg/g	2.09	0.38	1.12	53.44	0.87	41.57
DI	Cr μg/g	2.91	0.53	1.56	53.54	1.21	41.64
FOS	Cr μg/g	3.36	1.50	4.41	131.22	3.43	102.06
CCB	Cu μg/g	1014.4	702.1	2063.66	203.44	1605.07	158.23
DI	Cu μg/g	232.7	293.6	862.97	370.85	671.20	288.44
FOS	Cu μg/g	440.8	527.1	1549.47	351.54	1205.14	273.42
CCB	DDT	230.1	66.9	196.53	85.40	152.86	66.42
DI	DDT	555.6	226.3	665.10	119.71	517.30	93.11
FOS	DDT	352.8	64.5	189.44	53.75	147.35	41.81
CCB	Dieldrin	13.4	6.7	19.75	147.30	15.36	114.57
DI	Dieldrin	65.7	33.4	98.12	149.27	76.31	116.10
FOS	Dieldrin	27.0	14.1	41.45	153.45	32.24	119.35
CCB	Hg	423	207	0.61	143.84	0.47	111.88
DI	Hg	241	43	0.13	52.45	0.10	40.79
FOS	Hg	538	385	1.13	210.35	0.88	163.60
CCB	Ni μg/g	0.95	0.47	1.38	145.42	1.07	113.11
DI	Ni μg/g	0.80	0.59	1.74	218.88	1.35	170.24
FOS	Ni μg/g	1.60	1.40	4.10	256.12	3.19	199.20

Table B-5. Detectability of changes in chemical concentrations in lobster hepatopancreas.

		Average	STD	P95 Detectable Change		P80 Detectable Change	
CCB	PAH	4055	1034	3039.21	74.95	2363.83	58.29
DI	PAH	29708	6909	20308.80	68.36	15795.70	53.17
FOS	PAH	4060	385	1132.90	27.90	881.14	21.70
CCB	Pb μg/g	4.49	5.95	17.48	389.21	13.60	302.72
DI	Pb μg/g	0.38	0.10	0.28	74.46	0.22	57.92
FOS	Pb μg/g	0.28	0.14	0.41	150.71	0.32	117.22
CCB	PCB	1212	566	1664.46	137.31	1294.58	106.80
DI	PCB	3254	721	2118.33	65.11	1647.59	50.64
FOS	PCB	2070	515	1512.79	73.10	1176.61	56.85
CCB	Zn μg/g	100.6	45.0	132.16	131.33	102.79	102.14
DI	Zn μg/g	76.6	21.7	63.81	83.31	49.63	64.80
FOS	Zn μg/g	110.8	35.3	103.64	93.57	80.61	72.77

[1] Except as noted in the table.

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