1996 Toxics issue review

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1996 Toxics Issue Review

submitted to

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1.0 INTRODUCTION

1.1 Overview

For the 1996 Toxics Issues Review (TIR), a detailed evaluation was made of the levels and trends associated with selected toxics (PCBs, pesticides, mercury) in the current and future Massachusetts Water Resources Authority (MWRA) outfall effluent. The TIR utilized currently available data and documents to present an updated evaluation of the potential sources (e.g., effluent, riverine, combined sewer overflow (CSO), atmospheric) of these toxics. The evaluations also compared background sources to predicted future water column concentrations following discharge. Finally, the TIR evaluated potential effect of these contaminants on endangered species living in the Bays. The focus of the TIR report on effluent toxics and potential ecological effects was designed to help support decision-making on the upcoming NPDES permit for the future outfall. Major topics which the 1996 TIR addressed are described further below.

1.1.1 Evaluation of Organic Contaminants

An evaluation of the selected organic contaminants present in the current and future outfall discharge was made. The organic contaminants examined included PCBs, DDT, dieldrin, heptachlor, and hexachlorobenzene. This evaluation consisted of the following subtasks:

- summarization of the current levels of organic contaminants in surface water in Massachusetts and Cape Cod Bay and comparison to historical levels;
- description and estimation of potential loading sources of organic contaminants including present outfall discharge, riverine loading, atmospheric deposition, and/or other sources; and
- prediction of potential organic contaminant levels in future outfall discharge and mixed surface water concentrations in Massachusetts Bay.

1.1.2 Evaluation of Mercury

An evaluation of mercury present in the current and future outfall discharge was conducted. The evaluation consisted of the following subtasks:

- summarization of the current levels of mercury in surface water in Massachusetts and Cape Cod Bay and comparison to historical levels;

- description and estimation of potential loading sources of mercury including present outfall discharge, riverine loading, atmospheric deposition, and/or other sources; and

- prediction of potential mercury levels in future outfall discharge and mixed surface water concentrations in Massachusetts Bay and comparison of existing background loads (i.e., non-point sources) to predicted loading.

1.1.3 Evaluation of Potential Bioaccumulative Effects

An evaluation of the potential impact of bioaccumulative contaminants present in the current and future outfall discharge on endangered species (i.e., northern right whale) was made. The bioaccumulative contaminants examined included PCBs, pesticides, and mercury; as well as PAHs and metabolites of concern. The evaluation consisted of the following subtasks:

- summarize the current database on tissue residues in marine mammals in Massachusetts and Cape Cod Bay and compare to historical levels;

- compare the trophic structure and feeding habits of northern right whale food chain to comparable marine mammals (i.e, beluga whales) and discuss possible implications regarding exposure to outfall contaminants; and

- using available predicted contaminant levels in future outfall discharge, estimate incremental tissue residue, describe potential ecotoxicological effects to northern right whales, and identify areas of uncertainty in the analysis.

Four principal investigators contributed to the 1996 TIR. Dr. Michael Wade (Wade Research, Inc.) characterized historical, current, and future levels of organic contaminants (e.g., PCBs, 4 pesticides) presented in Section 2.0. Dr. Windsor Sung (Sung and Associates) provided an evaluation of mercury in the MWRA outfall and receiving waters (Section 3.0). Dr. Michael Moore (Woods Hole Oceanographic Institute) evaluated bioaccumulation, tissue residue levels, and potential impacts to marine mammals, especially the Northern Right Whale. Dr. David Mitchell (ENSR) edited the TIR and summarized major findings (Section 5.0).

2.0 CHLORINATED PESTICIDES AND PCBs

2.1 Introduction

Employing all available data and documents, this section of the MWRA Toxics Review provides a comprehensive evaluation of the potential current sources (e.g., effluent, riverine, atmospheric) of selected chlorinated pesticides and polychlorinated biphenyls (PCBs) to Massachusetts and Cape Cod Bays, and compares background sources to predicted future water column concentrations following discharge. Specific topics addressed include:

- Summarization of current levels of chlorinated pesticides and PCBs in surface water in Massachusetts and Cape Cod Bays and comparison to historical levels;

- Quantification of potential loading sources of chlorinated pesticides and PCBs to Massachusetts and Cape Cod Bays including present outfall discharge, riverine loading, atmospheric deposition, and dredged material disposal; and

- Prediction of the effect of future outfall discharge of the mixed surface water concentrations of chlorinated pesticides and PCBs in Massachusetts and Cape Cod Bays using experimental data on concentrations of chlorinated pesticides and PCBs from MWRA pilot plant studies.

DDTs, banned for use in the United States since 1972, and chlordane, whose use effectively ended within the United States in 1983, have become useful indicators for assessment of the effectiveness of regulatory actions on ambient levels of chlorinated hydrocarbon pollutants in the coastal marine environment. Since 1971, in a manner similar to DDTs, PCBs have been subjected to increasingly rigorous regulatory limitations designed to reduce or eliminate their presence in the manufacturing streams in the industrial United States. Taken together, ΣPCBs, ΣDDT, and Σchlordane concentrations provide quantitative temporal data to monitor the effectiveness of regulatory action on these classes of environmental pollutants. Concentrations of dieldrin, aldrin, lindane, and Mirex are sometimes included to provide a more complete picture of environmental contamination from chlorinated pesticides.

For this evaluation, the following definitions for chlorinated hydrocarbons and PCBs were used: total chlordane was comprised of the arithmetic sum (Σchlordane) of heptachlor, heptachlorepoxide, *cis*-chlordane, and *trans*-nonachlor; total DDT was comprised of the arithmetic sum (ΣDDT) of the two parent DDT isomers p,p'-DDT and o,p-DDT, as well as all breakdown products p,p'-DDE, o,p-DDE, p,p'-DDD, and o,p-DDD; and total polychlorinated biphenyls (ΣPCBs) included the 20 individual PCB congeners

monitored by MWRA. Concentrations of dieldrin, aldrin, lindane, and Mirex were assessed individually, as appropriate.

2.2 Current Levels of Pesticides and PCBs in Surface Water in Massachusetts and Cape Cod Bays

Direct measurement of ambient concentrations of chlorinated pesticides and PCBs in seawater is challenging. Differing field approaches have been developed over the years to produce data that could be considered representative of ambient or background concentrations (see for example, Morris et al., 1977; Scura and McClure, 1975; deLappe et al., 1983; and others). In order to assess pollutant fluxes in marine water, researchers often employ indirect measurement of ambient seawater concentrations: such as polycyclic aromatic hydrocarbons in sediments (Shiaris and Jambard-Sweet, 1986; Gschwend and Hites, 1981; Larsen et al., 1986; Boehm et al., 1988; and others) or organic contaminants in intertidal/subtidal molluscs (Farrington et al., 1980; O'Connor, 1990; and others). Coupled with differences in study design and/or field sampling approaches, improvements in the analytical technology of gas chromatography have resulted in the availability of more precise and accurate chemical data beginning in the decade of the 1980s.

In recent years, using high volume field collection procedures and improved analytical technologies, there have been several attempts to measure ambient concentrations of various organic compounds (polycyclic aromatic hydrocarbons (PAHs), chlorinated pesticides and PCBs) in Massachusetts and Cape Cod Bays seawater (for example, Wade et al., 1987; Hunt et al., 1988, USEPA, 1992, Menzie-Cura et al., 1995). When coupled with data from the ongoing National Oceanic and Atmospheric Administration (NOAA) National Status and Trends Mussel Watch program, some existing data trends for ambient concentrations of selected organic compounds in Massachusetts and Cape Cod Bays seawater are evident. Some of the data from these efforts are useful; some data have evident contamination and were not very useful in the evaluation of regulatory approaches to marine pollution regulation.

2.2.1 Direct Measurement Programs

Over the past few years, there have been a number of attempts to directly measure seawater concentrations of chlorinated pesticides and PCBs in Massachusetts Bay. One outcome of these efforts is extremely clear; since the late 1980s, meaningful concentrations of selected chlorinated hydrocarbons have become increasingly difficult to determine. In addition, sample contamination and sample volume problems have hindered such work.

2.2.1.1 U.S. Army Corps of Engineers - Massachusetts Bay Disposal Site -Mid-1980s

Concentrations of selected pollutants, including total PCBs, were determined by the U.S. Army Corps of Engineers as part of the Massachusetts Bay Disposal Site (MBDS) Draft Environmental Impact Statement (DEIS) in support of the designation of an ocean disposal dredged material disposal site in Massachusetts Bay (U.S. EPA, 1992). In this effort, total PCBs were reported at 0.012 μg/L, a value which may have been representative of total PCBs in seawater in Massachusetts Bay during the mid-1980s. From the data, it is not clear what PCB congeners were measured in this particular effort.

2.2.1.2 MWRA Outfall Siting Project - 1987

Initial efforts to directly measure dissolved and particulate PCBs in Massachusetts Bay seawater at one near-shore station and one offshore station in April 1987 as part of the MWRA Outfall Siting Project were only marginally successful when high blank levels were reported by an analytical subcontractor (Wade et al., 1987). Consequently, the 1987 data were not adequately precise to use as the basis for regulatory decisions.

During July and August 1987, in conjunction with a bioaccumulation monitoring program conducted by MWRA at selected locations in Boston Harbor and Massachusetts/Cape Cod Bays, several whole seawater samples were collected and analyzed for selected chlorinated organics. Declining seawater concentrations with increasing distance from Boston Harbor were demonstrated (Table 2-1). Total chlorinated pesticides at a reference station (UM-8) in outer Massachusetts Bay could not be detected; PCBs in outer Massachusetts Bay seawater were measured at 1.15 to 3.2 ng/L, while PCBs in seawater at the MWRA ocean outfall site (Station 5) were measured at 2.9 ng/L.

2.2.1.3 MWRA Outfall Siting Project - 1988

Another effort to measure chlorinated hydrocarbon concentrations in seawater as part of the MWRA Outfall Siting Project was reported by Hunt et al. (1988). In February 1988, triplicate 100 liter seawater samples were collected and extracted; concentrations of chlorinated hydrocarbons were measured using NOAA Mussel Watch analytical procedures (both in the particulate and the "dissolved" phases in seawater) at two depths at a single station in Massachusetts Bay under non-stratified (winter) conditions. Analytical results were more acceptable than those achieved in April 1987.

Table 2-2 shows the concentrations of individual chlorinated hydrocarbons. Concentrations of ΣDDTs in the dissolved phase were in the range of 0.26 - 0.54 ng/L, and 0.17 - 0.21 ng/L in the particulate phase for surface waters at Station H1 (depth = 7 m). In the deeper waters at Station H1 (depth = 80 m), concentrations of ΣDDTs in the dissolved phase were in the range of 0.16 - 0.54 ng/L, and 0.25 - 0.60

ng/L in the particulate phase. Concentrations of Σchlordanes in surface waters (depth = 7 m) were in the range of 0.02 - 0.22 ng/L for particulate phase, and 0.012 - 1.15 ng/L in the dissolved phase. At 80 m, Σchlordane concentrations were in the range of 0.04 - 0.16 ng/L for the particulate phase and 0.34 - 0.86 ng/L in the dissolved phase. Concentrations of ΣPCBs (as determined by 18 individual PCB congeners) in the surface water were in the range of 1.2 - 2.6 ng/L for the particulate phase, and 2.0 to 5.0 ng/L in the dissolved phase. In deeper waters, concentrations of ΣPCBs (as determined by 18 individual PCB congeners) in deeper waters at Station H1 were in the range of 4.2 - 8.2 ng/L for particulate phase, and 1.7 to 4.4 ng/L in the dissolved phase. Concentrations were low, as expected, but measurable in 1988.

2.2.1.4 U.S. Environmental Protection Agency - Massachusetts Bay Disposal Site - June 1992

In June 1992, measurement of seawater concentrations of chlorinated hydrocarbons was continued by collection and analysis of whole seawater samples collected from three locations in Massachusetts Bay as part of the designation of the MBDS (Battelle, 1992). Because the amount of seawater extracted was too small to produce meaningful results (only 2.5 liters were extracted), concentrations of PCB congeners and chlorinated pesticides were estimated to be less than 10 ng/L for ΣPCBs and less than 5 ng/L for chlorinated pesticides. These reported estimates were found to be too high for the data reported. Accordingly, the individual analytical results from Battelle (1992) were re-examined and recalculated to provide a greater detail in the breakdown among the analytes reported (Table 2-3). Concentrations of individual chlorinated hydrocarbons as reported by Battelle were used to calculate actual values for selected parameters. No assumptions regarding data quality were made. Detailed results were consistent with reported less than values; however, more detailed data do provide comparable data with other studies and allow intercomparison of similar data. Mean values for whole seawater concentrations were ΣPCBs-2.5 ng/L; ΣDDTs-1 ng/L; ΣChlordanes-3.4 ng/L and Σpesticides-7 ng/L. The analytical results have to be considered as high-side estimates only. As confirming analysis of tentative peak identifications was not completed, the analytical results are assumed to represent an overestimation.

2.2.1.5 U.S. Environmental Protection Agency/Commonwealth of Massachusetts - Massachusetts Bays Program - 1992

Recently, a concerted effort has been made as part of the Massachusetts Bays program sponsored by the U.S. Environmental Protection Agency (U.S. EPA) and the Commonwealth of Massachusetts to quantify pollutant sources in Boston Harbor and Massachusetts Bay. Two programs were completed that provide direct data on concentrations of chlorinated hydrocarbons in Massachusetts Bay. The first program is the "Non-point Source Runoff/PAH Loading Analysis" and the second program is "Organic Loadings from Merrimack River into Massachusetts Bays." Concentrations of chlorinated hydrocarbons were measured as part of the Merrimack River program in Massachusetts Bay using high volume techniques that provided particulate material collected from 100 to 300 liter water samples. Concentrations of chlorinated

hydrocarbons in six samples from 5 locations collected in Massachusetts Bay are reported in Table 2-4. Results of chemical analyses showed that ΣPCBs in particulate matter were in the range of 0.04 - 0.56 ng/L; Σchlordanes was measured at 0.001 ng/L in one sample (all others were not detected); and ΣDDTs, HCB and gamma-BHC were not measured to the detection limit of the study. Dissolved fractions were not measured in this study.

2.2.1.6 Massachusetts Bay Seawater Samples - 1995

Newly-released seawater concentrations for selected chlorinated pesticides and PCBs in Massachusetts and Cape Cod Bays were provided to the MWRA (pers. comm., Shea 1997). Details of the sampling and analysis are not yet available. Seawater samples were collected at MassBay Buoy B, at Stellwagen bank, in western Cape Cod Bay, in the waters off Cape Anne, and in Georges Bank (Table 2-5). Both "particulate" and "dissolved"phases were collected using a combination of 0.7 mm quartz filter and solid phase extraction by XAD-2 resin. Concentrations of Chlordanes, DDTs, PCBs, and PAH were reported at each station. Replicate seawater samples, collected at MassBay Buoy B, showed that the relative standard deviation generally was less than 100% of the mean values of the replicate concentrations for most analytes.

Reported concentrations of selected chlorinated pesticides and PCBs appear to represent the lowest concentrations yet reported for Massachusetts/Cape Cod Bays seawater. Within Massachusetts/Cape Cod Bays seawater, the total (sum of dissolved and particulate) Chlordanes were determined to range from a low of 14.5 pg/L at the Cape Cod-West station to a high of 27 pg/L at MassBay buoy B. Total DDTs were reported to range from a low of 1.6 pg/L at Cape Anne to a high of 5.3 pg/L at MassBay buoy B. Total PCBs were reported to range from a low of 235 pg/L at Stellwagen Bank to a high of 449 pg/L at the Cape Cod Bay-West station. Concentrations of PAH were within the expected range of 2.6 to 7.4 ng/L (Wade, MassBays Program, 1995).

2.2.2 Indirect Measurement Approaches

2.2.2.1 NOAA Mussel Watch Bioaccumulation Monitoring Results

Since 1986, NOAA has conducted a national program designed to monitor concentrations of selected organic and metallic pollutants in bottom-feeding fish, shellfish, and sediments in the coastal waters of the United States. Results have documented declining concentrations of contaminants in numerous ecologically important areas of the United States, including decreasing concentrations of ΣDDT and Σchlordanes in bivalves located in several parts of the country (O'Connor, 1992). However, a similar clear cut trend is not observed at NOAA National Status & Trends Mussel Watch stations in Boston Harbor and the Massachusetts and Cape Cod Bays. Figures 2-1, 2-2 and 2-3 show the data trends for Σchlordanes, ΣDDT and ΣPCBs at four Mussel Watch stations. While individual trends indicate some reduction since

the early monitoring years of 1986-1987, a consistent decreasing concentration trend at any single monitoring station is simply not evident from these data.

Review of these data trends show that there may have been a slight decrease in the flux of ΣPCBs, Σchlordanes, and ΣDDTs in the coastal Massachusetts and Cape Cod Bays system. On the other hand, the delivery of ΣPCBs at Brewster Island, Dorchester Bay, Deer Island and Hingham Bay NOAA Status and Trends mollusc monitoring stations may be varying around a steady state amount over the time period of 1986 to 1993. Thus, no conclusions can be reached about the concentrations of PCBs as measured by tissue residues in the recent past. Similar trends are evident for Σchlordanes and ΣDDTs for the same time period.

2.2.2.2 MWRA Bioaccumulation Monitoring Results, 1987-1995

Beginning in 1987 and then running continuously from 1991 through 1995, MWRA has conducted a bioaccumulation monitoring (biomonitoring) program in Boston Harbor and Massachusetts Bay that has documented a declining annual burden of various organic pollutants delivered to the coastal marine system. Annual concentration data mirror bioaccumulation data provided by NOAA for the Boston Harbor and MassBay Mussel Watch program monitoring stations, or provide even more dramatic decreases in selected chlorinated pesticides and PCBs than seen in the NOAA Mussel Watch data for Massachusetts coastal areas (Table 2-6a-c).

Comparison among the yearly MWRA biomonitoring data must be made carefully because the duration of deployments were not identical; the 1987 deployment lasted for 30 days, 1991-1994 deployment lasted for 60 days, and the 1995 deployment lasted for 50 days. The blue mussels initially were collected from Barnstable in 1987 and Gloucester in 1991-1995. Nonetheless, a decrease in pesticides and PCBs can be seen for both the Boston Harbor Control Station at the New England Aquarium, within the present Deer Island Outfall plume, and at the MWRA ocean outfall in Massachusetts Bay (Figure 2-4). At all stations during this biomonitoring effort, chlorinated pesticides hexachlorobenzene (HCB), heptachlor, heptachlorepoxide, and Mirex® were not detected or the results were near detection limits of the analytical method, a result consistent with the NOAA Mussel Watch data for this area.

2.2.3 Seawater Concentrations of Pesticides, Massachusetts and Cape Cod Bays, Mid-1990s

A summary of particulate PCBs and chlorinated pesticides measured in the Massachusetts Bays program is shown in Table 2-7. From these data, it is clear that at least one preliminary conclusion can be reached: there has been some decrease in the water column concentrations of PCBs and chlorinated hydrocarbons to the coastal ocean in Massachusetts and Cape Cod Bays from the mid-1980s to the mid-1990s.

These data also allow reasonable estimates to be made for selected chlorinated pesticides and PCBs in Massachusetts and Cape Cod Bays seawater in the mid-1990s using all available seawater data (Table 2-8). The Table 2-8 data agree well with published data for chlorinated pesticides in seawater in the vicinity of Sable Island, Nova Scotia for the 1988-1989 time period, as well as other areas of the world as summarized in Bidleman et al. (1992). The estimates presented in Table 2-8 will be used to complete a preliminary assessment of potential chlorinated hydrocarbon concentrations in relation to future MWRA outfall discharge and seawater concentrations in Massachusetts and Cape Cod Bays later in this section of the report.

2.3 Description of Potential Loading Sources of Organic Contaminants

Delivery of organic contaminants to the Massachusetts coastal marine environment occurs by a variety of mechanisms that have been documented to varying degrees for Massachusetts and Cape Cod Bays (Menzie-Cura 1991a; Menzie-Cura 1991b; MWRA 1995). Based on the volumes of water discharged, municipal sewage and sewage sludge discharged in Boston Harbor by the MWRA has long been thought of as the primary source of most of these organic contaminants. The pollution abatement program implemented by MWRA in Boston Harbor, involving the cessation of the discharge of sewage sludge from the existing Deer Island outfall, has reduced the loading of bacteria, concentrations of toxic contaminants, and sewage particles into the coastal marine environment (MWRA 1993). Positive effects from the cessation of sewage sludge discharge have been apparent (Blake et al., 1993).

There are significant mechanisms that influence to varying degrees how organic contaminants enter, leave, and become incorporated into various compartments within the Massachusetts and Cape Cod Bays ecosystem. As the MWRA pollution abatement program becomes more effective on a bay-wide basis, other sources of organic contaminants to the ecosystem will become increasingly more important. The major sources of organic contaminants include effluent discharge from publicly-owned treatment works (POTW), atmospheric deposition, dredged material disposal, river transport, non-point source runoff, contributions from other defined point sources, and import from the Gulf of Maine (Figure 2-5). Once delivered to the ecosystem, organic contaminants undergo a variety of processing reactions, dependent on phase relationships between dissolved and particulate material and biogeochemical interactions. Outputs from the ecosystem include loss from the system to the atmosphere, degradative losses within the system, burial to the sediments, and export to the Gulf of Maine (MWRA 1992; MWRA 1995a; Wade, Massachusetts Bays Program, 1995).

A detailed representation of individual processes that influence concentrations of organic contaminants in Massachusetts and Cape Cod Bays seawater is presented in Figure 2-6 (modified from a similar treatment by Farrington and Westall, 1986). Complete characterization of each of the individual routes, rates and reservoirs indicated in Figure 2-6 is needed. Detailed information that would be required to complete each portion of the source, processing and output terms for Massachusetts and Cape Cod Bays is not yet

available. There are data that would allow a preliminary assessment of major terms in the system. Recently, such an analysis was completed for polycyclic aromatic hydrocarbons (Wade, Massachusetts Bays Program, 1995). This section of the MWRA toxics report presents a similar approach for chlorinated hydrocarbons and PCBs in Massachusetts and Cape Cod Bays.

2.3.1 POTW Outfall Discharge

The MWRA outfall at Deer Island contributes the majority of waste water treatment plant discharge to Massachusetts and Cape Cod Bays, providing an average daily flow of 376 MGD. There are additional POTW effluent discharges to Massachusetts and Cape Cod Bays, including the South Essex Treatment Plant (30 MGD), Lynn Treatment Plant (80 MGD), and other smaller facilities along the coast of Massachusetts (e.g., Scituate, Marshfield, Plymouth, etc.) that contribute a combined average effluent flow of 10 MGD. Detailed characterizations of the chlorinated hydrocarbon concentrations of the MWRA effluent have been completed as part of an ongoing program at MWRA (MWRA, 1993; Hunt et al., 1995; MWRA, 1997) and are summarized in Table 2-9.

In the MWRA effluent characterization studies, the PCB data show a very high degree of variability (Figure 2-7, top plot) over time (Hunt et al., 1995; MWRA 1997). Reasons for this apparent variability were not apparent. Accordingly, a detailed review of individual PCB congener data was completed. This detailed review of the variability of individual PCB congener concentrations in the MWRA Deer Island effluent characterization studies revealed important data quality issues that have yet to be addressed.

Since starting effluent monitoring in 1994, PCB congener data show the effects of analytical interference that was handled differently by different analytical contractors. The variable interference was most notable for two PCB congeners, PCB18 and PCB180. Subsequent investigation revealed that in some samples, an effort was made to adjust for interferences and at other times adjustments were not made and PCB congener concentrations may have been artificially high. Such influences made annual comparison for trends and/or calculation of mean values suspect. To evaluate the effects of analytical interferences on the PCB data value calculated, PCB data were recalculated with PCB18 and PCB180 removed from the summation of individual PCB congeners. The second PCB concentration that was calculated was the sum of 18 individual PCB congeners, not 20 as originally reported (Hunt et al., 1995; MWRA 1997).

Removal of the data for PCB18 and PCB180 dramatically reduced the variability (as measured by the change in the % relative standard deviations (RSD) for PCB congeners) of the PCBs, especially for the 1995 data (Table 2-9). The fact that this variability was removed by simply removing two PCB congeners argues strongly for some type of artificial influence in the MWRA Deer Island effluent data for these two congeners. Accordingly, the mass budget of PCBs in Massachusetts and Cape Cod Bays employed PCBs as the sum of 18 individual PCB congeners, not the 20 congeners as originally reported.

When MWRA conducts additional effluent characterization studies of the Deer Island effluent for chlorinated pesticides and PCBs in the future, additional analytical steps should be added to separate PCBs from chlorinated pesticides and other potential interferences before the GC/ECD analysis is completed on each fraction. Once analytical data are generated that appear to be free from analytical interference, the new PCB effluent data should be compared to the adjustments made in this report to validate the adjustment.

