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Transport and Fate of Toxic Contaminants Discharged by MWRA into Massachusetts Bay

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TRANSPORT AND FATE OF TOXIC CONTAMINANTS DISCHARGED BY MWRA INTO MASSACHUSETTS BAY

by

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ERRATA SHEET

This report refers in Section 3.2.4 to modelling predictions used by EPA in the Supplemental Environmental Impact Statement (SEIS). The SEIS models are extensions of modelling approaches in the Environmental Impact Report (EIR). Examples of model outputs in the text include results from both the SEIS and the EIR predictions. The report's critique of the model assumptions is equally applicable to both the SEIS and EIR model results.

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EXECUTIVE SUMMARY

The Massachusetts Water Resources Authority (MWRA) is implementing a pollution abatement program in Boston Harbor that will greatly improve sewage treatment, that will move the effluent discharge from within the confines of Boston Harbor to an offshore outfall, and that has already eliminated the discharge of sewage sludge into Boston Harbor. These changes are expected to significantly improve water and sediment quality within the Boston Harbor area *and* the regional environment of Massachusetts and Cape Cod Bays.

The future outfall discharge will bring treated effluent containing toxic contaminants directly into Massachusetts Bay. Previous model projections suggest that this will cause little harm to the environment of Massachusetts Bay, but that very near the outfall diffuser there may be violations of certain water quality criteria and degradation of the benthic environment. However, because the major source of most toxic contaminants to Boston Harbor has been the MWRA outfall, there is concern that moving the outfall into Massachusetts Bay will simply transport Boston Harbor's problems out into Massachusetts Bay. This concern, combined with the uncertainty in predicting the fate and effects of toxic contaminants, provided the impetus to develop this report on updating our understanding of contamination-related processes in Boston Harbor and the Bays and to help answer the question "Are we transporting Boston Harbor's problems out into Massachusetts Bay?"

Toxic contaminants have been discharged from a variety of sources into Boston Harbor and Massachusetts Bay for over 350 years. Within Boston Harbor, the effluent from MWRA's primary treatment plants at Nut Island and Deer Island is the dominant source for most contaminants, with loading contributions ranging from about 1/3 of the total lead load to about 3/4 of the total polycyclic aromatic hydrocarbon (PAH) load. The MWRA sludge discharge (which ceased in December 1991) was the second largest source of most toxic contaminants.

Loading estimates for Massachusetts Bay indicate that Boston Harbor and the MWRA discharges (including sludge) have already been a significant source of contaminants to the Bay; results from a survey in October 1991 have shown that sewage contaminants are already widely distributed throughout the sediments of the Bays. However, it is also clear that there are other contaminant sources equal in magnitude to the MWRA discharges. For example, the dominant source of polychlorinated biphenyls (PCBs) and lead to Massachusetts Bay is deposition from the atmosphere. Discharges with permits under the National Pollutant Discharge Elimination System (NPDES) are the primary source of copper and contribute about half of the total PAH load, with the atmosphere contributing the other half. The MWRA outfall generally makes the largest contribution to the total NPDES load, but the contributions from other discharges are also significant and in some cases (cadmium and lead) are even higher than the MWRA load.

Within the last 20 years, total toxic loadings from the MWRA have leveled off and have even begun to decrease; loading of toxic metals has decreased about a factor of 5 over the last 10 years, and recent analysis of the MWRA effluent for organic contaminants (PCBs, pesticides, and PAHs) using state-of-the-art analytical methods indicates that most of these toxic compounds are 10 to 100 times lower than previously reported. There should be even more significant decreases in the loading of toxic contaminants to Boston Harbor and Massachusetts Bay with the continuing toxics reduction programs, cessation of dumping raw sewage sludge into the Harbor, and the great improvements in the treatment of sewage with the new primary and secondary sewage treatment facility at Deer Island. For example, the present sewage treatment removes less than 20% of the toxic contaminants from the effluent. When the primary treatment plant is operational in 1995, about 40% of the toxic material will be removed; secondary treatment will remove up to 85% of the toxic material.

The present MWRA outfalls discharge at the mouth of the Harbor. During the incoming tide, much of the contaminant load is brought into the Harbor where the quality of the water is degraded because there is very little dilution. Contaminated particles can be deposited in the calm areas of the Harbor (Deer Island Flats, for example). On the outgoing tide much of that same water, plus the effluent discharge, are moved directly into Massachusetts Bay. Dissolved contaminants and contaminated particles that have not settled (or have been resuspended) are transported out into western Massachusetts Bay. Mass balance calculations indicate that between 50% and 95% of the toxic contaminants entering Boston Harbor can be exported from the Harbor into Massachusetts Bay.

Estimates of contaminant residence times in Boston Harbor (that is, how long a contaminant resides in the water column of Boston Harbor) range from about 2 to 10 days. Contaminants that are primarily in the dissolved form appear to be most easily exported from the Harbor. Contaminants that are strongly associated with particles (lead and most organic contaminants) have greater retention in the Harbor. These calculations indicate that even contaminants entirely bound to particles would still have a significant export from the Harbor. This export could be through transport of suspended particles in the water and through dredging of particles that have settled to the bottom of Boston Harbor (and then disposed of at the Massachusetts Bay Disposal Site). The consistent high export from the Harbor, across a range of contaminants, supports the hypothesis that contaminants discharged into Boston Harbor are rapidly flushed out into Massachusetts Bay.

If most of the toxic contaminants that enter Boston Harbor are exported to Massachusetts Bay, then (1) there should be evidence of a decreasing gradient of contaminant concentrations with distance from Boston Harbor, and (2) there should be a fairly widespread distribution of toxic contaminants in Massachusetts Bay. Convincing evidence of both the gradients and the widespread contaminant distribution is recorded in the sediments, water column, and body burdens and physiological condition of fish and shellfish from Massachusetts and Cape Cod Bays. This evidence is consistent with the results of a companion study of the transport and fate of dissolved nutrients which concluded there was a similar export of nutrients from Boston Harbor.

The data discussed above and presented in this report provide compelling evidence that most of the toxic contaminants entering Boston Harbor find their way into Massachusetts Bay. Thus, one would expect very little change, regionally, by moving the outfall offshore. Clearly, there will be two local areas affected by this move. Boston Harbor will become cleaner, and the area immediately surrounding the new outfall will become more contaminated. Just how contaminated the new outfall area will become was the subject of numerous studies and models conducted by MWRA and EPA. These studies predicted very little impact — a few water quality criteria violations just outside the mixing zone and a degraded benthic environment in the vicinity of the outfall. Even so, an evaluation of the assumptions used in these models has shown that the predictions were very conservative (protective). Alternative assumptions (e.g., contaminant removal mechanisms) are offered in this report and various possible effects of altering the assumptions are presented. In addition to these theoretical considerations, there is empirical evidence that the EPA modeling yielded predictions of contaminant concentrations that are too high (conservative). If contaminant concentrations predicted at the present outfall are compared to actual observations, metals in the water column appear to be overestimated by up to a factor of 40 and organic contaminants by up to a factor of 90. Contaminant accumulation predicted in sediments near the future outfall may also be significantly overestimated. In both cases, this report offers reasons why these predicted contaminant concentrations tend to be too high and how existing models could be improved. Implications for developing a focused and efficient monitoring program for the future MWRA outfall are also given.

ACKNOWLEDGMENTS

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CONTENTS

Executive Summary ii
Acknowledgments v
List of Tables vii
List of Figures viii

1.0 INTRODUCTION

2.0 TRANSPORT AND FATE OF TOXIC CONTAMINANTS DISCHARGED BY MWRA INTO MASSACHUSETTS BAY
2.1 Overview of the Transport and Fate of Contaminants 2-1
2.1.1 Physical Transport Processes 2-2
2.1.2 Fate of Organic Contaminants 2-4
2.1.3 Fate of Trace Metals 2-5
2.1.4 Biological Factors 2-5
2.2 Estimates of Contaminant Loading 2-6
2.3 Present Role of Boston Harbor: Export of Loads to Massachusetts Bay 2-13
2.3.1 Water Column Concentrations 2-14
2.3.2 Retention of Chemicals within Boston Harbor Sediments 2-16
2.3.3 Export via Dredging and Disposal to the Massachusetts Bay Disposal Site 2-21
2.3.4 Evidence from the Neighboring Recipient System: Gradients into Western Massachusetts Bay 2-22
2.4 Present Situation Relative to Future Outfall 2-24
2.4.1 Analogous Features Influencing Transport and Fate 2-24
2.4.2 Additional Factors to Consider in Making Predictions for Change at the Proposed Outfall Site 2-25
2.4.2.1 Seasonal Feature of Particle Accumulation 2-25
2.4.2.2 Enhanced Role of Pelagic Removal Processes (Including Zooplankton) 2-25
2.4.2.3 Effects of Stratification 2-26
2.4.2.4 Spatial Scale of Concern 2-26

3.0 POTENTIAL IMPACT RESULTING FROM SEWAGE DISCHARGE AT THE NEW OFFSHORE OUTFALL
3.1 Water Quality 3-1
3.2 Sediment Quality 3-15
3.2.1 Heterogeneity 3-16
3.2.2 Chemical Removal Rates in Seawater 3-22
3.2.3 Post-Depositional Redistribution of Contaminants 3-23
3.2.4 SEIS Modeling and Associated Uncertainties 3-23
3.3 Expected Changes Resulting from Moving the Outfall Offshore 3-30

CONTENTS (continued)

LIST OF FIGURES

Figure

1-1 Massachusetts and Cape Cod Bays 1-2

2-1 Processes that influence the transport and fate of toxic contaminants in the marine environment 2-3

2-2 Estimates of contaminant loading to Massachusetts Bay 2-8

2-3 Metal loading from the MWRA effluents over the last decade 2-10

2-4 Contaminant concentrations in the MWRA effluent that were estimated in the SEIS for primary treatment (EPA, 1988) compared to those measured by Shea (1992) 2-11

2-5 Residence times for several contaminants in Boston Harbor water 2-15

2-6 Relationship between dissolved metals in the water at Deer Island and the total metal load exported from Boston Harbor 2-20

2-7 *Clostridium perfringens*, a tracer of sewage particles, in Massachusetts and Cape Cod Bays sediments 2-23

3-1 Contaminant concentrations in the water predicted in the SEIS at President Roads (PR) for primary (-1) and secondary (-2) treatment are compared to observed (OBS) concentrations. Concentrations in northern Massachusetts Bay (MB), in continental shelf water (REF), and for the most stringent water quality criteria (WQC) are also given.
- (a) Aldrin 3-5
- (b) Arsenic 3-6
- (c) Copper 3-7
- (d) DDT 3-8
- (e) Dieldrin 3-9
- (f) Fluorene 3-10
- (g) Heptachlor 3-11
- (h) Mercury 3-12
- (i) PCB 3-13

3-2 Contour plot of contaminant concentrations near the new MWRA offshore outfall
- (a) Arsenic (μg/g, dry weight) 3-17
- (b) Cadmium (μg/g, dry weight) 3-18
- (c) Copper (μg/g, dry weight) 3-19
- (d) Polycyclic aromatic hydrocarbons (μg/g, dry weight) 3-20
- (e) Total organic carbon (%) 3-21

LIST OF FIGURES (continued)

Figure

3-3 Sedimentation rates (g/m^2/day) predicted in the SEIS: Site 5, primary treatment, stratified conditions 3-25

3-4 Contaminants in the effluent discharge can be associated with three different fractions (dissolved, colloidal, and particulate), all of which can have very different rates of transport away from the diffuser and to the bottom sediments 3-26

3-5 Simple model prediction for concentration of an effluent constituent in bottom sediments near the future outfall is shown (top) along with the hypothetical effects (dashed lines) of modifying the assumptions used in the SEIS and STFP models (solid lines) 3-28

CONTENTS (continued)

4.0 RECOMMENDATIONS FOR IMPROVEMENT OF FURTHER MODELING EFFORTS
4.1 Contaminant Loading 4-1
4.1.1 Effluent Chemical Quality and Flow Rate 4-1
4.1.2 Dissolved vs. Particulate Fractions in the Effluent 4-1
4.2 Water Column Processes 4-3
4.2.1 Partitioning between Dissolved and Particulate Fractions of the Effluent as It Initially Mixes with Seawater 4-3
4.2.2 Rates of Contaminant Removal from the Water Column by Particle Transport and Pelagic-Benthic Interactions 4-3
4.2.3 Relationship between Particle Sizes and Contaminant Burdens, with Respect to Settling Rates 4-4
4.3 Deposition and Related Processes 4-4
4.3.1 Variability in Bioturbation Depths and Mixing Rates as a Function of Depth within the Sediment 4-4
4.3.2 Spatial and Temporal Variability in Rates of Background Sedimentation 4-4
4.3.3 Post-Depositional Particle Transport Processes and Secondary Alteration of Contaminant Burdens on Transportable Solids 4-5
4.3.4 Significance of Biofiltration and Biodeposition in Modifying Chemical Fate and Transport 4-5
4.4 Ongoing and Future Work that Addresses Critical Information Needs 4-6

5.0 REFERENCES

LIST OF TABLES

Table
2-1 Estimated secondary treatment contaminant loadings (μg/s) from several outfalls 2-9

2-2 Retention in accumulated bottom sediments in Boston Harbor 2-18

3-1 Constituents that exceed EPA ambient water quality criteria 3-2

4-1 To improve model predictions of contaminant transport and fate, information is needed on critical points in three categories 4-2

1.0 INTRODUCTION

The Massachusetts Water Resources Authority (MWRA) is overseeing the construction of a new Deer Island Wastewater Treatment Plant. The new offshore outfall will be located in Massachusetts Bay approximately 15 km from the Deer Island Plant at a water depth of 32 m (Figure 1-1). Greatly improved sewage treatment, cessation of sewage-sludge discharge to Boston Harbor, and removal of the wastewater discharge from within the confines of Boston Harbor are expected to significantly improve water and sediment quality within the Boston Harbor area *and* the regional environment of Massachusetts and Cape Cod Bays (EPA, 1988). Operation of the new outfall is scheduled to begin in July 1995, initially with effluent from upgraded primary treatment; secondary treatment is scheduled to be phased in beginning October 1996.

Discharge from the future outfall will bring treated effluent containing contaminants (including those which may act as toxicants) directly into Massachusetts Bay. Model projections suggest that this will cause little harm to the environment of Massachusetts Bay, but that very near the outfall diffuser there may be violations of certain water quality criteria and degradation of the benthic environment (EPA, 1988). However, because the major source of most contaminants to Boston Harbor has been the MWRA outfall, there is concern that moving the outfall into Massachusetts Bay will simply transport Boston Harbor's problems out into Massachusetts Bay. There is also considerable public concern about the outfall's impact on shorelines and other important natural resources in open waters (e.g., Stellwagen Bank).

These concerns, combined with the uncertainty in predicting the fate and effects of contaminants, have led MWRA to make toxic contamination a prime issue addressed by the MWRA outfall monitoring program (MWRA, 1991). However, a wealth of information is already available on toxic contaminants in Boston Harbor and, to a lesser extent, Massachusetts and Cape Cod Bays. This information can be used to update understanding of contamination-related processes in the Bays and to help answer the question "Are we transporting Boston Harbor's problems out into Massachusetts Bay?" To answer this question, we need accurate estimates of past, present, and future loadings of toxic contaminants to Boston Harbor and Massachusetts Bay, an understanding of the transport and fate of contaminants in the two water bodies, and knowledge of how various biological organisms in these areas will respond if exposed to different types and levels of contaminants.

The primary objective of this report is to address whether the mitigation procedures being implemented by the MWRA to clean up Boston Harbor (toxic reduction program, sludge abatement, primary and secondary sewage treatment, and discharge of effluent through the future offshore outfall) are likely to cause any detectable changes to the loading or ambient concentrations of toxic contaminants in the Massachusetts and Cape Cod Bays region.

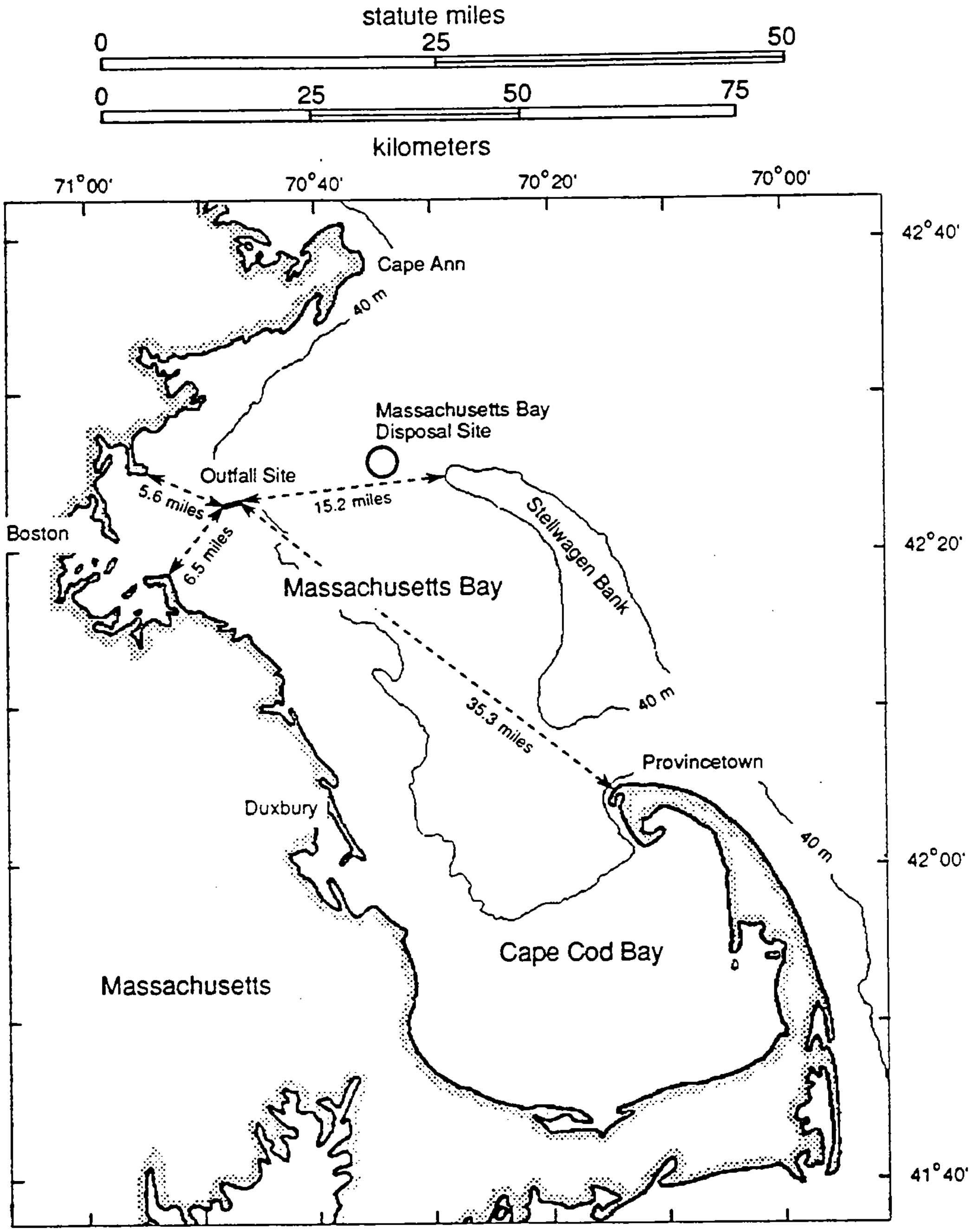


Figure 1-1. Massachusetts and Cape Cod Bays

In addressing this issue, several questions about the present situation in Boston Harbor and how things will change with the future outfall are answered:

> What has been the historical loading of toxic contaminants to Boston Harbor and Massachusetts Bay and how will improved treatment of the effluent affect the loading of toxic contaminants?
>
> What happens to the toxic contaminants entering Boston Harbor today and how much is exported to Massachusetts Bay?
>
> How does the loading of toxic contaminants from MWRA discharges compare to other sources?
>
> How will the effluent dilution and dispersion differ between the present and future outfalls?
>
> What are the similarities and differences between the environment at the present and future outfall sites?
>
> Were there any important processes left out of previous predictions of impact?

A secondary objective of this report is to provide a focus to the discussion of toxic contaminants discharged from the future outfall into Massachusetts Bay and to provide guidance for efficiently monitoring the amounts and effects of these toxic contaminants.

In this report, information from recent studies on toxic contamination in Boston Harbor, Massachusetts Bay, and elsewhere is used to examine the potential impact of the offshore outfall on Massachusetts Bay. Predictions of potential impact were made previously, during the outfall site-designation process conducted as part of the Secondary Treatment Facilities Plan, or STFP (MWRA, 1988), and the Supplementary Environmental Impact Statement, or SEIS (EPA, 1988). Generally the most conservative (most protective) assumptions were used in the STFP and SEIS models in the hope that predictions of impact would be an upper bound. We have reexamined some of these assumptions to identify what would be needed to make more accurate predictions of potential impact.

The remainder of the report is structured as follows:

Chapter 2	Processes that influence the transport and fate of toxic contaminants in Boston Harbor and Massachusetts Bay
Chapter 3	The potential impact resulting from movement of the MWRA outfall from Boston Harbor offshore to Massachusetts Bay
Chapter 4	Information needed to improve the predictions of impact on Massachusetts Bay

2.0 TRANSPORT AND FATE OF TOXIC CONTAMINANTS DISCHARGED BY MWRA INTO MASSACHUSETTS BAY

2.1 OVERVIEW OF THE TRANSPORT AND FATE OF CONTAMINANTS

The transport and fate of contaminants in the marine environment are controlled by physical processes that can transport both dissolved and particulate contaminants, geochemical processes that can alter the form and behavior of the contaminant, and biological processes that can alter the rate of both the physical and geochemical processes. Biological processes such as uptake, accumulation, metabolism, and elimination of contaminants regulate the effects on an organism in response to a given level and type of exposure. Some important contaminant transport and fate processes in the marine environment are summarized below.

The transport and fate of contaminants in the marine environment involve four media — atmosphere, water, sediment, and biota. The importance of each of these media (and the processes that take place within and among them) depends on the chemical properties of the contaminant and the form in which the contaminant enters the environment. The atmosphere can be a significant source of contaminants to the water through both dry and wet deposition (see Section 2.2), and can also act as a sink for more volatile contaminants. Water is the primary short-term medium for most contaminants that enter the marine environment through direct discharge (sewage effluent). However, most contaminants have a high affinity for particulate matter and, therefore, their long-term fate is controlled more by the transport and deposition of particulate matter to bottom sediments. Various organisms within the biological community interact, to some extent, with air, water, and sediment. Biology has a central role in transport of contaminants through removing them from the water column, converting them to particulate matter, and burying and mixing them into bottom sediments. Organisms can also metabolize certain compounds, hastening their decay in the environment.

For all but the more volatile contaminants, bottom sediments are the principal contaminant sink in the marine ecosystem. The fate of contaminants in the sediment depends on sediment resuspension and lateral transport, burial and bioturbation, biogeochemical transformation processes, and partitioning between the water and sediment. Thus, our ability to model the fate of contaminants in the marine environment depends on our estimates of rates of sedimentation, sediment resuspension, bioturbation, contaminant transformation, and sediment-water partitioning. Unfortunately, these rates often vary significantly with time and space and can be dominated by episodic events such as storms or changes in the redox status of the sediments.

2.1.1 Physical Transport Processes

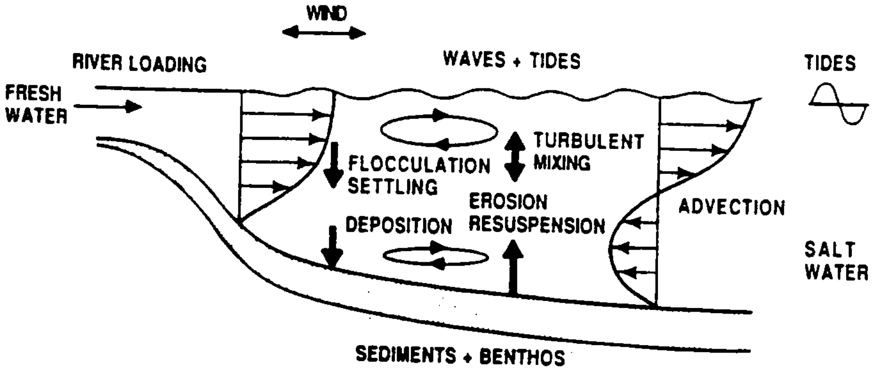
The most important physical processes related to the transport of effluent contaminants in the marine environment are advection, turbulent mixing, flocculation, settling, and resuspension (Figure 2-1a). Advection is governed by a three-dimensional flow field that is both time-dependent and turbulent, and in an estuarine environment is driven by tide, wind, and density gradients (Sheng, 1989). Turbulent mixing can be caused by wind, breaking of surface or internal waves, shearing motion at the bottom, unstable stratification, etc. In shallow estuarine environments, turbulence may exist over the entire water column, and certainly exists throughout the zone of initial dilution around a large wastewater outfall. Turbulence within the bottom boundary layer plays an important role in transporting particles via resuspension, whereas turbulent mixing throughout the water column can influence the flocculation and settling of particles. The turbulent mixing process is complex and random, and can be difficult to predict, particularly in regions of rough bottom (e.g., western Massachusetts Bay) or in the nearfield of an outfall plume (Sheng, 1989).

Wastewater effluent and marine sediments contain particles of various sizes, ranging from submicrometer colloidal particles (inorganic and macromolecular-organic) to large flocs or sands exceeding hundreds of micrometers. Flocs are formed when fine-grained particles are brought into frequent contact. Turbulent mixing, high ionic strength, and high organic content favor the formation of flocs, making sewage effluent discharge to marine systems an ideal environment for floc formation. Flocculation can lead to orders of magnitude increase in settling velocity.

