

An Integrated Model of Organic Chemical Fate and Bioaccumulation in the Hudson River Estuary

PREPARED FOR
THE HUDSON RIVER FOUNDATION

Kevin J. Farley
Robert V. Thomann
Thomas F. Cooney, III
Darin R. Damiani
James R. Wands

Environmental Engineering Department
Manhattan College
Riverdale, New York 10471

March 1999

Abstract

Accumulation of toxic chemicals (including PCBs, PAHs, dioxins, furans, chlordane, DDT, dieldrin, mercury, and cadmium) in the Hudson River/New York-New Jersey Harbor ecosystem is impacting fisheries, and is limiting port activities associated with dredging and disposal of contaminated sediments (Harbor Estuary Program 1996). In order to assess the effect of various loads on toxic contamination in the ecosystem, we have developed a large space scale, seasonal time scale model for contaminant fate and bioaccumulation of PCBs, PAHs, and dioxin. The model includes: (1) a fully time-variable chemical transport and fate model for calculating PCBs, PAHs, and dioxin in the water column and sediments; (2) a time-variable, age-dependent food chain model for calculating PCB concentrations in phytoplankton, zooplankton, small fish, perch, and striped bass; and (3) Biota-Sediment-Accumulation-Factors (BSAFs) for calculating PCB, PAH, and dioxin concentrations in dredged material test organisms (*Macoma nasuta* and *Nereis virens*).

The model was initially used in evaluating the transport, fate, and subsequent food chain bioaccumulation of PCBs in the Hudson River Estuary. In this evaluation, model results were found to provide a good description of field data for PCB homologue concentrations in the water column, surface sediments, and perch, and for total PCB concentrations in striped bass. Seasonal hydrology, sediment transport, and striped bass migration patterns were found to play key roles in determining the fate and bioaccumulation of PCBs. The general behavior for PCB fate and bioaccumulation in the estuary is as follows:

During Spring high flows, disproportionate amounts of PCBs are transported into the estuary from the Upper Hudson and deposited in sediments or transported down to the harbor without significant volatile loss. For low to moderate flows that usually occur during the remainder of the year, sediments serve as a source of PCBs to the overlying water via desorption from contaminated sediments during tidal resuspension. Because of the longer-term retention of PCBs in sediments, surface sediment PCB concentrations and dissolved PCB concentrations in the overlying water appear to be decreasing slowly in time in the mid estuary with little or no seasonal variation. PCB accumulations in fish, which are ultimately driven by dissolved PCB concentrations and accumulation in lower trophic species, are also decreasing slowly in time. The large variation in observed PCB concentrations in striped bass, and in particular, the unusually high concentrations of PCBs that are observed in some striped bass are in part related to the presence of a non-migrating subpopulation in the estuary. Based on the model calculations, the upper Hudson load appears to contribute 60-80% of the PCB sediment contamination and 50% of the perch and striped bass contamination in the tidal freshwater and mid estuary. Average PCB concentrations in 2-5 and 6-15 year old striped bass are presently at or below the Food and Drug Administration (FDA) limit of 2 $\mu\text{g/g}$ (wet wt) but the 95%ile concentrations are expected to remain above the safe limit for the next few years.

Subsequent model applications were performed to evaluate the effects of contaminant loading on PCB, PAH, and dioxin accumulation in New York-New Jersey Harbor sediments and in the dredged material test organisms (*Macoma abita* and *Nereis virens*). A preliminary assessment of harbor sediments was made based on available field data and showed exceedances of present tissue concentration guidelines for dredged material test organisms (*Macoma nasuta* and *Nereis virens*) would occur in 17% of the harbor for PCBs and in 3% of the harbor for PAHs. Similar results were determined from modeling studies. In these evaluations, a preferential accumulation of PCBs and PAHs in Newark Bay sediments was determined and was in part attributed to the effective trapping of contaminants in the bay by high concentrations of sewage-derived and phytoplankton organic carbon. For PAHs, further analysis of sediment contamination should be performed with site-specific loading data for New York-New Jersey Harbor, and the PAH narcosis-based tissue concentration guideline for dredged material disposal should be reviewed.

For the most toxic dioxin congener, 2,3,7,8 tetrachlorinated dibenzo-*p*-dioxin (TCDD), sediment samples from many of the harbor regions would likely cause an exceedance of Category 1 guidelines, and sediment samples from Newark Bay and from New York Harbor (near the Marine Ocean Terminal) would likely fail to meet requirements for Category 2 dredged material. From modeling studies, TCDD contamination in the harbor appears to be only minimally related to present discharges from combined sewer overflows (CSOs) and storm water outflows (SWOs), and are more likely the result of past discharges of TCDD into the Lower Passaic River (and the associated high levels of TCDD sediment contamination in the Lower Passaic and Newark Bay). Further evaluations of past TCDD loading should be performed with specific attention given to comparison of model results and sediment data for the New York Harbor segment.

The research contained in this report has been financed in part through a research grant from the Hudson River Foundation, with contributing funds from the New York-New Jersey Port Authority, and US Environmental Protection Agency, Region 2. The views expressed in this report do not necessarily reflect the beliefs or opinions of the sponsoring agencies, and they assume no responsibility for the contents or use of the information herein.

Table of Contents

Chapter 1: Introduction	1
PCB Contamination in Hudson River Striped Bass: Background	1
PCB, PAH, Dioxin Contamination in NY-NJ Harbor Sediments: Background ..	8
General Description of Modeling Approach	16
Chapter 2: Fate and Bioaccumulation of PCB Homologues: Model Development ...	19
Hydrodynamic Transport	21
Sediment Transport	24
Organic Carbon Distributions	28
Chemical Transport and Fate	30
PCB Loads	38
PCB Initial Conditions in Sediments	41
Bioaccumulation Modeling	45
PCB Initial Conditions in Fish	52
Chapter 3: Fate and Bioaccumulation of PCB Homologues: Model Results	53
Previous Modeling Results	53
Post-Audit Evaluation	53
Recalibrated PCB Homologue Model	53
Recalibration Results for the Water Column and Surface Sediments	55
Time History/Projections for PCB Water Column and Sediment Conc.	64
PCB Fluxes	67
Organic Carbon Effects on PCB Fate	72
Recalibration Results and Time History/Projections for PCBs in Perch	73
Effect of Migration on PCB Accumulation in Striped Bass	78
Effect of Non-Migration on PCB Accumulation in Striped Bass	86
Component Load Analysis	92
Summary and Conclusions	99
Chapter 4: PCB, PAH, and Dioxin Contamination of Dredged Sediments from New York- New Jersey Harbor	101
Introduction	101
Biota-Sediment-Accumulation Factors (BSAFs)	102
Preliminary Assessment of NY-NJ Harbor Dredged Material	109
PCB Model Results	119
PAH Model Results	119
Dioxin Model Results	132
Summary and Conclusions	137
References	139
Appendix A: Model Parameters and Other Supporting Material	149
Appendix B: Notation	169

Acknowledgments

Support for this research was provided by the Hudson River Foundation, with contributing funds from the New York-New Jersey Port Authority and the Environmental Protection Agency Region 2. The technical support and insights of Dennis Suszkowski and John Waldman of the Hudson River Foundation is gratefully acknowledged. Special thanks to the many individuals who provided us with data and with their technical insights including: Darvene Adams, Seth Ausubel, Joel Baker, Henry Bokuniewicz, Richard Bopp, Bruce Brownawell, John Connolly, Steve Chillrud, Dominic DiToro, Steve Eisenreich, Jay Field, Ed Garvey, Rocky Geyer, John Haggard, Carlton Hunt, Simon Litten, Jim Lodge, Robin Miller, Tom Parkerton, Pat Phillips, Jim Rhea, Lisa Rossman, Dave Secor, Larry Skinner, Ron Sloan, John St. John, and Doug Tomchuck.

Chapter 1

Introduction

Accumulation of toxic chemicals (including PCBs, PAHs, dioxins, furans, chlordane, DDT, dieldrin, mercury, and cadmium) in the Hudson River/New York-New Jersey Harbor ecosystem is impacting fisheries, and is limiting port activities associated with dredging and the disposal of contaminated sediments (Harbor Estuary Program 1996). In order to assess the effect of various loads on toxic contamination in the ecosystem, we have developed a large space scale, seasonal time scale model for contaminant fate and bioaccumulation of PCBs, PAHs, and dioxin. The model includes: (1) a fully time-variable chemical transport and fate model for calculating PCBs, PAHs, and dioxin in the water column and sediments; (2) a time-variable, age-dependent food chain model for calculating PCB concentrations in phytoplankton, zooplankton, small fish, perch, and striped bass; and (3) Biota-Sediment-Accumulation-Factors (BSAFs) for calculating PCB, PAH, and dioxin concentrations in dredged material test organisms (*Macoma nasuta* and *Nereis virens*).

In the remainder of this chapter, background information is presented on PCB contamination of Hudson River striped bass, and on PCB, PAH, and dioxin contamination of dredged sediments from New York-New Jersey Harbor, and is followed by a general description of the overall modeling approach. Application of the model has primarily focused on PCBs because of large gaps in the available data for PAH and dioxin loadings, and to a lesser extent, gaps in observed contaminant concentrations in harbor water, sediments, and biota. For this reason, model development will be described in terms of PCB calculations in Chapter 2. Model results and projections for the fate and bioaccumulation of PCBs in Hudson River striped bass will be discussed in Chapter 3, and will be followed in Chapter 4 by application of the fate and bioaccumulation model for PCBs, PAHs, and dioxin in dredged sediments from New York-New Jersey Harbor.

PCB Contamination in Hudson River Striped Bass: Background

From approximately 1947 to 1977, General Electric (GE) facilities along the Upper Hudson at Hudson Falls (River Mile 197) and nearby Fort Edwards (River Mile 195) used PCBs in the manufacturing of electrical capacitors. According to a U.S. Environmental Protection Agency (EPA)(Limno-Tech et al. 1996), 0.2 to 1.3 million pounds of PCBs were discharged from the GE facilities into the Upper Hudson River between 1957 and 1975. Downstream migration of PCBs over the 40-mile stretch between Hudson Falls and Federal Dam was greatly enhanced in 1973 with the removal of the Fort Edwards Dam (River Mile 195) and subsequent high water discharges in April, 1974 and April, 1976 (Chillrud 1996). In 1976, the New York State Department of Environmental Conservation (NYS DEC) imposed a ban on fishing in the Upper Hudson River and on commercial fishing for striped bass in the estuary due to the potential risk posed by consumption of PCB-contaminated fish.

Although PCB use at the GE facilities was curtailed in 1977, data on PCB loads show that approximately five to ten pounds per day of PCBs were being discharged from the Upper Hudson in the early 1990s. These upstream loads, which have subsequently been reduced to approximately one pound per day, have been primarily attributed to scouring of PCB contaminated sediments from an old water intake structure, PCB oil seeps through the fractured bedrock underlying the Hudson Falls facility (Rhea et al. 1997) and/or leaching of PCBs from contaminated sediments (e.g., in Thompson Island Pool) (Garvey and Hunt 1997). In addition to the Upper Hudson source, PCB sources from wastewater treatment plant discharges, rainfall runoff, and direct atmospheric sources may also be important. For example, PCB loadings from New York City wastewater treatment plant effluents and combined sewer overflows were estimated to be on the order of 0.55 pounds per day from 1994-95 monitoring data (Chen 1995).

Concentrations of PCBs in surface sediments (0-2 cm) in the Hudson River Estuary are given in Figure 1-1. In this plot, total PCB concentrations are subdivided into homologues (i.e., groups with similar number of chlorines substituted on the biphenyl structure) and are presented for mono- through octa-chlorobiphenyl (CB). The data show a gradient in PCB concentrations from the upper portions of the estuary down into New York-New Jersey Harbor, particularly for the lower chlorinated homologues. This decrease in PCB sediment concentrations suggests dilution of PCB-contaminated sediments by cleaner, downstream sediment sources and/or PCB loss from the estuary by volatilization and/or degradation. An increase in total PCB sediment concentrations is also evident around River Mile 0, and as discussed later in this report, may reflect PCB and/or organic carbon source loadings from the New York City metropolitan area. Dissolved water column concentrations in the upper and mid-estuary region are in the range of 10-20 ng/L (TAMS/Gradient 1995; Brownawell 1997).

PCB homologue concentrations in white and yellow perch from an August 1993 survey of the estuary are presented in Figure 1-2. Although the gradient in PCB perch concentrations does not seem to be as dramatic as the PCB sediment gradient (Figure 1-1), PCB perch concentrations varied from a high of 11 $\mu\text{g/g}$ wet weight in the tidal freshwater region to a low of 3 $\mu\text{g/g}$ wet weight further downstream in the mid estuary. In comparison to the surface sediment data (Figure 1-1), the perch data show a clear shift to higher chlorinated compounds.

A time history of total PCB concentrations in striped bass is shown for the tidal freshwater (Figure 1-3), the mid estuary (Figure 1-4), and the harbor region (Figure 1-5). For the tidal freshwater and mid estuary, median concentrations (as shown by the centerline of the box-and-whisker) show declines in the late 1980s, leveling off or possible increases in PCB concentrations in the early 1990s, and subsequent declines in the mid 1990s. Throughout the period, highest median PCB concentrations are observed in the tidal freshwater region, with lower concentrations in the mid estuary and harbor regions.

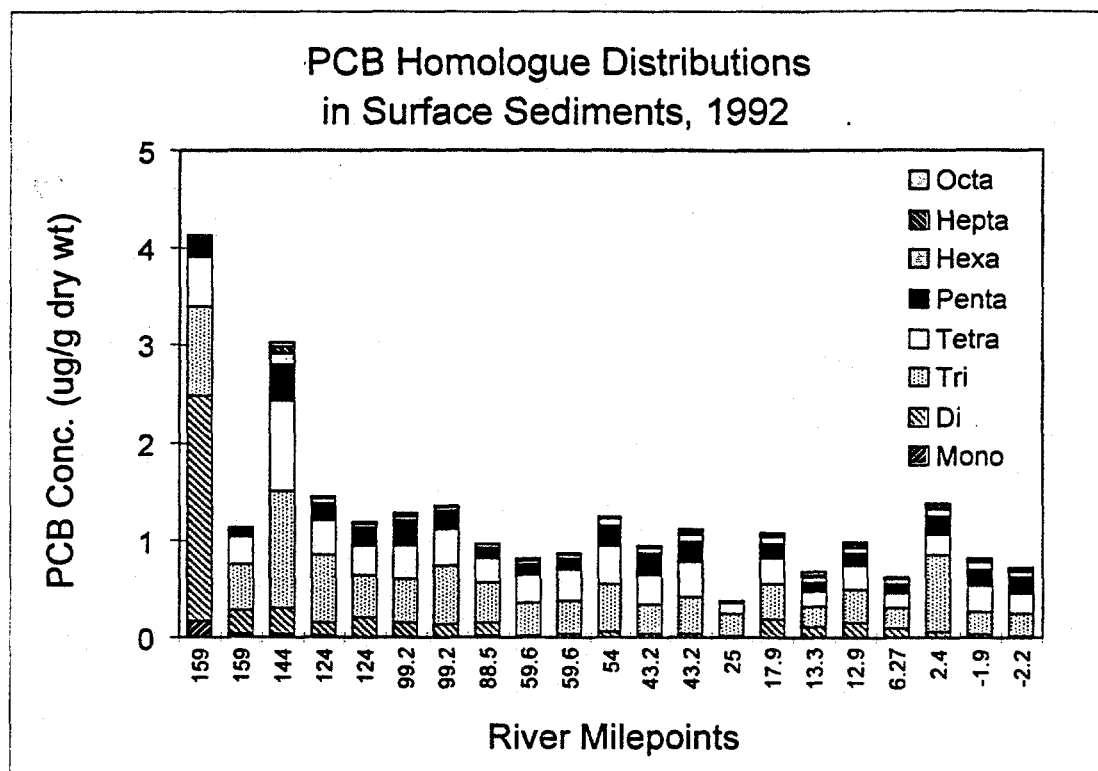


Figure 1-1. PCB concentrations and homologue distributions for surface sediments in the Hudson River Estuary (8/92 - 11/92). [Data from TAMS/Gradient (1995).]

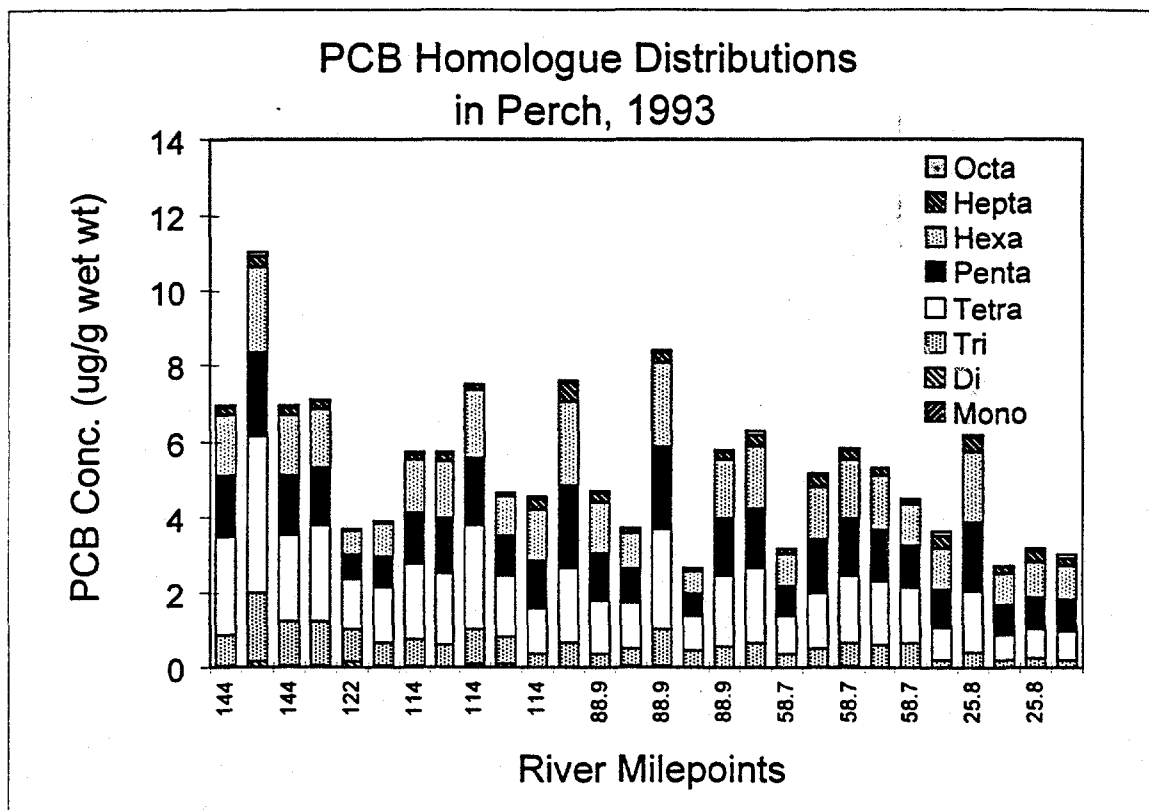


Figure 1-2. PCB concentrations and homologue distributions for white and yellow perch in the Hudson River Estuary (8/93). [Data from TAMS/Gradient (1995).]

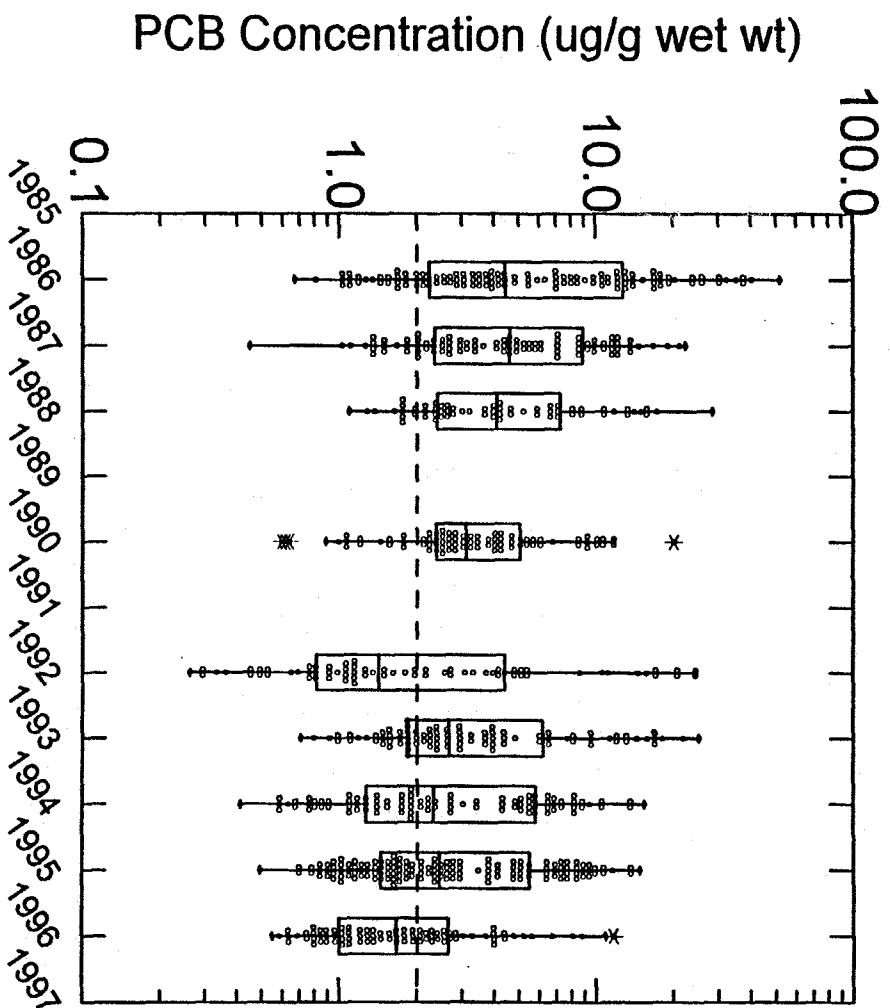


Figure 1-3. 1986-1997 time history of total PCBs in striped bass from the tidal freshwater Hudson. [Data from NYS DEC as reported in TAMS/Gradient (1995).]

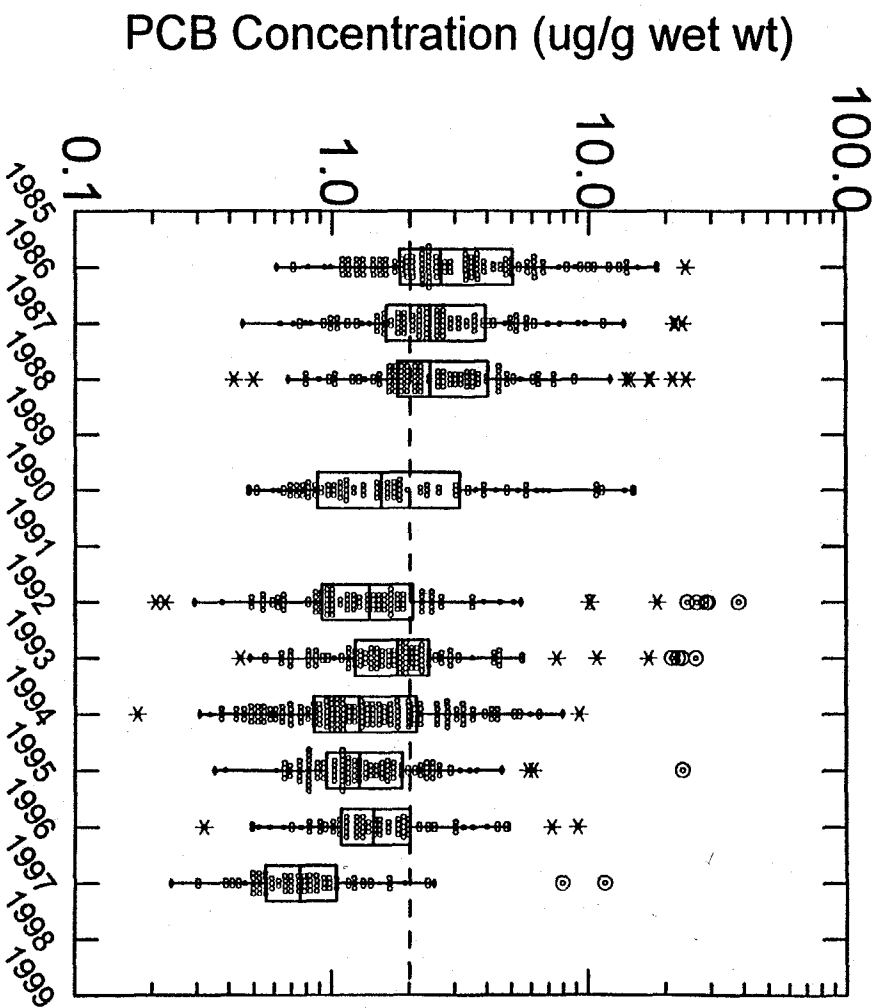


Figure 1-4. 1986-1997 time history of total PCBs in striped bass from the Hudson mid estuary. [Data from NYS DEC as reported in TAMS/Gradient (1995).]

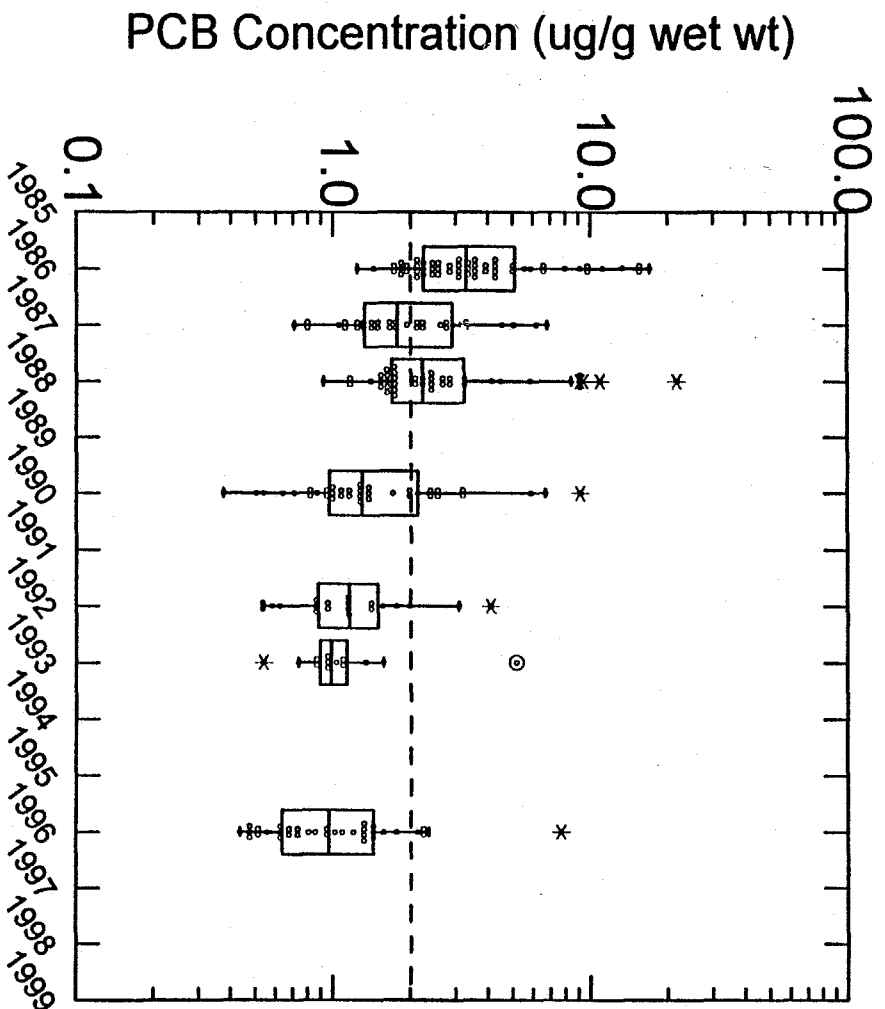


Figure 1-5. 1986-1997 time history of total PCBs in striped bass from New York-New Jersey Harbor. [Data from NYS DEC as reported in TAMS/Gradient (1995).]

For any given year, large variations in striped bass PCB concentrations are observed, with some fish still exceeding the Food and Drug Administration (FDA) limit of 2 µg/g wet weight. As discussed in Chapter 3, the large variations in striped bass PCB concentrations are attributed to interspecies variability, and are partially explained by striped bass migration behavior and the presence of a non-migrating subpopulation of striped bass.

PCB, PAH, and Dioxin Contamination in NY-NJ Harbor Sediments: Background

Several million cubic yards of sediment are dredged each year from shipping channels and berthing areas in New York-New Jersey Harbor (Suszkowski 1995). In the past, a large portion of the dredged material (approximately 80% by volume) was classified as Category 1 (i.e., non-bioaccumulative and non-toxic) and dumped at an offshore disposal site in the Atlantic Ocean. Testing requirements for the ocean disposal of dredged material from New York-New Jersey Harbor however were revised in the early 1990s (USACE New York District/EPA Region II 1992) and now include protocols for amphipod toxicity testing and dioxin analyses. These changes have resulted in reclassification of a larger portion of dredged material as Category 2 (i.e., bioaccumulative but not toxic and limited to ocean dumping with capping) or Category 3 (i.e., toxic and/or bioaccumulative and not acceptable for ocean disposal) (Suszkowski 1995).

PCBs, PAHs, and dioxins (along with mercury) have been identified as probable causative agents in contamination of dredged material (Suszkowski 1995). Sources of PCBs into the harbor include tributary loadings (particularly from the Upper Hudson), wastewater treatment plant discharges, combined sewer overflows (CSOs), storm water outflows, rainfall runoff, and direct atmospheric loading. Total PCB concentrations in harbor sediments (as given by the 1993-94 New York-New Jersey R-EMAP surface (0-2 cm) sediment data (Adams et al. 1998)) are shown in Figure 1-6. In this plot, total PCBs were taken as two times the summation of 20 measured PCB congeners (Adams et al. 1998), and median concentrations are shown to vary by approximately two orders of magnitude for various regions around the harbor. For example, median concentrations (as shown by the centerline of the box-and-whisker) are given as 0.003 µg/g(dry wt) in the New York Bight to approximately 0.5 µg/g(dry wt) in the Upper Harbor and Newark Bay.

PAHs (polycyclic aromatic hydrocarbons) comprise a large class of compounds containing two or more benzene rings with at least two common carbons. Principle sources of PAHs to the harbor include: incomplete combustion of fossil fuels; processing byproducts from oil and coal (e.g., wood distillation, manufacturing of roof and road tar, refining and application of creosote); spills of petroleum products; and runoff from urban highways. Based on log octanol-water partitioning coefficients, many PAHs are expected to bind strongly to solids and be deposited in sediments. Total PAH concentrations in harbor sediments (as given by the summation of 23 PAHs in the 1993-94 R-EMAP surface sediment data (Adams et al. 1998)) are given in Figure 1-7. As shown, total PAH

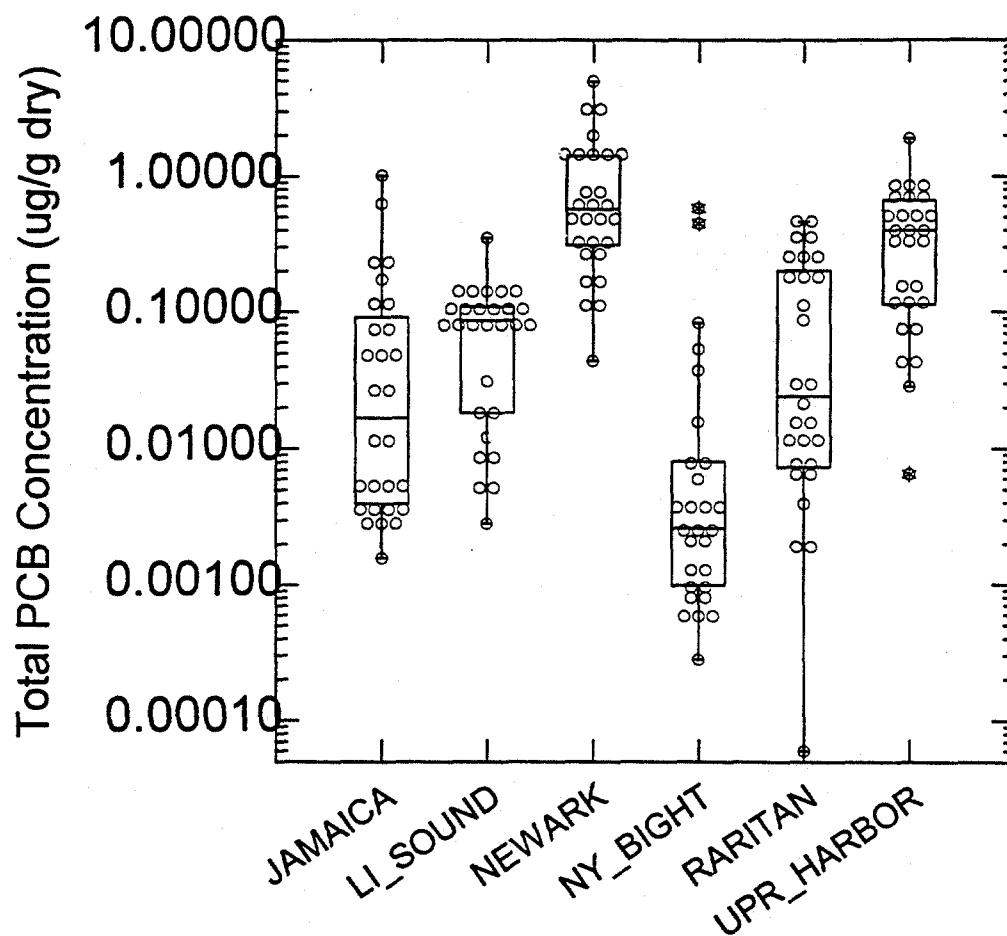


Figure 1-6. 1993-94 total PCB concentrations in surface sediments of New York-New Jersey Harbor. [Data from 1993-94 R-EMAP study (Adams et al. 1998).]

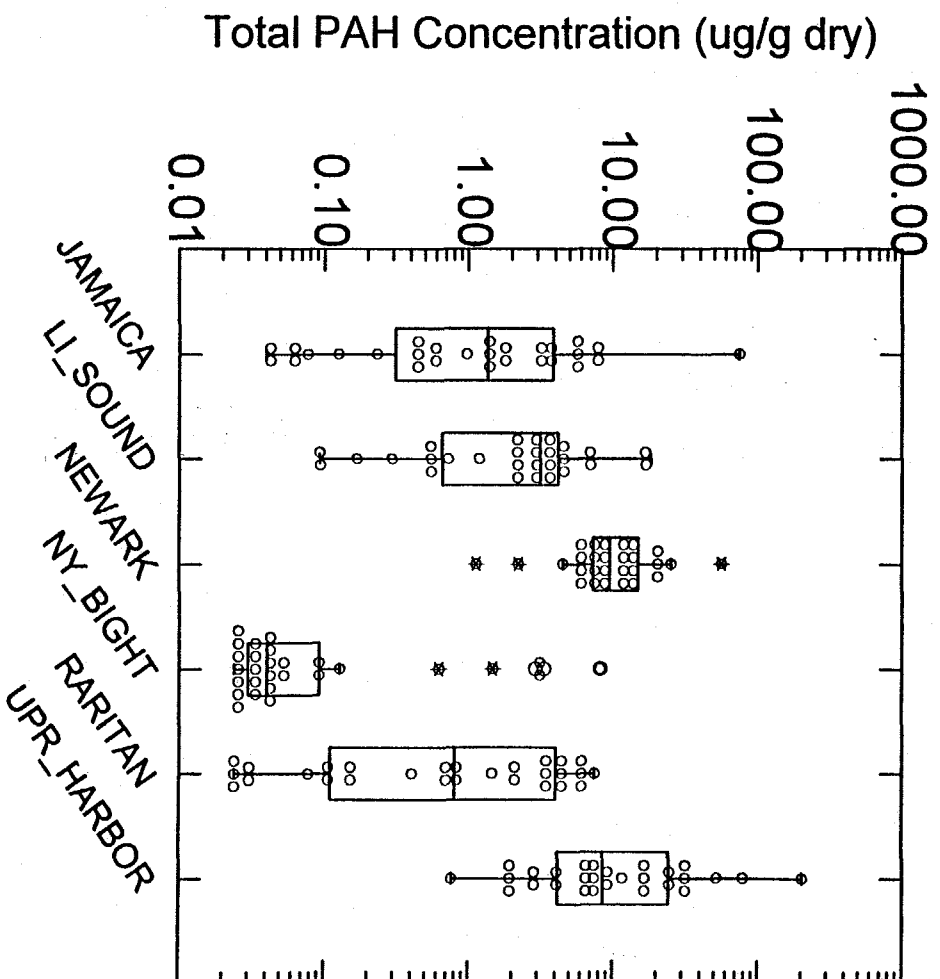
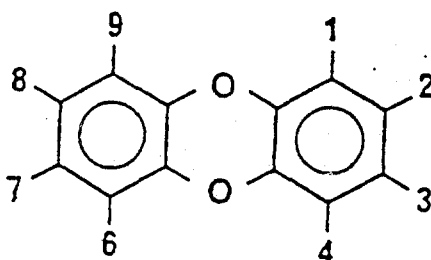


Figure 1-7. 1993-94 total PAH concentrations in surface sediments of New York-New Jersey Harbor. [Data from the 1993-94 R-EMAP study (Adams et al. 1998).]

concentrations for various regions around the harbor show large spatial variations with medians of 0.04 $\mu\text{g/g}$ (dry wt) in the New York Bight and 10 $\mu\text{g/g}$ (dry wt) in the Upper Harbor and Newark Bay.

In addition to sorption, chemical transformation of PAHs may occur through microbial degradation, and possibly photolysis. PAH accumulation in aquatic organisms is also a concern and may result in a higher degree of toxicity due to narcosis (Di Toro et al. 1999). Biomagnification of PAHs in aquatic food chains has not been observed. This finding is likely the result of biotransformation of PAHs in higher trophic organisms (Broman et al. 1990). Transformation of PAHs in aquatic organisms however may result in the production and accumulation of toxic intermediates and byproducts. In addition, byproducts such as octahydrochrysenes, which is believed to be a byproduct of sewage sludge degradation, may confound the identification of petroleum-contaminated areas by measurements of PAH levels in organisms (Farrington et al. 1986).

Polychlorinated dibenzo-*para*-dioxins (dioxins) represent a family of 75 congeners having various chlorine substitution patterns on an almost planar tricyclic aromatic structure as shown below:



The relative toxicity of dioxins is expressed in terms of Toxicity Equivalency Factors (TEFs) (Table 1-1). Based on TEFs, dioxin compounds of greatest concern are 2,3,7,8-tertachlorinated dibenzo-*p*-dioxin (2,3,7,8-TCDD) and higher chlorinated congeners that have chlorines at the 2,3,7,8 positions.

Principal sources of dioxins to the harbor include: combustion sources such as municipal and hazardous waste incinerators, and fossil-fuel power plants; chemical manufacturing where dioxins are a byproduct in the manufacture of chlorinated phenols, PCBs, phenoxy herbicides such as 2,4,5-T; and chlorination processes where the chlorination of naturally-occurring phenolic compounds may result in the formation of dioxins (e.g. during the manufacture of bleached pulp and paper products). The distribution of dioxin congeners in the different types of discharges and emissions are highly source specific (Rappe and Buser 1989). For example, incinerator emissions contained a wide variety of PCDD congeners while the 2,3,7,8-TCDD was found to be the major dioxin byproduct from 2,4,5-T production.

Table 1-1. Toxicity Equivalency Factors (TEFs) for Polychlorinated dibenzo-para-dioxins (dioxins).	
Compound	TEF
Mono-, Di, and Tri-CDDs	0
2,3,7,8-TCDD	1
Other TCDDs	0
2,3,7,8-PeCDD	0.5
Other PeCDDs	0
2,3,7,8-HxCDDs	0.1
Other HxCDDs	0
2,3,7,8-HpCDD	0.01
Other HpCDDs	0
OCDD	0.001

Because dioxins are very hydrophobic ($\log K_{ow}$'s > 6), they are expected to bind strongly to solids and be deposited in sediments. A compilation of field data for dioxin contamination for surface sediments around the harbor is shown in Figure 1-8 for 2,3,7,8 tetra-chloro-dibenzo-para-dioxin (TCDD) and octa-chloro-dibenzo-para-dioxin (OCDD). High levels of the more toxic TCDD are found in lower Passaic River and are attributed to past discharges of 2,3,7,8-TCDD from an industrial site in Newark where chlorinated phenols were produced (Bopp et al. 1991). A gradient in TCDD concentrations is clearly noted with median concentrations decreasing from a high of 350 pg/g(dry wt) in the lower Passaic River to 6 pg/g(dry wt) in the New York Bight. Conversely, OCDD concentrations are approximately equal to 2,000 pg/g(dry wt) throughout the study area suggesting that OCDD is from regional (combustion) sources (Bopp et al. 1991).

Bioaccumulation of dioxins is believed to be primarily through the food chain (Servos et al. 1992), with large differences in congener distributions reported for various trophic levels (Rappe and Buser 1989)(Servos et al. 1992). For example, only 2,3,7,8-substituted congeners are found in fish while 2,3,7,8-substituted congeners and a variety of other congeners are found in crabs. This difference may be attributed to the ability of fish to metabolize dioxins that do not have chlorines substituted in the 2,3,7,8 positions

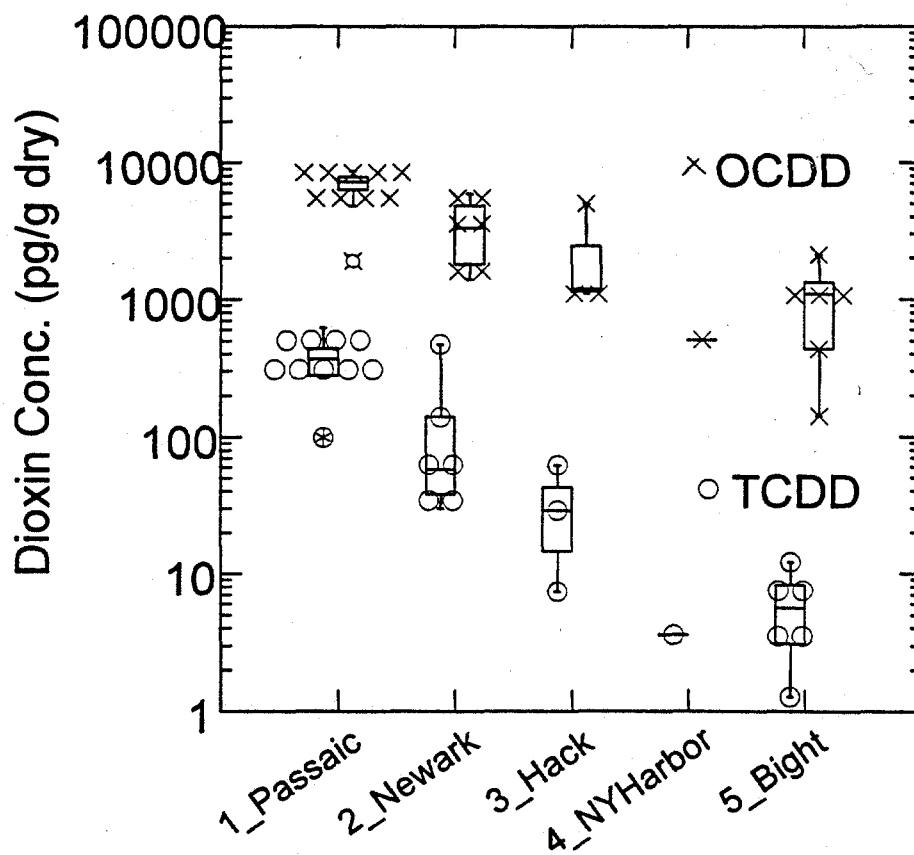


Figure 1-8. 2,3,7,8-TCDD and OCDD concentrations in sediments in New York-New Jersey Harbor. [Data from Long et al. (1995).]

(Servos et al. 1992). Dioxin congeners have generally been found to exhibit low assimilation efficiencies (i.e. less than 15%) (Servos et al. 1992). 2,3,7,8-TCDD however appears to be an exception with a reported assimilation efficiency of 50%.

Potential impacts associated with dredging contaminated sediments from the harbor and disposing dredged material in the ocean are determined from bioaccumulation and toxicity testing. For bioaccumulation, 28-day sediment exposure tests are conducted using *Macoma nasuta* (a clam) and *Nereis virens* (a worm) as test organisms. A summary of current tissue guidelines used in assessing PCB, PAH, and dioxin bioaccumulation in macoma and nereis is given in Table 1-2. Note that dioxin guidelines given in Table 1-2 are specified for both potential food chain bioaccumulation [1 pg/g(wet wt)] and toxicity [10 pg/g(wet wt)].