2.3.2 Riverine Transport

Transportation of river-borne organic pollutants has been the subject of recent investigations for trace metals (Wallace et al., 1993) and organic compounds (Menzie-Cura, et al., 1995). The Menzie-Cura et al. report contained descriptions of organic contaminants that are provided in Table 2-10. Most analyses of incoming or outgoing river water completed in that program resulted in undetected concentrations of all individual PCB congeners and selected pesticides to a detection limit of 0.2 ng/L of river water. Some PCBs and chlorinated pesticides were detected in coastal estuarine sediments at the mouth of the Merrimack River (Menzie-Cura et al., 1995) indicating some transport of chlorinated hydrocarbons to the coastal marine environment from rivers in Massachusetts. Data are summarized in Table 2-10. Average whole river water ΣPCBs and Σpesticide concentrations for the fresh water layer were comparable, with ΣPCBs of 2 - 5 ng/L and Σpesticides of 0.13 - 0.14 ng/L. In the absence of data from other rivers in Massachusetts, a reasonable expectation is that these concentrations are approximately representative of average river water discharge to Massachusetts and Cape Cod Bays.

With an average daily discharge rate of 5,800 MGD, the Merrimack River is the major source of fresh water flow into the Massachusetts and Cape Cod Bays system. One of the principal results of satellite reconnaissance completed during the Menzie-Cura et al. (1995) work in the Merrimack River was documentation that the river discharge interacted with the Massachusetts and Cape Cod Bays system approximately half of the time. Other rivers that discharge directly or indirectly into the Massachusetts and Cape Cod Bays system include the Charles River (370 MGD), Mystic River (75 MGD) Danvers River (14 MGD), and a combination of other coastal freshwater discharges (1000 MGD). Total freshwater input into the Massachusetts and Cape Cod Bays system from rivers is estimated at 7,200 to 7,300 MGD.

2.3.3 Atmospheric Levels of PCBs and Chlorinated Pesticides

Previously, in constructing the annual balance of pollutants in Boston Harbor (Menzie-Cura 1991b) and Massachusetts and Cape Cod Bays (Menzie-Cura 1991a), atmospheric concentrations of PCBs from Martha's Vineyard and Georges Bank were the only data available (Harvey and Steinhauer, 1973). At this time, use of these historical data was considered inappropriate and an examination of more recent data was conducted.

2.3.3.1 Wet and Dry Fallout

Golumb et al. (1995) presented an assessment of the atmospheric deposition of toxic metals and polyaromatic hydrocarbons into Massachusetts Bay from September 1992 - September 1993 at two sites surrounding Massachusetts and Cape Cod Bays (i.e., Nahant and North Truro). As part of this effort, PCBs were analyzed in several samples. The authors reported that due to sampling problems, no PCBs (to the detection limits of the analytical procedure) were found in several wet or dry fallout samples. The analytical detection limit for individual PCB congeners was reported to be 1 ng/L. Consequently, no direct measurements of deposition of PCBs and chlorinated hydrocarbons are available.

Swackhamer and Armstrong (1986) presented an estimate of the atmospheric and non-atmospheric contributions of PCBs to Lake Michigan using sediment PCB accumulation rates and mass balance calculations from atmospheric concentrations. Mass balance results showed that wet washout of vapor phase pesticides and dry deposition of particulate phase pesticides were minor contributions in relation to overall particulate rain-out. Using undetected values for wet and dry fallout from Golumb et al., (1995), insignificant to very minor amounts of ΣPCB and Σpesticide net flux can be expected for Massachusetts and Cape Cod Bays.

Latimer (1996) presents an evaluation of the significance of atmospheric deposition of PCBs to Narragansett Bay that provides data on atmospheric concentrations of PCBs in that estuaries air. Major results showed the presence of seasonal differences in PCB concentration in air, with greatest concentrations in the summer months over the 1992 to 1994 time period. Results showed that on the average, wet and dry atmospheric deposition of PCBs to Narragansett Bay were almost equal in magnitude. Overall, however, atmospheric processes (dry deposition, wet washout, rain-out) were documented to account for only 2% of the total inputs from all sources of PCBs to Narragansett Bay.

Rainfall input to Massachusetts and Cape Cod Bays is assumed to be in approximate annual balance with evaporative losses to the atmosphere.

2.3.3.2 Vapor Phase Interactions

Atmospheric transport of PCBs and chlorinated hydrocarbons is the predominant transport mechanism throughout the world. Concentrations of PCBs and chlorinated hydrocarbons in the global atmosphere do not reflect localized transport of pesticides but can reflect usage patterns from thousands of miles away (Knap et al., 1991). The most recent data for a location closest to the Massachusetts and Cape Cod Bays system that could be considered marine in origin for the time period 1988-1989 were obtained from Bidleman et al. (1992) for Sable Island, Nova Scotia. Concentrations of hexachlorohexane (ΣHCHs), ΣPCBs, ΣDDTs, toxaphene, Σchlordanes, dieldrin, and endosulfan I and II were reported (Table 2-11). Comparison of these data with data from Bermuda (Knap et al., 1991) over approximately the same time

period suggest that the Nova Scotia data are more reasonable for this time period in Massachusetts and Cape Cod Bays. Using such atmospheric concentrations and a washout rate such as that presented by Golumb et al. (1995) for Massachusetts and Cape Cod Bays provides an estimate of the flux of PCBs and chlorinated hydrocarbons.

2.3.3.3 Summary of Atmospheric Interactions

Following Swackhamer and Armstrong (1986), there were little hard data from Massachusetts to estimate the net flux to Massachusetts and Cape Cod Bay seawater. It is considered most reasonable to assume that it was likely that the net flux of ΣPCBs to seawater is positive in colder winter months. During the warmer summer months, the net flux of ΣPCBs to seawater from the atmosphere can be expected to be negative, i.e., a net loss of PCBs to the atmosphere in the warmer months (Latimer, 1996). Reviewing the available ΣPCB data, it can be expected that there would be little net change in ΣPCBs in seawater from Massachusetts and Cape Cod Bays from atmospheric interactions on an annual basis. However, this must be considered only the crudest of conclusions until this issue can be examined in a more complete manner - hopefully with high quality and site-specific Massachusetts and Cape Cod Bays marine atmospheric concentrations and seawater data.

2.3.4 Non-Point Source Transport/Combined Sewer Overflows

Considerable work on the transport of organic contaminants from non-point sources and combined sewer overflows (CSOs) to Boston Harbor and Massachusetts and Cape Cod Bays has been completed over the last five to six years (Battelle 1990; MWRA, 1993; Menzie-Cura et al., 1995; and others). None of these works have produced data on PCB congeners and chlorinated pesticides. At the present time, the best source of non-point source data for PCBs and chlorinated pesticides inputs to the Massachusetts and Cape Cod Bays system remains the Menzie-Cura report utilizing data from the 1980s (1991a). ΣPCB concentrations in stormwater runoff were estimated to be on the order of 1-5 ng/L (mean value 2.5 ng/L), with Σpesticide concentrations estimated to be <1 ng/L for Massachusetts coastal areas.

The average annual coastal runoff was estimated at 60 MGD (Wade, Massachusetts Bays Program, 1995); non-point source stormwater runoff was estimated to contribute 50% of this value and CSO discharge was estimated to contribute the remaining 50%.

2.3.5 Dredged Material Waste Disposal

The U.S. Army Corps of Engineers maintains an ongoing navigation improvement program for the navigable waters of coastal Massachusetts. By far, the largest dredging project planned for Massachusetts is the Boston Harbor Navigation Improvement and Berth Dredging Projects (BHNIBDP) (U.S. Army Corps of Engineers and Massachusetts Port Authority, 1995). In this program, a total of 2,871,000 cubic

yards (CY) of material will be removed from the main shipping channel and tributaries of Boston Harbor. Of this total, 1,134,000 CY of sediment in the silt size fraction will be removed and disposed of at various sites in Boston Harbor and other coastal sites. The silt material has been documented to contain concentrations of ΣPCBs and Σpesticides higher than at other cleaner areas of Boston Harbor. Sites which are expected to contribute the most contaminated sediments for dredged material disposal in Boston Harbor are the Conley, Eastern Minerals, Moran, Mystic and North Jetty berths (see Table 2-12). Disposal of the contaminated sediments from these berths was approved only for subaqueous disposal sites within Boston Harbor and extreme coastal Massachusetts Bay. Disposal of contaminated dredge material out into Massachusetts Bay at the MBDS was not authorized.

Modeling efforts completed for this project (U.S. Army Corps of Engineers and Massachusetts Port Authority, 1995) showed that impacts from PCB-contaminated dredged material disposal would be minimal. The release of small amounts of PCBs from contaminated silts in the immediate vicinity of the disposal activities was estimated at <10 ng/L.

In order to derive some estimate of pollutant transport from dredging, it was estimated that a total of 25% of the cleaner sediment material, not slated for containment in disposal sites, would have been contaminated with higher concentrations of PCBs. For purposes of the calculation of an annual budget, it was assumed that 2% of the sediment disposed of somewhere in Massachusetts Bay would be lost to the surrounding water column. Concentrations of ΣPCBs were assumed to be 0.1 μg/g dry weight of sediment in the material that is going to be disposed of at the MBDS. A total loss from dredged material disposal of 2% of solids was assumed for all sediments to be disposed of at the MBDS. Data on chlorinated pesticides showed that non-detected values were obtained uniformly for all sediment samples analyzed to a detection limit of 1.0 μg/g dry weight. It was considered that the detection limit of 1.0 μg/g was too high (by a factor of 10 to 50) to produce meaningful data for chlorinated pesticides in Boston Harbor sediments. Using these assumptions, an upper limit of pollutant transport from the dredging of Boston Harbor was calculated.

2.3.6 Identified Massachusetts Coastal Point Sources

Menzie-Cura and Associates (1991a) reviewed the EPA's NPDES data base for total amounts of effluent discharge from permitted point sources in Massachusetts. Concentrations of ΣPCBs were assessed at 4 ng/L in point source flows estimated at 175 MGD or 235 x 10^{12} L yr^{-1}. ΣPesticides were not calculated for permitted point sources.

2.3.7 Import from the Gulf of Maine

The volume of Massachusetts and Cape Cod Bays has been calculated at 1.4×10^{11} m^3 or 1.4×10^{14} L. Given a generally-accepted half life of seawater in Massachusetts Bay as 0.5 yr, the volume of seawater exchanged annually between Massachusetts and Cape Cod Bays with the Gulf of Maine is approximately 2.8×10^{14} L, or $280{,}000 \times 10^9$ L.

Seawater concentrations of ΣPCBs and Σpesticides can be established from data presented in Table 2-8, as well as data published in Bidleman *et al.* (1992). The concentration of ΣPCBs can be estimated to be ~0.3 to 0.6 ng/L and the concentration of Σpesticides ~0.5 ng/L. However, net addition from transport of dissolved and particulate chlorinated hydrocarbons from the Gulf of Maine is probably balanced by net transport out of Massachusetts and Cape Cod Bays to a first approximation. Therefore, in the absence of other data, net addition of ΣPCBS and chlorinated pesticides from Gulf of Maine seawater influx into Massachusetts and Cape Cod Bays was estimated to be zero. Recent data from Shea (1997) would seem to validate this assumption.

2.4 Estimation of Current and Future Loadings to Massachusetts and Cape Cod Bays

Loadings of ΣPCBs and chlorinated pesticides were calculated for mixed surface water in Massachusetts and Cape Cod Bays using the parameters previously outlined in this subsection. ΣPCBs flux calculations are presented in Table 2-13; Σpesticide flux calculations are presented in Table 2-14.

2.4.1 Discharge of MWRA Deer Island and Other POTWs to Massachusetts Bay

Total discharge from MWRA's Deer Island outfall was calculated to be 137×10^9 gallons per year, or 520×10^9 liters per year (L yr^{-1}). Discharge from other POTWs was calculated for South Essex to be 41×10^9 L yr^{-1}, for Lynn to be 110×10^9 L yr^{-1}, and for other POTW discharges to be 13×10^9 L yr^{-1}. Concentrations of ΣPCBs and chlorinated pesticides under current outfall discharge conditions (Table 2-9) were used for the MWRA effluent source term (MWRA, 1993; Hunt *et al.*, 1995; MWRA, 1997). Concentrations for additional POTWs (S. Essex, Lynn, Other) are based on available data (Menzie-Cura and Associates, 1991a). Multiplying annual discharge volumes for each identified POTW source by measured concentrations of ΣPCBs results in the calculated amounts of ΣPCBs to Massachusetts and Cape Cod Bays on an annual basis as shown in Tables 2-13 and 2-14. [Note that the ΣPCBs presented in Table 2-13 are reported for both 18 and 20 congeners (see Section 2.3.1)]. Further, the loadings reflect projected secondary treatment levels based on pilot plant results (Table 2-15)].

2.4.2 Atmospheric Deposition

There are no contemporary site-specific atmospheric data available for Massachusetts and Cape Cod Bays. The closest comparable atmospheric data are from the Gulf of Maine and Sable Island, Nova Scotia. Wet and dry deposition data from Nahant and Truro for ΣPCBs are not available to the detection limits of the combined field and analytical methodology. Consequently, low contributions from both sources had to be assumed. Similarly, the question of whether Massachusetts and Cape Cod Bays are a net source or a net sink of ΣPCBs to the atmosphere is still open. For purposes of determination of relative contribution, a situation close to balance was assumed, with a small loss of ΣPCBs to the atmosphere was assumed mainly because of the amounts of ΣPCBs discharged into Massachusetts and Cape Cod Bays by Massachusetts-based POTWs.

2.4.3 Dredged Material Disposal

For purposes of determination of relative contribution from the BHNIBDP to Massachusetts and Cape Cod Bays, it was assumed that the entire project would be completed in one year, and that a total of 2% of the total amount of dredged material would find its way into the water column. Concentrations of ΣPCBs in clean sediment were assumed to be 0.1 μg/g dry weight, and that approximately 1.2×10^6 m^3 of this sediment would be disposed of in Massachusetts Bay at the MBDS. Accordingly, the flux of ΣPCBs and Σpesticides estimated to come from the disposal of dredged material in one year was deliberately overestimated for this comparison.

2.4.4 River Outflow

Mass fluxes of ΣPCBs from major Massachusetts rivers were estimated from available whole water concentration data for ΣPCBs (Menzie-Cura, 1995). River flows were obtained from estimates made by Menzie-Cura (1991a; 1995).

2.4.5 Non-point Sources/Combined Sewerage Outflows

Mass fluxes of ΣPCBs from non-point sources/CSOs were calculated from data provided by Menzie-Cura (1991a).

2.4.6 Identified Point Sources

Mass fluxes of ΣPCBs from identified point sources were calculated from data provided by Menzie-Cura (1991a).

2.4.7 Exchange with Gulf of Maine

ΣPCBs concentrations in seawater coming from the Gulf of Maine are assumed to be within limits of determination/detection, and most likely indistinguishable from Massachusetts and Cape Cod Bays seawater exiting Massachusetts Bay. For the purposes of this assessment, and until more specific and reliable seawater ΣPCB data are available, net contributions of seawater exchange with the Gulf of Maine were assumed to be zero.

2.4.8 Results of Current Loading Calculations

Source calculations for ΣPCBs into the Massachusetts and Cape Cod Bays system under present (1992-1996) MWRA Deer Island outfall conditions are presented in Table 2-13. Results of the source calculations, within the limitations and assumptions discussed above, show the dominance of the present MWRA Deer Island outfall discharge into the Bays system. The MWRA discharge is the largest source identified for ΣPCBs based on 20 congeners, with the Merrimack River identified as the next largest source, mainly because the Merrimack River is the largest source of freshwater inflow to the Massachusetts and Cape Cod Bays system.

Source calculations for Σpesticides into the Massachusetts and Cape Cod Bays system under present (1992-1996) MWRA Deer Island outfall conditions are presented in Table 2-14. An identical set of assumptions that was made for ΣPCBs was made for Σpesticide source calculations. Results showed that under present conditions, the MWRA Deer Island outfall contributes the majority of Σpesticides to Massachusetts and Cape Cod Bays. The mean value of Σpesticides under present discharge conditions was determined to be 31 ng/L. Lindane, which is currently commercially utilized locally, accounts for the majority of the Σpesticides in the MWRA effluent (see Table 2-9). The Merrimack River source term was a factor of 10 lower than the MWRA Deer Island effluent in this estimate.

2.4.9 Estimation of Future Loadings to Massachusetts and Cape Cod Bays

Net source contributions of ΣPCBs and chlorinated pesticides were calculated for mixed surface water in Massachusetts and Cape Cod Bays using the secondary treatment effluent data following assumptions previously outlined in subsection 2.4.1 above. ΣPCB and chlorinated pesticide effluent concentrations from MWRA's Pilot Plant studies (Table 2-15) were used for the MWRA effluent source term under each condition (MWRA, 1995). Results for PCBs (Table 2-13) showed the expected effects of secondary treatment on the concentrations of ΣPCBs and chlorinated pesticides on MWRA's Deer Island effluent. Net source calculations were reduced from the current primary treatment estimate of 29 kg yr^{-1} to 5 kg yr^{-1} (or 7.3 kg/yr to 3 kg/yr for the 18 PCB congener term). Under secondary treatment conditions, the Merrimack River outflow becomes the largest source term for ΣPCBs, and disposal of the dredged material from the BHNIBDP becomes of comparable value.

Results of the Σpesticides (Table 2-14) net source contributions under secondary treatment conditions showed the effect of the expected reduction in Σpesticide concentrations in the MWRA effluent. The MWRA Deer Island effluent becomes of comparable order of magnitude with the Merrimack River. The net effect of atmospheric interactions is not estimated in this analysis.

2.5 Comparison of Annual Fluxes from Source Loading Terms with Selected Removal Mechanisms

This section compares the concentrations of annual source terms for ΣPCBs and Σpesticides from predicted source terms (MWRA Deer Island outfall discharge, riverine flows, and non-point sources/combined sewage overflows) with selected removal mechanisms (sedimentation, atmospheric interaction, and outflow to the Gulf of Maine). There are two assumptions that are made in removal mechanisms simply because there are no data to provide a more refined assessment. The first assumption is that inflow from the Gulf of Maine is approximately equal to outflow to the Gulf of Maine (i.e., net exchange = 0). The second assumption is that the atmospheric processes involved in ΣPCB and Σpesticides between winter and summer processes (Swackhamer and Armstrong, 1986) are approximately equal on an annual basis. As the annual net contribution from atmospheric processes may well be close to zero, atmospheric removal was not considered further.

Under these two assumptions, net burial of ΣPCBs and Σpesticides into Massachusetts and Cape Cod Bays sediments becomes the primary removal mechanism. Concentrations of ΣPCBs and Σpesticides in sediment depositional areas of Massachusetts and Cape Cod Bays and sediment accumulation rates can be used to calculate annual removal (Wade et al., 1989). With a sediment accumulation rate of 0.1μg cm^{-2} yr^{-1}, and surface concentrations of 16 - 55 ng/g for ΣPCBs and 4 - 28 ng/g for Σpesticides (2 x ΣDDTs), sediment accumulation rates can be compared with annual source terms to determine if there is a balance between source and removal terms. Removal of ΣPCBs throughout the entire Massachusetts and Cape Cod Bays system was calculated as 0.1 cm yr^{-1} x 0.42 g cm^{-3} x 16 - 55 ng/g x 3.7 x 10^{13} cm^2 = 2.5 - 8.5 x 10^4 g yr^{-1} or 25 to 85 kg yr^{-1} (see Table 2-16).

Given the large source term of the MWRA Deer Island outfall, the assumption that the net gaseous transfer into Massachusetts Bay seawater from the atmosphere is a negative 7 to 14 kg ΣPCBs per year (i.e., there is a net loss of ΣPCBs to the atmosphere from seawater under present MWRA discharge conditions) is probably reasonable. The assumption that the BHNIBDP dredged material disposal into Massachusetts Bay will be effective over the entire year was somewhat important in the total balance. The fact that this assumption is most likely completely unreasonable is the only saving grace for this assumption.

Table 2-16 provides a comparison of selected source terms under various MWRA Deer Island treatment schemes and other source terms with sediment removal rates for ΣPCBs in Massachusetts and Cape Cod

Bays sediment. The fact that the total of source terms were approximately equal to the calculated removal rate supports the initial assumptions that atmospheric and inflow/outflow terms may not play a major controlling function in the overall balance of ΣPCBs in the Massachusetts and Cape Cod Bays system.

Removal of Σpesticides throughout the entire Massachusetts and Cape Cod Bays system was calculated as 0.1 cm yr^{-1} x 0.42 g cm^{-3} x 4 - 28 ng/g x 3.7 x 10^{13} cm^2 = 0.6 - 4.4 x 10^4 g yr^{-1} or 6 to 44 kg yr^{-1}. Table 2-17 provides a comparison of selected source terms under various MWRA Deer Island treatment schemes and other source terms with sediment removal rates for Σpesticides in Massachusetts and Cape Cod Bays sediment. Again, there was an approximate balance between the selected source terms and sediment removal for Σpesticides, supporting the initial assumptions that atmospheric and inflow/outflow terms do not yet play major controlling functions in the overall balance of Σpesticides in the Massachusetts and Cape Cod Bays system.

2.6 Summary - Where Does This Leave Us?

From a comprehensive review of all available water column and biomonitoring studies completed in the Boston Harbor, Massachusetts and Cape Cod Bays area since extensive marine monitoring programs began in the mid-1980s, it is clear that the annual delivery of selected organic pollutants has decreased over time. Some of this decrease in annual delivery can be ascribed to the MWRA having stopped the discharge of sewage sludge from the Deer Island outfall in 1989.

However, it is apparent from the assembled chemical data that cessation of discharge of sewage sludge by the MWRA was not the only seminal event in the history of marine pollution discharge into Boston Harbor and Massachusetts/Cape Cod Bay; simply put, there are other forces at work that govern the distribution and re-distribution of marine pollutants that have only begun to be quantified. As the coastal marine ecosystem re-adjusts to decreases in water column concentrations for selected pollutants resulting from MWRA discharge reduction, it can be expected that some environmental compartments that were once considered ultimate sinks for marine pollutants may in the future become sources instead.

For example, there is an accumulation of hundreds of years of marine pollution resident in Boston Harbor and Massachusetts/Cape Cod Bays sediments which could be released into the water column in response to numerous types of environmental events, including the decrease in their overlying water column concentrations. Perhaps equally important, the role of the marine atmosphere in Massachusetts/Cape Cod Bays is poorly understood at present, but it can be expected to play an increasingly important role in the mobilization and re-mobilization of waterborne and airborne marine pollutants. Another important and probably confounding influence on the water concentrations of chlorinated pesticides and PCBs would be an expected temperature influence on the solubilization of these organic compounds between summer and winter months.

At this time, we simply do not know where we are in the system's re-adjustment processes. We do not have sufficiently precise chemical data to enable us to understand the partitioning between water and air reservoirs that is underway. The major effects can be found at the two critical interfaces in the system, the air-water and sediment-water interfaces. Currently, we have no site-specific data on the factors controlling movement of these types of organic compounds in Massachusetts and Cape Cod Bays. As planned and new environmental studies proceed and data are reported, it may become eventually apparent where in the historic equilibration process the system was as this report was prepared. Now, the only conclusion that can be reached with any degree of certainty regarding chlorinated hydrocarbons in the Boston Harbor and Massachusetts/Cape Cod Bays marine system is that in the decade of the 1990s the marine system was not static. Dramatic and system-wide adjustments were underway at the time.

Table 2-1. Concentrations (ng/L) of the Arithmetic Sum (Σ) of Chlordanes, ΣDDTs, ΣPolychlorinated Biphenyls and ΣPolynuclear Aromatic Hydrocarbons in Massachusetts and Cape Cod Bays Surface Water Determined in the 1987 MWRA Biomonitoring Studies. Concentrations Given as the Mean Value (Standard Deviation).

Sampling	ΣChlordanes	ΣDDTs	ΣPCBs	ΣPAH
MWRA Outfall (Station 5)	0.00 (0.0)	0.01 (0.02)	2.94 (2.1)	-
Nut Island	0.44 (0.45)	0.07 (0.07)	10.5 (18.3)	-
Deer Island Light	0.50 (0.24)	0.57 (0.62)	6.5 (2.7)	-
MassBay Reference Station				
UM-8 Upper	0.00 (0.0)	0.00 (0.0)	1.15 (1.63)	89.0 (68)
UM-8 Lower	0.00 (0.0)	0.00 (0.0)	3.2 (4.5)	230.5 (162)

Table 2-2. Mean Concentrations (ng/L) of Selected Chlorinated Pesticides and PCBs in Seawater Collected at Station H1 in Massachusetts Bay in February 1988.