Resuspension of surficial-layer bottom sediments can be induced by an increase in hydrodynamic stress or a weakening of sediment resistance and depends on both hydrodynamics and the physicochemical characteristics of the sediment. Likewise, deposition depends on both hydrodynamics and the properties of the suspended sediments. Our understanding of the deposition-resuspension-transport-redeposition process is very limited, but we do know that the biological community in the sediments may influence each step of the physical process.

Considering the bottom heterogeneity in western Massachusetts Bay, there are clearly net erosional and net depositional areas that should be considered in predictive modeling of material (contaminant) fate and transport. Some ongoing sediment trap studies (Bothner *et al.*, 1990) may provide field data to estimate the importance of sediment resuspension and lateral transport in affecting longer term transport of particles in Massachusetts Bay.

a. Physical processes



b. Biogeochemical processes

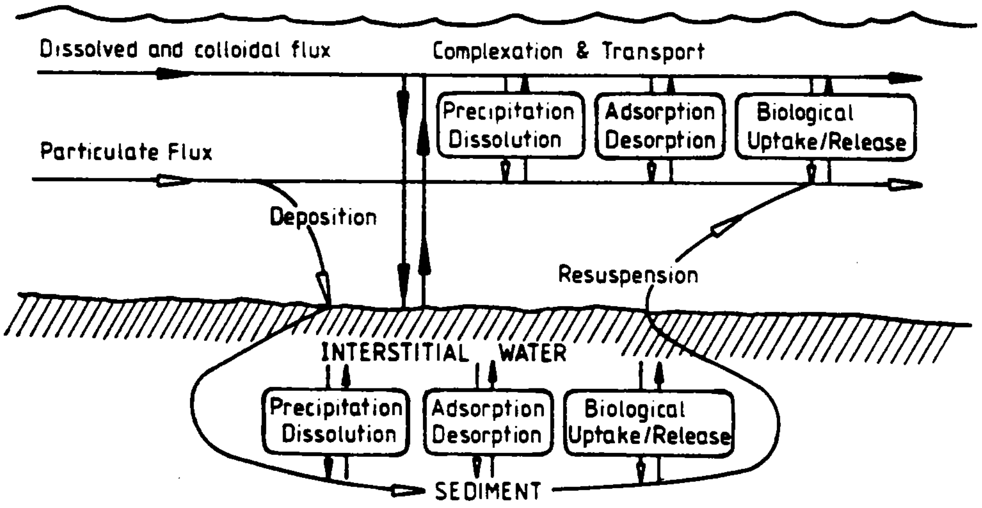


Figure 2-1. Processes that influence the transport and fate of toxic contaminants in the marine environment

2.1.2 Fate of Organic Contaminants

Physicochemical processes that can influence the fate of organic contaminants include sorption-desorption, volatilization, hydrolysis, photolysis, halogenation-dehalogenation, oxidation-reduction, and metabolic transformations. The importance of each of these processes is highly dependent on both the contaminant and its environment. The organic contaminants in the MWRA effluent that have been identified as potential problems (EPA, 1988) fall into five classes: (1) volatile hydrocarbons and halocarbons, (2) chlorinated pesticides, (3) polychlorinated biphenyls (PCBs), (4) phthalate esters, and (5) polynuclear aromatic hydrocarbons (PAHs).

In general, volatile organic contaminants are relatively soluble in water; do not sorb to particles to any great extent; are readily vaporized into the atmosphere (over weeks); and undergo hydrolysis, photolysis, and biodegradation at moderate rates. The use of chlorinated pesticides has been reduced considerably over the past 20 years and, not surprisingly, they are rarely detected in the MWRA influent or effluent (see Section 2.2). Their fate is governed primarily by sorption to particles and subsequent selective dechlorination by bacteria in sediments. Most are also moderately volatile, some are moderately soluble in water (e.g., toxaphene), and some undergo fairly rapid hydrolysis (e.g., heptachlor) and/or photolysis. Use of PCBs has also been long discontinued, however their persistence in the environment makes them ubiquitous. PCBs have a very low water solubility and corresponding high affinity for the organic carbon coating of particles. They are moderately volatile, but otherwise are only very slowly degraded by chemical or biological means. In anoxic sediments, selective microbial dechlorination has been shown to be surprisingly fast, but half-lives are still on the order of years. Phthalate esters have a high affinity for organic particles and are consumed by bacteria with relative ease, resulting in short half-lives in the water column. PAHs range from the relatively volatile, water-soluble, and biodegradable naphthalene (2-ring PAH) to the nonvolatile, recalcitrant, and highly particle-bound benzo(ghi)perylene (6-ring). Therefore, relative changes in the distribution of individual PAHs need to be considered when discussing the fate of these compounds.

Many other organic contaminants are enriched in sewage relative to other contaminant sources, but do not pose a threat to the environment. Fecal sterols (e.g., coprostanol) and linear alkyl benzenes (impurities found in detergents) are enriched in sewage and follow biogeochemical fate pathways similar to those of many of the toxic organic contaminants. Thus they can be used to "fingerprint" sewage and trace the transport and fate of the effluent in the environment (Eganhouse *et al.*, 1988). Measurements of these compounds in conjunction with those of toxic contaminants are particularly useful in discriminating sources of contaminants in sediments or suspended particulate matter. Unfortunately, little is known about the levels and chemical composition of organic sewage tracers (or their ratios to other contaminants) in the MWRA effluent or in sediments of Boston Harbor and Massachusetts Bay.

An additional and potentially important mechanism for transporting organic contaminants is complexation (or sorption) with organic colloids (Figure 2-1b). Colloidal-phase material can be an important route for transport of organic contaminants in the interstitial waters of sediments (Brownawell, 1986; Brownawell and Farrington, 1986; Wu, 1986) and colloidally-bound contaminants could be a less bioavailable form of the contaminant (McCarthy *et al.*, 1985). Sewage effluent also contains very high levels of organic colloids. Association of organic contaminants with colloidal material in the effluent could provide an important short-term (days) mechanism of transport within the water column. Colloids are too small to settle (though they can readily sorb to particles or flocculate) and thus would tend to follow the same pathway as dissolved contaminants after discharge from an outfall. Transport of colloidally bound contaminants from the sewage outfall deserves serious consideration as a major transport mechanism and as a mechanism of rendering organic contaminants nonbioavailable.

2.1.3 Fate of Trace Metals

The short-term fate of trace metals in the marine environment is more complicated than that of non-ionic, hydrophobic organic contaminants because most trace-metal transformation processes take place at much faster rates than organic-contaminant transformation processes. The primary reactions include acid-base, complexation, oxidation-reduction, sorption-desorption, and precipitation-dissolution. Perhaps the most important consideration in modeling the fate and effects of trace metals in the water column is the fractionation between particulate and dissolved forms (Figure 2-1b). The transport mechanisms and bioavailability of dissolved metals are much different from those of particulate metals, but, unlike hydrophobic organic contaminants, there are significant amounts of both dissolved and particulate metals in the effluent, water column, and interstitial water of sediments. In addition, different metals behave differently as they are discharged from an organic-rich, freshwater effluent into the marine environment. Over short time periods (days to weeks) some dissolved metals act conservatively (Cu, Cd, Zn), whereas others are scavenged by particles and removed from the water column (Pb, Hg). The presence of large amounts of dissolved, colloidal, and particulate organic matter in the effluent makes modeling the fate of metals very complicated, but first-order predictions can be based simply on dissolved and particulate forms of metals.

2.1.4 Biological Factors

Biological activity within the sediments can result in physical mixing (bioturbation) through the movement of macrobenthic organisms or through gas ebulation associated with microbial mineralization processes (e.g., methane formation). Organisms in the water column and/or benthos can hasten the rate of scavenging of dissolved and particulate contaminants. Feeding and the production of feces or pseudofeces can "package" material (including contaminants) into a form that sinks faster than dissolved or smaller particulate material and also offers fresh surfaces for contaminant adsorption. Many bacteria common to marine sediments can transform otherwise immobile forms of Hg and Pb to highly mobile and toxic forms through methylation. Numerous metabolic activities can enhance the degradation of organic contaminants and alter the form of trace metals. The products of these biologically mediated reactions can either increase or decrease a contaminant's bioavailability and toxicity.

Biological activity can influence contaminant fate through indirect processes as well. Photosynthetic and respiratory activities can result in changes to pH, oxidation-reduction (redox) potential, and other chemical characteristics of water and sediment. These changes can have dramatic effects on the speciation, mobility, and toxicity of contaminants and often have large temporal (and many times spatial) variability. For example, marine systems often have high rates of sulfate reduction (to sulfide) in the warmer months. The production of sulfide leads to the precipitation of some metals (e.g., Cu, Cd, Pb, Zn, Hg) and remobilization of others (e.g., Fe, Mn, Cr), although many shorter term processes are also known (e.g., dissolution of iron oxyhydroxides that release adsorbed copper to the water, which is then precipitated by sulfide, and later remobilized by complexation with polysulfides and organic sulfides). The formation of metal sulfides obviously decreases metal solubility and has been shown to decrease the toxicity of sediment-bound metals. Changes in redox chemistry also can be accompanied (and are often caused) by a shift in organism populations that can then affect the fate of contaminants. For example, a shift from aerobic to anaerobic bacteria in sediment can enhance the rate of dechlorination of PCB while retarding the rate of oxidation of PAH.

2.2 ESTIMATES OF CONTAMINANT LOADING

Toxic contaminants have been discharged from a variety of anthropogenic sources into Boston Harbor and Massachusetts Bay for more than 350 years. Although reliable measurements of the types and amounts of contaminants have only been made over about the last 20 years, there is evidence (e.g., from contaminant profiles in sediment cores) that the loading of contaminants increased over the last 200 years as a result of increasing population and industrialization.

Estimates of the present loading of contaminants to Boston Harbor (Menzie *et al.*, 1991) and Massachusetts Bay (Menzie-Cura, 1991) are available. The contaminant sources for which data were compiled by Menzie *et al.* (1991) for Boston Harbor include MWRA effluent and sludge, runoff [including discharge from combined sewer overflows (CSO)], rivers, air, groundwater, direct discharges with permits granted under the National Pollutant Discharge Elimination System (NPDES), and airport runoff. These estimated loads to Boston Harbor are very similar to those presented in the *State of Boston Harbor: 1990* (MWRA, 1990). Within Boston Harbor, the MWRA effluent is the dominant source for most contaminants, with loading contributions ranging from about 1/3 of the total lead load to about 3/4 of the total aromatic hydrocarbon load. The MWRA sludge discharge (which ceased in December 1991) was the second largest source of most toxic contaminants.

The sources of toxic contaminants compiled for Massachusetts Bay include the atmosphere (wet and dry deposition), land runoff, Boston Harbor (CSOs, runoff, sediment resuspension and transport, sewage sludge and effluent), sources north of Cape Ann (e.g., Merrimack River), other NPDES discharges (e.g., Lynn and Salem), the Massachusetts Bay Disposal Site (MBDS), and other disposal sites (Menzie-Cura, 1991). Loading estimates for Massachusetts Bay indicate that Boston Harbor and the MWRA discharges (including sludge) are already a significant source of contaminants to the Bay, but that there are other sources of equal magnitude. For example, the dominant source of PCBs and Pb to Massachusetts Bay is deposition from the atmosphere (Figure 2-2). NPDES discharges (black pie slice in Figure 2-2) are the primary source of Cu and contribute about half of the total PAH load; the atmosphere contributes the other half. The MWRA outfall generally makes the largest contribution to the total NPDES load (see Table 2-1), but the contributions from other discharges are also significant and in some cases (Cd and Pb) will be even higher than the MWRA load with secondary treatment (MWRA, 1988).

Within the last 20 years, total toxic loadings to Boston Harbor and Massachusetts Bay have leveled off and have even begun to decrease due to smaller population increases, the enactment and enforcement of new, stricter environmental regulations, and toxic reduction programs that have been implemented by the MWRA and others. An example of this recent decrease in toxic loading is the sharp decline in loading of toxic metals from the MWRA effluent (Figure 2-3). In addition, recent analysis of the MWRA effluent for organic contaminants (PCBs, pesticides, and PAHs) using state-of-the-art analytical methods (Shea, 1992) indicates that many of the toxic organic compounds identified as being of concern (EPA, 1988) are 10 to 100 times lower than previously reported, with dieldrin being the only exception (see Figure 2-4). The lower concentrations reported by Shea (1992) are primarily the result of using analytical methods that reduce matrix interferences and provide very low detection limits. Actual reductions in organic contaminant concentrations may have taken place over the last 10 years, as has been the case with metals (Figure 2-3). However, because data based on low-detection-limit methods are not available from previous years, it is difficult to quantify the actual reductions in organic contaminant concentrations. As concentrations of contaminants in the effluent continue to decline due to source-reduction efforts by MWRA and greatly improved treatment, the ability to detect contaminants in the effluent could become more even problematic in the future (particularly using standard EPA analytical methods).

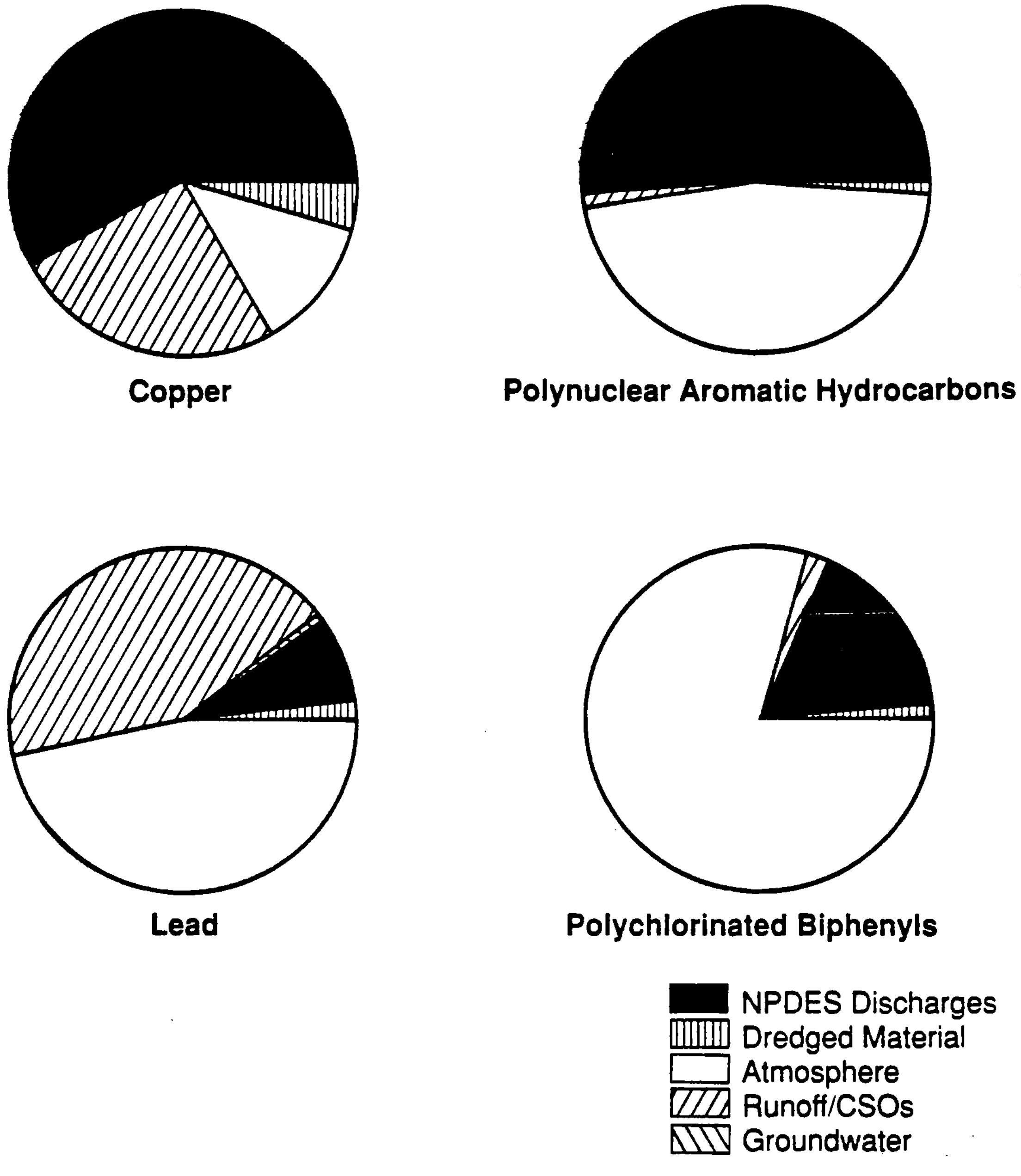


Figure 2-2. Estimates of contaminant loading to Massachusetts Bay [from Menzie *et al.*, 1991]

Table 2-1. Estimated Secondary Treatment Contaminant Loadings ($\mu g/s$) from Several Outfalls (MWRA, 1988).

	LYNN	SESD[a]	SWAMPSCOTT	CSOs[a]	MWRA
arsenic	<988		<6,730	4,325	20,000
cadmium	26,000	12,100	<82.5	2,075	22,100
chromium	25,900	23,600	2,300	10,940	111,600
copper	58,100	31,200	11,700	30,900	378,100
lead	172,000	37,100	3,660	77,090	157,000
mercury	1,950	4,790	24.4	1,373	6,500
nickel	94,000		1,800	21,630	282,500
silver	1,790		<80.5	1,081	9,400
zinc	347,000	47,200	73,700	129,300	1,092,400
selenium	<49,100		<3,360	80,860	140,000

[a] SESD: South Essex Sewage District; CSOs: Combined Sewer Overflows

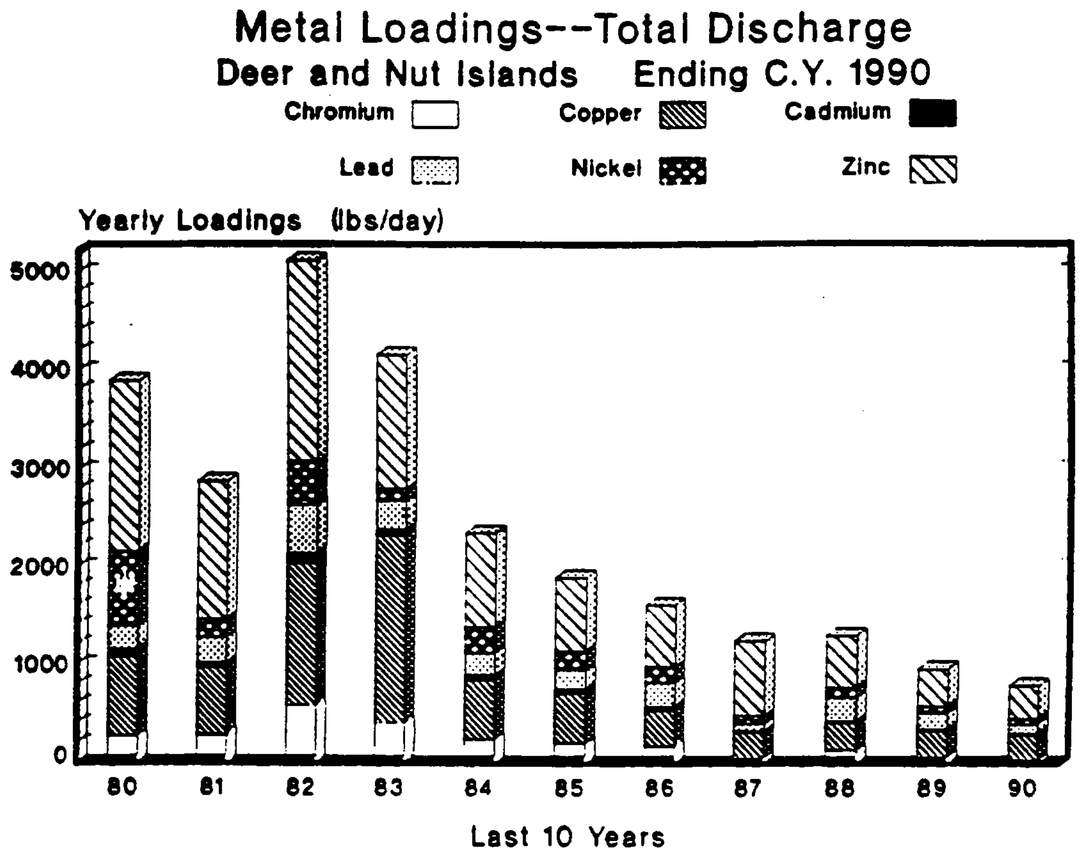


Figure 2-3. Metal loading from the MWRA effluents over the last decade. [MWRA, 1991, unpublished data]

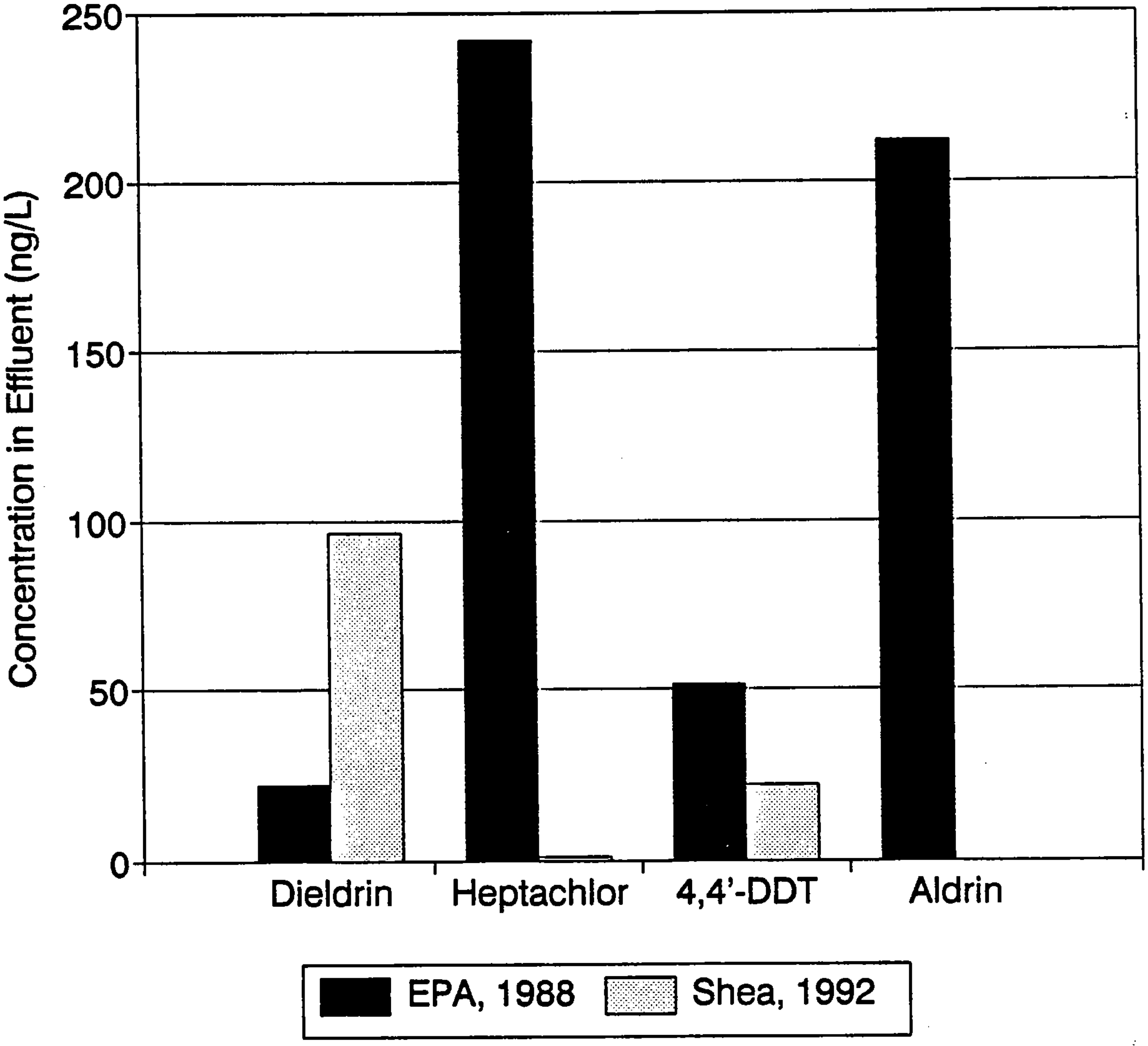


Figure 2-4. Contaminant concentrations in the MWRA effluent that were estimated in the SEIS for primary treatment (EPA, 1988) compared to those measured by Shea (1992)

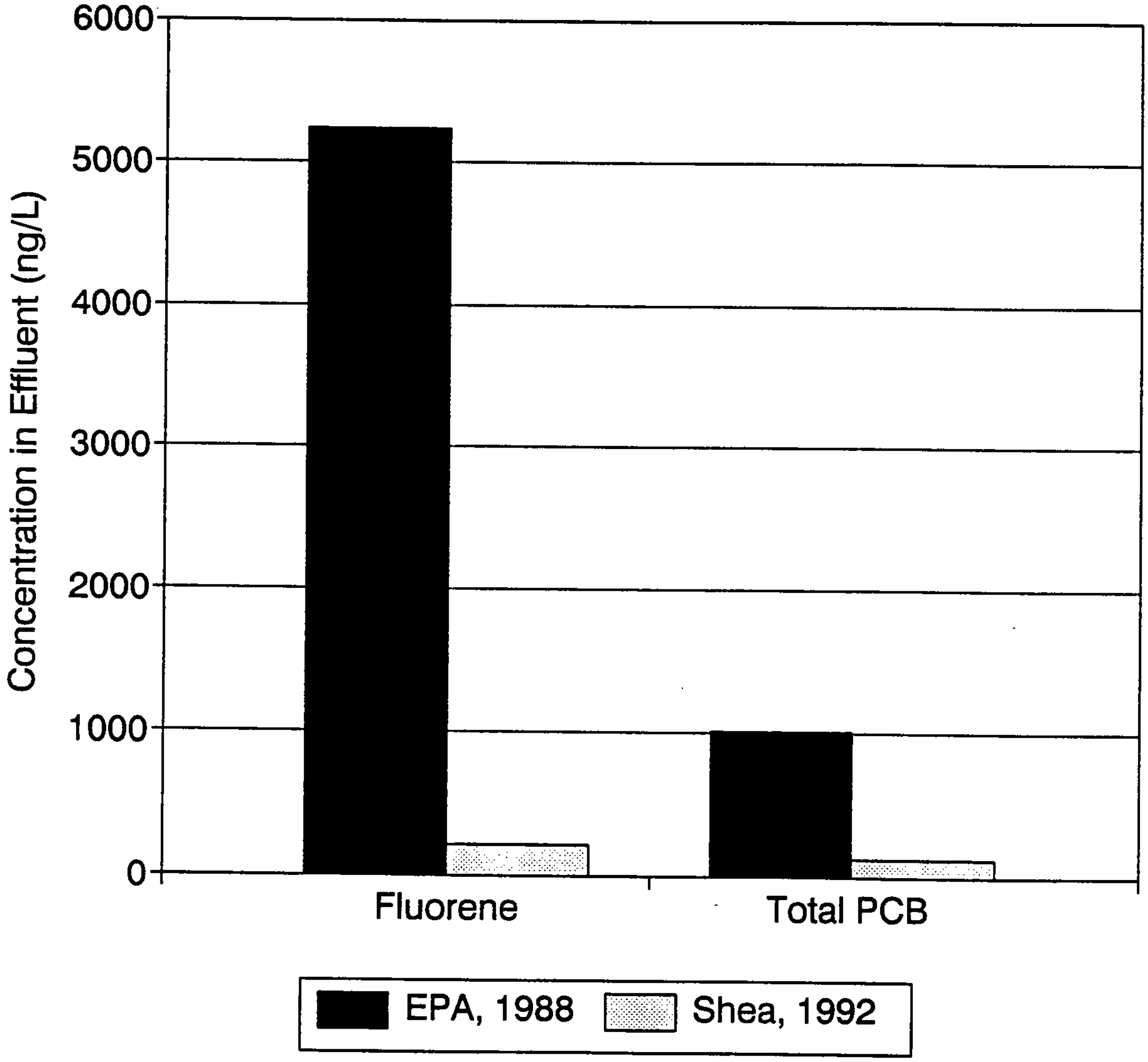


Figure 2-4 (continued). Contaminant concentrations in the MWRA effluent that were estimated in the SEIS for primary treatment (EPA, 1988) compared to those measured by Shea (1992)

Over the next several years, there should be even larger decreases in the loading of toxic contaminants to Boston Harbor and Massachusetts Bay due to the toxics reduction programs, cessation of dumping sewage sludge into the Harbor, and great improvements in the treatment of sewage with the new primary and secondary sewage treatment facility at Deer Island. The present sewage treatment at the Deer and Nut Island facilities removes less than 20% of the toxic contaminants from the effluent. When the primary treatment plant is operational in 1995, about 40% of the toxic material will be removed; secondary treatment will remove up to 85% of the toxic material (MWRA, 1988).