Table 1-2. PCB, PAH, and Dioxin Bioaccumulation Tissue Concentration Guidelines for Dredged Material from New York-New Jersey Harbor.	
	Tissue Concentrations
PCBs	0.1 µg/g(wet wt) clam ⁽¹⁾ 0.4 µg/g(wet wt) worm ⁽¹⁾
PAHs	40 µg/g(wet wt) ⁽²⁾
Dioxins: (2,3,7,8-TCDD)	1 pg/g(wet wt) ⁽¹⁾ 10 pg/g(wet wt) ⁽³⁾
⁽¹⁾ Regional matrix values for Category 1 dredged material (Pabst and Tavolaro 1998). ⁽²⁾ Ecological non specific narcosis effects levels (Pabst and Tavolaro 1998). ⁽³⁾ Regional matrix values for Category 2 dredged material (Rossman 1998).	

A direct measure of toxicity is also determined from 10-day sediment exposure tests using *Ampelisca abita* as the test organism. Results for ampelisca toxicity testing from the 1993-94 R-EMAP surface sediment monitoring program are shown in Figure 1-9. For toxicity results, ampelisca survival rates of greater than 80% are typically considered acceptable and indicative of non-toxic sediments. Results of the R-EMAP study show that 24 out of 168 (or 14% of) surface sediment samples from around the harbor region fall below this criterion and are considered toxic. At the present time, there is no acceptable

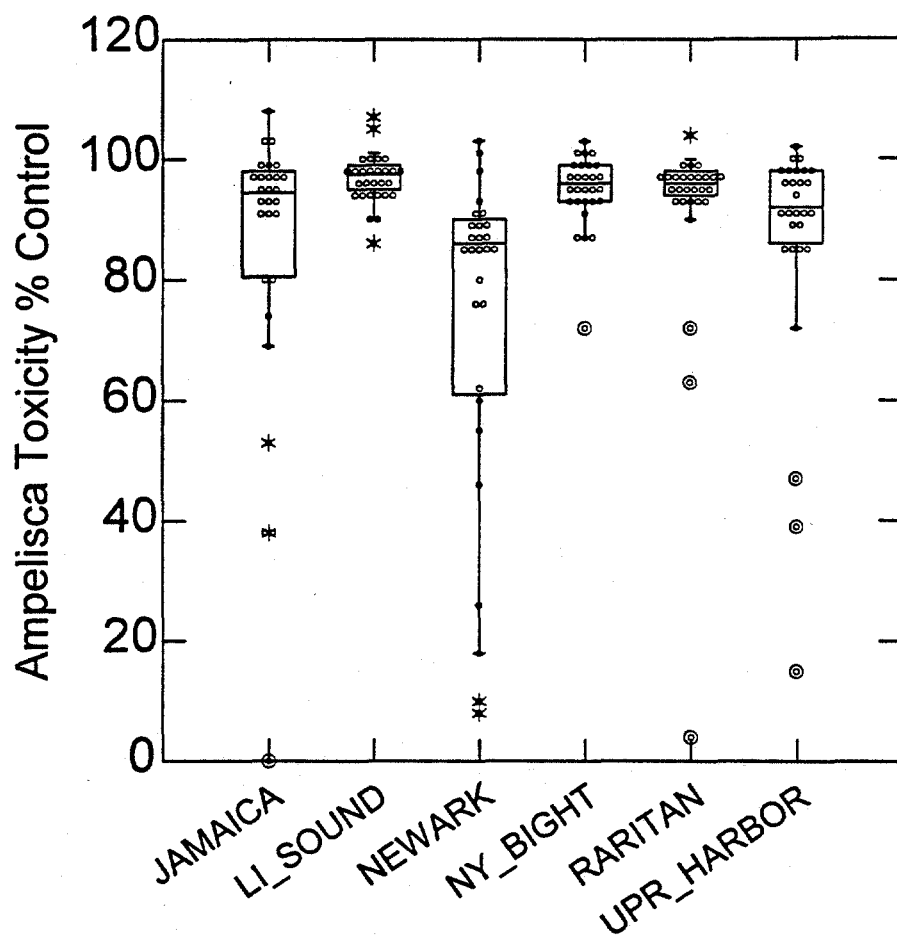


Figure 1-9. 1993-94 *Ampelisca* abita toxicity testing results for surface sediments in New York-New Jersey Harbor. [Data from 1993-94 R-EMAP study (Adams et al. 1998).]

approach to relate sediment concentrations (as given in Figures 1-6 through 1-8) to bioaccumulation or toxicity in test organisms.

Finally, results presented in Figures 1-6 through 1-9 provide a good indication of contamination levels and toxicity for surface sediments around the harbor. For dredged material testing, sediments are collected over the full project depth (typically one to several meters) and are likely to show higher contaminant levels due to historical sources of PCBs, PAHs, and dioxin.

General Description of Modeling Approach

The overall objective of our study was to develop a mathematical model to assess the impacts of external chemical loadings into the Hudson River/New York-New Jersey Harbor ecosystem on: (1) PCB concentrations in striped bass, and (2) PCB, PAH, and dioxin accumulation and toxicity in dredged material test organisms. The overall mass balance modeling approach used in this analysis can be considered in terms of the five components (or submodels) as outlined in Figure 1-10. As shown in the figure, hydrodynamic model calculations serve as the basis of the overall calculation and are used to define water transport through the system. Since many of the chemicals of concern in this study have a strong affinity for particulate (POC) and dissolved organic carbon (DOC), organic carbon model calculations are needed to compute the partitioning of contaminants among freely dissolved, particulate, and DOC phases. Sediment transport model calculations are also necessary to describe the settling, resuspension, burial, and water column transport of particle-bound contaminants in the system.

In the chemical fate model calculation, physical-chemical processes (e.g., sorption, volatilization, hydrolysis, photodegradation) and biochemical processes (e.g., aerobic degradation, anaerobic dechlorination) are considered with hydrodynamic, organic carbon, and sediment transport to determine the effects of point and nonpoint sources on contaminant concentrations in the sediments and in the overlying water. Results of contaminant concentrations in the sediments and overlying water are then used in bioaccumulation model calculations to determine contaminant concentrations in sediment dwelling organisms and fish.

In this modeling approach, calculated water and sediment contaminant concentrations can be compared directly to applicable water and sediment quality standards. Contaminant body burdens in bottom dwelling organisms and fish can be compared to acceptable bioaccumulation levels for dredged material test organisms and FDA values for seafood, respectively. Results from the fate and bioaccumulation model calculations can also serve as input in calculations for human and ecological risk.

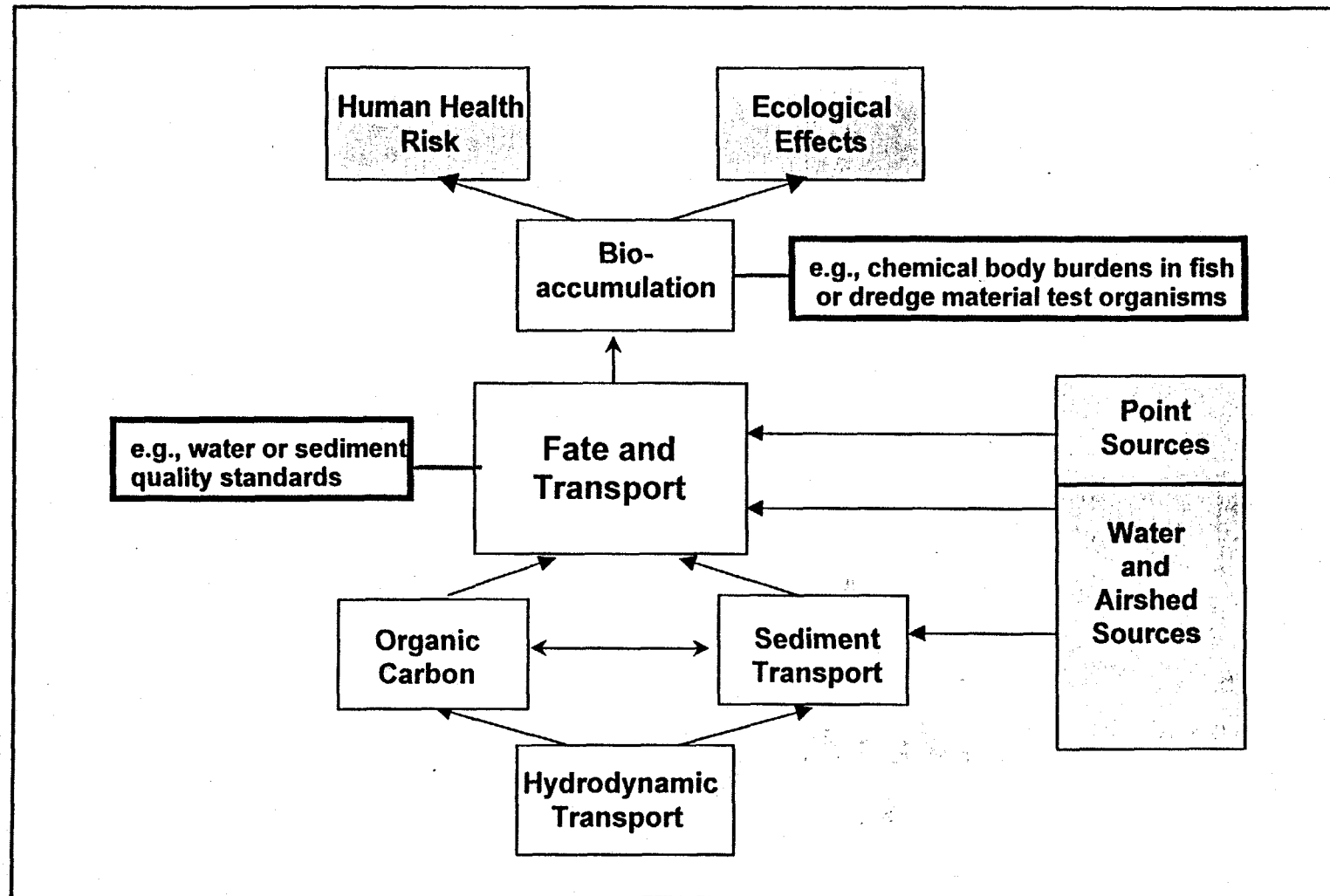


Figure 1-10. Components (or submodels) of the overall modeling approach.

Within this overall mass balance modeling approach, the primary focus of our proposed studies has been in the development and application of models for chemical fate and bioaccumulation. Hydrodynamic, organic carbon, and sediment transport were therefore not modeled directly but rather were specified based on field observations, other modeling work, or simple mass balance calculations as discussed in the subsequent chapter on model development for PCB homologues. Chemical fate and bioaccumulation calculations were performed using the WASTOX model (Connolly 1991; Connolly and Thomann 1985).

Chapter 2

Fate and Bioaccumulation of PCB Homologues: Model Development

The purpose of our studies was to construct a model for evaluating the transport, fate, and subsequent food-chain bioaccumulation of PCBs in the Hudson River Estuary. Towards this end, model calculations were developed using WASTOX (Connolly 1991b)(Connolly and Thomann 1985) and PCB concentrations in striped bass were taken as final endpoints. Particular emphasis in model development was given to management questions related to: (1) the relative impacts of PCBs sources from the Upper Hudson and from wastewater discharges from the New York City area, and (2) a time projection for the reopening of the Hudson River striped bass fishery.

Long-term time scales (seasons and decades) were chosen for model application based on the decades-long extent of PCB inputs, the long-term "memory" of the sediment, the life span of striped bass, and the long-term projection period. The geographic extent of the model was specified from the Federal Dam at Troy out into the New York Bight and Long Island Sound with large spatial segmentation of the water column based on estuarine mixing behavior and the migration patterns of striped bass (see Figure 2-1). Underlying each water column segment was placed two to fourteen sediment segments with thicknesses ranging from 0.5 cm (in low deposition zones) to 2.5 cm (in high deposition zones) for a total of 30 water column segments and 120 sediment segments. The geometry of each segment is from Thomann et al (Thomann et al. 1989) and is given in Table A-1 of the Appendix.

Although total PCB concentration was used as a state variable in many previous model studies (e.g., New Bedford Harbor (Connolly 1991a)), this approach was not considered adequate for this study based on the large differences in physical-chemical and biochemical behavior of PCB congeners. Modeling the transport, fate, and bioaccumulation of a large number of congeners over decadal time periods in a 150 segment model however was not considered readily tractable. As a compromise solution, model calculations presented below were performed for the five PCB homologue groups (di-CB through hexa-CB) that contain the largest mass of PCBs.

With this as a basis for modeling PCBs in the Hudson River/New York-New Jersey Harbor ecosystem, a PCB homologue fate and bioaccumulation model was developed by Thomann et al (Thomann et al. 1989) as part of a previous grant from the Hudson River Foundation. The PCB homologue model for the Hudson River Estuary was initially run using a long-term average annual hydrology and sediment transport. Model calibration was performed for the period of 1946 (before PCB contamination) through 1987 using total PCB and Aroclor data for the water column, sediments, and striped bass. Model parameters and details of the calibration are given in Thomann et al (Thomann et al. 1989); (Thomann et al. 1991). Model projections for PCB concentrations in striped bass were then made assuming exponentially-decreasing loads from the Upper Hudson and the

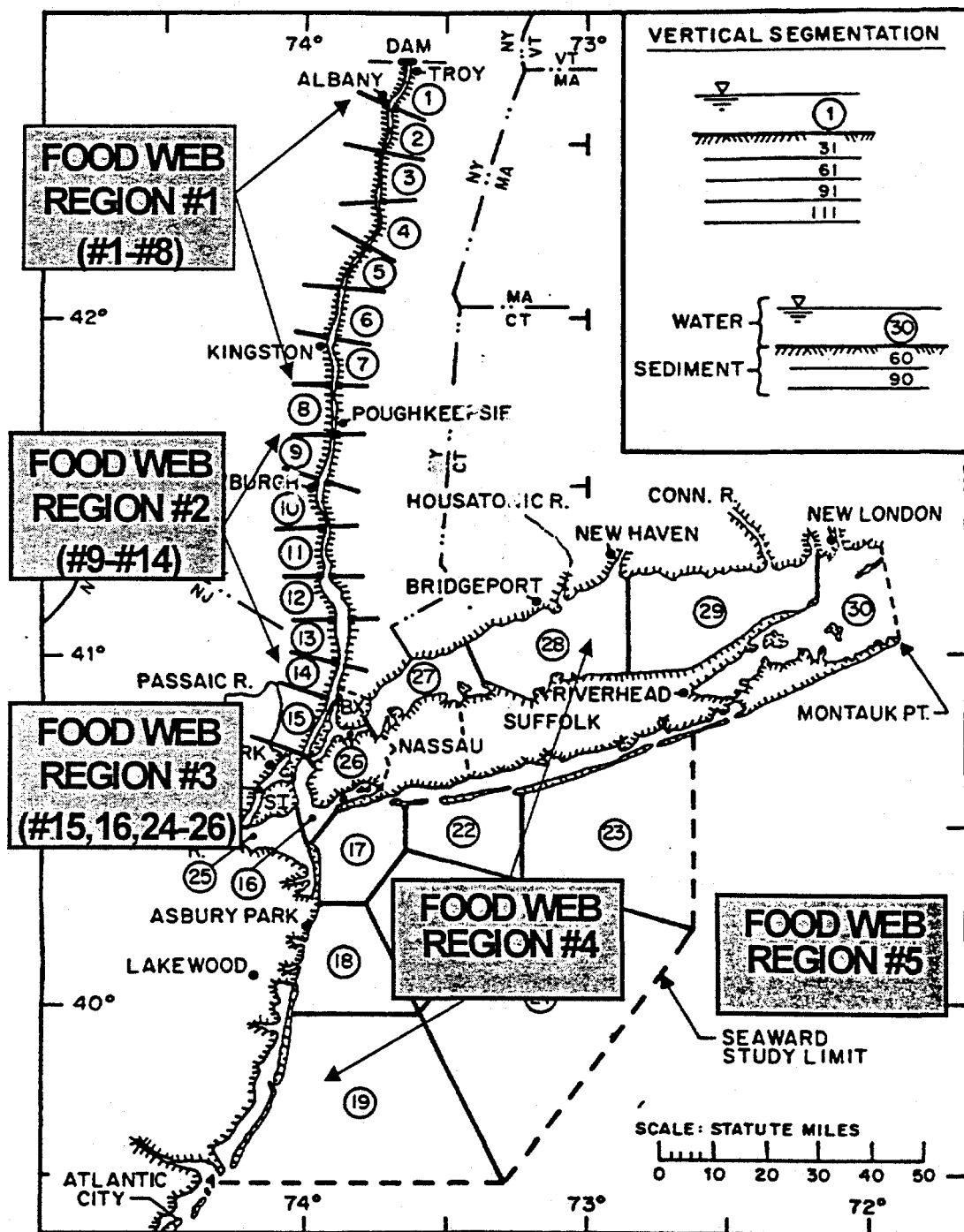


Figure 2-1. Large-space scale model for PCBs in the Hudson River Estuary, New York-New Jersey Harbor, New York Bight, and Long Island Sound. [Model segments are aggregated in five food web regions for bioaccumulation model calculations.]

downstream sources. Results of this work indicated that, under a no-action alternative, 50% of the striped bass would be below the Food and Drug Administration (FDA) limit of 2 µg of total PCB/g of fish (wet weight) by 1992 and 95 percent of the striped bass would be below the FDA limit by 2004.

As an extension of the initial modeling studies, we conducted a post-audit evaluation of PCB homologue model calculations by comparing PCB water column, sediment, and striped bass model results to data that have been collected since the 1987 model calibration. In this evaluation, we found that the model provided a reasonable description of PCB striped bass data (Figure 2-2). The model calculations however overestimated suspended solids concentrations typically observed in the estuary and severely underestimated PCB concentrations in the water column and in surface sediments (see Figure 2-3).

In response to these deficiencies in the model, the PCB homologue model was restructured to include seasonal hydrology and sediment transport. Modifications in relationships for PCB partitioning to organic carbon and phytoplankton were also made to correspond to more current formulations, and model parameters (e.g., flows, sediment transport, organic carbon, PCB loads, volatilization, bioaccumulation) were reviewed as part of the model recalibration. Descriptions of hydrodynamic transport, sediment transport, organic carbon distributions, chemical transport and fate calculations, PCB loads, initial conditions, and bioaccumulation modeling are given below, along with final parameter values used in the recalibrated PCB homologue model. Results and projections for the recalibrated PCB homologue model are presented in Chapter 3 for simulation periods starting in 1987.

Hydrodynamic Transport

Hydrodynamic transport in estuaries can be determined from simple hydrodynamic calculations (e.g., for the James River Estuary (O'Connor et al. 1983)), complex hydrodynamic models (e.g., for Chesapeake Bay (Johnson et al. 1993)), or from observations and simple flow balance calculations. For the original PCB homologue model (Thomann et al. 1989), flows were defined from a flow balance using average annual flow conditions from U.S. Geological Survey (USGS) river and tributary inflow records, surface runoff estimates, wastewater treatment plant discharge, and in the case of net circulation in the New York Bight, using observation of McLoughlin et al (as contained in O'Connor et al (O'Connor et al. 1977)). Dispersion coefficients, which describe horizontal mixing between adjacent water column segments, were calibrated using measured salinity concentrations. A summary of flows and dispersion coefficients for average-annual hydrologic conditions are given in Thomann et al (Thomann et al. 1989).

For the recalibrated PCB homologue model, hydrodynamic transport was specified on a seasonal basis throughout the system based on results from the Blumberg-Mellor three-dimensional, intratidal model (ECOM3D) that were developed as part of the System-

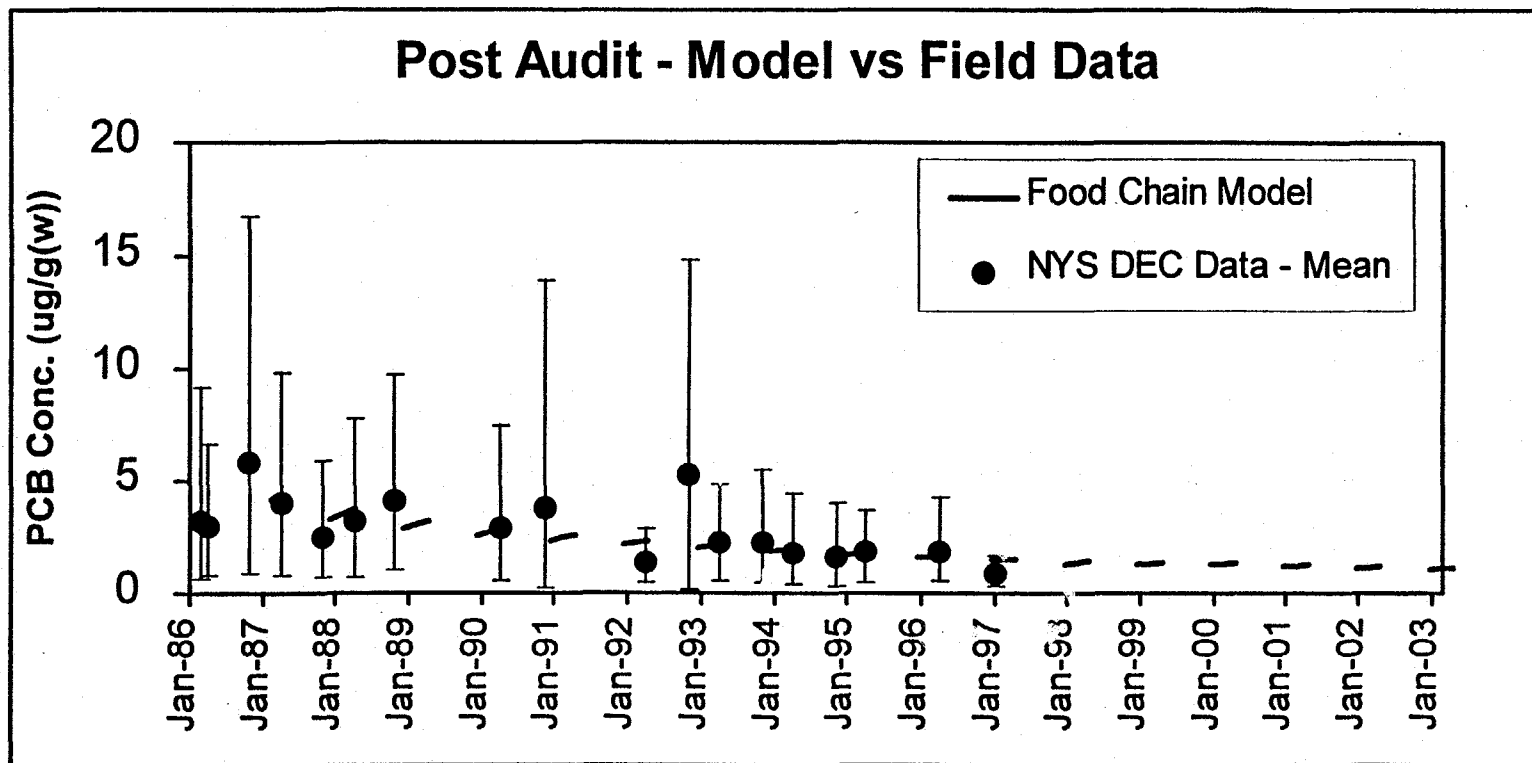


Figure 2-2. Post-audit evaluation of the original PCB homologue model (Thomann et al. 1989) for total PCBs in striped bass. [Data from NYS DEC as reported in TAMS/Gradient (1995).]

Post-Audit Evaluation

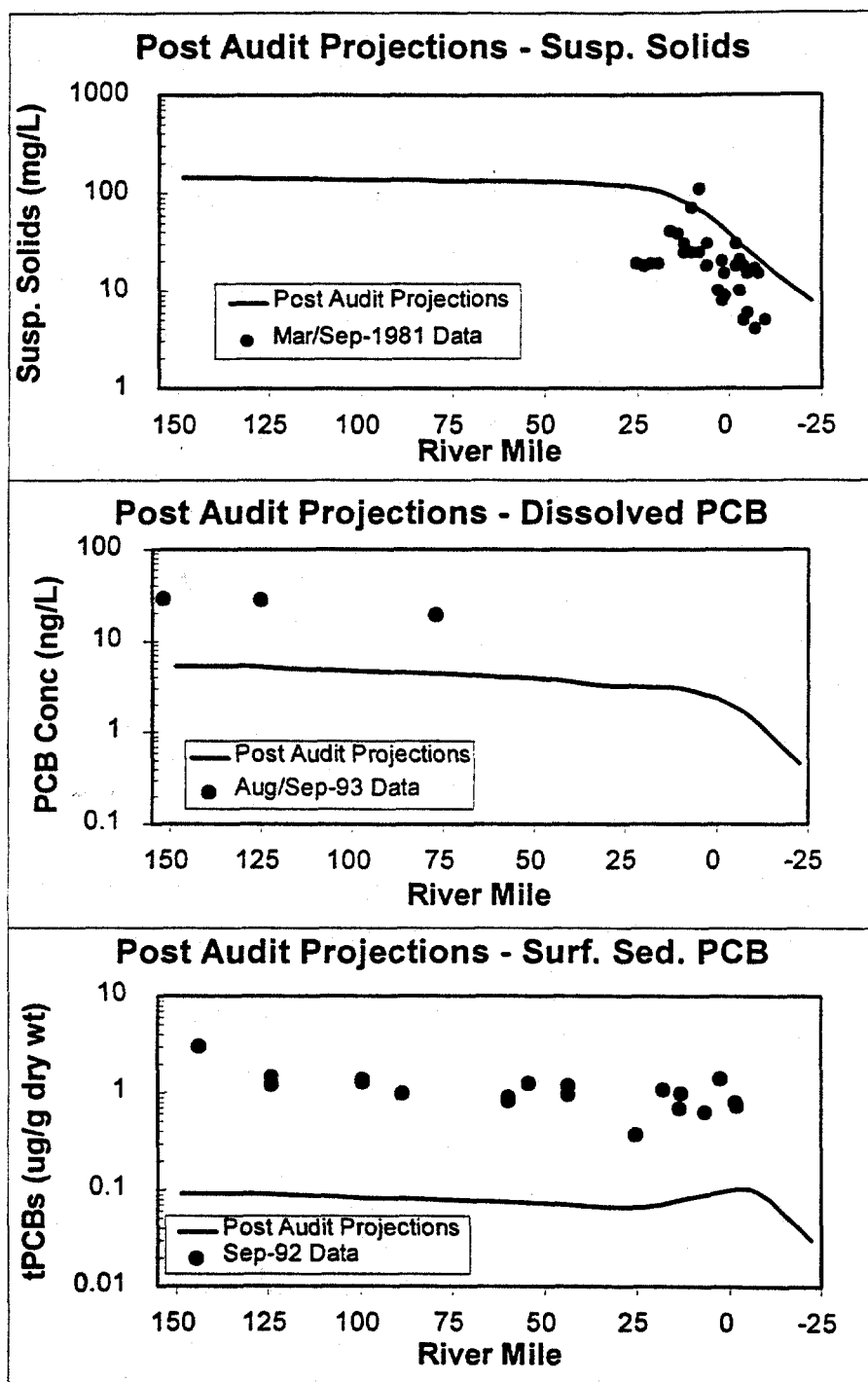


Figure 2-3. Post-audit evaluation of the original PCB homologue model (Thomann et al. 1989) for suspended solids, and for PCBs in the water column and surface sediments. [Data for suspended solids from Coch and Bokuniewicz (1986) as reported in HydroQual (1995); data for PCBs from TAMS/Gradient (1995).]

Wide-Eutrophication-Model (SWEM) study (HydroQual 1998). For this purpose, calibrated hydrodynamic model results for the 1989 water year were collapsed on to our 30 segment water column grid and seasonally-averaged to obtain Spring high flow (March-April) and Summer-Fall-Winter low flow (May-February) conditions (HydroQual 1998). The 1989 water year flows (with an combined annual discharge of 12,470 cfs from the Upper Hudson and Mohawk Rivers) were taken to be representative of an average hydrologic year for the study area and were repeated annually in the PCB simulations. Dispersion coefficients, which were taken from the original calibration by Thomann et al (Thomann et al. 1989) for average-annual hydrologic conditions, provided a good description of salinity distributions for the low flow period (Figure 2-4). As expected, the calculated salinity distribution is translated toward the ocean for the high flow period. Listings of seasonal flows and dispersion coefficients used in the recalibrated PCB homologue model are given in Tables A-2 and A-3, respectively.

Sediment Transport

Sediment transport models describing particle settling, coagulation, resuspension, and burial have been applied to rivers and estuaries with some success (e.g., Gailani et al (Gailani et al. 1991)), but these models typically require large data collection efforts for model parameterization and calibration. Solids balances have also been used in a number of studies such as our application for the Hudson River/New York-New Jersey Harbor system. In this approach, solids transport in a given segment is defined by the following mass balance equations

$$V_i \frac{dm_i}{dt} = \sum Q_{ij} m_j - \sum Q_{ji} m_i + \sum E_{ij} (m_j - m_i) + W_m - w_s A_{s_i} m_i + w_u A_{s_i} m_{sed_i} \quad (1)$$

$$V_{sed_i} \frac{dm_{sed_i}}{dt} = w_s A_{s_i} m_i - w_u A_{s_i} m_{sed_i} - w_b A_{s_i} m_{sed_i} \quad (2)$$

where Eq 1 represents the change in the mass of solids in water column segment 'i' with time and is equal to the mass rate of solids flowing into and out of segment 'i', plus the mass rate of solids dispersing into and out of segment 'i', plus the input of solids from external sources (e.g., tributary inflow), minus the loss of solids from the water column by settling, plus the gain of solids to the water column by resuspension. Eq 2 is a similar expression for the change in the mass of solids in a surface sediment segment with time which is equal to the addition of solids by settling from the overlying water, minus losses of solids by resuspension and burial. (A complete listing of terms used in the equations is given in the Appendix B.) Since the hydrolysis and decomposition of particulate organic matter is not considered as part of the solids balance in Eqs 1 and 2, these equations are

Salinity Distribution: Hudson River Transect

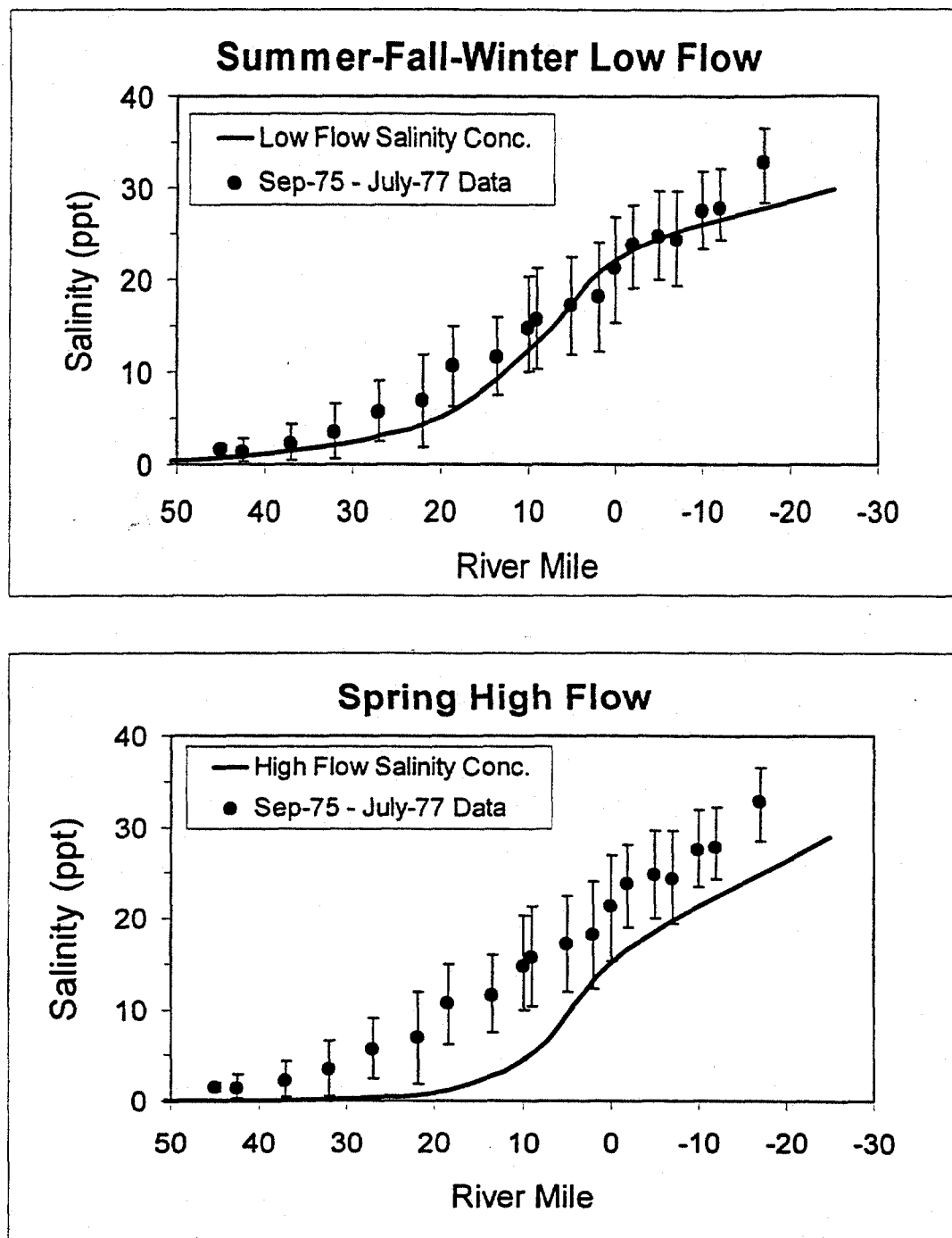


Figure 2-4. Calculated salinity distributions for the recalibrated PCB homologue model.
[Data from Hydrosience (1978).]

applicable to fixed solids concentrations (see adjustment below for *in-situ* phytoplankton production).

For the average-annual solids balance in the original PCB homologue model (Thomann et al. 1989), solids loadings were estimated for rivers, surface runoff, treatment plant discharges, barge disposal, and atmospheric loading. Estimates of the net sedimentation rates were made from an areal weighting of the sedimentation regions described by Bopp (Bopp 1979). For this calculation, sedimentation rates for dredged areas, coves and broad shallow areas, and sub-tidal banks were taken as 10 cm/yr, 1 cm/yr, and 0.1 cm/yr, respectively. Estimates of harbor dredging were taken from Olsen et al (Olsen et al. 1984). Sediment concentrations were determined to increase from 500,000 mg/L at the surface to 700,000 mg/L at 25 cm below the sediment surface based on analysis of more than 40 sediment cores (Thomann et al. 1989). Suspended solids concentrations at the Long Island Sound boundary and at the New York Bight boundary were set at 1.0 mg/L and 0.5 mg/L, respectively. These values, along with a specified settling velocity of 10 ft/day, were used to calculate average-annual suspended solids concentrations and sediment resuspension rates. Calculated average-annual suspended solids concentrations were approximately 100 mg/L in the tidal freshwater Hudson and mid-estuary region, and decreased to approximately 20 mg/L in the harbor (Thomann et al. 1989).

As shown previously (Figure 2-3), calculated average-annual suspended solids concentrations in the original PCB homologue model overestimated suspended solids concentrations typically observed in the estuary. For this reason, sediment transport for the recalibrated PCB homologue model was specified on a seasonal basis by considering Spring high flow (March-April) and Summer-Fall-Winter low flow (May-February) periods. For the Upper Hudson and Mohawk Rivers, the annual sediment load was taken as 462,000 Mtons/yr, based on 1971-90 monthly estimates of suspended sediment yields from Phillips (Phillips 1998). This value is approximately four times lower than the sediment discharge rate used in the original PCB homologue model of Thomann et al (Thomann et al. 1989). For the Upper Hudson and Mohawk Rivers, 65% of the solids were assumed to discharge during the two month Spring high flow period, based on an evaluation of maximum 60-day running average for solids discharge rates for the Mohawk at Cohoes during the 1978-1996 period. The remaining 35% of the solids were assumed to discharge during the ten month low flow period.

Annual sediment loads from the Lower Hudson watershed (from Albany to New York City area) were estimated by Swaney et al (Swaney et al. 1998) to be 206,000 Mtons/yr. This load was distributed equally among segments #2 through #14 in the model calculations. Annual sediment loads for the New Jersey tributaries (i.e., the Passaic, Hackensack, and Raritan Rivers) were taken from HydroQual (HydroQual 1991) and totalled 71,400 Mtons/yr. Annual sediment loads for the Connecticut tributaries (primarily the Connecticut River) were taken from Farrow et al (Farrow et al. 1986) and total 135,900 Mtons/yr. Seasonal distribution of 65% of solids being discharged during the two month Spring high flow and 35% of the solids being discharged during the ten month Summer-

Fall-Winter low flow period were assumed for the Lower Hudson, New Jersey, and Connecticut tributaries to mimic the Upper Hudson and Mohawk Rivers.

Solids loads from WWTPs, CSOs, storm water discharge, and industry were taken from HydroQual (HydroQual 1991) and totalled 169,500 Mtons/yr. WWTPs and industrial discharges were assumed to be constant throughout the year. CSOs and storm water discharges were considered to vary seasonally following the 65%-35% distribution that was specified above for tributary loadings.

Atmospheric loading of solids was calculated assuming an average deposition rate of 105 kg/ha/year (HydroQual 1991). This value was multiplied by the water surface area to determine atmospheric loadings for each segment. Based on these calculations, atmospheric loading contributes 35,400 Mtons/yr, with only a small portion of the load (1,500 Mtons/yr) entering the HudsonRiver/New York-New Jersey Harbor segments (Segments 1-16, 24-26). The remainder of the atmospheric solids loads was inputted to the New York Bight and Long Island Sound segments.

In-situ production of solids by phytoplankton was calculated assuming an average productivity rate of 1.0 gC/m²-day. To calculate the mass of fixed solids produced per day, this value was multiplied by the water surface area, a stoichiometric conversion factor of 2.6 g dry wt/gC, and a factor of 0.2 (to account for an 80% decomposition of phytoplankton organic solids in the water column and surface sediment layer). Based on these calculations, the total load of fixed solids from phytoplankton production is 3,542,500 Mtons/yr, again with only a small portion of the production (35,400 Mtons/yr) occurring in the HudsonRiver/New York-New Jersey Harbor segments. An overall summary of solids loadings into each model segment is given in Table A-4 of the appendix.

Estimates of the net sedimentation rates were again made from an areal weighting of the sedimentation regions described by Bopp (Bopp 1979). For this calculation, average-annual sedimentation rates were taken to be 20% of the values used in Thomann et al (Thomann et al. 1989) (i.e., 2 cm/yr for dredged areas, 0.2 cm/yr for coves and broad shallow areas, and 0.02 cm/yr for sub-tidal banks).

Estimates of harbor dredging of fine-grained material were computed from Army Corps of Engineers dredging records (with new construction and dredging of sand from the entrance of Ambrose Channel not considered in the computation) (Suszkowski 1998). Average-annual dredging rates were calculated for the last five years of record (1987 through 1991) to coincide with model simulations starting in 1987. The average annual dredging rate for this period was 2.45 million cubic yards (MCY). This value was converted into mass per time measurement by multiplying the 2.45 MCY by 81.4% (to account for differences in barge volume and in-place sediment volume) and an assumed solids concentration of 400,000 mg/L for dredged sediment (Suszkowski 1998). Based on these calculations, the total amount of dredged material removed from the harbor annually is 595,900 Mtons/yr. As in Thomann et al (Thomann et al. 1989), dredging from the tidal freshwater portion of the Hudson was assumed to be 10% of the harbor value. The

dredged material accumulation in the tidal freshwater portion of the river was distributed in the sediments underlying Segments #1 through #7 following the distribution given in Thomann et al (Thomann et al. 1989). Tabulated results for dredged sediment are given in Table A-5 in the Appendix.

Accumulation of bottom sediment and dredged material in the Hudson River, New York-New Jersey Harbor, Long Island Sound, and the New York Bight is expected to occur at varying rates throughout the year. For our seasonal solids balance, seasonal variations in the accumulation of bottom sediment and dredged material were therefore specified as follows: For the Hudson River (Segments 1-14), which is largely affected by tributary inflows, 65% of the accumulation is considered to occur in the two month Spring high flow period and 35% to occur during the remainder of the year. For the harbor and Long Island Sound (Segments 14-16, 24-30), which are affected by both tributary and seasonally-constant solids loadings, 45% of the accumulation is considered to occur in the two month Spring high flow period and 55% to occur during the remainder of the year. For the New York Bight (Segments 17-23), which is less affected by tributary loadings, 25% of the accumulation is considered to occur in the two month Spring high flow period and 75% to occur during the remainder of the year. A summary of annual sediment and dredged material accumulation rates, and seasonal distributions for accumulation is given in Table A-6 in the appendix.

Finally, sediment concentrations were again specified as increasing from 500,000 mg/L at the surface to 700,000 mg/L at 25 cm below the sediment surface based on analysis of more than 40 sediment cores (Thomann et al. 1989). Suspended solids concentrations at the Long Island Sound boundary and at the New York Bight boundary were set at 1.0 mg/L and 0.5 mg/L, respectively. These values, along with a specified settling velocity of 10 ft/day, were used to calculate seasonal distributions in suspended solids concentrations and sediment resuspension rates (see Table A-7 in the appendix). Calculated suspended solids concentrations are presented in Figure 2-5 for a transect along the mainstem of the Hudson River, through New York-New Jersey Harbor, and out into the New York Bight. As shown in Figure 2-5, calculated suspended solids concentrations for the Spring high flow and Summer-Fall-Winter low flow periods generally show a good correspondence with the field measurements of (Coch and Bokuniewicz 1986). Notable exceptions are the higher observed suspended solids concentrations for River Mile 5-20 for the low flow period which are likely associated with a turbidity maximum and are difficult to simulate using a vertically-integrated water column model.

Organic Carbon Distributions

PCBs and other hydrophobic organic contaminants have a strong affinity to sorb onto particulate (POC) and dissolved organic carbon (DOC). For the original PCB homologue model (Thomann et al. 1989), POC was expressed in terms of the fraction organic carbon (f_{oc}), where f_{oc} was specified as 0.072 in the water column and 0.024 in the sediments based on data from Olsen et al. (Olsen et al. 1984). For the water column, DOC

Suspended Solids Distribution: Hudson River Transect

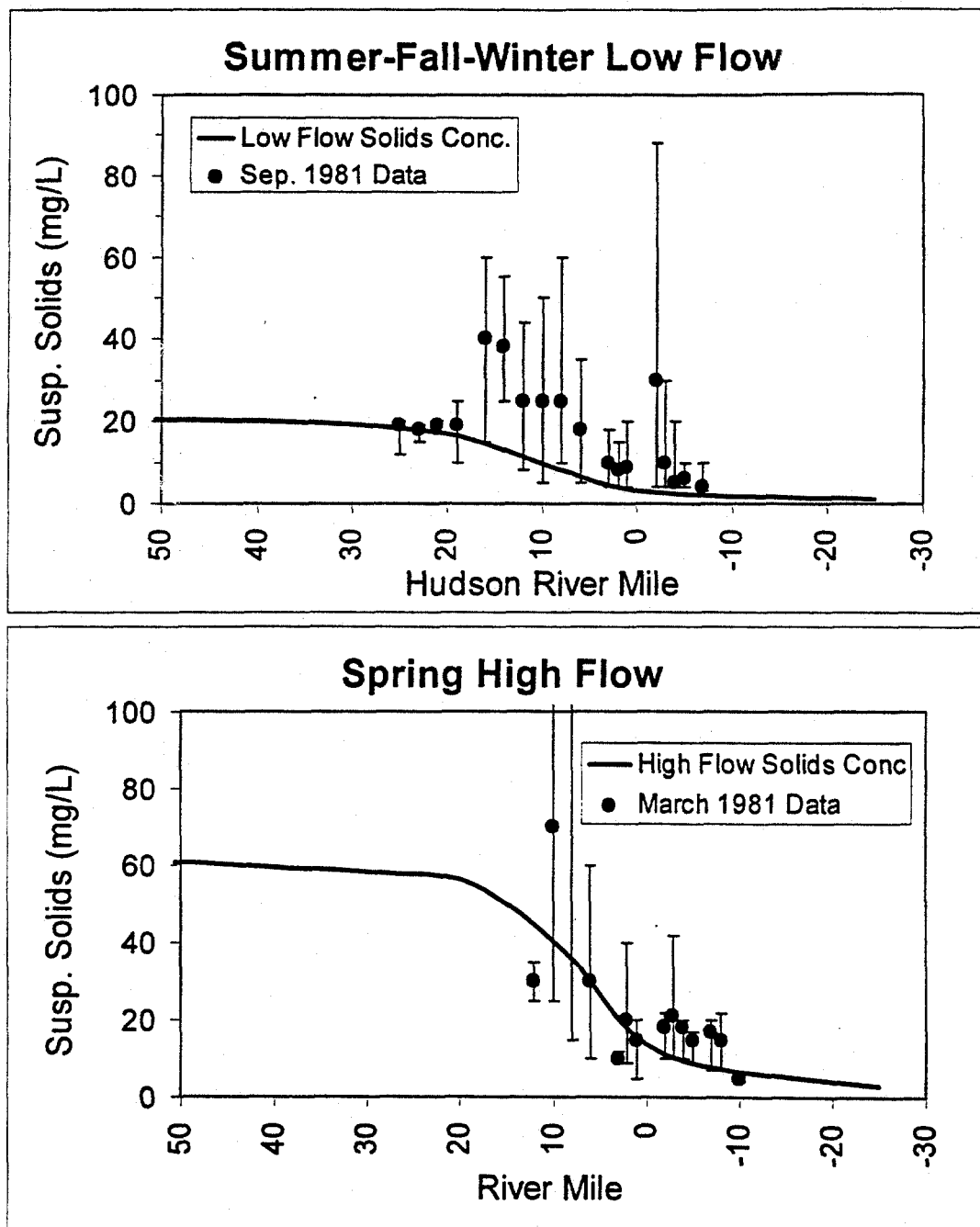


Figure 2-5. Calculated suspended solids distributions for the recalibrated PCB homologue model. [Data from Coch and Bokuniewicz (1986) as reported in HydroQual (1995).]

was not explicitly considered in the binding of PCBs. For the sediments, DOC in the porewater was taken as 10 mg/L.