Particulate Phase (ng/L)	Surface (7 m)					Bottom (80 m)				
	Rep 1	Rep 2	Rep 3	Mean	SD	Rep 1	Rep 2	Rep 3	Mean	SD
Hexachlorobenzene (HCB)	0.045	0.047	0.05	0.047	0.003	0.023	0.035	0.029	0.029	0.006
Lindane	0.76	0.028	0.034	0.274	0.421	0.004	0	0	0.001	0.002
Heptachlor	0.108	0.032	0	0.047	0.055	0.116	0.119	0.019	0.085	0.057
Heptachlorepoxide	0	0	0	0.000	0.000	0	0	0	0.000	0.000
Chlordane	0.017	0.17	0.011	0.066	0.090	0.029	0.02	0.016	0.022	0.007
trans-Nonachlor	0.013	0.02	0.01	0.014	0.005	0.016	0.014	0.007	0.012	0.005
Aldrin	0	0.084	0.136	0.073	0.069	0	0	0	0.000	0.000
Dieldrin	0.016	0.016	0.012	0.015	0.002	0	0	0	0.000	0.000
o,p-DDE	0	0	0	0.000	0.000	0	0	0	0.000	0.000
p,p'-DDE	0.027	0.027	0.019	0.024	0.005	0.112	0.081	0.047	0.080	0.033
o,p-DDD	0.036	0.049	0.03	0.038	0.010	0.142	0.109	0.055	0.102	0.044
p,p'-DDD	0.078	0.083	0.076	0.079	0.004	0.279	0.199	0.121	0.200	0.079
o,p-DDT	0	0	0	0.000	0.000	0.062	0.064	0.03	0.052	0.019
p,p'-DDT	0.015	0.055	0.045	0.038	0.021	0	0	0	0.000	0.000
Mirex	0	0	0	0.000	0.000	0	0	0	0.000	0.000
ΣDDTs	0.156	0.214	0.17	0.180		0.595	0.453	0.253	0.434	
ΣChlordanes	0.138	0.222	0.021	0.127		0.161	0.153	0.042	0.119	
ΣPesticides	1.115	0.611	0.423	0.716		0.783	0.641	0.324	0.583	

Table 2-2. Mean Concentrations (ng/L) of Selected Chlorinated Pesticides and PCBs in Seawater Collected at Station H1 in Massachusetts Bay in February 1988 (Continued).

Dissolved Phase (ng/L)	Surface (7 m)					Bottom (80 m)				
	Rep 1	Rep 2	Rep 3	Mean	SD	Rep 1	Rep 2	Rep 3	Mean	SD
Hexachlorobenzene (HCB)	10.494	5.284	8.188	7.989	2.611	9.37	9.061	8.711	9.047	0.330
Lindane	0.758	0.437	0.366	0.520	0.209	0.38	0.38	0.246	0.335	0.077
Heptachlor	1.094	0	0	0.365	0.632	0.36	0.566	0.206	0.377	0.181
Heptachlorepoxide	0	0	0	0.000	0.000	0	0	0	0.000	0.000
Chlordane	0	0.108	0	0.036	0.062	0.13	0.18	0.086	0.132	0.047
trans-Nonachlor	0.058	0.035	0.012	0.035	0.023	0.06	0.109	0.044	0.071	0.034
Aldrin	0	0	0	0.000	0.000	0	0	0	0.000	0.000
Dieldrin	0.083	0.055	0.089	0.076	0.018	0.05	0.09	0.113	0.084	0.032
o,p-DDE	0	0	0	0.000	0.000	0	0	0	0.000	0.000
p,p'-DDE	0.051	0.039	0.033	0.041	0.009	0.03	0.054	0	0.028	0.027
o,p-DDD	0.248	0.089	0.052	0.130	0.104	0	0.213	0.242	0.152	0.132
p,p'-DDD	0	0	0	0.000	0.000	0	0	0	0.000	0.000
o,p-DDT	0	0	0	0.000	0.000	0	0	0	0.000	0.000
p,p'-DDT	0.237	0.188	0.173	0.199	0.033	0.13	0.225	0.298	0.218	0.084
Mirex	0	0	0	0.000	0.000	0	0	0	0.000	0.000
ΣDDTs	0.536	0.316	0.258	0.370		0.16	0.492	0.54	0.397	
ΣChlordanes	1.152	0.143	0.012	0.436		0.55	0.855	0.336	0.580	
ΣPesticides	13.023	6.235	8.913	9.39		10.51	10.878	9.946	10.44	

Table 2-2. Mean Concentrations (ng/L) of Selected Chlorinated Pesticides and PCBs in Seawater Collected at Station H1 in Massachusetts Bay in February 1988 (Continued).

Particulate Phase (ng/L)	Surface (7 m)					Bottom (80 m)				
	Rep 1	Rep 2	Rep 3	Mean	SD	Rep 1	Rep 2	Rep 3	Mean	SD
PCB08	0	0	0	0.000	0.000	0.065	0.104	0.051	0.073	0.027
PCB18	0.223	0.087	0.151	0.154	0.068	0.053	0.215	0.059	0.109	0.092
PCB28	0.258	0.1	0.186	0.181	0.079	0.298	0.38	0.068	0.249	0.162
PCB52	0.777	0.276	0.512	0.522	0.251	0.55	0.814	0.213	0.526	0.301
PCB44	0.448	0.083	0.203	0.245	0.186	0.075	0.326	0.077	0.159	0.144
PCB66	0	0	0	0.000	0.000	0	0	0	0.000	0.000
PCB101	0.092	0.082	0.068	0.081	0.012	0.62	0.533	0.305	0.486	0.163
PCB118	0.06	0.06	0.03	0.050	0.017	0.217	0.222	0.096	0.178	0.071
PCB153	0.152	0.166	0.141	0.153	0.013	1.687	1.555	0.893	1.378	0.425
PCB105	0	0	0	0.000	0.000	0	0	0	0.000	0.000
PCB138	0.086	0.116	0.105	0.102	0.015	1.268	1.138	0.742	1.049	0.274
PCB187	0.045	0.055	0.051	0.050	0.005	0.58	0.544	0.357	0.494	0.120
PCB128	0.017	0.033	0.03	0.027	0.009	0.14	0.11	0.059	0.103	0.041
PCB180	0.121	0.079	0.08	0.093	0.024	1.049	1.006	0.678	0.911	0.203
PCB170	0.113	0.051	0.058	0.074	0.034	0.635	0.637	0.386	0.553	0.144
PCB195	0.054	0.005	0.008	0.022	0.027	0.133	0.17	0.09	0.131	0.040
PCB206	0.077	0.005	0.014	0.032	0.039	0.123	0.223	0.046	0.131	0.089
PCB209	0.047	0.01	0.002	0.020	0.024	0.125	0.242	0.055	0.141	0.094
ΣPCBs	2.57	1.208	1.639	1.81		7.553	8.219	4.175	6.671	

Table 2-2. Mean Concentrations (ng/L) of Selected Chlorinated Pesticides and PCBs in Seawater Collected at Station H1 in Massachusetts Bay in February 1988 (Continued).

Dissolved Phase (ng/L)	Surface (7 m)					Bottom (80 m)				
	Rep 1	Rep 2	Rep 3	Mean	SD	Rep 1	Rep 2	Rep 3	Mean	SD
PCB08	0	0	0	0.000	0.000	0	0	0	0.000	0.000
PCB18	2.912	0.962	0.964	1.613	1.125	0.95	0.831	0.661	0.814	0.145
PCB28	0.376	0.307	0.329	0.337	0.035	0.42	0.646	0.316	0.461	0.169
PCB52	0.436	0.353	0.343	0.377	0.051	0.79	1.771	0	0.854	0.887
PCB44	0	0.155	0	0.052	0.089	0	0	0	0.000	0.000
PCB66	0	0	0	0.000	0.000	0	0	0	0.000	0.000
PCB101	0.348	0.196	0.188	0.244	0.090	0.28	0.411	0.211	0.301	0.102
PCB118	0.096	0.09	0.05	0.079	0.025	0.07	0.188	0.104	0.121	0.061
PCB153	0.104	0.176	0.164	0.148	0.039	0.14	0.41	0.283	0.278	0.135
PCB105	0	0	0	0.000	0.000	0	0	0	0.000	0.000
PCB138	0.392	0.218	0	0.203	0.196	0.2	0	0	0.067	0.115
PCB187	0	0.052	0	0.017	0.030	0	0	0	0.000	0.000
PCB128	0	0	0	0.000	0.000	0	0	0	0.000	0.000
PCB180	0.119	0.085	0	0.068	0.061	0.04	0.148	0.096	0.095	0.054
PCB170	0	0	0	0.000	0.000	0	0	0	0.000	0.000
PCB195	0.081	0.011	0	0.031	0.044	0	0	0	0.000	0.000
PCB206	0	0.012	0	0.004	0.007	0	0	0	0.000	0.000
PCB209	0.117	0.051	0	0.056	0.059	0	0	0	0.000	0.000
ΣPCBs	4.981	2.668	2.038	3.23		2.89	4.405	1.671	2.989	

Table 2-3. Concentrations (ng/L) of selected chlorinated pesticides and PCBs in triplicate[1] whole seawater collected from three stations near the Massachusetts Bay Disposal Site (MBDS) in May 1992.

Analyte Classification	Mean Value (N = 12)
ΣPCBs	2.49
ΣDDTs	1.07
ΣChlordanes	3.43
HCB	0.56
gamma-BHC	1.84
ΣPesticides	7.05

[1] At one station, two sets of triplicate samples were collected, resulting in 12 field samples collected.

ΣPCBs: Sum of 20 individual congeners.

ΣDDTs: Sum of 2,4'-DDE, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, and 4,4'-DDT.

ΣChlordanes: Sum of heptachlor, heptachlorepoxide, *cis*-chlordane, and *trans*-nonachlor.

ΣPesticides: Sum of HCB, gamma-BHC, heptachlor, aldrin, heptachlorepoxide, 2,4'-DDE, *cis*-chlordane, *trans*-nonachlor, dieldrin, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, 4,4'-DDT, and Mirex.

Data from Battelle (1992).

Table 2-4. Concentrations (ng/L) of PCBs and selected chlorinated pesticides in seawater in the particulate phase only, collected from Massachusetts Bay in 1992 as part of the Massachusetts Bays Program.

Station No.	Date Sampled	ΣPCBs	ΣDDTs	ΣChlordanes	HCB	gamma-BHC	ΣPesticides
MassBay 2	05/21/92	0.12	0	0	0	0	0
MassBay 4	05/21/92	0.56	0	0	0	0	0
MassBay 5	05/21/92	0.04	0	0	0	0	0
MassBay 6	05/21/92	0.33	0	0	0	0	0
MassBay 8	05/21/92	0.007	0	0	0	0	0
MassBay 6	11/23/92	0.134	0	0.001	0	0	0.001

ΣDDTs: Sum of 2,4'-DDE, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, and 4,4'-DDT.

ΣChlordanes: Sum of heptachlor, heptachlorepoxide, *cis*-chlordane, and *trans*-nonachlor.

ΣPesticides: Sum of HCB, gamma-BHC, heptachlor, aldrin, heptachlorepoxide, 2,4'-DDE, *cis*-chlordane, *trans*-nonachlor, dieldrin, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, 4,4'-DDT, and Mirex.

Data from Menzie-Cura *et al.* (1994).

Table 2-5. Concentrations (pg/L) of the Arithmetic Sum (Σ) of Chlordanes, ΣDDTs, Σ Polychlorinated Biphenyls and Σ Polynuclear Aromatic Hydrocarbons in Massachusetts and Cape Cod Bays Surface Water in December 1995 (after Shea, 1997). Where Available, Data Given as the Mean Value Concentration (Standard Deviation).

Sampling Location	Phase	ΣChlordanes	ΣDDTs	ΣPCBs	ΣPAH
MassBay Buoy "B"	Dissolved (mean, SD)	23.0 (8.0)	4.2 (2.0)	231.0 (111)	2985 (1573)
	Particulate (mean, SD)	4.0 (3.0)	1.1 (0.7)	38.5 (22)	3536 (1715)
	Total (mean, SD)	27.0 (12)	5.3 (3.0)	269.5 (155.5)	6521 (3415)
MassBay Stellwagen	Dissolved	9.4	2.5	189	1497
	Particulate	5	0.4	46.5	2233
	Total	14.4	2.9	235.5	3730
Cape Ann	Dissolved	16	1.4	300	2763
	Particulate	6	0.2	54.5	3252
	Total	22	1.6	354.5	6015
Cape Cod Bay-West	Dissolved	11.5	2.0	385	3432
	Particulate	3	0.7	64	3968
	Total	14.5	2.7	449	7400
Georges Bank	Dissolved	4.7	1.4	29.3	813
	Particulate	0.5	0.3	5.8	1772
	Total	5.2	1.7	35.1	2585

Table 2-6. Pre-Exposure and Post-Exposure Concentrations (ng/g) of Selected Chemical Constituents in Blue Mussels (*Mytilus edulis*) Used by the MWRA in Biomonitoring Studies in Boston Harbor and Massachusetts Bay. The Control Site was the Proposed Outfall Location in MassBay (Station 5) in 1987 and for 1992 - 1995; in 1991, it was in Gloucester.

Year	Pre-Exposure Concentration	Control	Boston Harbor (at NE Aquarium)	Deer Island	Nut Island
	ΣPCBs				
1987	317	227	-	630	604
1991	77	77	477	199	
1992	65	44	652	133	
1993	no data	110	596	321	
1994	107	89	500	161	
1995	94	gear lost	441	172	
	Σ DDTs				
1987	52	30	-	63	51
1991	28	28	94	48	
1992	15	12	103	25	
1993	no data	30	130	63	
1994	27	19	86	50	
1995	29	gear lost	92	45	
	α-chlordane				
1987	8.7	6.7	-	21.5	19.5
1991	2.4	2.5	19	10.3	
1992	1.9	12.7	19	6.9	
1993	2.9	3.8	10.5	8.2	
1994	3.5	3.6	12.8	13.8	
1995	2.6	gear lost	11.7	7.5	

Note: The 1987 deployment lasted for 30 days, 1991-1994 deployment lasted for 60 days, and the 1995 deployment lasted for 50 days. The mussels initially were collected from Barnstable in 1987 and Gloucester in 1991-1995.

Table 2-6. Pre-Exposure and Post-Exposure Concentrations (ng/g) of Selected Chemical Constituents in Blue Mussels (*Mytilus edulis*) Used by the MWRA in Biomonitoring Studies in Boston Harbor and Massachusetts Bay. The Control Site was the Proposed Outfall Location in MassBay (Station 5) in 1987 and for 1992 - 1995; in 1991, it was in Gloucester (Continued).

Year	Pre-Exposure Concentration	Control	Boston Harbor (at NE Aquarium)	Deer Island	Nut Island
	Dieldrin				
1987	6.6	3.6	-	11.4	7.6
1991	<1.4	2.3	9	2.9	
1992	<1.0	1.2	6.7	2.7	
1993	<2.9	2.2	4.5	3.4	
1994	<1.2	2	15.6	10.4	
1995	1.5	gear lost	6.9	3.2	
	Lindane				
1987	1.8	0.8	-	5.5	0.8
1991	<1.5	<2.2	<3.2	<2.5	
1992	<1.0	<1.0	<1.9	<1.3	
1993	no data	<1.7	2.3	2.7	
1994	<0.9	<0.6	<2.2	1.6	
1995	0.7	gear lost	0.9	1	
	trans-Nonachlor				
1987	7.7	6.2	-	18	15.8
1991	<1.4	<1.5	<2.5	8.9	
1992	2.1	2.5	21.3	9.3	
1993	4.8	4	11	10.7	
1994	4	3.8	11	11.2	
1995	0.6	gear lost	9	4.2	

Note: The 1987 deployment lasted for 30 days, 1991-1994 deployment lasted for 60 days, and the 1995 deployment lasted for 50 days. The mussels initially were collected from Barnstable in 1987 and Gloucester in 1991-1995.

Table 2-6. Pre-Exposure and Post-Exposure Concentrations (ng/g) of Selected Chemical Constituents in Blue Mussels (*Mytilus edulis*) Used by the MWRA in Biomonitoring Studies in Boston Harbor and Massachusetts Bay. The Control Site was the Proposed Outfall Location in MassBay (Station 5) in 1987 and for 1992 - 1995; in 1991, it was in Gloucester (Continued).

Year	Pre-Exposure Concentration	Control	Boston Harbor (at NE Aquarium)	Deer Island	Nut Island
	Σ PAH				
1987	581	465	-	2344	683
1991	217	228	2570	1207	
1992	216	129	3545	1934	
1993	188	166	1321	665	
1994	264	122	2255	848	
1995	214	gear lost	1444	761	
	Σ LMW PAH				
1987	-	-	-	1221	
1991	113	74	239	516	
1992	80	61	199	427	
1993	66	66	110	169	
1994	106	61	79	217	
1995	105	gear lost	206	340	
	Σ HMW PAH				
1987	-	-	-	1123	
1991	104	154	2330	691	
1992	136	69	3347	1507	
1993	122	101	1210	496	
1994	158	61	2174	631	
1995	109	gear lost	1238	421	

Note: The 1987 deployment lasted for 30 days, 1991-1994 deployment lasted for 60 days, and the 1995 deployment lasted for 50 days. The mussels initially were collected from Barnstable in 1987 and Gloucester in 1991-1995.

Table 2-7. Concentrations (ng/L) in Mean Values or Ranges of Selected Chlorinated Pesticides and Total Polychlorinated Biphenyls (ΣPCBs) Over Time in Massachusetts and Cape Cod Bays.

Analyte Classification	Mid-1980s	1987	1988	1992	1992	1995
ΣPCBs	12	7.3 1.15 - 2.94	5 - 9	<2.49	0.04 - 0.56	0.24 - 0.45
ΣDDTs	-	- 0.00 - 0.01	0.55 - 0.83	<1.1	0	0.002 - 0.005
ΣChlordanes	-	- 0.00	0.56 - 0.70	<3.4	0.001	0.015 - 0.028
HCB	-	-	8.0 - 9.1	<0.56	0	0.05 - 0.09
γ - BHC (Lindane)	-	-	0.001 - 0.3	<1.8	0	0.04 - 0.06
ΣPesticides	-	- 0.00 - 0.01	10 - 11	<7	0.001	0.12 - 0.4

ΣPCBs: Sum of 20 individual congeners.

ΣDDTs: Sum of 2,4'-DDE, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, and 4,4'-DDT.

ΣChlordanes: Sum of heptachlor, heptachlorepoxide, *cis*-chlordane, and *trans*-nonachlor. (Note that 1995 seawater data also include *trans*- (γ-) chlordane, oxychlordane, and *cis*-nonachlor).

ΣPesticides: Sum of HCB, γ-HCH (Lindane), α-HCH, aldrin, dieldrin, endrin, Mirex, ΣChlordanes, and ΣDDTs.

Table 2-8. Mid-1990s Estimated Concentration Ranges (ng/L) of Selected Chlorinated Pesticides and Total PCBs (∑PCBs) in Massachusetts and Cape Cod Bays Seawater.

Analyte Classification	Estimated Concentration Range
∑PCBs	0.2 - 0.5
∑DDTs	≤0.1
∑Chlordanes	≤0.1
HCB	<0.1
γ-BHC (Lindane)	<0.1

∑PCBs: Sum of 20 individual congeners.

∑DDTs: Sum of 2,4'-DDE, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, and 4,4'-DDT.

∑Chlordanes: Sum of heptachlor, heptachlorepoxide, *cis*-chlordane, and *trans*-nonachlor.

Table 2-9. Concentrations (ng/L) of Selected Chlorinated Pesticides and Total PCBs (ΣPCBs) in Effluent from the MWRA Deer Island Treatment Plant as Reported in Three Separate Technical Reports.

Analyte Analyte Classification	1993/1994[1] Mean (Range)	1995[2] Mean (Range)
Aldrin	0.35 (0 - 5.3)	0.11 (0 - 2.3)
Chlordane (ΣChlordanes)	5.58 (0.24 - 58.5) 10.13 (3.9 - 74.3)	2 (0 - 3.7) 6.2 (4 - 57)
Dieldrin	1.43 (0 - 5.2)	0
4,4'-DDT (ΣDDTs)	12.1 (0.8 - 12) 31.07 (11.8 - 56.8)	6.5 (0 - 22) 11.8 (0 - 30)
Endrin	4.16 (0 - 57.8)	0
Heptachlor	0.04 (0 - 1.3)	2.5 (0 - 15)
γ - BHC (Lindane)	13.9 (7.4 - 25.4)	24.2 (0 - 160)
ΣPCBs (20 congeners)	57.8 (14.1 - 125.1)	56.5 (7 - 225)
ΣPCBs (18 congeners)	39.2 (6.6 - 88.8)	19.3 (4 - 53.2)

1 - Deer Island Effluent Characterization and Pilot Treatment Plant Studies, June 1993 - November 1994. MWRA Technical Report Series No. 95-7.

2 - Deer Island Effluent Characterization Studies, January 1995 - December 1995. Draft Report. Routine Monitoring Results only.

ΣChlordanes: Sum of heptachlor, heptachlorepoxide, *cis*-chlordane, and *trans*-nonachlor.

ΣDDTs: Sum of 2,4'-DDE, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, and 4,4'-DDT.

ΣPCBs: Sum of listed individual congeners.

Table 2-10. Mean Concentrations (ng/L) of Selected Chlorinated Pesticides and Total PCBs (ΣPCBs) in the Merrimack River Water Over the Time Period 1992-1993.

Analyte Classification	Average Fresh Water Layer	Average Salt Water Layer
ΣPCBs	2.21	5.10
ΣDDTs	0	0
ΣChlordanes	0	0
HCB	0.05	0.13
γ-BHC (Lindane)	0.09	0
ΣPesticides	0.14	0.13

ΣPCBs: Sum of 20 individual congeners.

ΣDDTs: Sum of 2,4'-DDE, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, and 4,4'-DDT.

ΣChlordanes: Sum of heptachlor, heptachlorepoxide, *cis*-chlordane, and *trans*-nonachlor.

ΣPesticides: Sum of HCB, gamma-BHC, heptachlor, aldrin, heptachlorepoxide, 2,4'-DDE, *cis*-chlordane, *trans*-nonachlor, dieldrin, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, 4,4'-DDT, and Mirex.

Table 2-11. Mean Concentration and Ranges (pg/m^3) of Selected Chlorinated Pesticides and Total PCBs (ΣPCBs) in the Northwest Atlantic Atmosphere at Sable Island, Nova Scotia, 1988-1989.

Analyte Classification	Mean Concentration	Analytical Range
ΣHCHs	250	121 - 364
ΣPCBs	52	2 - 107
ΣDDTs	4.5	2.1 - 6.8
Toxaphene	38	20 - 71
ΣChlordanes	15	7.7 - 25
Dieldrin	8	8.7 - 12
Endosulfan I and II	54	1.4 - 159

ΣHCH: Sum of a-HCH and g-HCH.

ΣPCBs: Sum of 38 individual congeners.

ΣDDTs: Sum of 2,4'-DDE and 4,4'-DDT.

ΣChlordanes: Sum of heptachlor, heptachlorepoxide, *cis*-chlordane, *trans*-chlordane, *cis*-nonachlor, and *trans*-nonachlor.

Table 2-12. Mean Concentration (µg/g dry weight) of Total PCBs (ΣPCBs), ΣPesticides, and Total Organic Carbon (%) in the Boston Harbor Sediments that will be Dredged in the Boston Harbor Navigation Improvement and Berth Dredging Projects.

Site Name	ΣPCBs	ΣPesticides	Total Organic Carbon
Army Base	1.1	ND	3.3
Boston Edison	0.34	ND	7.1
Conley	1.86	ND	2.7
Eastern Minerals	2.05	ND	5.8
Gulf Oil	0.75	ND	2
Moran	1.9	ND	4.5
Mystic Piers	2.22	ND	4.5
North Jetty	2.55	ND	3.2
Reference	0.01	ND	3.4
Control	0.024	ND	0.33

ΣPCBs: Sum of 20 individual congeners.

ΣPesticides: Aldrin, Chlordane, pp-DDT, DDE, DDD, Dieldrin, Endosulfan I, Endosulfan II, Endosulfan Sulfate, Endrin, Endrin Aldehyde, Heptachlor, Heptachlor Epoxide, Hexachlorocyclohexane, Methoxychlor, and Toxaphene.

ND: Not Detected in sediment to a detection limit of 1 µg/g dry weight.

Table 2-13. Annual Source Volumes [(L yr^{-1}) or (m^3 yr^{-1})], ΣPCB Mean Source Concentrations [(ng L^{-1}) or (μg g^{-1})], and ΣPCB Annual Flux Net (kg yr^{-1}) Calculations from Identified Sources to Massachusetts and Cape Cod Bays System Under Present (1992-1996) and Planned (1998 and Beyond) MWRA Deer Island Outfall Conditions.