The contaminant loading from the MWRA effluent used by Menzie-Cura (1991) for their estimates of loading to Boston Harbor are in reasonably good agreement with the SEIS predictions for primary treated effluent to be discharged in 1995; the latter were based on measurements of the influent and expected removal efficiencies (EPA, 1988). Although these estimates are probably too high for many organic contaminants (and some metals), we will use these values to make some calculations on Harbor contaminant fate and subsequent transport to Massachusetts Bay.

2.3 PRESENT ROLE OF BOSTON HARBOR: EXPORT OF LOADS TO MASSACHUSETTS BAY

For many potentially toxic contaminants, a large fraction of the present loading to the whole of Massachusetts Bay enters through coastal embayments. Because Boston Harbor is the major embayment receiving very high loads of many chemicals, it is necessary to understand how the Harbor has influenced existing contaminant conditions in the vicinity of the future outfall site. This understanding is highly relevant to evaluating the future conditions in western Massachusetts Bay when much of the chemical discharge is shifted offshore with the new outfall and the role of the Harbor as a geographic point source and chemical conduit to the Bay is diminished. There is more information on contaminants in the Harbor than at the future outfall site. Examination of chemical fate and transport dynamics in the Harbor may help identify critical factors that influence a contaminant's fate and partitioning in the environment (e.g., amount dissolved in water vs. sorbed to suspended particles or in sediments) and would be invaluable in predicting chemical behavior at the offshore site. Thus, for several reasons, we discuss the fate of discharged contaminants and their export, under present conditions, from Boston Harbor to Massachusetts Bay.

The difficulty of constructing a reliable mass balance for contaminants input into open, dynamic coastal waters is well known. Uncertainties in chemical budgets therefore exist for shallow embayments the size of Boston Harbor; however, budgetary accounting will be even more difficult in a more open and dispersive environment like the future outfall site. Nevertheless, from the data at hand, one can approximate how much of a certain contaminant may pass through Boston Harbor to its offshore receiving waters. The approximation here is based on three lines of evidence: (1) existing concentrations in the water column, (2) chemical concentrations and estimated accumulation in sediments, and (3) chemical gradients from the Harbor to offshore.

2.3.1 Water Column Concentrations

Using an estimate of the chemical load of a compound or element to a water body, one can make simple calculations to predict resultant water column concentrations. By adding input mass into a volume of water and correcting for the residence time of that water, the resulting concentration offers a prediction that presumes the element mixes and flushes as does input water (i.e., it is said to be "conservative") and there is no environmental mechanism (removal or otherwise) for biogeochemical modification of the *in situ* concentration. If the observed concentration is similar to the prediction, the contaminant may be acting conservatively and therefore exchanging with water between embayment and offshore. Conservatism in this case implies small retention within the Harbor.

Using estimates of load to the Harbor for different contaminants (from Menzie *et al.*, 1991) and assuming a probable range of water residence time from 2 to 10 days (cf. Kelly, 1991; Signell, 1991), *in situ* total concentrations of several metals (Cu, Cd, Ni, Pb, Zn) were in the range predicted. [Observed concentrations shown in Figure 2-5 are from Wallace *et al.* (1988a) and MWRA (1988).] For the nine contaminants examined (Figure 2-5), a 3- to 4-day residence time appears indicated on average. The longer residence times for Cu and Cr could indicate that an additional source is needed (e.g., remobilization from sediment), that a significant source of these metals is from an area with a longer residence time (e.g., inner harbor), and/or that particulate removal is less active for these elements. The shorter residence time for PAH could indicate that particulate-bound PAH are deposited rapidly, that the source term is overestimated (e.g., using 1/2 detection limit for MWRA effluent), and/or that the dominant source is to a water mass with shorter residence time (e.g., MWRA outfalls in President and Nantasket Roads). Sung (1991) has used the presumption of conservatism of dissolved Cu and Zn to calculate about a 3-day residence time for these elements. Slightly longer residence times (6 to 10 days) were found for volatile halogenated organic compounds (VHOCs) during a tracer study conducted in the Northern Harbor (MWRA, 1988), but these could be due to an underestimate of the load to Boston Harbor (the only source term was the Deer Island effluent). Signell (1991) has estimated a range of Harbor water residence times from less than 3 days to almost 17 days, with an average for the whole Harbor around 8 days.

The discrepancies between water mass and contaminant residence times illustrate the fact that Boston Harbor cannot be treated as a homogeneous water body with contaminant input that is instantaneously mixed throughout the Harbor. The shorter residence times shown in Figure 2-5 support the hypothesis that because the present MWRA outfalls are in regions of rapid flushing, the contaminants in the MWRA effluent should be exported from Boston Harbor relatively fast. Alternatively, short geochemical residence times (less than water) implies fast removal to sediments (see Section 2.3.2).

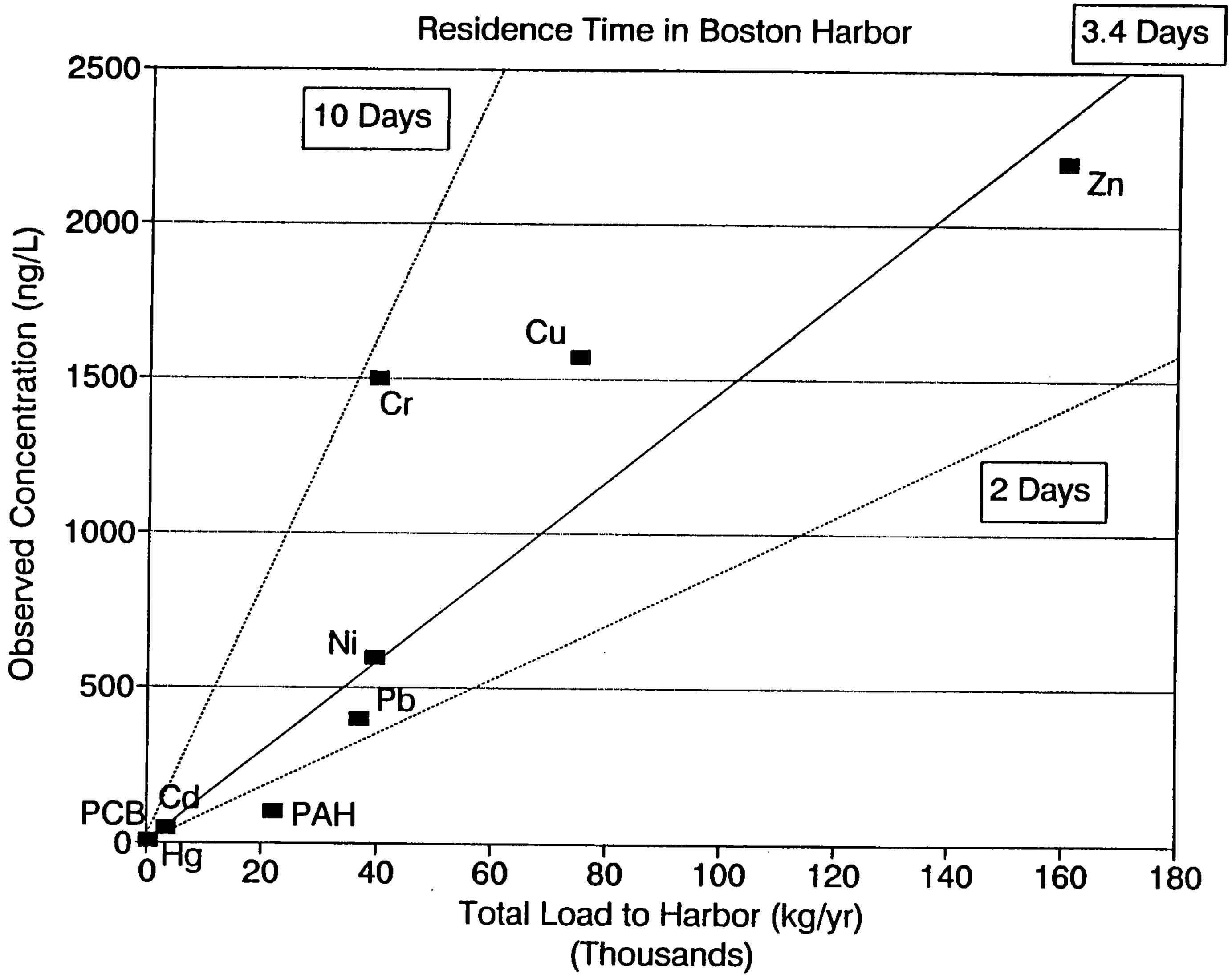


Figure 2-5. Residence times for several contaminants in Boston Harbor water.

There is uncertainty in the estimates of both the load and the average *in situ* Harbor concentrations (each over time and space), as well as some variation in the water residence time over seasons and over spatial scales within the Harbor. Because of these and other uncertainties, it is difficult to use the rough correspondence between observed and predicted values to arrive at a reliable quantitative prediction of export. A more rigorous model of water exchange between the Harbor and Bay, such as that being developed by Signell (1991), would be required to develop more accurate quantitative predictions of export. Though even with such a model, the variability in time and space likely will leave considerable uncertainty in export predictions developed in this manner. Regardless, data on the water column concentration of an array of contaminants strongly indicate substantial flushing to Massachusetts Bay of a number of chemicals now discharged into Boston Harbor.

Material delivered to Boston Harbor can have several fates other than transport offshore.

- It can be volatilized (directly or through methylation) and exported to the atmosphere (only viable for some compounds).
- It can accumulate in the water.
- It can accumulate in the bottom sediments.
- It can accumulate in larger biota (and be harvested).

The atmosphere is a poor sink for elements or compounds that essentially lack gaseous forms at environmental temperatures (e.g., most forms of metals), as well as for nonvolatile (and most semi-volatile) organic compounds. Accumulation and/or export in harvested biomass is a very small fraction of input. For example, using annual statistics on lobster harvest (Leigh Bridges, personal communication) and contaminant concentrations in lobsters (MWRA, 1990; Wallace *et al.*, 1988b), the contaminant removal via lobster harvest is on the order of 0.001% of a given contaminant's input. In part because the Harbor does flush rapidly, chemicals have little chance of accumulating in the water column (even if they do not act conservatively), except as associated with particles. Because particles constantly in suspension will be subject to offshore transport, chemical retention within the Harbor is principally due to particle settling and incorporation into the bottom sediments. Thus, as an alternative method for assessing the export of chemicals, one can attempt to estimate accumulation within sediments and, by difference from input, infer export.

2.3.2 Retention of Chemicals within Boston Harbor Sediments

To estimate the principal sink for metals and persistent nonvolatile organic contaminants input to Harbor waters, one first needs an estimate of net sedimentation — the mass or equivalent depth of sediment accumulating within a period of time, usually a year. This can be a difficult number to obtain. An estimate of total loading of suspended solids to the Harbor, calculated from Menzie *et al.* (1991) is about 9.8 x 10^{10} g/y, or about 900 g/m^2/y if spread evenly across the whole Harbor (108 km^2). Making some assumptions, as a maximum this represents roughly 0.1 cm/y in bottom sediments, which is in the range of the most recent estimates of Harbor-wide sediment burial rates. Most of the input of solids comes from effluent and sludge, which are very high in organic content. Once in the environment, much of this organic matter is consumed (the environmental equivalent of secondary sewage treatment). A rough calculation suggests that on the order of half of the estimated mass of suspended solids could be lost (converted to CO_2 and breakdown products) before long-term burial in the sediments. This loss would decrease the estimate of load available for net sediment accumulation to about 0.05 cm/y. However, the total load of solids estimated by Menzie *et al.* (1991) did not include input from marine erosion or input of marine muds from offshore; the combination of these sources could be on the same order as Menzie's estimated total load (H. Knebel, personal communication, 1992). On the other hand, dredging records (see Appendix) indicate that removal of sediments from Boston Harbor also approaches the total load of solids given by Menzie *et al.* (1991). On the whole, these two mechanisms of input and export may approximately balance.

For purposes of calculating chemical retention in the Harbor, we assume that all of the solids coming into the Harbor (900 g/m^2/y or 0.1 cm/y by the present estimate) are indeed trapped and that dredging offsets any input from erosion or transport from offshore. This is clearly a maximum that does not account for organic mass loss or transport out of the Harbor. [Recent estimates of sediment transport out of the Harbor are as high as 75% (E. Adams, personal communication, 1992).]

Using concentrations for various metals and organic contaminants measured throughout sediments in the Harbor, it is possible to get a first-order estimate of retention in sediments by multiplying concentration by sedimentation. This approach has uncertainties and possible sources of errors, including dilution of the input signal by bioturbation. Considering that the chemical and solid input to the Harbor has been either constant or decreasing for several decades and that the few depth profiles available do not show sharp concentration decreases with depth, the problem of diluting a chemical input signal as a result of bioturbation is probably minor.

Caveats in hand, we see that the mass calculated to be retained in sediments is generally not a large fraction of inputs (Table 2-2), supporting the water-column-based evidence of substantial advective transport. Some variation does appear across contaminants — the calculated percent retained mostly ranges from a few percent to about 30% for chemicals for which there are fairly reliable loading estimates. The highest suggested retention is for As (47%), although that value is based on effluent- and sludge-loading estimates that have considerable uncertainty (see above, Section 2.2) and does not include other sources of input to the Harbor (such as rivers) which for most other contaminants are comparatively less.

Table 2-2. Retention in accumulated bottom sediments in Boston Harbor.

Contaminant	Concentration[a] (ppm, dry wt)	Mass Retained[b] (kg/yr)	Mass Loading[c,d] (kg/yr)	% Retained
PAH[e]	5.45 (78)	534 (7,644)	18,427 to 21,230	2.5 (41)
Cu	93.3	9,143	71,972 to 78,190 [117,713]	12 to 13 [8]
Pb	113	11,074	35,369 to 38,574 [39,990]	29 to 31 [28]
Zn	179	17,542	152,891 to 168,166 [174,666]	10 to 11 [10]
Cd	2.33	228	~3,353 to 3,391 [7,781]	~7 [3]
Hg	1.21	119	~405 [745]	~29 [16]
As	11.9	1,166	[2483]	[47]
Cr	123	12,054	[40,065]	[30]
Ag	2.94	288	[10,761]	[3]
Ni	30.9	3028	[39,569]	[7]
PCB	0.48	47	400	12
DDT[f]	0.037	3.6	not available	
N	~3,000[g]	~2.9 x 10^5	~12,997,000	~2

[a] Calculated as area-weighted average for Boston Harbor from surface sediment concentrations given in MWRA (1990; Table 5.1) and assuming areas for different regions — Inner Harbor: 1×10^7 m^2, Northwest Harbor: 4.1×10^7 m^2, Central Harbor: 3.8×10^7 m^2, Southeast Harbor: 1.9×10^7 m^2.

[b] Sediment concentration multiplied by the estimated solids loading (900 $g/m^2/yr$) and the total area (10.8×10^7 m^2)for the Harbor.

[c] Unbracketed values are total loading from data summary of Menzie *et al.* (1991).

[d] Brackets are 10-yr average (1980-89) for MWRA effluent and sludge (A. Pancic, pers. comm.).

[e] Numbers in parenthesis include all "hot spots".

[f] Includes DDT and metabolites. The data set is very small.

[g] Estimate from other eutrophic areas; value is similar to Harbor dredged material (EPA, 1989).

In general, the variation in retention across contaminants relates well, with an exception or two, to some known differences in biogeochemistry. Take, for example, elements with suggested retention greater than 25% (Cr, Pb, Hg). The dissolved forms of these are known to be "particle-reactive" in coastal seawater and have summer temperature half-removal times of about 2 to 3 days (on the order of or less than the water residence time) (Santschi *et al.*, 1983). Moreover, this group in general has a high sediment-water distribution coefficient and thus a large fraction in particulate rather than dissolved form. In contrast, the elements with poorest retention (especially Ag, Ni, and Cd) have slow half-removal times (weeks to months during New England summer water temperatures) that would not allow much *in situ* removal within the time frame of the water residence time. Moreover, these elements in general have higher fractions in dissolved forms than in particulate forms. Copper and zinc appear to be intermediate in retention and this roughly coincides with their chemistry.

Arsenic seems a somewhat peculiar case: in terms of half-removal rates it might be grouped with Cd (Santschi *et al.*, 1983), yet its retention in Boston Harbor appeared highest. Based on the As/Ni ratio in loads, relative to the As/Ni ratio in Harbor sediments, it appears that As could be selectively retained. Another explanation would be that the total Harbor load is higher than the loading estimates used here (e.g., riverine input). If selective retention via active removal from the water does occur, this process must be included in any models predicting As concentration in water and sediments; note that the SEIS modeling assumed conservative behavior of *all* metals.

With minor exceptions, we postulate that retention of contaminants in Harbor sediments is a function of the particulate fraction in the load as additionally modified by subsequent particle-dissolved-colloidal partitioning once mixed with seawater and also by environmental half-removal times relative to water residence times. This is demonstrated in Figure 2-6 where the percent of contaminant exported from the Harbor (1 minus the percent retained from Table 2-2) is plotted against the percent of the contaminant in the dissolved phase (from Wallace *et al.*, 1988a) for four metals.

Regardless of the accuracy of the retention estimate, the relative contaminant retentions generally coincide with past studies and theoretical considerations. In addition, Figure 2-6 suggests that even if contaminants are almost entirely bound by particles, more than half of the contaminant load would still be exported from the Harbor; therefore, additional export mechanisms, such as dredging or suspended-particle transport, also could be very important. This result is consistent with the particle settling model used in the SEIS (EPA, 1988), which estimated that at least 50% of the particles from primary effluent would not settle within about 1 week, roughly the residence time of Boston Harbor water.

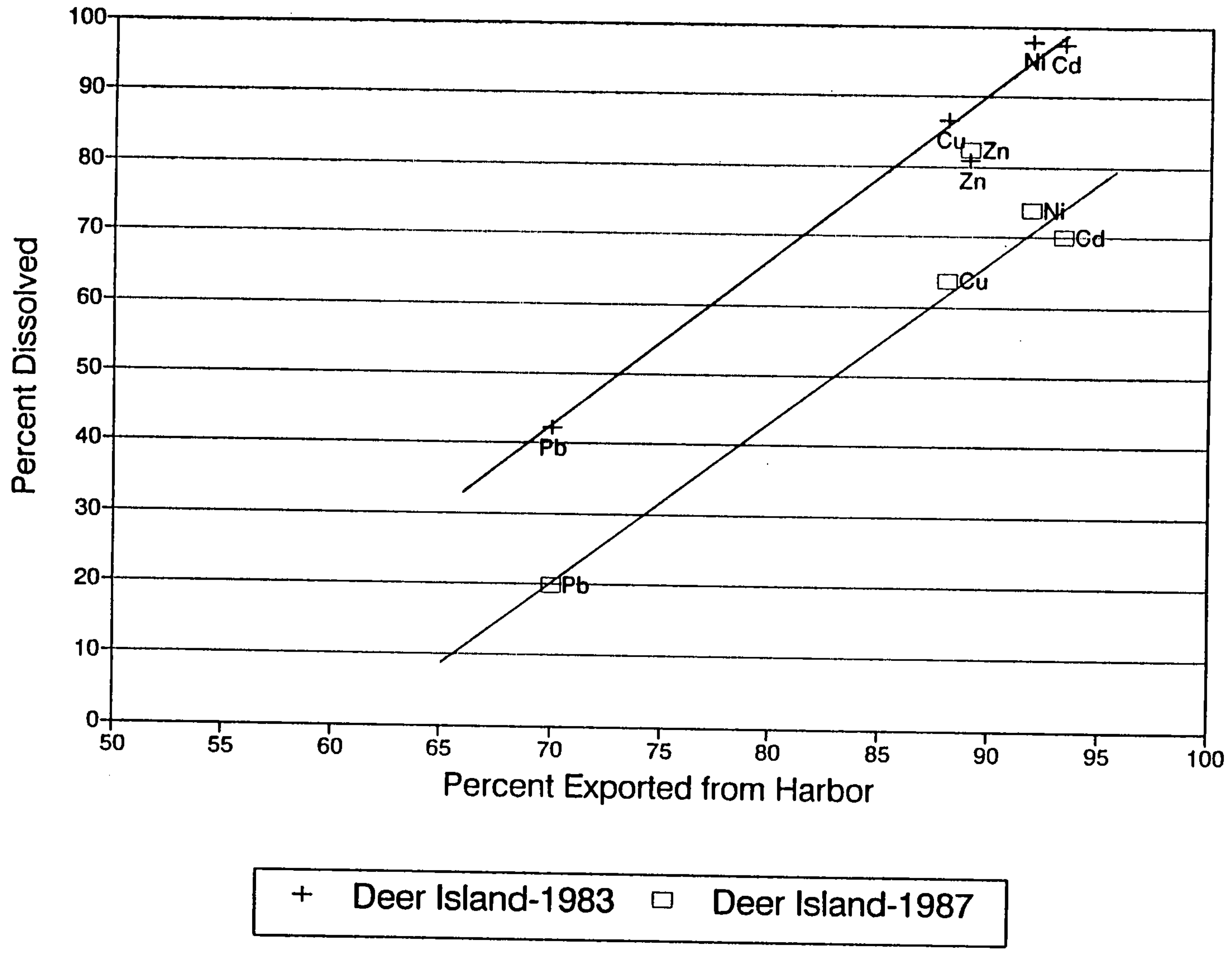


Figure 2-6. Relationship between dissolved metals in the water at Deer Island and the total metal load exported from Boston Harbor. [data are from Wallace *et al.*, 1988 and MWRA, 1988]

We have a basic dilemma, however, because Boston Harbor appears to have a high retention of solids but a low retention of contaminants. This has also been observed in selected smaller areas within the Harbor (Gallagher *et al.*, 1990). One explanation may be that contaminants are highly concentrated into slow-settling particles and dissolved or colloidal fractions, which are high in organic content and represent a relatively small fraction of the nondegradable (inorganic) mass of solids input. There may be other mechanisms, but it is not possible to resolve this discrepancy at present.