For the recalibrated PCB homologue model, POC and DOC concentrations in the water column were determined from 1989 SWEM results (HydroQual 1998). The SWEM results were collapsed onto our 30 segment water column grid and averaged over the two month (March-April) Spring high flow and ten month (May-February) Summer-Fall-Winter low flow periods (HydroQual 1998). The resulting spatial distributions of POC and DOC along the mainstem of the Hudson River, through New York-New Jersey Harbor, and out into the New York Bight are given in Figure 2-6. Note that the increased organic carbon concentrations near River Mile 0 are largely the result of sewage discharge of organic matter and nutrients (with some nutrient-enriched phytoplankton production) from the New York metropolitan area. The effect of increased organic carbon concentrations on PCB transport and fate is discussed in Chapter 3.

For our model calculations, POC is again expressed in terms of the fraction organic carbon (f_{oc}). Listing of POC, DOC, fixed suspended solids concentrations, and computed f_{oc} values are given in Tables A-8 and A-9 of the appendix for the Spring high flow and Summer-Fall-Winter low flow periods. [Note that since our solids balance is based on fixed (inorganic plus refractory organic) solids, f_{oc} values in excess of one are possible, particularly for areas of the harbor and the bight where concentrations of phytoplankton and sewage organic carbon are greater than fixed solids concentrations.] Since f_{oc} and DOC values are specified as constants for each segment in the model input for WASTOX (Connolly 1991b), the ten month (May-February) Summer-Fall-Winter low flow values were taken as representative of the entire year. For the sediments, the fraction organic carbon was again taken as 0.024 based on data from Olsen et al. (Olsen et al. 1984) and DOC in the porewater was again taken as 10 mg/L.

Chemical Transport and Fate

Following evaluations for hydrodynamic, sediment transport, and organic carbon, the transport and fate of toxic organic chemicals (such as PCBs) in estuaries can be expressed by a set of mass conservation equations--one for each water column and sediment segment based on the schematic diagram shown in Figure 2-7. For a vertically well-mixed water column, the mass conservation equation for total (i.e., freely-dissolved, DOC-bound plus particulate) concentration of a specific chemical constituent (e.g., a PCB homologue) can be written for each water column segment as

$$V_i \frac{dC_i}{dt} = \sum Q_{ij} C_j - \sum Q_{ij} C_i + \sum E_{ij} (C_j - C_i) + W_{C_i} - w_{s_i} A_{s_i} m_i \Gamma_i + w_{u_i} A_{s_i} m_{sed_i} \Gamma_{sed_i} - k'_f A_{s_i} (C_{dis+DOC_{sed_i}} - C_{dis+DOC_i}) + k'_v A_{s_i} \left(\frac{P_C (MW)}{K_H} - C_{dis_i} \right) - k_i V_i C_{dis_i} \quad (3)$$

Organic Carbon Distributions

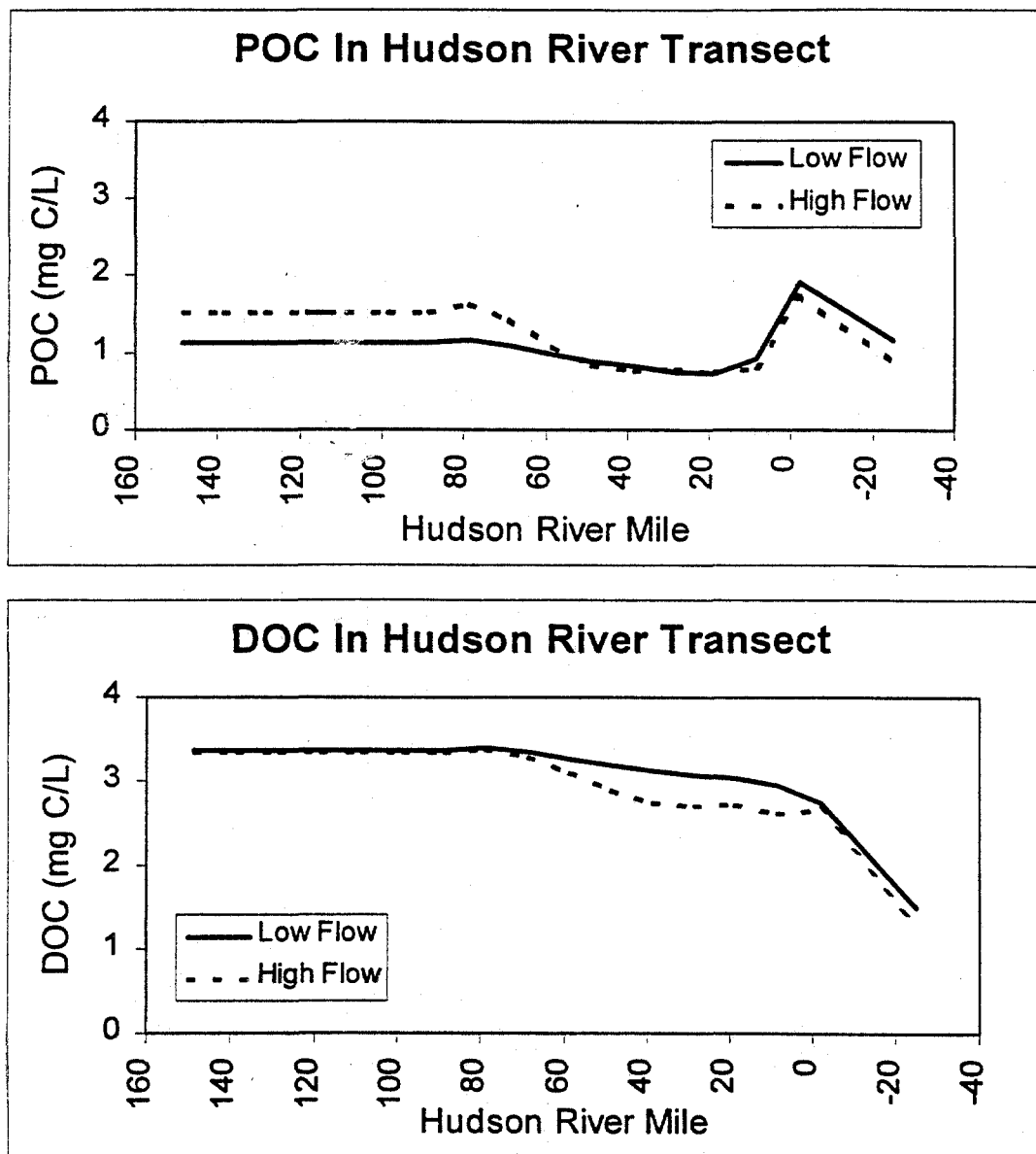


Figure 2-6. Spatial distributions of POC and DOC from 1989 SWEM model results. Profiles shown for the mainstem of the Hudson River, through New York-New Jersey Harbor, out into the New York Bight.

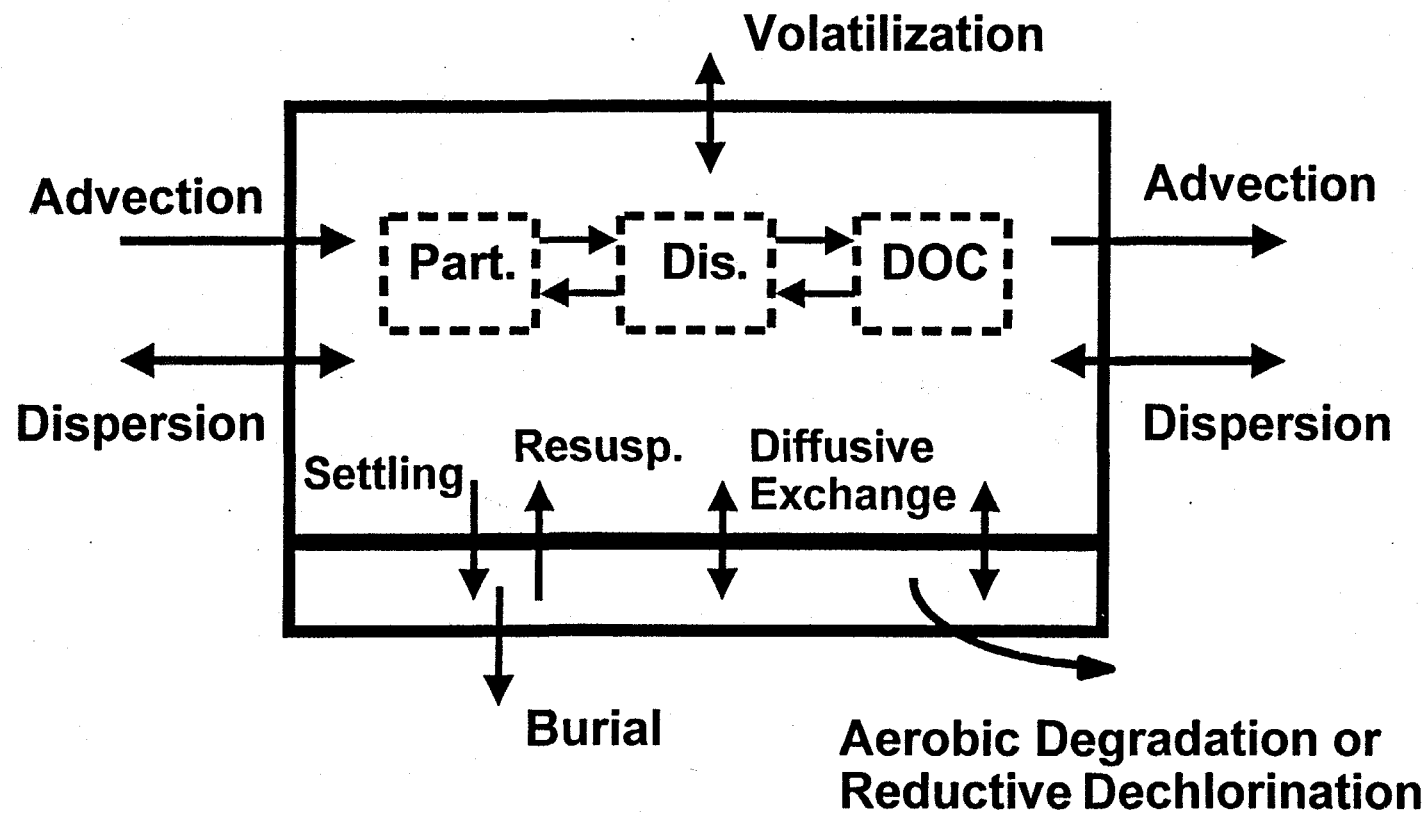


Figure 2-7. Schematic diagram of processes affecting the transport and fate of PCB homologues in the water column and surface sediments.

where the first term represents the time rate of change of the freely-dissolved, DOC-bound, plus particulate chemical concentration in the water column; the second and third terms represent the mass rate of chemical flowing into and out of segment 'i', respectively; the fourth term represents chemical entering or leaving segment 'i' by dispersion; the fifth term represents chemical input into segment 'i' from an external source (e.g., tributary input or wastewater discharge); the sixth term represents chemical loss from the water column by settling; the seventh term represents chemical gain from resuspension; the eighth term represents diffusive exchange between freely-dissolved and DOC-bound concentrations in the water column and pore waters; the ninth term represents the transfer of chemical across the air-water interface (i.e., volatilization); and the last term represents transformation losses from the water column (e.g., by aerobic degradation). (A complete listing of terms used in the equations is given in Appendix B.)

Similar equations can be written for chemical concentrations in the sediments. For example, total (freely-dissolved, DOC-bound plus particulate) concentration for a chemical constituent in a stationary surface sediment layer can be written as

$$\begin{aligned}
 V_{sed_i} \frac{dC_{sed_i}}{dt} = & w_{s_i} A_{s_i} m_i \Gamma_i - w_{u_i} A_{s_i} m_{sed_i} \Gamma_{sed_i} - w_{b_i} A_{s_i} m_{sed_i} \Gamma_{sed_i} \\
 & - k'_f A_{s_i} (C_{dis+DOC_{sed_i}} - C_{dis+DOC_i}) - k_{sed_i} V_{sed_i} C_{dis_{sed_i}} \\
 & + k'_f A_{s_i} (C_{dis+DOC_{deep\ sed_i}} - C_{dis+DOC_{sed_i}})
 \end{aligned} \tag{4}$$

where the first term represents the change in the total (freely-dissolved, DOC-bound plus particulate) chemical mass with time; the second term represents the gain of chemical by settling from the overlying water, the third and fourth terms represent the loss of chemical by resuspension and burial into deeper sediments, respectively; the fifth term represent diffusive exchange of freely-dissolved and DOC-bound chemical with the overlying water; and the sixth term represents transformation losses from the sediments (e.g., by anaerobic dechlorination); and the last term represents diffusive exchange of freely-dissolved and DOC-bound chemical with the deeper sediment pore water. Chemical gain, e.g., by dechlorination of higher chlorinated congeners, is also possible.

Since sorption plays an important role in determining the partitioning of chemical between the freely-dissolved, DOC-bound, and particulate phases, its definition is essential in describing the various flux terms in Eqs 3 and 4. In modeling studies, sorption reactions are usually assumed to be fast compared to other environmental processes and are typically modeled as instantaneous (or equilibrium) reactions (e.g., Ambrose et al (Ambrose et al. 1988); Mossman et al (Mossman et al. 1988); O'Connor (O'Connor 1988a)). For contaminant transport studies, chemical partitioning of hydrophobic organic contaminants between the freely-dissolved and particulate phases is considered to be a function of the the fraction organic carbon on the solids and the octanol-water partitioning coefficient (K_{ow}) (Karickhoff 1981).

For the original PCB homologue model (Thomann et al. 1989), the partition coefficient for the dissolved (freely-dissolved plus DOC-bound) and particulate phase was described using the Particle Interaction Model (DiToro 1985)

$$K_d = \frac{f_{oc} K_{ow}}{1 + \frac{f_{oc} K_{ow} m}{1.4}} \quad (5)$$

in terms of f_{oc} , K_{ow} , and the solids concentration (m). In this formulation, K_d increases linearly with octanol-water partitioning for low K_{ow} compounds and approaches a constant value of $1.4/m$ for high K_{ow} compounds. [This relationship can be used to mimic the effects of DOC binding on the apparent partitioning of chemical between the dissolved (freely dissolved plus DOC-bound) and particulate phases.] Following Thomann and Salas (Thomann and Salas 1986), the computed K_d value was then increased by a factor of two to account for irreversibly bound PCBs. $\log K_{ow}$ values used in the original PCB homologue model (Thomann et al. 1989) are given in the second column in Table 2-1 and vary from 4.9 for di-CB to 6.7 for hexa-CB.

For the recalibrated PCB homologue model, chemical partitioning between the freely-dissolved, DOC-bound, and particulate phases is explicitly considered. Here, chemical concentrations in the various phases can be expressed in terms of the total chemical concentrations using the equilibrium partitioning relationships [$K_d = \Gamma/C_{dis}$ and $K_{DOC} = (C_{DOC}/DOC)/C_{dis}$] and the total mass concentration equation [$C = \phi C_{dis} + \phi C_{DOC} + \Gamma m$] as:

$$C_{dis} = \frac{C}{\phi + \phi K_{DOC} DOC + K_d m} \quad (6)$$

$$C_{DOC} = \frac{K_{DOC} (DOC) C}{\phi + \phi K_{DOC} DOC + K_d m} \quad (7)$$

$$C_{part} = \Gamma m = \frac{K_d m C}{\phi + \phi K_{DOC} DOC + K_d m} \quad (8)$$

where ϕ is the porosity (and is approximately equal to one in the water column). Similar expressions can be written for freely-dissolved, DOC-bound, and particulate phase chemical concentrations for the sediment layers.

Table 2-1: log Kow Values for PCB Homologue Model				
	log Kow (L/kg) Values			
	^a Original PCB Homologue Model	^b Weighted-Average K _{ow} from Surface Sediment Data	^b Weighted-Average K _{ow} from Perch Data	Recalibrated PCB Homologue Model
di-CB	4.9	5.07	4.96	5.0
tri-CB	5.4	5.61	5.63	5.6
tetra-CB	5.8	6.01	5.97	6.0
penta-CB	6.2	6.42	6.46	6.45
hexa-CB	6.7	6.83	6.85	6.85
Notes: ^a Kow's for original PCB homologue model from Thomann et al. (Thomann et al. 1989). ^b Weighted-average K _{ow} 's from observed congener distributions for 1992 high-resolution surface sediment data and 1993 eco-perch data (TAMS/Gradient 1995) and PCB congener K _{ow} values of Hawkins and Connell (Hawker and Connell 1988).				

Partitioning between the freely-dissolved and particulate phases is assumed to be directly related to the octanol-water partition coefficient as:

$$K_d = f_{oc} K_{ow} \quad (9)$$

which is equivalent to Eq 5 when the second term in the denominator is small (i.e., for low K_{ow} compounds). f_{oc} values used for our evaluations were discussed in the previous section. K_{ow} values for the various homologue groups were determined from weighted averages of observed congener distributions in 1992 high-resolution surface sediments and 1993 eco-perch data (TAMS/Gradient 1995) and K_{ow} values for individual PCB congeners (as reported in Hawkins and Connell (Hawker and Connell 1988)). Computed K_{ow}'s from the observed surface sediment and perch congener concentrations are in good agreement. Final K_{ow} values chosen for the recalibrated PCB homologue model are given

in the last column of Table 2-1 and are approximately 0.2 log units higher than values used in the original model.

Partitioning of PCBs between the freely-dissolved and DOC-bound phases was also assumed to be directly related to the octanol-water partition coefficient as:

$$K_{DOC} = a_{DOC} K_{ow} \quad (10)$$

where the factor a_{DOC} was set at 0.1 for the water column and 1.0 for the sediment segments. The lower a_{DOC} value for the water column was used to account for differences in PCB partitioning to lower molecular weight DOC compounds and octanol. Note that an order of magnitude decrease in the PCB partitioning to DOC is consistent with results presented by Garvey et al (Butcher et al. 1998) for the Upper Hudson.

Volatilization rate coefficients are typically calculated using the two layer model of the air-water interface (Butcher et al. 1998). For rivers and estuaries, mass transfer coefficients for the water side of the interface can be calculated using the O'Connor-Dobbins formula (O'Connor and Dobbins 1958)

$$k_v' = \sqrt{\frac{D_w U}{h}} \quad (11)$$

where D_w is the molecular diffusivity of PCBs in water; U is the average water velocity (and is often taken as the average tidal velocity in estuaries); and h is the mean water depth. Mass transfer coefficients for the air side of the interface are often estimated from water evaporation rates (with a correction for differences in the molecular diffusivity of the organic chemical and water vapor in air).

For PCBs in the Hudson River Estuary, transfer through the water side of the interface was assumed to control the overall transfer rate. In the original PCB homologue model (Thomann et al. 1989); (Thomann et al. 1991), a PCB volatilization rate of 0.6 m/day was estimated using an assumed oxygen transfer rate of 1.0 m/day times a molecular weight adjustment of $(32/300)^{0.25}$.

A similar approach was used in estimating the volatilization rate for the recalibrated PCB homologue, except the exponent for the molecular weight adjustment was taken as 0.355 based on a square root dependency of molecular diffusivity in the O'Connor-Dobbins formula (O'Connor and Dobbins 1958) and a molecular diffusivity-molecular weight relationship of

$$D_w \propto MW^{-0.71} \quad (12)$$

as given in Schwarzenbach et al (Schwarzenbach et al. 1993). The oxygen transfer rate was again taken as 1.0 m/day (based on computed values in HydroQual (HydroQual 1996)) and resulted in a computed volatilization rate coefficient of approximately 0.5 m/day. This value is consistent with results of Clark et al (Clark et al. 1996) for gas exchange rates as determined from a sulfur hexafluoride and helium-3 tracer study in the tidal freshwater Hudson. (Using the square root dependency for molecular diffusivity and the molecular diffusivity-molecular weight relationship given in Eq 12, the 2.8 to 9.2 cm/hr CO₂ gas exchange rates reported by Clark et al. (Clark et al. 1996) correspond to a PCB volatilization rate of 0.34 to 1.1 m/day.)

Dissolved chemical exchange between pore water and the overlying water column is dependent on the detailed hydrodynamic structure at the water-sediment interface and can be greatly enhanced by bioturbation. For most applications, the pore water exchange rate coefficient (k'_p) is specified by some multiple of molecular diffusivity (e.g., 10 times) divided by the thickness of the sediment segment. For our studies, we used a value of approximately two times the molecular diffusivity of PCBs (i.e., 10⁻⁵ cm²/sec). Since settling and resuspension of particle-bound PCB dominated the chemical transfer rates across the water-sediment interface, the transport and fate of the PCB homologues were not found to be sensitive to the actual selection of pore water exchange rate coefficient.

Transformations of toxic organic chemicals can occur in the aquatic systems by hydrolysis, photolysis, biodegradation, and reductive dechlorination reactions. Although PCBs were originally thought to be refractory organic compounds, a number of studies have show that certain PCB congeners may be degraded under aerobic conditions or microbially dechlorinated under anaerobic conditions in aquatic environments (Abramowicz 1990). The major conclusion from aerobic degradation studies (Bedard et al. 1986); (Bedard et al. 1987a); (Bedard et al. 1987b) is that biodegradation of PCBs can occur by the attack of a dioxygenase enzyme at an unchlorinated 2,3 (or 5,6) site or at an unchlorinated 3,4 (or 4,5) site. These attacks result in cleavage of the biphenyl ring and can be carried out by a variety of naturally-occurring bacteria. Congeners with chlorines at both ortho (2,6) positions on either ring are generally not degraded as readily as congeners lacking this characteristic.

Under anaerobic conditions, organisms leave the biphenyl ring intact while removing chlorines from the ring, thereby producing less chlorinated congeners. Although details of the dechlorination process are not fully understood, reductive dechlorination has been shown to proceed primarily through the selective removal of meta (3,5) and para (4) chlorines (Abramowicz 1990; Abramowicz et al. 1993; Quensen et al. 1988; Rhee et al. 1993a; Rhee et al. 1993b; Rhee et al. 1993c). In the Hudson, aerobic degradation and anaerobic dechlorination have been observed in the upper river (above the Federal Dam at Troy). However, in the estuary, aerobic degradation and anaerobic dechlorination are

not thought to be important—for aerobic degradation, due to low oxygen concentrations in sediments, and for anaerobic dechlorination, possibly due to PCB concentrations being below a “threshold” value.

PCB Loads

PCB concentrations in sediments and in the overlying water are dependent of PCB loadings and initial conditions. In most contamination studies, direct information on chemical loadings is not available and various estimates are employed in calculating loads. For the original PCB homologue model (Thomann et al. 1989), estimates of PCB loads over Federal Dam at Troy for 1947-1975 were determined from PCB concentrations in sediment cores upstream of the dam (Thomann et al. 1989). For 1976 to 1987, PCB loads over the dam were determined from measured water column concentrations and flows. Downstream PCB loads were determined from few measurements of PCBs in wastewater treatment plant effluents, wastewater treatment plant influents (to estimate loads from combined sewer overflows), tributaries, and rainfall (to estimate direct atmospheric inputs) (Thomann et al. 1989). For model projections (1987 to 2012), upstream and downstream loads were assumed to decline at exponential rates of 0.28 and 0.057 per year, respectively.

For the recalibrated PCB homologue model, loadings over Federal Dam at Troy were specified on a monthly basis. For January 1987-April 1991, PCB homologue loads were determined by adjusting the exponentially decreasing average-annual load function for total PCBs given in Thomann et al (Thomann et al. 1989) to monthly loadings assuming 68% of the annual PCB load entered during the two month Spring high-flow period and 32% entered during the remaining ten months of the year. (This assumed seasonal distribution in PCB loading is based on information given in Limno-Tech et al. (Limno-Tech et al. 1996).) Homologue distributions for the January 1987-April 1991 time period were taken to be equal to measured distributions in May 1991. For May 1991-December 1997, monthly PCB loading were calculated from measured concentrations and estimated flows at Thompson Island Dam (O'Brien & Gere 1997). For this calculation, flows at Thompson Island Dam were taken to be 1.06 times gaged flows at Fort Edwards based on incremental drainage area (Connolly 1997). Monthly PCB loadings for all years after 1997 were assumed to follow 1997 monthly loads. Based on these computations, monthly-averaged PCB loads (for di-CB through hexa-CB) from the Upper Hudson were as high as 20-50 lbs/day during 1991-92 and have since decreased to approximately 1.5 lbs/day in 1997 (Figure 2-8).

All other PCB homologue loads (i.e., for the Mohawk River, the New Jersey tributaries, New York City and New Jersey wastewater treatment plants (WWTPs), combined sewer outflows (CSOs), and direct atmospheric loads) were taken to be constant throughout the model simulation period and were calculated as follows: For the Mohawk River, PCB homologue loads were determined from the average annual flow for the 1989 water year times the average measured concentrations of PCB homologues in the Mohawk

Monthly-Averaged PCB Load from the Upper Hudson

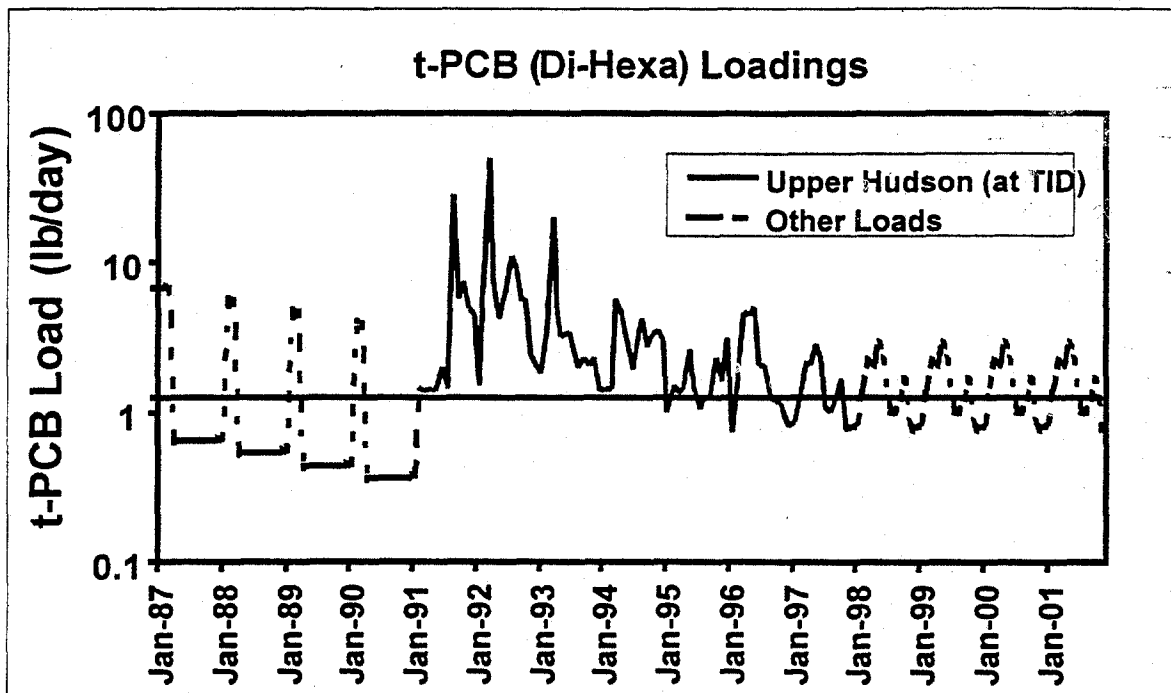


Figure 2-8. Monthly-averaged loads for total (di- through hexa-) PCBs from the Upper Hudson. [Upper Hudson loads based on measured PCB concentrations at Thompson Island Dam (O'Brien & Gere 1997).]

River (TAMS/Gradient 1995). Since measured PCB concentrations were not available for the Passaic, Hackensack, and Raritan Rivers, the averaged measured PCB homologue concentrations in the Mohawk were assumed to be representative of concentrations in the New Jersey tributaries. The Mohawk River PCB homologue concentrations were then used along with the 1989 water year average annual flows of the Passaic, Hackensack, and Raritan Rivers to calculate PCB loads for the New Jersey tributaries. Tabulated results for PCB homologue loadings from the Mohawk, Passaic, Hackensack, and Raritan Rivers are given in Table A-10 in the appendix. Based on these computations, the Mohawk River accounts for 0.37 lbs/day of total PCBs, and the New Jersey tributaries accounts for 0.27 lbs/day of total PCBs.

Wastewater Treatment Plant (WWTP) PCB homologue loads were determined from plant flow rates times average effluent concentrations for two 1994 dry weather events (Battelle 1995)(Battelle 1995b). Tabulated results for the fourteen New York City and the twelve New Jersey WWTPs are given respectively in Tables A-11 and A-12 in the appendix. Based on these calculations, the New York City WWTPs discharged 0.384 lb/day of total PCBs during the 1994 sampling period, with Newtown Creek contributing 36% of the total. The New Jersey WWTPs contributed 0.186 lb/day during the 1994 sampling study, with Passaic Valley contributing 56% of the total. It should be noted that these loadings for the New York City and New Jersey WWTPs are approximately a factor of two less than the projected 1994 PCB loadings used in the original PCB homologue model projections by Thomann et al (Thomann et al. 1989).

For New York City combined sewer overflows (CSOs), PCB homologue loads were calculated for individual treatment plant service areas. PCB concentrations in the overflows were taken as the average treatment plant influent concentrations for two wet weather events (Battelle 1995b). These homologue concentrations were then multiplied by CSO estimated flows (which were computed using the New York City 208 Rainfall-Runoff Model and reported in HydroQual (HydroQual 1991)) to obtain PCB homologue loads. Since only a total flow was reported for New Jersey CSOs in HydroQual (HydroQual 1991), PCB homologue concentrations in the New Jersey overflows were determined by multiplying the total CSO flow by an average of the twelve treatment plant influent concentrations for two wet weather events (Battelle 1995). The entire New Jersey CSO load was considered to enter Segment #24 (which encompasses the lower Passaic and Hackensack Rivers, Newark Bay, the Arthur Kill, and the Kill van Kull). Tabulated results for the PCB homologue loads from the New York City and New Jersey CSOs is given in Table A-13 in the appendix. The computed results show that CSOs account for 0.309 lbs/day of total PCBs to the harbor. This loading estimate is also less than the projected 1994 PCB loadings used in the original PCB homologue model projections by Thomann et al (Thomann et al. 1989).

Direct PCB load estimates from wet deposition and back diffusion from atmospheric gases were calculated as follows: Average total PCB concentrations in the New Jersey urban (New Brunswick) and coastal (Sandy Hook) atmosphere in early 1998 have been reported as 395 pg/m³ and 227 pg/m³, respectively (Brunciak et al. 1998). These values

are comparable to atmospheric PCB concentrations measured west of Minneapolis/St. Paul, MN in the winter of 1991-92 (Franz and Eisenreich 1998). PCB homologue concentrations that were reported for the air on 12/11-12/91 and for rain on 12/12/91 west of Minneapolis/St. Paul, MN (Franz and Eisenreich 1998) were therefore used to compute PCB homologue loads. For PCB loads by wet deposition, the 12/12/91 rainfall data were multiplied by the water surface area and an average annual rainfall for the New York-New Jersey region of 42.7 inches/yr (Thomann et al. 1989). For PCB loadings associated with back diffusion from the atmosphere, the 12/11-12/91 air data were first divided by a dimensionless Henry's law constant (which was taken as 4×10^{-3} based on congener-specific values reported in Dunnivant et al (Dunnivant et al. 1992) and represents a low estimate for the Henry's law constant and hence a high estimate for PCB loading). These values were then multiplied by the water surface area and a PCB interfacial gas exchange rate coefficient of 0.5 m/day. Tabulated results for atmospheric loadings are given in Tables A-14 through A-16 in the appendix. Based on these calculations, direct atmospheric loadings from wet deposition and back diffusion from atmospheric gases accounts for 1.8 lbs/day. Only 4% of this loading was directly applied to the Hudson River and New York-New Jersey Harbor. The remaining 96% of the atmospheric load was applied to the New York Bight and Long Island Sound, primarily as wet deposition.

A summary of total PCB loads for the 1987-97 time period is given in Figure 2-9. As shown, total PCB loading to the estuary and harbor region were dominated by the Upper Hudson during the early 1990s. Upper Hudson loads however have continued to decrease from the early 1990s and for 1997 are estimated to be slightly less than half the total PCB load. A closer look at PCB loads by homologue is given in Figure 2-10 and shows that increased loads from the Upper Hudson during the early 1990s is largely attributed to lower chlorinated homologues (i.e., di- through tetra-CBs). Increased loads of the higher chlorinated homologues (i.e., penta- and hexa-CBs) from the Upper Hudson are also noted during the early 1990s. The Mohawk, the New Jersey tributaries, and New York City-New Jersey WWTPs and CSOs contribute a significant portion of the higher chlorinated homologues, particularly during the late 1980s and the mid to late 1990s.

PCB Initial Conditions in Sediments

PCB initial conditions in sediments for model simulations beginning in 1987 were assigned based on measured concentrations in dated sediment cores. For this determination, cesium-137 sediment profiles were examined in the 1992 high-resolution cores (TAMS/Gradient 1995) and used to determine average deposition rates for each core (by assigning peak cesium-137 concentrations to sediments deposited in the early 1960s). The depth horizon for 1987 deposited sediments were then determined (by multiplying the deposition rate by a five year period). Since the 1992 high-resolution cores were collected in sediment deposition zones and tended to include higher organic carbon sediments, PCB homologue data for the 1987 horizon was organic carbon normalized. Organic carbon normalized values were then multiplied by an average sediment organic carbon content value of 2.4% to provide a segment-wide average concentration for 1987

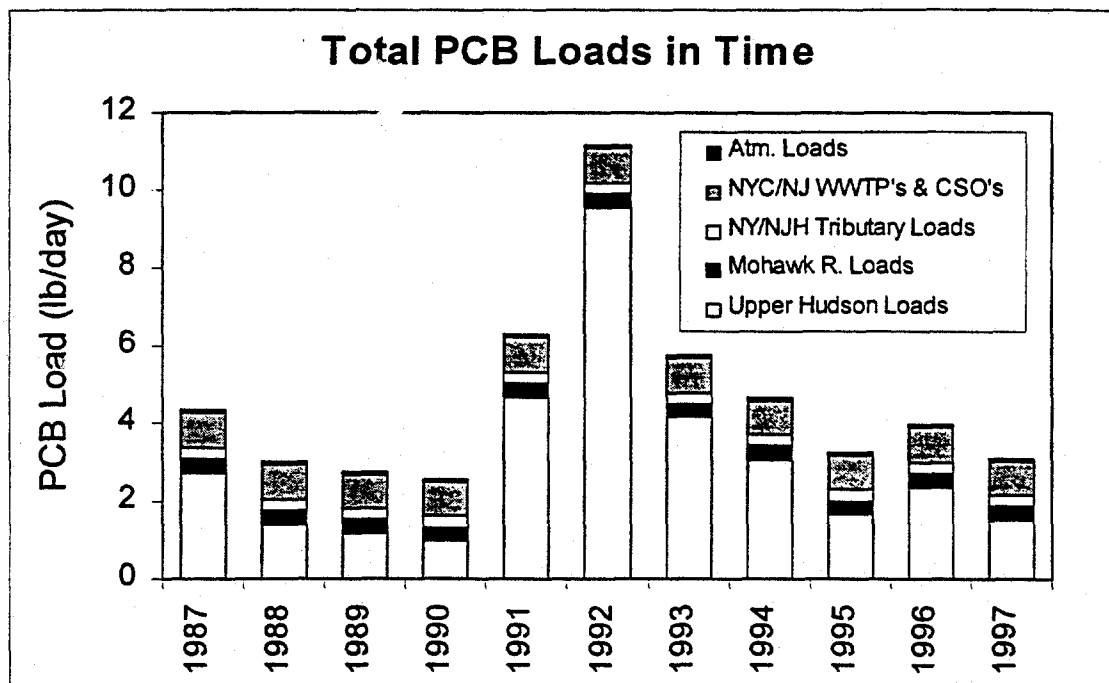


Figure 2-9. Summary of total PCB loads for the 1987-1997 time period.

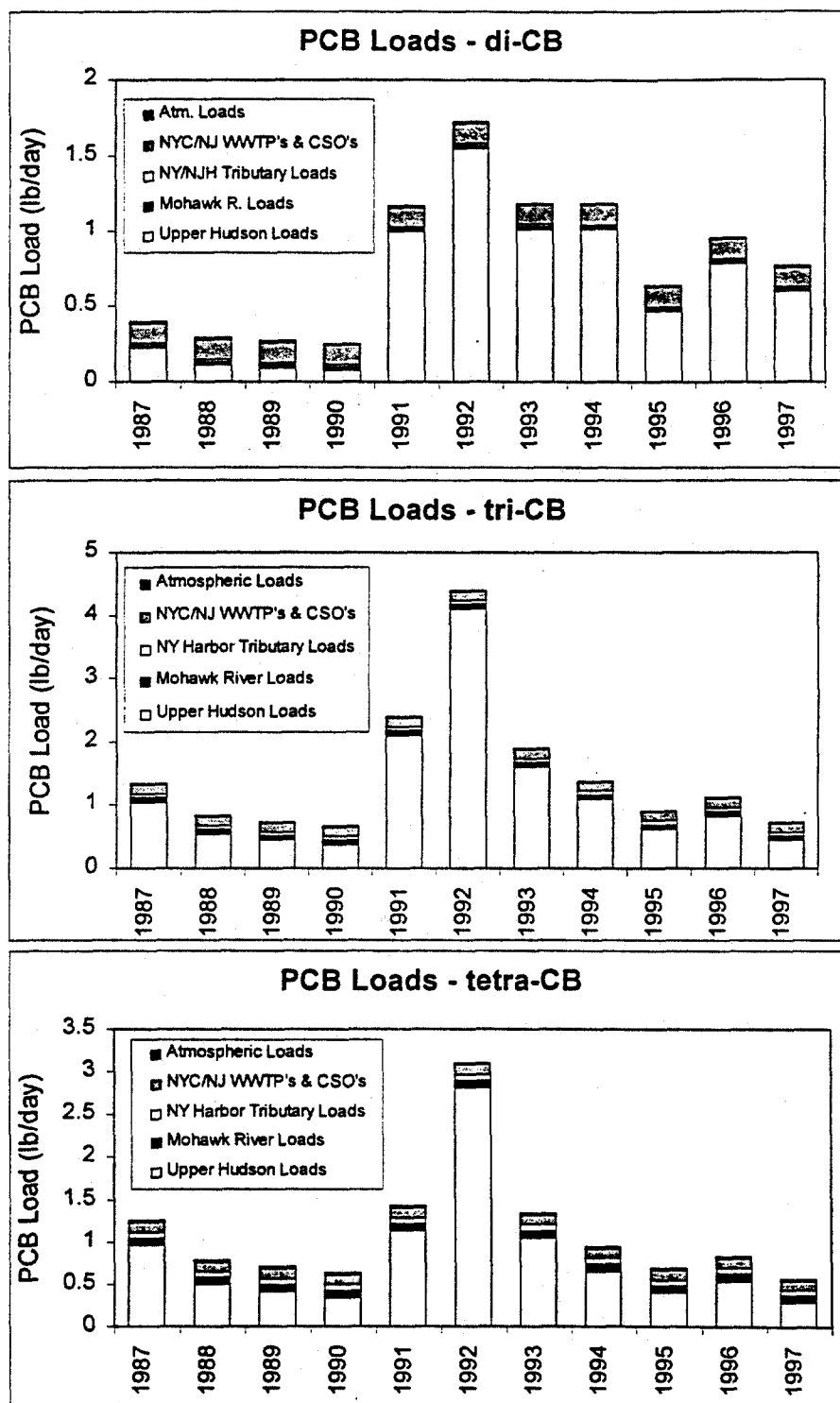


Figure 2-10. Summary of PCB homologue loads for the 1987-97 time period.

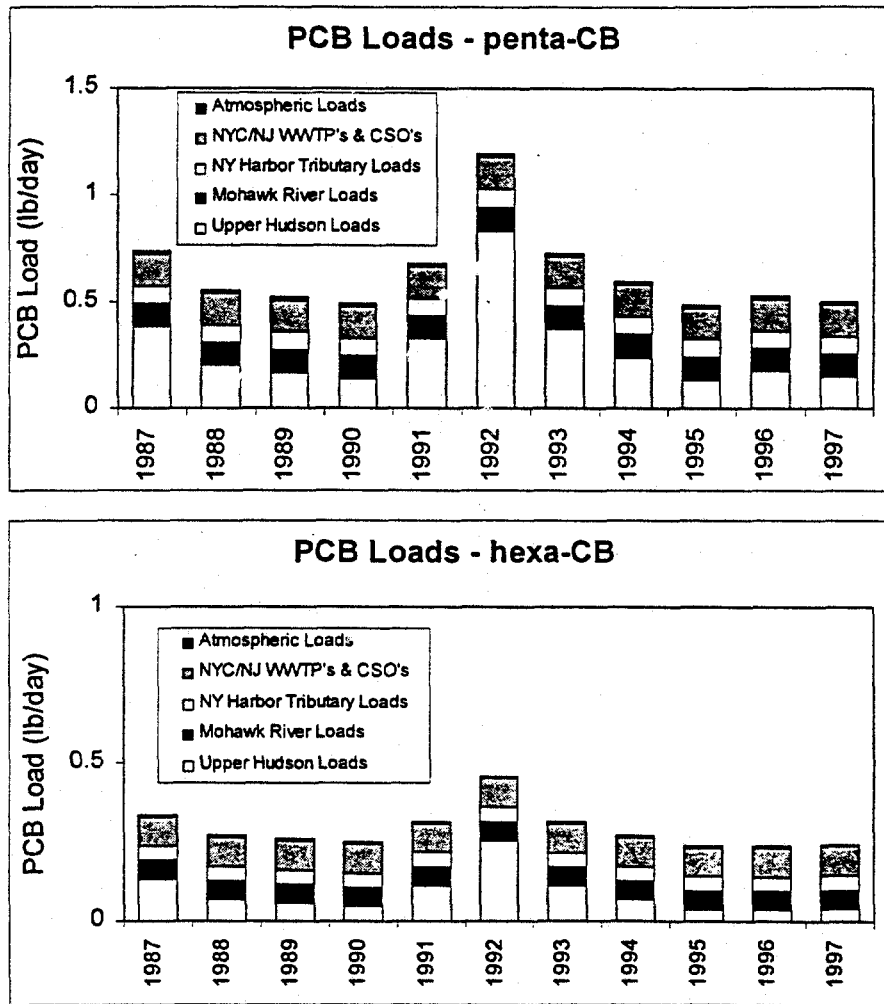


Figure 2-10 (continued).

initial conditions. For model segments missing sediment cores, PCB homologue concentrations were assigned using the closest upstream sediment core. Final values for the 1987 PCB initial conditions in sediments are given for each segment in Table A-17 of the appendix.

Bioaccumulation Modeling

The accumulation of toxic organic chemicals in aquatic organisms has been modeled at several levels of detail. In the simplest formulation, a partition coefficient (i.e., the bioconcentration factor or BCF) is used to define the chemical concentration in the organism relative to its dissolved concentration in water. A more complex formulation is the food chain model (Thomann et al. 1992b)(Thomann et al. 1992a). In this approach, the chemical accumulation within a given organism in the food chain is viewed as a dynamic process that depends on direct uptake from the water, ingestion of contaminated prey, depuration (from urine excretion and egestion of fecal matter), and metabolic transformation of the chemical within the organism.

Model equations for the uptake and release of toxic organic chemicals into a given organism are typically written in terms of μg chemical per g wet weight of organism (v_k) (Thomann et al. 1992b)(Thomann et al. 1992a). The general form of this equation is given as

$$\frac{dv_k}{dt} = k_{u_k} C_{dis} - k_{b_k} v_k + \sum \alpha_{ki} I_{ki} v_i - [k_e + k_m + k_g] v_k \quad (13)$$

where the first term represents the change in chemical concentration in organism 'k' with time; the second term represents the direct uptake of chemical from the water phase by diffusion across an external cell or gill membrane; the third term represents back diffusion of the chemical across the membrane; the fourth term represents chemical uptake through the ingestion of contaminated food or prey and is dependent on the chemical assimilation efficiency (α_{ki}) and the consumption rate (I_{ki}) for organism 'k' feeding on organism 'i'; and the last term represents decreases in chemical concentrations in organism 'k' due to excretion (k_e), metabolic transformation (k_m), and growth dilution (k_g). In this equation, growth dilution is included as a loss term to account for the reduction in chemical concentration in the organism due to the increase in the size of the organism. (A complete listing of terms used in Eq 13 is given in the Appendix B.)