Identified Sources	Volume	Mean Concentration ΣPCB Annual Flux	
		PCB20 / PCB18	PCB20 / PCB18
POTW Outfalls			
*MWRA Deer Island (Primary)	520 X 10^9	56 / 14	29.1 / 7.3
South Essex (Primary)	41 X 10^9	56 / 14	2.3 / 0.6
Lynn (Secondary)	110 X 10^9	10 / 10	1.1 / 1.1
Other (Primary)	13 X 10^9	56 / 14	0.73 / 0.2
**MWRA Deer Island (Secondary)	520 X 10^9	10 / 6	5.2 / 3.1
Atmospheric Deposition			
Wet Washout		ND	< 1
Dry Deposition		ND	< 1
Net Gaseous Transfer (flux: 2 - 4 μg m^{-2} yr^{-1})			-7 to -14 (est.)
Dredged Material Disposal			
BHNIP (2% Loss on disposal)	1.15 X 10^6	0.1	4.6
River Outflow			
Merrimack River (5,800 MGD)	8,020 X 10^9	2	8.0
Charles River (370 MGD)	512 X 10^9	2	1.0
Mystic River (75 MGD)	104 X 10^9	2	0.21
Danvers River (14 MGD)	19 X 10^9	2	0.04
Others (1000 MGD)	1,382 X 10^9	2	2.8
Non-point Sources (60 MGD)	83.0 X 10^9	4	0.3
Point Sources (170 MGD)	235 X 10^9	4	0.9
Exchange with Gulf of Maine	-	-	0

Table 2-14. Annual Source Volumes [(L yr^{-1}) or (m^3 yr^{-1})], ΣPesticides Mean Source Concentrations [(ng L^{-1}) or (μg g^{-1})], and ΣPesticides Annual Flux Net (kg yr^{-1}) for Identified Sources to Massachusetts and Cape Cod Bays System Under Present (1992-1996) and Planned (1998 and Beyond) MWRA Deer Island Outfall Conditions.

Identified Sources	Volume	Mean Concentration	Annual Flux
POTW Outfalls			
*MWRA Deer Island (Primary)	520 X 10^9	31	16.1
South Essex (Primary)	41 X 10^9	31	1.3
Lynn (Secondary)	110 X 10^9	8	0.9
Other (Primary)	13 X 10^9	31	0.4
**MWRA Deer Island (Secondary)	520 X 10^9	8	4.2
Atmospheric Deposition			
Wet Washout			Unknown
Dry Deposition			Unknown
Net Gaseous Transfer			Unknown
Dredged Material Disposal			
BHNIP (2% Loss on disposal)	1.15 X 10^6	0.1	0.02
River Outflow			
Merrimack River (5,800 MGD) (Estimate 50% to MassBay)	8,020 X 10^9	0.3	1.2
Charles River (370 MGD)	512 X 10^9	0.3	0.16
Mystic River (75 MGD)	104 X 10^9	0.3	0.03
Danvers River (14 MGD)	19 X 10^9	0.3	0.006
Others (1000 MGD)	1,382 X 10^9	0.3	0.41
Non-point Sources (60 MGD)	83.0 X 10^9	0.4	0.03
Point Sources (170 MGD)	235 X 10^9	0.5	0.12
Exchange with Gulf of Maine	-	-	0

Table 2-15. Concentrations (ng/L) of Selected Chlorinated Pesticides and Total PCBs (ΣPCBs) in Primary and Secondary Effluent from the MWRA Deer Island Pilot Treatment Plant as Reported in Deer Island Pilot Plant Effluent Characterization Studies, January 1995 - December 1995 (MWRA, 1995).

Analyte Classification	Primary Effluent Mean (SD)	Secondary Effluent Mean (SD)
ΣChlordane	4.8 (4.6)	0.97 (1.89)
ΣDDTs	10.0 (7.0)	0.56 (1.18)
ΣPesticides	31.3 (18.4)	7.9 (4.2)
ΣPCBs (20 congeners)	56.5 (84.3)	10.4 (15.2)
ΣPCBs (18 congeners)	14.2 (8.9)	6.04 (9.7)

ΣChlordanes: Sum of heptachlor, heptachlorepoxide, *cis*-chlordane, and *trans*-nonachlor.

ΣDDTs: Sum of 2,4'-DDE, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, and 4,4'-DDT.

ΣPesticides: Sum of HCB, Gamma-BHC, Heptachlor, Aldrin, Heptachlorepoxide, DDMU o,p-DDE, Chlordane, trans-Nonachlor, Dieldrin, p,p'-DDE, o,p-DDD, Endrin, p,p'-DDD, o,p-DDT, p,p'-DDT, and Mirex.

ΣPCBs: Sum of individual congeners, as discussed in text.

Table 2-16. Annual Source Calculations of ΣPCBs for Selected Sources (POTW Outfalls, River Outflows, Point Sources, Dredged Material Disposal, and Non-point Sources/Combined Sewage Overflows) Compared with Sedimentation Removal.

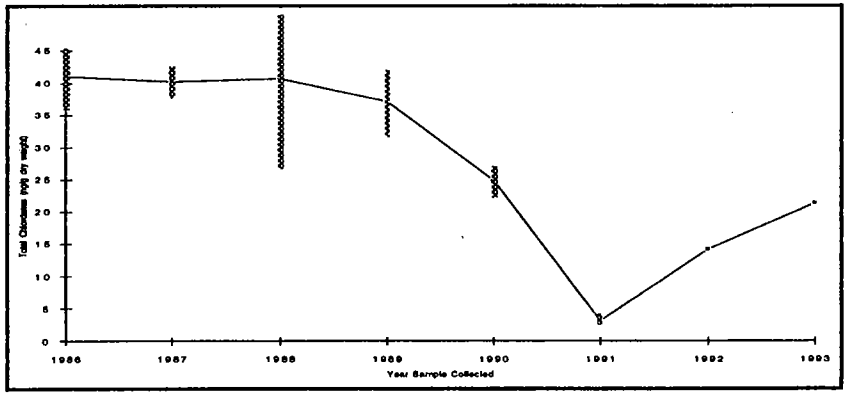
Sources	Annual Source (kg/yr^{-1}) Present Day Conditions (Primary)	Secondary
POTW Outfalls		
MWRA Deer Island (Primary)	29.1	5.2
South Essex (Primary)	2.3	2.3
Lynn (Secondary)	1.1	1.1
Other (Primary)	0.73	0.73
Dredged Material Disposal		
BHNIP (2% Loss on disposal)	4.6	4.6
River Outflow		
Merrimack River (5,800 MGD) (Estimate 50% to MassBay)	8.0	8.0
Charles River (370 MG)	1.0	1.0
Mystic River (75 MGD)	0.21	0.21
Danvers River (14 MGD)	0.04	0.04
Others (1000 MGD)	2.8	2.8
Non-point Sources (60 MGD)	0.3	0.3
Point Sources (170 MGD)	0.9	0.9
Total Sources:	51.1	27.1
Total Removal:		25 to 85 $kg\ yr^{-1}$

Table 2-17. Annual Source Calculations of ΣPesticides for Selected Sources (POTW Outfalls, River Outflows, Point Sources, Dredged Material Disposal, and Non-point Sources/Combined Sewage Overflows) Compared with Sedimentation Removal.

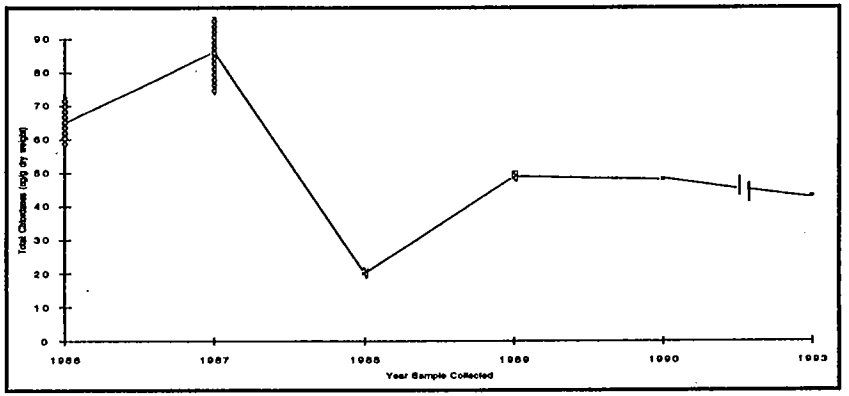
	Annual Source (kg/yr^{-1})	
Sources	Primary	Secondary
POTW Outfalls		
MWRA Deer Island	16.1	4.2
South Essex (Primary)	1.3	1.3
Lynn (Secondary)	0.9	0.9
Other (Primary)	0.4	0.4
Dredged Material Disposal		
BHNIP (2% Loss on disposal)	0.02	0.02
River Outflow		
Merrimack River (5,800 MGD) (Estimate 50% to MassBay)	1.2	1.2
Charles River (370 MG)	0.16	0.16
Mystic River (75 MGD)	0.03	0.03
Danvers River (14 MGD)	0.006	0.006
Others (1000 MGD)	0.41	0.41
Non-point Sources (60 MGD)	0.03	0.03
Point Sources (170 MGD)	0.12	0.12
Total Sources:	20.7	8.8
Total Removal:	6 to 44 kg/yr^{-1}	

Figure 2-1. Trends in ΣChlordane Concentrations (ng/g dry weight) in Blue Mussels During Late 1980s to Early 1990s. Data from the NOAA Status & Trends Program at Four Separate Boston Harbor Monitoring Stations.

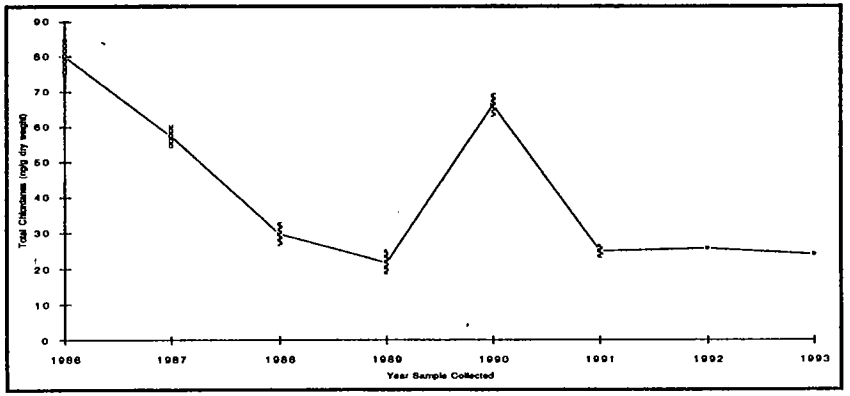
Brewster Island, 8 years, from 1986 to through 1993.



Dorchester Bay, 6 years, from 1986 to through 1993.



Deer Island, 8 years, from 1986 to through 1993.



Hingham Bay, 8 years, from 1986 to through 1993.

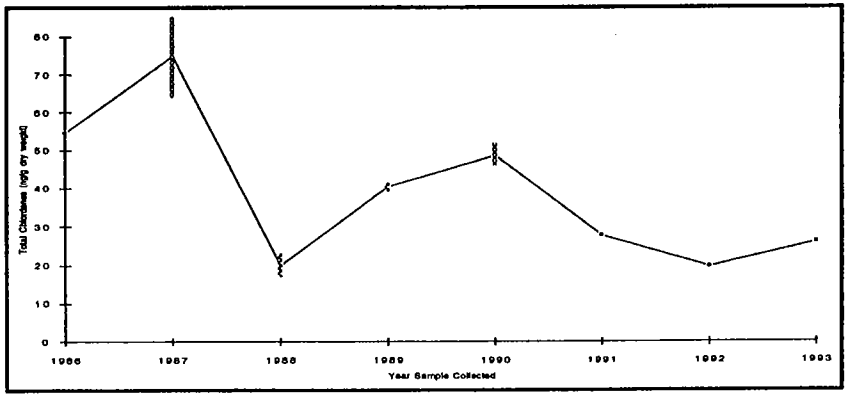
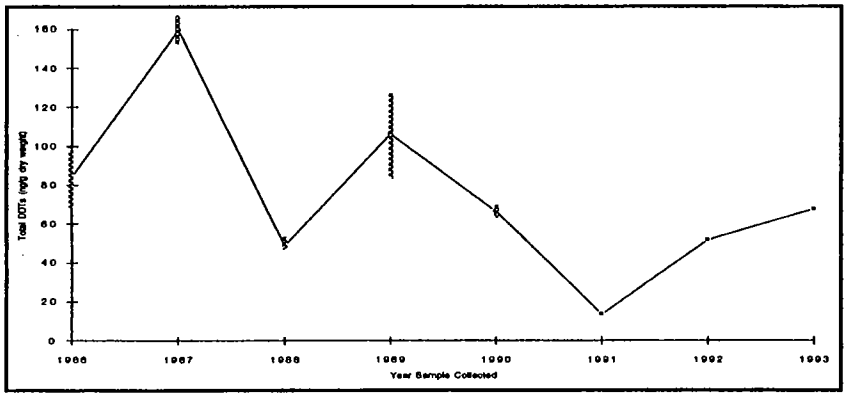
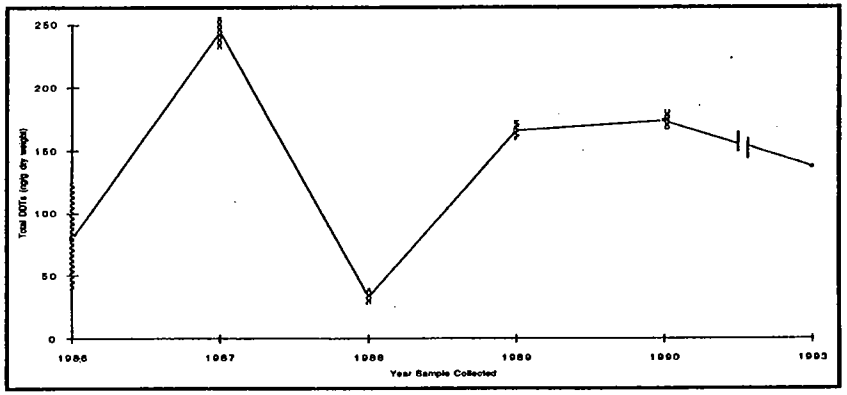


Figure 2-2. Trends in ΣDDT Concentrations (ng/g dry weight) in Blue Mussels During Late 1980s to Early 1990s. Data from the NOAA Status & Trends Program at Four Separate Boston Harbor Monitoring Stations.

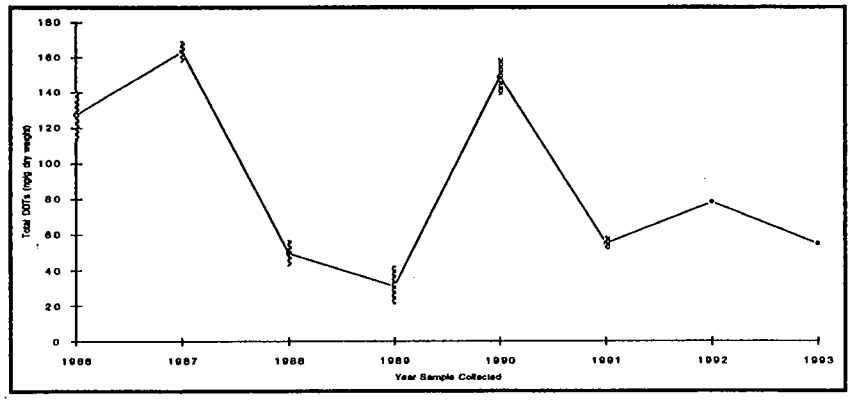
Brewster Island, 8 years, from 1986 through 1993.



Dorchester Bay, 6 years, from 1986 through 1993.



Deer Island, 8 years, from 1986 through 1993.



Hingham Bay, 8 years, from 1986 through 1993.

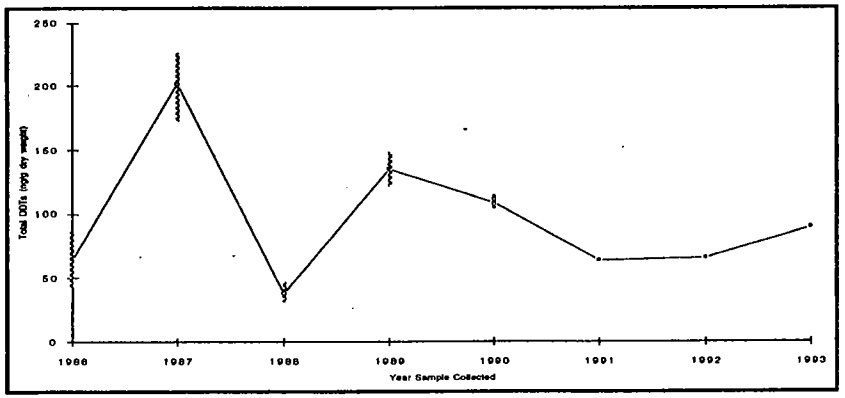
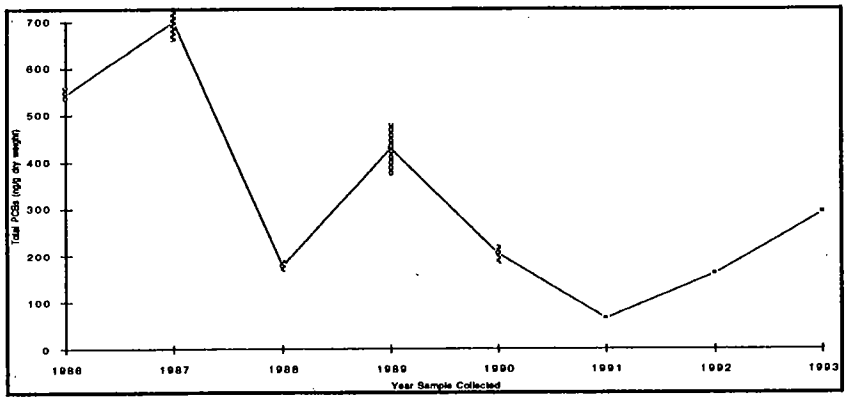
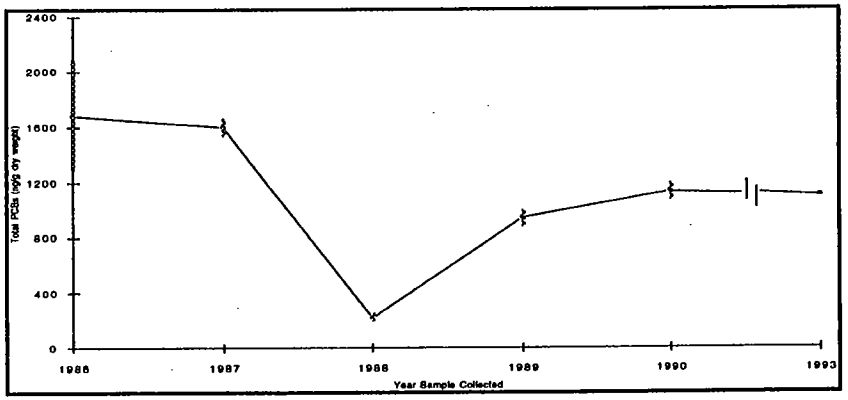


Figure 2-3. Trends in ΣPCB Concentrations (ng/g dry weight) in Blue Mussels During Late 1980s to Early 1990s. Data from the NOAA Status & Trends Program at Four Separate Boston Harbor Monitoring Stations.

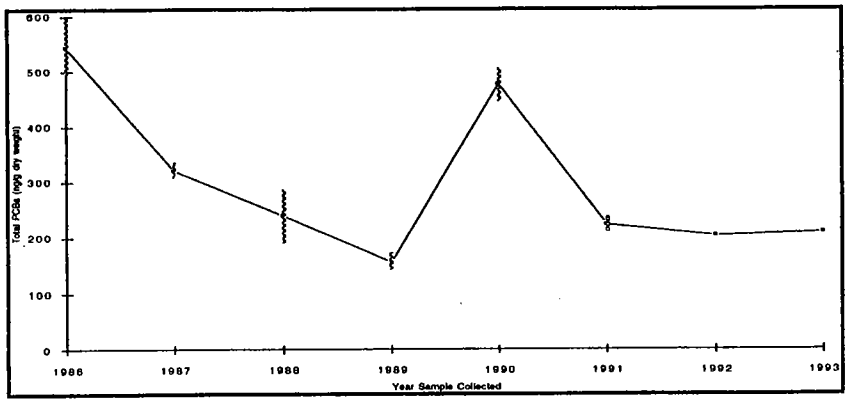
Brewster Island, 8 years, from 1986 to through 1993.



Dorchester Bay, 6 years, from 1986 to through 1993.



Deer Island, 8 years, from 1986 to through 1993.



Hingham Bay, 8 years, from 1986 to through 1993.

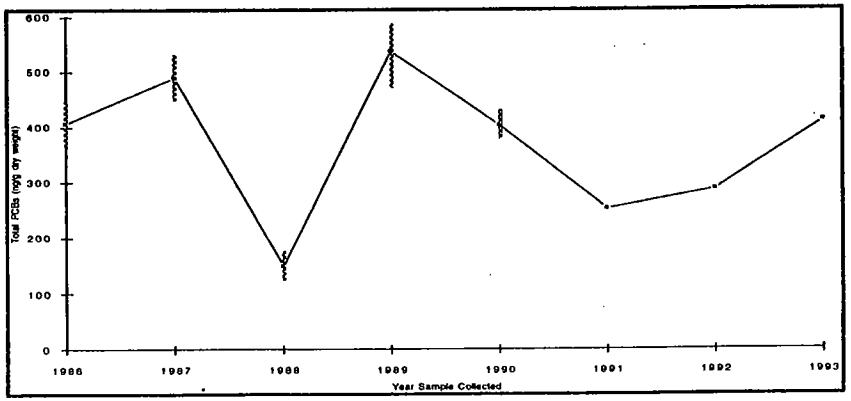


Figure 2-4. Annual Variability of ΣPCBs, ΣDDTs, and ΣPAH Assimulated over time in the MWRA Biomonitoring Program for Boston Harbor (New England Aquarium), Deer Island Outfall, and MassBay Station 5.

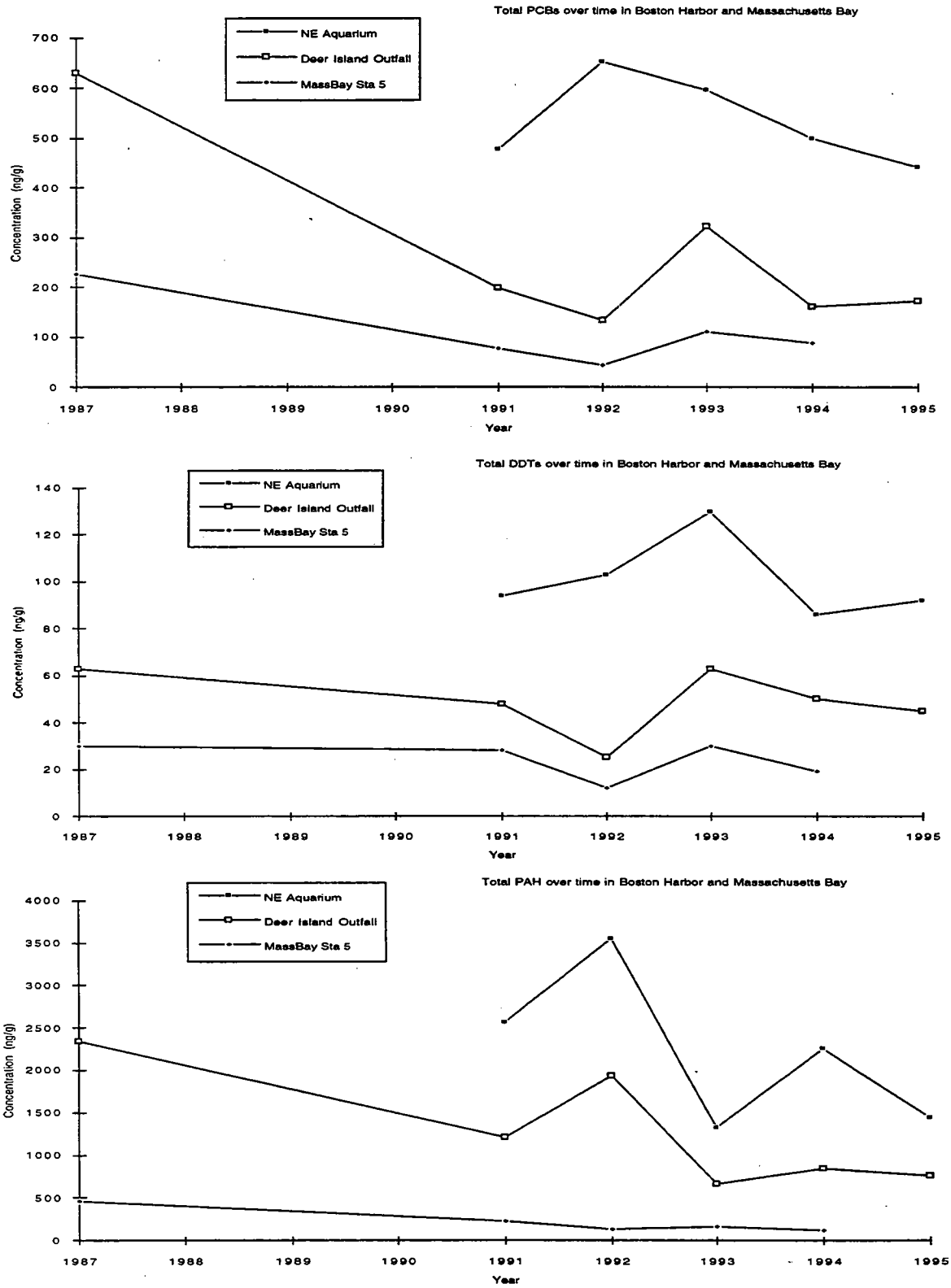


Figure 2-5. Description of Sources, Processing and Output Processing Functions for Organic Contaminants in the Massachusetts and Cape Cod Bays System.