Differences in retention efficiencies (Table 2-2) may illustrate the basic difficulty of constructing mass balances as much as they provide unqualified quantitative evidence of high rates of contaminant export. It is certainly difficult to claim that a calculated retention efficiency is accurate to better than plus or minus 50%; but the relative ordering should be maintained even with improved understanding of sedimentation rates of solids, etc. In addition, the consistent conclusion of high export across a range of contaminants provides additional credibility to the hypothesis that there is rapid flushing of contaminants discharged from the present MWRA outfalls into Massachusetts Bay.

2.3.3 Export via Dredging and Disposal to the Massachusetts Bay Disposal Site

If much of the loading for most contaminants is indeed rapidly flushed to the offshore, such flushing is only one export mechanism. On a long-term basis, the amount dredged appears to be equivalent to the load of solids over the same period, by available estimates (EPA, 1989). Assuming the dredged mass has contaminant concentrations similar to the average surface-sediment values in the Harbor, essentially all of that fraction of chemical retained in Harbor sediments is also eventually removed.

In this case, the assumption can be validated and the evidence is very clear. The primary location for disposal of Harbor-dredged sediments, the Massachusetts Bay Disposal Site (MBDS), indeed has contaminant concentrations that mirror the levels in Boston Harbor, element for element (Shea *et al.*, 1991), and concentrations are similar also to the contaminant content of dredged material (which is measured before disposal) (EPA, 1989).

Thus, it seems that one way or another Massachusetts Bay has received most of the chemical loads that have entered Boston Harbor over the recent past. What is not exported in water exchange (of dissolved or slowly settling particulate-bound contaminants) to the near-coastal region is transported further offshore, directly to sediments in deeper waters via disposal of dredged sediment. Much of the fraction of contaminants that does remain in Boston Harbor ends up in the small (relative to Massachusetts Bay) depositional sites that have been well documented as hot spots and consequently exhibit signs of impact (e.g., Deer Island Flats).

2.3.4 Evidence from the Neighboring Recipient System: Gradients into Western Massachusetts Bay

If there is indeed poor retention of the present contaminant load to the Harbor (the majority of which is MWRA discharge), there should be some evidence of a gradient either in water column or sediment concentrations (or both) extending from the Harbor offshore, particularly for contaminants that have relatively small loads from other sources.

Such evidence is recorded in the sediments. The concentrations in the sediments (normalized to grain size or total organic carbon) at various sites offshore are much less than in the Harbor (Shea *et al.*, 1991). In the vicinity of the future outfall and shoreward to Boston Harbor, the sedimentary environment is extremely heterogeneous, ranging from silt-clay soft-bottom patches to gravel/boulders over short distances. Therefore, appearance of a uniform sediment gradient of chemical contamination from the Harbor is complicated by patchiness of depositional and highly turbulent bottom environments. Nevertheless, contaminants such as As, Cu, Hg, Ag, Zn, and PCB, once properly normalized, do generally show a decrease from Boston Harbor (Shea *et al.*, 1991; see also Table 2-2 and Figure 3-2). A transect in the sediment from the Harbor across Broad Sound to Nahant (MWRA, 1992) has recorded a sharp decline in concentrations of Cu and number of spores of the bacterium *Clostridium perfringens*, a good tracer of sewage particles. In addition, a sediment survey conducted in October 1991 (MWRA, manuscript in preparation) has shown that *C. perfringens* spores are widely distributed throughout Massachusetts and Cape Cod Bays, with clear gradients away from Boston, Lynn, and Salem Harbors (Figure 2-7), the three dominant point sources of sewage contamination in the region. Note that most nonpoint sources of sewage contamination (e.g., septic systems on Cape Cod) will not contribute to input of *C. perfringens* to the Bay because there is very poor exchange of particles (spores) between ground water and Bay water.

There is also good evidence that contaminant (metal, PCB, PAH, and pesticide) concentrations in the water column (MWRA, 1988; L. Pitts, personal communication) and in fish and shellfish tissue (MWRA, 1988; Battelle, 1990; Boehm *et al.*, 1984) exhibit a gradient from Boston Harbor to Massachusetts Bay. A similar gradient in the prevalence of histological abnormalities (e.g., vacuolation of cells) in winter flounder was found by Moore *et al.* (1992), with prevalences decreasing in the order Lynn Harbor > Boston Harbor > Massachusetts Bay > Cape Cod Bay.

Unfortunately, data are not available at sufficient spatial resolution to develop contoured illustrations of a concentration gradient of water column (or tissue) contamination away from a Harbor source, as was done by Kelly (1991) for nutrients. However, during the STFP studies, eight volatile halogenated organic compounds (VHOCs) were used as tracers of the present effluent discharge as it mixed into Massachusetts Bay (MWRA, 1988). The VHOCs were measured in the effluent and the receiving waters along several transects from Boston Harbor through Massachusetts Bay. The data

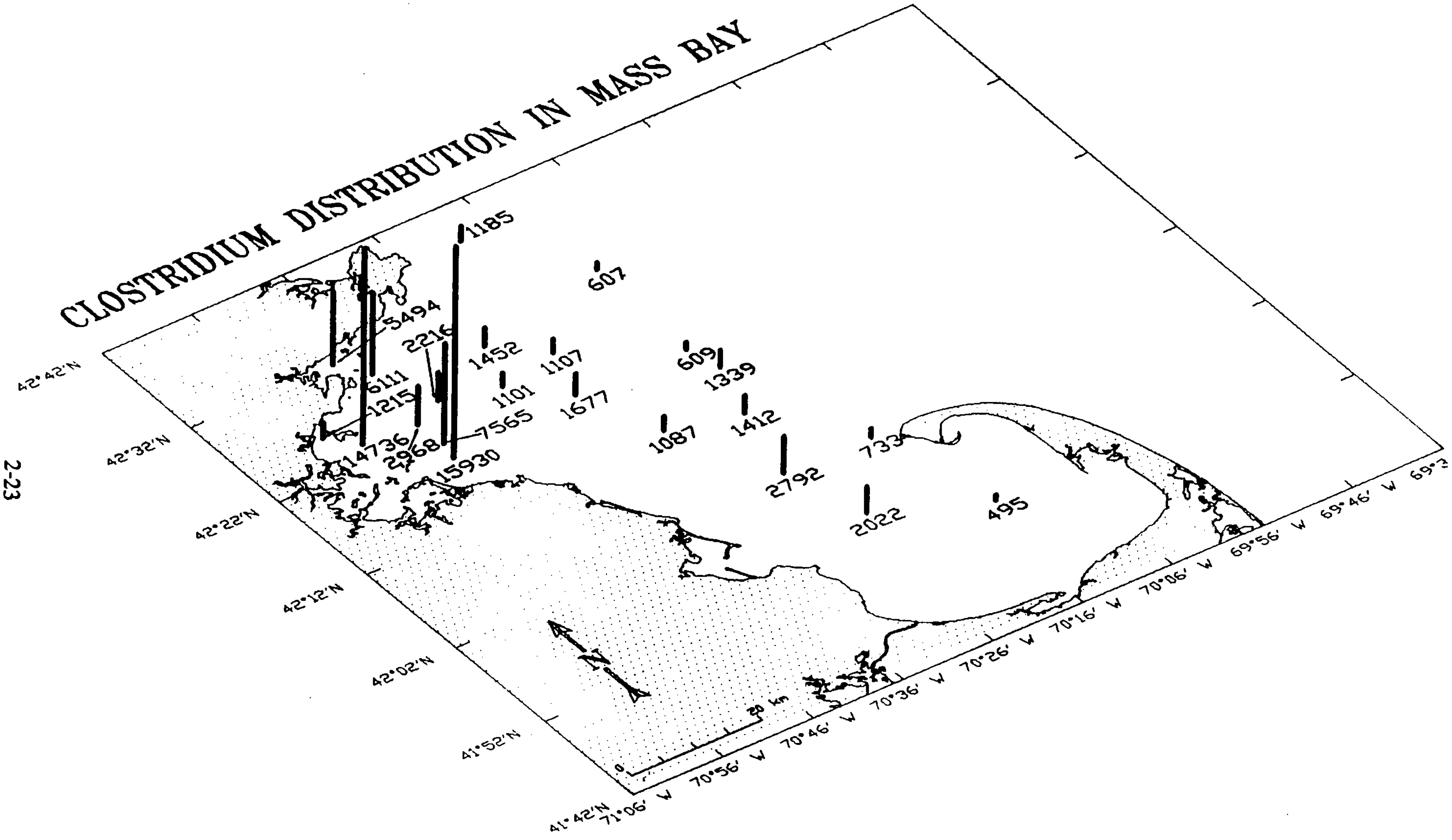


Figure 2-7. *Clostridium perfringens*, a tracer of sewage particles, in Massachusetts and Cape Cod Bays sediments (spores/g, dry weight, normalized to total organic carbon)

indicated a consistent decrease in VHOC concentration with distance from Boston Harbor (10-fold decrease in concentration over about 10 miles), supporting the idea that most of the dissolved contaminants (there is very little sorption of VHOCs to particles) entering Boston Harbor are transported a significant distance out into Massachusetts Bay.

It is interesting to note that concentrations of individual VHOCs were less than 0.2 ppb right near the Deer Island outfall. This value is far below any water quality criteria for aquatic life or human health. Thus, it would be hard to envision any significant problems associated with VHOCs at the future outfall in Massachusetts Bay.

2.4 PRESENT SITUATION RELATIVE TO FUTURE OUTFALL

2.4.1 Analogous Features Influencing Transport and Fate

Two principal phenomena that are apparent for Boston Harbor should be kept in mind when predicting outfall effects at the future offshore site, and comparing these to the present situation. The first is the inferred effect of water flushing in effectively exporting (and diluting) loads. The second is the existence of differences in contaminant fate based on differences in chemical forms and behavior.

In the more open, deeper, and (intentionally chosen) more dispersive offshore environment near the future outfall (relative to the present outfall), contaminant export should be at least as high as in Boston Harbor within a spatial area similar to the Harbor (i.e., about 100 km^2). In principle, because of greater water depth alone, the initial dilution at the future site should provide water column concentrations lower than those seen in Boston Harbor. Besides dilution effects, large-scale circulation patterns in the Bay influence water near the future outfall site much more than water near Boston Harbor (MWRA, 1988). The result is increased flushing, which reduces the residence time (and steady state concentrations) of contaminants in much of Massachusetts and Cape Cod Bays (MWRA, 1988; see also Figure 1.4 in MWRA, 1992).

The character of the discharge (dissolved versus particulate fractions) and the environmental partitioning between dissolved and particulate forms (which affects settling behavior) both are important to predictions of environmental fate. Particle settling dynamics and water residence times may vary between the inshore and offshore situations, and this could influence the environmental concentrations of some elements relative to others. Even so, retention in deposited sediments around the outfall would be expected to be most efficient for those chemicals that either enter the environment largely in a particulate form or are highly particle reactive once in the environment. Among metals, these might include Pb and Hg (and perhaps Cu and As). A gradient away from the outfall (in the absence of post-depositional transport) might appear as changing metal ratios as a function of time and distance, simply reflecting differences in removal rates from the water.

2.4.2 Additional Factors To Consider in Making Predictions for Change at the Future Outfall Site

The fate of chemicals within Boston Harbor does not provide a perfect analogue for Massachusetts Bay. Differences that could affect geochemical dynamics are not limited to physical processes and water depth, though these were prime factors that varied in the SEIS site evaluation and are indeed important. Several additional factors are briefly described next.

2.4.2.1 Seasonal Feature of Particle Accumulation

The Harbor sediments show a high degree of spatial heterogeneity, and this is at least as true for the bottom areas surrounding the outfall. A major difference between the Harbor and Bay, however, relates to temporal variability in depositional and erosional processes. Unlike the Harbor, the Bay stratifies vertically and a seasonal accumulation of deposits, even in hard-bottom areas, seems prevalent during summer stratification (MWRA, 1988). This material accumulates only temporarily and is resuspended (during autumn) to be transported elsewhere. A fair amount of "digestion" and chemical modification may therefore occur before the settled matter is incorporated into the sediment at long-term accumulation sites. The seasonal feature of particle accumulation has not yet been explicitly considered in attempting to simulate long-term spatial patterns of sediment contamination surrounding the future outfall.

2.4.2.2 Enhanced Role of Pelagic Removal Processes (Including Zooplankton)

In the Harbor, the shallowness of the water engenders a large role for benthic-pelagic interactions. In the deeper Bay, a pelagic community dominates, and in general zooplankton may play a more significant role in regulating phytoplankton as well as in facilitating particle removal via fecal pellets. The role of zooplankton in the Bay, although conceivably much more significant than in the Harbor, is not known and may impact some geochemical dynamics involving toxic contaminants.

2.4.2.3 Effects of Stratification

Stratification can have direct effects on contaminant fate by reducing vertical mixing, and hence vertical particle-transport, and also by decoupling the more energetic and biologically active surface layer from the bottom layer. Indirect effects of stratification can also impact geochemical behavior in many other ways. These include reduced potential for photochemical processes (if chemicals do not mix to surface irradiance levels), changed 0_2-pH environment in trapped subpycnocline waters, and minimized impacts of phytoplankton activity on element speciation and distribution.

2.4.2.4 Spatial Scale of Concern

The openness of the Bay system surrounding the outfall site makes geochemical budgeting (a prerequisite for rigorous transport/fate modeling) a task far more difficult than for Boston Harbor with its relatively constrained points of water exchange. Given uncertainties in budgeting materials in the Harbor, we must firmly understand the critical geochemical processes to include in any predictive model that examines the future outfall in the open Bay.

3.0 POTENTIAL IMPACT RESULTING FROM SEWAGE DISCHARGE AT THE NEW OFFSHORE OUTFALL

In this chapter, some of the SEIS predictions (EPA, 1988) are examined in light of estimated loadings, measured effluent concentrations, transport and fate processes, and potential effects of the various contaminants. To focus the discussion on important biogeochemical processes, we have assumed that the physical oceanographic model used in the SEIS is not a dominant source of error in the predicted contaminant distributions; thus, we discuss potential refinements of the physical transport model only qualitatively. This assumption can be tested (and predictions verified or modified) upon completion of the circulation model being developed by the USGS (Bothner *et al.*, 1990).

3.1 WATER QUALITY

One of the environmental criteria that will be used to assess impact of the new outfall discharge is whether conditions just outside the mixing zone comply with water quality criteria (WQC) set by the Environmental Protection Agency (EPA). The WQC include chronic and acute values for the protection of aquatic life. These WQC, which are generally technically defensible and widely used throughout the United States, have been adopted by the Commonwealth of Massachusetts as water quality standards. Two other types of WQC also can be used for this comparison: human toxicity criteria and human-health-risk carcinogenicity criteria (usually 10^{-5} or 10^{-6} risk factors). These criteria are based on lifetime exposure and fish consumption. They have not been adopted as standards by Massachusetts (and thus are not presently enforceable), and they are less frequently used for comparison, in part because of questions concerning their technical defensibility (Kelly and Cardon, 1991).

The SEIS reported predicted concentrations of contaminants at the edge of the mixing zone at each potential outfall site (for primary and secondary treatment) and compared them to all four types of criteria. All sites were predicted to have some criteria violations, with the fewest being at Site 5 (Table 3-1). One way to assess the validity of the SEIS predictions is to compare the predicted concentrations of contaminants that exceed criteria at President Roads (the present outfall site) with observed concentrations.

Table 3-1. Constituents that exceed EPA Ambient Water Quality Criteria.

PRIMARY TREATMENT EFFLUENT

Criteria	Chemical	Exceedances at Sites	Reason
CMC	copper	PR, 2, 3, 4	MWRA
CCC	mercury	all	MWRA
CCC	4,4' - DDT	all	MWRA
CCC	dieldrin	PR	MWRA
CCC	heptachlor	all	MWRA
CCC	PCBs	PR, 2, 3, 4	MWRA
10^{-6} carcinogenicity	arsenic	all	ambient
10^{-6} carcinogenicity	4,4' - DDT	all	MWRA
10^{-6} carcinogenicity	dieldrin	all	MWRA
10^{-6} carcinogenicity	fluorene	all	MWRA
10^{-6} carcinogenicity	heptachlor	all	MWRA
10^{-6} carcinogenicity	aldrin	all	MWRA
10^{-6} carcinogenicity	PCBs	all	ambient
10^{-5} carcinogenicity	arsenic	all	ambient
10^{-5} carcinogenicity	4,4' - DDT	all	MWRA
10^{-5} carcinogenicity	dieldrin	PR	MWRA
10^{-5} carcinogenicity	heptachlor	PR,2	MWRA
10^{-5} carcinogenicity	aldrin	all	MWRA
10^{-5} carcinogenicity	PCBs	all	ambient

CMC: Criterion Maximum Concentration (acute)
CCC: Criterion Continuous Concentration (chronic)
PR: President Roads (present outfall location)

Table 3-1 (continued)

SECONDARY TREATMENT EFFLUENT

Criteria	Chemical	Exceedances at Sites	Reason
CMC	copper	PR	MWRA
CCC	mercury	all	MWRA
10^{-6} carcinogenicity	arsenic	all	ambient
10^{-6} carcinogenicity	4,4' - DDT	all	MWRA
10^{-6} carcinogenicity	dieldrin	PR	MWRA
10^{-6} carcinogenicity	heptachlor	PR, 2, 3	MWRA
10^{-6} carcinogenicity	aldrin	all	MWRA
10^{-6} carcinogenicity	PCBs	all	ambient
10^{-5} carcinogenicity	arsenic	all	ambient
10^{-5} carcinogenicity	4,4' - DDT	PR	MWRA
10^{-5} carcinogenicity	PCBs	all	ambient

CMC: Criterion Maximum Concentration (acute)
CCC: Criterion Continuous Concentration (chronic)
PR: President Roads (present outfall location)

In Figure 3-1, the SEIS model predictions (EPA, 1988) for President Roads (PR-1 and PR-2 in Figure 3-1, for primary and secondary treatment, respectively) are compared to actual ambient concentrations at President Roads (PR-OBS), Massachusetts Bay (MB), and a reference site (REF). These plots provide a means of evaluating the ability of the model to accurately predict ambient concentrations at President Roads. Observed data for the MB site (MWRA, 1988) and REF site (Furness and Rainbow, 1990) provide lower bounds of concentrations within the Bay (MB site is at the northern end of Stellwagen Basin) and in typical coastal Atlantic waters (REF). The most stringent water quality criteria values (primarily 10^{-6} carcinogenicity risk factors) are also indicated on each plot to evaluate the potential for exceedances, and standard EPA method detection limits are indicated to evaluate the ability to enforce these criteria using analytical methods required by EPA.

For primary treated effluent at President Roads, the SEIS initial-dilution-model predictions exceed the actual concentrations of contaminants measured at the site in all cases. For secondary treatment, predictions exceed observed values for all but two contaminants (dieldrin and PCBs). In most cases, the predicted concentrations are higher than the observed concentrations by more than a factor of 10. In addition, the predicted ambient concentrations for metals (Cu, Cd, Ni, Pb, and Zn) after primary treatment exceed concentrations measured by Wallace *et al.* (1988a) within the Deer Island effluent plume (not shown), and observed concentrations of VHOCs (MWRA, 1988) are about 10 times less than predicted (not shown). The observed concentrations in President Roads exceed the most stringent WQC for As, DDT, dieldrin, and PCBs. Note that all of these WQC are 10^{-6} human-health risk factors rather than chronic or acute marine aquatic life criteria (i.e., those adopted by Massachusetts), and that all of these concentrations are below the detection limit of the standard EPA method required under current regulations. Importantly, the observed contaminant concentrations in President Roads do not exceed any of the acute or chronic WQC adopted by Massachusetts (not shown).

Based on discussion in Section 2, there are three primary factors that would yield this level of overestimate.

- Overestimate of contaminant concentrations (and therefore loading) in the effluent
- Underestimate of contaminant removal terms in the SEIS (STFP) modeling
- Underestimate of the effluent initial dilution

Any overestimate of metal loading would result in consistently high predictions for all sites evaluated in the SEIS. Assuming that any missing removal terms or poor dilution modeling would result in an equal error at each site, the SEIS initial dilution model could provide reliable relative predictions among sites, which was its stated purpose (EPA, 1988). However, the water column predictions appear to overestimate metals by up to a factor of 40 (Hg) and organic compounds by up to a factor of 90 (aldrin). Assuming that the degree of overestimation for the future outfall discharge (near Site 5) is the same as at President Roads, the only WQC exceedances at the future outfall site would be

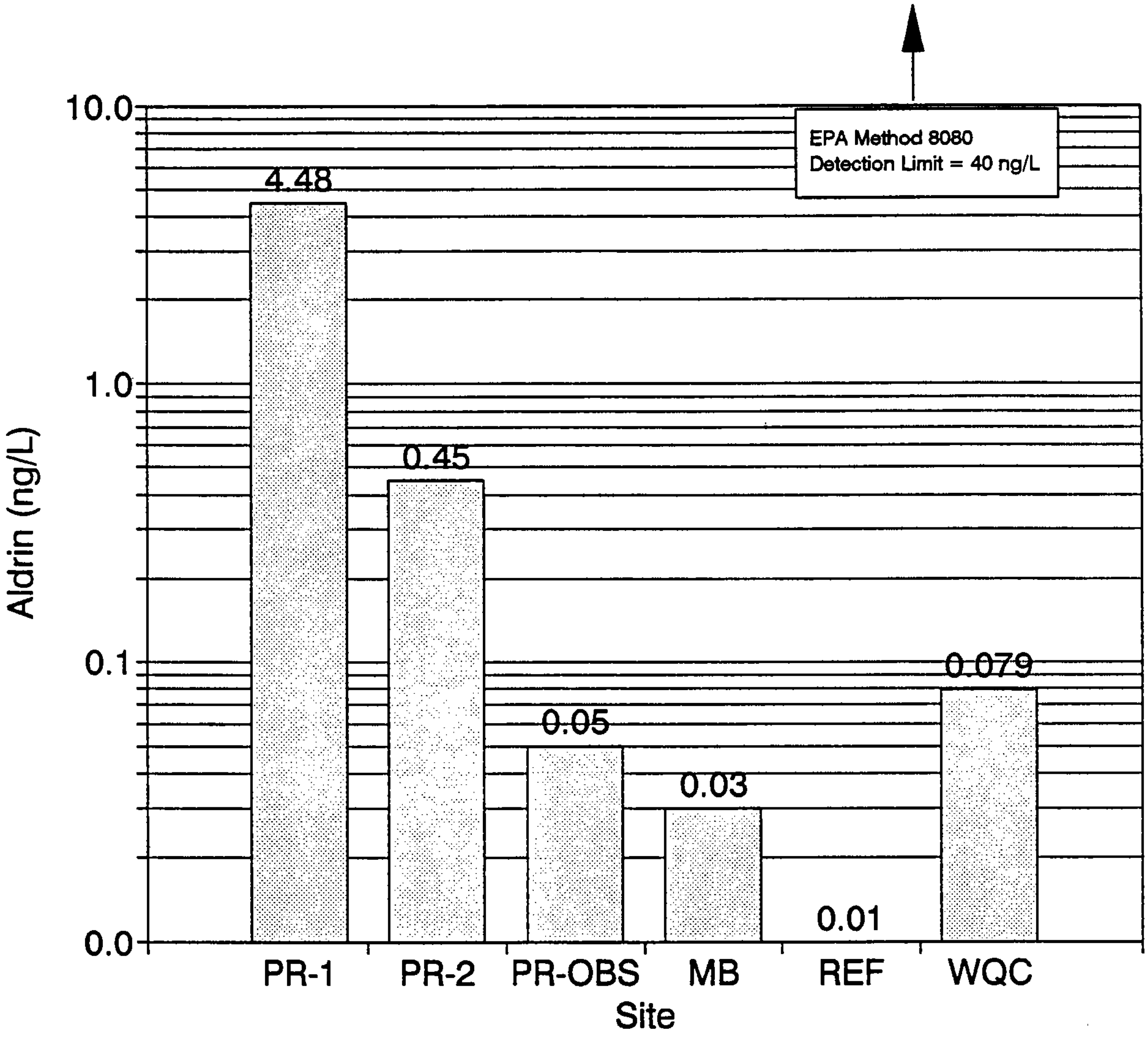


Figure 3-1. Contaminant concentrations in the water predicted in the SEIS at President Roads (PR) for primary (-1) and secondary (-2) treatment are compared to observed (OBS) concentrations. Concentrations in northern Massachusetts Bay (MB), in continental shelf water (REF), and for the most stringent water quality criteria (WQC) are also given.
a. Aldrin