In applying this model to striped bass in the Hudson River Estuary, Thomann et al (Thomann et al. 1989); (Thomann et al. 1991) developed a time-variable, age-dependent striped bass food chain model. The model included a five component, pelagic food chain that consists of phytoplankton, zooplankton, small fish, seven age classes of perch, and seventeen age classes of striped bass (Figure 2-11). As shown in the figure, phytoplankton serve as the base of the food chain. Phytoplankton are preyed upon by a

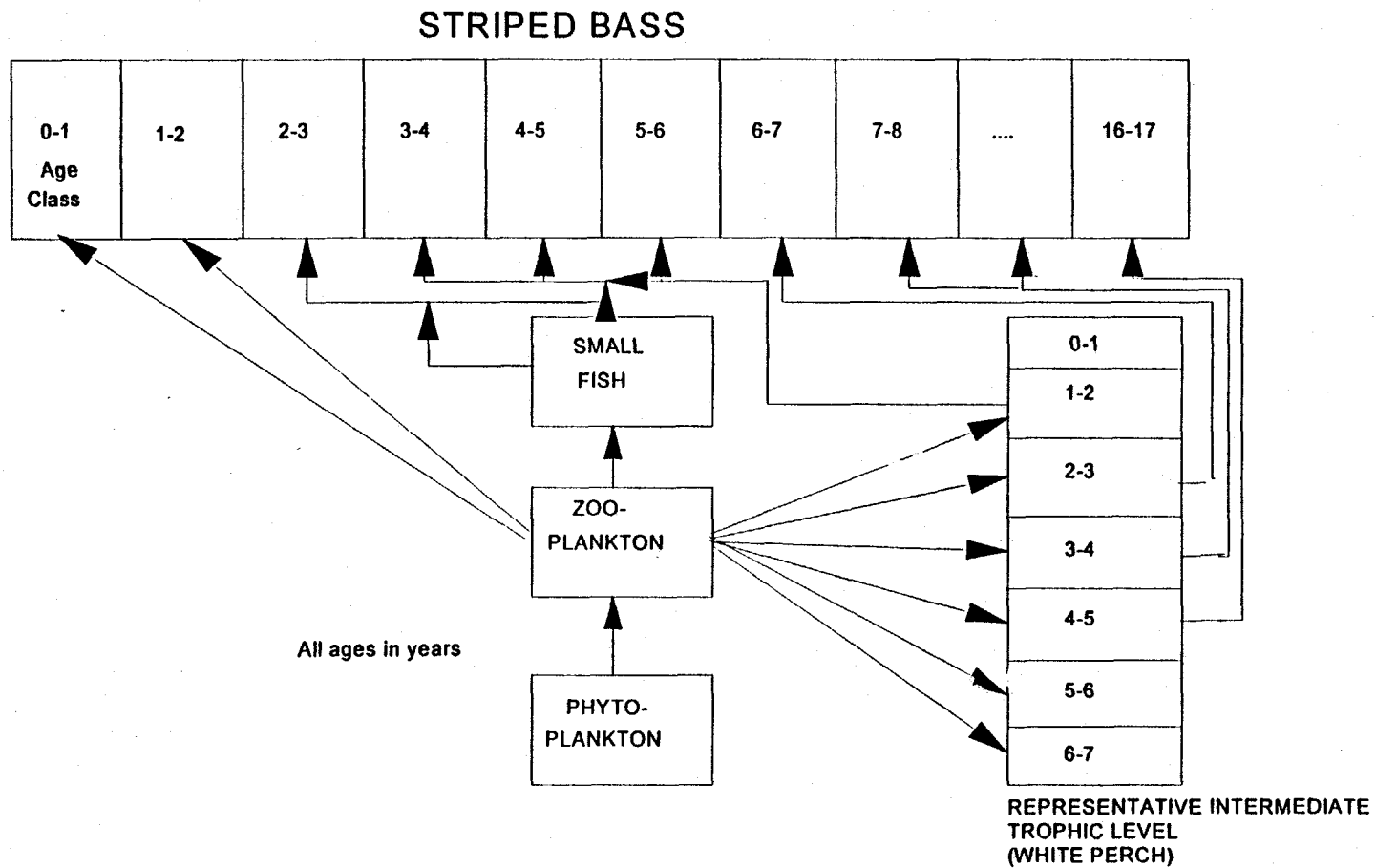


Figure 2-11. Schematic diagram of food web interactions in the PCB striped bass bioaccumulation model.

zooplankton compartment, the characteristics of which is considered to be represented by *Gammarus*. The small fish compartment, which feeds on zooplankton, is meant to reflect a mixed diet of fish of about 10 g in weight and includes 0-1+ tomcod and herring. Perch is considered as a representative size-dependent prey of the striped bass and is assumed to feed exclusively on zooplankton. Based on feeding studies where stomach contents of striped bass were examined (Gardinier and Hoff 1982; O'Connor 1984; Setzler et al. 1980), the 0-2 year old striped bass are assumed to feed on zooplankton; 2-5 year old striped bass are assumed to feed on a mixture of small fish and 0-2 year old perch; and 6-17 year old striped bass are assumed to feed on 2-5 year old perch.

In the original PCB homologue model (Thomann et al. 1989), PCB homologue concentrations in the phytoplankton are considered to be in equilibrium with freely-dissolved water concentrations and are described using a constant bioconcentration factor (BCF) of 30 L/g(wet), based on the phytoplankton BCF data of Oliver and Niimi (Oliver and Niimi 1988) (Figure 2-12). For higher trophic organisms, PCB homologue concentrations are computed using Eq 13 and the following parameter specifications.

Growth rates were determined from results of Poje et al (Poje et al. 1988) for zooplankton; from a generalized growth-weight relationship for small fish (Thomann et al. 1989); from the age-weight data of Bath and O'Connor (Bath and O'Connor 1982) for perch; and from the age-weight data of Setzler et al (Setzler et al. 1980) and Young (Young 1988) for striped bass. Details of age-dependent weights and growth rates are given in Thomann et al (Thomann et al. 1989) and are summarized in Table A-18 in the appendix.

Respiration rates for zooplankton, small fish, perch, and striped bass were estimated using formulations given in Thomann and Connolly (Thomann and Connolly 1984) and Connolly and Tonelli (Connolly and Tonelli 1985). Details of respiration rates, along with lipid content, dry weight fractions, and food assimilation efficiency, are given in Thomann et al (Thomann et al. 1989) and are summarized in Table A-19 in the appendix. These values are used with the PCB homologue-specific parameters for K_{ow} , gill transfer efficiency (β), and chemical assimilation efficiency from food (α) to calculate gill uptake rates (k_u), back diffusion rates (k_b), and food ingestion rates (I). K_{ow} , α , and β values for di- through hexa-CB are given in Table 2-2.

Gill uptake rate coefficients (k_u) were determined as a function of the respiration rate in oxygen equivalents (R_{O_2}), the gill transfer efficiency (β), and the dissolved oxygen concentration (C_{O_2}) as:

$$k_u = \beta \frac{R_{O_2}}{C_{O_2}} \quad (14)$$

where C_{O_2} is taken as 70% of oxygen saturation (which is a function of temperature and salinity according to Eqs 6.15 and 6.16 in Thomann and Mueller (Thomann and Mueller

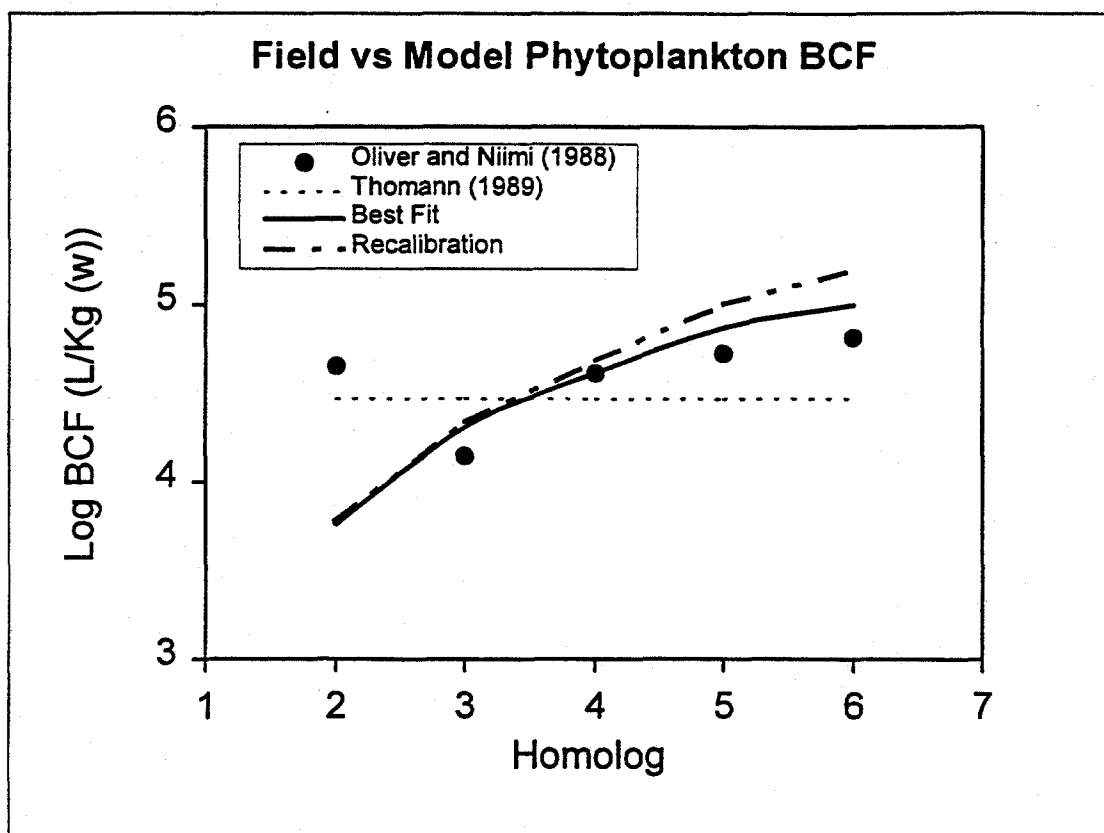


Figure 2-12. Phytoplankton BCF as a function of PCB homologue. [Data from Oliver and Niimi (1988).]

Table 2-2. Homologue Specific Octanol-Water Partition Coefficients, Bioconcentration Factors, and Chemical Assimilation Efficiencies for the Hudson River Estuary Striped Bass Food Chain Bioaccumulation Model ^a

Homologue	log K _{ow} L/kg	Phyto- plankton BCF L/kg(lipid)	Gill Transfer Efficiency (β)	Chemical Assimilation Efficiency (α)	BCF L/g(wet)		
					Zooplankton and small fish $f_{lipid} = 0.06$	White perch $f_{lipid} = 0.04$ ($f_{lipid} = 0.06$)	Striped Bass $f_{lipid} = 0.07$ ($f_{lipid} = 0.06$)
di-CB	4.9 (5.0)	4.48 (3.77)	0.5 (0.4)	0.71 (0.53)	4.8 (6.0)	3.2 (6.0)	5.6 (6.0)
tri-CB	5.4 (5.6)	4.48 (4.34)	0.5 (0.4)	0.80 (0.53)	15.1 (23.9)	10.0 (23.9)	17.6 (23.9)
tetra-CB	5.8 (6.0)	4.48 (4.68)	0.45 (0.4)	0.80 (0.53)	37.9 (60.0)	25.2 (60.0)	44.2 (60.0)
penta-CB	6.2 (6.45)	4.48 (5.0)	0.45 (0.4)	0.63 (0.53)	95.1 (169.1)	63.4 (169.1)	110.9 (169.1)
hexa-CB	6.7 (6.85)	4.48 (5.2)	0.33 (0.4)	0.35 (0.53)	300.7 (424.8)	200.5 (424.8)	350.4 (424.8)

^a Values without parentheses represent values used in the original PCB homologue model (Thomann et al. 1989). Numbers in parentheses represent final values used in the recalibrated PCB homologue model.

1987)). Back diffusion rate coefficients (k_b) were determined as a function of the gill uptake rate coefficient (k_u), the lipid content of the organism (f_{lipid}), and K_{ow} as:

$$k_b = \frac{k_u}{f_{lipid} K_{ow}} \quad (15)$$

Food ingestion rates (I) were determined as a function of the respiration rate in wet weight equivalents (R), the organism growth rate (k_g), and the food assimilation efficiency (a) as:

$$I = \frac{R + k_g}{a} \quad (16)$$

Lastly, metabolic transformations of PCBs in the striped bass food chain were considered negligible (i.e., $k_m = 0$).

In bioaccumulation model calculations for the Hudson, migration of striped bass adds a further complication in specifying time-dependent exposure concentrations. Migration patterns used in the initial calculations were assigned using Waldman (Waldman 1988a; Waldman 1988b) and are described in Thomann et al (Thomann et al. 1989; Thomann et al. 1991) and summarized in Figure 2-13. As shown, striped bass are born on May 15 of each year and the yearlings are assumed to remain in the mid estuary (as defined by River Mile points 18.5 to 78.5). 2-5 year old striped bass are considered to migrate from the mid estuary into New York/New Jersey Harbor in June and spend the summer months (July through September) in Long Island Sound and the New York Bight. Lastly, 6-17 year old striped bass are assumed to spend most of their year in the open ocean, but migrate into Long Island Sound and the New York Bight around March 15 and return to the mid estuary around April 15 to spawn. They remain in the mid estuary until the middle of July.

Changes in bioaccumulation parameters for the recalibrated PCB homologue model are given as follows: First, the lipid content for perch was adjusted from 0.04 to 0.06 based on 1993 field data (TAMS/Gradient 1995), and the lipid content for striped bass was adjusted from 0.07 to 0.06 based on NYSDEC monitoring data (TAMS/Gradient 1995). The resulting changes in BCF values for perch and striped bass are listed in Table 2-2. Second, for phytoplankton BCFs, PCB homologue concentrations in phytoplankton were again assumed to be in equilibrium with freely-dissolved water concentrations and were considered to be controlled by the diffusive uptake, back diffusion, and growth dilution. With these assumptions, the phytoplankton BCF can be obtained from Eq 13 as:

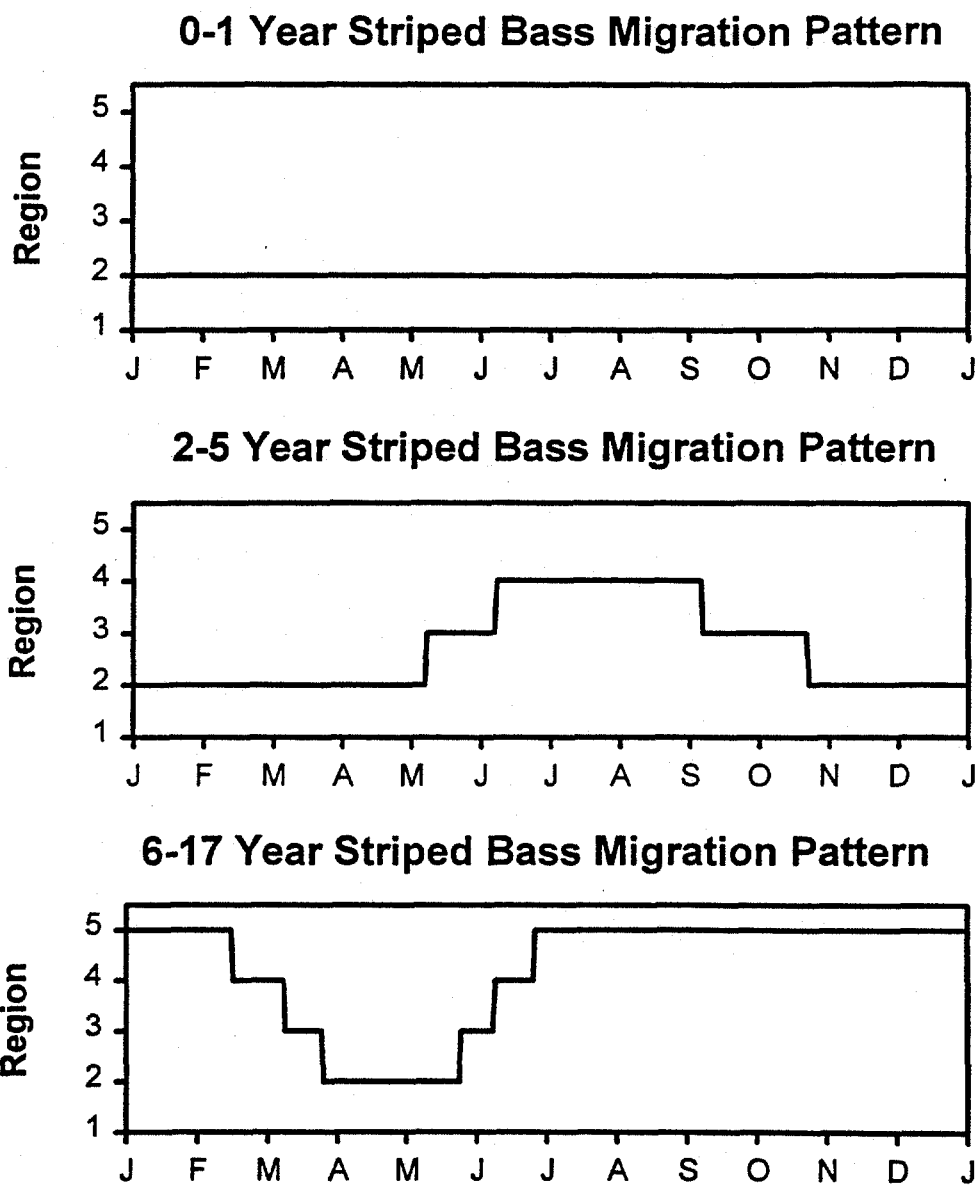


Figure 2-13. Striped bass migration patterns used in the PCB bioaccumulation model.

$$BCF = \frac{k_{u-phyto}}{k_{b-phyto} + k_{g-phyto}} \quad (17)$$

Since $k_b = k_u / (f_{lipid} K_{ow})$, Eq 14 can be expressed as

$$BCF = \frac{k_{u-phyto}}{k_{b-phyto} + k_{g-phyto}} = \frac{f_{lipid} K_{ow}}{1 + \left(\frac{k_{g-phyto}}{k_{u-phyto}} \right) f_{lipid} K_{ow}} \quad (18)$$

where the lipid fraction for phytoplankton (f_{lipid}) is taken as 0.06; K_{ow} values are given in Table 2-2 for each PCB homologue; and the ratio of $k_{g-phyto}/k_{u-phyto}$ was adjusted to a value of 0.007 kg(wet)/L as a "best fit" to the phytoplankton BCF data of Oliver and Niimi (Oliver and Niimi 1988) as shown in Figure 2-12. As discussed in Chapter 3, this ratio was subsequently reduced to 0.004 kg(wet)/L in model calibration to fit PCB homologue data in perch. Phytoplankton BCF values for the final calibration are given in parentheses in Table 2-2. Lastly, the gill transfer efficiency (β) and the chemical assimilation efficiency of PCBs from contaminated food (α) were set equal to constant values across the homologue groups based in part on findings of McKim et al (McKim et al. 1985)(Erickson and McKim 1990) showing that gill transfer efficiency is constant for chemicals with log K_{ow} 's between 5 and 7. As discussed in Chapter 3, final values of β and α were determined by model calibration and are given as 0.4 and 0.53, respectively (see values in parentheses in Table 2-2).

PCB Initial Conditions in Fish

PCB initial conditions in zooplankton, small fish, and perch for model simulations beginning in 1987 were assigned based on previous modeling results of Thomann et al (Thomann et al. 1989). After preliminary model runs, PCB homologue concentrations in perch were adjusted to help eliminate any sharp gradients in PCB perch concentrations during the first year of the model simulation. Final values for the 1987 PCB initial conditions in zooplankton, small fish, and perch are given for each food web region in Table A-20 of the appendix. No information was available on PCB homologue concentrations in striped bass or on the depuration of PCB homologues from striped bass during migration. PCB homologue concentrations in striped bass were therefore initially set to zero. This assignment may cause a lower prediction of PCB concentrations in striped bass, but only for the first few years of the model simulation.

Chapter 3

Fate and Bioaccumulation of PCB Homologues: Model Results

Previous Modeling Results

The PCB homologue fate and bioaccumulation model was originally developed by Thomann et al (Thomann et al. 1989) as part of a previous grant from the Hudson River Foundation. The model was initially run for the Hudson River Estuary using long-term average annual hydrology and sediment transport. Model calibration was performed for the period of 1946 (before PCB contamination) through 1987 using total PCB and Aroclor data for the water column, sediments, and striped bass. Model parameters and details of the calibration are given in Thomann et al (Thomann et al. 1989; Thomann et al. 1991). Model projections for PCB concentrations in striped bass were then made assuming exponentially-decreasing loads from the Upper Hudson and the downstream sources. Results of this work indicated that, under a no-action alternative, 50% of the striped bass would be below the Food and Drug Administration (FDA) limit of 2 µg of total PCB/g of fish (wet weight) by 1992 and 95 percent of the striped bass would be below the FDA limit by 2004.

Post-Audit Evaluation

As an extension of the initial modeling studies, we conducted a post-audit evaluation of PCB homologue model calculations by comparing PCB water column, sediment, and striped bass model results to data that have been collected since the 1987 model calibration. As discussed in Chapter 2, we found that the model provided a reasonable description of PCB striped bass data (see Figure 2-2). The model calculations however overestimated suspended solids concentrations typically observed in the estuary and underestimated PCB concentrations in the water column and in surface sediments (see Figure 2-3).

Recalibrated PCB Homologue Model

In response to these deficiencies, the PCB homologue model was restructured to include seasonal hydrology and sediment transport. Modifications in relationships for PCB partitioning to organic carbon and phytoplankton were also made to correspond to more current formulations, and model parameters and coefficients for flows, sediment transport, organic carbon distributions, PCB loads, volatilization, and bioaccumulation were reviewed as part of the recalibration effort. Full descriptions of hydrodynamic transport, sediment transport, organic carbon distributions, chemical transport and fate calculations, PCB

loads; initial conditions, and bioaccumulation model calculations were previously given in Chapter 2.

For model recalibration and projections, calculations were performed for simulation periods beginning in 1987. Final calibrations were obtained by slight adjustment of the volatilization rate coefficient for the fate model, and by adjustment of the gill transfer efficiency (β), the chemical assimilation efficiency (α), and the phytoplankton uptake rate/growth rate ratio ($k_{u-phyto}/k_{g-phyto}$) for the bioaccumulation model (see Table 3-1 and discussions in Chapter 2 for final chemical parameters). Field data used for recalibration included 1993 PCB homologue water column data, 1992 and 1993 PCB homologue surface sediment data (TAMS/Gradient 1995), 1993 and 1995 PCB homologue perch data (McGroddy et al. 1997; TAMS/Gradient 1995), and 1987 through 1997 total PCB striped bass data (TAMS/Gradient 1995).

Table 3-1. Summary of Chemical/Biochemical Parameters Used in the Recalibrated PCB Homologue Model.	
log K_{ow}	5.0 (di-CB) 5.6 (tri-CB) 6.0 (tetra-CB) 6.45 (penta-CB) 6.85 (hexa-CB)
a_{DOC} (water column)*	0.1
a_{DOC} (sediments)*	1.0
Volatilization rate coefficient (k_v)*	0.5 m/day
gill transfer efficiency (β)*	0.4
Chemical assimilation efficiency (α)*	0.53
Phytoplankton uptake rate/growth rate ratio ($k_{u-phyto}/k_{g-phyto}$)	0.004 kg(wet)/L
* Values are applied to all PCB homologues.	

Recalibration Results for the Water Column and Surface Sediments

Longitudinal distributions of dissolved PCB concentrations in the water column are shown in Figure 3-1 for April and August 1993. Model results for both periods show similar behavior with a slow decrease in dissolved PCB concentrations through the tidal freshwater and mid estuary regions (RM 154 to RM 13.5). These decreases are largely due to dilution from the lower Hudson tributaries and losses from volatilization. (A more complete description of seasonal behavior is given subsequently in the section on PCB fluxes.) In the New York/New Jersey Harbor and in the New York Bight Apex (RM 13.5 to RM -20), dissolved PCB concentrations decrease more rapidly as a result of tidal exchange with the ocean. Seasonal differences in dissolved PCB concentrations are also apparent (Figure 3-1) with higher concentrations in April associated with higher loadings from the upper river during the Spring of 1993.

As shown in Figure 3-1, the August 1993 model results provide a reasonable representation of the August/September 1993 dissolved water column data. A more detailed comparison of PCB homologue data and calculated results for the three sampling locations is given in Figure 3-2. For di-CBs, model results are higher than observed concentrations. This difference is likely a result of using measurements at Thompson Island Dam (TID) to estimate Upper Hudson tributary loading and not accounting for the potential loss of di-CB by volatilization and/or aerobic degradation in the 35 mile stretch between TID and Federal Dam at Troy. For higher chlorinated homologues, model results are lower than observed concentrations--most notably for RM 77. Reduction in the volatilization rate coefficient (below a value of 0.5 m/day) could be made to further improve model fits for the higher chlorinated homologues. This adjustment however was not made because water column concentrations are subject to short-term variability in flow and the actual flows in August 1993 were lower than the average low flow used in the model simulation. For model recalibration, greater emphasis was therefore placed on model fits for sediments and fish, which serve as better integrators of long-term behavior.

Longitudinal distributions of PCB concentrations in the surface sediments are shown in Figure 3-3 for April and August 1993 on both a dry weight and an organic carbon normalized basis. As shown in the top panel, sediment results show a decrease through the tidal freshwater and mid estuary regions (RM 154 to RM 13.5) with concentrations of in excess of 1 $\mu\text{g/g}$ (dry wt) in the Albany area and approximately equal to 0.5 $\mu\text{g/g}$ (dry wt) near RM 20. Further downstream (RM 0), an increase in PCB sediment concentrations occurs, and as discussed subsequently, may reflect PCB and/or organic carbon source loadings from the New York metropolitan area. Differences in the April and August profiles are negligible and are indicative of the slow response of sediments to changing PCB loadings.

August 1993 field data (which were collected randomly using box cores) are also shown in Figure 3-3. Although the field data exhibit a fair amount of scatter, the model results provide a good description of the central tendency of the data. Our attempt to organic carbon normalize the field data did not appear to reduce scatter (lower panel of

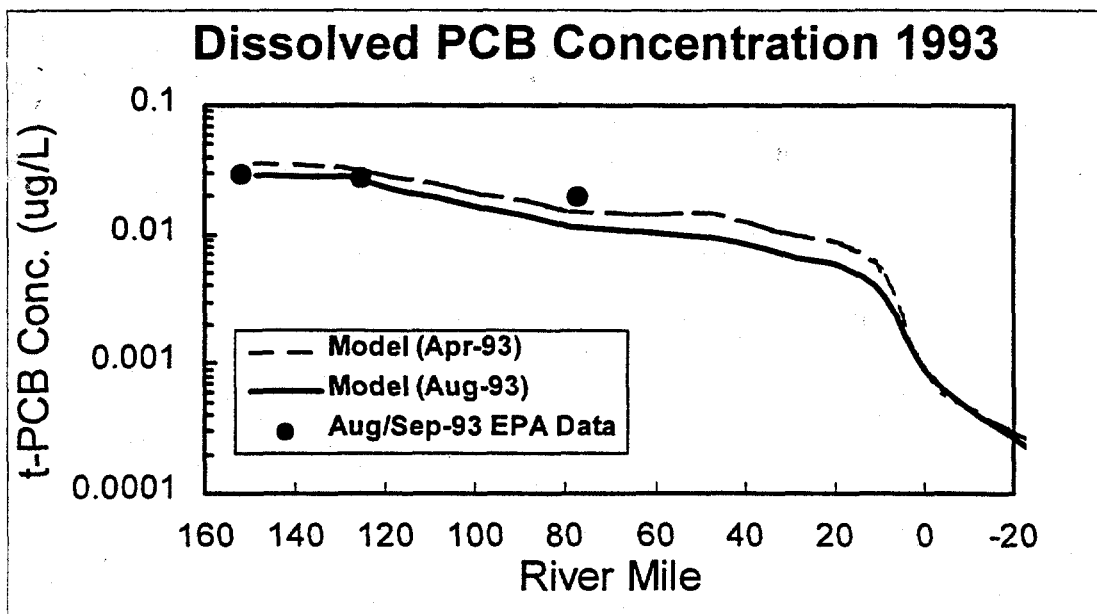


Figure 3-1. Model-field data comparison for 1993 dissolved PCB concentrations in the main stem Hudson. [Data from 1993 EPA Region 2 water column sampling (TAMS/Gradient 1995).]

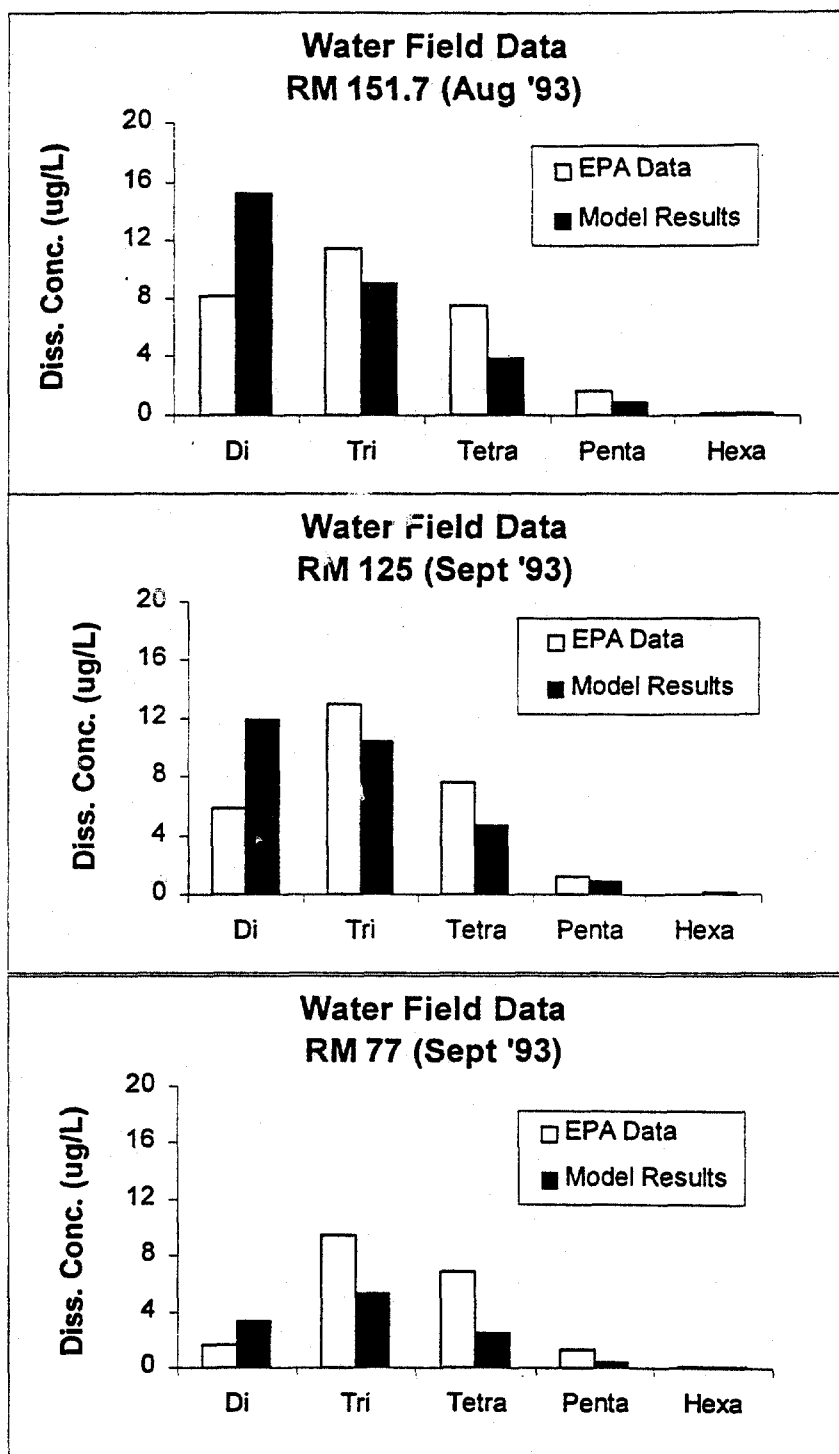


Figure 3-2. Model-field data comparison for 1993 dissolved PCB homologue concentrations. [Data from 1993 EPA Region 2 water column sampling (TAMS/Gradient 1995).]

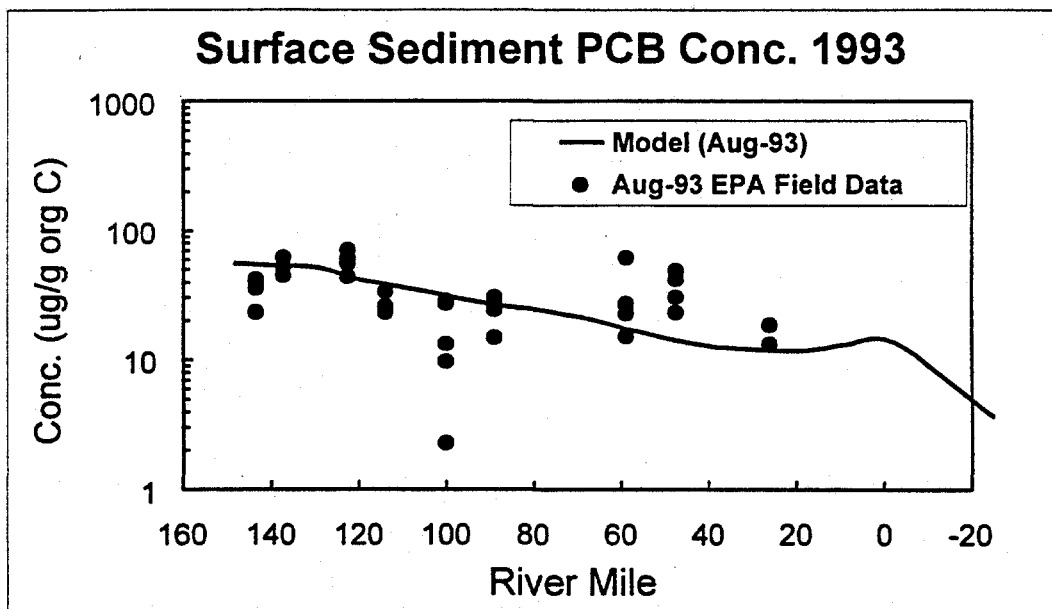
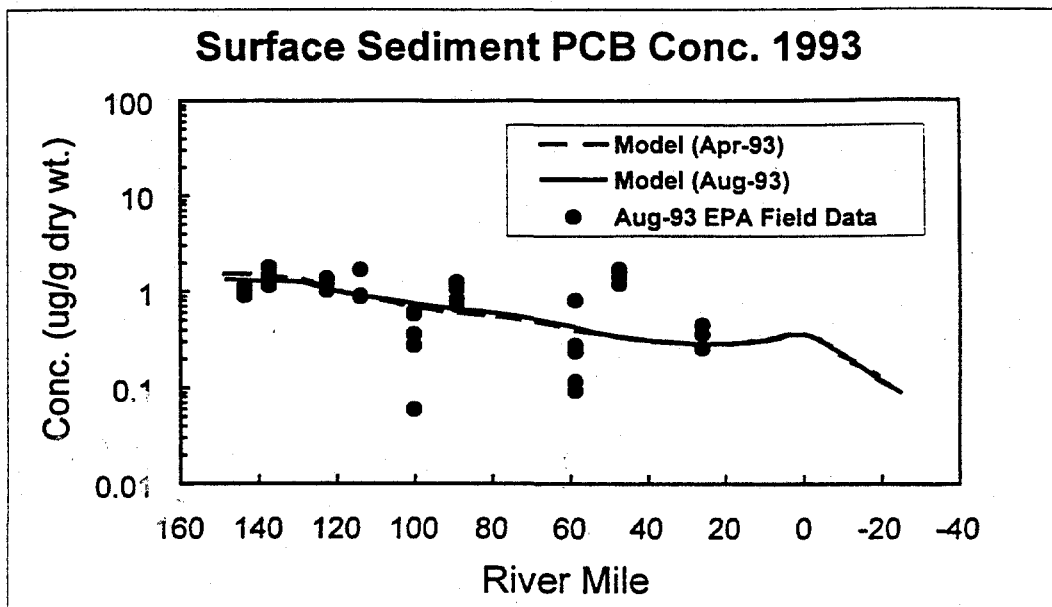


Figure 3-3. Model-field data comparison for 1993 total PCB surface sediment concentrations. [Data from 1993 EPA Region 2 Eco-sediment samples(TAMS/Gradient 1995).]

Figure 3-3). This suggests that variations in the 1993 PCB sediment data are not simply related to organic carbon content but rather are highly dependent on sediment deposition patterns in the river bed (i.e., high deposition zones with recently deposited PCBs versus low deposition zones with aged PCB deposits).

Final comparison of model results to sediment field data was therefore performed using the August 1992 field data. For these data, sediment cores were collected in soft muds from expected deposition zones and core tops were analyzed for beryllium-7 (a short-lived radioisotope) to confirm that sediments were recently deposited. A plot of PCB concentrations in surface sediments for the August 1992 field data is shown in Figure 3-4 along with model results. The field data, which have been organic carbon normalized to account for the higher organic carbon content of the soft mud sediment field samples, are in good agreement with model results for total PCBs, with the possible exceptions for samples from RM 88.9, RM 58.7, RM 2.4.

A more detailed comparison of PCB homologue data and calculated results for August 1992 is given in Figure 3-5 for ten coring locations. Overall, model results provide a good description of the homologue data, particularly for the tetra-, penta-, and hexa-CBs. Two notable exceptions for the higher chlorinated homologues are as follows:

1. The sediment sample for RM 25.8 was obtained from a sediment core taken in Piermont Marsh. Because of exchange fluxes between the main portion of the river and marsh area are likely to be controlled by dissolved (and not particulate) transport, the observed PCB homologue distribution in the organic-rich marsh sediments is skewed toward the lower chlorinated homologues.
2. Concentrations of PCB homologues in the Newtown Creek sediment sample are approximately two times greater than model results. Since the model results are representative of the entire East River segment (from the Battery to Throgs Neck), it is not surprising the sediments in the proximity of the Newtown Creek Wastewater Treatment Plant would exhibit higher concentrations.

For the lower chlorinated homologues, particularly for tri-CB, the following should be noted:

1. Concentrations of tri-CB in sediment samples from RM 88.9 and RM 58.7, which are enriched with BZ#28 and BZ#31, are two and a half times higher than model results. Possible explanations could include the presence of BZ#28 and BZ#31 as possible products of partial dechlorination, or analytical measurement error in quantifying BZ#28 and BZ#31 concentrations (e.g., matrix interference).
2. Concentrations of tri-CB in the RM 2.4 sediment sample are exceptionally high and may indicate an analytical measurement error (e.g., matrix interference).

A summary of August 1992 sediment results is presented in the top panel of Figure 3-6 as a direct comparison of calculated and observed concentrations. As shown, the

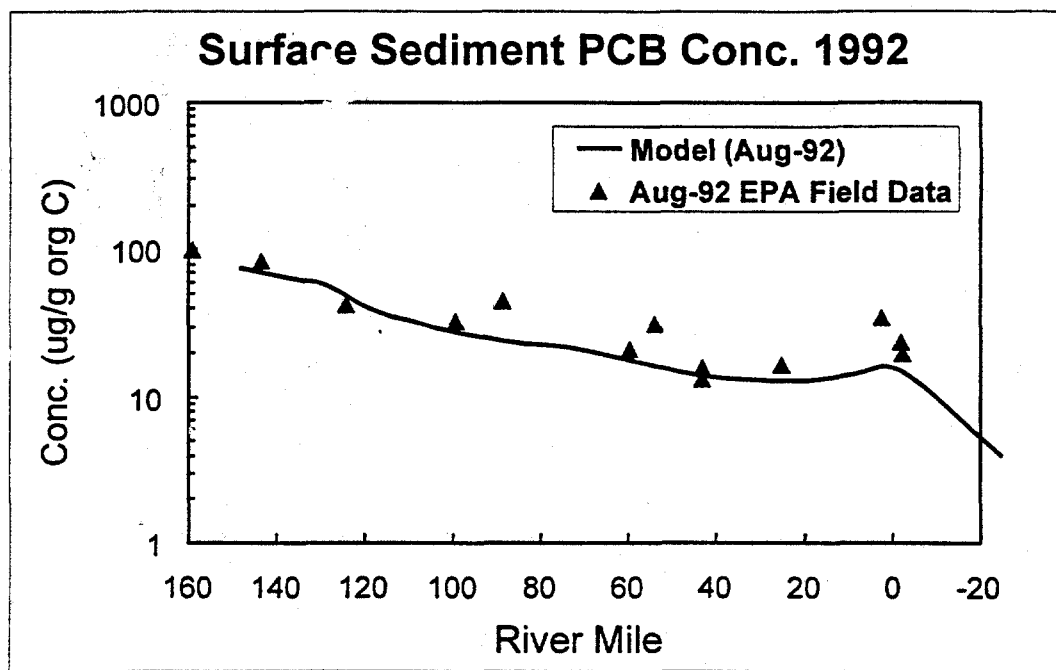


Figure 3-4. Model-field data comparison for 1992 total PCB surface sediment concentrations. [Data from EPA Region 2 high-resolution core samples (TAMS/Gradient 19950.)]

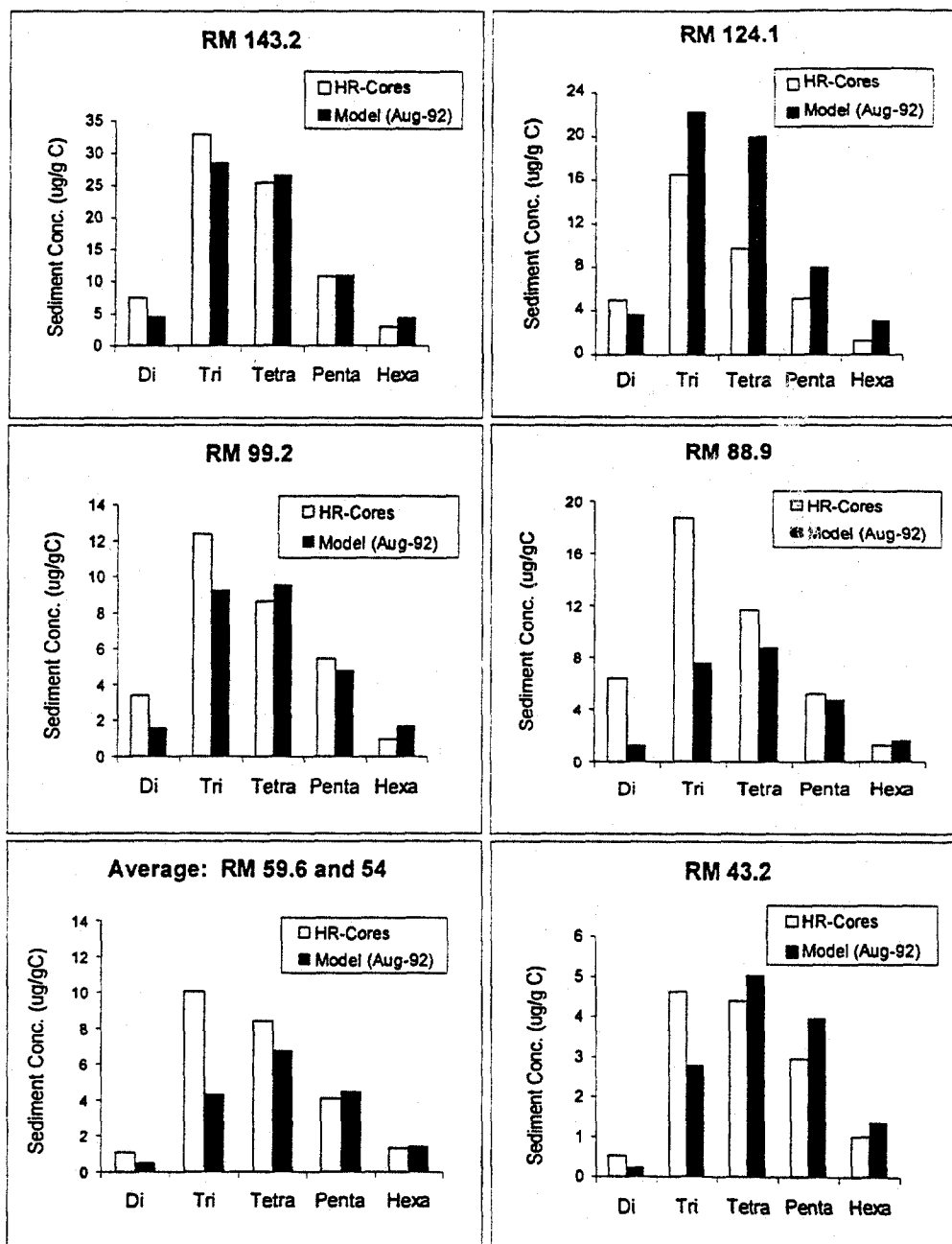


Figure 3-5. Model-field data comparison for 1992PCB homologue concentrations in surface sediments. [Data from EPA Region 2 high-resolution sediment core samples (TAMS/Gradient 1995).]