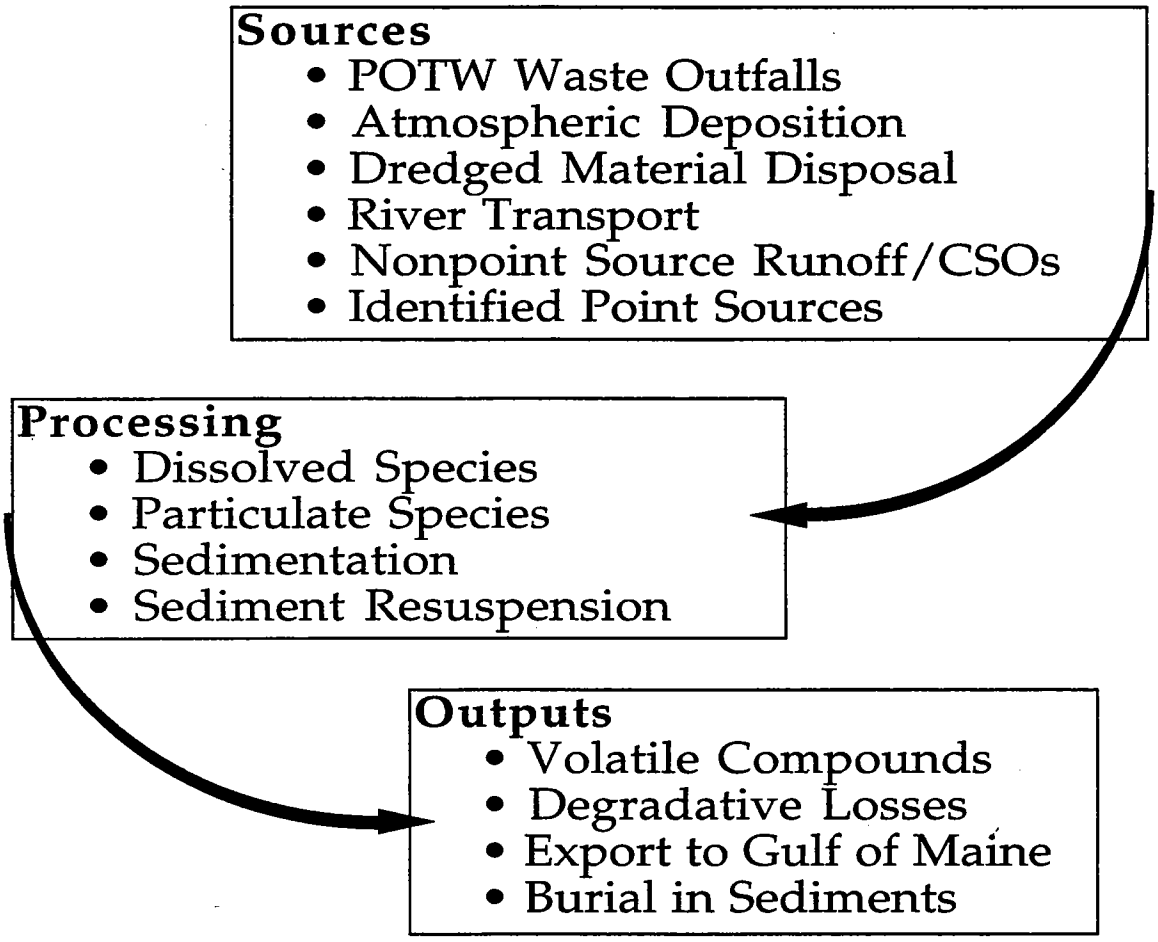


Figure 2-6. Diagrammatic Representation Of Sources, Processing and Output Processing Functions for Organic Contaminants in the Massachusetts and Cape Cod Bays System (after Farrington and Westall, 1986).

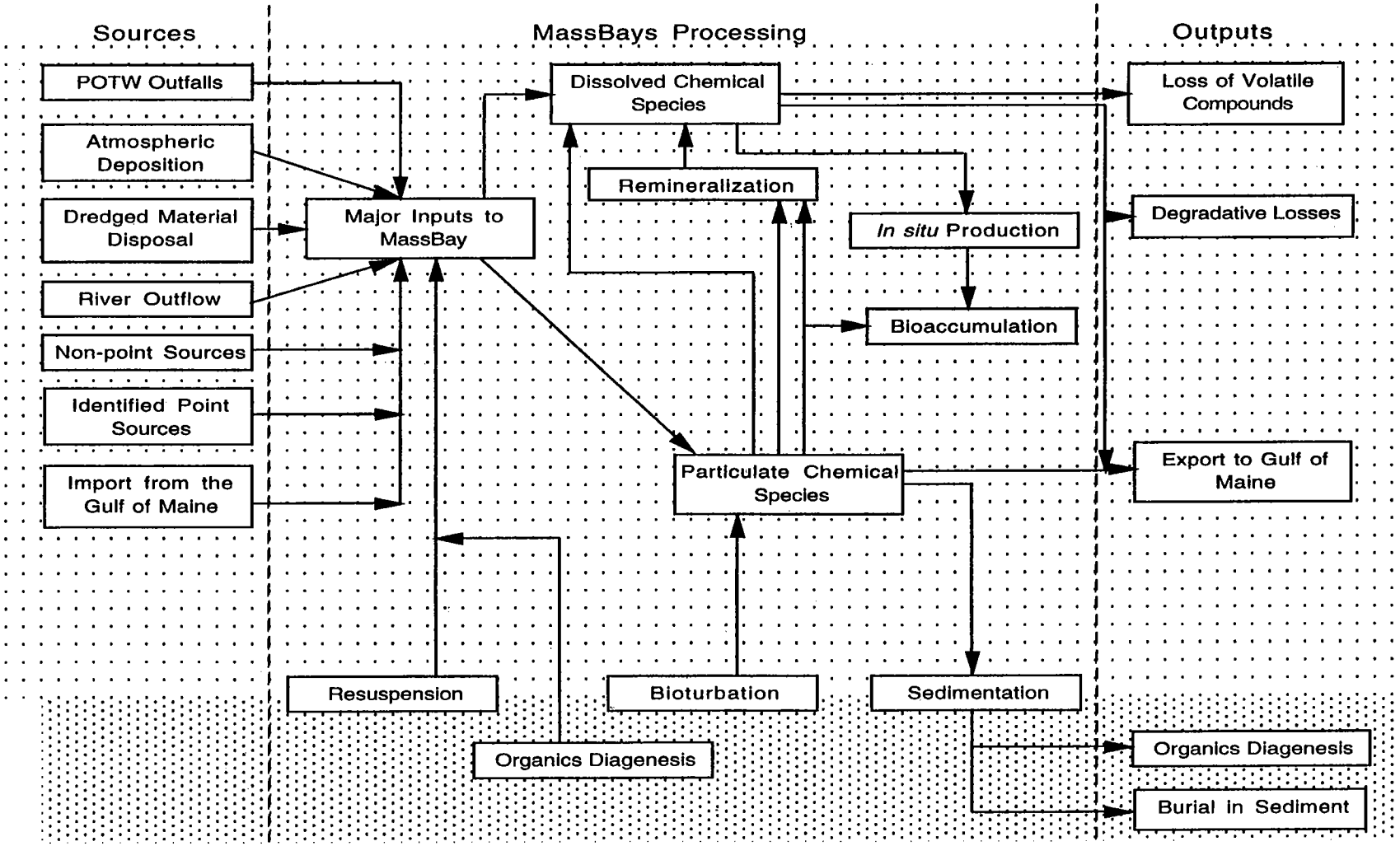
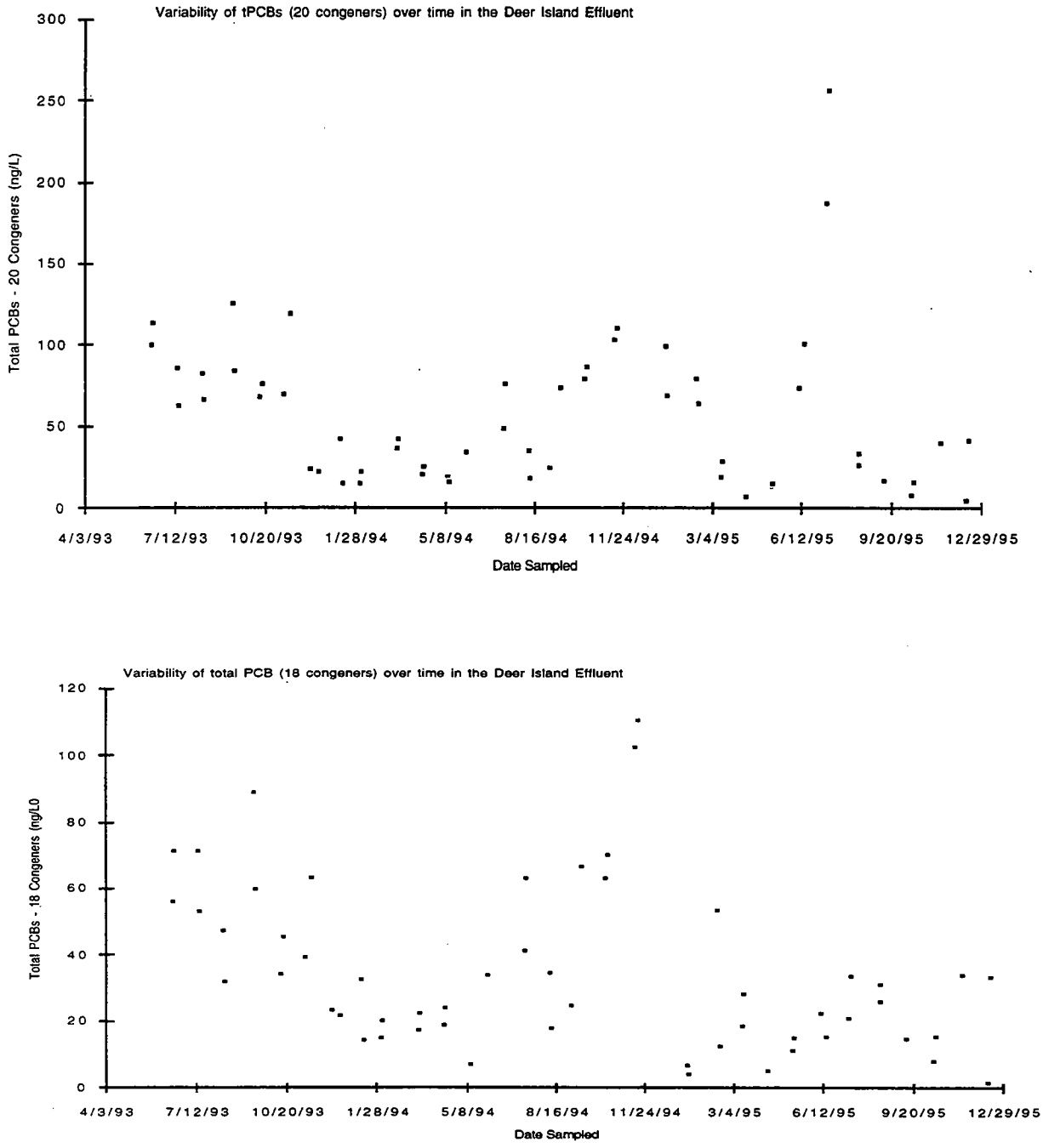


Figure 2-7. Comparison of Variability of ΣPCBs (20 congeners) in the Deer Island Effluent to Variability of ΣPCBs Adjusted for Suspected Analytical Interferences (18 congeners).



3.0 EVALUATION OF MERCURY

As part of the evaluation of the present and future MWRA outfall effluent, a characterization of mercury was made. This characterization included an evaluation of the present sources of mercury to Boston Harbor (Section 3.6) as well as the potential influence of the future MWRA outfall on the mercury load to Massachusetts Bay (Section 3.7).

3.1 Introduction

Mercury is a virulent poison that is readily absorbed through the respiratory tract, the gastrointestinal tract, and the skin. Organic forms of mercury can bioaccumulate or build up in concentration over time in tissues of living organisms, such as fish, shellfish, and lobster. Concerns over human consumption of these aquatic organisms has led to careful monitoring of mercury levels. This is of further concern because certain groups consume significantly higher amounts of fish than the average citizen. Also, pregnant women who eat contaminated fish can pass mercury to their unborn children, who are particularly susceptible to its toxic effects.

The discharge of mercury into coastal waters can impact public health by bioaccumulation into seafood. For reference, the Federal Food and Drug Administration (FDA) limit for mercury in fish and lobster is 1.0 ppm on a wet weight basis. Mercury levels measured by the Massachusetts Division of Marine Fisheries since 1982 reported an average of 0.04 ppm in pollock, 0.19 ppm in flounder and a maximum level of 0.19 ppm in soft-shell clams from the Boston Harbor area (MADEP, 1996). Schwartz et al. (1993, 1995) measured mercury and other metals in winter flounder, American lobster and bivalve molluscs from Boston Harbor, Salem Harbor and coastal Massachusetts. The latest results show that mercury was lowest in bivalve molluscs, averaging at 0.035 ppm. American lobster from Boston Harbor contained from 0.15 to 0.25 ppm mercury. Winter flounder from all areas averaged 0.14 ppm mercury. A recent survey by the Minnesota Department of Agriculture (MDA, 1995) found an average mercury concentration of 1.13 parts per million (ppm) in 27 samples of swordfish.

3.2 The Difficulty of Quantifying Mercury Levels

Mercury presents unique analytical challenges due to its low ambient concentrations, its speciation and its complicated ecosystem dynamics. Windom et al. (1991) discussed the difficulty in analysis of historical trends for trace metals. These could be complicated by improper sampling and analytical procedure, which could contaminate the samples and give false high concentrations. This is particularly true for mercury, where ambient levels are often below ng/l levels but typical detection levels are set at 200 ng/l. For example, it was difficult to quantify the relationship between clean-up efforts versus sample

contamination when measured mercury concentrations in Vandercook Lake, Wisconsin decreased from more than 200 ng/l in 1983 to about 50 ng/l in 1985 - 1986 and then to 0.5 ng/l in 1986 as progressively better and cleaner techniques for sample collection and handling were adopted (Zillioux et al., 1993). Similarly, atmospheric deposition of mercury over land areas decreased from the 1975 estimate by Garrels et al. of 88.9 micrograms per square meter per year ($\mu gm^{-2}yr^{-1}$) to the 1994 estimate of Mason et al. of 15.8 $\mu gm^{-2}yr^{-1}$ for latitudes 30° to 70°N. Accordingly, this report will focus only on reports after 1990 with documented quality control/assurance such that the quantification of mercury at below ng/l levels is possible. Reports where analytical measurements may be suspect will be mentioned only if they treated certain important issues not otherwise addressed but, if included, will be noted as such. This includes, but is not limited to, the following list: the CSO monitoring program in the late 1980's, the local limits study in 1992, and NPDES program of MWRA prior to 1993 etc.

An additional difficulty in quantifying the ecological and public health impacts of mercury is that it can exist in various forms. Researchers have distinguished between total and dissolved inorganic mercury as well as total and dissolved methyl mercury. In addition, the term reactive mercury is sometimes used. This review will focus on total mercury only.

3.3 Mercury Levels in Massachusetts Coastal Surface Water

Mercury was analyzed in various parts of Boston Harbor and Massachusetts Bay as part of the MWRA Secondary Treatment Facilities Plan (STFP, 1988). Mercury in shallow waters from Broad Sound and Massachusetts Bay were reported to average 2.1 (±0.16) and 3.6 (±0.8) ng/l, respectively (number in parenthesis is the standard deviation), using analytical techniques with a detection limit of 0.8 ng/l. Measurements were also performed in July and August 1987 as part of a bioaccumulation study as shown in Table 3-1. It should be noted that Station 5 is close to the future outfall location. Many of these measurements were below detection limits, which apparently varied from about 4.9 to 9.9 ng/l. (Note: Table 3-1 through Table 3-3 report total mercury.) Geometric means were used for the following discussion because contaminant concentrations typically follow log-normal distributions.

The Central Artery/Third Harbor Tunnel project also monitored for mercury in various parts of Boston Harbor and the geometric mean concentrations are summarized in Table 3-2. In addition, measurements from Fort Point Channel in November 1995 showed values between 20 and 30 ng/l both upstream and at the mouth of the channel. It can be seen that mercury levels increased from 1992 and decreased by 1995. This may reflect the effects of construction activities (dredging for the Third Harbor Tunnel and activities on Spectacle Island) as well as sampling/analytical problems. Surface water sampling around Spectacle Island in 1996 showed mercury values from 3 to 8 ng/l (see Table 3-2).

These measurements from Boston Harbor/Massachusetts Bays can be compared to those measured by Battelle (1991) for USEPA in the New York/New Jersey Harbor areas to evaluate ambient conditions.

Their results are summarized in Table 3-3. As an additional point of reference, the average mercury concentration in surface waters of the open ocean is reported at about 0.4 ng/l (Li, 1991).

3.4 Mercury Loading from MWRA

The MWRA contribution to mercury loads has been estimated to be 643.8 kg/yr (STFP, 1988) from primary treated and 205 kg/yr for secondary treated effluents. An in-house study by MWRA (1989) estimated a total discharge of 416.9 kg/yr into the harbor with the following breakdown: 16% industry; 67% POTW (279 kg/yr); and 17% CSO. Menzie and Cura (1991) estimated a total loading of 220 kg/yr from MWRA (effluent and sludge) into Boston Harbor. The MWRA NPDES FY93 report estimated a mercury loading of 120.9 kg/yr into Boston Harbor from effluent only. Sludge discharge into Boston Harbor ceased in 1991.

In response to some of the concerns discussed in Section 3.2, MWRA initiated an effluent characterization study in 1993 utilizing careful sampling procedures and low detection limits. Figure 3-1 shows the concentration of mercury in Deer Island effluent from June 1993 to August 1996. The final measurement of 340 ng/l in August 1996 is suspect and subject to further quality control/assurance. The Deer Island effluent concentration may be combined with flow data to estimate monthly mercury loading over the period 7/94-6/95 (see Table 3-4).

The geometric mean mercury loading from Deer Island effluent in the period of July 1994 to June 1995 was 0.197 lb/d, equivalent to 32.6 kg/yr. It has been estimated that the Nut Island flow is about 50% of that of Deer Island and the total MWRA mercury loading can be estimated by multiplying the Deer Island load by a factor of 1.5. Since 1994, the effluent characterization report has estimated total MWRA loading (combined Deer Island and Nut Island flows) using this methodology. These estimated combined flows are presented in Table 3-5.

It is useful to note that 50 kg/yr is about equivalent to a concentration of 104 ng/l Hg at a combined flow of 376 MGD (average of Deer Island and Nut Island flows from 1992 to 1995). The 1991 study for the New York/New Jersey Harbor cited in Table 3-3 sampled fifteen sewage effluents, reporting a geometric mean of 26 ng/l for mercury. This probably consisted of both primary and secondary treated effluents. The mercury level in secondary treated effluents from the Minneapolis area were reported by Balogh, et al. (1997) to be about 10 ng/l.

The MWRA pilot treatment program also employed low detection limits and provided valuable information (MWRA, 1995b; 1996b) for projecting future effluent mercury concentrations when secondary treatment is implemented and the 1995 results are summarized in Table 3-6. The geometric mean secondary effluent mercury concentration was 32 ng/l, indicating that secondary treatment can remove close to 52% of the primary effluent mercury load. Application of this removal efficiency to the estimated

combined effluent load provides an estimate of the projected future load. The projected mercury load after secondary treatment is implemented is then 50 kg/yr * (1.0-0.52) or about 24 kg/yr, compared to the STFP projection of 205 kg/yr.

3.4.1 Potential Further Decreases in MWRA Mercury Loading

A study by Burnam et al. (1994) reported a mercury load to the POTW of 0.8 lb/d in Sacramento, with 50% of that load from residential sources, equivalent to 181 g/d. The MWRA 1992 (Camp, Dresser and McKee) evaluation and estimation of toxic pollutants estimated a total mercury influent load of 340 g/d and that about 26% of the influent mercury load came from residential sources (88 g/d). These results are qualified in that the Sacramento study did not report on the sampling and analytical procedures used for measurement of mercury, and the MWRA study's detection limits for mercury was 200 ng/l, and relied on default values and statistical estimation for concentrations because many samples were below detection limits.

Further decreases in the mercury load in the MWRA influent waste streams are possible. One of the most common uses of mercury is in dental amalgams. The use of mercury amalgam in the USA in late 1960 and mid 1975 was estimated by Clark (1992) to be about 9 metric tons, which is about 45 milligram per capita per year. Epstein and Skavroneck (1996) reported a 1993 study in the Duluth area that showed a mercury loading of 6.3 kg/yr from dental offices to the Western Lake Superior Sanitary District. The amount of mercury discharged is about 0.056 grams per capita per year or 153 µg per capita per day if we assumed a population for Duluth/Superior of 112,627. A similar extrapolation to MWRA with a service population of about 2.5 million indicates that dental amalgam can be responsible for up to 383 grams of mercury per day in the MWRA system. The MWRA Toxic Reduction and Control Department has studied the discharge of mercury to the sewer from dental offices. The preliminary results indicate that the total mercury load from dental offices ranges from 46 g/d to 271 g/d (MWRA TRAC, 1997).

It is therefore conceivable that the total MWRA effluent mercury load can be decreased by pollution prevention at the source through the use of some sort of modified retention device or "trap" at the dentist's office, similar to traps for retention of silver wastes in photo finishing laboratories, or through changes in dental amalgam composition to decrease the mercury content. Assuming that dental amalgam contributes about thirty percent of the MWRA influent mercury loading (100 g/d), and that this could be decreased by a factor of two, then the influent mercury load may be reduced by 15%. Thus, the amount of mercury discharged could be decreased to 20.4 kg/yr after secondary treatment is implemented (a secondary effluent concentration of about 27 ng/l).

3.5 Other Sources of Mercury Loading to Boston Harbor and Massachusetts Bay

Other sources of mercury to Boston Harbor and Massachusetts Bay include point sources such as industries with NPDES permits or other POTWs, riverine inputs (under average conditions or storm conditions), combined sewer overflow (CSO) discharges, and atmospheric deposition. These sources are described further below.

3.5.1 Industrial Sources

An MWRA 1989 in-house study estimated that up to 16% of the total 416.9 kg/yr mercury load into Boston Harbor came from direct industrial discharges. This suggests that approximately 66.7 kg/yr was due to these sources. A more recent study (MWRA, 1994) indicates that virtually all of this mercury is currently routed to the MWRA facility. For the purpose of this review, it was assumed that a trace amount (i.e., 1 to 2 kg/yr) may be discharged from various minor industrial discharge (e.g., cooling water and/or stormwater runoff).

3.5.2 Rivers

It was surprising that virtually no direct measurement of mercury has been performed recently in large coastal rivers in Massachusetts. Most monitoring programs have focused on the quantification of mercury in fish tissue. As previously mentioned, measurements from the Fort Point Channel in November 1995 showed values between 20 and 30 ng/l, both upstream and at the channel mouth. Mason et al. (1994) assumed a riverine concentration of 5 ng/l in their global mercury model. A study by the state of Wisconsin (Webb, 1993) on background metal concentrations reported a range of 0.65 to 6.12 ng/l for mercury, with an arithmetic mean of 3.04 ng/l from fourteen samples. Values of 14 ng/l to 40 ng/l for the Hudson River and the Arthur Kill River were reported by Battelle (1991). In contrast, mercury concentration in the Madeira River (a Brazilian Amazon river with documented mercury contamination from gold mining) has been measured by Nriagu et al. (1992) at an average of only 24.6 ng/l (12 samples with a standard deviation of 4.5) once careful attention was paid to sampling and analytical techniques.

The typical mercury concentration in coastal rivers was assumed to be no more than 10 ng/l for this report. The average river flow into Boston Harbor used for this calculation was 36 cubic meters per second (Menzie-Cura, 1995). The riverine mercury load to Boston Harbor is thus calculated to be 11.4 kg/yr.

3.5.3 Combined Sewer Overflows

The 1989 MWRA in-house study also estimated that up to 17% of the total 416.9 kg/yr mercury load into Boston Harbor came from CSOs. The CSO study utilized standard analytical techniques with 200 ng/l as a detection limit for mercury. In addition to the high method detection limit, it is unknown whether the highly concentrated "first flush" samples were properly sampled and processed. The CSO mercury load is thus estimated at 70.9 kg/yr assuming these qualifications. More recent data from a study utilizing more sensitive detection limits (50 ng/l) and better defined flow estimates from actual NPDES measurements from the period 1995-present, showed average mercury concentration of 350 ng/l from the major CSOs discharging into Boston Harbor and an average flow of 1.69 billion gallons per year, which translates to a loading of 2.3 kg/yr (Hall, 1997). Accordingly, this last estimate was considered more representative.