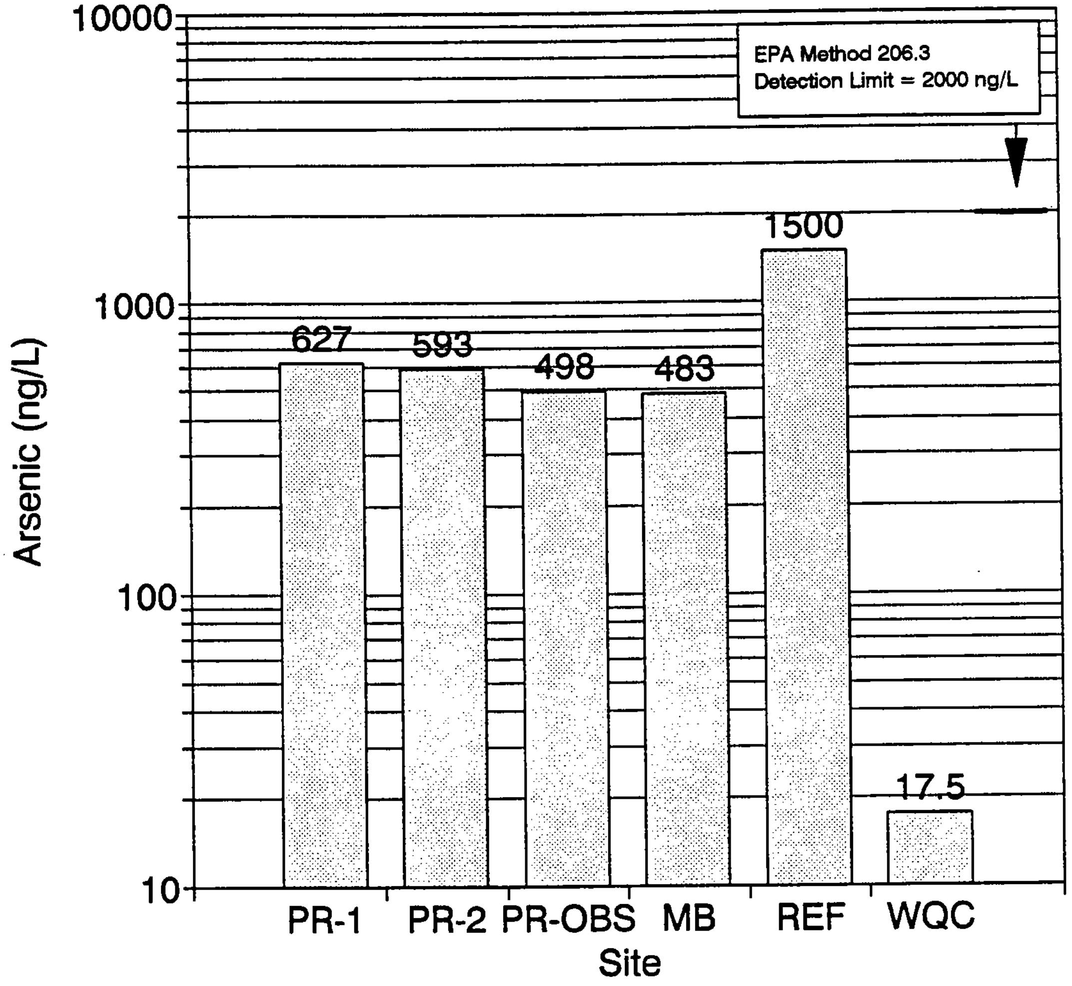


Figure 3-1. Contaminant concentrations in the water predicted in the SEIS at President Roads (PR) for primary (-1) and secondary (-2) treatment are compared to observed (OBS) concentrations. Concentrations in northern Massachusetts Bay (MB), in continental shelf water (REF), and for the most stringent water quality criteria (WQC) are also given.
b. Arsenic

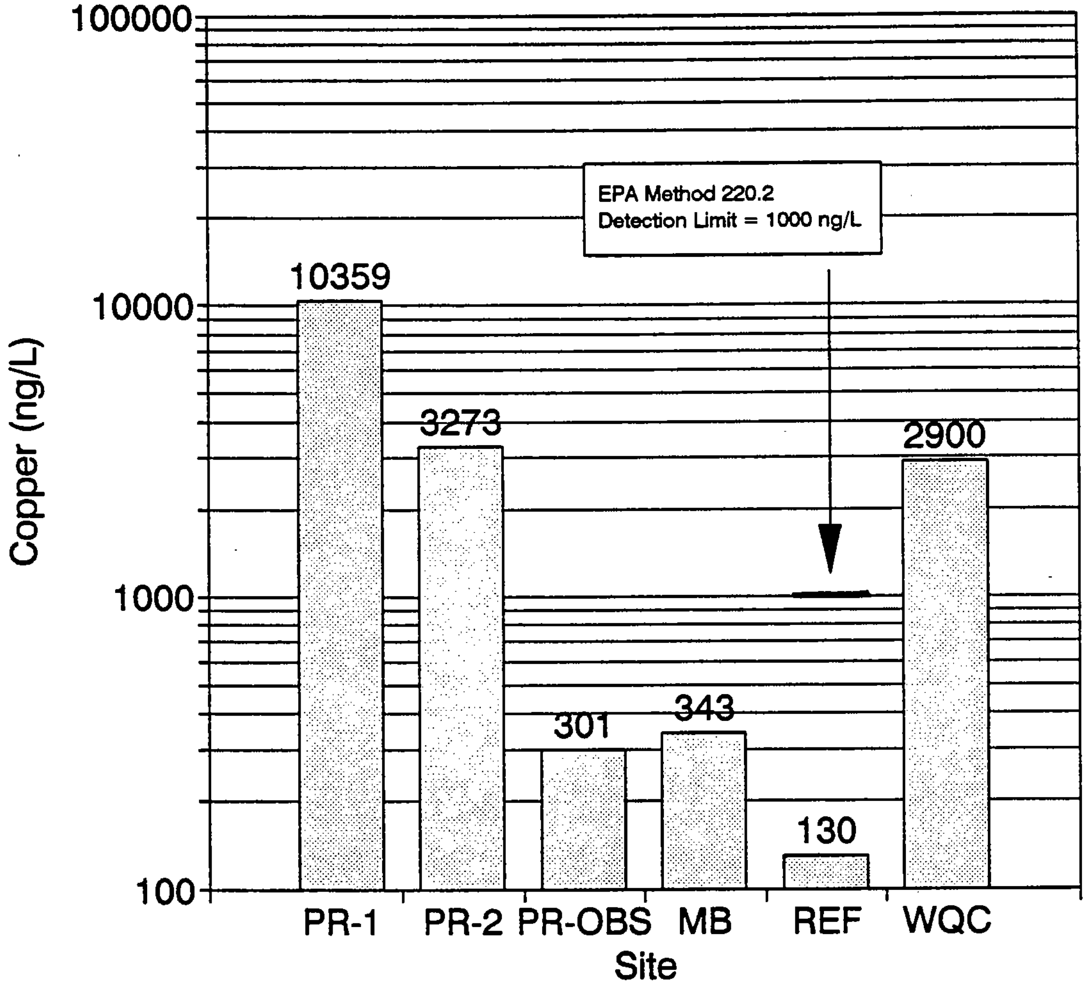


Figure 3-1. Contaminant concentrations in the water predicted in the SEIS at President Roads (PR) for primary (-1) and secondary (-2) treatment are compared to observed (OBS) concentrations. Concentrations in northern Massachusetts Bay (MB), in continental shelf water (REF), and for the most stringent water quality criteria (WQC) are also given.
c. Copper

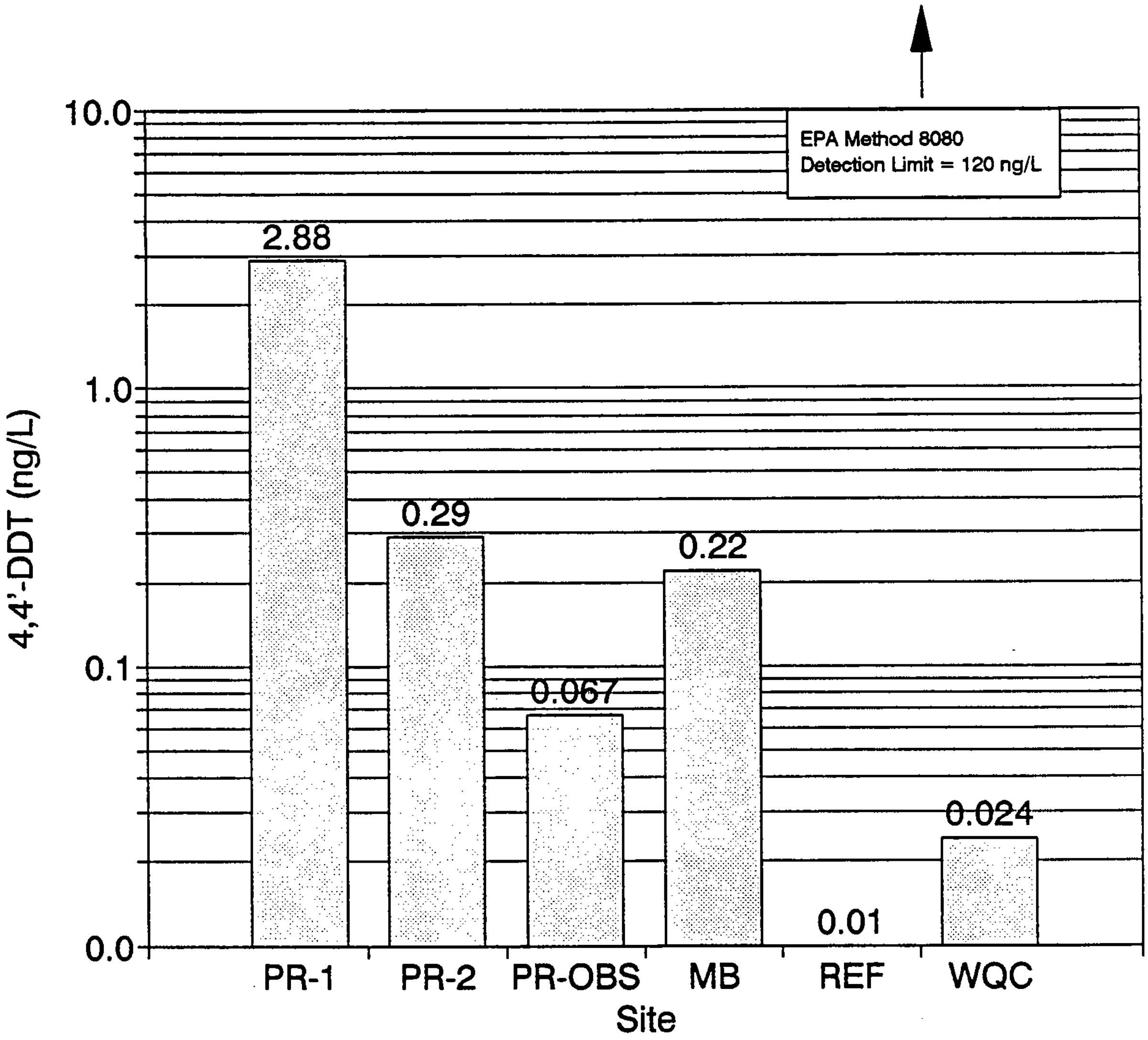


Figure 3-1. Contaminant concentrations in the water predicted in the SEIS at President Roads (PR) for primary (-1) and secondary (-2) treatment are compared to observed (OBS) concentrations. Concentrations in northern Massachusetts Bay (MB), in continental shelf water (REF), and for the most stringent water quality criteria (WQC) are also given.
d. DDT

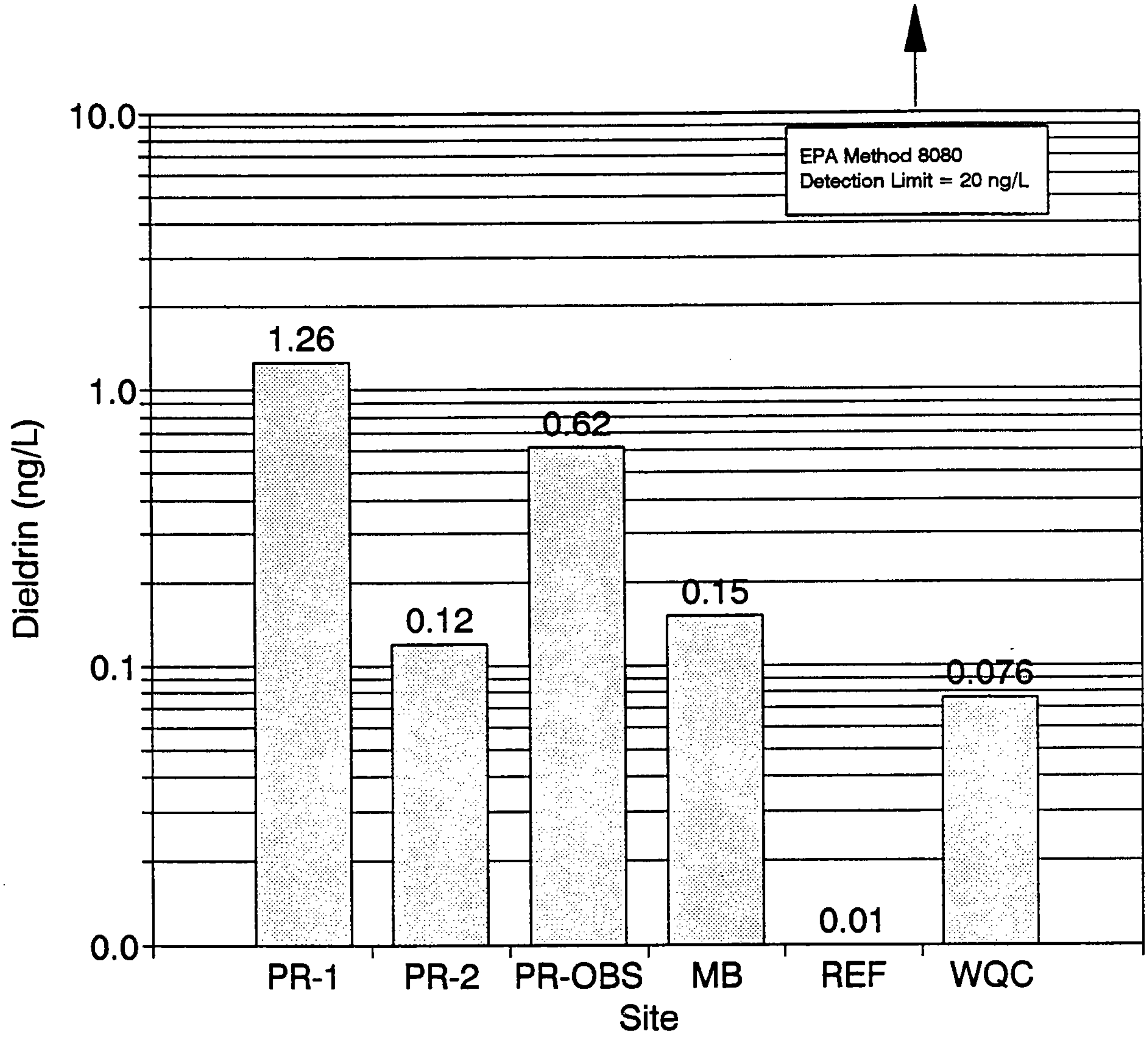


Figure 3-1. Contaminant concentrations in the water predicted in the SEIS at President Roads (PR) for primary (-1) and secondary (-2) treatment are compared to observed (OBS) concentrations. Concentrations in northern Massachusetts Bay (MB), in continental shelf water (REF), and for the most stringent water quality criteria (WQC) are also given.
e. Dieldrin

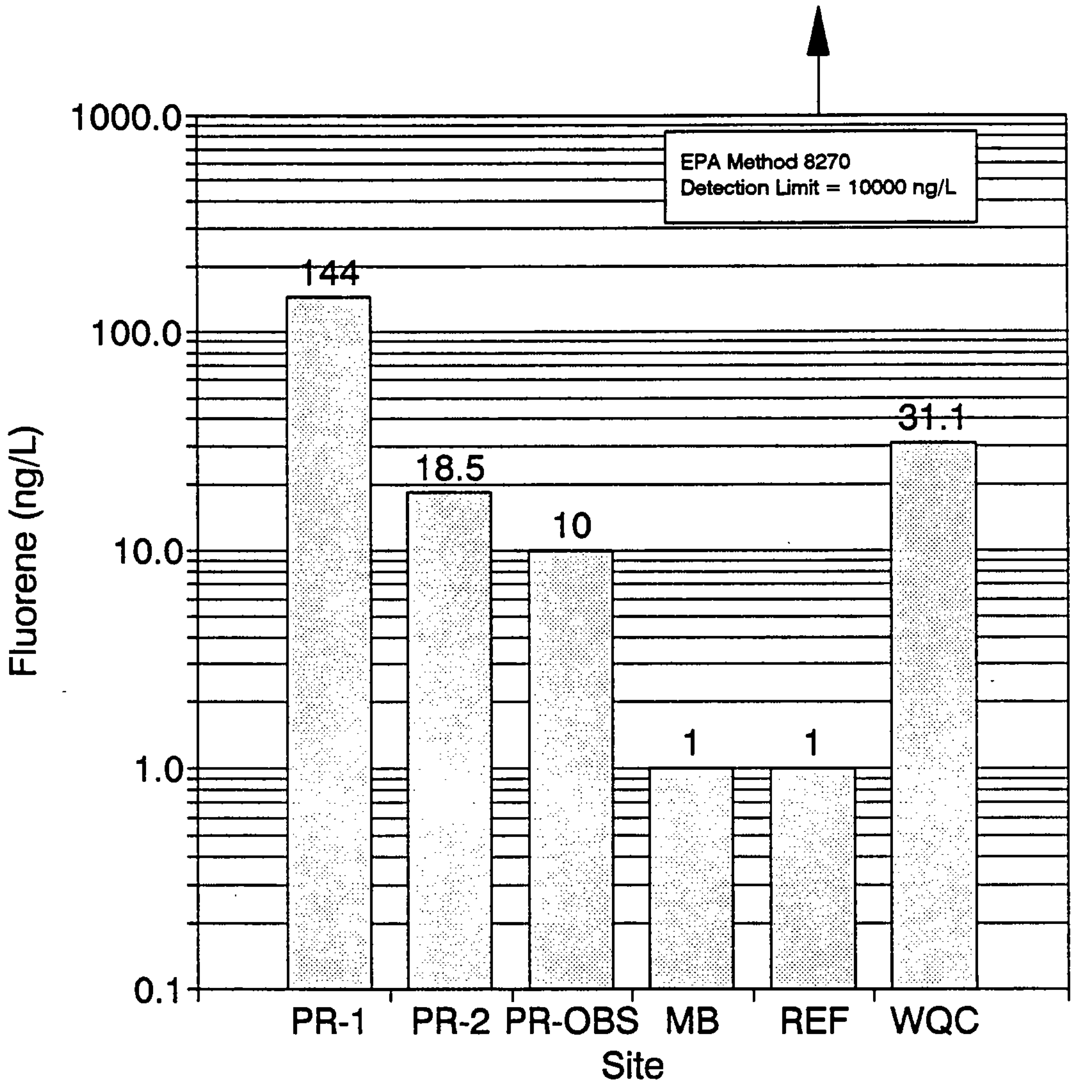


Figure 3-1. Contaminant concentrations in the water predicted in the SEIS at President Roads (PR) for primary (-1) and secondary (-2) treatment are compared to observed (OBS) concentrations. Concentrations in northern Massachusetts Bay (MB), in continental shelf water (REF), and for the most stringent water quality criteria (WQC) are also given.
f. Fluorene

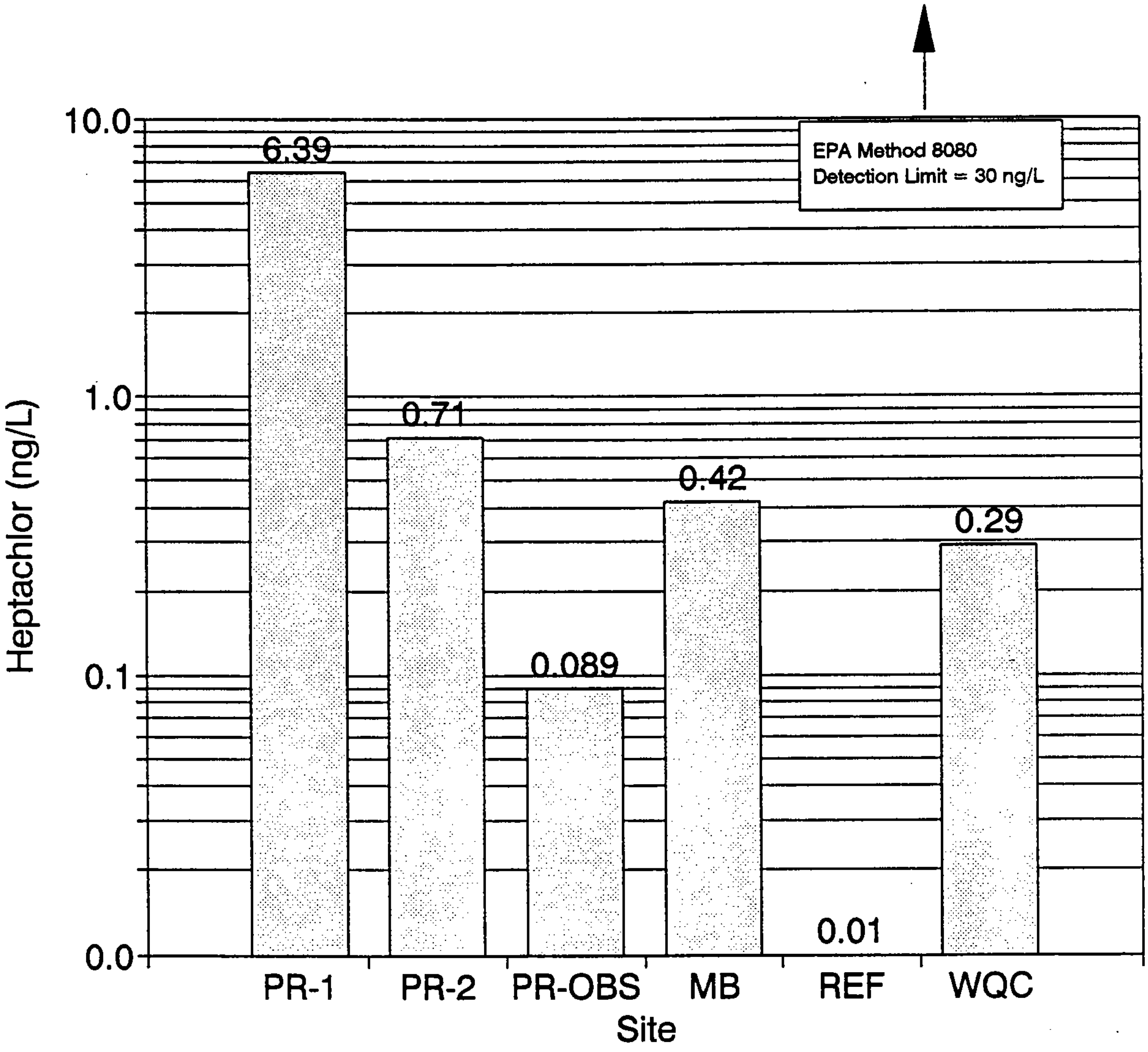


Figure 3-1. Contaminant concentrations in the water predicted in the SEIS at President Roads (PR) for primary (-1) and secondary (-2) treatment are compared to observed (OBS) concentrations. Concentrations in northern Massachusetts Bay (MB), in continental shelf water (REF), and for the most stringent water quality criteria (WQC) are also given.
g. Heptachlor

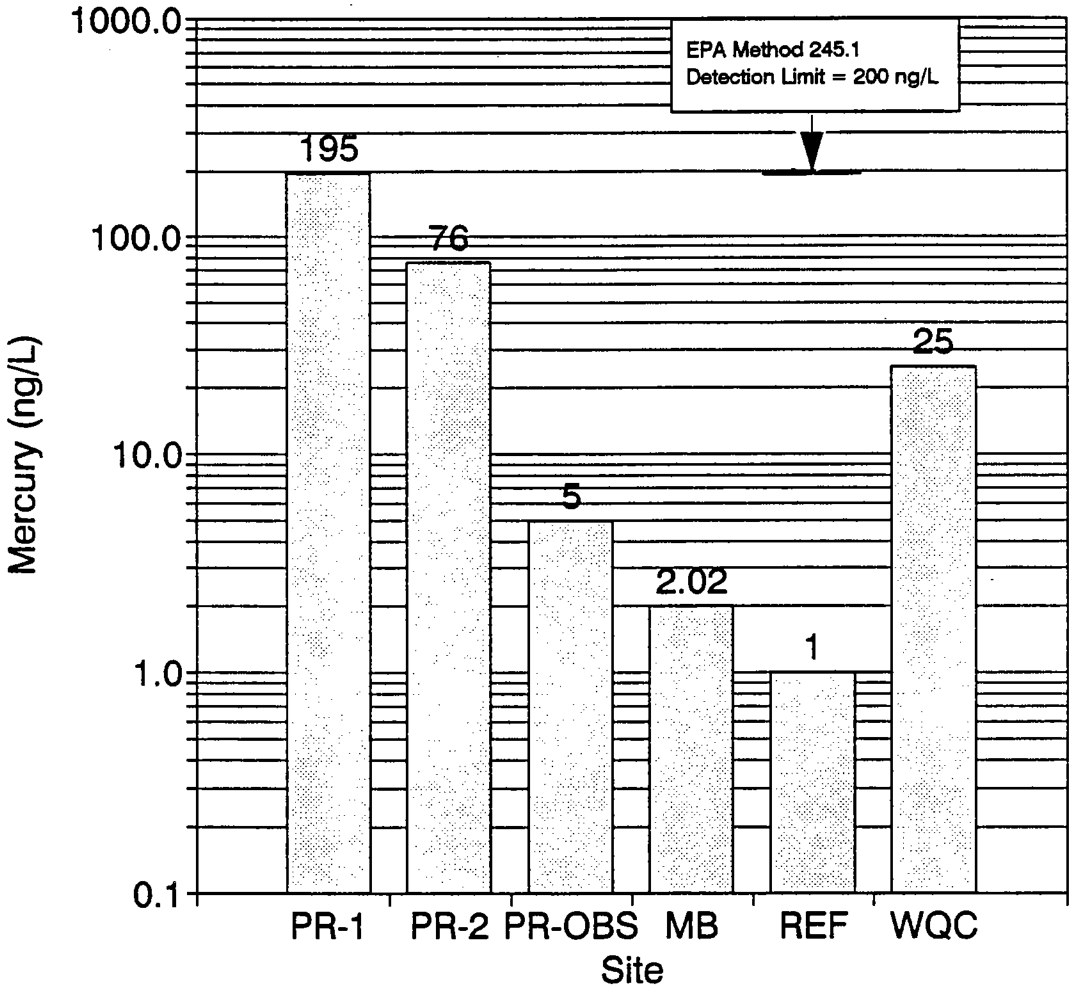


Figure 3-1. Contaminant concentrations in the water predicted in the SEIS at President Roads (PR) for primary (-1) and secondary (-2) treatment are compared to observed (OBS) concentrations. Concentrations in northern Massachusetts Bay (MB), in continental shelf water (REF), and for the most stringent water quality criteria (WQC) are also given.
h. Mercury