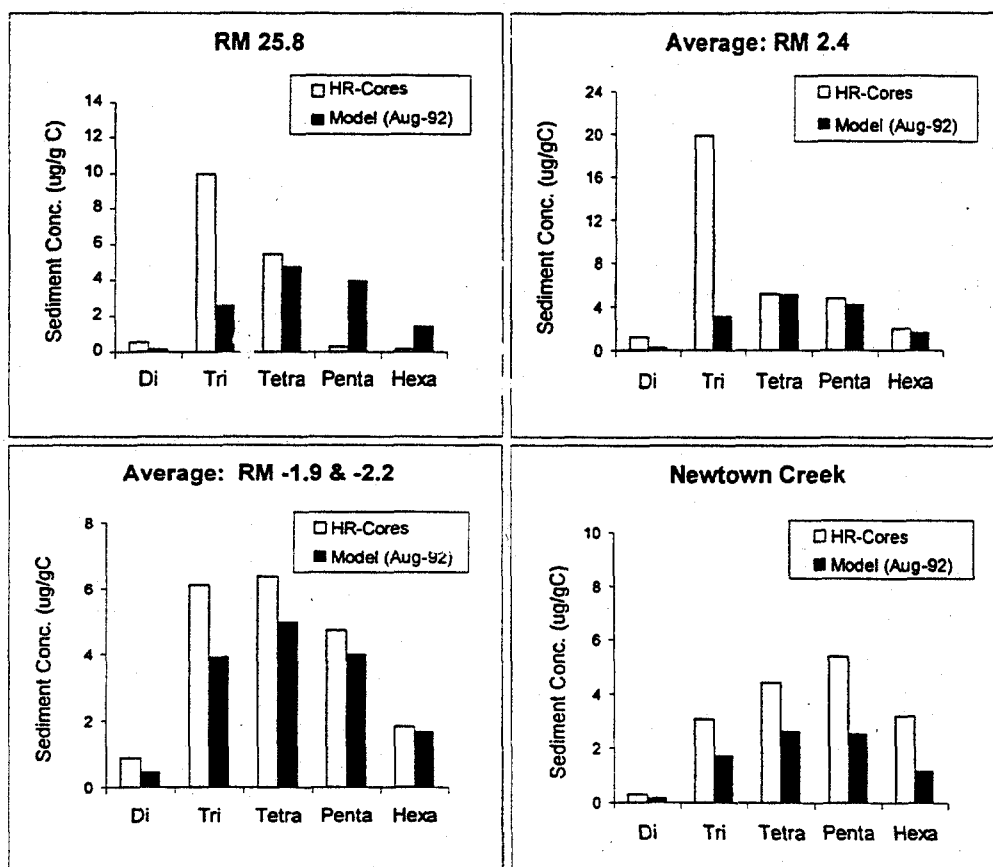


Figure 3-5 (continued). Model-field data comparison for 1992PCB homologue concentrations in surface sediments. [Data from EPA Region 2 high-resolution sediment core samples (TAMS/Gradient 1995).]

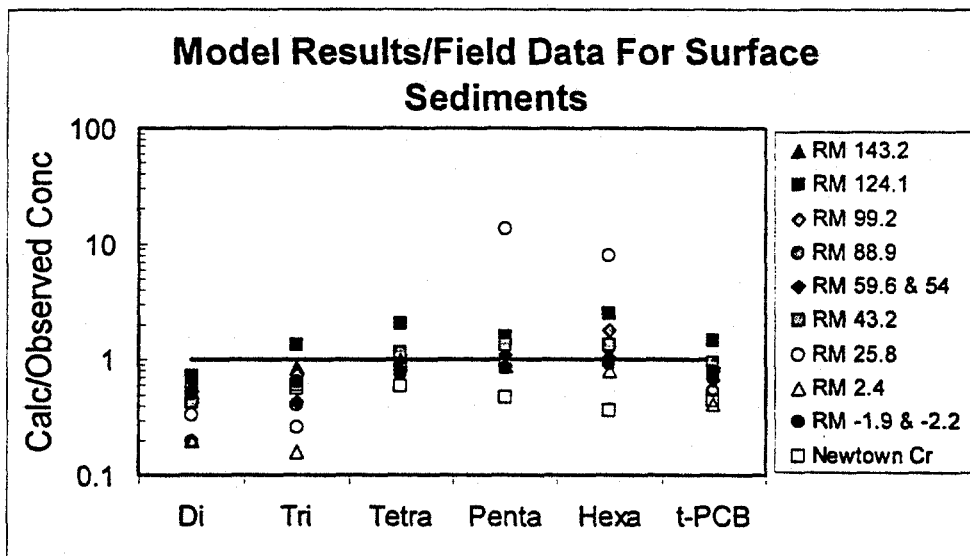
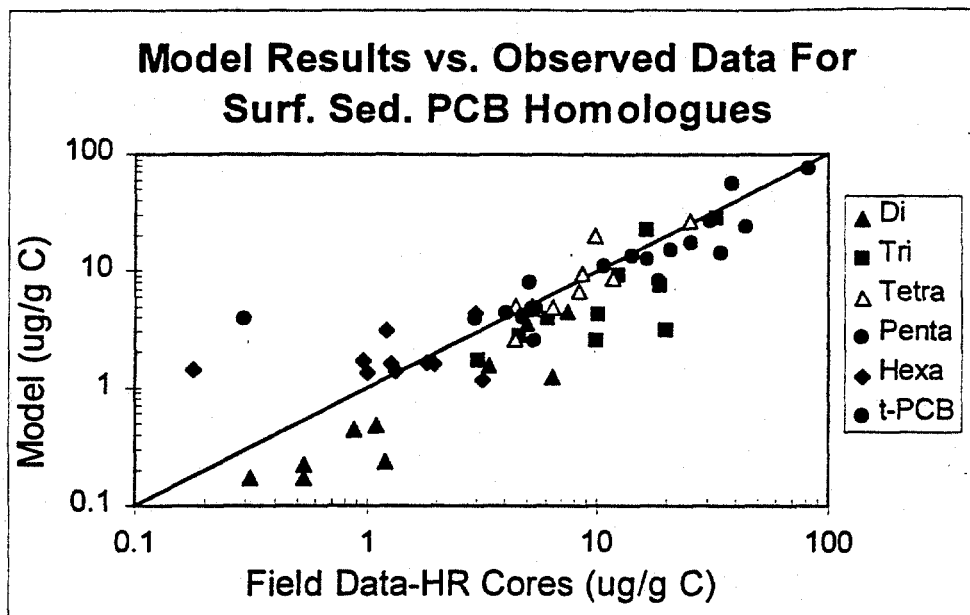


Figure 3-6. Summary of 1992 model-field data comparisons.

correspondence of calculated and observed concentrations for total PCBs and for individual homologues is quite good. Calculated/observed concentration ratios for each homologue are shown in the bottom panel of Figure 3-6 (where results for RM 25.8 (Piermont Marsh), RM 2.4, and Newtown Creek are expected to be outliers and denoted by open symbols). For the lower chlorinated homologues, model calculations tend to be somewhat biased on the low side as shown in the bottom panel of Figure 3-6 with calculated/observed ratios of approximately 0.6 for di-CB and approximately 0.7 for tri-CB. For the higher chlorinated homologues, model calculations provide a good description for tetra-CB, penta-CB, and hexa-CB, all with calculated/observed ratios of approximately one. These results are particularly encouraging since the higher chlorinated homologues are expected to preferentially accumulate in striped bass.

Time History/Projections for PCB Water Column and Sediment Concentrations

Time history/projections for dissolved PCB concentrations in the water column are given in Figure 3-7 for a model simulation period of January 1987 to December 2001. For model segment #3 (RM 128.5), dissolved PCB concentrations decreased during the period of exponentially decreasing loads in the late 1980s and early 1990s. A large peak in dissolved PCB concentrations occurred in September/October 1991, corresponding to the flooding and scouring of highly contaminated PCB sediments from the Allen Mills water intake structure at RM 197 on the Upper Hudson (Rhea et al. 1997). Subsequent peaks in dissolved PCB concentrations occurred during 1992 and are consistent with elevated PCB loadings from the Upper Hudson (previously given in Figure 2-7) that are believed to be associated with seepage of PCB oil from General Electric's Hudson Falls plant site. After 1992, PCB concentrations are shown to slowly decrease with a seasonal response associated with higher PCB loadings from the Upper Hudson during the Spring high flow period (see Figure 2-7).

Similar behavior for dissolved PCBs is shown for model segment #7 (RM 88.5). At this location, however, the peaks in PCB concentrations are highly attenuated. This is primarily due to the continuous interaction of the overlying water with sediments (through settling and tidal resuspension) and the large capacity of the sediments to sorb PCBs and dampen downstream response. This dampening effect is further highlighted downstream in model segment #11 (RM 48.5) where the effects of increased loads are no longer obvious. A seasonal response in model segment #11 however is noted as shown by increased dissolved PCB concentrations during the Spring high flow period.

Time histories/projections for PCB concentrations in surface sediments below model segments #3 (RM 128.5), #7 (RM 88.5), and #11 (RM 48.5) are shown in Figure 3-8.

Dissolved PCB Concentrations

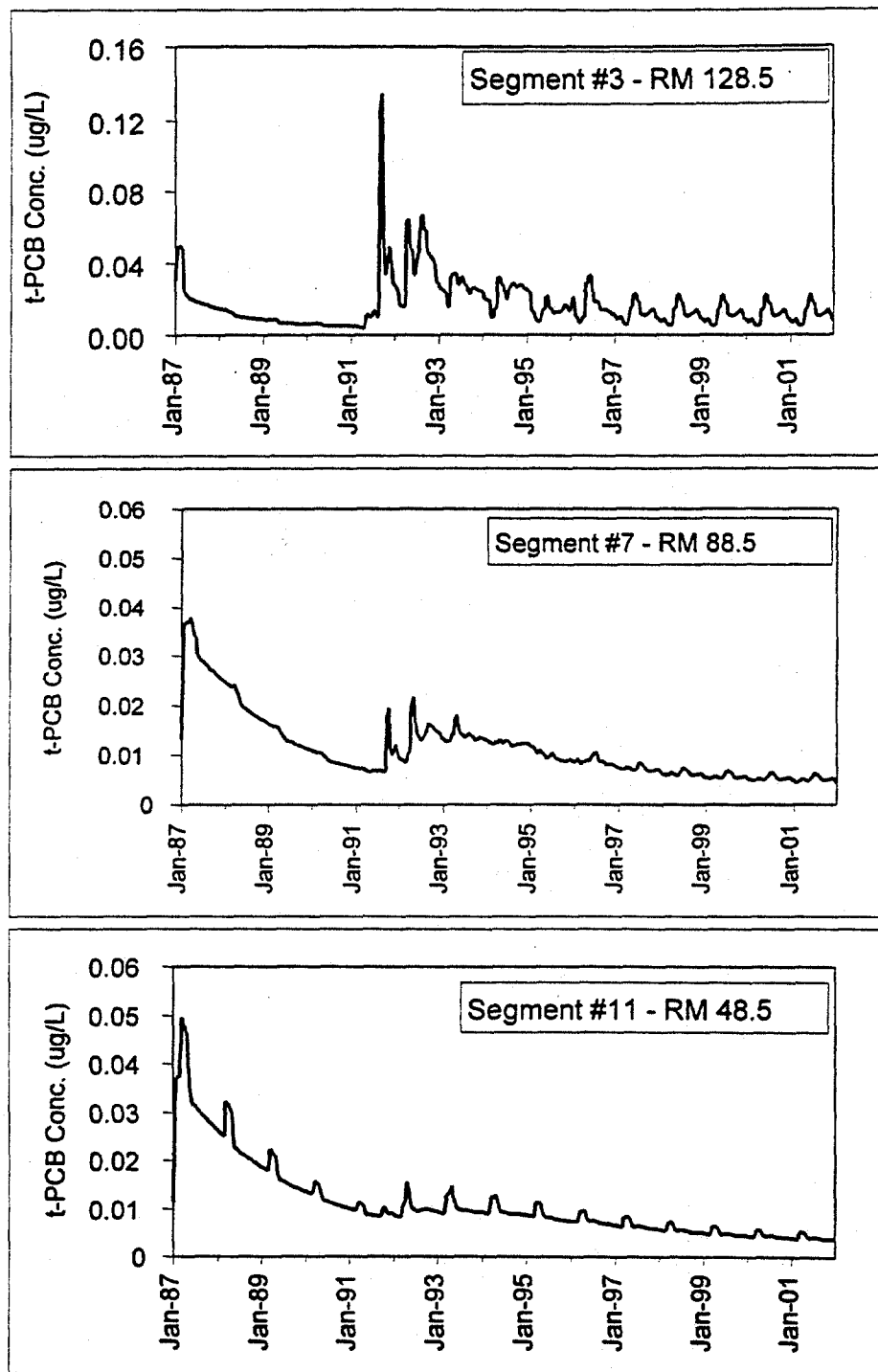


Figure 3-7. Time histories/projections for dissolved PCB concentrations in the water column for RM 128.5, 88.5, and 48.5.

Surface Sediment PCB Concentrations

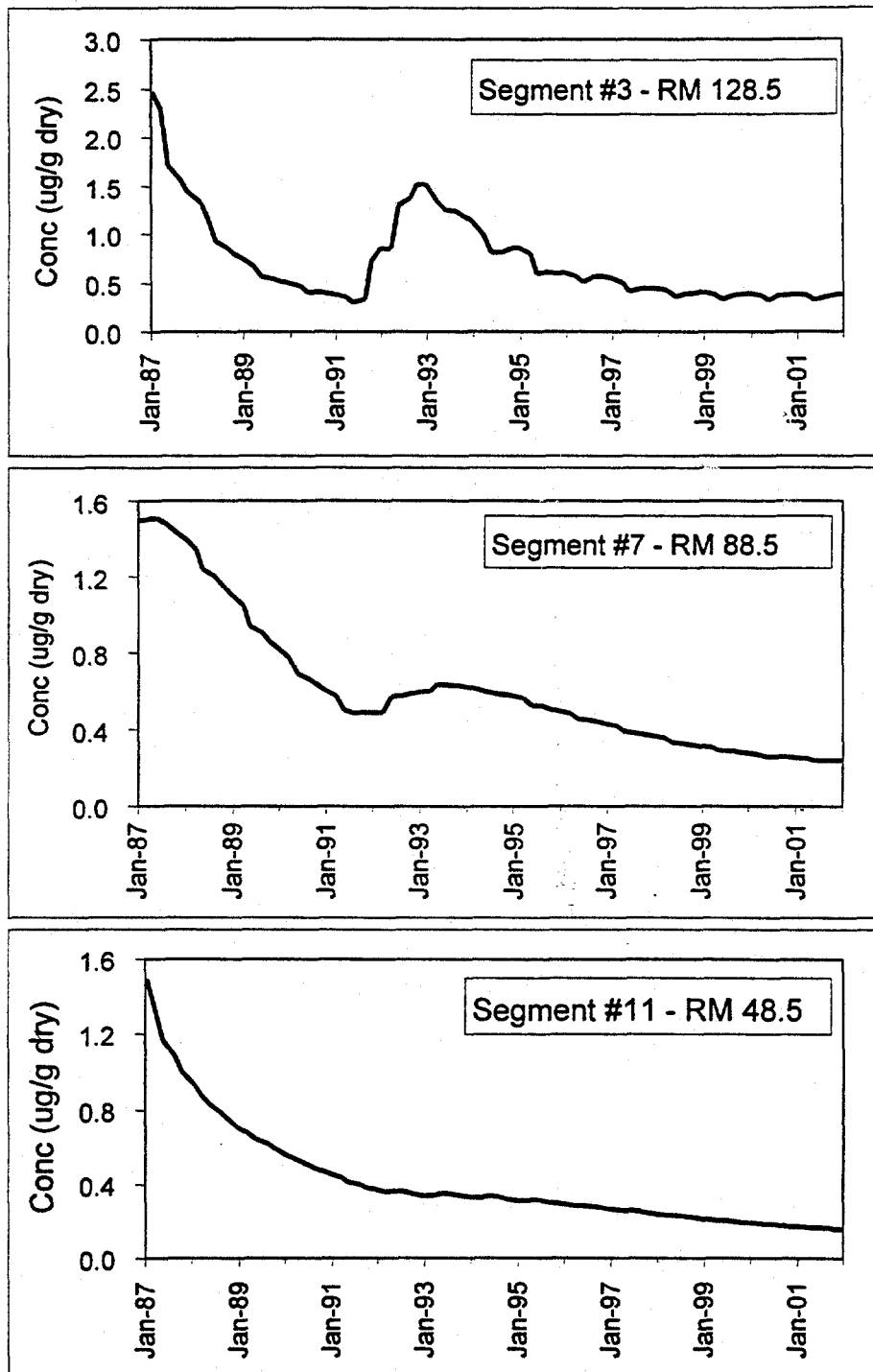


Figure 3-8. Time histories/projections for PCB concentrations in surface sediments for RM 128.5, 88.5, and 48.5.

As expected, PCB responses in the sediments show slower rates of change due to the large capacity of the sediments to sorb PCBs. Seasonal variations in the sediments are therefore not noticeable. Sediment responses to increased PCB loadings from the Upper Hudson during 1991-92 varied with distance downstream. For sediments underlying model segment #3 (RM 128.5), an increase in PCB concentrations is readily apparent with peak PCB concentrations of 1.5 µg/g (dry wt) occurring in December 1992. For model segment #7 (RM 88.5), the peak PCB concentration in the underlying sediments is highly attenuated and occurs in May 1993 (six months after the peak at RM 128.5). Further downstream in model segment #11 (RM 48.5), the effects of the 1991-92 PCB loadings from the Upper Hudson are not discernible.

PCB Fluxes

PCB flux balance diagrams were constructed to obtain a better understanding of the transport and fate of PCBs in the estuary on a total mass basis. Annual and seasonal balances for 1992 and 1997 are shown in Figures 3-9 through 3-12 for the Hudson River Estuary from Federal Dam at Troy (RM 153.5) to approximately the New York City line (RM 13.5). For 1992 (Figure 3-9), which was a period of high PCB loading, approximately 3500 pounds of PCBs entered the estuary from the Upper Hudson and Mohawk Rivers. (In comparison, atmospheric PCB inputs that entered the estuary from direct precipitation on the water surface and back diffusion from the atmosphere were negligible.) Of this 3500 pounds from tributary loading, 45% of the PCBs was lost from the estuary by volatilization, 30% was accumulated in the surface sediments, 5% was accumulated in buried sediments, and 20% was transported down to New York/New Jersey Harbor by advection and tidal dispersion.

On a seasonal basis (Figure 3-10), about half of the 1992 PCB load entered the estuary in the two month Spring high flow period and half of the PCB load entered during the ten month Summer-Fall-Winter low flow period. A similar breakdown is shown for PCBs transported down to New York/New Jersey Harbor. During the Spring high flow period, PCB loss by volatilization (16%) is minimized and more of the incoming PCB load is accumulated in surface (51%) and deep (6%) sediments. This is the result of two factors: (1) the hydraulic residence time in the estuary is shorter during the high flow period; and (2) the higher suspended solids concentrations that accompany the Spring high flow limit the volatilization flux (since a smaller fraction of water column PCBs is in the dissolved phase) and enhance the settling flux (since a larger fraction of PCBs is in the particulate phase). For the ten month summer-fall-winter low flow period, the hydraulic residence time in the estuary is much longer and a larger fraction of PCBs is in the dissolved phase due to low suspended solids concentrations. During this period, a large loss of PCBs by volatilization (73%) occurs and smaller percentages of PCB accumulate in surface (10%) and buried (3%) sediments.

For 1997 (Figure 3-11), which is during the period of decreasing PCB loads, approximately 680 pounds of PCBs entered the estuary from the Upper Hudson and

PCB Mass Balance 1992 (lbs)

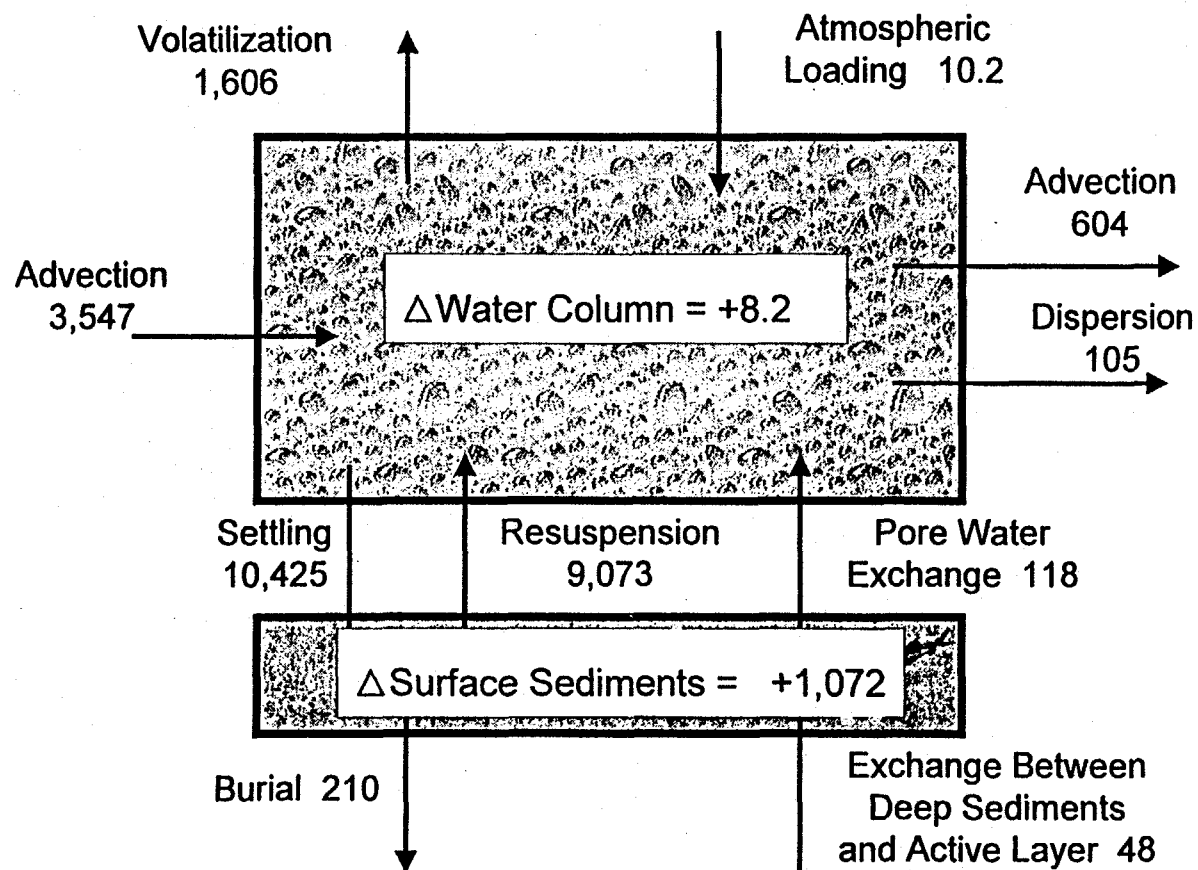


Figure 3-9. Annual PCB flux balance for 1992.

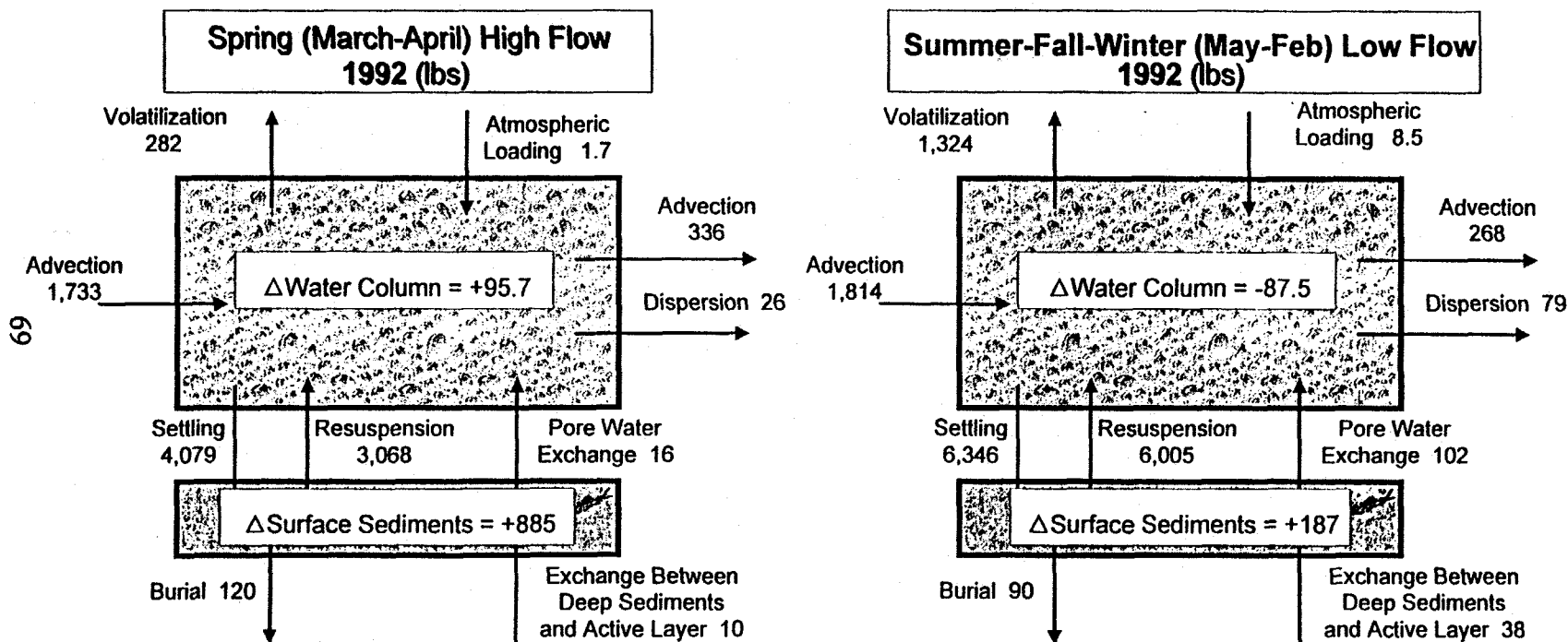


Figure 3-10. Seasonal PCB flux balance for 1992.

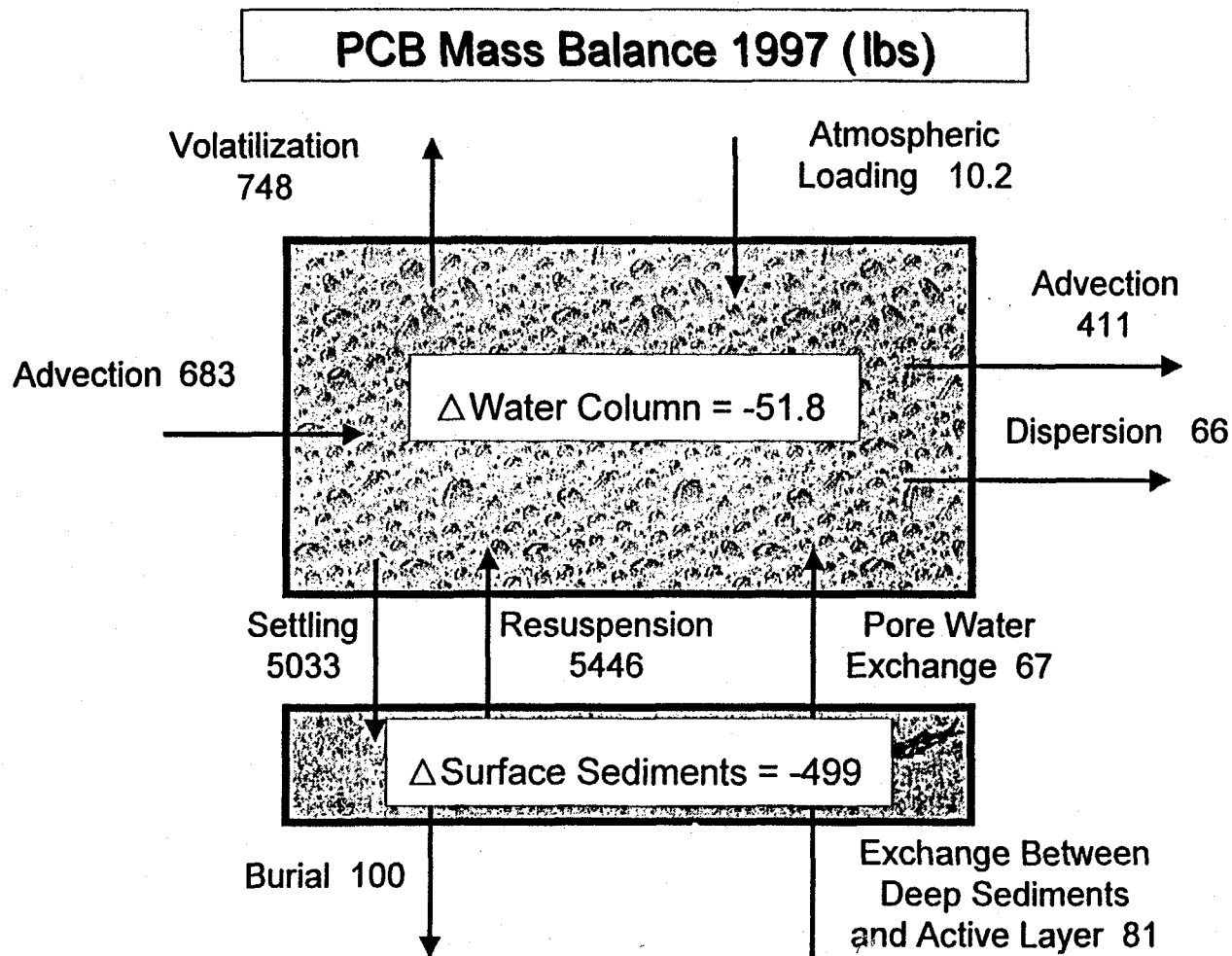


Figure 3-11. Annual PCB flux balance for 1997.

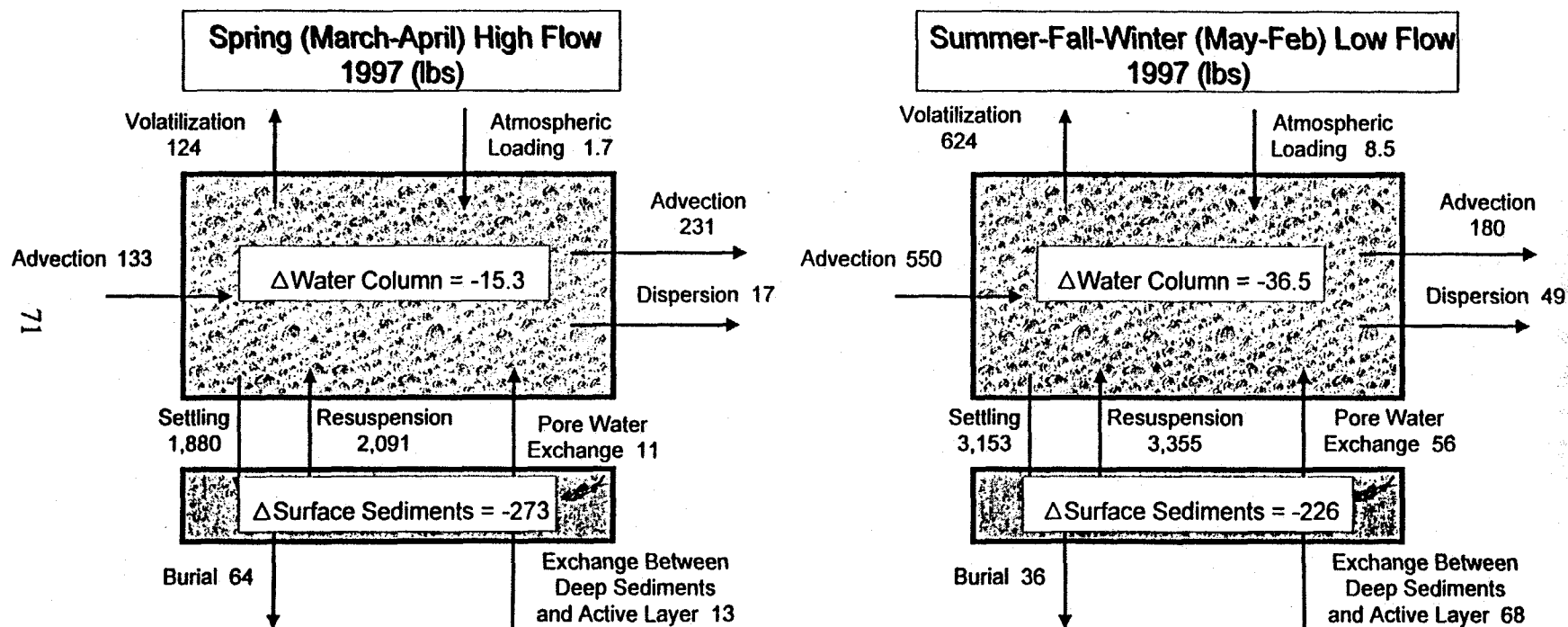


Figure 3-12. Seasonal PCB flux balance for 1997.

Mohawk Rivers. An additional 480 pounds of PCBs entered the water column from the underlying sediments (i.e., resuspension minus settling plus pore water exchange). Of the total 1160 pounds to enter the water column, 65% of the total was lost by volatilization and the remaining 35% was transported down to New York/New Jersey Harbor by advection and tidal dispersion.

On a seasonal basis (Figure 3-12), only 20% of the 1997 PCB load from the Upper Hudson and Mohawk Rivers entered the estuary during the two month Spring high flow period. (For 1997, the highest PCB loads occurred during May, June, and July.) About half of the PCB mass that re-entered the water column from contaminated sediments occurred during the Spring high flow and half during the remaining ten month Summer-Fall-Winter low flow period. Similar to 1992, volatilization was less important during the two month Spring high flow period but was the dominant removal mechanism during the ten month Summer-Fall-Winter low flow period. Transport of PCBs to New York/New Jersey Harbor was again equally divided between the two flow periods.

Based on the flux balance results, the effects of seasonal transport on the fate and transport of PCBs through the estuary can, in general terms, be summarized as follows. During the Spring high flow period, disproportionate amounts of PCBs and suspended sediments typically enter the estuary. Because hydraulic residence times in the estuary are shorter and suspended solids concentrations are greater during high flow periods, PCB loss by volatilization is minimized and most of the incoming PCB load is accumulated in sediments or transported down to the harbor. This Spring-time accumulation of PCBs in sediments (without significant volatile loss) results in a relatively small longitudinal gradient in sediment PCBs on a μg PCB per g (dry wt) basis. During the Summer-Fall-Winter low flow period (with longer residence times and lower suspended solids concentrations), volatilization becomes the primary removal pathway. As PCBs are lost from the water column during their transport through the estuary, they are replenished in the downstream sections by desorption from contaminated sediments that are continually resuspended by tidal motion. The net result is a significant loss of PCBs by volatilization during the low flow period without an appreciable decrease in the downstream PCB water column concentrations.

Organic Carbon Effects on PCB Fate

Since PCBs have a strong affinity to sorb onto organic carbon, increased organic carbon concentrations in New York/New Jersey Harbor may have a significant effect on fate and transport of PCBs. To examine this possibility, model simulations were compared for the following two cases: (1) our base case where f_{oc} values in the water column were set equal to 0.078 in the tidal freshwater and mid estuary region, and were increased in the Harbor and Bight in accordance with SWEM results (HydroQual 1998) as given in Table A-9; and (2) f_{oc} values in the water column were set equal to 0.078 throughout all regions of the model domain. (For both simulations, DOC values in the water column were set

equal to SWEM results (HydroQual 1998) and f_{oc} and DOC values in the sediments were taken as 0.024 and 10 mg/L, respectively.)

A comparison of PCB surface sediment concentrations for the two model simulations is given in Figure 3-13 for the New York/New Jersey Harbor region. PCB concentrations in sediments underlying model segments #1-13 (not shown), #14 (HR Yonkers) and #15 (HR Manhattan) are nearly identical for the two simulations. For the remaining segments, particularly for the Upper Harbor, Newark Bay, and Raritan Bay, PCB concentrations in the surface sediments are approximately a factor of two greater for our base case which includes higher POC concentrations in New York/New Jersey Harbor and Bight. This suggests that the higher POC concentrations are effective in binding PCBs and transporting them to the sediments. Therefore, increased PCB concentrations that have been observed in Upper Harbor and Newark Bay sediments are in part associated with increased water column concentrations of sewage-derived and phytoplankton organic carbon.

Recalibration Results and Time History/Projections for PCBs in Perch

As shown previously in Figure 2-1, bioaccumulation model calculations were performed for five food web regions: Food Web Region 1 representing the tidal freshwater portion of the estuary (RM 153.5 to 73.5), Food Web Region 2 representing the mid estuary (RM 73.5 to 13.5), Food Web Region 3 representing New York/New Jersey Harbor, Food Web Region 4 representing Long Island Sound and the New York Bight, and Food Web Region 5 representing the open ocean. A full description of the five component food chain model (phytoplankton, zooplankton, small fish, perch, and striped bass) used in the bioaccumulation calculations is given in Chapter 2. Recalibration of the bioaccumulation model was initially performed using 1993 and 1995 PCB perch homologue data and then tested using 1987 through 1997 total PCB perch and striped bass data.

Calibration of the bioaccumulation model was as follows: Model results for tri-CB accumulation in perch (which are relatively insensitive to the phytoplankton uptake rate/growth rate ratio ($k_{u-phyto}/k_{g-phyto}$)) were first fit to 1995 field observations by adjusting the chemical assimilation efficiency (α) and the gill transfer efficiency (β). (In our calibration, α and β values were held constant for all PCB homologues and were assumed to be the same for zooplankton, small fish, perch, and striped bass.) Since tri-CB exposure concentrations were varying slowly in 1995, the α/β ratio was more important than the actual α and β values in fitting the field data. The ratio of α/β was then held constant, and α and β values were adjusted proportionality to obtain a good fit of the 1993 field data for tri-CB. For this time period, exposure concentrations in the tidal freshwater and mid estuary were varying and tri-CB accumulations in perch were sensitive to the actual kinetic rate coefficients. Phytoplankton BCF values were then adjusted (by decreasing the $k_{u-phyto}/k_{g-phyto}$ ratio) until a good fit of the field data was obtained for the higher chlorinated homologues. Final parameters used in the bioaccumulation model for the PCB homologues are included in Table 3-1 and model results are presented below.

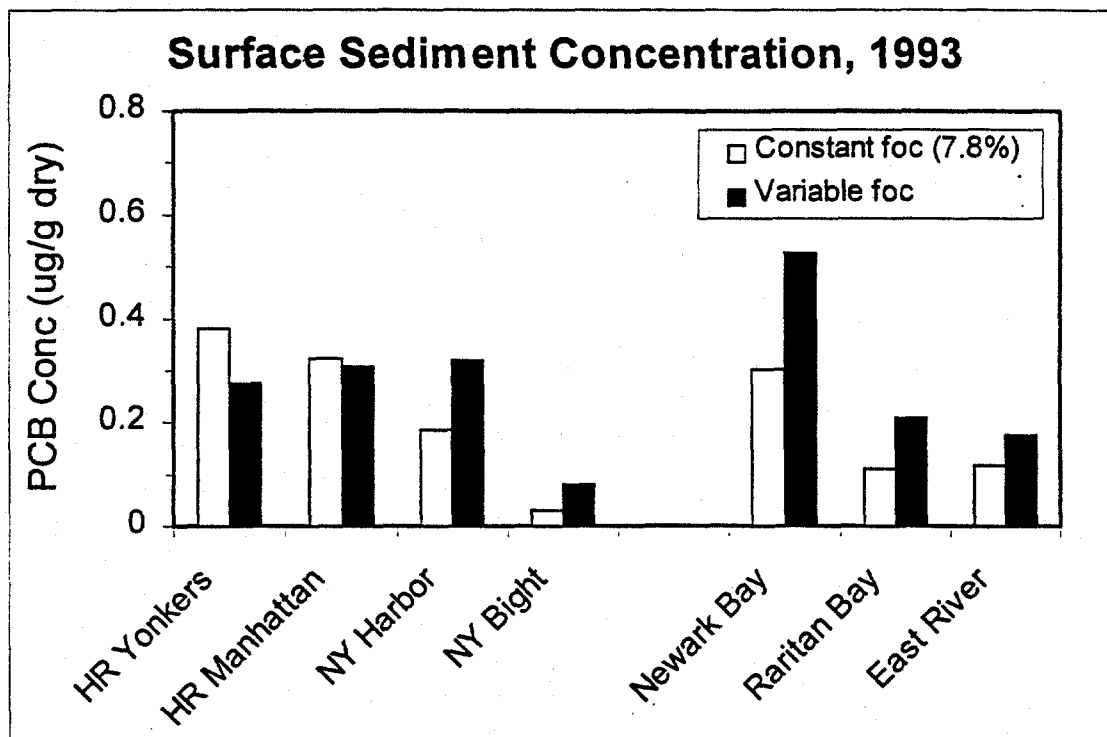


Figure 3-13. Comparison of 1993 PCB concentrations in surface sediments for model simulations with variable water column f_{oc} (base case) and constant f_{oc} of 7.8%.

Because of the limited data available for PCB accumulations in phytoplankton, zooplankton, and small fish, we begin our discussions with perch. Calculated distributions for total PCBs in perch are shown in Figure 3-14 for the 1987-2001 simulation period. For Food Web Region 1 (the tidal freshwater Hudson), PCB concentrations in perch decreased during the period of exponentially decreasing loads from 1987-1991. Peaks in PCB concentrations in perch approaching 10 µg/g (wet wt) occurred in the Fall of 1991 and 1992, and are followed by decreasing PCB concentrations in perch for the remainder of the model simulation. These results are in accordance with temporal distributions of dissolved PCB exposure concentrations in the tidal freshwater Hudson (see results for model segments #3 and #7 in Figure 3-6) which in turn are believed to be related to scouring of contaminated sediments from the Allen Mills water intake structure (September 1991) and seepage of PCB oil from the General Electric's Hudson Falls plant site (Rhea et al. 1997).

For Food Web Region 2 (the mid estuary), PCB concentrations in perch slowly decline for most of the simulation period. (The initial increase in the PCB concentrations is likely the result of underestimating initial conditions for PCBs in perch.) Also of note for Region 2 is that the effects of the increased loadings from the Upper Hudson during 1991 and 1992 are no longer obvious. These results are consistent with temporal distributions of dissolved PCB exposure concentrations in the mid estuary (see results for model segment #11 in Figure 3-6) which show a large dampening of 1991-92 PCB peak responses further down the estuary.

For Food Web Region 3 (NY/NJ Harbor) and Food Web Region 4 (NY Bight and Long Island Sound), model results show similar declines for PCB concentrations in perch. Note that PCB concentrations in perch are much lower in these regions (as indicated by the change in concentration scales on Figure 3-14).

A comparison of field data to model results is also given in Figure 3-14 for Food Web Regions 1 and 2. As shown, the bioaccumulation model, which was calibrated to the 1993 and 1995 EPA/NOAA field data (solid symbols on Figure 3-14), provides a good fit to these data points. The NYS DEC data (open symbols on Figure 3-14) were at times 40% lower than the EPA/NOAA field data and our model results. Differences in the EPA/NOAA field data (and our model results) and the NYS DEC data may in part be reconciled by lipid normalization. This suggests that differences in the EPA/NOAA field data (and our model results) and NYS DEC field data may be related to differences in lipid extraction procedures used in the laboratory analyses.