3.5.4 Atmospheric

The technical advisory group to the Massachusetts Executive Office of Environmental Affairs (EOEA) reportedly recommended in 1988 the use of an estimated atmospheric deposition rate of 330 micrograms per square meter per year ($\mu gm^{-2}yr^{-1}$) for mercury from the atmosphere to the sea surface. Table 3-7 shows more recent estimates for atmospheric mercury deposition rates in various locations. An inspection of Table 3-7 indicates that an atmospheric deposition rate of 330 $\mu gm^{-2}yr^{-1}$ for mercury from the atmosphere to the sea surface represents an overestimate for this rate. The value of 40 to 80 $\mu g/m^2/yr$ used by MADEP was selected as more representative. The Boston Harbor surface area is about one hundred (100) square kilometers. Applying the estimated deposition rate, the atmospheric load of mercury to Boston Harbor is about 4 to 8 kg/yr.

3.6 Total Loading to Boston Harbor

Sung (1991) has used a simple analysis to relate harbor-wide averages of trace metals (copper and zinc) to metal loading rates. The harbor-wide concentration of copper and zinc was found to be approximately equal to 3.8×10^{-12} days/liter multiplied by the metal loading in grams/day. The factor 3.8×10^{-12} was related to the volume of water and the average residence time of water in Boston Harbor. This factor will increase if POTW discharges in the early to mid 1980s were slightly overestimated. It was suggested that the POTW loads were high by a factor of 2 to 3 (Hall, 1997) and so the proportion factor could be as high as 11.4×10^{-12} days/liter. Recent estimates of hydraulic residence time in the harbor ranged from 2.5 to 9 days and the harbor volume has been estimated as between 6.2 to 7.2×10^8 m^3 (Signell and Butman, 1992; MWRA, unpublished). This can be combined to be a range of 3.5 to 14.5×10^{-12} d/l for the slope factor. This encompasses the range of 3.8 to 11.4×10^{-12} d/l used for this analysis. The Central Artery/Third Harbor Tunnel project monitoring results described previously indicated a harbor-wide mercury concentration range of about 3 to 30 ng/l. If mercury behaved similarly to copper and zinc, the

range of 3 to 30 ng/l would imply a loading of 0.79 to 7.9 kg/d, or 288 to 2881 kg/yr of mercury, using the 3.8 x 10^{-12} factor. Table 3-8 summarizes the mercury loads described in Section 3.5 and compares that to this implied mercury load.

The mercury load accounting for Boston Harbor (Table 3-8) of 69 to 99 kg/yr is about the same order of magnitude as the lower range of 288 kg/yr described above, 11.4 x 10^{-12} days/liter multiplied by 188 g/d (69 kg/yr) equals 2 ng/l. Referring to Table 3-2, this appears reasonable for the monitoring results of 1992. It appears that 30 ng/l is too high a value for a harbor-wide mercury concentration. The reasons for this apparent overestimate are not known but sampling errors may have occurred in the 1993 to 1995 monitoring results or mercury may not have followed the same dilution analysis as copper and zinc. If the same dilution analysis used by Sung (1991) holds, then an unaccounted mercury load on the order of magnitude of nearly 2,000 kg/yr is produced. In order to account for this value, it would be necessary to have a large, otherwise unmeasured source. For example, the amount of contaminated soils and sediments mobilized by the Central Artery/Third Harbor Tunnel project would need to be on the order of about two million metric tons per year if the mercury concentration of mobilized contaminated soil and sediments was about one ppm. It is unlikely that such a major source has not been previously addressed. The most reasonable explanation is that the harbor-wide concentration of mercury is closer to 5 ng/l, and that the monitored mercury concentrations include higher concentrations which reflect local impacts from construction activities. The best estimate of MWRA contribution to mercury loading to Boston Harbor is thus from 50 to 75 kg/yr out of 69 to 99 kg/yr or about 73 to 76% of the total mercury load if the above analyses and assumptions are correct. This analysis also assumed no loss terms (volatilization, sedimentation).

3.7 Effect of Moving the MWRA Discharge to Massachusetts Bay

One of the obvious consequences of the MWRA discharge being moved to Massachusetts Bay is that the MWRA contribution to total mercury load to the Bay will decrease on a percentage basis. The secondary treated effluent will have an estimated mercury concentration of about 27 (if the prevention of dental amalgam pollution has been implemented) to 32 ng/l. Table 3-9 summarizes the expected contributions from various sources once the discharge has been moved. This was calculated by assuming a total river flow into Massachusetts Bays at 180.4 cubic meters per second (5.7 x 10^{12} l/yr) and an average river mercury concentration of 10 ng/l. An area of four thousand square kilometers for Massachusetts Bays and a range of 40 to 80 $\mu g/m^2/yr$ was used to calculate the atmospheric deposition loading. Other POTW discharges were calculated by a total flow of 1.9 m^3/s of primary treated effluent of 100 ng/l (mainly south Essex) and a flow of 2.5 m^3/s of secondary treated effluent of 32 ng/l (Lynn). A total CSO flow (1.69 billion gallons per year) assumed at 350 ng/l was used to calculate the CSO load. The flow values were obtained from Menzie-Cura (1995). The total mercury load to Massachusetts Bays was estimated at 243 to 406 kg/yr. The future contribution from the MWRA outfall is about 4 to 6% of the total mercury load to the Massachusetts Bays.

The acute and chronic ambient water quality criteria (AWQC) for mercury in marine waters are 2100 and 25 ng/l respectively. The secondary treated effluent will be well below the acute AWQC and will meet the chronic AWQC with minimal dilution. The anticipated regulatory dilution is 52 for acute and 68 for chronic conditions. The projected secondary effluent mercury concentration of about 27 to 32 ng/l will be diluted to less than the chronic water quality criteria and near ambient levels within the immediate vicinity of the outfall. In fact, it is likely that the concentration will fall to ambient levels within 50 feet of the outfall.

3.8 Implications for Monitoring

The implications of the effluent mercury for monitoring efforts is considered in this section. The current mercury concentration in the vicinity of the present outfall is close to 4 ng/l. Minimal dilution of the projected secondary effluent of 27 to 32 ng/l will make it indistinguishable from ambient conditions. Monitoring for changes in the ambient mercury concentration in seawater would be a real challenge and would likely be practically indistinguishable from natural variation.

In addition, monitoring for changes in the sediment environment poses similar difficulties. Secondary treated effluents typically contain 20 to 30 mg/l total suspended solids (TSS). Mercury is commonly associated with fine particulates. So the suspended particulates will have a mercury concentration of about 32 ng/l divided by 30 mg/l, or 1.06 ppm, which is then subject to transport, settling and dilution. A similar analysis applies to primary treated effluents (about 100 ng/l mercury and TSS of 100 mg/l). The mean and median mercury concentration in Boston Harbor sediments were reported by Manheim and Hathaway (1991) to be 1.07 and 1.3 ppm respectively. Unpublished results by Wallace showed sediment samples with mercury concentration greater than 1.0 ppm in the outfall vicinity already, although most samples were below 0.5 ppm. Thus, it appears that neither water quality measurements nor sediment monitoring will be sufficient to detect the small predicted changes in ecosystem characteristics due to MWRA effluent mercury. In particular, these methods are insufficiently sensitive to provide a reliable means to ensure a lack of impacts to sensitive marine receptors.

Since the quantification of water quality or sediments will not allow the detection of subtle changes in ecosystem quality, an alternative means of assessing potential mercury impacts is needed. It would seem that mercury impacts are better and more dependably monitored by the use of bioaccumulation and the relation of body burdens of indicator organisms to observed toxicological endpoints of target organisms. This method is already being used as an integral part of the MWRA monitoring program in the Fish and Shellfish Monitoring Program. Application of measured tissue residues to infer potential adverse ecological impacts to sensitive marine receptors is discussed in Section 4 of this report.

3.9 Conclusions

The current and future MWRA discharge of mercury into Massachusetts Bay is a small percentage of the total mercury load into the system. The projected secondary effluent concentration of 27 to 32 ng/l will be diluted to less than the chronic AWQC for mercury in the immediate vicinity of the outfall. Due to limitation in the ability to detect trends in water column or sediments, mercury impacts are best monitored by the use of bioaccumulation and relating body burdens of indicator organisms to observed toxicological endpoints of target organisms.

It is also recommended that mercury concentration should be measured in the Merrimack River to better quantify the contribution by rivers. This should be done during the spring flood season to capture the largest suspended sediment load.

TABLE 3-1

Total Mercury in Massachusetts Seawater

Station	Latitude	Longitude	Mean total Hg in ng/Kg (n = number of samples)
Deer Island Light	42° 20.36'N	70°57.71'W	< 16.3 (n = 7)
Deer Island Piling	42° 20.90'N	70° 57.71'W	< 14.3 (n = 7)
Nut Island Spindle	42° 17.62'N	70° 57.44'W	< 13.2 (n = 7)
Station 5	42° 22.67'N	70° 47.00'W	< 19.8 (n = 5)
Massachusetts Bay	42° 25.30'N	70° 38.77'W	< 9.5 (n = 8)

TABLE 3-2

Central Artery/THT Monitoring for Mercury

Station	1992	1993	1994	1995	1996
Charles River	3.2 (n = 2)		36.8 (n = 4)		
Aquarium	5.3 (n = 6)		43.6 (n = 5)	28 (n = 2)	
Reserve Channel	4.4 (n = 2)		51.6 (n = 4)		
Spectacle Island		44.8 (n = 4)		27.2 (n = 2)	4.5 (n = 4)

Geometric mean concentration in ng/l (n = number of samples)
CA/THT (1997) Spectacle Island Water Quality 1996 Revised Monitoring Program. November 1996 Sampling

TABLE 3-3

Ambient Mercury Levels in New York/New Jersey Harbor

	Geometric mean total Hg in ng/l (n = number of samples)
Oceanic	2.4 (n = 4)
Bight area	6.3 (n = 9)
Estuarine	6.7 (n = 7)
Riverine, Arthur Kill	39.5 (n = 9)
Riverine, Hudson	14.1 (n = 12)

TABLE 3-4

Mercury Loading from Deer Island Effluent

Month	Deer Island Effluent Hg Loading (lb/d) and (kg/yr)
Jul 94	0.22 (36)
Aug 94	0.38 (63)
Sep 94	0.28 (46)
Oct 94	0.25 (41)
Nov 94	0.24 (40)
Jan 95	0.11 (18)
Feb 95	0.08 (13)
Mar 95	0.13 (22)
Apr 95	0.17 (28)
May 95	0.39 (65)
Jun 95	0.16 (26)

TABLE 3-5

Total MWRA Mercury Loading

Year	MWRA Hg load (kg/yr) from effluent characterization	Calculated
1993	63 (Uhler et al., 1994)	215 (Alber and Chan, 1993)
1994	75	
1995	50	

TABLE 3-6

1995 Pilot Secondary Treatment Program Mercury Results

Month	Removal Efficiency (%)	Secondary effluent Hg (ng/l)
January	62	17
March	86	10
May	25	60
June	15	71
July	17	62
August	-3	74
September	98	8
October	24	140
November	82	18
December	87	16

TABLE 3-7

Summary of Atmospheric Deposition Rates for Mercury

Area	Atmospheric Deposition Rate for Hg in micrograms per sq. meter per year	Source
Western Mediterranean	50	Clark, 1992
South Atlantic Bight	240	Clark, 1992
Tropical North Atlantic	21	Clark, 1992
Pettaquamscutt estuary, RI	22	Mason et al., 1993
Latitudes 30 -70°N	7.7	Mason et al., 1994
Lake Champlain, VT	15	MADEP, 1996
Massachusetts	40 to 80	MADEP, 1996

TABLE 3-8

Summary of Estimated Mercury Loads to Boston Harbor

Source of mercury to Boston Harbor	Loading (kg/yr)
MWRA	50 to 75 from 1993 to 1995
Industrial	1 to 2
Rivers	11.4
CSO	2.3
Atmospheric	4 to 8
TOTAL	68.7 to 98.7, rounded to 69 to 99

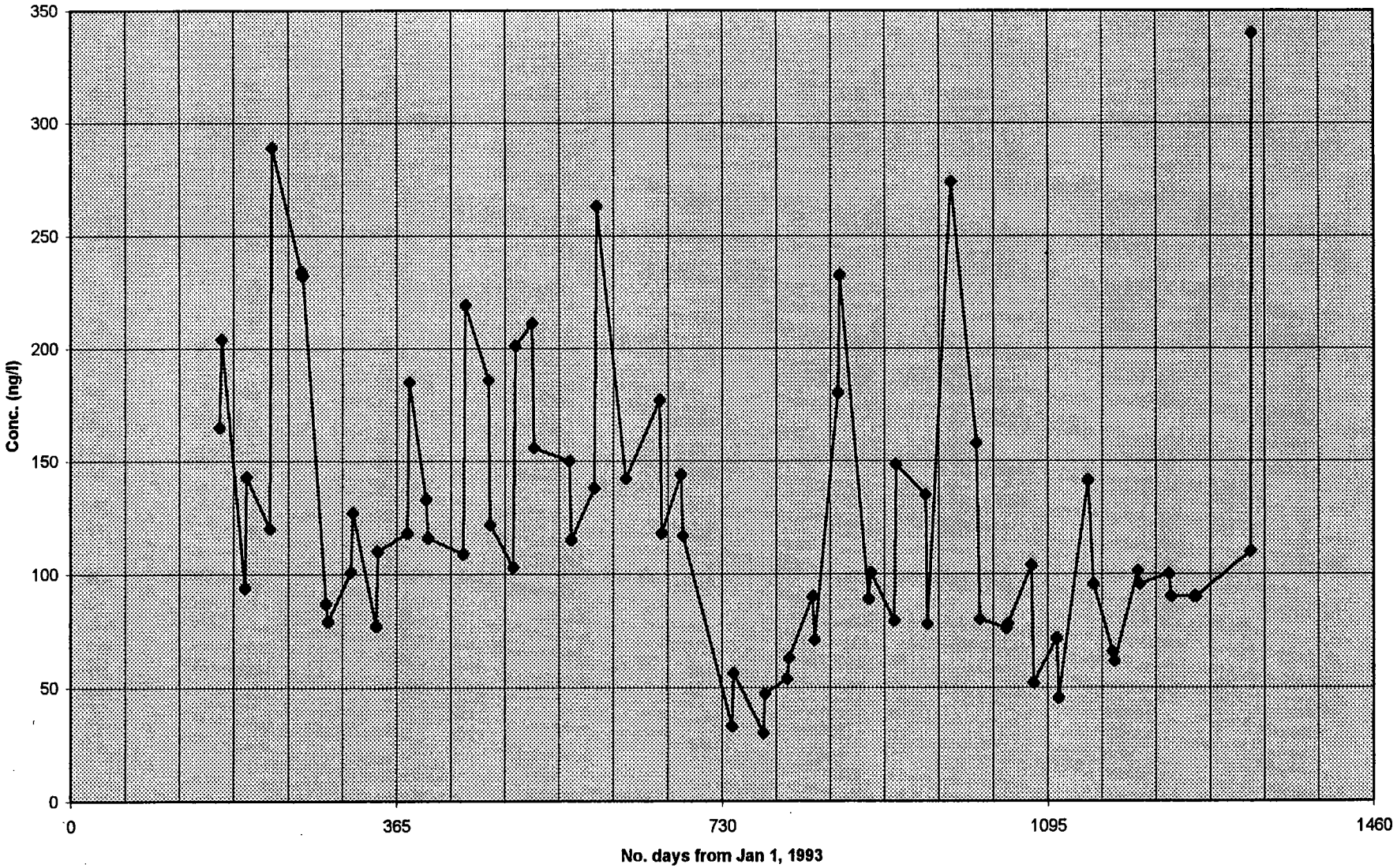
TABLE 3-9

Summary of Mercury Loads to Massachusetts Bays

Source of mercury	Loading (kg/yr)
MWRA	14 to 16.6
Rivers	56.9
Atmospheric	160 to 320
Other POTWs	8.4
CSO	2.3
Others (assume the same as Boston Harbor)	1 to 2
TOTAL	242.6 to 406.2, rounded to 243 to 406

Figure 3-1

Mercury in Deer Island effluent



4.0 EVALUATION OF POTENTIAL BIOACCUMULATIVE EFFECTS

This section presents an evaluation of the potential impact of bioaccumulative contaminants present in the current and future outfall discharge on the most endangered of the local marine mammal species, the northern right whale. The bioaccumulative contaminants examined include PCBs, pesticides, and mercury as well as non-bioaccumulative PAHs.

4.1 Overview

In 1993 a comprehensive review was prepared of the possible scenarios that might lead to impacts by the MWRA sewage outfall on endangered species (U.S. EPA 1993). That document reviewed the physical environment, the natural history of federally-listed "endangered" species known to frequent Massachusetts and Cape Cod Bays, and assessed the possible nutrient, biotoxin and chemical impacts to those species. The primary conclusion of that review was that past and future impacts would be negligible. The primary reasons for this were: 1) the lack of endangered species habitat in the near vicinity of the current or future outfall, and 2) the absence of any association between observed mortalities and the MWRA discharges. This conclusion was presented with detailed supporting data. Data was presented on all of the relevant endangered species, namely right, blue, fin, humpback, sperm and sei whales, three sea turtle species and one sturgeon species.

The first assumption, that no appropriate habitat is adjacent to the near field of the future outfall, has been more recently tested by increasingly systematic observations during MWRA Survey Cruises from 1993 to the present. In 1993 (Hamilton, 1993) reported no sightings during a single 3 day cruise. A summary of incidental observations during 1993 (Bechtold and Hunt, 1994) reported sightings in the "far field", but none in the "near field," with near- and farfield being defined in terms of the future MWRA outfall location. In both 1995 and 1996 there was a dedicated marine mammal observer aboard all water column survey cruises. Those data are summarized in Table 4-1. It can be seen that the nearfield area is marine mammal habitat, including, at least transiently, endangered species (fin and right whales). A single right whale was sighted April 5 1996 between near field stations N10 and N11.

The second assumption, that lack of linkage between MWRA effluent and mortalities leads to zero impact of MWRA effluent, fails to consider chronic sub-lethal effects such as: failure to gain adequate body condition to reproduce, immunosuppression, and other chronic non-lethal effects. That assumption will be examined in the latter part of this review.

The general questions that remain are:

1) Do current or future MWRA outfall effluent chemical contaminants in general pose a threat to marine mammal species now known to frequent the outfall area as shown above?
2) What biological effects of those contaminants might be important?
3) What proportion of the regional threat that might exist should be assigned to the MWRA outfall? This last question is addressed elsewhere in this review, primarily in Sections 2.0 and 3.0.

This section will present information in the following three major sections:

1. Data on tissue residues or body burdens in marine mammals from the Gulf of Maine and other areas;
2. A review of the feeding ecology of target species; and
3. An assessment of potential toxic effects in marine mammals including right whales.

These sections are followed by conclusions, recommendations, and supporting tables and figures.

4.2 Data Availability, Quality, and Interpretation

The assessment is focused on the northern right whale, given its highly endangered status, and the presence of substantial critical habitat in Cape Cod Bay. The amount of relevant data for this species is limited to one published paper (Woodley et al., 1991) and unpublished data (Moore et al.). However, there are also a number of other marine mammals in Massachusetts and Cape Cod Bay, for which there is somewhat more information available on a regional and global basis. Those data are used as background material.

Assignation of biological significance to measurements of specific chemical contaminant burdens in animals from the wild is extremely difficult. The factors that contribute to this problem include:

1. The presence of a complex mixture of inorganic and organic contaminants in all samples. The manner and extent to which compounds and elements interact, synergise and antagonize is usually poorly understood.
2. Gender, reproductive state, time of year, migration history and age all impact both the likely contaminant burden and the likely effects of contaminants present. Most marine mammal contaminant burden and effects data sets are severely limited by sample size.
3. Laboratory analysis and data reduction methods are often poorly comparable between laboratories, although this has recently become less of a problem as methods become increasingly standardized and inter-calibrated.

4.3 Summary of the Current Database on Tissue Residues in Marine Mammals in the Gulf of Maine with a Comparison to Historical Levels

There is a scarcity of data from the Gulf of Maine on contaminant burden in target marine mammal species. Moore, Weisbrod, Miller, Shea and Stegeman have a manuscript in preparation that describes contaminant burdens, and cytochrome P4501A expression in pilot whales and white sided dolphins stranded on Cape Cod. There are data on three Gulf of Maine harbor porpoises, one from Boston Harbor, and two from Boothbay harbor, ME, as described in 4.3.1 below. There are also data available for a pilot whale from Cape Cod Bay archived at the National Institute of Standards, as described in 4.3.2 below.

To attempt to put each species and its tissue residues into context, we first present a summary of a comprehensive consistent data set for mercury (Table 4-2) and selected organic contaminants (Table 4-3) from marine mammals sampled around the British Isles. This is then followed by regional comparisons for harbor porpoise, pilot whales, and balaenid whales (right/ bowhead). Additional material on winter flounder and blue mussels are also presented for reasons given below. The data in table 4-2 show that the single baleen whale analyzed was an order of magnitude lower for liver mercury than the toothed whales. Other data would predict that were right whales to be in this data set, they would be even lower. The data in Table 4-3 show comparable levels in the odontocetes examined, with highest levels in the bottlenose dolphin.

4.3.1 Regional Comparison for Harbor Porpoise *Phocaena phocaena*

The first regional comparison for a target marine mammal was conducted for harbor porpoises. Table 4-4 shows a comparison of organic contaminants in harbor porpoise blubber. The amount of data for Gulf of Maine animals is sparse, but it appears that these animals are comparable to the Gulf of St. Lawrence, the US West Coast, and the British Isles, and higher than animals from Newfoundland.

Table 4-5 shows mercury burdens for Harbor porpoises from the Gulf of Maine compared to the British Isles. Burdens are somewhat lower in the Gulf of Maine, although the sample size is very small.

4.3.2 Pilot Whales *Globicephala spp.*

Table 4-6 shows organic chemical contaminants in pilot whale compared around the world. Both long-finned (*Globicephala melas*), and short-finned (*Globicephala macrorhyncus*) are included. The single animal from Cape Cod has low levels in comparison to all other areas. This observation is supported by unpublished data (Moore et al.) on a larger sample set from other pilot whales stranded on Cape Cod. This relationship is probably driven by the fact that pilot whales in New England spend most of their time on the edge of the continental shelf, and only venture into coastal areas quite rarely - see 4.4.3.2 below. The two animals reported in 1975 from Rhode Island and Maine (Taruski et al., 1975) strongly suggest

that PCB burdens in this species may well have been higher in the 1970's, although we will never know, given the small sample size. Furthermore, these differences may reflect changes in analytical technique.

Table 4-7 shows mercury levels in pilot whales from Newfoundland and the Faroes. There is no heavy metal data available for Cape Cod animals. The high levels in the animals from Newfoundland suggest that anthropogenic influences may be less in this case.

4.3.3 Balaenid Whales *Eubalaena* spp. and *Balaena mysticetus*

There is very little contaminant data available in northern right whales (*E. glacialis*). The only published data are summarized in Table 4-8 (Woodley et al., 1991). These data are compared in the table to comparable data from bowheads (*B. mysticetus*) sampled in the subsistence hunt in Alaska. Bowheads are of a similar feeding ecology to northern right whales. They are primarily zooplanktivores. It seems that right whale values are low, but not as low as the bowheads, with exception of the bowhead PCB values which appear high. Most of the bowhead values for additional organic contaminants not shown in Table 4-8 are non-detects (Bratton et al., 1990). There are also unpublished data Westgate et al. (pers. comm.), that confirm the organochlorine concentrations in northern right whales being less than other toothed cetacean species listed above. Additional compounds in this later dataset include toxaphenes. A more recent set of biopsy samples collected in 1994 to 1996 has been analyzed for cytochrome P4501A induction in blubber endothelial cells (Moore et al., 1995), and will soon be analyzed for organochlorine content (D. Shea and A. Weisbrod, pers. comm.). These data, in comparison with similar samples from 14 other cetacean species show a level of P4501A induction that is higher than would have been predicted on expected organochlorine burden. Thus the possibility exists that this biochemical change reflects exposure to non-bioaccumulating compounds that also induce CYP1A such as polyaromatic hydrocarbons. Unfortunately there are no chemical or biochemical data from right whale populations elsewhere in the world where PAH exposure would be less. This situation may change after completion of a sampling effort in southern right whale habitat in South Georgia in February 1997 (Moore et al.). At this time it is reasonable to conclude that published data do not suggest major organochlorine burdens in this species, but that a recent abstract (Moore et al., 1995) of biochemical data on CYP1A induction leaves an open question of the role of chemicals such as PAH. This later data will be published with support from the MWRA over the next 12 months.