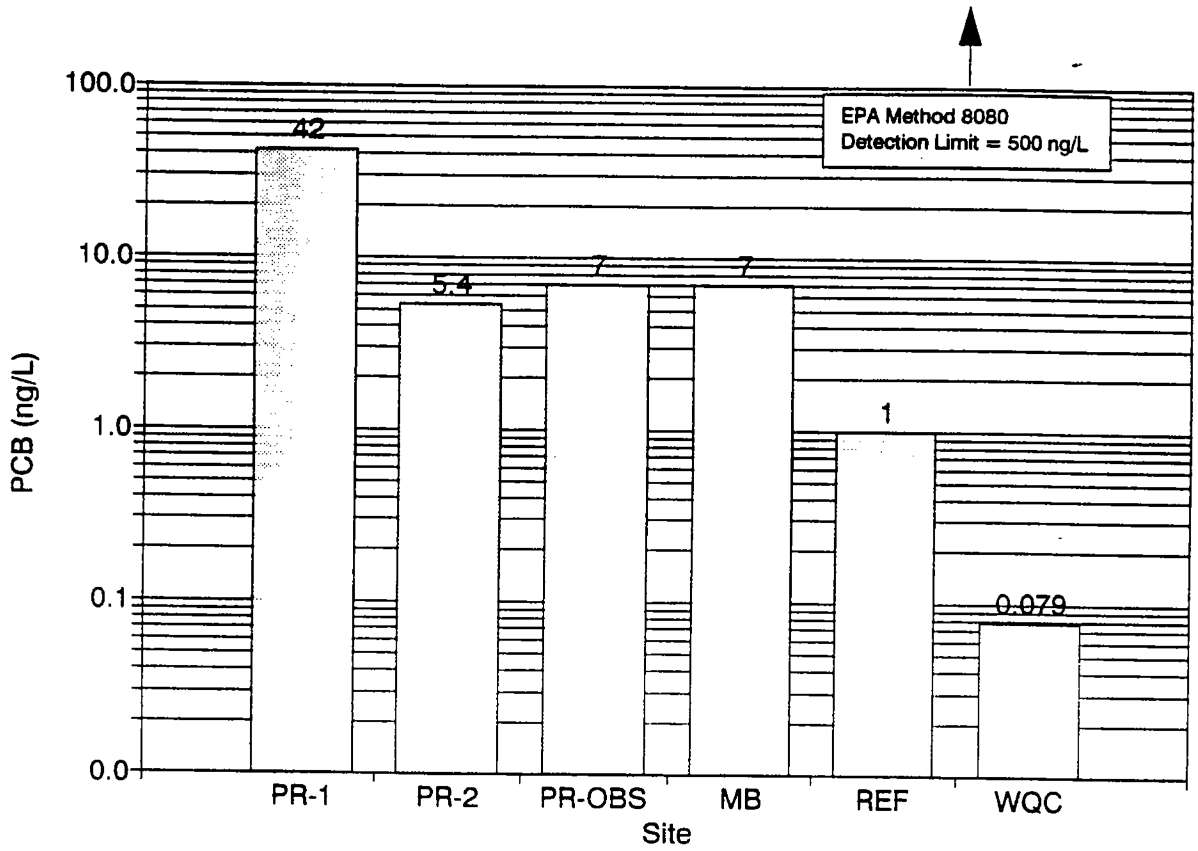


Figure 3-1. Contaminant concentrations in the water predicted in the SEIS at President Roads (PR) for primary (-1) and secondary (-2) treatment are compared to observed (OBS) concentrations. Concentrations in northern Massachusetts Bay (MB), in continental shelf water (REF), and for the most stringent water quality criteria (WQC) are also given.
i. PCB

for the 10^{-6} risk factor for As and PCB. Both of these criteria are currently exceeded at reference sites in Massachusetts Bay and in typical ocean water, and both of these criteria are being reviewed by EPA because of possibly inflated potency factors used in the criteria derivation. Note also, that the predicted concentrations outside the mixing zone at Site 5 (EPA, 1988) actually exceed the present levels in Boston Harbor for all contaminants except dieldrin; for several metals the predictions even exceed the levels in the Deer Island effluent plume (Wallace *et al.*, 1988a).

The overestimate of effluent contaminant concentrations could be a very significant error for some contaminants, but it is one that can be easily verified and corrected. A recent interlaboratory comparison study of analyses of 21 municipal sewage effluents in the metropolitan New York area (Battelle, 1991) found a greater variability and a high bias (overestimate) for several metals when using standard EPA methods for analysis. The study concluded that overestimates of metal concentrations in sewage effluent are common for two reasons:

1. Detection limits for the standard EPA methodology used for effluent analysis are too high — often on the same order as the concentration in the effluent.
2. Many laboratories that routinely sample and analyze effluent do not adequately control contamination, nor do they correct for or minimize positive interferences that are common in this type of matrix.

These observations underscore the need to use analytical methodologies that provide lower detection limits and better data quality than those offered by standard EPA methods, particularly for Hg and many of the organic contaminants. For example, the standard EPA Method 245.1 for Hg has a nominal detection limit of 200 ng/L, but typical sewage effluent concentrations are often less than about 10 ng/L (see Battelle, 1991), whereas the EPA chronic marine WQC is 25 ng/L. Using 1/2 the detection limit (i.e., 100 ng/L) would overestimate Hg concentrations at least 10-fold in this case. The error is not quite as large for most other metals, but may be even larger for organic contaminants. Data from Shea (1992) indicate that many organic contaminants in the MWRA effluent may have been overestimated by a factor of 100 or more (see Figure 2-4).

Based on the results of the analysis of contaminant concentrations in the MWRA effluent (Shea, 1992), results of the modeling reported in the SEIS (EPA, 1988), cessation of sewage sludge discharge, and the greatly improved sewage treatment at the new Deer Island facility, the concentrations of contaminants in the water column of Massachusetts and Cape Cod Bays are likely to decrease except in the immediate vicinity of the outfall. Contaminant concentrations should be monitored in this "nearfield" area to verify the expected compliance with WQC and facilitate the modeling of longer term transport and fate. Otherwise, monitoring for contaminants should be restricted to state-of-the-art analysis of the effluent and any areas of possible contaminant accumulation (i.e., sediment or biota). There is no evidence that indicates a need to monitor contamination in the water column far away from the mixing zone, unless the intent is to study transport and fate processes.

3.2 SEDIMENT QUALITY

The physical and chemical environment of bottom sediments influences the benthic biological community, which in turn may affect plankton communities and demersal fisheries. For a number of reasons, benthic infaunal monitoring continues to be a mainstay of environmental assessment and monitoring in the marine environment. However, in contrast to the water column, the sediments are not yet governed by quality criteria, though several approaches are being considered (Shea, 1988).

There is relatively poor information on what concentrations of specific contaminants affect different benthic populations and communities. The additional problem with a complex discharge like a sewage effluent is that many chemicals potentially may act (cumulatively or interactively) as a toxic stress to resident organisms or produce tissue burdens passed through the demersal food web. Yet, as a primary sink for metals and persistent organic compounds, accumulations of contaminants in sediments are a main mechanism for longer term effects and must not be overlooked. A sense, even if coarse, of toxicant concentrations that would ensue in sediments could aid in projecting potential consequences of a new offshore outfall.

There are significant limitations to predictability in the case of sediment contaminant concentrations over space and time. As examples, these include

- Extreme environmental variability in spatial distribution of bottom type
- Seasonal variability in particle settling, sediment resuspension, and transport
- Non-uniformity, across chemicals, in spatial distribution in surface sediments
- Contaminant-specific variability in water column removal via particle deposition
- Post-depositional processes that resuspend and transport particles after initial deposition

These factors are briefly highlighted before examining projections and SEIS modeling efforts.

3.2.1 Heterogeneity

Spatial and temporal heterogeneity of the Massachusetts Bay benthic environment near the future outfall is one important feature that makes projections difficult. The diversity in the sedimentary environment has been well documented — boulders, gravel, and pockets of silt-clay soft bottom coexist as a mosaic of patches on virtually all spatial scales recently used to characterize the bottom (Shea *et al.*, 1991; Bothner *et al.*, 1990). Regarding temporal heterogeneity, various studies have described a seasonal accumulation of deposition to bottom areas (including hard-rock bottoms) that can occur during stratified water column conditions when bottom waters are relatively quiescent (Battelle, 1987). These accumulations are indeed temporary to some bottom areas, evidenced by scouring and post-depositional removal from hard-bottom areas during autumn and winter storms.

Perhaps as a partial consequence of the environment's physical variability, but also due to each chemical's particular sources and behavior within the Massachusetts Bay environment, there is a striking variability in present background levels of different chemicals in sediments near the future outfall (Shea *et al.*, 1991). For example, from observed data, the distribution of As (Figure 3-2a) in sediments just inshore, north and south, of the future outfall diffuser field is very different from that of other contaminants such as PAH (Figure 3-2d). A southern site had very high As (reason undetermined) and otherwise a fairly uniform distribution just west of the future outfall. PAH at the southern station was very low, whereas the transect directly towards Boston Harbor from the future outfall site had two distinct peaks in concentration amidst three valleys.

Other chemicals have different distributions. For Cd (Figure 3-2b), the northernmost site (toward Salem) showed the highest levels recorded throughout this whole field, no large enrichment at the high As site, no peak and valley pattern like the one for PAH, and a broad general trend towards increasing concentrations westward towards the Boston shore. Copper (Figure 3-2c) in general showed the least relative variability throughout the middle of the described field, had higher but not atypical values at the high Cd site (north), and had distinctly lowest values at the high As site (south). The pattern for Cu, although less pronounced than Cd, may suggest slightly enriched levels towards western and northern shorelines with known metal sources (Boston, Lynn, and Salem Harbors).

Some metals (Cd especially, but also Cu) may have relatively poorer retention efficiency within inshore embayments (Table 2-2 and Figure 2-5); accordingly their distribution offshore should more effectively show an inshore-offshore gradient (e.g., Cd in Figure 3-2b). With chemicals having lesser "signals" from inshore, in theory the environmental variability could be strong enough to obscure shoreward source signals. However, contaminant distributions in sediments of Western Massachusetts Bay are confounded by an additional source — there is a possibility that contaminated sediments at the MBDS have been resuspended and transported to other depositional sites.

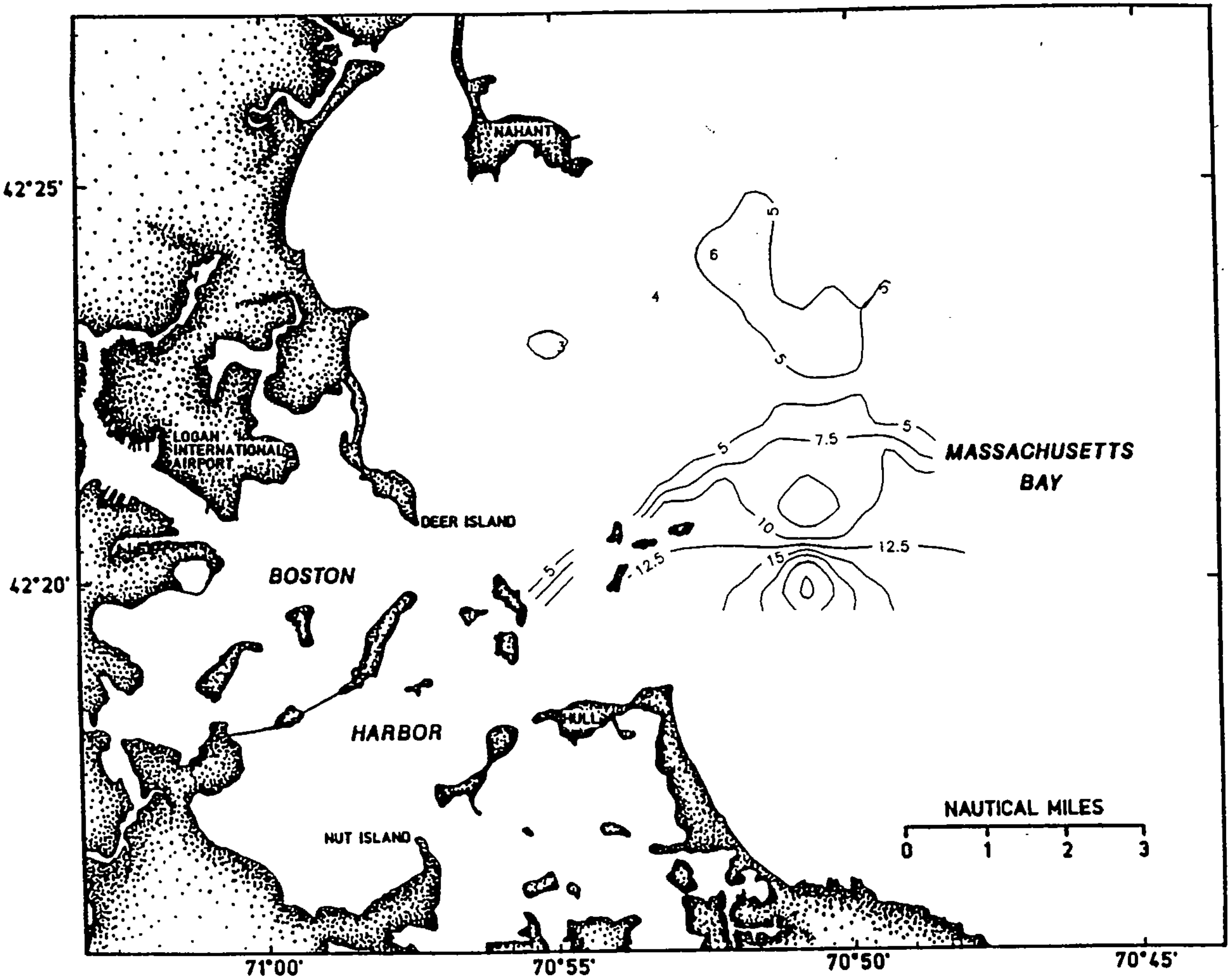


Figure 3-2. Contour plot of contaminant concentrations near the new MWRA offshore outfall (modified from Shea *et al.*, 1991).
a. Arsenic (μg/g, dry weight)

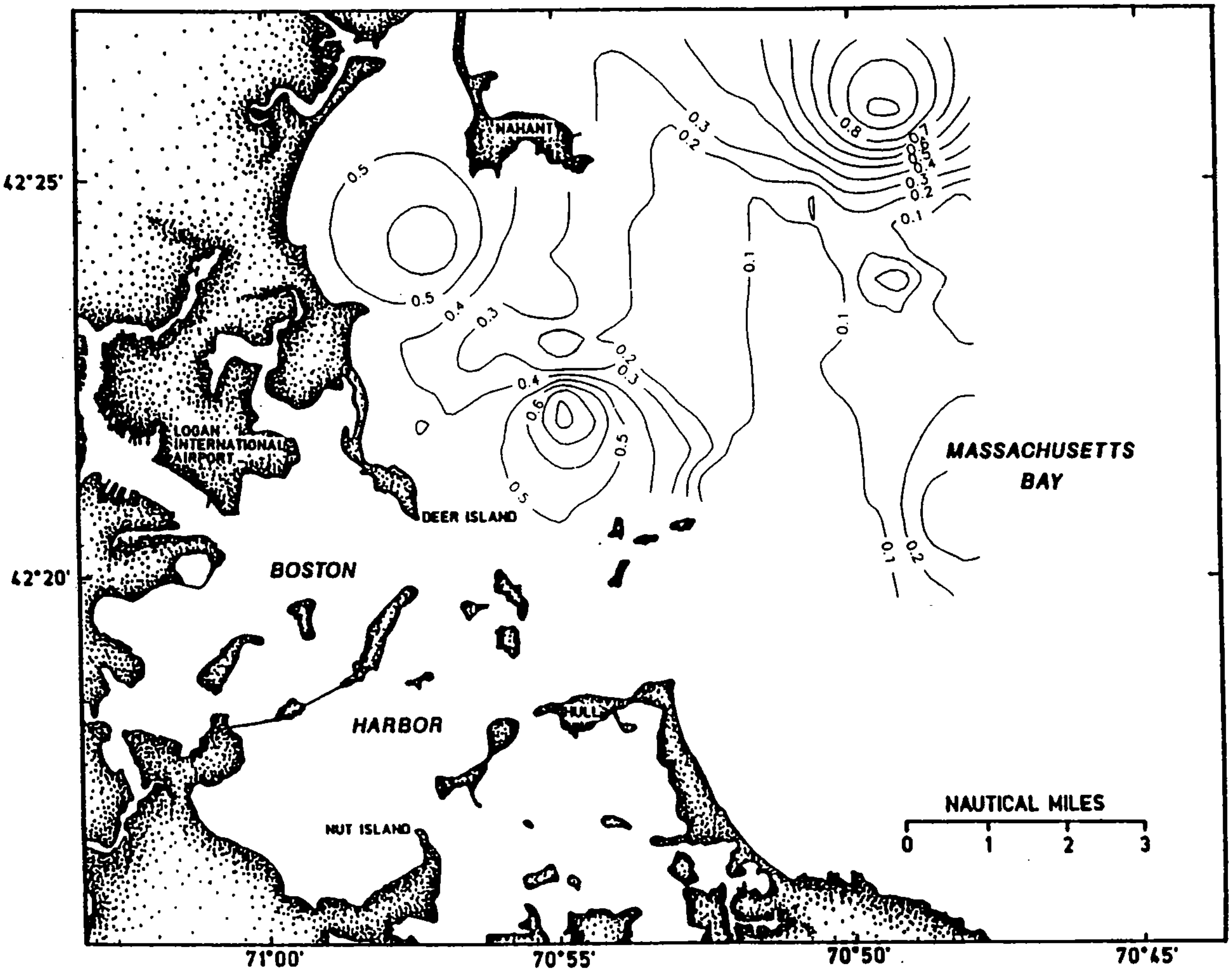


Figure 3-2. Contour plot of contaminant concentrations near the new MWRA offshore outfall (modified from Shea *et al.*, 1991).
b. Cadmium (μg/g, dry weight)

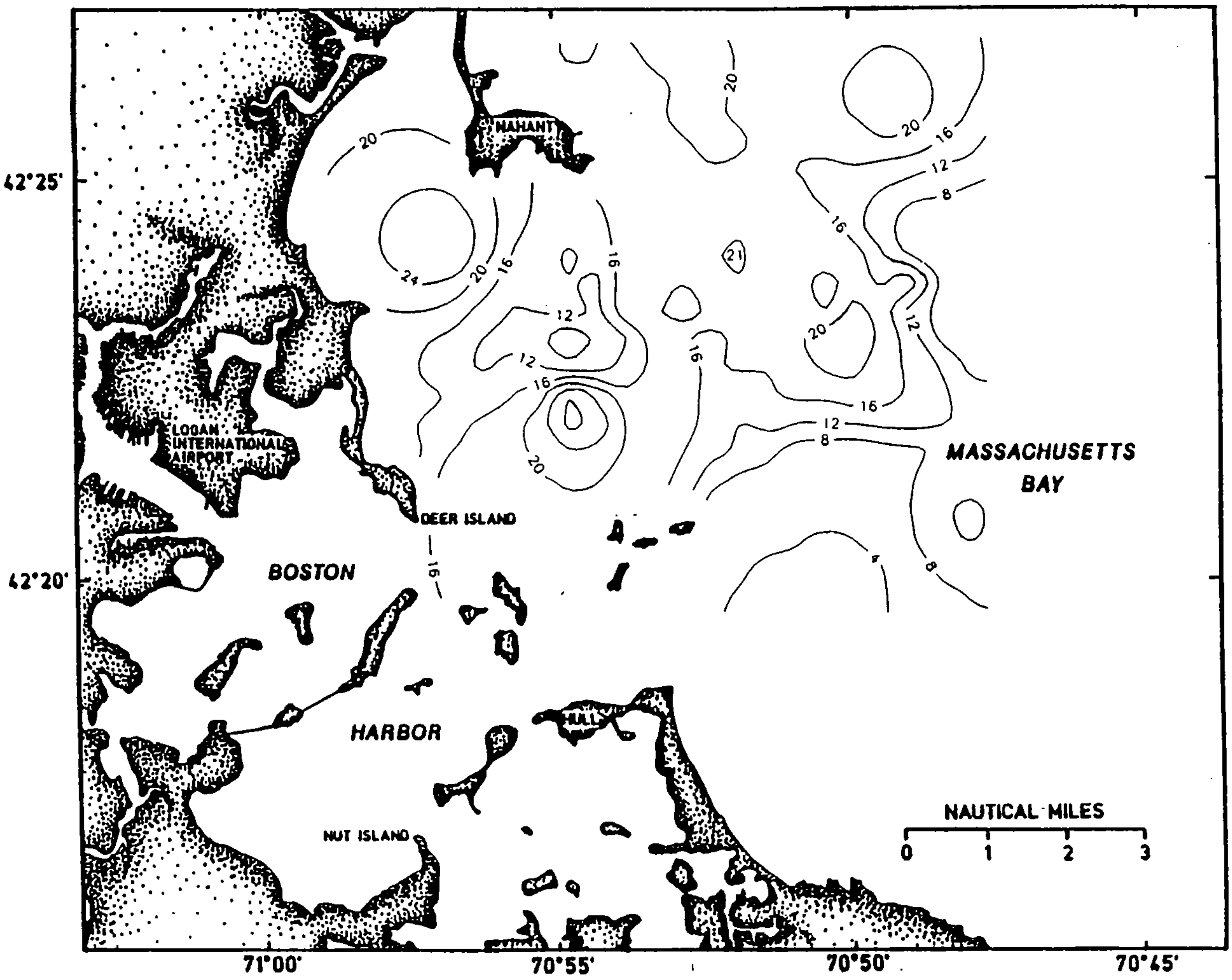


Figure 3-2. Contour plot of contaminant concentrations near the new MWRA offshore outfall (modified from Shea *et al.*, 1991).
c. Copper (μg/g, dry weight)

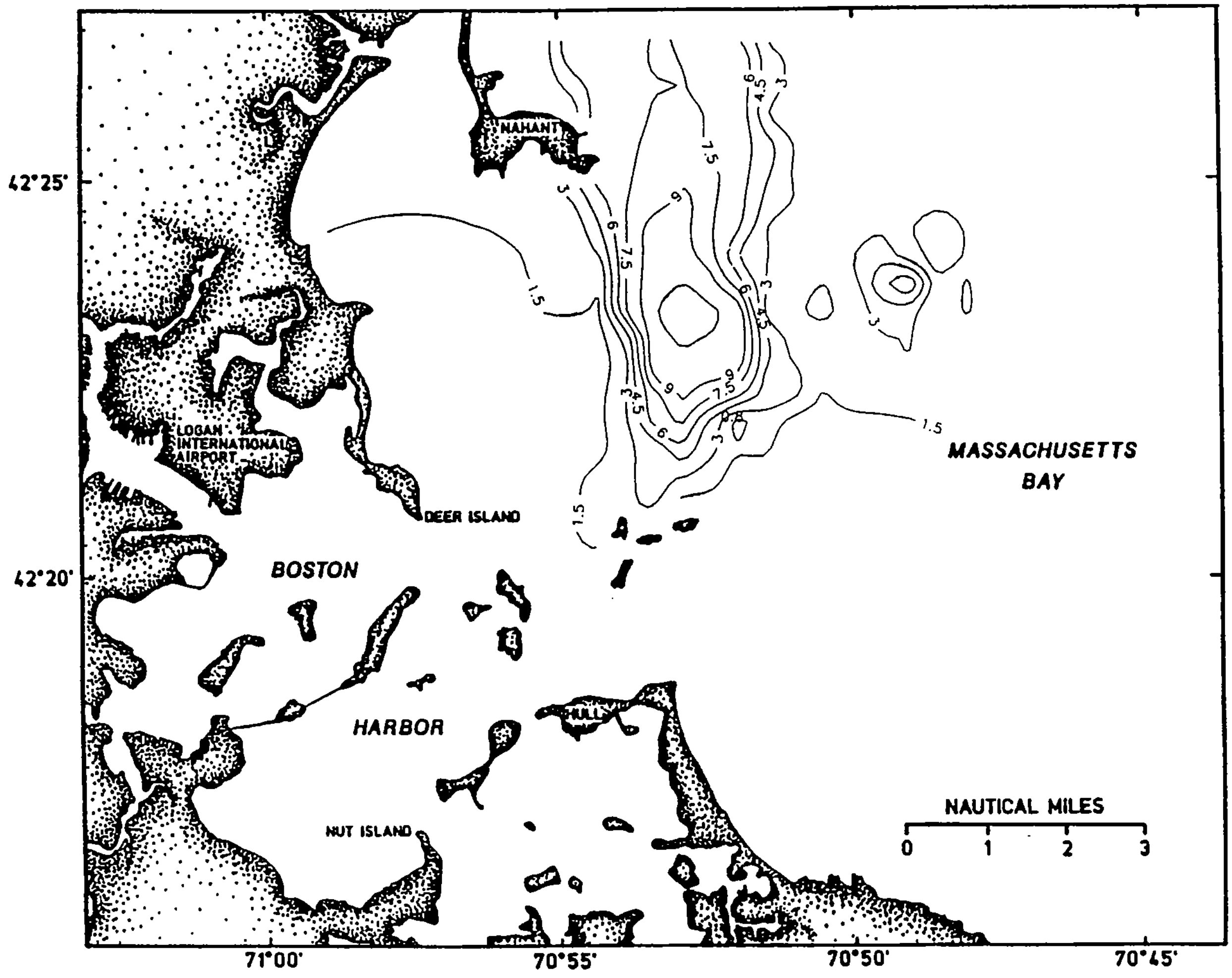


Figure 3-2. Contour plot of contaminant concentrations near the new MWRA offshore outfall (modified from Shea *et al.*, 1991).
d. Polycyclic aromatic hydrocarbons ($\mu g/g$, dry weight)