A more detailed comparison of the 1993 and 1995 EPA/NOAA field data and model results was made on a homologue basis and is given in Figure 3-15. As shown, the model provides a very good description of PCB concentrations in perch in both the tidal freshwater Hudson and mid estuary region. Finally, a shift in the homologue distribution to higher chlorinated PCBs is noted from a comparison of PCB homologue results for perch to the homologue distributions previously shown for dissolved PCB water concentrations (Figure 3-2) and sediment concentrations (Figure 3-4)

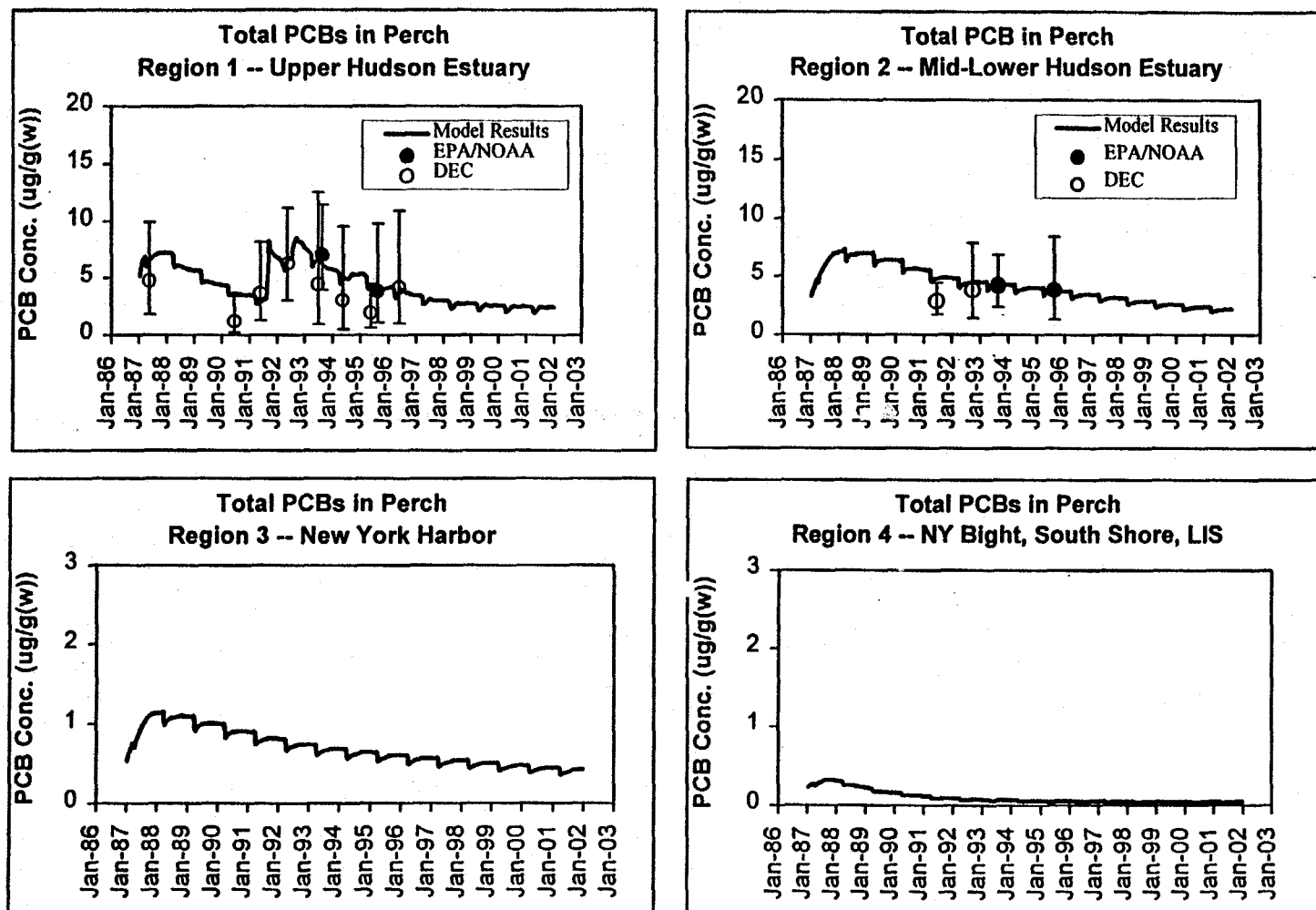


Figure 3-14. Model-field data comparisons for total PCB concentrations in perch. [Data for NYS DEC samples and 1993 EPA/NOAA samples from TAMS/Gradient (1995); data for 1995 EPA/NOAA samples from McGroddy (1997).]

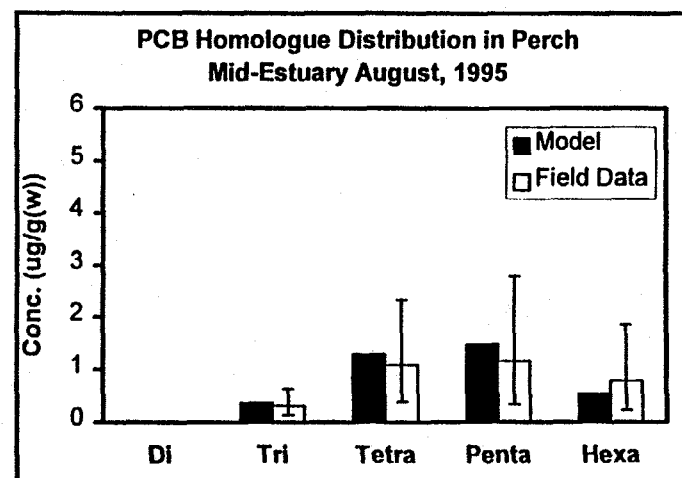
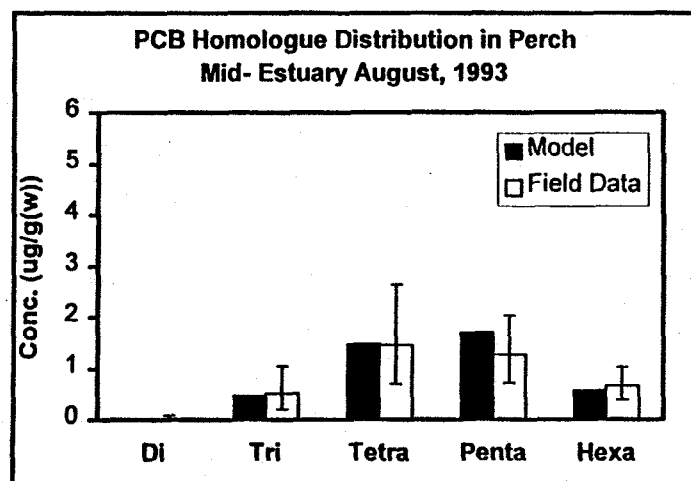
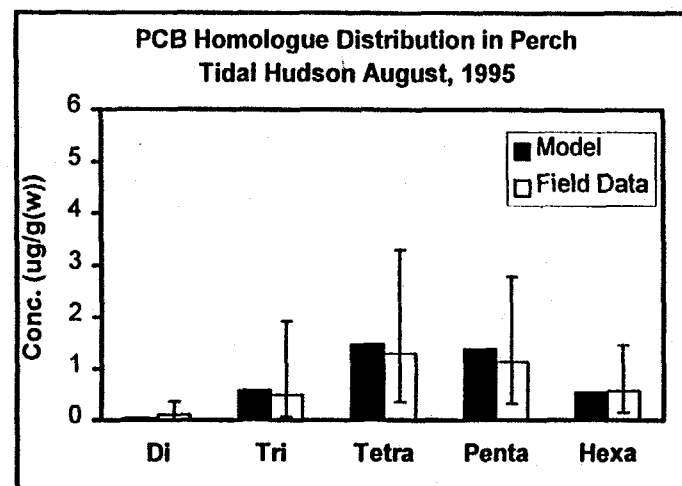
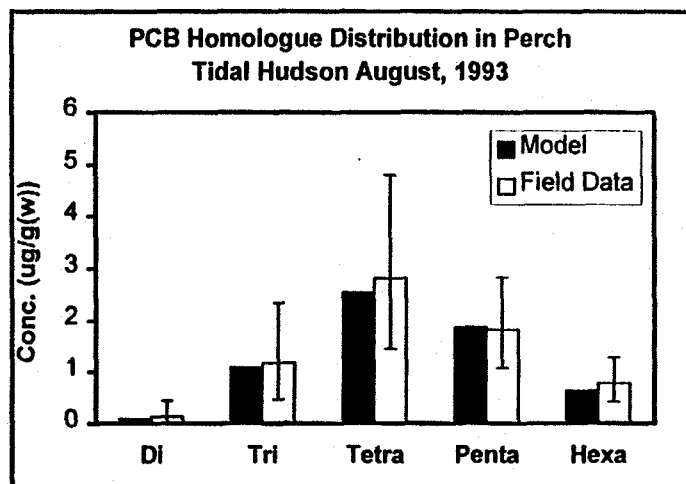


Figure 3-15. Model-field data comparisons for 1993 and 1995 PCB homologue concentrations in perch for the tidal freshwater and mid estuary. [Data for 1993 samples from TAMS/Gradient (1995); data for the 1995 samples from McGroddy (1997).]

Effect of Migration on PCB Accumulation in Striped Bass

As discussed previously in Chapter 2, striped bass feed on a mixed diet of zooplankton (for 0-2 year old striped bass), small fish and perch (for 2-5 year old striped bass), and perch (for 6-17 year old striped bass). Since striped bass are a migratory fish (see Figure 2-13), the PCB content of their prey varies significantly over the course of the year, with high concentrations while striped bass are in the mid estuary (Food Web Region 2) and lower concentrations while striped bass are in the New York Bight, Long Island Sound (Food Web Region 4) and the open ocean (Food Web Region 5).

Simulated PCB model response for a striped bass cohort born in 1987 is given in Figure 3-16. As shown, the 1987 striped bass cohort accumulate relatively high levels of PCBs [5-7 $\mu\text{g/g}$ (wet wt)] during their first two years of life in the mid estuary (solid line indicating Food Web Region 2). As the cohort ages (2-5 year olds), they migrate from the mid estuary (Food Web Region 2) into New York/New Jersey Harbor (open squares indicating Food Web Region 3) and spent the summer months in the New York Bight and Long Island Sound (open triangles indicating Food Web Region 4). In the Bight and Long Island Sound, striped bass ingestion of PCBs is reduced (by feeding on less contaminated prey). This is accompanied by a significant decrease [approximately 1-2 $\mu\text{g/g}$ (wet wt)] in striped bass PCB concentrations as the fish lose PCBs by depuration and further reduce their concentration by growth dilution. Each year, the 2-5 year old striped bass return to the mid estuary in the late Fall and remain there until June. In the mid estuary (solid line indicating Food Web Region 2), striped bass PCB concentrations increase approximately 1-2 $\mu\text{g/g}$ (wet wt) as the fish again feed on more contaminated prey.

This seasonal trend is slightly modified in later years (6-17 year olds), as striped bass spend a large portion of the year in the open ocean (open circles indicating Food Web Region 5). The 6-17 year old striped bass migrate into Long Island Sound and the New York Bight (open triangles indicating Food Web Region 4) in the middle of March and return to the mid estuary (solid line indicating Food Web Region 2) in the middle of April to spawn. They remain in the mid estuary until the middle of July, and it is during this time that their PCB concentrations increase approximately 0.5-1 $\mu\text{g/g}$ (wet wt). Most of this 0.5-1 $\mu\text{g/g}$ (wet wt) of PCBs is lost during migration to the open ocean (open circles indicating Food Web Region 5). There is however a slight increase in striped bass PCB concentrations on a year-to-year basis.

A more detailed analysis of PCB accumulation in the 1987 striped bass cohort is shown in Figure 3-17 for individual PCB homologues. Differences in homologue behavior are noted. At one extreme, there is almost a complete loss of di- and tri-CBs during migration into the New York Bight, Long Island Sound (open triangles indicating Food Web Region 4) and the open ocean (open circles indicating Food Web Region 5). At the other extreme, there is a relatively small decrease in hexa-CB concentrations during migration to the open ocean. Since this decrease in hexa-CB concentrations is less than hexa-CB

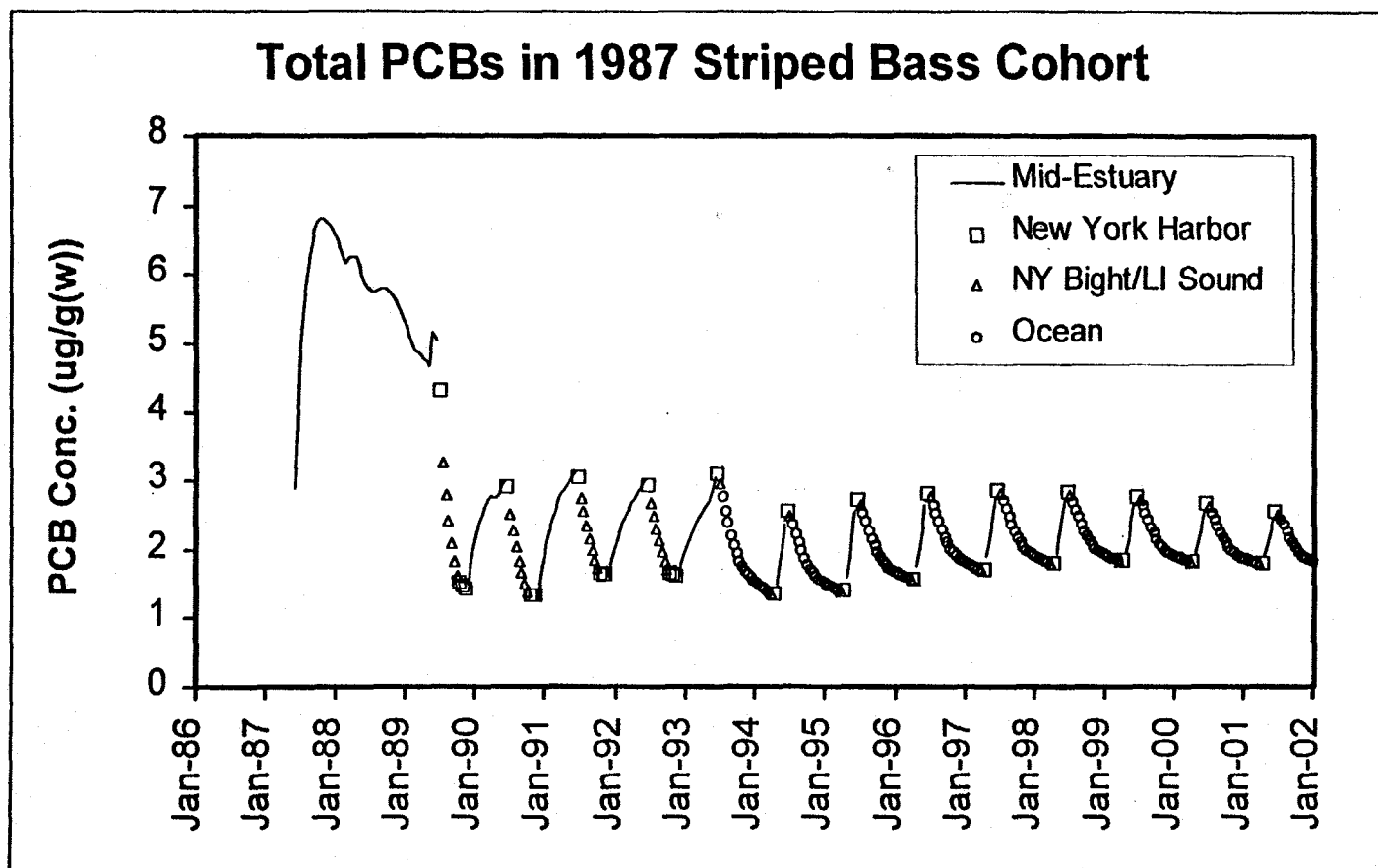


Figure 3-16. Calculated PCB concentrations in a 1987 striped bass cohort. [Assumed migration patterns previously given in Figure 2-13.]

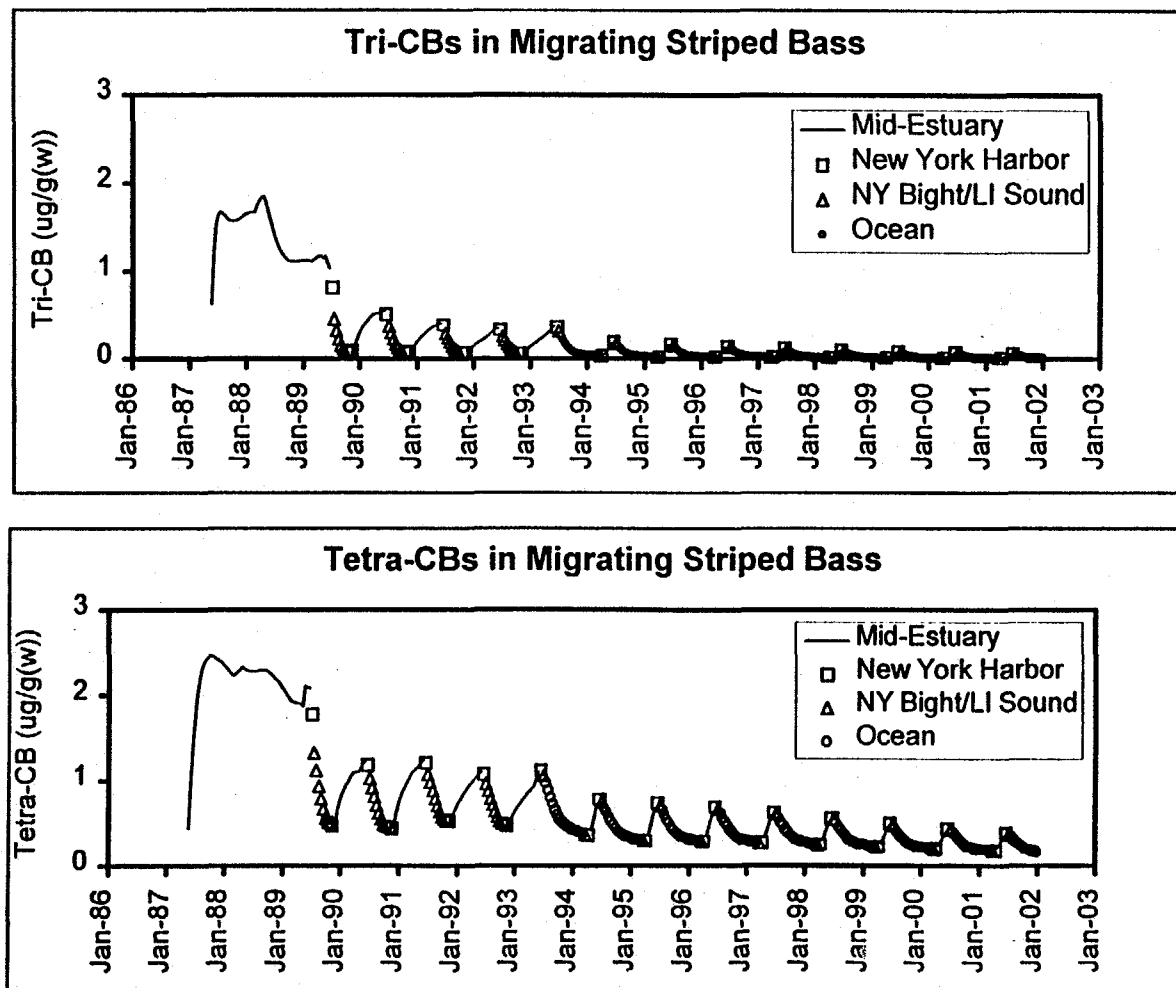


Figure 3-17. Calculated PCB homologue concentrations for a 1987 striped bass cohort.
[Assumed migration patterns previously given in Figure 2-13.]

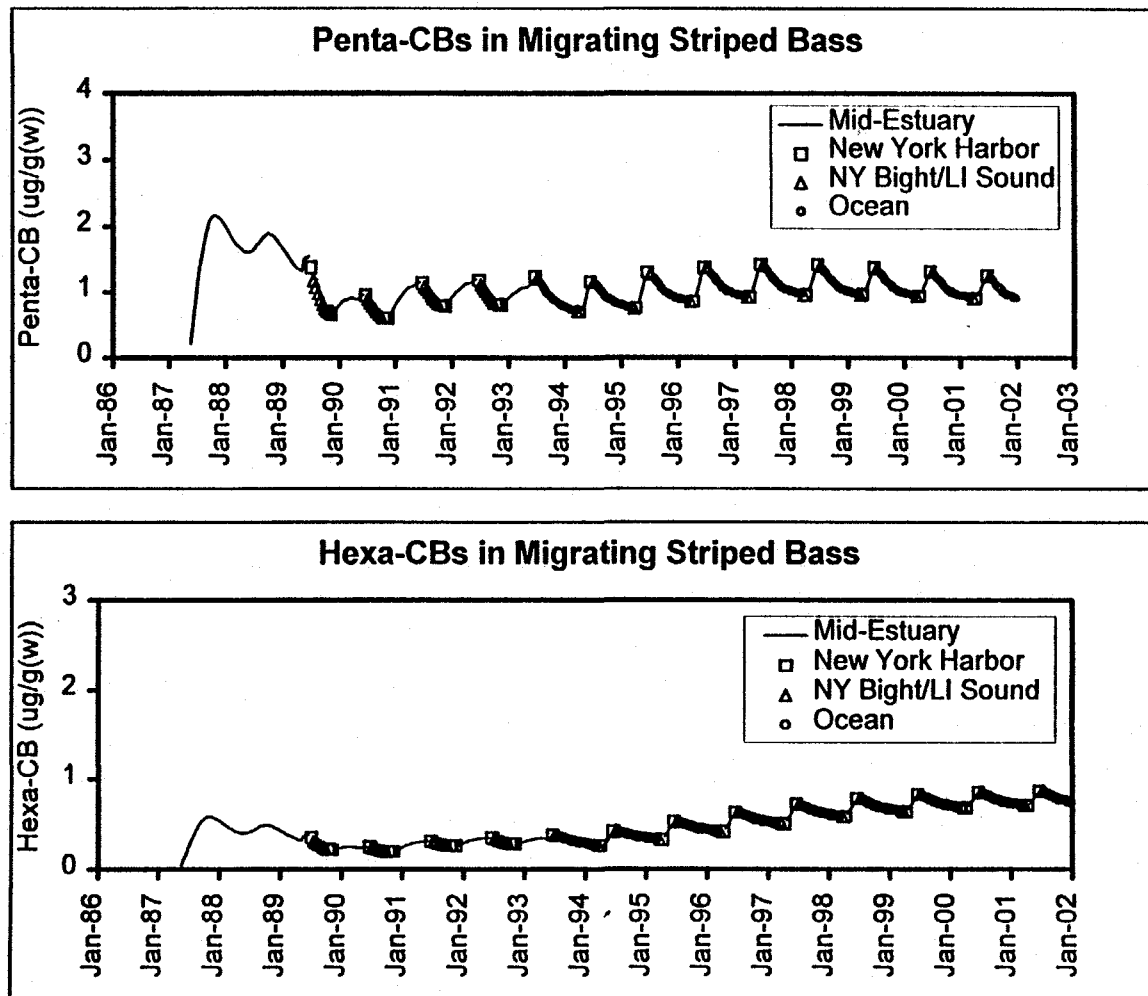


Figure 3-17 (continued). Calculated PCB homologue concentrations for a 1987 striped bass cohort. [Assumed migration patterns previously given in Figure 2-13.]

accumulation during migration into the mid estuary (solid line indicating Food Web Region 2), there is a continuing increase in hexa-CB concentrations over the years.

The differential behavior in PCB homologues in Figure 3-17 is related to their hydrophobicity (and the inverse relationship for depuration rates and K_{ow} given in Eq 15). For the lower chlorinated (low K_{ow}) homologues, depuration rates are relatively fast and the rapid decrease in PCB concentrations is primarily due to depuration. For higher chlorinated (high K_{ow}) homologues, there is a greater affinity for the homologue to remain in fish lipids and depuration rates become smaller. The decrease in PCB concentrations is much slower, and in the limit, is solely dependent on growth dilution. The net effect is an expected shift in PCB homologue distributions to higher chlorinated homologues in older migratory fish.

Comparisons of the PCB striped bass model simulation results and 1987-97 field data (TAMS/Gradient 1995) are shown for 2-5 year old (Figure 3-18) and in 6-15 year old (Figure 3-19) striped bass in the mid estuary (Food Web Region 2). Simulation results are denoted by disconnected lines to represent only the portion of the year that striped bass are in the mid estuary. Field data are presented as seasonal (3-month) average concentrations with 5 and 95%iles. For the 2-5 year old striped bass, model results provide a good match to most of the average observed concentrations. Model results also correspond reasonably well to field data for 6-18 year old striped bass.

A closer look at the 2-5 year old striped bass results (Figure 3-18) shows that a good correspondence between the model calculations and average field data was not always obtained in the Fall, particularly for 1989, 1990, and 1992. To help explain the discrepancy between model results and Fall field data, probability distributions for PCB accumulations in striped bass are presented in Figure 3-20 for the 1990 and 1992, Spring and Fall catches. For the Spring catches, PCB concentrations in striped bass follow log normal distributions and results suggest that a single population of striped bass is dominant in the mid estuary. For Fall catches, particularly in 1992, results suggest that two populations of striped bass with different behavior are present in the mid estuary. As discussed in the next section, a subpopulation of non-migrating striped bass, which accounts for a large portion of the striped bass catch in the Fall, is present along with the migrating fish.

Since the non-migrating striped bass are expected to have higher PCB accumulations (from continuous exposure to contaminated prey), average PCB concentrations in the migrating striped bass were recalculated (after eliminating the top two points from the 1990 distribution and the top four points from the 1992 distribution). New average PCB concentrations for Fall 1990 and Fall 1992 are plotted as open squares on Figure 3-18 and are shown to correspond well with model calculations. Model results therefore appear to provide a good description of both seasonal behavior and long-term trends in PCB accumulations in migrating striped bass. An analysis of non-migrating striped bass is given below.

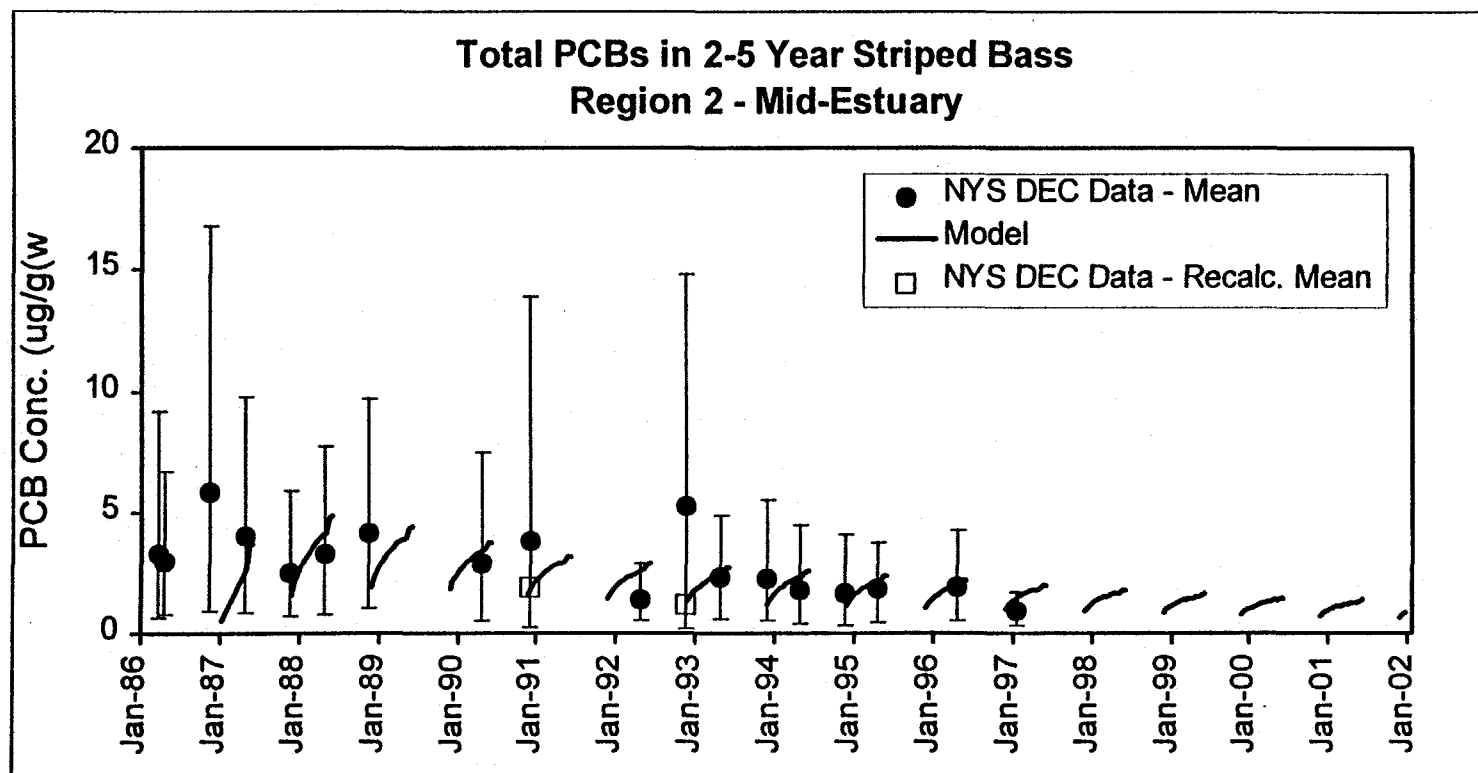


Figure 3-18. Model-field data comparisons for PCB concentrations in 2-5 year old striped bass. [Data from NYS DEC (TAMS/Gradient 1995).]

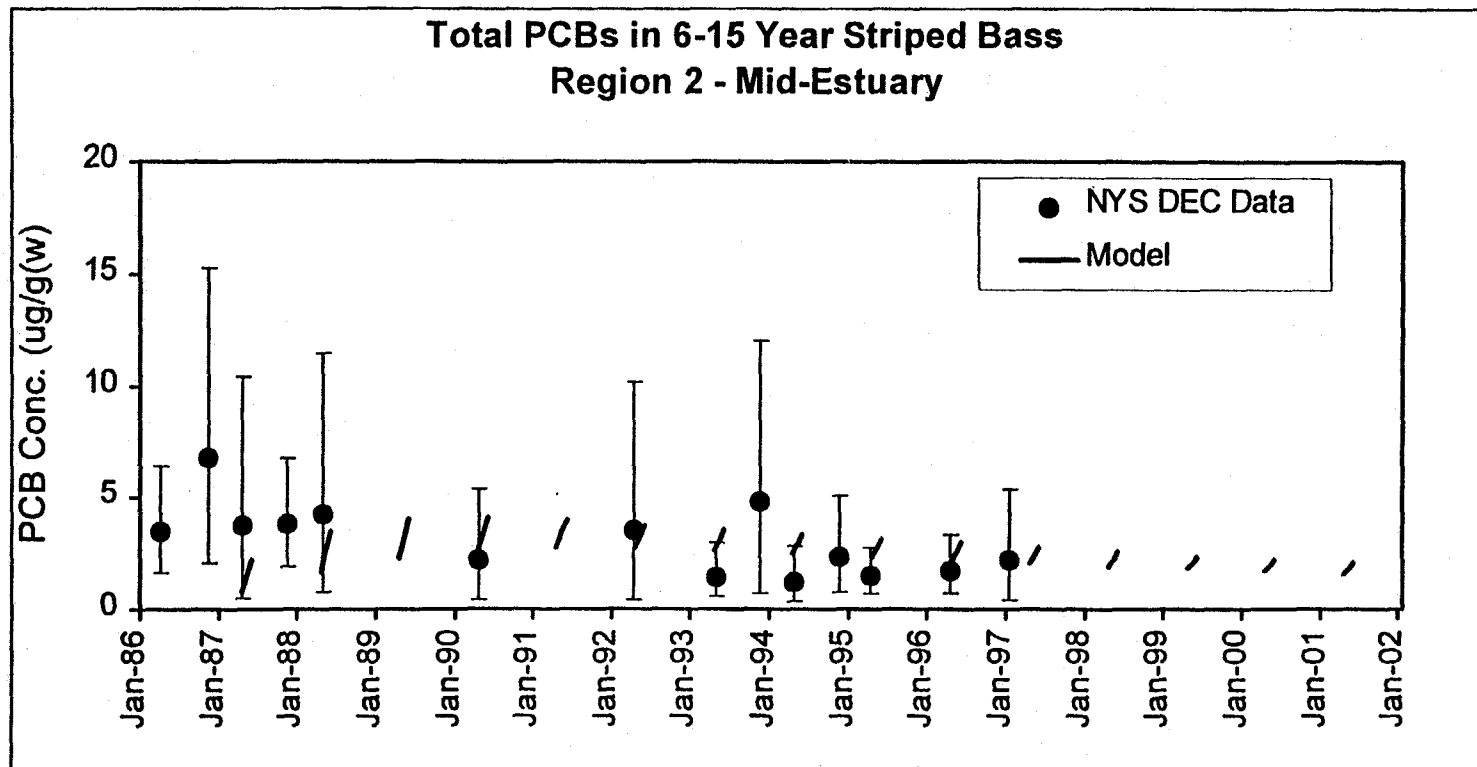


Figure 3-19. Model-field data comparisons for PCB concentrations in 6-15 year old striped bass. [Data from NYS DEC (TAMS/Gradient 1995).]

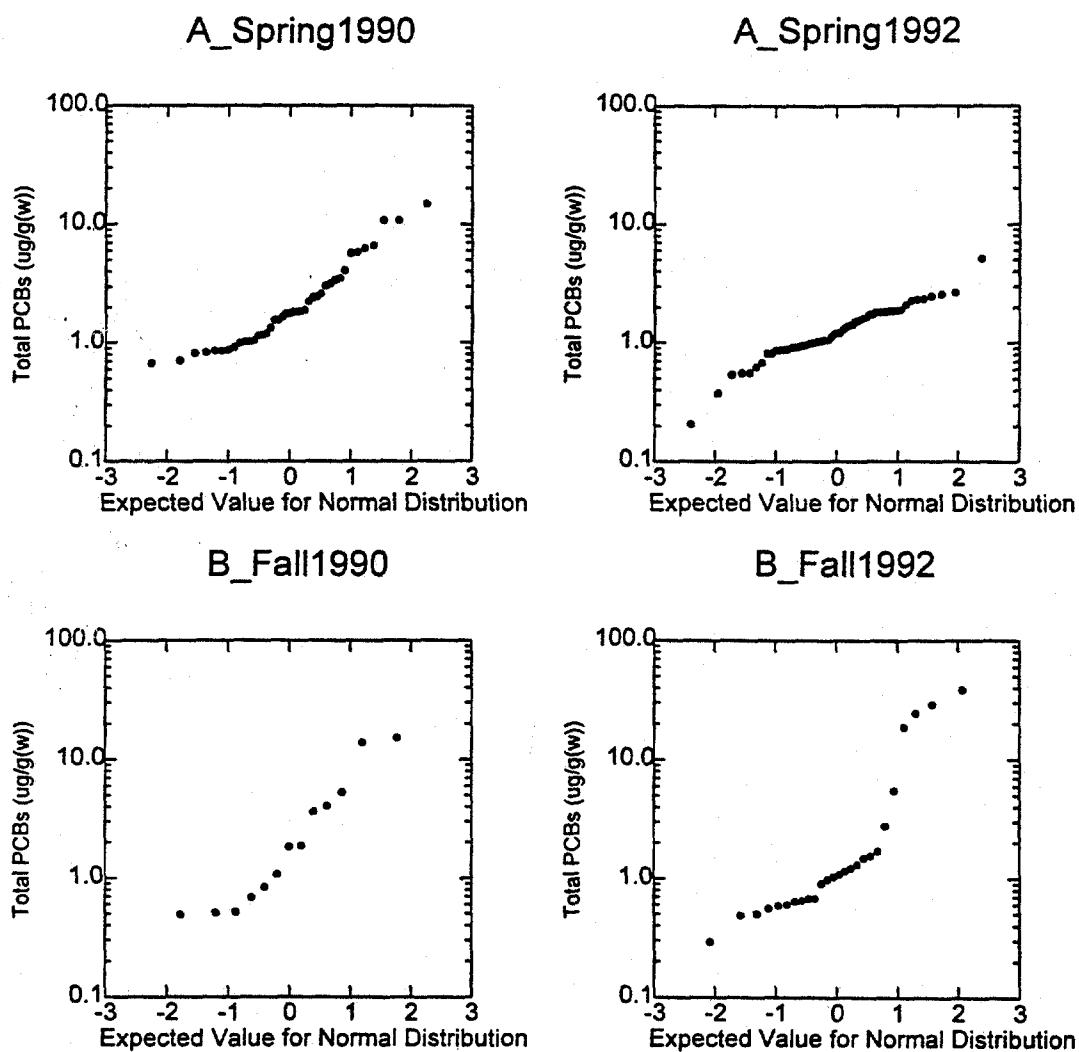


Figure 3-20. Probability distributions for PCBs in 2-5 year old striped bass for 1990 and 1992, Spring and Fall Catches. [Data from NYS DEC as reported in TAMS/Gradient (1995).]

Effect of Non-Migration on PCB Accumulation in Striped Bass

In recent studies, Secor and Piccoli (Secor and Piccoli 1996) have shown that strontium:calcium ratios in striped bass otoliths can be used to estimate the life history (in terms of salinity environment) for individual striped bass. Representative results of their work (Figure 3-21) show a wide range in migration behavior for two male striped bass. One striped bass (Figure 3-21, top panel) is shown to have followed a somewhat expected migration pattern, spending his first few years in the mid estuary and migrating out to ocean waters later in life. The other striped bass (Figure 3-21, bottom panel) is shown to have never left the tidal freshwater region of the estuary. This latter migration behavior is believed to be representative of a subpopulation of non-migrating (predominantly male) striped bass.

Since the non-migrating male subpopulation would have continuous exposures to high PCB concentrations, they are expected to have higher PCB accumulations. Our breakdown of PCB concentrations in female and male striped bass (Figure 3-22) shows a higher PCB average concentration and a more noticeable presence of outliers for males (as indicated by the open circles on Figure 3-22). These results are consistent with the notion of a portion of the male population stays in the estuary year round while the remainder of the males and most of the females follow a seasonal migration pattern.

To examine the effect of limited migration behavior on PCB accumulation in striped bass, model calculations were performed for a subpopulation of non-migrating striped bass in the mid-estuary. Results of this calculation are shown by the continuous lines in Figure 3-23 for 2-5 and 6-15 year old striped bass. As expected, results for the non-migrating fish show much higher PCB accumulations, with 1994 concentrations of approximately 5 $\mu\text{g/g}$ (wet wt) for the 2-5 year olds and 15-20 $\mu\text{g/g}$ (wet wt) for the 6-15 year olds. These results correspond reasonably well to the distribution outliers that were previously shown in Figure 3-20 and 3-22 for the 1987-97 PCB striped bass data.

Similar model calculations were performed for a subpopulation of non-migrating striped bass in the tidal freshwater Hudson (Food Web Region 1). Calculated PCB concentrations for the 2-5 year old and 6-15 year old striped bass in the tidal freshwater region are shown in Figure 3-24 and are similar in magnitude to results for the mid estuary (Figure 3-23). The response of 2-5 year old striped bass in the tidal freshwater region to increased Upper Hudson loads in the early 1990s however is readily apparent.

Finally, comparison of homologue distributions in non-migrating and migrating striped bass are given in Figure 3-25. For non-migrating striped bass (top panel of Figure 3-25), there is a shift from less chlorinated PCB homologues in the tidal freshwater region to more chlorinated PCB homologues in the mid estuary. This trend is consistent with the field data of Ashley and Baker (Ashley et al. submitted), and is due to the enrichment of lower chlorinated homologues in the Upper Hudson PCB load. In the mid estuary (bottom panel of Figure 3-25), PCB homologue distributions for migrating striped bass vary but are on average appear to be quite similar to the distribution for the non-migrating fish, making

Salinity Regime versus Age

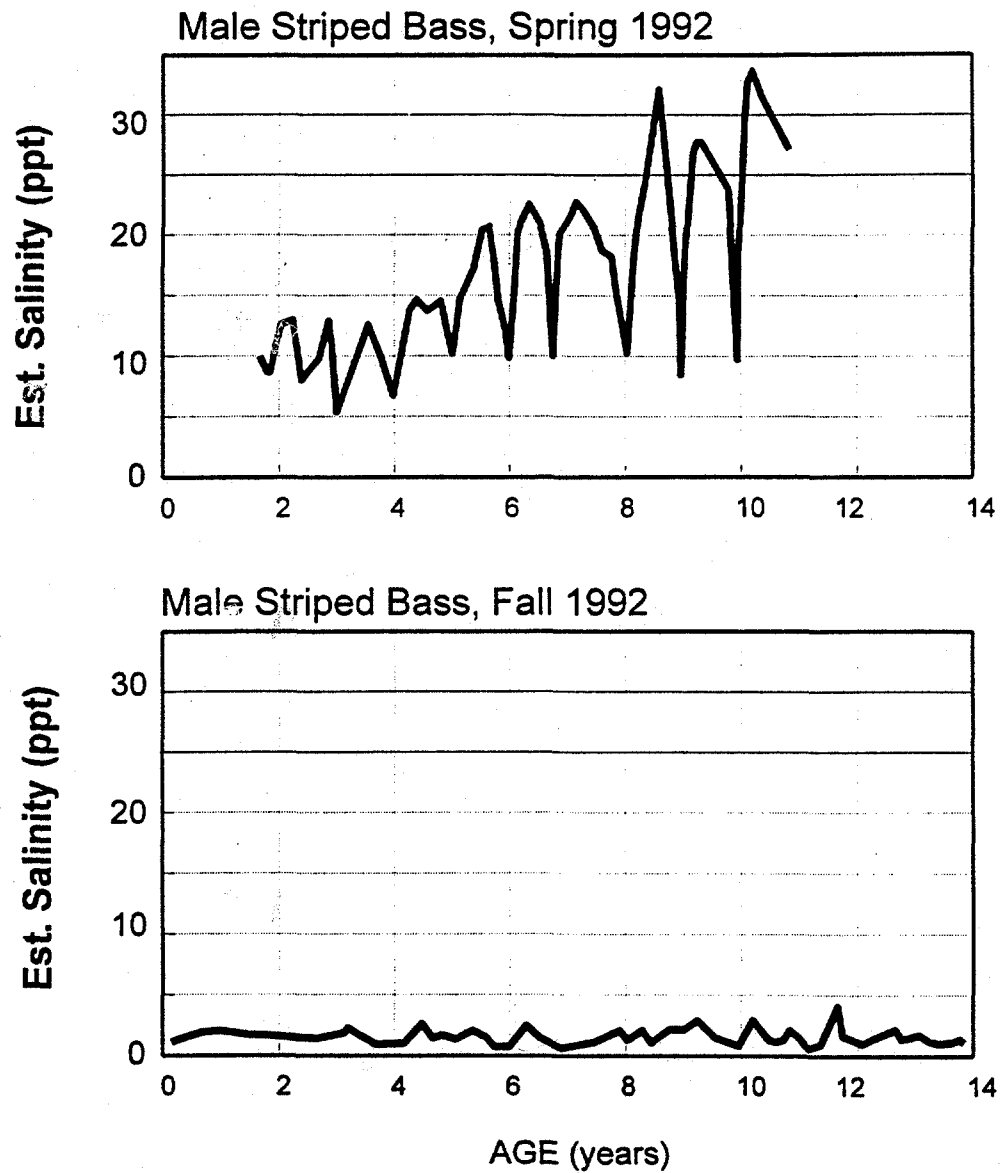


Figure 3-21. Estimated life histories for individual striped bass (in terms of salinity environment). [From strontium:calcium ratios in striped bass otoliths (Secor and Piccoli 1996).]

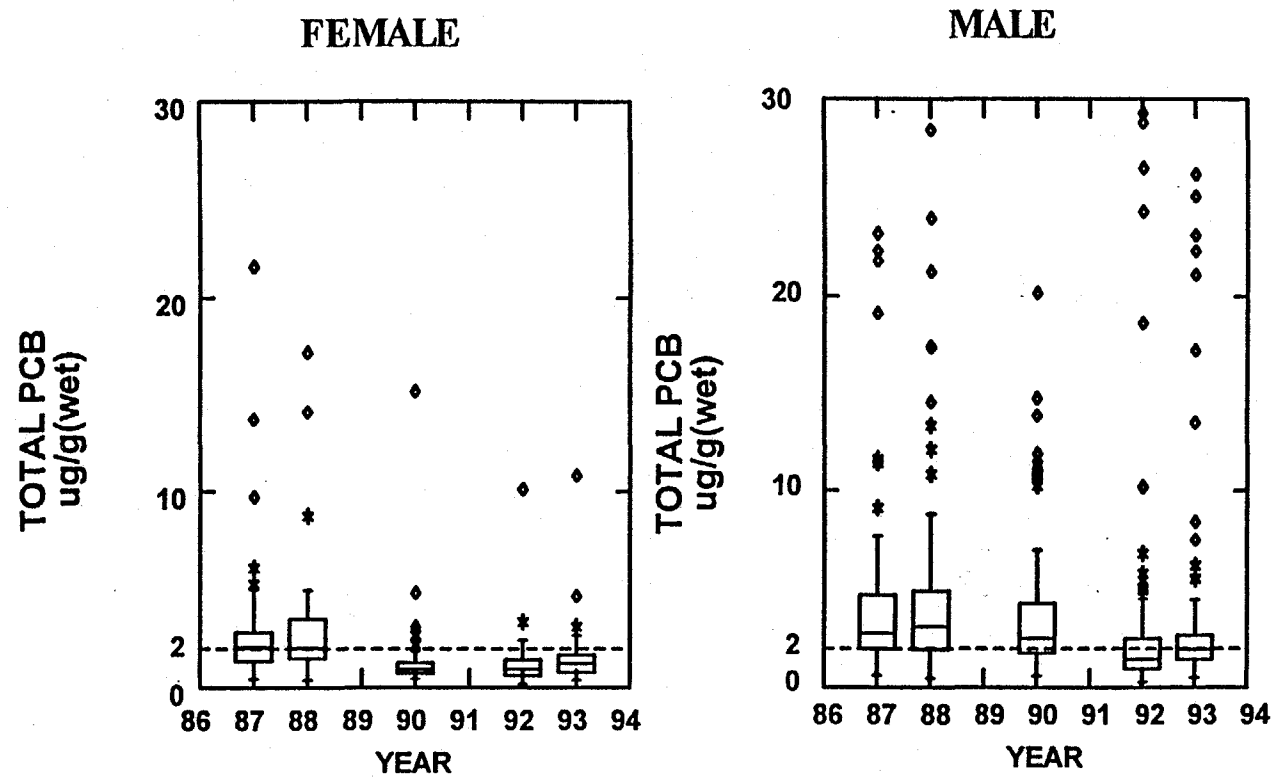


Figure 3-22. Variations in PCB concentrations in 2-5 year old female and male striped bass caught in the mid estuary. [Data from NYS DEC as reported in TAMS/Gradient (1995).]