To our knowledge there are no heavy metal data available for right whales. A review of available data for other baleen whale species has been published (O'Shea and Brownell, 1994). Low levels have been reported for the liver of Alaskan bowheads (Bratton et al., 1990). In these bowheads total mercury in the livers ranged from 0.001 to 0.009 ppm. The highest liver mercury reported for a baleen whale is in minke whales off Greenland, at 2.7 ppm.(Hansen et al., 1990). These values were considerably lower than for belugas and narwhals from the same area.

4.3.4 Winter Flounder

Not a marine mammal, but a vertebrate of significance in the habitat of concern to this review is the winter flounder. The MWRA has funded extensive analyses of chemical burden and effect in this species from 1990 to the present (Mitchell et al., 1996; Moore et al., 1996). The relevant observations from this species to this review are the distinct reduction of contaminant burden and effect with increasing distance from the major population centers in the Greater Boston area. This observation is strictly relevant only to benthic -feeding organisms, but given the major role of the fallout of organic matter from the water column in generating benthic contaminant loads, it is a reasonable proxy in predicting the likely differences in exposure to marine mammals feeding in the water column of Cape Cod Bay as compared to near Boston or at the future outfall site.

4.3.5 Blue Mussels

In 4.3.3 above, we discuss the possible role of PAH in the biochemical effects reported in the northern right whale biopsies (Moore et al., 1995). Data for PAH burden in vertebrate animals is uninformative as they effectively metabolize most of these non-halogenated aromatic compounds. The metabolites either form macromolecular adducts, or they are excreted in urine and bile. In the absence of data from such excreta we can only approximate an assessment of PAH exposure to marine vertebrates. One such source of data is studies of PAH bioaccumulation by caged mussels conducted by the MWRA (Downey, 1994), as shown in Table 4-9.

These data are a good proxy for right whale exposure, were they to be feeding at the deployment sites, given the filter-feeding habit of this species. There is no expectation of bioaccumulation of PAH given the likely metabolic capacity of these vertebrates, but there will be a steady throughput of these compounds.

The future outfall site looks to be a low risk area at the time of sampling, but it must be considered that once the effluent is discharged at this site there will be a potential for increased PAH exposure in the near field.

Consideration of PAHs is critical in this assessment, as in contrast to the halogenated hydrocarbons, which are mostly under severe or total use restrictions, PAHs are the inevitable combustion products of the vast majority of our industrial and domestic heating and transportation in the northeast. Thus as the population grows, the ongoing input of PAHs to the coastal zone will be maintained. The role of PAHs in the marine environment is poorly understood, resulting in large part from their failure to bioaccumulate, which in turn makes exposure hard to monitor.

One further data source will soon be submitted for publication from the laboratory of D. Shea (North Carolina State University) in collaboration with M. Moore. There will be data available on water and plankton PCB, pesticide and PAH concentrations for stations from Cape Cod Bay, Bay of Fundy and Georges Bank. Preliminary analysis of this data set indicates a significantly lower contaminant burden in samples from Georges Bank, as compared to the inshore stations. Thus it can be assumed that northern right whales are consuming plankton that are more contaminated than would be the case if they had a major feeding habitat further offshore.

4.4 Comparison of the Trophic Structure and Feeding Habits of Northern Right Whale Food Chain to Comparable Marine Mammals; Discussion of Possible Implications Regarding Exposure to Outfall Contaminants

Information pertinent to this issue for the four species reviewed (right whale, harbor porpoise, pilot whale, beluga whale) includes: 1) the nature of the diet, 2) the proportion of the year spent in the Massachusetts Bays in general, and in the vicinity of the outfall in particular and 3) the chemical characteristics of other habitats visited during other parts of the migration cycle.

4.4.1 Right Whale

4.4.1.1 Right Whale Diet

Right whales filter feed on zooplankton. The primary species appears to be *Calanus finmarchicus* (Mayo and Marx, 1990). They also feed on other calanoid copepods and northern krill, *Meganichthyphanes* (Goodyear, 1996), at least in the Bay of Fundy.

4.4.1.2 Right Whale Residency in the Massachusetts Bays

Right whales are found in Cape Cod Bay in between December and April (Kraus et al., 1986; Winn et al., 1986). Sightings start in late December. They feed in the Great South Channel , off Nantucket, in the spring, and are found off Nova Scotia and in the Bay of Fundy from June to September. Animals that breed are found off the coasts of Georgia and Eastern Florida December through March. The location of non-breeding animals during the winter is unclear. The current satellite tagging program of the New England Aquarium right whale program should address this issue (Kraus pers. comm.)

4.4.1.3 Right Whale Exposure in Other Habitats

Recent satellite tagging studies (Christopher Slay, pers. comm.) suggest that right whales spend a substantial amount of time close to shore. A track (Figure 4-1) of a mother with calf that was tagged off

the Georgia coast in February of 1996 shows a close proximity to the coast. Thus they will be exposed to contaminants in food and water throughout their coastal migration. Contaminant exposure will include atmospheric deposition, riverine input, including pulp mill effluent both in the Bay of Fundy, and in Georgia, and all of the industrial/suburban/ urban effluent of the Eastern seaboard, in addition to sewage outfalls, such as that currently discharged at Deer Island in Boston Harbor, and soon to be located 9 miles further east.

4.4.2 Harbor Porpoise

4.4.2.1 Harbor Porpoise Diet

The summer diet, which consists almost exclusively of large herring found in the northern Gulf of Maine and the Bay of Fundy (Recchia and Read, 1989), shifts in the winter to include a greater diversity of species and size classes. The winter/spring diet is dominated by silver hake (*Merluccius bilinearis*). Herring (Atlantic, but probably alewife and blueback too), pearlsides, butterfish, and pollock also seem to be important in the winter/spring diet (Read et al., 1994). Additionally calves are known to eat euphausiids (Smith and Read, 1992).

4.4.2.2 Harbor Porpoise Residency in the Massachusetts Bays

Strandings of harbor porpoise in the Massachusetts Bays occur primarily in February through April (Greg Early, pers. comm., Damon Gannon, pers. comm.). In response to a request for a current assessment of the state of knowledge of porpoise migration in the NW Atlantic an authority in the species, A. Read (pers. comm. - Duke University) responded as follows: "The annual migration pattern of harbor porpoises has not yet been fully elucidated. From data on by-catches, sightings and satellite telemetry of individual movements we can piece together the following outline. The majority of the population spends the summer months (July - October) in the northern Gulf of Maine and Bay of Fundy. Animals then move southwest, generally along the 50-fathom line off the coast of Maine to the coast of New Hampshire and Cashes Ledge, where they spend the fall. The winter and spring distribution are poorly known, but at least some adults appear to remain in the Gulf of Maine, while many juveniles move south along the mid-Atlantic coast. Harbor porpoises are seen frequently in Massachusetts Bay only during the spring months, when they may move as far inshore as inner Boston Harbor."

4.4.2.3 Harbor Porpoise Exposure in Other Habitats

This piscivorous species is likely to be exposed to contaminants via the diet throughout its feeding lifecycle.

4.4.3 Pilot Whales

4.4.3.1 Pilot Whale Diet

Pilot whales are primarily teuthophagus (squid eaters) but their diet is considerably more diverse than initially reported. The long- finned squid (*Loligo pealei*) is their primary prey during March and April. Short-finned squid (*Ommastrephidae*), histioteuthid squid, clupeids, and mackerel (*Scomber scombrus*) make up most of the remainder (Gannon, 1995; Gannon et al., 1997). In respect to Massachusetts waters there are a number of caveats (D. Gannon, pers. comm.): no studies have been made of pilot whale food habits from Massachusetts Bay/Gulf of Maine; the vast majority of specimens used in food habits studies were collected in March and April, with a few taken in February and May; no information exists on the summer or fall diet; no adult male stomach has ever been sampled from U.S. waters. In spite of the above, it can be said that species that pilot whales are known to eat that occur in Massachusetts Bay include (in no particular order): *Loligo pealei, Illex illecebrosus, Clupea harengus, Merluccius bilinearis, Scomber scombrus, Squalus acanthias*, and *Urophycis spp.*

4.4.3.2 Pilot Whale Residency in the Massachusetts Bays

Pilot whales strand in the Massachusetts Bay primarily during the months of November and December (G. Early, pers. comm.). Sightings of animals in the Bays by whale watching boats are primarily in the Fall (D. Gannon, pers. comm.).

4.4.3.3 Pilot Whale Exposure in Other Habitats

When not inshore, pilot whales are primarily distributed over the outer half of the continental shelf (Payne and Heinneman, 1993).

4.4.4 Beluga

Beluga are not found in New England waters, but they are discussed here, as they are arguably the best case of contaminant-associated disease in wild marine mammals. The population of this species in the Gulf of St. Lawrence has suffered impaired reproduction, and has an elevated prevalence of tumors reported (Martineau et al., 1987; Martineau et al., 1988; Muir, 1990). The association between PAH and HAH exposure and these changes has been inferred.

4.4.4.1 Beluga Diet

Beluga are carnivores. They eat whatever fish and invertebrates are plentiful and easy to catch (Hazard, 1988).

4.4.4.2 Beluga Residency in the Massachusetts Bays

This area is outside of the range of beluga.

4.4.4.3 Beluga Exposure in Other Habitats

The Gulf of St. Lawrence appears to be a significant source of beluga contamination. Populations in the Arctic appear to be approximately 10x less contaminated than those in the St Lawrence (Muir et al., 1990).

4.4.5 Summary of Life History Implications for Potential Exposure

Chemical risk in the major right whale habitat in Eastern Cape Cod Bay, is best represented in the flounder data set as mentioned in 4.3.4 above, by Station 5 (Moore et al., 1996), situated between Wellfleet and Provincetown. Data from flounder taken at this station has been consistently at or close to baseline for the NW Atlantic, as defined by the same species on Georges Bank (Boehm and Hirtzer, 1982; Moore, 1991). Thus the flounder data, from a species that has a much less extensive migration than the marine mammals under review here, suggest that it is only when marine mammals venture into the western part of Massachusetts Bay that they will be likely to encounter increased contaminant exposure above baseline for the Gulf of Maine.

The single sighting of the tagged right whale shown in Figure 4-1 is a rare event in Western Massachusetts Bay. Between December and April, right whales are consistently encountered in the eastern half of Cape Cod Bay. Thus the current distribution of this species appears not to expose it to a contaminant burden that is significantly augmented by contaminants in the MWRA discharge within the near field. However, as described elsewhere in this review (see Sections 2.0 and 3.0), the current MWRA discharge represents a very important source of organic contaminant (e.g., PCBs and pesticides) loading to the Massachusetts Bay, Cape Cod Bay system. To that extent it is important to consider the contaminants in the MWRA effluent as being responsible for any possible effects on the right whale.

This percentage has to be further divided by the proportion of the year that individual whales spend in Massachusetts waters. Of the approximately 300 animals in the population, between 20 and 60 of them spend about 45 days a year in Cape Cod Bay (P. Hamilton, pers. comm). Therefore individuals that enter the bay spend 12% (45/365) of the year in the bay. Assuming equal exposure in all habitats they will thus receive 12% of their chemical loading from the bay. On a population basis, this represents between 0.7 and 2.4 % (20/300 x 12% to 60/300 x 12%) of the total chemical loading to the population, given the same assumption. One current unknown is whether activation of the future outfall will induce a significant change in marine mammal population distribution as a result of altered nutrient concentrations. Maintenance of the existing marine mammal observer presence on water column survey cruises should

ensure detection of this possibility, however it has been concluded previously (U.S. EPA, 1993) that such a scenario is unlikely.

4.5 Potential Ecotoxicological Effects to Northern Right Whales, Using Predicted Contaminant Levels in Future Outfall Discharge; Estimated Incremental Tissue Residue: Areas of Uncertainty

4.5.1 Predicted Contaminant Levels in Future Outfall Discharge

The scheduled implementation of the future MWRA outfall discharge will alter the existing loading of organic and inorganic contaminants to Massachusetts and Cape Cod Bays. Two changes will be caused by the implementation: (1) a shift in the location of the outfall and (2) change in the quality of the effluent as secondary treatment is phased in. The combined effect of these two changes will be to bring a less concentrated effluent closer to the potential marine mammal habitat. Since the effluent is expected to be quickly mixed into the water column, the effect of the closer outfall may result in more rapidly distribution of effluent into the Bays. However, it should be noted that the main concern with the contaminants of concern is with potential long-term chronic exposure and bioaccumulative effects through the food chain. Therefore, it is the average mixed water concentrations, rather than transient, localized conditions, which are important.

Analyses conducted in Sections 2.0 and 3.0 indicate that implementation of secondary treatment on the MWRA effluent should reduce the total loading to the Bays by approximately 46% for PCBs and 58% for pesticides. Mercury will be reduced by at least 52% in the MWRA effluent outfall. The projected MWRA effluent mercury load is only 4-6% of the estimated total mercury load. The results of these analyses suggest that existing ambient water quality in the Bays for PCBs, pesticides, and mercury should, at worst, stay at existing levels, or more likely, decrease as the secondary treatment of MWRA effluent is implemented.

4.5.1.1 PCBs

A recent review of PCBs in aquatic organisms (Niimi, 1996), states PCB concentrations in coastal waters to be in the low ng/l (i.e., 10^{-9} g/l) range, and in offshore the pg/l (i.e., 10^{-12} g/l) range. Plankton concentrations are usually in the low µg/kg range. Concentrations in fish range from the low µg/kg values to low mg/kg values for fish from more contaminated systems. The significance of PCB residues in mammals has been recently reviewed (Kamrin and Ringer, 1996). Health effects in mammals can include teratogenic changes, infertility, embryo and fetal mortality, abortion, low birth weight and behavioral changes. Experimental studies in mink have shown liver levels above 4 ppm and fat levels above 10 ppm to be lethal. Lower concentrations have been associated with a diversity of reproductive problems.

However it is important to realize that chronically acquired burdens in wild animals may well exceed burdens shown to be detrimental in experimental animals if the wild animals have inherited or acquired a resistance to elevated chemical exposure. Such resistance is being found to be widespread in fish from heavily contaminated sites (Weis and Weis, 1989). Nonetheless, study of seals fed herring from the relatively contaminated Baltic sea, as compared to herring from the North Sea, showed significant immunosuppression associated with exposure to elevated PCB and other chemical burdens in the Baltic fish (Ross et al., 1995). Indeed the concentrations of 10's of ppm's of total PCB's in many of the marine mammals reviewed in Section 4.3 above would strongly suggest that such mechanisms are at work in marine mammals. There has been speculation and controversy over whether such chemical effects contributed to the various morbillivirus epizootics reported in marine mammals from various parts of the world in the past decade (Aguilar and Borrell, 1994; De L. Swart et al., 1994). Knowledge of chronic burdens being present and consistent with ongoing life in no way excuse our society from being concerned about the implications of such levels, and from exploring possible ways of reducing them.

4.5.1.2 Pesticides

Pesticides include compounds such as DDT, DDD, DDE, aldrin, dieldrin, endrin, heptachlor, chlordane, mirex and others. In recent reviews (Peakall, 1996) of the likely effects of these compound on wild vertebrates, changes associated with exposure to these compounds include effects on reproduction and behavior.

4.5.1.3 PAHs

In contrast to PCBs, pesticides and mercury, PAHs do not bioaccumulate to a marked degree. They are readily metabolized by vertebrates with resultant excretion. However the larger four and five ring compounds could have significant toxic effects given chronic exposure, with reproductive, immunological and other systemic effects comparable to halogenated compounds, depending on the specific structure activity relationships.

4.5.1.4 Mercury

Inorganic mercury enters the aquatic environment either by natural release, which occurs primarily through degassing of the earth's crust, or from anthropogenic sources (Law, 1996). This author provides a review of mercury in marine mammals. The inorganic form is methylated by microorganisms in freshwater and marine sediments. The methylated forms are more readily bioaccumulated, and most toxic. Marine mammals can demethylate mercury, and store it in the inorganic form. Transplacental transfer occurs of the methylated forms. Concentrations increase with age, being less than 1 ppm mg/kg wet weight in fetal marine mammals. The liver is the primary site of accumulation. Short term (< 100 days) experimental studies of feeding mercury to seals (Ronald et al., 1977) induced chronic renal failure and death.

However, chronic accumulation to higher levels appears to be non-lethal in many marine mammals. Selenium is known to be an antagonist to methylmercury toxicity, and it seems that marine mammals are able to tolerate extremely high levels of liver methylmercury by sequestering inert mineralized deposits of mercuric selenide in the liver (Andre et al., 1991).

4.6 Assessment of Potential Effects of MWRA Outfall on Right Whales

There are two related questions to address in this assessment:

1. Are northern right whales significantly exposed to toxic contaminants?
2. What proportion of those contaminants are derived from the MWRA outfall?

4.6.1 Are Northern Right Whales Significantly Exposed to Toxic Contaminants?

The only published data relating to the question of exposure are analyses of contaminants in blubber biopsies reported by Woodley et al. (Woodley et al., 1991). These data show a relatively low level of bioaccumulation as compared to other marine mammals, see Section 4.3.3 above. Total PCB ranged from non-detectable to 1.9 mg/kg wet weight. In a review of contaminant burdens in baleen whales it was concluded that contaminant burdens were less of an immediate problem than habitat and human-induced mortality (O'Shea and Brownell, 1994). In this review the ranges of total PCB concentrations were compared between different species. Total PCB concentrations ranged from non-detectable to 30 mg/kg wet weight. Highest burdens were in single fin and minke whales from the St Lawrence River, and a minke whale from California. Levels in animals from the Southern ocean were consistently at 0.01 mg/kg or lower.

One further anomalous observation that has not yet been fully explored is the unpublished data on cytochrome P4501A expression in blubber endothelial cells sampled by biopsy (Moore et al., 1995). As discussed above in 4.3.3, these data may suggest a significant PAH exposure in right whales, possibly as a result of their filter feeding habit. At this time such a suggestion is speculative, but is currently being investigated further, in part with support from the MWRA and the Regional Marine Research Program of the Gulf of Maine (Moore et al., unpublished data) to look closer at right whale chemical dynamics in the Gulf of Maine, and by collecting reference southern right whale biopsy samples from the Southern Ocean in February 1997.

The levels reported by Woodley are obviously lower than those shown for toothed whales in section 4.3 above. However, it is important to consider differences in life history that may underlie the low levels in baleen whales, and may actually underestimate the potential risk from contaminant exposure. Baleen whales tend to migrate to tropical and subtropical latitudes to breed and give birth. In contrast, toothed whales in temperate latitudes tend to not migrate more than a few hundred miles through the year,

maintaining a relatively steady input of food. Thus baleen whales will have an annual opportunity to depurate during their seasonal starvation, in contrast odontocetes will have a year round intake of contaminants. This seasonal starvation could lead to enhanced systemic exposure as the blubber and muscle lipid is depleted

4.6.2 Evidence for Biological Effects of Chemical Contaminants in Marine Animals

The possibility that contaminants are affecting marine mammals (Reijnders, 1995) has been supported in studies of seals (Reijnders, 1986) and cetaceans (Subramanian et al., 1987). A disease syndrome thought to be associated with exposure to chemical contaminants is endemic in beluga in the St. Lawrence estuary (Martineau et al., 1988). It is thought that the beluga feed significantly on contaminated eels migrating from Lake Ontario, which are probably a major contaminant source (Hodson et al., 1994). Effects possibly exacerbated by chemical exposure include tumors, nocardiosis and viral dermatitis (Martineau et al., 1988). There have been suggestions that the reproductive physiology in these animals has been chemically compromised but these suggestions have been questioned (Addison, 1989). But in general, the role of lipophilic contaminants in the failure of many whale stocks to recover after the cessation of commercial exploitation remains an open issue. Growing experimental and environmental evidence implicate both PAH and HAH in reproductive toxicity. PAH and HAH may cause impairment by altering metabolism of hormones, by parent compounds or metabolites acting either as estrogens or as anti-estrogens, or by being activated to toxic products that may damage reproductive tissues or developing tissues. Studies of the effects of PAH and HAH on reproduction have been described in many taxa. Aromatic and halogenated hydrocarbons have been shown to induce gonadal dysplasia and infertility in mammals (Mclachlan et al., 1981). A study in mice showed that benzo(a)pyrene, an archetypal PAH, inhibited corpus luteum formation in the ovary (Mille et al., 1992). A field study of Forsters terns in Lake Michigan showed reduced hatchability and parental attentiveness in birds from a PCB contaminated site (Kubiak et al., 1989). Likewise, PAH and HAH have been shown to affect reproduction in fish (Thomas, 1990).

There has also been extensive speculation that some of the mass mortalities of seals and dolphins have resulted from organochlorine-induced immunosuppression facilitating subsequent fatal viral infections. In particular, morbilliviruses such as the phocid and delphinid distemper viruses have been prominent in the past few years. Recently the first evidence supporting this hypothesis has become available (Ross et al., 1994). It is quite possible that these viral agents are dormant as subclinical infections in particular species that are apparently not susceptible to fatal epidemics with this virus, such as the long-finned pilot whale (Duignan et al., 1995).

4.6.3 Effects of Mercury

In Florida the livers of stranded bottlenose dolphins, *Tursiops truncatus*, were examined and abnormalities correlated with the level of mercury (Rawson et al., 1993). The range of values for the liver was 0.01 to 443 µg/g wet weight. Abnormalities were found when levels exceeded 61 µg/g.

In the Faroe Islands pilot whales contribute significantly to the human diet. A recent landmark study (Weihe et al., 1996) has shown some disturbing trends. Pilot whale muscle contains an average of 3.3 µg/g of which half is methylmercury. Blubber concentration is 30 µg/g. These concentrations lead to a dietary intake close to the WHO recommended weekly intake limit of 0.3 mg. Mercury burden in human maternal hair and umbilical cord blood have shown positive correlation's with several later neurobehavioral tests in the children of these mothers. Studies are currently underway for relationships between PCB burden and the same battery of tests.

It is obviously impractical to repeat this kind of study to look for cognitive effects of contaminants in cetacea, but the above study certainly raises the question as to whether contaminants may affect both the health and behavior of marine mammals in the wild.

4.6.4 Reproductive Status of Northern Right Whales

The northern right whale population is failing to recover as it should, in spite of the absence of whaling mortality. The total population numbers only 300 individuals, of which about half are sexually mature. The mean increase of numbers of animals in the population is 2.5% per year (Knowlton et al., 1994). In contrast, the South American and South African populations, in the less polluted South Atlantic, have rates of 6.8% (Best, 1990) and 7.5% (Payne et al., 1990) respectively. Likewise bowheads from Alaska are known to have a rate of 5.2% (Koski et al., 1993). Only 38% of the mature female northern right whales are known to be reproductively successful in the NW Atlantic population (Brown et al., 1994), in contrast to 54% in the SW Atlantic population (Payne et al., 1990). Furthermore, observations of abnormal external genitalia have been made in the NW Atlantic population (Kraus, pers. comm.). The recruitment failure of the NW Atlantic population probably has multiple causes: about half of the shortfall has been accounted for by the combined effects of ship collisions and fishing gear entanglement (Kraus, 1990). The remaining factor(s) are thought to include a loss of viability from inbreeding, and/or reproductive failure resulting from exposure to toxic chemicals. Attempts to consider the inbreeding issue have been inconclusive (Schaeff et al., 1993). The lower levels of contaminants in southern right whales may be predictive of a chemical impact on NW Atlantic right whales. Certainly of the bowhead and the northern and southern right whales, the northern right whale occupies the most industrialized habitat.

4.6.5 Copepods

Copepods are lipid rich crustaceans, and the prey of right whales, *C. finmarchicus*, consist of 30-40% lipid dry weight (Marshall and Orr, 1972). This suggests that they could accumulate lipophilic foreign compounds, and thereby contribute substantially as a reservoir and source to other parts of the system. Experimentally, copepods are known to accumulate PCB's and eliminate them via fecal pellets and eggs (McManus et al., 1983). A study has shown that zooplankton fecal pellets were the major route by which PAH's reached the sediments of a coastal bay (Prahl and Carpenter, 1979). When a coastal copepod, *Centropages* was exposed to crude oil (Cowles and Remillard, 1983), egg viability was reduced, but accumulation did not occur. These authors also concluded that this organism could metabolize petroleum hydrocarbons, though did not show it directly. In other studies copepods were exposed to naphthalene (Harris et al., 1977) (Gyllenberg, 1981). The compound was accumulated, and was apparently metabolized to a degree. Thus metabolism and reproductive effects of PAH can occur in planktonic copepods. In this way contaminants may have an indirect impact on the survival of northern right whales by way of impacting population growth of their prey species. A better knowledge of the chronic dose levels that may lead to chemical suppression of copepods is essential before the significance of this discussion can be determined.

4.6.6 Right Whale Body Condition

Physical oceanographic, biotic, and natural and anthropogenic chemical factors presumably impact copepod patch density. These factors therefore will impact the ability of right whales to graze sufficiently to gain adequate body condition for successful conception, gestation, lactation and recruitment. Thus the limitation of right whale survival may be nutritionally based, with a number of interacting primary factors underlying the net plankton patch density availability.