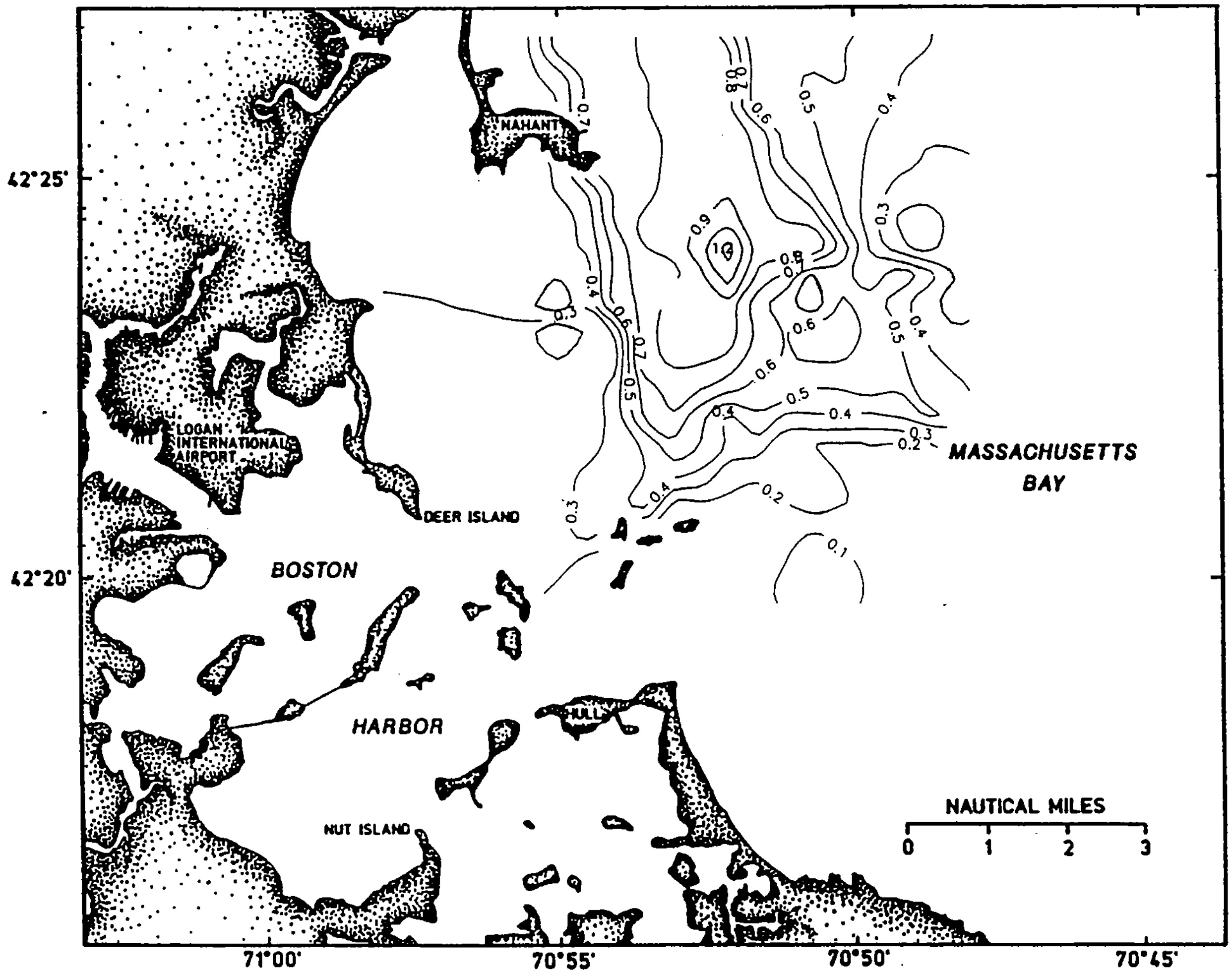


Figure 3-2. Contour plot of contaminant concentrations near the new MWRA offshore outfall (modified from Shea *et al.*, 1991).
e. Total organic carbon (%)

From present evidence, we can only speculate that the distributional pattern is determined by variation in the relative dominance of two factors: source strength vs. *in situ* environmental processes. Nevertheless, at the extreme of virtual "control" by environmental variability, one might expect great chemical-specific variability in dispersion of chemicals and a nonuniform pattern that mimics the present diversity of chemical distributions. At the other extreme, complete "control" by a source would be most likely to be evident very close to the source. The most highly correlated distributions of source chemicals might be expected very close to the diffuser, where the influence of the source strength is strongest and environmental processes play a lesser role in partitioning. With distance, some selective source strength diminution (across chemicals, due to their degree of particle-reactivity and removal from the water column) becomes more probable. If so, some chemicals will likely be better indicators for farfield transport than others, and chemicals that are unique to the discharge (e.g., linear alkyl benzenes) may have great utility in clarifying these processes.

It is worth noting here that the SEIS particle-settling rates and chemical accumulations in sediments predict that only about 10% of the diffuser-discharged chemicals will be retained in surrounding sediments up to spatial scales nearly the size of the Harbor. The rest will remain suspended in the water column or be transported to depositional sites in the "farfield," including outside of the Massachusetts and Cape Cod Bays region.

3.2.2 Chemical Removal Rates in Seawater

Variability due to *in situ* environmental processes relates to the behavior of chemicals in seawater and how much and how fast they may be removed via particle adsorption and deposition to the benthos. As discussed above, there is a spectrum from very particle-reactive chemicals to those that exist primarily as dissolved forms in seawater. The half-times for chemical removal to sediments can vary as a function of water depth, temperature, concentrations of suspended solids and dissolved organic compounds, salinity, flocculation processes, plankton productivity, benthic filter-feeding, particle resuspension rates, etc. For example, most chemicals in riverine freshwater solution, when mixed with seawater, will undergo a phase change from dissolved to particulate form, some fraction of which may then rapidly settle from solution (Santschi *et al.*, 1983).

The farfield transport and fate modeling in the STFP/SEIS included a provision for particle removal of some organic contaminants, wrapped into an all-process-inclusive "decay" time. We will not fully assess whether the decay category for each organic constituent is realistic for the specific conditions at the outfall, but this concern is just one of many uncertainties, addressed below, that could modify the projected sediment concentrations. However, all metals were assumed to be "conservative." This is clearly not the case for all metals within time frames appropriate for consideration of the loading from the future outfall (less than 2 weeks). Omission of decay processes is a principal limitation to the reliability of SEIS projections as a site-specific prediction (as opposed to a comparative siting exercise; see Section 2). Assuming conservative behavior for some metals (most probably Pb, Hg, Cr, and perhaps As) would overestimate water column concentrations and underestimate the load carried rapidly to bottom sediments. Conversely, more conservative trace metals (Cu, Cd, Ni, Zn) are less affected by these processes and thus, for these metals, the model used in the STFP/SEIS is probably more accurate.

3.2.3 Post-Depositional Redistribution of Contaminants

We often view the sediments as a long-term integrator that records some of the activity occurring in the overlying water column. However, a feature becoming increasingly recognized is that some areas can store appreciably more of this record than others, and over the long term, there are bottom sites that tend to focus particles and their associated contaminants. This is confounding to budgeting chemical fate in Boston Harbor (cf. Gallagher *et al.*, 1990). Given the environmental heterogeneity of Massachusetts Bay, just described above, there can be little doubt that redistribution of sediments occurs, especially between summer and winter (stratified and unstratified conditions). Various workers (e.g., Shea *et al.*, 1991) have discussed certain areas as hydrodynamically "low kinetic energy areas" where fine silt-clay particles and organic material may focus. Post-depositional transport can occur through bottom-current transport of physically or biologically resuspended particles or bedload transport; regardless of mechanism, this transport is a major *in situ* environmental force (see Section 3.2.1 above) to influence the distributional patterns of chemicals across the seabed. Omission of this process surely could be misleading by suggesting a spatial uniformity in the form of a smooth (normal) distribution of contaminants that will not be fully realized even if, on average, a spatial gradient is observable.

3.2.4 SEIS Modeling and Associated Uncertainties

On the general level, omission of post-depositional transport creates an obvious limitation in the use of the SEIS modeling as a compelling and supportable prediction. On a more specific level, omission of removal terms for metals, many of which have removal half-lives well within the 20-day time frame used for the most rapidly "decaying" constituents modeled in the SEIS, is also a serious deficiency. It is not sufficient to evaluate the effect of process inclusions upon the predicted sediment concentrations, however, because this would assume no variance from the assumptions used in the exercise. These assumptions are numerous, have a bearing on the prediction, and would be borne out by a sensitivity analysis of the model. Here, we will simply point out some principal assumptions and qualitatively suggest how slight modifications could alter the predictions; we restrict the analysis to a qualitative form because, as was true during development of the SEIS, there is little information to justify one assumption over another. The possible assumptions that we suggest are untested, unverifiable, or constant when they should be varying; as a whole, they do not give one great confidence in the projections as a site-specific prediction. On the other hand, the assumptions used in the STFP/SEIS modeling were generally the most conservative (at least with respect to the water column) and thus may form a reasonable upper bound of impact.

The model used to simulate sediment concentrations was relatively straightforward (EPA, 1988). Solids from the outfall were allowed to settle, and classes of different settling rates were apportioned, based on previous studies, to appropriate fractions of the total load of solids. The fraction of each contaminant associated with the solids was calculated based on theoretical grounds. Input solids, with their calculated contaminant concentrations, settled out from the water along trajectories essentially determined by the depth of water (whole water column or whole minus a 15-m superpycnocline layer), tidal oscillation, and net drift rates. Subsequently the resultant pattern of projected distribution (Figure 3-3) reflects a tidal elliptical pattern with major and minor axes and peak deposition grading from the foci to the outer edges, with slight distortion of a perfect ellipse due to net drift.

As an aside, note that only a small fraction of the particles in all simulations settle within this ellipse (Figure 3-4); about 40% of the solids introduced in primary effluent are assumed *never* to settle within all of Massachusetts Bay. Indeed, the simulations for primary and secondary effluents over 6-month or 5-year time frames usually resulted in a small fraction (perhaps 9 to 15% of the loaded solids) being retained in sediments within an area smaller than Boston Harbor. If reliable, this estimate of deposition, based on knowledge of effluent-particle gravitational settling rates, would seem to point out a basic difference between Boston Harbor, with its apparent high particle-retention efficiency (Section 2.3), and the more dispersive character offshore, even under relatively quiescent stratified conditions. Everything else being equal, offshore predictions of contaminant concentrations should, *at maximum*, be similar to the most depositional environments in Boston Harbor, but on the average (i.e., the spatial scale of the whole Harbor) should be much less offshore. SEIS model predictions of maximum sediment contaminant concentrations accumulated over only 5 years (versus decades or a century in the Harbor) generally exceed the average and, in some cases, even the maximum for Boston Harbor (MWRA, 1988). Such high predictions are particularly surprising in view of the lower background sediment values and the suspected deep mixing that may be present in parts of the Bay near the future outfall (Bothner *et al.*, 1990). In addition, the solids loading from the new primary treated effluent will be less than half of the present discharge, and secondary effluent will have about one-third of the new primary effluent solids. Thus, everything else is not equal, and — contrary to the STFP/SEIS model — the sediment accumulations near the future outfall should be significantly less than the most depositional areas of Boston Harbor.

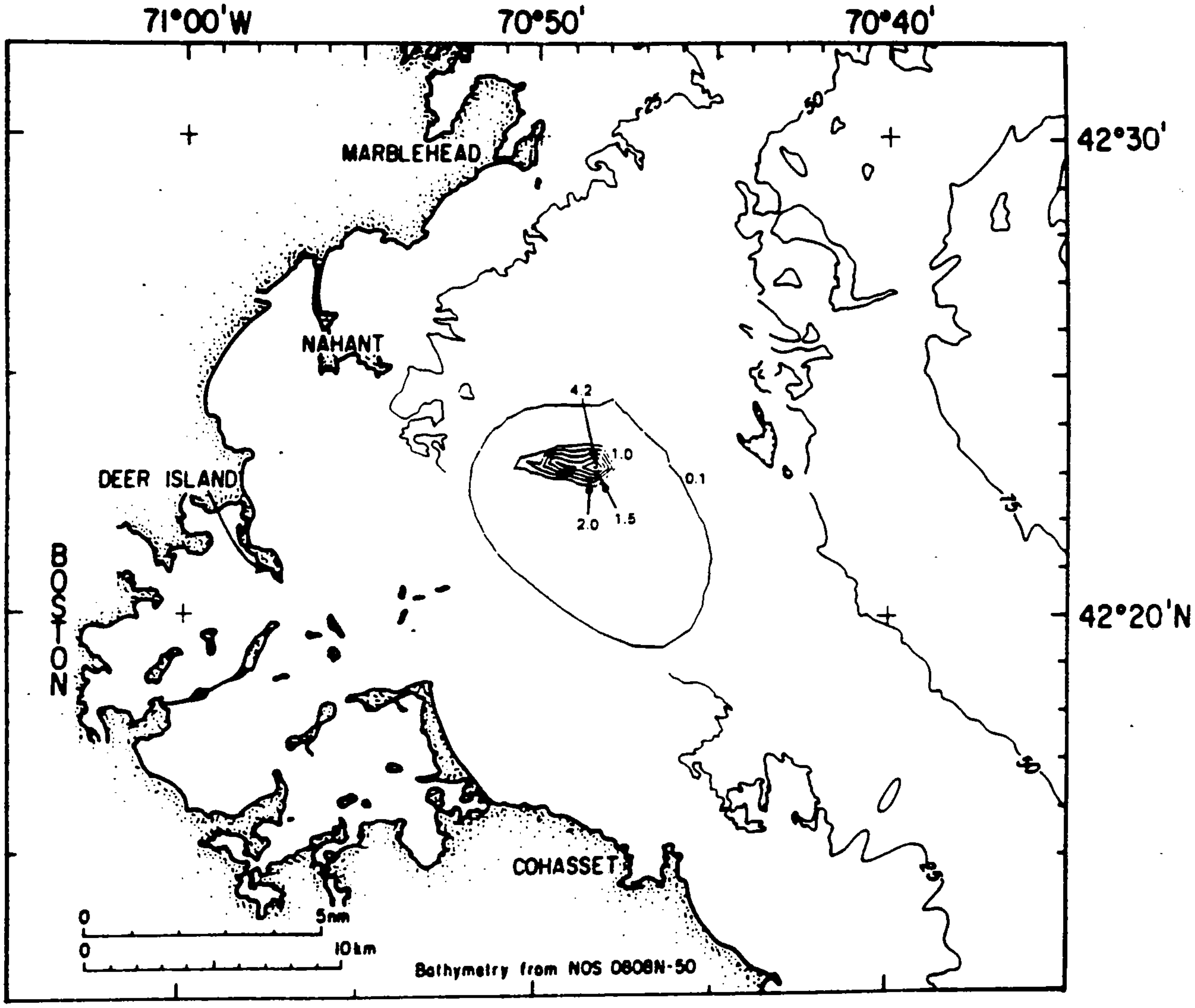


Figure 3-3. Sedimentation rates (g/m^2/day) predicted in the SEIS: Site 5, primary treatment, stratified conditions (from EPA, 1988). Average sedimentation rate in Boston Harbor is about 2.5 g/m^2/day.

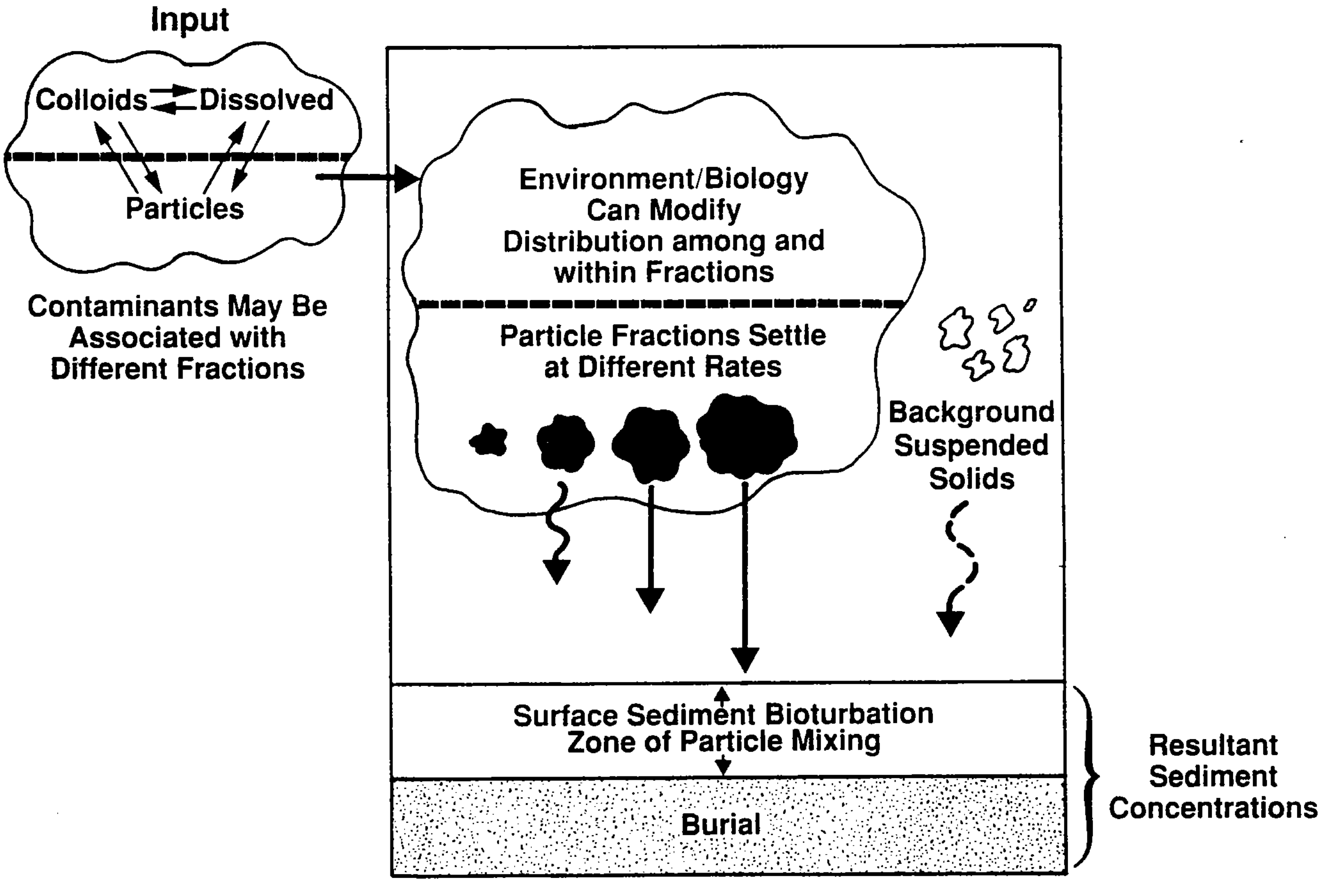


Figure 3-4. Contaminants in the effluent discharge can be associated with three different fractions (dissolved, colloidal, and particulate), all of which can have very different rates of transport away from the diffuser and to the bottom sediments. Even particles of different size fractions can behave quite differently, with larger particles settling quickly and smaller particles never settling within Massachusetts or Cape Cod Bays. Smaller particles also can be enriched in contaminants relative to larger particles. Thus, smaller particles can have a very small contribution to total solids loading, but a significant contribution to contaminant loading. Once particles settle, bioturbation and settling of less contaminated background solids can dilute the resulting concentrations in the bottom sediment.

Next in the model, contaminant concentrations of the outfall-discharged solids are mixed into sediments. Fallout of "natural" background solids and their associated contaminants is also added to the outfall load and mixed with the 3 cm of existing background surface sediments (with assumed average contaminant burden). An alternate simulation simply added the outfall-generated deposition to the presumed background deposition to simulate settling to a hard-bottom area initially clear of settled particles.

In Figures 3-4 and 3-5, we note various possible effects of altering the assumptions used in modeling. For example

- If we assume that contaminants are more strongly associated with finer particles or colloids, the fact that smaller particles can be transported farther results in a broader dispersion of contaminants. This is a mechanism that can partially explain the relatively high contaminant export from Boston Harbor. This is a testable assumption (by simply measuring contaminants in size fractionated suspended particles) that has not been tested.
- If we change the depth of sediment mixing, the rate of background sedimentation, and the processes that remove particles, we also change our hypothetical distribution of contaminants.
- If we assume spatial variability in any model parameter (e.g., biological mixing or particle focusing), we produce a patchy, rather than a smooth, distribution of contaminant concentrations.

It is now possible to add up these effects to say the predictions overestimate or underestimate concentrations of a particular contaminant. In Figure 3-5 we illustrate the essential features of the sediment deposition model, and in Section 4.3 we suggest where improvements in knowledge or assumptions would aid in the predictive capability. Figures 3-4 and 3-5 are intended to portray a cumulative uncertainty suggesting that the modeling results were clearly a very conservative (i.e., protective) estimate and, therefore, there is little reason to perform further simulations on sediment contamination, other than to better quantify the modifications suggested above. If additional modeling to simulate sediment contamination in highly depositional areas was required, more information on these environmental processes might be required.

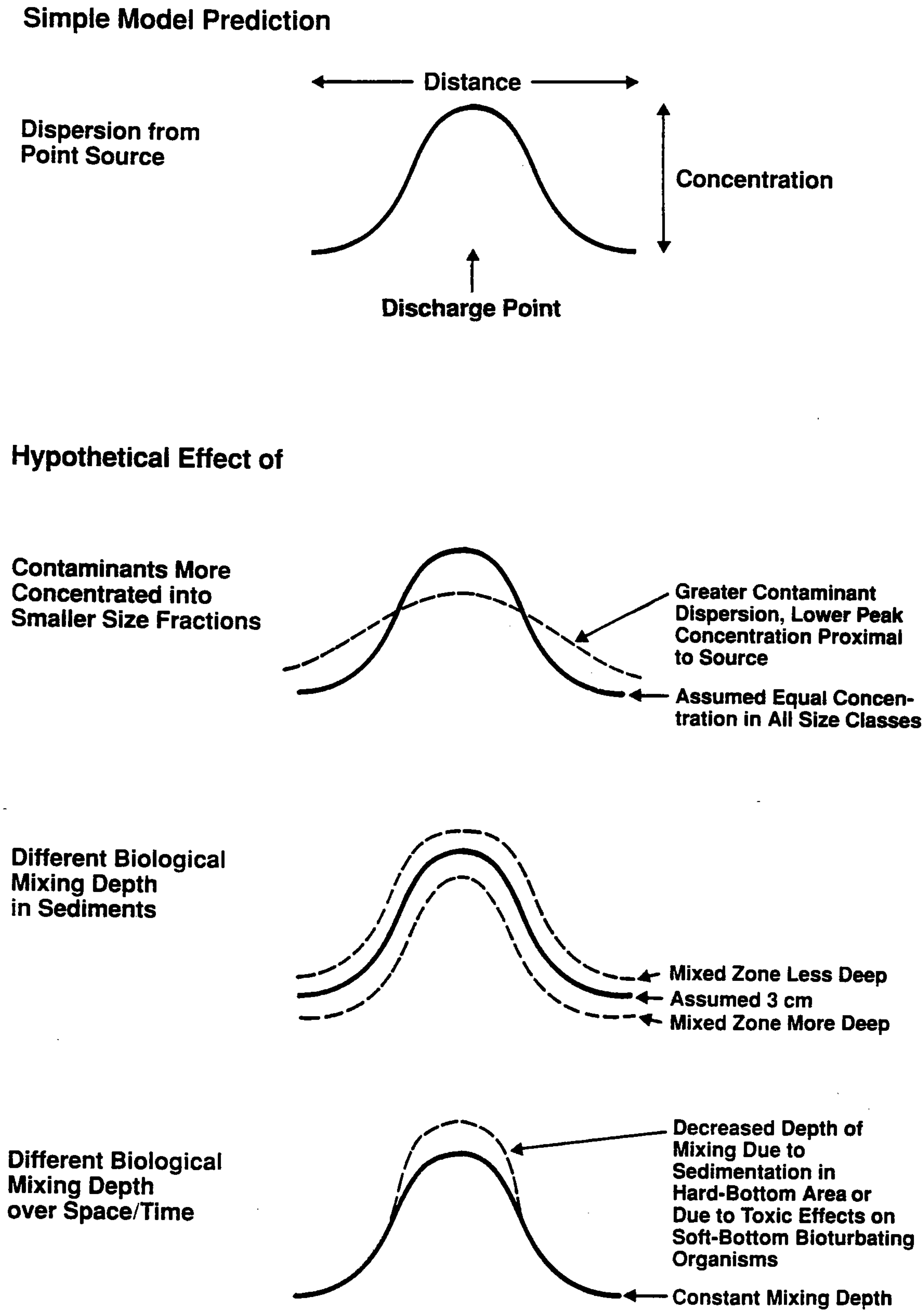
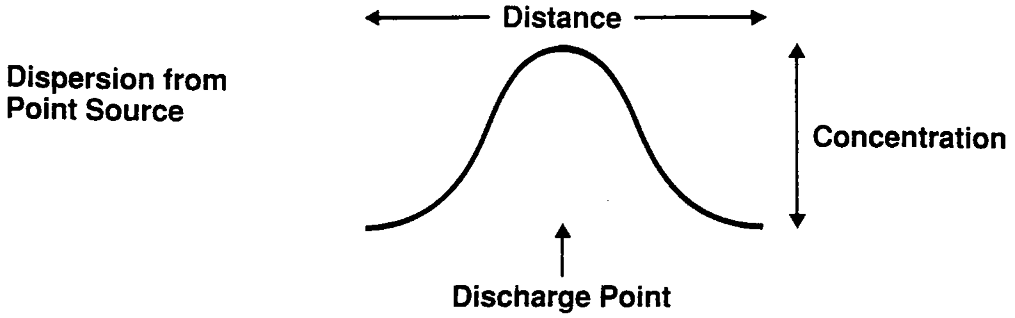


Figure 3-5. Simple model prediction for concentration of an effluent constituent in bottom sediments near the future outfall is shown (top) along with the hypothetical effects (dashed lines) of modifying the assumptions used in the SEIS and STFP models (solid lines).