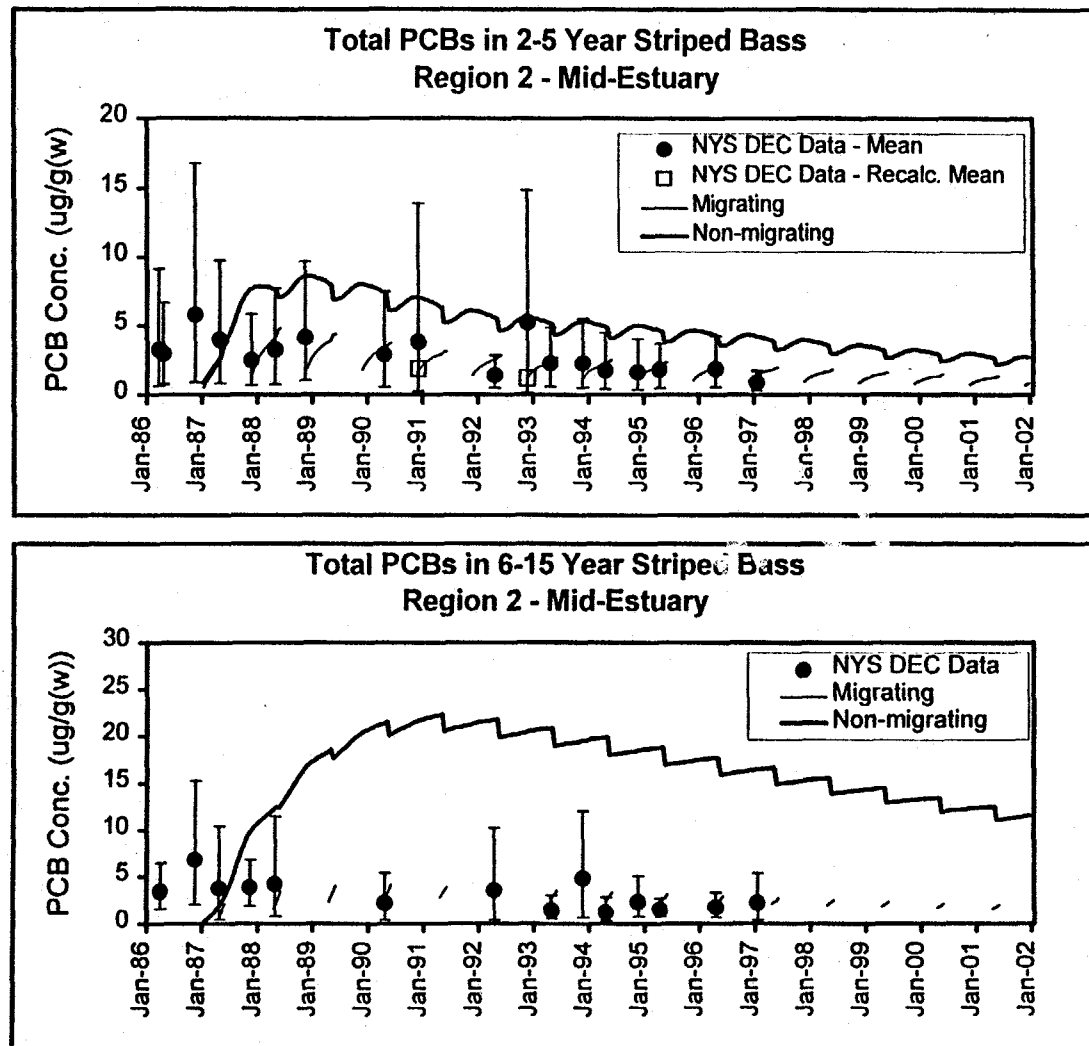


Figure 3-23. Calculated PCB accumulations for a subpopulation of non-migrating 2-5 and 6-15 year old striped bass in the mid estuary. [Data from NYS DEC as reported in TAMS/Gradient (1995) is shown for comparison.]

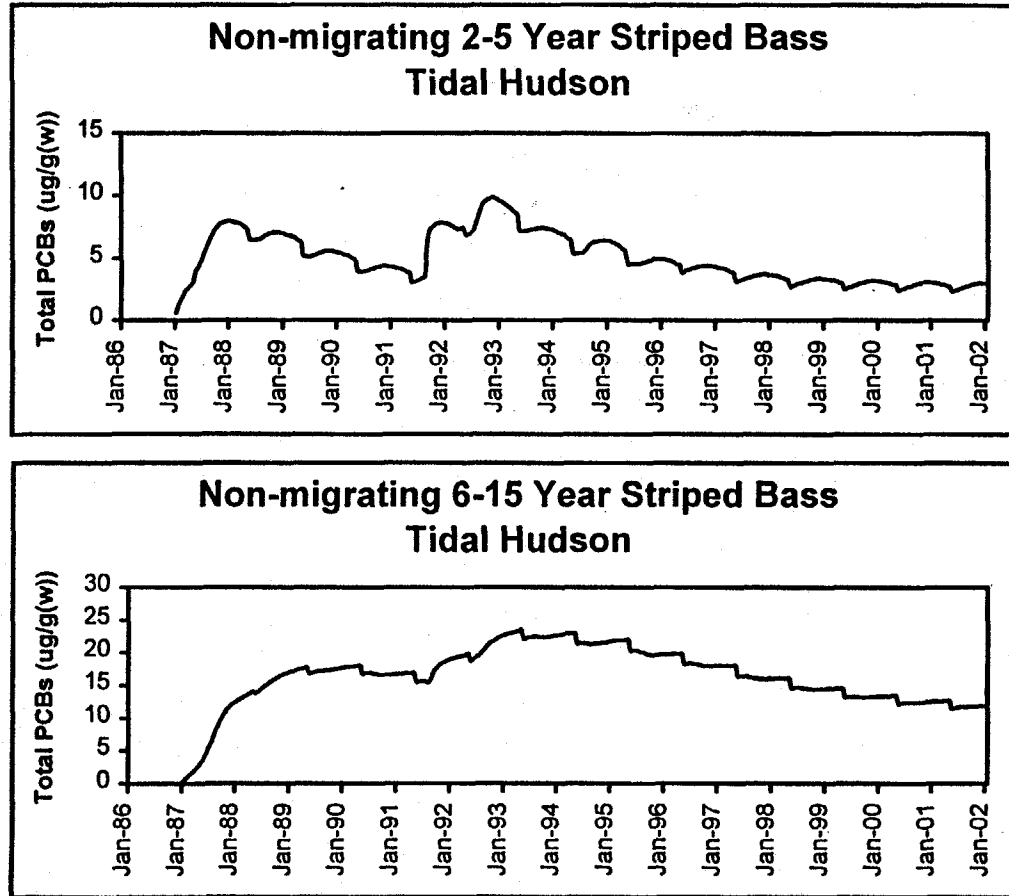


Figure 3-24. Calculated PCB accumulations for a subpopulation of non-migrating 2-5 and 6-15 year old striped bass in the tidal freshwater Hudson.

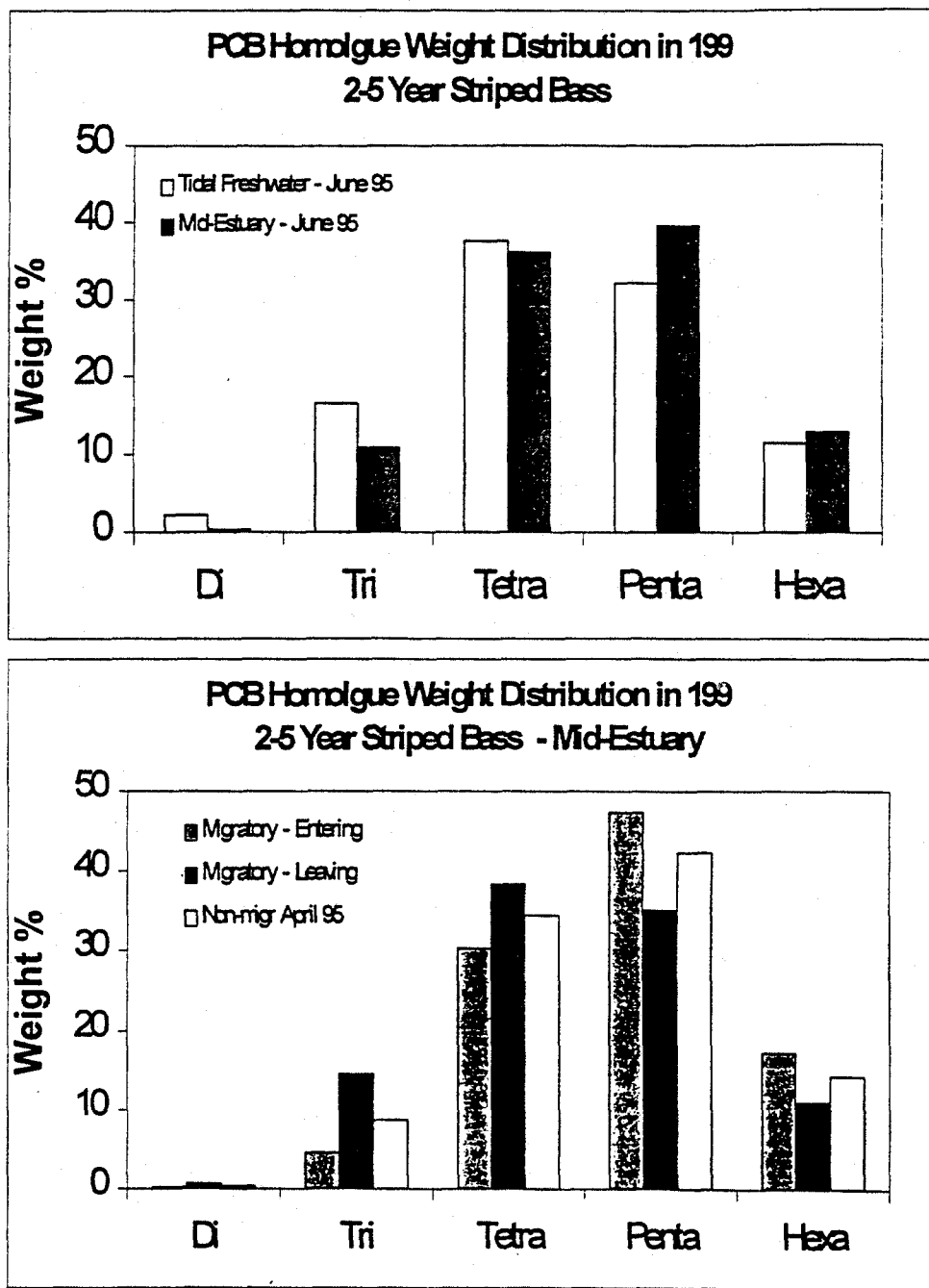


Figure 3-25. Comparisons of homologue distributions in non-migrating striped bass from the tidal freshwater and mid estuary (Upper Panel) and of homologue distributions in migrating and non-migrating striped bass in the mid estuary (Bottom Panel).

it difficult to distinguish between migrating and non-migrating striped bass based on homologue distributions. In this region, variations in homologue distributions for the migratory fish occur as follows: As striped bass enter the mid estuary, they are deficient in lower chlorinated PCB homologues due to the more rapid depuration of lower chlorinated homologues during migration to the New York Bight, Long Island Sound, and the open ocean. Migrating striped bass however become enriched in lower chlorinated PCB homologues during their stay in the mid estuary and leave the mid estuary slightly enriched in the more chlorinated PCB homologues.

Component Load Analysis

Time histories/projections for PCB concentrations in sediments and fish were performed to examine the relative impacts of various PCB source loadings on PCB sediment and fish accumulations, and to determine a time projections for the reopening of the Hudson River striped bass fishery. Three cases were considered in these evaluations: (1) the 1987 PCB initial conditions in sediments plus all PCB loads [including the Upper Hudson, Mohawk River, New Jersey tributaries, wastewater treatment plant (WWTPs), combined sewer overflows (CSOs), storm water outflows (SWOs), and atmospheric deposition], (2) the 1987 PCB initial conditions in sediments plus all PCB loads except the Upper Hudson load, and (3) only the 1987 PCB initial conditions in sediments.

Model results for total PCBs in surface sediments are given in Figure 3-26 for 1992, 1997, and 2001. For 1997 and 2001, 60-80% of the PCB sediment response in the tidal freshwater and mid estuary is due to the Upper Hudson load. PCB responses in harbor sediments are more equally divided among the Upper Hudson load, WWTP, CSOs, SWOs, and tributary loadings from the New York-New Jersey Harbor region, and the 1987 PCB initial conditions in sediments. Of particular note is the longer-term memory of harbor sediments to the 1987 initial condition. This response is primarily due to longer time-scales associated with the downstream passage of contaminated sediments from the tidal freshwater and mid estuary to the harbor, and as discussed previously, the effect of PCB trapping by higher POC concentrations in the harbor region.

A break-down of 1997 sediment responses for PCB homologues is shown in Figure 3-27. As shown, sediment concentrations of di-, tri-, and tetra-CB in the tidal freshwater and mid estuary are predominantly associated with the Upper Hudson load. For penta- and hexa-CB, approximately 50% of the PCB sediment response in the tidal freshwater and mid estuary is due to the Upper Hudson load. As discussed below, this difference in homologue response (which is largely determined by the prevalence of di-, tri-, and tetra-CB in the Upper Hudson load) is important in determining the effects of various loadings on PCB accumulation in fish.

Corresponding model results for total PCBs in perch and migrating striped bass are given for the mid estuary region in Figure 3-28. In this figure, PCB concentrations in perch

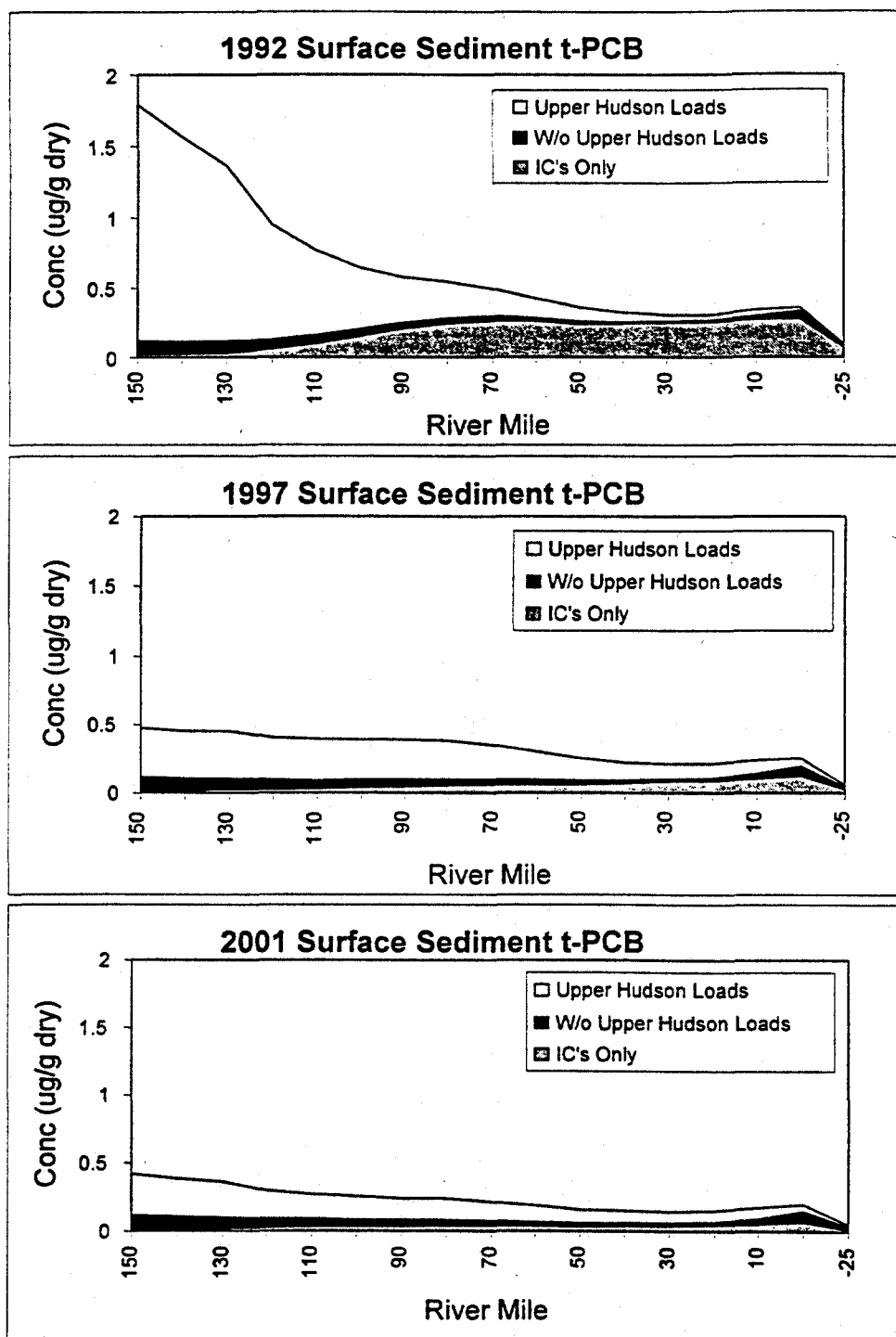


Figure 3-26. Model results for total PCBs in surface sediments for 1992, 1997, and 2001 due to (1) 1987 initial PCB concentrations in sediments; (2) 1987 initial conditions plus all PCB loads except the Upper Hudson load; and (3) 1987 initial conditions plus all PCB loads including the Upper Hudson load.

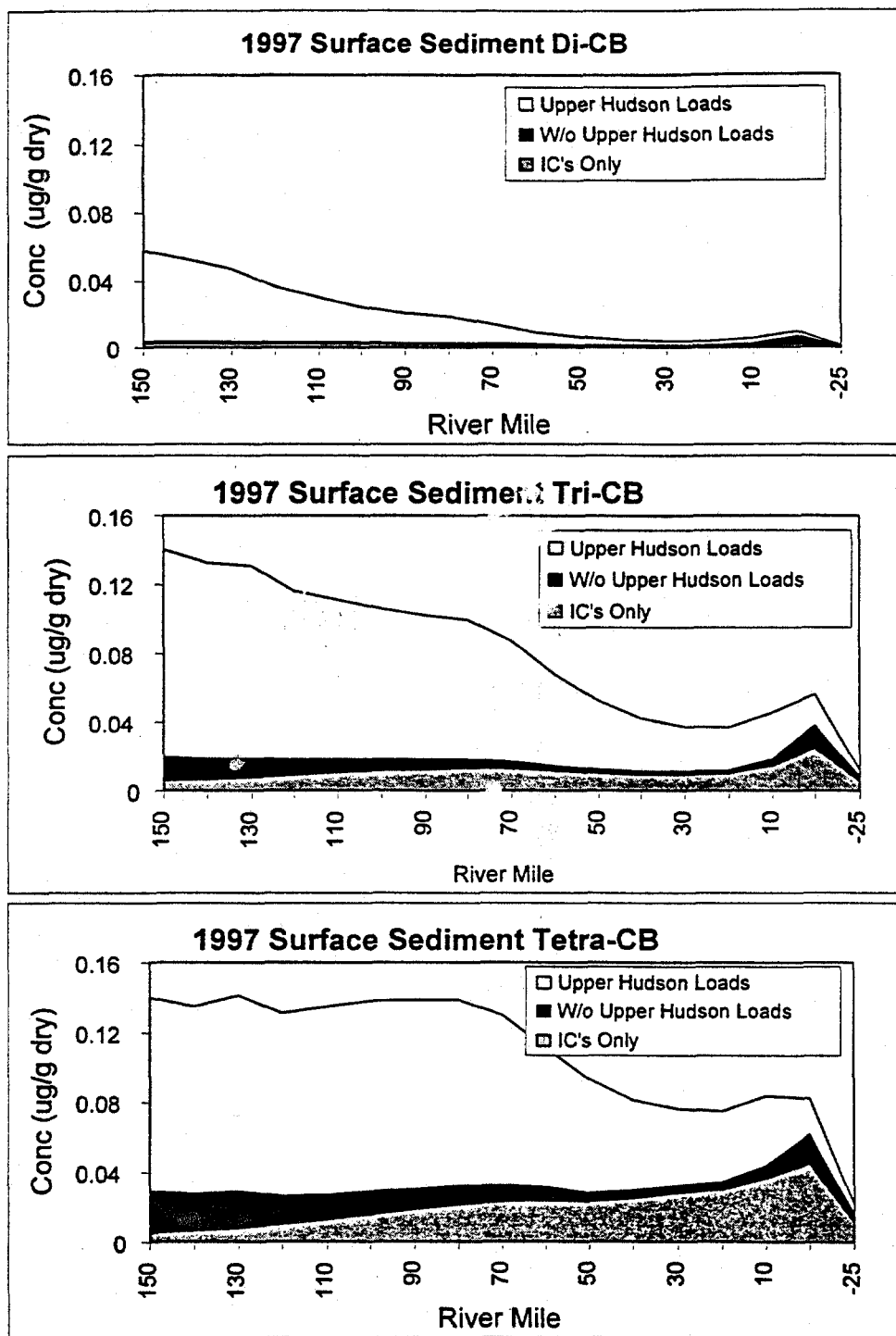


Figure 3-27. Model results for PCB homologues in surface sediments for 1992, 1997, and 2001 due to (1) 1987 initial PCB concentrations in sediments; (2) 1987 initial conditions plus all PCB loads except the Upper Hudson load; and (3) 1987 initial conditions plus all PCB loads including the Upper Hudson load.

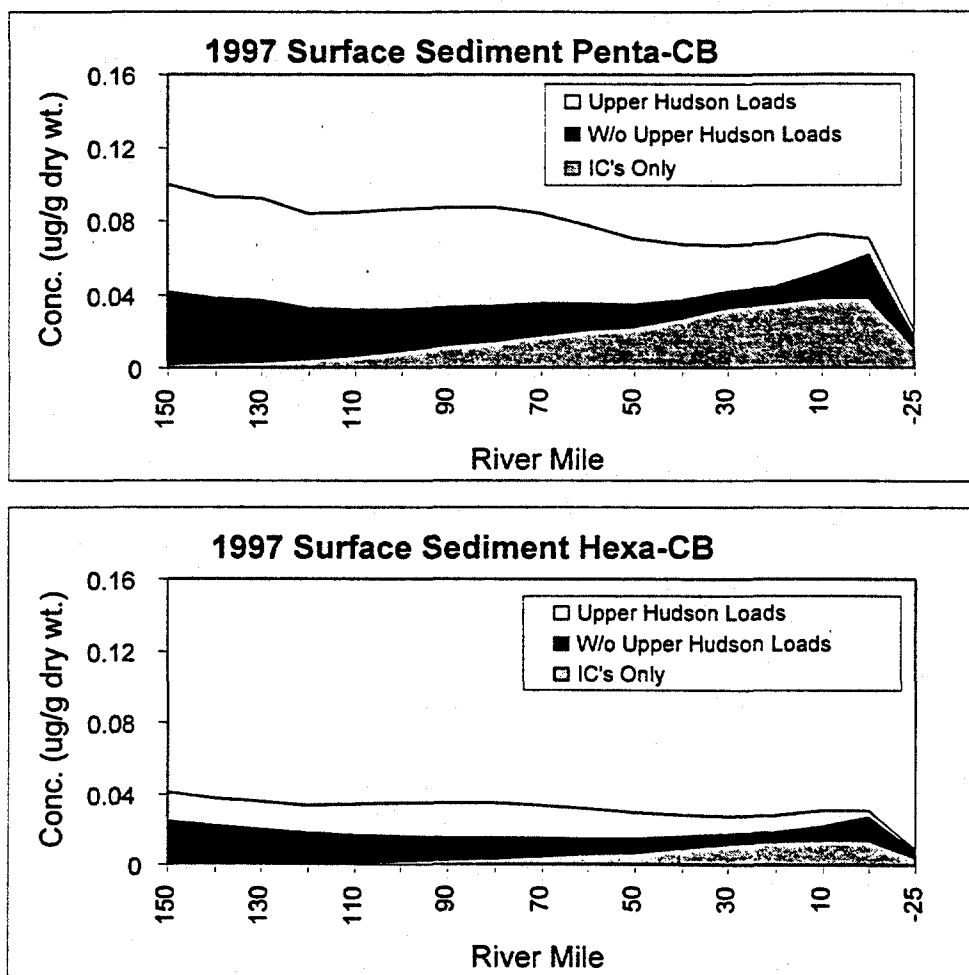


Figure 3-27 (continued). Model results for PCB homologues in surface sediments for 1992, 1997, and 2001 due to (1) 1987 initial PCB concentrations in sediments; (2) 1987 initial conditions plus all PCB loads except the Upper Hudson load; and (3) 1987 initial conditions plus all PCB loads including the Upper Hudson load.

are representative of average tissue concentration over the entire year, while PCB concentrations in striped bass represent average tissue concentrations for periods when migrating striped bass are in the mid estuary. 5 and 95%iles for PCB concentrations in striped bass are also included in the figure and were computed following the approach in Thomann (Thomann et al. 1989), where a log normal distribution of PCB concentrations in striped bass is assumed based on an analysis of existing field data. 5 and 95%ile concentrations are then determined from the average PCB concentration as computed by the model and the coefficient of variation using the following relationships:

$$\sigma_{\ln x}^2 = \ln [1 + \delta_x^2] \quad (1)$$

$$\mu_{\ln x} = \ln \mu_x - \frac{1}{2} \sigma_{\ln x}^2 \quad (2)$$

$$x_{5,95\%ile} = \exp [\mu_{\ln x} + \sigma_{\ln x} Z] \quad (3)$$

where $\sigma_{\ln x}^2$ is the variance of the log transformed concentrations, δ_x is the coefficient of variation (given as σ_x/μ_x), σ_x is the standard deviation, μ_x is the arithmetic mean, $\mu_{\ln x}$ is the log transformed mean concentration, $x_{5,95\%ile}$ are the 5 and 95%ile concentrations, and Z is the standard normal deviate and is equal to -1.64 and +1.64 in computing the 5 and 95%iles, respectively. The coefficient of variation is specified based on an analysis of historical data and results for seasonal catches of 2-5 and 6-15 year old striped bass are given in Figure 3-29. Based on these results, δ_x for the 2-5 and 6-15 year old striped bass were specified as 1.0 and 0.8, respectively.

Fish results for 1992 (Figure 3-28) show that average concentrations in perch, 2-5 year old striped bass, and 6-15 year old striped bass are all above the Food and Drug Administration (FDA) guideline of 2 $\mu\text{g/g}$ (wet wt). During this early period in the simulation, PCB concentrations in fish are largely affected by PCB contaminated sediments that were present at the start of the 1987-2001 simulation period.

By 1997, average PCB concentrations in 2-5 year old striped bass are less than 2 $\mu\text{g/g}$ (wet wt) but the 95%ile value is above the FDA guideline. Average PCB concentrations in perch and older striped bass are still above 2 $\mu\text{g/g}$ (wet wt). Approximately 50% of the PCB concentration in fish during this period is associated with the Upper Hudson load. This is consistent with the preferential accumulation of higher chlorinated PCBs in fish and previous results showing that the Upper Hudson load is contributing approximately 50% of the PCB sediment response for the higher chlorinated homologues.

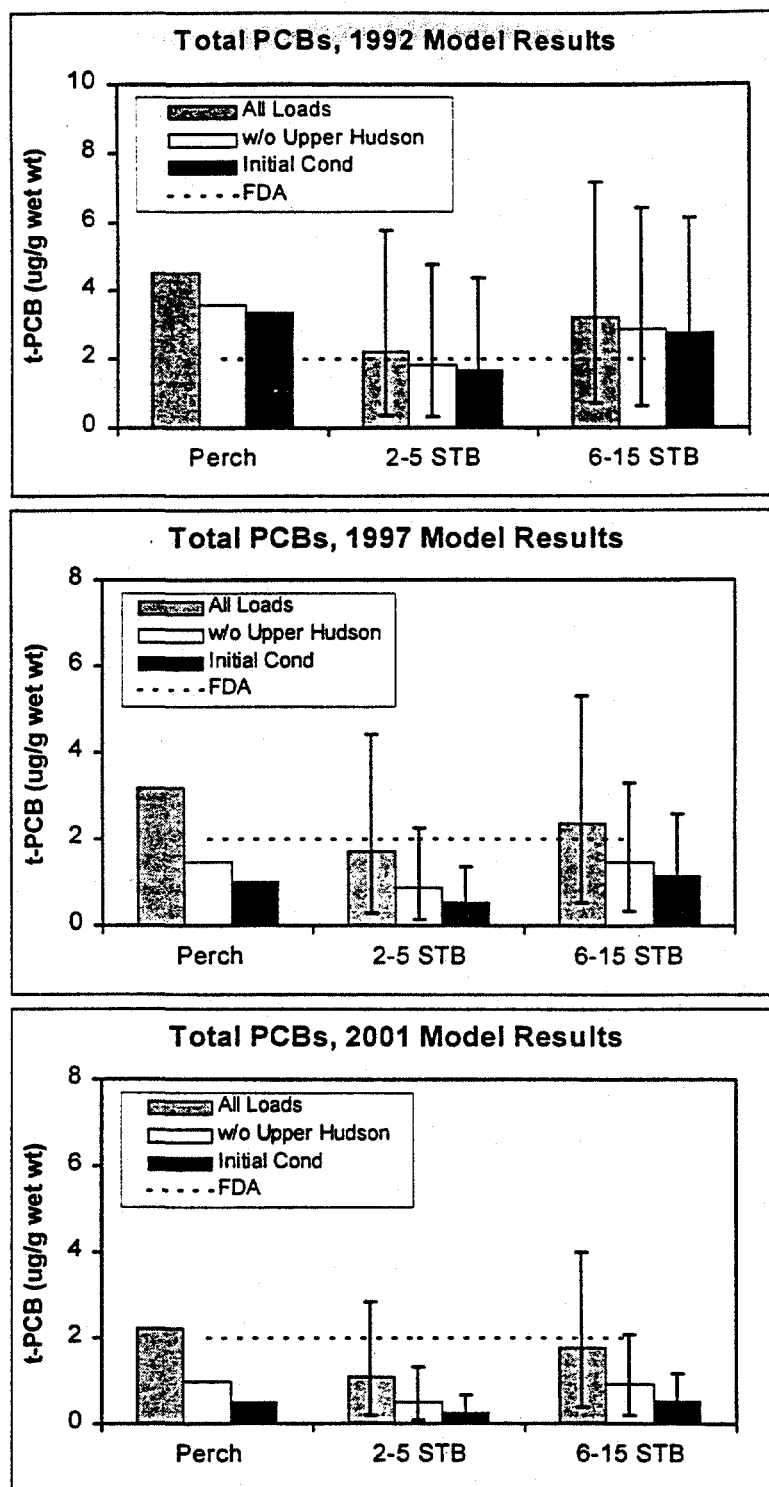


Figure 3-28. Model results for average (with 5 and 95%ile) PCB concentrations in perch and migrating striped bass for 1992, 1997, and 2001 due to (1) 1987 initial PCB concentrations in sediments; (2) 1987 initial conditions plus all PCB loads except the Upper Hudson load; and (3) 1987 initial conditions plus all PCB loads including the Upper Hudson load.

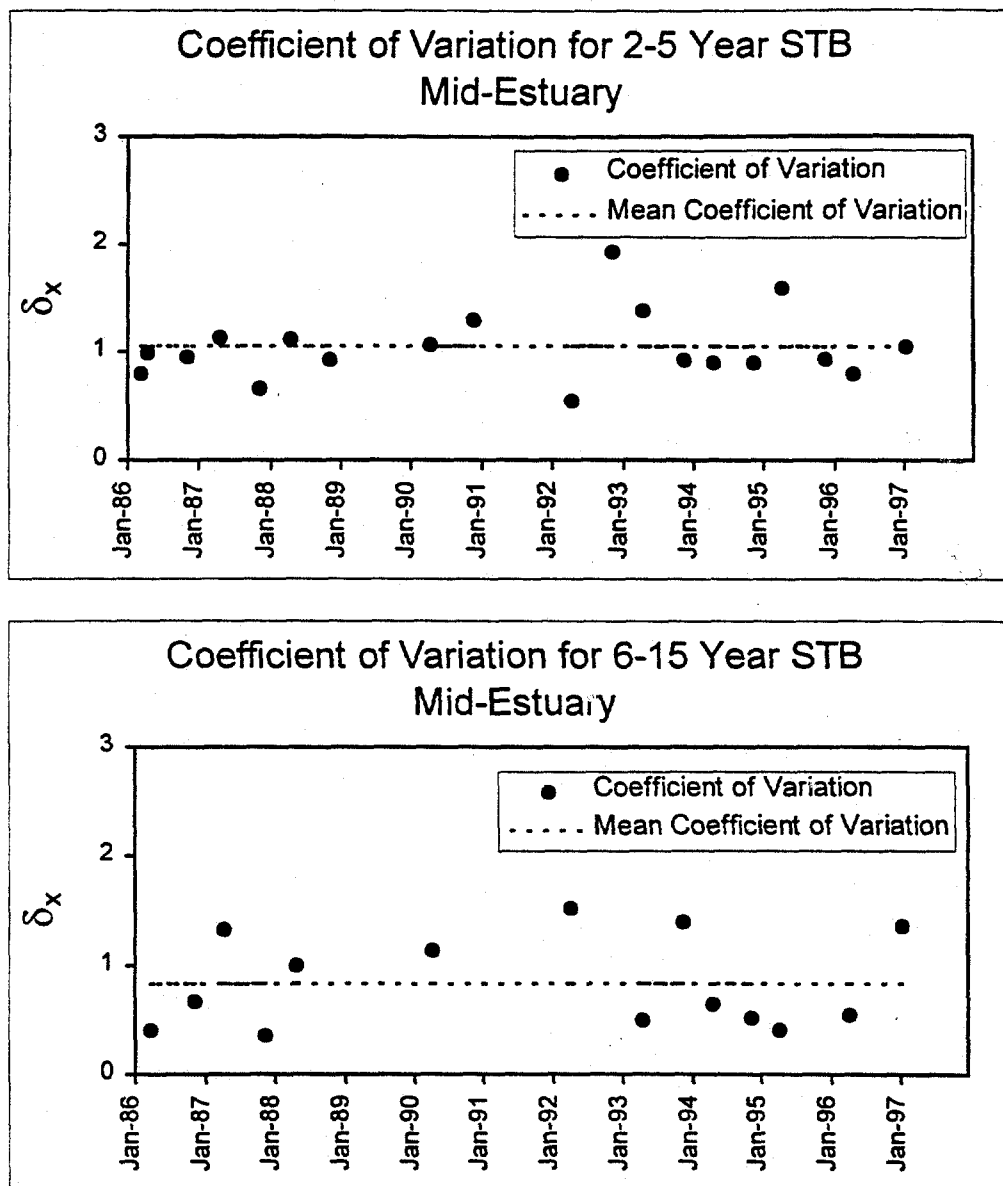


Figure 3-29. Coefficient of variation for total PCB concentrations in 2-5 and 6-15 year old striped bass caught in the mid estuary.

For the 2001 projection, average and 95%ile concentrations in the 2-5 year old and 6-15 year old striped bass are below the FDA limit. The 95%ile concentrations however are expected to still be above 2 $\mu\text{g/g}$ (wet wt), particularly for the older fish. The slower response in 6-15 year old striped bass to decreasing PCB exposure concentrations is likely due to slow depuration of the higher chlorinated homologues and smaller reduction in PCB concentrations by growth dilution in older fish.

Summary and Conclusions

The PCB homologue model of Thomann et al (Thomann et al. 1989) was restructured and now includes descriptions for seasonal hydrology and sediment transport. The model, which was calibrated by adjusting four modeling coefficients (k_v' , α , β , $k_{u\text{-phyto}}/k_{g\text{-phyto}}$), provides a good description of field data for PCB homologue concentrations in the water column, surface sediments, and perch, and for total PCB concentrations in striped bass. Key findings in our model evaluations included;

1. Seasonal hydrologic and sediment transport have a significant effect on the fate of PCB homologues. During the Spring high flow period, disproportionate amounts of PCBs typically enter the estuary and are accumulated in sediments or transported down to the harbor without significant volatile loss. This results in a relatively small longitudinal gradient in sediment PCBs on a μg PCB per g (dry wt) basis. During the Summer-Fall-Winter low flow period, volatilization becomes the primary removal pathway. As PCBs are lost from the water column during their transport through the estuary, they are replenished in the downstream sections by desorption from contaminated sediments that are continually resuspended by tidal motion. The net result is a significant loss of PCBs by volatilization during the low flow period without an appreciable decrease in the downstream PCB water column concentrations.
2. Increased POC concentrations in New York-New Jersey Harbor, which are associated with increased water column concentrations of sewage-derived and phytoplankton organic carbon, are effective in binding PCBs and transporting them to the sediments.
3. A rationale approach for calibrating the bioaccumulation coefficients (α , β , and $k_{u\text{-phyto}}/k_{g\text{-phyto}}$) was developed based on differences in PCB homologue behavior, time-variable exposure history in the tidal freshwater and mid estuary, and striped bass migration behavior.
4. Striped bass migration patterns have a large effects on PCB accumulation. Model calculations and field data suggest that PCB concentrations in migrating 2-5 striped bass increase by 1-2 $\mu\text{g/g}$ (wet wt) during their time in the mid estuary. Older striped bass tend to preferentially loss lower chlorinated homologues by depuration during their migration to less contaminated ocean waters. This results in a shift to higher chlorinated PCBs in the homologue distributions for the older striped bass.

5. Collaborative evidence from outliers in PCB striped bass distributions to chemical analysis of otoliths (Secor and Piccoli 1996) support the presence of a (predominately male) non-migrating striped bass subpopulation in the Hudson River Estuary. PCB concentrations are much higher in the non-migrating fish due to the continuous exposure to contaminated prey.
6. Average concentrations in 2-5 year old and 6-15 year old striped bass are expected to fall below the FDA limit of 2 $\mu\text{g/g}$ (wet wt) before the year 2001. In 2001, 95%ile concentration in 2-5 year old and 6-15 year old striped bass are expected to still exceed the FDA limit, particularly for the older fish. This is most likely due to slow depuration rates for higher chlorinated homologues and smaller growth dilution effects for the older fish.
7. The Upper Hudson load appears to contribute 60-80% of the PCB sediment contamination and 50% of the PCB perch and striped bass contamination in the tidal freshwater and mid estuary. These results are consistent with the lower fraction of higher chlorinated homologues in the Upper Hudson load, and the preferential accumulation of higher chlorinated homologues in fish.

Chapter 4

PCB, PAH, and Dioxin Contamination of Dredged Sediments from New York-New Jersey Harbor

Introduction

Testing requirements for the ocean disposal of dredged material from New York-New Jersey Harbor were revised in the early 1990s (USACE New York District/EPA Region II 1992) and now include protocols for amphipod toxicity testing and dioxin analyses (Suszkowski 1995). As a result, a larger portion of the dredged material from the harbor is considered unacceptable for unrestricted ocean disposal. The current classification of New York-New Jersey dredged material was previously given in Chapter 1 and includes: Category 1 (no bioaccumulation or toxicity and acceptable for unrestricted ocean disposal), Category 2 (bioaccumulation but no toxicity and limited to ocean disposal with capping), and Category 3 (toxicity and/or bioaccumulation and not acceptable for ocean disposal).

PCBs, PAHs, and dioxins are three of the contaminants considered as potential causative agents in the contamination of dredged material (Suszkowski 1995). The aim of our studies therefore was to develop a quantitative modeling approach that can be used to relate PCB, PAH, and dioxin contaminant loadings to specific dredged material endpoints (including the bioaccumulation of PCBs, PAHs, and dioxins in the test organisms *Macoma nasuta* and *Nereis virens*, and *Ampelisca abita* toxicity). The modeling approach, which was previously outlined in Figure 1-10, includes a chemical fate and bioaccumulation model component as described below.

Chemical fate modeling calculations, which include seasonal hydrodynamic, organic carbon, and sediment transport, chemical partitioning to POC and DOC, and volatilization, were previously described in Chapter 2. The model calculations were initially developed for PCB homologues and used to identify the effects of various PCB loadings on sediment contamination in Chapter 3. Similar calculations will be presented later in this chapter for PAHs and dioxin using the calibrated PCB homologue model with appropriate changes in contaminant loadings, initial conditions, and log K_{ow} values in the model input file.

Results from the chemical fate model are used in defining exposure concentrations for bioaccumulation model calculations. In this step, PCB, PAH, and dioxin concentrations in the test organisms, *Macoma nasuta* and *Nereis virens*, are determined by multiplying the sediment contaminant concentrations by chemical-specific Biota-Sediment-Accumulation Factors (BSAFs). Calculated contaminant concentrations in macoma and nereis can be compared directly to PCB, PAH, and dioxin tissue concentration guidelines for dredged material for New York-New Jersey Harbor (see Table 1-2). Finally, PAH accumulations, which can elicit a narcotic response in organisms, is tested as an indicator of increased organism toxicity. Evaluation of BSAFs for PCBs, PAHs, and dioxins is presented below

and is followed by a preliminary assessment of the bioaccumulative potential and toxicity of harbor sediments. The chapter concludes with model results for the effects of PCB, PAH, and dioxin loadings on contaminant accumulations in the dredged material test organisms, *Macoma nasuta* and *Nereis virens*.

Biota-Sediment-Accumulation Factors (BSAFs)

The accumulation of toxic contaminants in dredged material test organisms may be viewed as a dynamic process that depends on direct uptake from the water, food ingestion, depuration (from back diffusion, urine excretion and egestion of fecal matter) and metabolic transformation of contaminants within the organism. For *Macoma nasuta* and *Nereis virens*, contaminant transfer is expected to occur relatively fast (on the order of weeks) and steady state is often assumed. Under these conditions, bioaccumulation of contaminant in organisms may be expressed in terms of the biota-sediment-accumulation factor (BSAF):

$$BSAF = \frac{V_{lipid}}{\Gamma_{oc}} \quad (1)$$

where the BSAF is expressed as the ratio of contaminant concentration in the organism (V_{lipid}) in $\mu\text{g/g(lipid)}$ to contaminant concentration in the sediments (Γ_{oc}) in $\mu\text{g/g(oc)}$.

BSAFs for PCBs and PAHs were determined using 28-day bioaccumulation data from six Army Corps of Engineers, New York Region permit applications (Bayridge Reach A, Bayridge Reach B, Red Hook, Bronx River, Shoal Harbor, and Shark River). Sediment concentrations and tissue concentrations for macoma and nereis were measured for the 22 PCB congeners and 16 PAH compounds. Sediment concentrations were normalized to organic carbon content using measured total organic carbon (TOC) values. Lipid contents for the macoma and nereis however were not reported. Calculation of lipid-normalized organism concentrations were therefore made using literature values for lipid content (see Table 4-1). For analyses presented below, lipid contents of 0.59% and 2.1% were assigned for macoma and nereis, respectively, based on data from (Gardiner et al. 1996).

Correlations plots for chemical accumulations in macoma and nereis are shown in Figure 4-1 for the 22 PCB congeners and 16 PAH compounds. For PCBs, the concentrations in macoma are approximately two times greater than the concentrations in nereis. This may be indicative of differences in uptake efficiencies, some degree of PCB metabolism in nereis, or variations in actual and assumed lipid contents. For PAHs, chemical accumulations are an order of magnitude greater in macoma than in nereis and are likely due to greater rates of PAH metabolism in nereis.