4.7 Conclusions

Based on the data and information presented in this review, a number of conclusions were reached. These include:

1. Organic contaminant burdens in right whales are lower than in toothed whales.

2. Contaminant burdens are highest in marine mammals found close to heavily populated areas of the globe. Those burdens appear to be lower in the late 1980's and 1990's in comparison to the 1970's.

3. The projected contaminant loading in the future MWRA outfall will represent from 5% to 20% of the total annual load to the Massachusetts Bays system. To that extent the outfall will contribute to the burden acquired by right whales in that habitat.

4. Of whales that visit Cape Cod Bay, assuming equal chemical exposure in all habitats, they will receive 12% of their annual loading whilst in that bay.

5. Levels at which contaminant effects have been associated with contaminant exposure are far higher than have been shown in right whales, or predicted, even if their habitat moves closer to the future outfall site.

6. In spite of the seemingly low level of risk from contaminant effects in right whales it is important to realize that the overall loading of contaminants to the coastal zone should decrease if at all possible. Thus the MWRA should continue its source reduction efforts to the greatest extent possible.

4.8 Recommendations

Two recommendations emerged from this review. These are stated below.

1. There is a major lack of data on contaminant burden and effect in marine mammals from the Gulf of Maine. This, along with linked studies of possible biological effects should be addressed. Studies should be conducted on contaminant burden and effect in toothed and baleen whales in this area.

2. There are substantial numbers of single and mass stranded and fishery bycaught cetaceans that die in the Gulf of Maine each year. These animals should be investigated systematically for possible cause(s) of death, and for evidence of chronic non-lethal contaminant associated pathology. Currently these animals are largely catalogued for natural history data, and some other limited sampling agendas.

TABLE 4-1

Summary of the number of marine mammal sightings during MWRA water column cruises from the mouth of Boston Harbor to the east edge of the near field station grid during 16 days at sea in each of 1995 and 1996.

Species	1995 (Wennemer and Hickey, 1996)	1996 (MWRA, unpublished data)
Harbor Seals	3	7
Harbor Porpoise	0	15*
Fin Whales	4	1
Minke Whales	8	3*
Right Whale	0	1
Unidentified	5	0
Total	20	27

*These sightings may involve repeat sightings of some of the same animals within the same day.

TABLE 4-2

Mercury concentrations in marine mammal liver from around the British Isles 1988-1992 µg/g wet weight (Law 1994)

Species	N	Mean or Range	SD
Grey Seal	40	77.39	112.55
Harbor Seal	30	47.49	43.82
Minke	1	1.80	
Common dolphin	42	16.18	27.21
Grampus	1	1.30	
Pilot whale	2	25 - 23	
White sided dolphin	3	10.63	14.27
Spotted dolphin	7	7.81	7.43
Bottlenosed dolphin	6	7.54	6.25
Harbor porpoise	106	14.54	29.56

TABLE 4-3

Organic chemical concentrations in blubber from marine mammals from around the British Isles µg/g wet wt (Law 1994)

Species		Total 25 PCB's	HCB	a-HCH	g-HCH	p , p ' - DDE	p,p'-DDT	p,p'-TDE	Dieldrin
Grey Seal		55.70				4.80			
	N	1				1			
Common dolphin	Mean	16.06	0.15	0.15	3.91	4.81	1.78	1.17	1.34
	SD	22.14	0.14	0.10	7.04	6.86	2.38	0.86	1.06
	N	69	58	47	58	59	48	48	24
Grampus		10.10	0.12	0.02	0.05	1.20	0.34	0.99	0.94
	N	1	1	1	1	1	1	1	1
Pilot whale	Range	44.80	0.13			11.30			
		28.90	1.02			16.60			
	N	2	2			2			
White beaked dolphin		27.0		0.22	0.11		4.20	4.30	4.00
	N	1		1	1		1	1	1
Spotted dolphin		18.40	0.55			8.20			
	N	1	1			1			
Bottlenose dolphin	Range	119.30	0.22			36.80			
		10.9	0.36			3.30			
	N	2	2			2			
Harbor porpoise	Mean	17.90	0.35	0.06	0.17	3.59	1.20	1.83	3.54
	SD	19.48	0.29	0.04	0.12	3.93	1.64	1.56	3.03
	N	98	97	39	41	98	42	43	41

TABLE 4-4

Organic contaminant burdens in harbor porpoise blubber µg/g wet wt (mean ± SD)

Site (N)	tPCB	ppDDE	Reference
Boston Harbor, MA			(Stein et al. 1992)
Male (1)	33.0	5.6	
Boothbay Harbor, ME			
Female (2)	22.0 - 13.0	3.9 - 4.8	
Bay of Fundy/ME 1989-1991			(Westgate et al. 1997)
Males (55)	17.28 ± 11.18	3.49 ± 1.27	
Females 53)	11.38 ± 4.81	2.67 ± 1.13	
Gulf of St Lawrence 1989-1991			(Westgate et al. 1997)
Male (31)	10.64 ± 5.43	2.99 ± 1.41	
Female (31)	7.15 ± 3.85	1.92 ± 0.96	
Nfld. 1991			(Westgate et al. 1997)
Males (18)	5.24 ± 2.51	1.77 ± 0.86	
Females (11)	5.49 ± 4.37	1.51 ± 1.17	
US West Coast 1971-1986			(Calambokidis and Barlow 1991)
Males (17)	18.2 ± 16.6	37.7 ± 33.1	
Females 26)	11.7 ± 10.1	28.5 ± 28.1	
British Isles 1989-1992			(Kuiken et al. 1994; Law 1994)
Males (48)	21.8 ± 22.0	4.2 ± 3.7	
Females (46)	14.2 ± 16.7	28.5 ± 28.1	
Scandinavian Waters			(Granby and Kinze 1991)
Males (11)	12.2 ± 8.1	7.5 ± 6.7	
Females 16)	13.7 ± 12.1	6.5 ± 5.5	

TABLE 4-5

Inorganic contaminant burdens in harbor porpoise blubber µg/g wet wt (mean ± SD)

Site (N)	Cd	Cu	Hg	Reference
Boston Harbor, MA				
Male (1)	0.068	17.0	.061	(Stein et al. 1992)
Boothbay Harbor, ME				
Female (2)	0.078 - 0.3	5.7 - 8.8	1.2 - 2.5	(Stein et al. 1992)
British Isles				
(106)	0.20 ± 0.060	2.4 ± 8.9	14.5 ± 7.4	(Law et al. 1991)

TABLE 4-6

Organic chemical contaminants in pilot whale blubber µg/g wet wt.

Site, species, sex, (N)	t PCB	ppDDE	Reference
Cape Cod, MA, USA			
Long finned			
Female (1)	1.30	0.45	(NOAA 1995)
Rhode Island, USA			(Taruski et al. 1975)
Old female (1)	114	187	
Maine, USA			(Taruski et al. 1975)
Juvenile male (1)	42	21	
Japanese Pacific Coast 1985 Short finned			(Tanabe et al. 1987)
Male (5)	6.2 ± 1.7	14.4 ± 5.0	
Female (24)	3.1 ± 3.2	7.1 ± 6.9	
France - Atlantic/Med. coasts			(Alzieu and Duguy 1979)
Long finned			
(7)	189 ± 298	69.6 ± 110.6	
Grand Beach Newfoundland			(Muir et al. 1988)
Long finned			
Male (5)	9.03 ± 3.8	5.65 ± 3.71	
Female (9)	3.46 ± 3.34	4.7 ± 5.30	
Faroes Long finned			
(46)	15.17		(Borrell and Aguilar 1993)
(44)	39.90		
(90)	27.40		

TABLE 4-7

Mercury in Pilot Whale Liver - wet µg/g

Site	N	Hg	Reference
Grand Beach Newfoundland			(Muir et al. 1988)
	13	105 ± 98.2	
Pt. Leamington Newfoundland			(Muir et al. 1988)
	26	62.7 ± 87.8	
Faroe Islands			(Caurant et al. 1993)
Different schools	40	84.1 ± 92.0	
	52	56.0 ± 82.7	
	19	61.9 ± 56.8	
	11	52.1 ± 37.7	

TABLE 4-8

Contaminant burdens in northern right whale blubber biopsies from the Bay of Fundy (1988/1989) and bowhead blubber from Alaska µg/g wet wt.

	tDDT	Dieldrin	PCBs
Right Whales (Woodley et al. 1991)			
Adult (6)	0.21	0.03	0.7
Adult females (3)	0.03	0.01	0.4
Immature males (3)	0.07	0.04	0.9
Calf (2)	0.01	0.01	0.3
Unknowns (4)	0.17	0.04	1
Bowheads (Bratton et al. 1990) (N=6) (1986 & 1988) (Overton et al. 1985) (N=8) (1979 & 1980)	All nondetects*	0.033 and* 5 nondetects	3.6 ± 0.8

*Level of detection not cited in reports.

TABLE 4-9

Total PAH burden (ng/g) in caged blue mussels 1987 to 1995 (MWRA, various reports)

Predeployment	Site	Mean	SD
	Gloucester (n=6)	280	149
Postdeployment			
	Future Outfall (n=4)	221	164
	Deer Island (n=6)	1,293	693
	Inner Harbor (Discovery) (n=5)	2,227	907

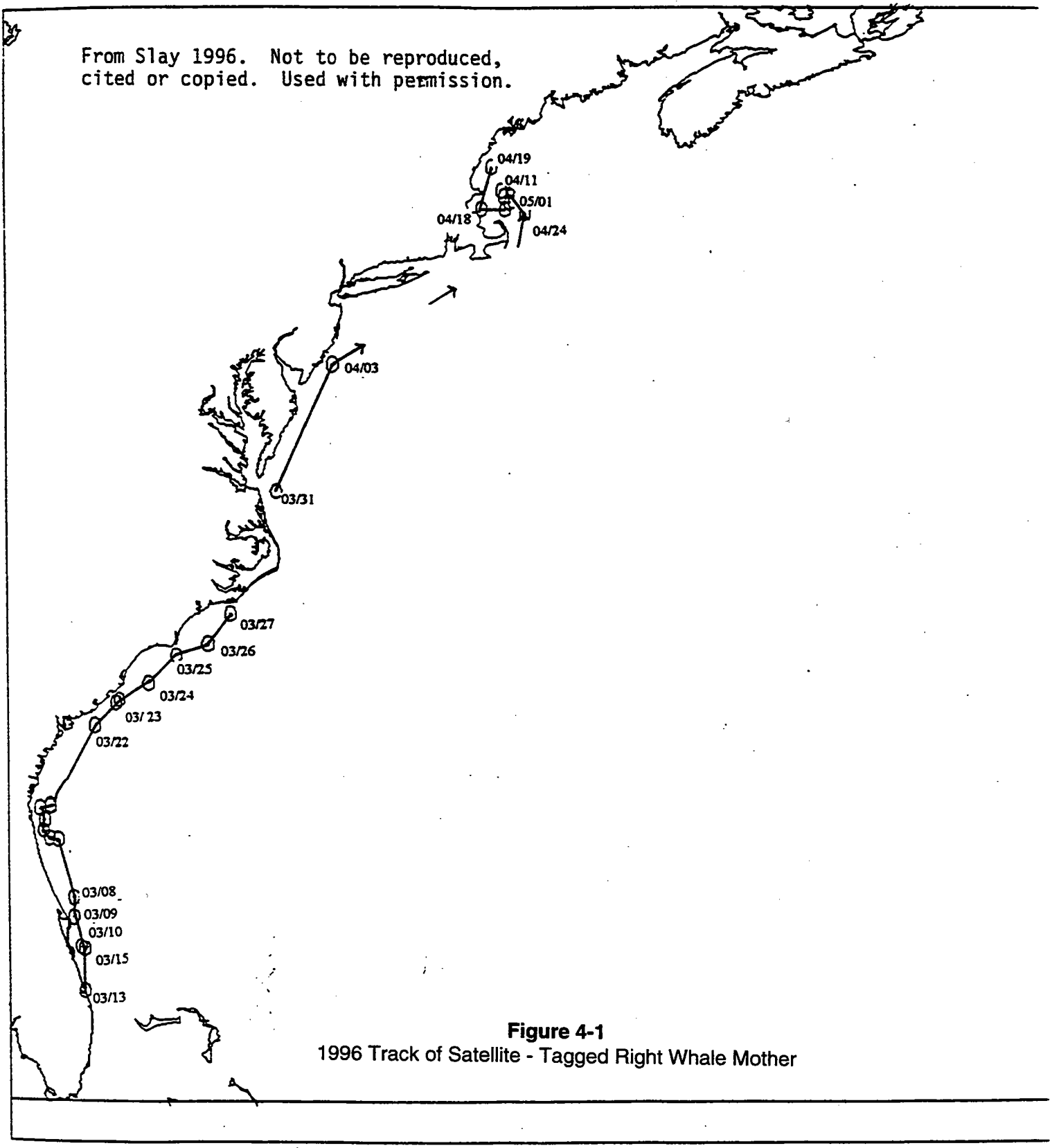


Figure 4-1
1996 Track of Satellite - Tagged Right Whale Mother

Right whale #1812 and calf.

5.0 CONCLUSIONS

For the 1996 MWRA Toxics Issues Report (TIR), an evaluation was made of the levels and trends associated with selected toxics (PCBs, pesticides, mercury) under the current and future scenarios of the Massachusetts Water Resources Authority (MWRA) outfall effluent, as well as the potential for such levels to adversely impact sensitive marine receptors (i.e., threatened/endangered species), particularly the northern right whale. This evaluation utilized currently available data and documents to present an updated characterization of the magnitude of potential sources (e.g., effluent, riverine, atmospheric, combined sewer overflows (CSOs), atmospheric) of these toxics. These results may be used to help evaluate whether the current effluent limitations specified in the NPDES permit for the MWRA future outfall are sufficiently protective of marine receptors.

The 1996 MWRA TIR Report was organized into three major sections: an evaluation of organic contaminants (pesticides, PCBs) in the MWRA effluent (Section 2.0); an evaluation of mercury in the MWRA effluent (Section 3.0); and the potential bioaccumulative effects (Section 4.0). The major findings of each of these sections are discussed below.

5.1 Evaluation of Organic Contaminants

An evaluation of selected organic contaminants present in the current outfall and modeled in the future discharge was made from current or best available data. The toxics considered included PCBs, DDT, dieldrin, heptachlor, and hexachlorobenzene. Review of available historical data indicated uneven quality of data for direct comparison, due to elevated analytical detection limits, sample size, sampling approaches, etc. Despite these qualifiers, there appears to have been decreases in the levels of PCBs and pesticides from the mid-1980s to the early 1990s (see Table 2-7).

Based on available data, estimates were made of the major current sources of organic contaminant loading to Massachusetts Bay including the MWRA outfall, other coastal POTWs, river flow, atmospheric deposition, dredged material disposal, CSOs and other non-point sources, other permitted dischargers, and advective transport from the Gulf of Maine. [It should be noted that the Merrimack River is the dominant contributor among the coastal rivers]. Comparison of these sources (Table 2-13) indicates that, for ΣPCBs (20 congeners), the current MWRA outfall is the most significant discharge, representing about 56% of the total annual load (estimated at 52 kg/yr) to Massachusetts and Cape Cod Bays. Next in importance were river outflow (23%), dredged material disposal (9%), and other coastal POTWs (8%). All other sources constituted 4%. For pesticides entering the Bays (Table 2-14), the current MWRA outfall is again the dominant source loading, accounting for 77% of the total annual load (estimated at approximately 21 kg/yr). River outflow contributed 9%, other coastal POTWs (12%), and all other sources constituted about

2%. These comparisons clearly indicate the current MWRA discharge outfall is by far the largest source of PCBs and pesticides entering Massachusetts and Cape Cod Bays.

A similar analysis was made to project how future organic loadings to the Bays might look. Comparison of MWRA pilot plant primary treatment effluent results with the most current (1995) effluent data supports the use of the pilot plant data as a reasonable predictor of effluent quality for both primary and secondary treatment (see Tables 2-13 and 2-14). An estimate of the future MWRA outfall PCB and pesticide load to Massachusetts Bay was made using effluent quality data from pilot plant testing. Application of predicted primary and secondary treatment effluent data indicates an 82% decrease of PCBs as the effluent shifts from primary to secondary treatment, with other pesticides decreasing from 75 to 94% (Table 2-15).

Using these predicted values (and keeping all other loading sources constant), the future MWRA outfall will be reduced to about 18% of the total PCB load (estimated future total load is about 28.2 kg/yr) to Massachusetts and Cape Cod Bays (Table 2-13). Under this future scenario, river outflow becomes the dominant source (43%) of PCBs, with dredged material disposal (16%) and other POTWs (15%) being important, and all other sources contributing about 8%. The future loading scenario for pesticides does not change as dramatically, as the MWRA outfall will still accounts for the largest (48%) of the estimated future load (about 9 kg/yr), followed by other POTWs (30%), river outflow (21%), and all other sources (1%) (Table 2-14). These analyses are consistent in predicting that the future MWRA outfall will undergo significant decreases in organic contaminants and will provide future loads to the Bays which are comparable to other sources (rivers).

These findings are accompanied by several caveats which need to be considered as potential sources of uncertainty. First, estimates of many of the source terms (e.g., estimates of non-point source and CSO flows) are based on scarce or historic data or may rest on many simplifying assumptions and thus are best approximations. There is little current surface water data for some selected contaminants in remote portions of Massachusetts and Cape Cod Bays, so that deepwater concentrations (Table 2-8) were simply estimates which generally agreed with published Gulf of Maine data. Similarly, atmospheric concentrations of chlorinated pesticides and PCBs from Massachusetts Bay were not available. Therefore available Gulf of Maine data (i.e., Sable Island, WS data) were judged comparable and used. It may be expected that the concentrations of these organics may be more elevated near Boston. Finally, there is insufficient information to indicate whether the Bays act as a net source or sink for atmospheric flux of PCBs and pesticides. In the current model, it was assumed that the Bays act as a minor source to the atmosphere (i.e., there is a flux out of the water and net loss (output) from the Bays). There are insufficient data to strongly support this contention and high quality atmospheric sampling should be conducted to confirm or refute this assumption. Due to the volatile nature of many of the organic contaminants of interest, a consistent cross-media sampling program which samples air, effluent, and ambient sea water should be considered.

5.2 Evaluation of Mercury

The mercury load present in the current and future MWRA outfall discharge was evaluated on the basis of best available information. Evaluation of the historic mercury data indicated potential problems due to highly variable analytical detection levels during the time period inspected, an incomplete use of "clean methods" for sampling and processing, analysis of variable mercury species, etc. Review of current and historic mercury loadings to Boston Harbor from the MWRA outfall provides evidence that levels have decreased from >400 kg/yr in the late 1980s to at or below 100 kg/yr. The last three years (1993-95) indicate values from 50-75 kg/yr (Table 3-5). A value of 104 ng/l appear representative for the current MWRA outfall mercury concentration (see Figure 3-1), while mercury values in central Massachusetts Bay were estimated at about 4 ng/l.

Estimates were made of the major current sources of mercury to Boston Harbor including the MWRA outfall, industrial discharges, riverine transport, atmospheric deposition, CSOs, and other non-point sources (Table 3-8). The amount of mercury in the current MWRA outfall was estimated to be approximately 50 to 75 kg/yr. Assuming the worst case (i.e., MWRA effluent = 75 kg/yr), the relative contribution of these sources are: the MWRA outfall (75%), CSOs and other non-point sources (2%), industrial discharges (2%), river flows (11%) and atmospheric deposition to 8%.

The scheduled implementation of secondary treatment of MWRA sewage is expected to decrease effluent mercury by 52%, based on the pilot plant test results (Table 3-6). This would reduce the mercury content of the discharged effluent to about 32 ng/l. Additional decreases in mercury loading in the future may be possible due to increased efforts to contain any easily identified source - dental amalgams. Application of simple retention traps may provide an additional reduction of the MWRA effluent down to about 27 ng/l. The range of total mercury in the future MWRA outfall is thus estimated at 32 to 27 ng/l, dependent on the level of pollution prevention.

Using the projected secondary effluent quality, estimates were made of future loadings to Massachusetts and Cape Cod Bays once the MWRA offshore outfall is active (Table 3-9). The total future load is estimated at 243-406 kg/yr. The sources of mercury to the Bays include atmospheric deposition (66-79%), river outflows (14-23%), combined industrial discharges, CSOs, and non-point sources (3-5%) and the MWRA outfall (4-6%). It can be seen that the future MWRA outfall will be a minor source of the mercury loading to the Bay.

The predicted mercury levels in the surface water near the future outfall were compared to both acute and chronic ambient water quality criteria. The modeled mercury values in the effluent are far below the acute value, while the chronic AWQC is expected to be met near the discharge and well within the limits of the mixing zone.

5.3 Evaluation of Potential Bioaccumulative Effects

The potential impact of bioaccumulative contaminants present in the current and modeled future MWRA outfall discharge on selected endangered species was reviewed, with special attention to the northern right whale. The information considered included available tissue residue data on marine mammals from the Gulf of Maine and other areas, the feeding ecology of the target species, and a preliminary assessment of the potential toxic effects due to bioaccumulative contaminants. The bioaccumulative contaminants examined included PCBs, pesticides, and mercury, as well as PAHs and metabolites of concern.

Review of the contaminant tissue residue data indicated little specific information for marine mammals in Massachusetts and Cape Cod Bays. Therefore tissue residue data from other areas was considered to provide a framework for the levels of organics and heavy metals found in animals from a variety of locations including the North Atlantic and British Isles (Tables 4-2 through 4-7). Species-specific information on the northern right whale was limited to data on tissue residues of DDT, dieldrin, and PCBs in specimens from the Bay of Fundy (Table 4-8).

Comparison of the available marine mammal database indicates that, in general, body burdens in marine mammals appears to have declined from the 1970's to the late 1980's and 1990's. This decline may reflect both decreased loadings to marine waters and/or more sensitive analytical techniques. As a species, the northern right whales have lower reported body burdens than toothed whales (e.g., pilot whales, dolphins).

While there has been a confirmed sighting of a right whale in the vicinity of the future outfall during 1996 (Table 4-1), there is little evidence to suggest that right whales frequent the area. It was estimated that only 20 to 60 animals (about 7-20% of the right whale population) spend significant time in Cape Cod Bay waters. The average residency period is reported as 45 days or 12% of the year (see Figure 4-1 for typical migratory track). Thus, it may be expected that exposure to Cape Cod Bay waters (and any diluted MWRA contaminants) is about 12% of the cumulative exposure of the whales to contaminants. As noted above, this would only hold true for the maximally exposed individuals, since the majority of right whales (i.e., 80 to 93%) do not frequent Cape Cod Bay.

Previous analyses (see Section 2.0 and 3.0) indicate that implementation of secondary treatment on the MWRA effluent will reduce the total loading to the Bays by approximately 46% for PCBs and 58% for pesticides. Mercury will be reduced by at least 52% in the MWRA effluent outfall. Indeed, the future MWRA effluent mercury load appears to be an insignificant source for Massachusetts and Cape Cod Bay at 4 to 6% of the estimated total mercury load.

No existing evidence exists to suggest that baleen whales in Massachusetts and Cape Cod Bays are subject to levels of PCBs and pesticides capable of producing ecotoxicological effects. Nonetheless, levels may

be of greater concern in toothed whales, such as porpoises (Stein et al., 1992) and white-sided dolphis (Moore et al., unpublished data). The possibility for subtle, chronic effects is not completely ruled out due to our poor understanding of the effects of chronic low-level exposure.

Based on inferences from the available tissue residue data, the temporally-limited exposure period, and the lack of identified adverse ecological effects from the present MWRA effluent discharge, it appears that the future MWRA effluent outfall poses little risk, if any, to the sensitive marine mammals in Massachusetts and Cape Cod Bays.

5.4 Recommendations

Major areas of uncertainty were noted during the course of the review and analysis conducted for the 1996 Toxics Issues Report. Several general recommendations emerged as the means to reduce the uncertainty and provide more quantitative data in the assessment of MWRA effluent toxics. These recommendations are listed below.

1. There is a need for additional selected sampling of the water concentrations of contaminants of concern (mercury, pesticides, PCBs), particularly in assessment of non-MWRA sources and Gulf of Maine advective import and export fluxes, to update and confirm assumed or predicted values from earlier studies.

2. In the future, as the MWRA contribution to the Massachusetts Bay system declines, the influence of atmospheric fluxes will become more important. High quality air sampling should be conducted within the Bays system to address the present lack of knowledge. If air quality sampling is initiated, common atmospherically-transported pesticides should be monitored in air, MWRA effluent and ambient seawater (e.g., Bidleman et al., 1992).

3. Based on the acquisition of the updated information discussed above, a more rigorous modeling of the toxic budgets of the Bays may be warranted.

4. The potential linkage of observed contaminant tissue residue and potential ecological effects in marine mammals from the Gulf of Maine and Massachusetts and Cape Cod Bays should be further studied. In particular, the incidence of stranded or incidentally caught cetacean from the Gulf of Maine may provide an opportunity for investigating potential pathological effects of chronic exposure.

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