Simple Model Prediction



Hypothetical Effect of

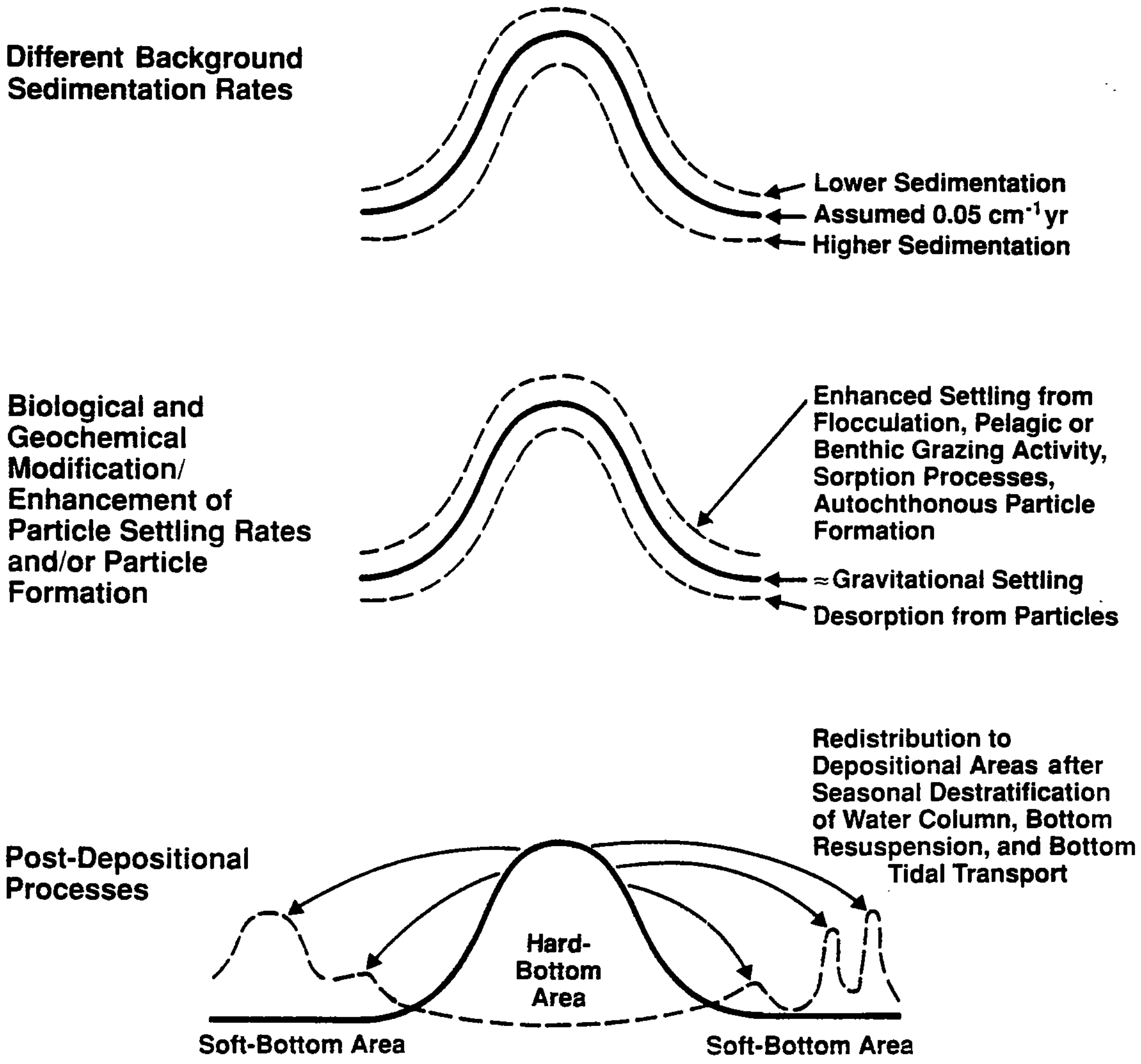


Figure 3-5. (continued) Simple model prediction for concentration of an effluent constituent in bottom sediments near the future outfall is shown (top) along with the hypothetical effects (dashed lines) of modifying the assumptions used in the SEIS and STFP models (solid lines).

3.3 EXPECTED CHANGES RESULTING FROM MOVING THE OUTFALL OFFSHORE

Boston Harbor, as an apparently inefficient chemical trap, currently functions as a strong source of contaminants to Massachusetts Bay. As a semi-point source to Broad Sound and westernmost Massachusetts Bay, the Harbor's export signal is rapidly diluted with distance into the Bay. This present source will diminish when effluent is diverted to the future offshore site. At present, the load that reaches the future outfall area is diffuse because of active volumetric mixing once outside the Harbor. When the ocean outfall is in operation, the load will become more of a direct offshore point source. This situation cannot be characterized as a simple trade-off of Massachusetts Bay (at the large scale) for Boston Harbor, because the Bay now receives much of the load it will receive in the new case. Clearly, the degree of export from the Harbor is the crux of the issue here.

In the new case, the length of the diffuser (2 km), tidal excursion, greater volume dilution (due to water depth), and greater mixing and advection (due to offshore currents) will all serve to "diffuse" the new "point source." If technical improvements raise effluent quality by removing an additional 10-30% of contaminants (about the minimum for primary treatment), then the new wastewater treatment plant would be functioning as Boston Harbor apparently is now, because retention of many elements is estimated to be about 10 to 30% (Table 2-2). Of course removal efficiencies for many contaminants could be as high as 85% after secondary treatment, and over the years Massachusetts Bay has been receiving not only poorly treated effluent, but also raw sewage sludge.

Some qualitative expectations of change due to simply moving the outfall offshore are given below.

- As in the Harbor, one could expect only a small portion of the load to be retained in nearby sediments, but some areas of elevated concentrations could be seen.
- Water column contaminant concentrations just outside the mixing zone are expected to be less than now seen in the Harbor. Predictions here suggest that water quality criteria will rarely, if ever, be exceeded in the water column, and then only immediately near the outfall. Contaminant concentrations should decrease throughout most of western Massachusetts Bay and Cape Cod Bay.
- Most of the qualitative modifications that are described herein for water column modeling and sediment simulations would result in predicted concentrations lower than those offered in the STFP/SEIS models. This must imply greater export from the local outfall area and dispersion throughout Massachusetts Bay, a situation that apparently exists now with the present outfall.
- Local effects could still occur, and some select areas could experience contaminant loads slightly higher than present. On a wider scale (i.e. Massachusetts and Cape Cod Bays), however, these consequences are no more likely than at present.

- As a final prediction, over the long term we expect a strong modification of the predicted smooth elliptical gradients around the diffuser, particularly for the sediments. Post-depositional processes will disperse material and create a more patchy distribution.

As discussed above, improvements to effluent quality also will have a dramatic effect on lessening potential impact to Massachusetts and Cape Cod Bays.

4.0 RECOMMENDATIONS FOR IMPROVEMENT OF FURTHER MODELING EFFORTS

Any model to predict chemical transport and fate of materials discharged at the future offshore outfall requires reliable data for a number of critical points. The foundation of the modeling effort is an adequate description of the physical oceanography, including time-varying vertical and horizontal processes. These processes can be classified by their time-space scale, including short-term variability (less than a day), intermediate-term variability (day to week, also week to month), and longer-term variability (greater than a month, also interannual). In the following description of critical information needs, we assume that these physical processes can be adequately described (see Bothner *et al.*, 1990 and Signell, 1992).

Critical points for chemical prediction can be divided into three semi-independent groups. The first group involves characterization of loading quantity and quality, the second deals with water-column events, and the third focuses on sediment processes. The points that need clarification in each group are listed in Table 4.1.

4.1 CONTAMINANT LOADING

4.1.1 Effluent Chemical Quality and Flow Rate

MWRA's effluent quality has improved dramatically over the last decade. Even in 1990, a number of contaminants, including some metals and organic compounds that have been highlighted by the SEIS as concerns, were often below detection limits using standard EPA methods. More sensitive chemical methods are available and would improve the reliability of loading quantity estimates. This has been clearly demonstrated for organic contaminants by Shea (1992); see Figure 2-4. This study also showed that, for most contaminants, short-term variability (over 1 week) is relatively small; similar measurements must be made over a longer time frame (year) to estimate long-term variability. With continued improvement in effluent quality expected through the 1990s, the need for better chemical characterization of effluent will become more critical. Any calculation of environmental burden due to a discharge begins with, and rests upon, the reliability of an input estimate.

4.1.2 Dissolved vs. Particulate Fractions in the Effluent

Evidence suggests (e.g., Figure 2-5) that local retention vs. farfield transport of a contaminant may be dictated in part by how much of the chemical in the effluent is associated with the dissolved fraction and how much is associated with the particulate fraction. Numbers on these proportions are generally not available for MWRA effluent. As a key to a major question on scaling and contaminant dispersion,

Table 4-1. To Improve Model Predictions of Contaminant Transport and Fate, Information Is Needed on Critical Points in Three Categories.

Category	Critical Point
Contaminant Loading	• Effluent contaminant concentrations and flow rate • Dissolved vs. particulate (and possibly colloidal) fractions in the effluent
Water Column Processes	• Partitioning between dissolved and particulate fractions of the effluent as it initially mixes with seawater • Rates of contaminant removal from the water column by particle transport and pelagic-benthic interactions • Relationship between particle sizes and contaminant burdens, with respect to settling rates
Deposition and Related Processes	• Variability in bioturbation depths and mixing rates as a function of depth within the sediment • Spatial and temporal variability in rates of background sedimentation • Post-depositional particle transport processes and secondary alteration of contaminant burdens on transportable solids (including material associated with hard-bottom areas as well as soft-bottom areas) • Significance of biofiltration and biodeposition in modifying chemical fate and transport

information on fractionation would be invaluable. The significance of colloidally bound contaminants on transport and bioavailability needs further consideration.

4.2 WATER COLUMN PROCESSES

4.2.1 Partitioning between Dissolved and Particulate Fractions of the Effluent as It Initially Mixes with Seawater

For modeling water quality, the SEIS had no explicit provision for conversion to particles once the effluent was in the environment, nor for any fractionation of contaminants to different size classes of particles (a constant concentration was implicitly assumed). These are processes that could have significant impacts on predictions of nearfield water quality and horizontal and vertical transport rates. The assumptions should be examined and data provided to improve them as needed.

4.2.2 Rates of Contaminant Removal from the Water Column by Particle Transport and Pelagic-Benthic Interactions

In the STFP/SEIS modeling, removal terms were included for organic contaminants, but not metals. As discussed, this is a deficiency that may create the impression of a water column more burdened with contaminants than will be the case. Because removal terms vary with environmental conditions including temperature, etc., analysis might also examine the sensitivity of predictions to some variation in constituents "decay" time or removal rate. Additionally, there is no provision in the present model for enhanced removal due to stimulation of water column biogeochemical processes that could modify rates of removal. For example, enhanced phytoplankton production and biomass (Kelly, 1991) could be followed by enhanced pelagic grazing pressure and fast removal by zooplankton fecal pellets. Such a process is probably more important at a deeper offshore site than it is in a shallow embayment like Boston Harbor, where benthic grazing activity can enhance removal rates. In essence, the presumption that "background" sedimentation continues unaltered in spite of water quality changes (that could have many causes — the outfall, global climate, etc.) may be too simplistic.

Removal terms, plus the expectation of additional partitioning of most elements to particles once mixed into the environment, would suggest that SEIS projections for water concentrations could be too high. As shown in Section 3, an SEIS model overestimate seems indicated where reasonably suitable environmental data are available for comparison to the model.

4.2.3 Relationship between Particle Sizes and Contaminant Burdens, with Respect to Settling Rates

The effect of contaminants being selectively associated with certain kinds and sizes of particles may not be uniformly important and may vary substantially from chemical to chemical. Thus, the qualitative impact of such a process on present water quality predictions is not known. The ramifications for spatial scale of transport seem significant and can be assessed.

4.3 DEPOSITION AND RELATED PROCESSES

4.3.1 Variability in Bioturbation Depths and Mixing Rates as a Function of Depth within the Sediment

Bioturbation of background sediments with new deposition generally dilutes the chemical signature of new outfall-generated sedimentation. This feature was well recognized and explicitly considered in the SEIS modeling. However, the presumption that a fixed 3-cm mixing depth is appropriate over the spatial frame of enhanced sedimentation is questionable. Recent and ongoing studies (Bothner *et al.*, 1990) suggest that mixing may sometimes occur to much greater depths (tens of centimeters), although one expects the intensity to be a function of depth. Because there is rather extreme variability in the geological nature and present background contamination in sediments surrounding the future outfall site (Section 3), a rather heterogenous distribution of mixing depths and rates must also be expected. In principle, to project even a "maximum/minimum" range within the immediate farfield of soft-sediment sites requires a reasonable description of the range of variability.

4.3.2 Spatial and Temporal Variability in Rates of Background Sedimentation

In the model, the mixture of natural sedimentation with outfall-generated sedimentation is dominated by the background deposition as one approaches the "edge" of the elliptical area of projected enhanced deposition. Thus, background rates become critical to the projection of the sediment inputs as one moves further afield. The model assumed that the background rates for specific contaminants were constant throughout the area, but if this were strictly true over longer time frames, the environment would not be so heterogeneous.

The contaminant concentrations used for background sedimentation in the model were based on the particle burdens observed in suspended particles. One problem with these values is that the measurements of contaminants associated with suspended particles in the water column around the site were very few. This is a possible source of error because variability — both seasonal and short-term in response to rapid biological events (e.g., Hunt and Smith, 1983) — is probably high. Perhaps more importantly, the bulk of particles in suspension are not settling to the bottom, and it is well known that particles that do fall (e.g., into sediment traps) do not necessarily have the same chemical signature as the bulk suspended solids. In short, at the margins of predicted dispersion of outfall chemicals, good knowledge of "background" contaminant sedimentation is a key factor.

4.3.3 Post-Depositional Particle Transport Processes and Secondary Alteration of Contaminant Burdens on Transportable Solids

Post-depositional processes as agents for redistribution were discussed extensively in Section 3. Their significance for the long-term fate and ultimate patchiness of contaminant distribution cannot be overstated. A primary emphasis here involves the seasonal cycle of stratification/mixing of the water column, and a projected dusting of outfall material on many areas that, over the time frame of the annual cycle, are not the final resting places of the toxicants. If longer term predictions are to be attempted, it will be necessary to know the content of material settling in the nearfield when bottom waters are relatively stagnant. An understanding of the chemical signature of this material and its *in situ* "digestion" during this period would aid in following the fate of contaminants secondarily transported to accumulating sediments, because this new "source" material for farfield long-term depositional sites will be different than the particle load from the effluent itself.

4.3.4 Significance of Biofiltration and Biodeposition in Modifying Chemical Fate and Transport

Finally, a note to emphasize that benthic filter-feeding organisms, depending on their density and filtration capacity, have potential to enhance the rate at which solids are removed from the water column. Furthermore, much of the filtered mass is not incorporated into tissue but ejected as feces and pseudofeces, thus altering the original suspended particles to a form that is usually easily resuspended and transported. Some of the organisms found in hard-bottom areas are capable of filter-feeding, as are many soft-bottom forms. The extent to which these organisms can alter contaminant distributions via either removal or post-depositional transport is not addressed in the SEIS/STFP models for the future outfall site, and its importance to chemical fate may need to be evaluated.

4.4 ONGOING AND FUTURE WORK THAT ADDRESSES CRITICAL INFORMATION NEEDS

Much of the information that is needed for an improved understanding of the transport and fate of contaminants discharged from the MWRA outfall (information discussed above and summarized in Table 4-1) will become available in the near future through ongoing studies being conducted by MWRA, the Massachusetts Bays Program, and other research organizations. For example, detailed effluent characterization has begun (Shea, 1992) and will continue under the MWRA Effluent Outfall Monitoring Program (MWRA, 1991). Removal of contaminants from the water column by settling particles is the subject of two separate studies being conducted by the University of Massachusetts and the USGS, respectively (cf. Bothner *et al.*, 1990). The USGS study is a joint program with the MWRA that is also investigating background sedimentation rates, post-depositional particle transport processes, and bioturbation depths and sediment mixing rates. Most of these studies will continue under the MWRA Effluent Outfall Monitoring Program (MWRA, 1991). Information that is gathered during the baseline period of the monitoring program (1992 through 1995) could be used to further refine the predictions given in the STFP/SEIS and the predictions presented above in this report.

5.0 REFERENCES

Battelle. 1987. Benthic reconnaissance. Final report on an ROV survey of Massachusetts Bay. Submitted to Camp Dresser and McKee/Stone & Webster Engineering Corporation for the Massachusetts Water Resources Authority. iii + 57 pp. + Attachment A.

Battelle. 1990. Phase IV final report, National Status and Trends Mussel Watch Program. Collection of bivalves and surficial sediments from coastal U.S. Atlantic and Pacific locations and analysis for organic chemicals and trace elements. Report to Department of Commerce, NOAA, Ocean Assessments Division, Rockville, MD.

Battelle. 1991. Results of the ambient and municipal sample interlaboratory comparison study in ambient waters and discharges to New York/New Jersey Harbor. Final report prepared for the Environmental Protection Agency under Work Assignment 2-202, Contract No. 68-C3-0105.

Boehm, P.D., W. Steinhauer, and J. Brown. 1984. Organic pollutant biogeochemistry studies Northeast U.S. Marine Environment. **Part 1:** The state of organic pollutant (PCB, PAH, coprostanol) contamination of the Boston Harbor — Massachusetts Bay — Cape Cod Bay system: Sediments and biota. **Part 2:** Organic geochemical studies in the Hudson Canyon and Gulf of Maine areas. Final report from Battelle Ocean Sciences to the National Oceanic and Atmospheric Administration.

Bothner, M.H., C.M. Parmenter, A.B. Brown, and R.P. Signell. 1990. Studies of circulation and pollutant transport in Massachusetts coastal waters. Department of the Interior, Geological Survey, Open-file Rep. 90-328. 33 pp.

Brownawell, B.J. 1986. The role of colloidal organic matter in the marine geochemistry of PCBs. Ph.D. dissertation, Woods Hole Oceanographic Institution/Massachusetts Institute of Technology Joint Program in Oceanography, Woods Hole, MA.

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

Eganhouse, R.P., D.P. Olanguer, B.R. Gould, and C.S Phinney. 1988. Use of molecular markers for the detection of sewage sludge at sea. *Mar. Env. Res.* 25:1-22.

EPA. 1988. Boston Harbor wastewater conveyance system. Draft Supplemental Environmental Impact Statement. Vol. I + II. Environmental Protection Agency, Region I, Boston, MA.

EPA. 1989. Evaluation of continued use of the Massachusetts Bay Dredged Material Disposal Site: Draft Environmental Impact Statement. Environmental Protection Agency, Region I, Boston, MA.

Furness, R.W. and P.S. Rainbow (eds). 1990. *Heavy Metals in the Marine Environment.* CRC Press, Boca Raton, FL. 256 pp.

Gallagher, E.D., G.T. Wallace, and R.P. Eganhouse. 1990. A synthesis of phase-I biological and chemical studies to identify the impact of the Fox Point CSO before modification. Draft final report submitted to Executive Office of Environmental Affairs, Boston, MA. 107 pp.

Hunt, C.D. and D.L. Smith. 1983. Remobilization of metals from polluted marine sediments. *Can. J. Fish. Aquat. Sci.* 40:132-142.

Kelly, J.R. 1991. Nutrients and Massachusetts Bay: A synthesis of eutrophication issues. MWRA Enviro. Quality Tech Rpt. Series No 91-10, June 1991. Massachusetts Water Resources Authority, Boston, MA. 66 pp.

Kelly, K.E. and N.C. Cardon. 1991. The myth of 10^{-6} as a definition of acceptable risk. Proceedings of the 84th Annual Meeting of Air & Waste Management Association, Vancouver, B.C., Canada.

McCarthy, J.F., B.D. Jimenez, and T. Barbee. 1985. Effect of dissolved humic material on accumulation of polycyclic aromatic hydrocarbons: Structure-activity relationships. *Aquat. Toxicol.* 7:15-24.

Menzie, C.A., J.J. Cura, J.S. Freshman, and B. Potocki. 1991. Boston Harbor: Estimates of loadings. MWRA Enviro. Quality Tech. Rpt. Series No. 91-3, February 1991. Massachusetts Water Resources Authority, Boston, MA. 108 pp.

Menzie-Cura & Associates. 1991. Sources and loadings of pollutants to Massachusetts Bay. Draft report prepared for the Massachusetts Bays Program. Massachusetts Coastal Zone Management, Boston, MA. 240+ pp.

Moore, M.J., B.R. Woodin, and J.J. Stegeman. 1992. Liver pathology of winter flounder: Boston Harbor, Massachusetts Bay, and Cape Cod Bay — 1991. Final report submitted to the Massachusetts Water Resources Authority, Boston, MA. 62 pp.

MWRA. 1988. Secondary Treatment Facilities Plan. Massachusetts Water Resources Authority, Boston, MA. March 1988.

MWRA. 1990. The State of Boston Harbor: 1990. Massachusetts Water Resources Authority, Boston, MA.

MWRA. 1991. Massachusetts Water Resources Authority Effluent Outfall Monitoring Plan Phase I: Baseline Studies. Massachusetts Water Resources Authority, Boston, MA. 45+ pp.

MWRA. 1992. The State of Boston Harbor: 1991. MWRA Enviro. Quality Tech. Rpt. Series No. 92-3, January 1992. Massachusetts Water Resources Authority, Boston, MA. 83pp.

Santschi, P.H., D.M. Adler, and M. Amdurer. 1983. The fate of particles and particle-reactive trace metals in coastal waters: radioisotope studies in microcosms. Pp 331-349 in C.S. Wong, E. Boyle, K.W. Bruland, J.D. Burton, and E.D. Goldberg (Eds.) *Trace Metals in Sea Water.* Plenum Press, New York, NY.

Shea, D. 1988. Developing national sediment quality criteria: Equilibrium partitioning of contaminants as a means of evaluating sediment quality. *Environ. Sci. Technol.* 22:1256.

Shea, D. 1992. Data report on organic contaminants in Deer Island and Nut Island influents and effluents. Submitted to the Massachusetts Water Resources Authority, Boston, MA.

Shea, D., D.A. Lewis, B.E. Buxton, D.C. Rhoads, and J.M. Blake. 1991. The sedimentary environment of Massachusetts Bay: Physical, chemical, and biological characteristics. MWRA Enviro. Quality Tech. Rpt. Series No. 91-6, June 1991. Massachusetts Water Resources Authority, Boston, MA. 139 pp.

Sheng, Y.P. 1989. Predicting the dispersion and fate of contaminated marine sediments. Pp 166-177 in *Contaminated Marine Sediments — Assessment and Remediation.* National Academy Press, Washington, DC.

Signell, R.P. 1991. Applications and limitations of depth-averaged circulation modeling in Boston Harbor, Massachusetts. To appear in Proceedings of the 2nd Estuarine and Coastal Modeling Conference, Tampa, FL, M. Spaulding (ed.). November, 1991.

Sung, W. 1991. Observations on the temporal variations of dissolved copper and zinc in Boston Harbor. *Civil Eng. Pract.* 6:99-108.

Wallace, G.T., J.H. Waugh, and K.A. Garner. 1988a. Metal distributions in a major urban estuary (Boston Harbor) impacted by ocean disposal. Pp 67-78 in Wolfe, D.A. and T.P. O'Connor (eds.), *Oceanic Processes in Marine Pollution, Vol. 5, Urban Wastes in Coastal Marine Environments.* Kreiger Publishing Co., Malabar, FL.

Wallace, G.T., R.P. Eganhouse, L.C. Pitts, and B.R. Gould. 1988b. Analysis of contaminants in marine resources. Technical report prepared for Massachusetts Department of Environmental Quality Engineering and the Environmental Protection Agency, Boston, MA. 145 pp.

Wu, S.C. 1986. Transport of hydrophobic organic compounds between water and natural sediment. Ph.D. Dissertation. Dept. of Civil Engineering, Massachusetts Institute of Technology. 231 pp.

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