COE Permit Data for Six Stations in NY-NJ Harbor

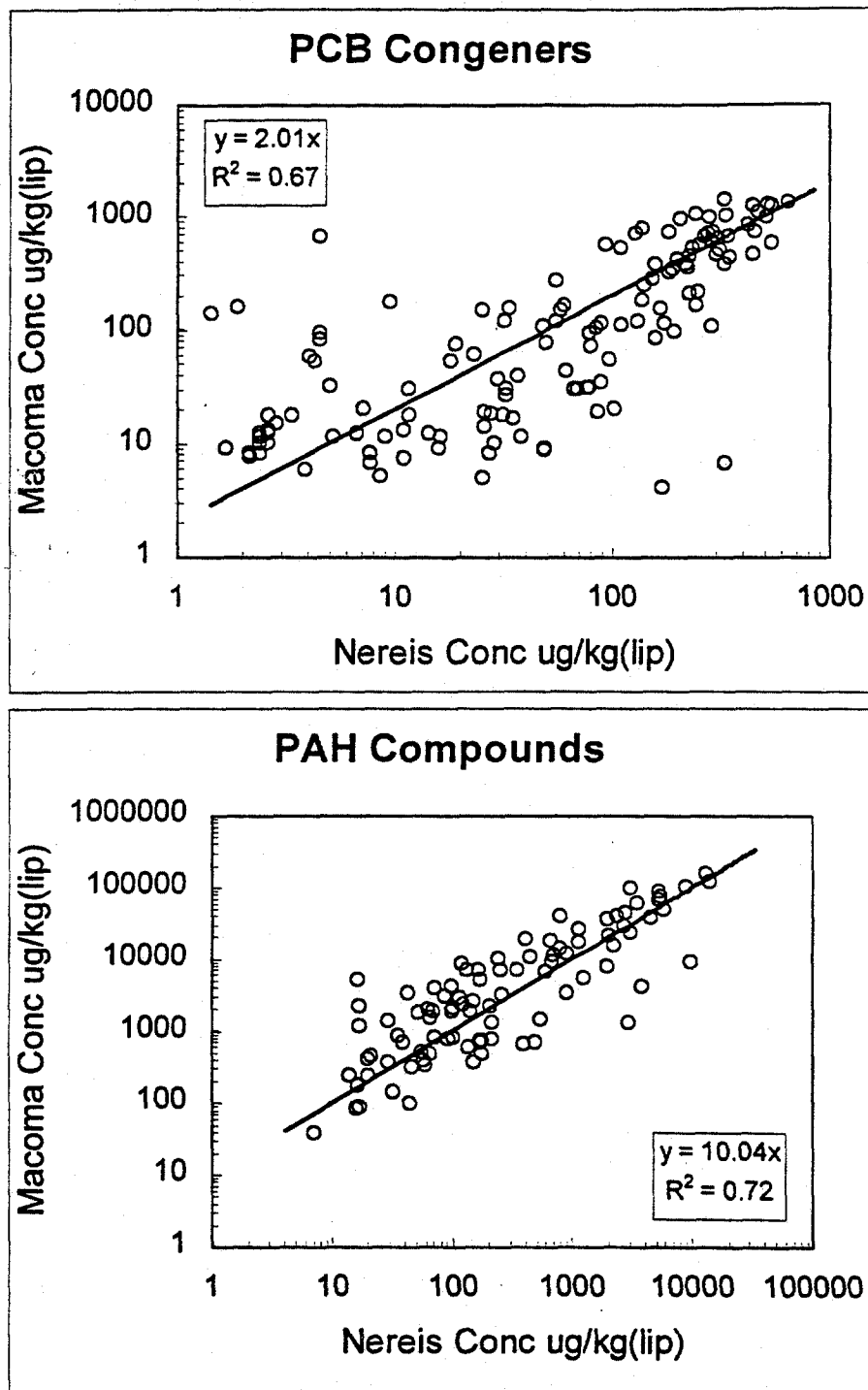


Figure 4-1. Correlations for chemical accumulations in *Macoma nasuta* and *Nereis virens* for 22 PCB congeners and 16 PAH compounds. [Data from six Army Corps of Engineers, New York Region permit applications.]

Table 4-1. Summary of Lipid Fractions for <i>Macoma nasuta</i> and <i>Nereis virens</i> .			
Organism	% Dry Weight	Lipid Content (% wet wt.)	Reference
<i>Macoma nasuta</i>		0.4 (0.16- 0.8) ¹	(Parkerton 1993)
	15	0.58, 0.59, 0.60	(Gardiner et al. 1996)
		2.2 (1.2 –3.1) ²	(McFarland 1995)
<i>Nereis virens</i>		1.3 (0.1 – 5.7) ¹	(Parkerton 1993)
	12.6	2.0, 2.1, 2.2	(Gardiner et al. 1996)
1. Median and range 2. Total dichloromethane-extractable lipids; average and range			

Correlations for chemical accumulations in the test organisms and concentrations in sediments is shown in Figure 4-2. A general trend of increasing chemical accumulation in the test organisms with increasing sediment concentrations is observed for both PCBs and PAHs. Differences between for trend lines for macoma and nereis, particularly for PAHs, again suggest greater rates of PAH metabolism in nereis as noted above. A scatter of data points about the trend lines is also apparent and may be due in part to a slow attainment of steady state, particularly for the higher K_{ow} compounds, and/or variability in bioavailable and non-bioavailable forms of chemical in sediments (e.g., PAHs in soot particles).

A closer look at PCB congener- and PAH compound-specific behavior is shown in Figure 4-3 and Figure 4-4, with computed BSAFs plotted as a function of $\log K_{ow}$ values. As shown, there is roughly an order of magnitude variation in BSAF values for individual PCB congeners and PAH compounds. Differences in median BSAFs as a function of $\log K_{ow}$ are smaller suggesting a greater variation in BSAFs from site to site than from chemical to chemical. For PCBs, BSAF values averaged approximately 0.4 kg(oc)/kg(lipid) for macoma and 0.2 kg(oc)/kg(lipid) for nereis. For PAHs, BSAF values were lower and averaged approximately 0.10 kg(oc)/kg(lipid) for macoma and 0.01 kg(oc)/kg(lipid) for nereis.

A similar evaluation of BSAFs for dioxins and furans was conducted using October 1994 New York Bight sediment and invertebrate data from Battelle (Battelle 1996). Dioxins and furans that were included in the study are listed in Table 4-2. Before analysis, sediment concentrations were normalized to organic carbon content values and invertebrate concentrations were normalized to lipid content using measured TOC and lipid concentrations.

COE Permit Data for Six Stations in NY-NJ Harbor

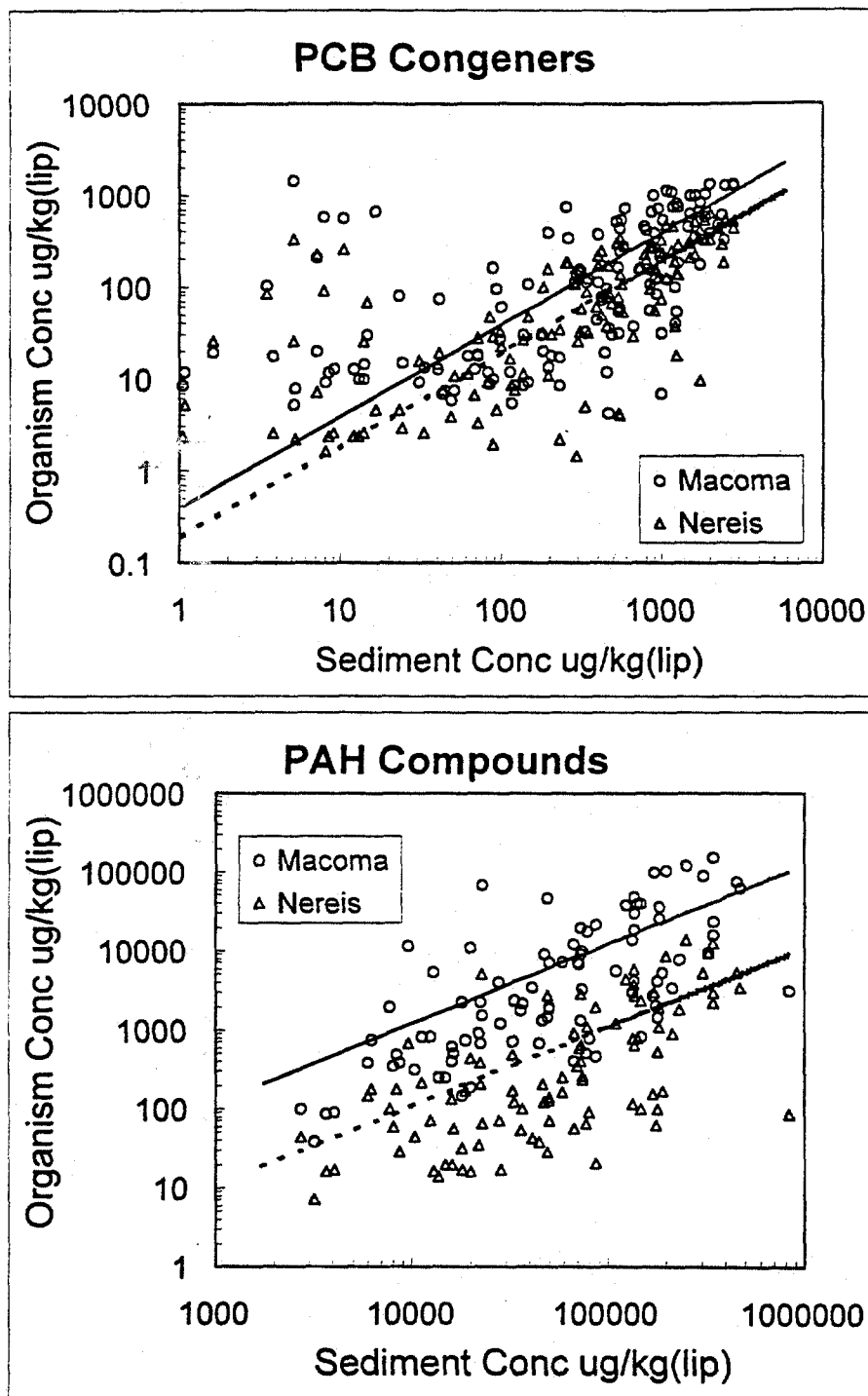


Figure 4-2. Correlations for chemical accumulations in *Macoma nasuta* and *Nereis virens* with concentrations in sediments for 22 PCB congeners and 16 PAH compounds. [Data from six Army Corps of Engineers, New York Region permit applications.]

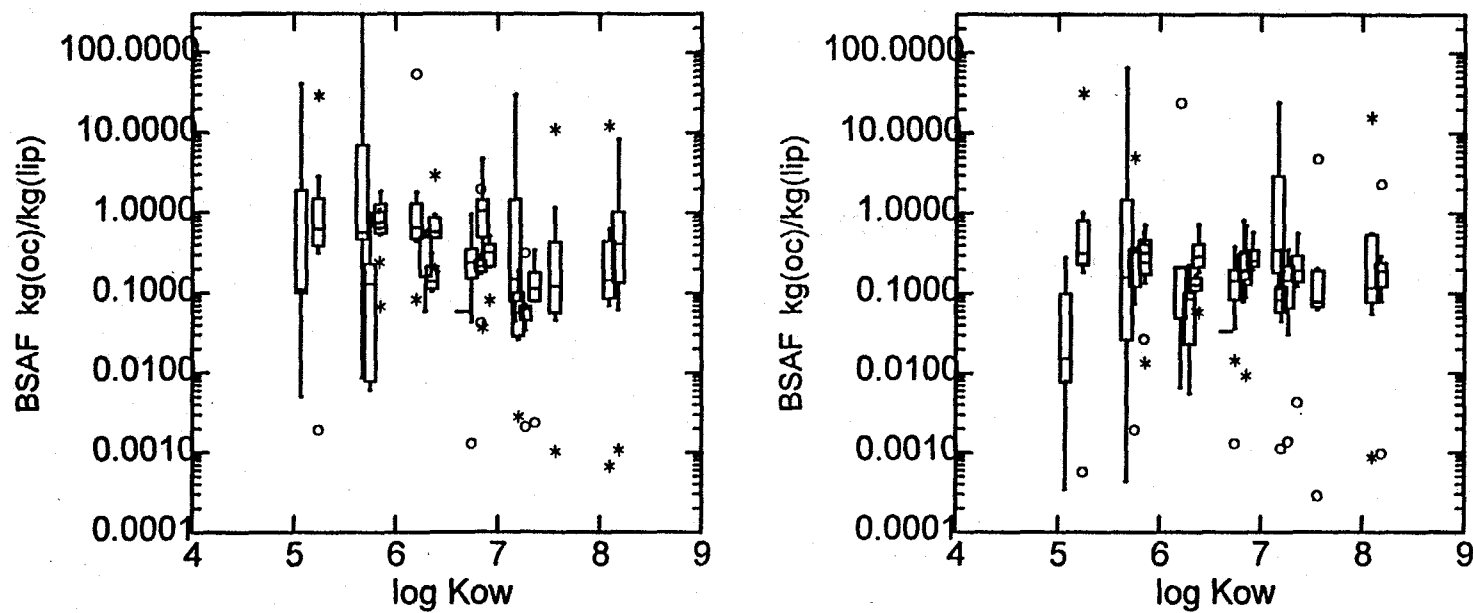


Figure 4-3. BSAFs for *Macoma nasuta* and *Nereis virens* as a function of log K_{ow} for 22 PCB congeners. [Data from six Army Corps of Engineers, New York District permit applications.]

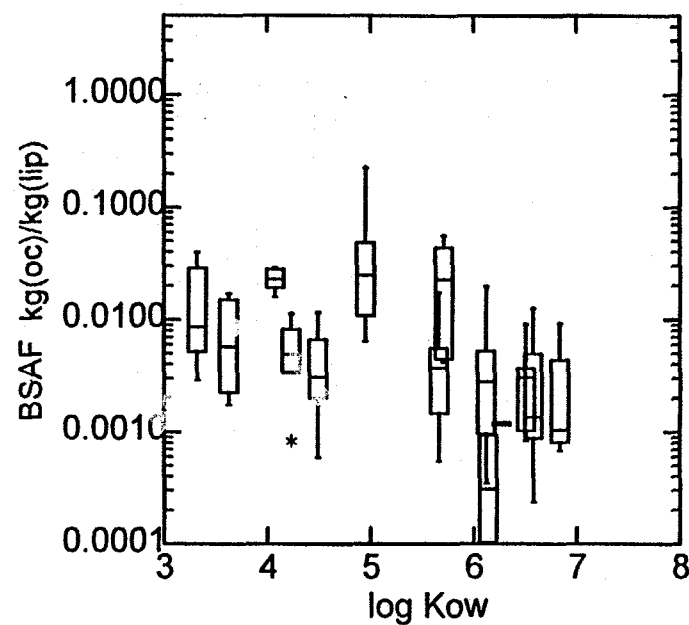
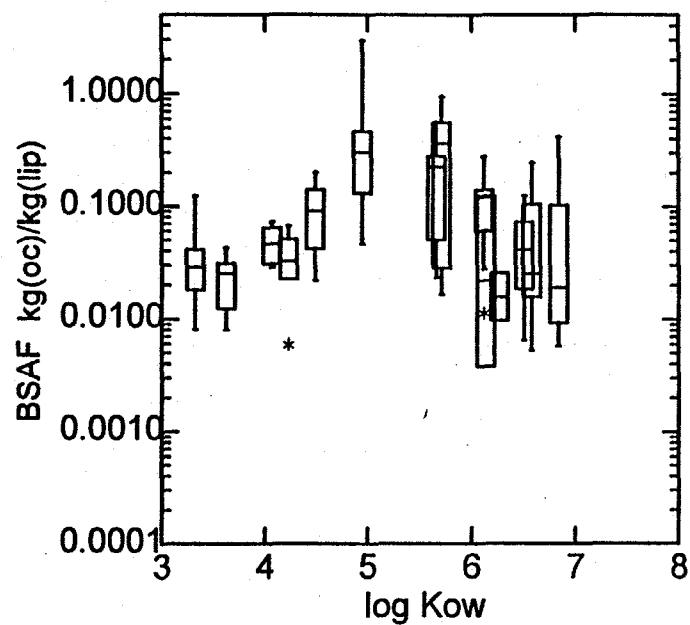


Figure 4-4. BSAFs for *Macoma nasuta* and *Nereis virens* as a function of $\log K_{ow}$ for 16 PAH compounds. [Data from six Army Corps of Engineers, New York District permit applications.]

**Table 4-2. log Kow and Computed Invertebrate BSAFs for Dioxins and Furans.
(Invertebrate and Sediment Data from the New York Bight (Battelle 1996).)**

Dioxins/Furans	log Kow	BSAF ⁽⁴⁾ kg(oc)/kg(lipid)
1,2,3,4,6,7,8 HpCDD	8.2 ⁽¹⁾	
1,2,3,4,7,8 HxCDD	7.8 ⁽¹⁾	below LOD
1,2,3,6,7,8 HxCDD	7.8 ⁽¹⁾	below LOD
1,2,3,7,8,9 HxCDD	7.8 ⁽¹⁾	below LOD
1,2,3,7,8 PeCDD	7.5 ⁽¹⁾	
2,3,7,8 TCDD	7.0 ⁽¹⁾	
OCDD	8.6 ⁽¹⁾	
1,2,3,4,6,7,8 HpCDF	6.6 ⁽²⁾	
1,2,3,4,7,8,9 HpCDF	6.6 ⁽²⁾	below LOD
1,2,3,4,7,8 HxCDF	6.6 ⁽³⁾	below LOD
1,2,3,6,7,8 HxCDF	6.6 ⁽³⁾	below LOD
1,2,3,7,8,9 HxCDF	7.0 ⁽³⁾	below LOD
1,2,3,7,8 PeCDF	6.8 ⁽³⁾	below LOD
2,3,4,6,7,8 HxCDF	6.6 ⁽³⁾	below LOD
2,3,4,7,8 PeCDF	6.2 ⁽³⁾	below LOD
2,3,7,8 TCDF	5.8 ⁽³⁾	
OCDF	8.8 ⁽³⁾	

⁽¹⁾ log Kow values from Cook et al (Cook et al. 1993).

⁽²⁾ log Kow values assumed

⁽³⁾ log Kow values from Morrison et al (Morrison et al. 1996)

⁽⁴⁾ BSAF values calculated for invertebrates in the New York Bight (data from Battelle (Battelle 1996)). BSAFs below LOD are listed for chemical concentrations below the limit of detection [of approximately 0.5 ng/kg(wet wt)] in invertebrates.

A correlation plot for chemical accumulations in invertebrates and concentrations in sediments is shown in Figure 4-5. For this data set, approximately 15% of the sediment samples and 50% of the invertebrate samples were at or below the limit of detection (LOD). For samples above the LOD, a general trend of increasing chemical accumulation in invertebrates with increasing sediment concentrations is observed. Congener-specific dioxin and furan BSAFs are given in Figure 4-6. Note that, for the limited number of congeners that are included on the figure, BSAF values appear to decrease for the higher K_{ow} congeners ($\log K_{ow} > 7$). This may be due to lower chemical assimilation efficiencies for high K_{ow} congeners. For the remaining congeners not included in Figure 4-6, invertebrate concentrations were at or below the limit of detection.

BSAFs for PCBs and PAHs calculated from the Army Corps of Engineers permit data are plotted as geometric mean values in Figure 4-7. These results are compared to the BSAF free energy relationship given by DiToro and McGrath (submitted):

$$BSAF = \frac{V_{lipid}}{\Gamma_{oc}} = K_{ow}^{-0.038} \quad (2)$$

Equation 23 provides a reasonable description of BSAFs for PCB (Figure 4-7) and also for dioxin congeners (not shown). The relationship given by DiToro and McGrath (DiToro and McGrath in prep), which does not consider the effects of chemical metabolism, clearly overpredicts BSAFs for PAHs that were computed from Army Corps of Engineers permit data. DiToro and McGrath (DiToro and McGrath in prep) however recommend using the higher BSAF estimates from their free energy relationship in calculating narcotic effects associated with the accumulation of PAHs in organisms. This is based on: (1) radio labeled benzo(a)pyrene (BaP) uptake experiments of Driscoll and McElroy (Driscoll and McElroy 1996) which showed that the reduction in the BSAF value due to BaP metabolism was largely compensated by the accumulation of BaP metabolites in the three polychaetes studied, and (2) the assumption that both the PAH parent compounds and their metabolites contribute equally to narcotic effects. PCB, PAH, and dioxin calculations performed in the remainder of the chapter are therefore based on the BSAF relationship given by Equation 23.

Preliminary Assessment of NY-NJ Harbor Dredged Material

A summary of current tissue concentration guidelines used in assessing the PCB, PAH, and dioxin bioaccumulation in 28-day exposure studies for dredged material from New York-New Jersey Harbor are given in Table 4-3. Based on these guidelines, equivalent sediment concentrations were calculated as follows:

$$\text{Equiv. Sediment Conc.} = \frac{\text{Tissue Conc. Guideline}}{f_{lipid} \text{ BSAF}} \quad (3)$$

Dioxin/Furan Concentrations in Invertebrate and Sediments

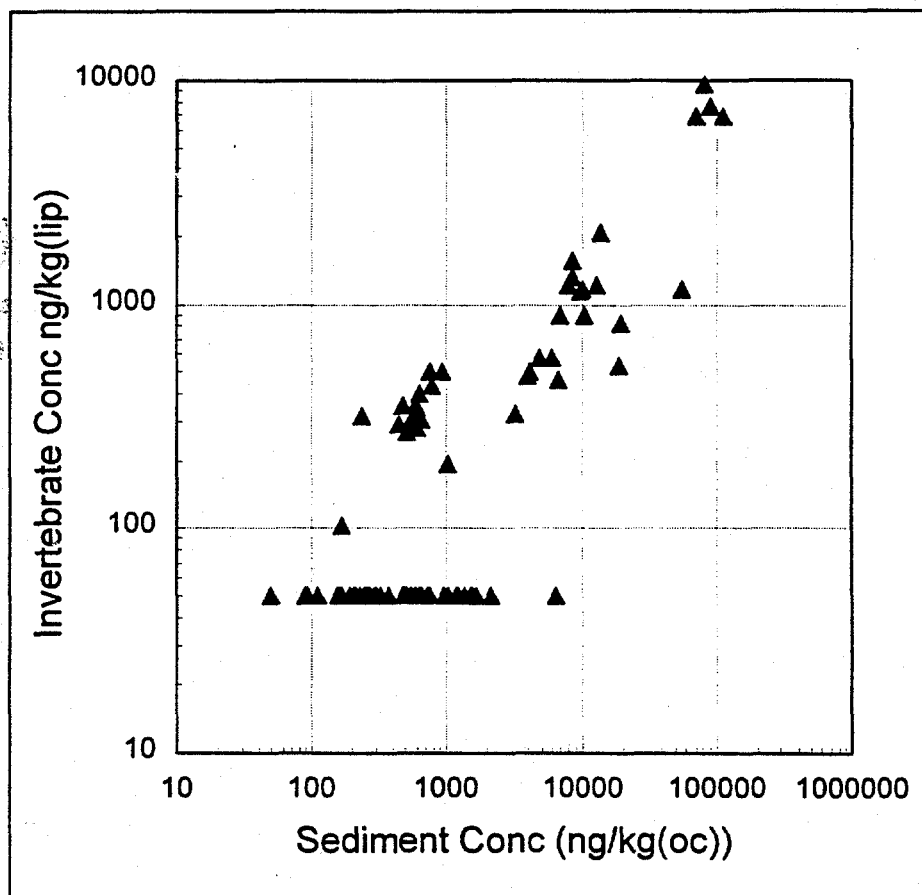


Figure 4-5. Correlation of dioxin/furan accumulations in invertebrates and concentrations in sediments. [Data from the New York Bight Mud Dump Site (Battelle 1996).]

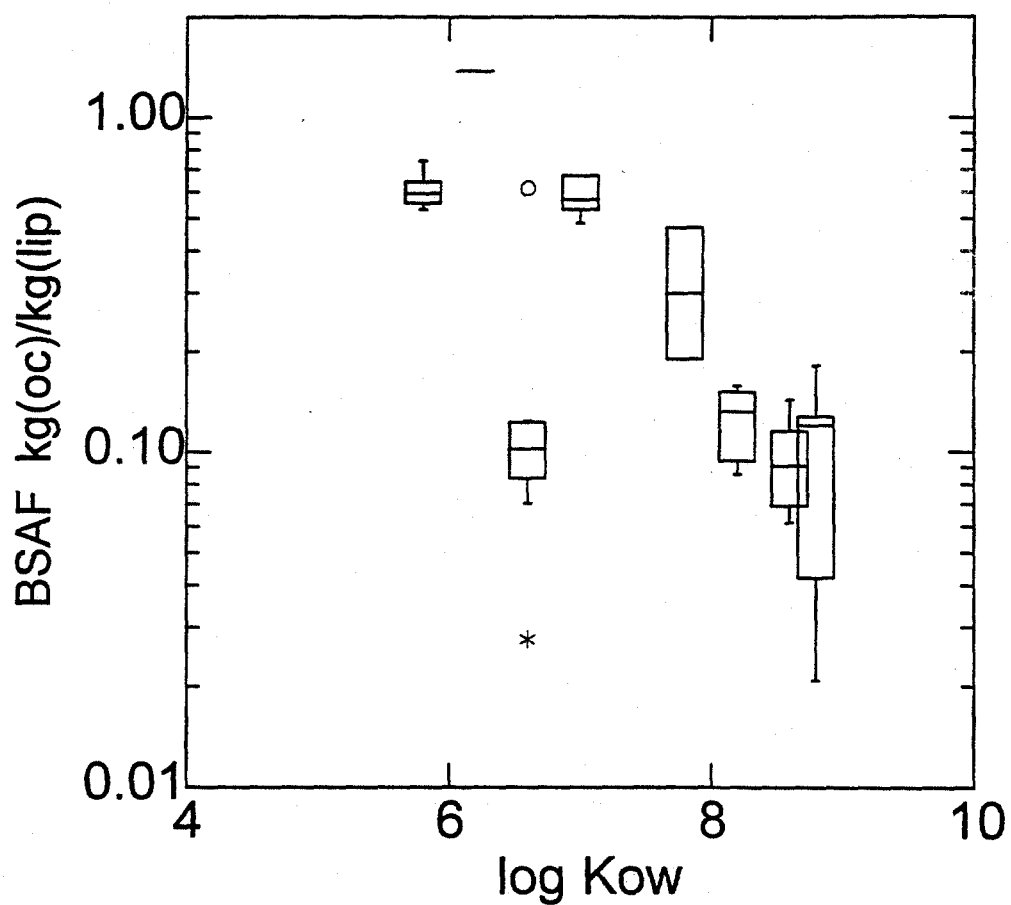


Figure 4-6. Congener-specific dioxin and furan BSAFs as a function of log K_{ow}. [Data from the New York Bight Mud Dump Site (Battelle 1996).]

Summary of BSAFs for PCBs and PAHs (Reported as Geometric Means)

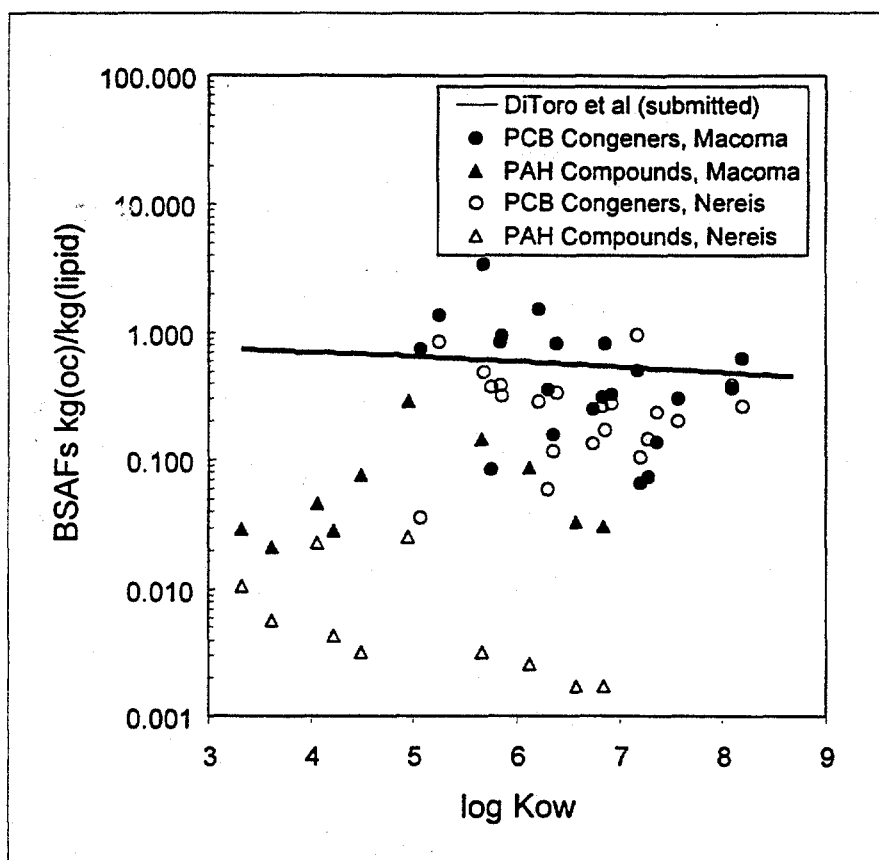


Figure 4-7. Summary of chemical-specific BSAFs for *Macoma nasuta* and *Nereis virens* as a function of log Kow for 22 PCB congeners and 16 PAH compounds. [Computed BSAFs are reported as geometric means from analysis of six Army Corps of Engineers, New York Region permit applications.] BSAF relationship of DiToro and McGrath (in prep) is given for comparison.

Table 4-3. PCB, PAH, and Dioxin Tissue Concentration Guidelines (and Equivalent Sediment Concentrations) for Dredged Material from New York-New Jersey Harbor.		
	Tissue Concentrations	Equivalent Sediment ⁽⁴⁾ Concentration Guidelines
PCBs	0.1 µg/g(wet wt) clam ⁽¹⁾ 0.4 µg/g(wet wt) worm ⁽¹⁾	29 µg/g(oc) clam 32 µg/g(oc) worm
PAHs	40 µg/g(wet wt) ⁽²⁾	10,400 µg/g(oc) clam 2,900 µg/g(oc) worm
Dioxins: (2,3,7,8-TCDD)	1 pg/g(wet wt) ⁽¹⁾ 10 pg/g(wet wt) ⁽³⁾	310 pg/g(oc) clam 88 pg/g(oc) worm 3,100 pg/g(oc) clam 880 pg/g(oc) worm
⁽¹⁾ Regional matrix values for Category 1 dredged material (Pabst and Tavolaro 1998). ⁽²⁾ Ecological non specific narcosis effects levels (Pabst and Tavolaro 1998). ⁽³⁾ Regional matrix values for Category 2 dredged material (Rossman 1998). ⁽⁴⁾ Equivalent sediment concentrations are calculated from specified tissue concentrations, lipid fractions of 0.0059 for clams (<i>Macoma nasuta</i>) and 0.021 for worms (<i>Nereis virens</i>) from data of (Gardiner et al. 1996), and Biota-Sediment Accumulation Factors (BSAFs) using Eq 23: for PCBs (log Kow ≈6), BSAF = 0.59; for PAHs (log Kow ≈5), BSAF = 0.65; for TCDD (log Kow≈7), BSAF = 0.54.		

where the BSAF is given by Equation 23 and is expressed in units of kg(oc)/kg(lipid), and lipid fractions for clams (*Macoma nasuta*) and for worms (*Nereis virens*) are taken as 0.0059 and 0.021, respectively, based on data of (Gardiner et al. 1996).

A preliminary assessment of PCB sediment contamination in the harbor was performed using the 1993-94 R-EMAP data for chemical concentrations in surface (0-2 cm) sediments (Adams et al. 1998). In this analysis, field data for total PCBs was taken as two times the summation of 20 measured PCB congeners (Adams et al. 1998) and normalized to the organic content of the sediments. Resulting concentrations are given in Figure 4-8 and show PCB sediment concentrations varying by over two orders of magnitude in the harbor region with medians of 0.8 µg/g(oc) in the New York Bight and 40 µg/g(oc) in Newark Bay. Equivalent sediment concentration for PCBs [given in Table 4-3 as

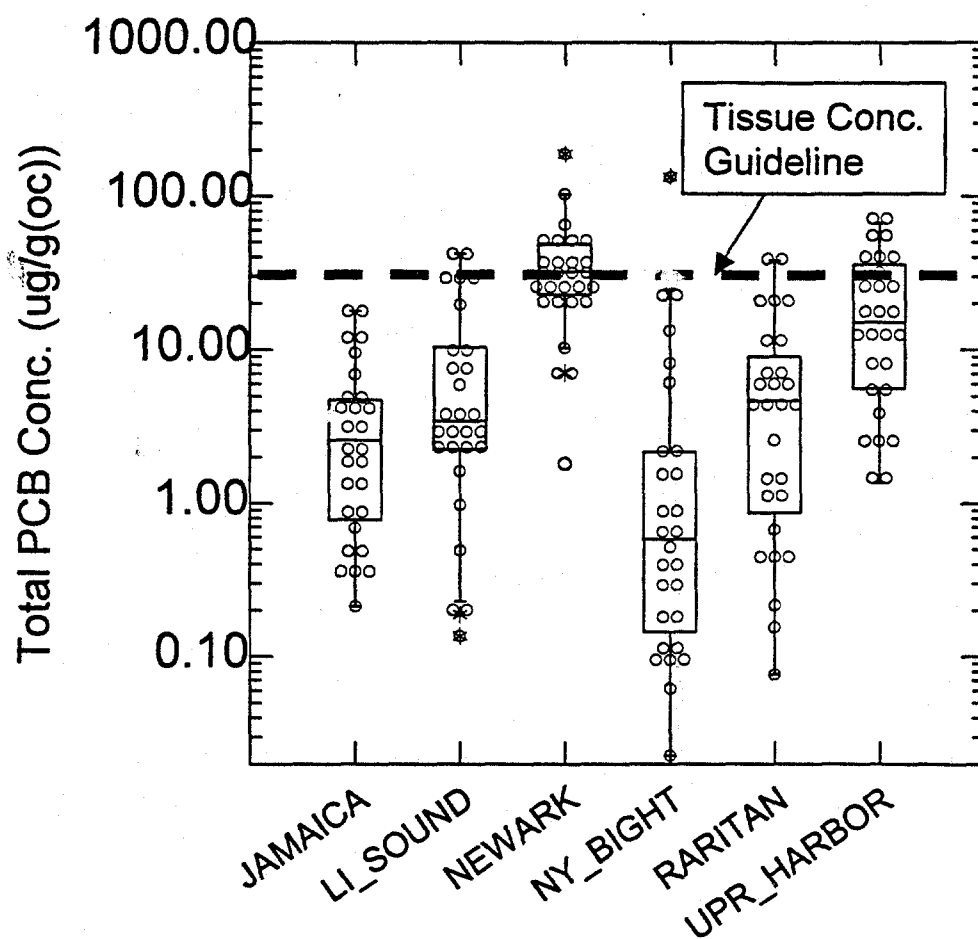


Figure 4-8. Preliminary assessment of PCB sediment contamination in New York-New Jersey Harbor. [Data from 1993-94 R-EMAP study (Adams et al. 19980.)]

approximately 30 µg/g(oc)] is included on Figure 4-8 as an indicator of potential exceedance for bioaccumulation of PCBs in the dredged material test organisms. As shown, 28 out of 168 samples (or 17%) exceeded the equivalent sediment guideline and are not expected to meet requirements for Category 1 dredged material. The largest number of exceedances occurred in Newark Bay followed by Upper New York Harbor.

A compilation of field data for the most toxic dioxin congener [2,3,7,8 tetra-chloro-dibenzo-para-dioxin (TCDD)] in surface sediments is shown in Figure 4-9. High levels of TCDD are found in lower Passaic River and are attributed to past discharges of TCDD from an industrial site in Newark where chlorinated phenols were produced (Bopp et al. 1991). The highest levels of TCDD are found in the lower Passaic River with median concentrations of 8,000 pg/g(oc) and decrease to approximately 600 pg/g(oc) in the New York Bight. Equivalent TCDD sediment concentrations for Category 1 and Category 2 dredged material (see Table 4-3) are included in Figure 4-9 for comparison. According to this analysis, nearly all of the samples exceed the Category 1 equivalent TCDD sediment concentrations for *Nereis virens* and *Macoma nasuta*. Many of the samples would also fail to meet requirements for Category 2 dredged material with samples in the lower Passaic River and parts of Newark Bay exceeding our calculated Category 2 TCDD sediment guidelines for *Nereis nasuta* by roughly an order of magnitude.

An assessment of PAH contamination in harbor sediments was also performed using the 1993-94 R-EMAP surface sediment data (Adams et al. 1998). Total PAH concentrations (which were taken as the summation of 23 PAHs and normalized to the organic carbon content of the sediments) are shown in Figure 4-10. The data show large spatial variations with medians of 20 µg/g(oc) in the New York Bight and 800 µg/g(oc) in Upper New York Bay. Equivalent PAH sediment concentration guidelines of 10,400 µg/g(oc) for *Macoma nasuta* and 2,900 µg/g(oc) for *Nereis virens* (Table 4-3) are also included in Figure 4-10 for comparison. As shown, 5 out of 168 samples (or 3%) exceed the equivalent sediment guideline for *Nereis virens* and 0 out of 168 samples exceed the guideline for *Macoma nasuta*.

Of the five sediment samples showing an exceedance of the PAH guideline for *Nereis virens* (see Figure 4-11), two samples also showed low ampelisca survival rates. Of the three remaining samples, two were very close to the PAH guideline for nereis (and high ampelisca survival rates may merely reflect differences in sensitivity of ampelisca and nereis to PAH narcotic effects); the other sample contained high concentrations of higher molecular weight PAHs (and bioavailability of the compounds may have been limited by solubility constraints). Twenty two sediment samples with total PAH concentrations below the narcosis-based guideline exhibited less than 80% ampelisca survival rates, strongly suggesting that the present narcosis-based guideline is too high or that other causative agents of ampelisca toxicity are present, particularly in Newark Bay sediments. Since PAH narcosis guidelines alone do not sufficiently describe observed instances of ampelisca toxicity, the following discussions of model results will focus on tissue concentration guidelines given in Table 4-3.

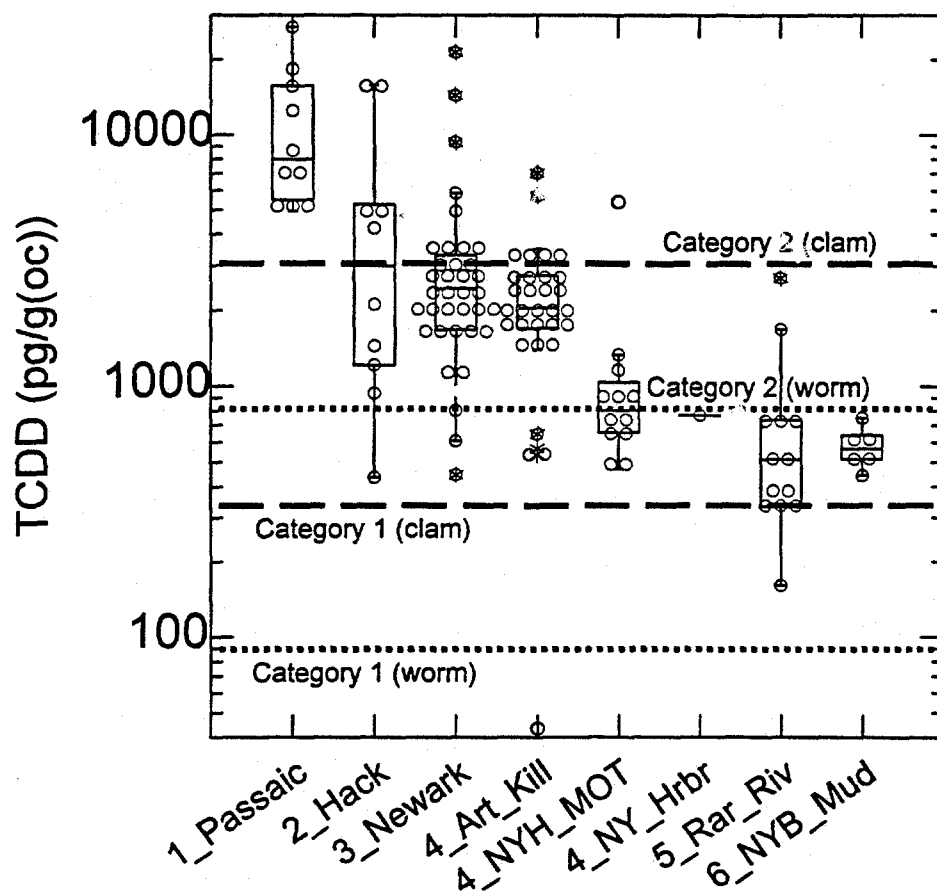


Figure 4-9. Preliminary assessment of dioxin [2,3,7,8 tetra-chloro-dibenzo-para-dioxin (TCDD)] sediment contamination in New York-New Jersey Harbor. [Data from Long et al. (1995) for the Passaic, Hackensack, Newark Bay, Arthur Kill, and NY-NJ Harbor; data from Army Corps of Engineers, New York Region permit applications for NY-NJ Harbor Military Ocean Terminal; data from Battelle (1996) for the NY Bight Mud Dump Site.]

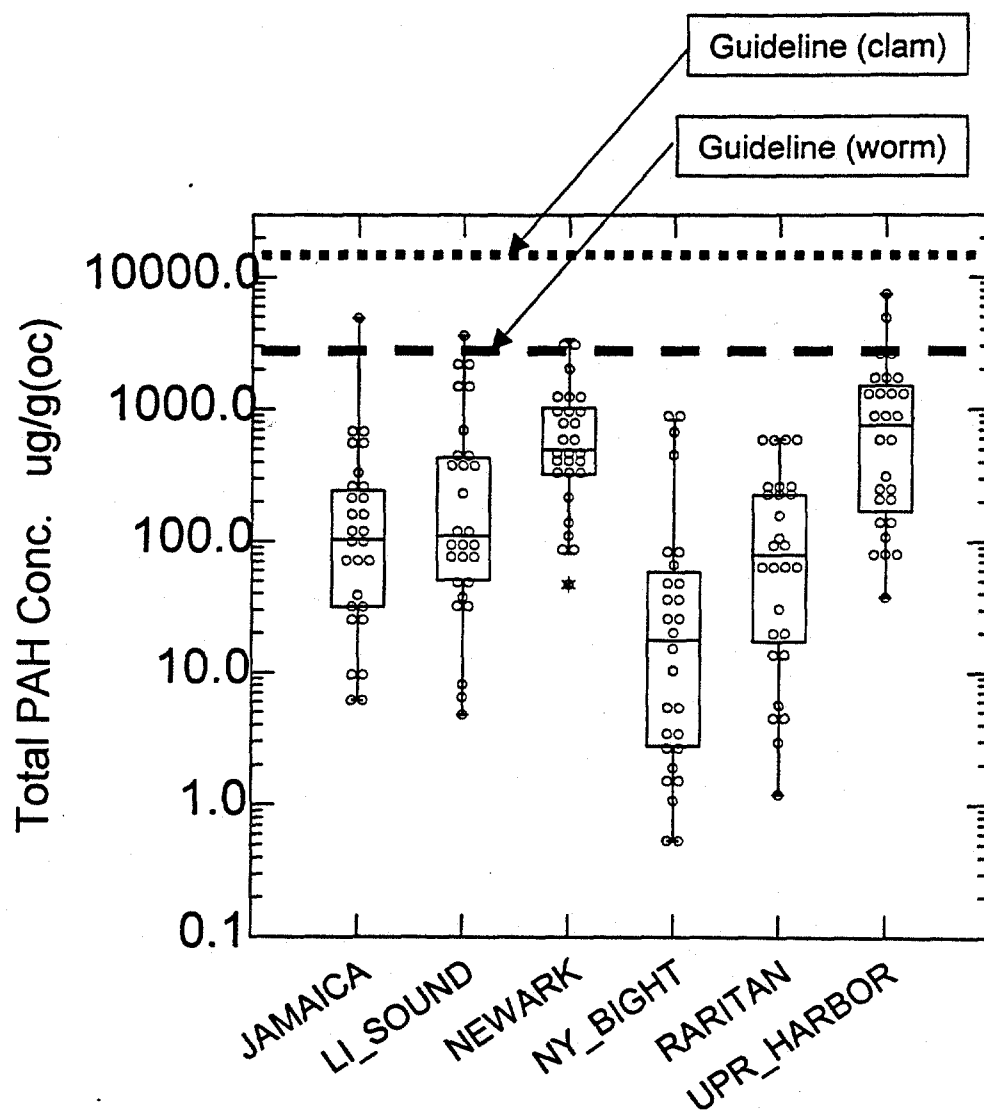


Figure 4-10. Preliminary assessment of PAH sediment contamination in New York-New Jersey Harbor. [Data from 1993-94 R-EMAP study (Adams et al. 1998).]

Ampelisca Survival versus Total PAH Concentration

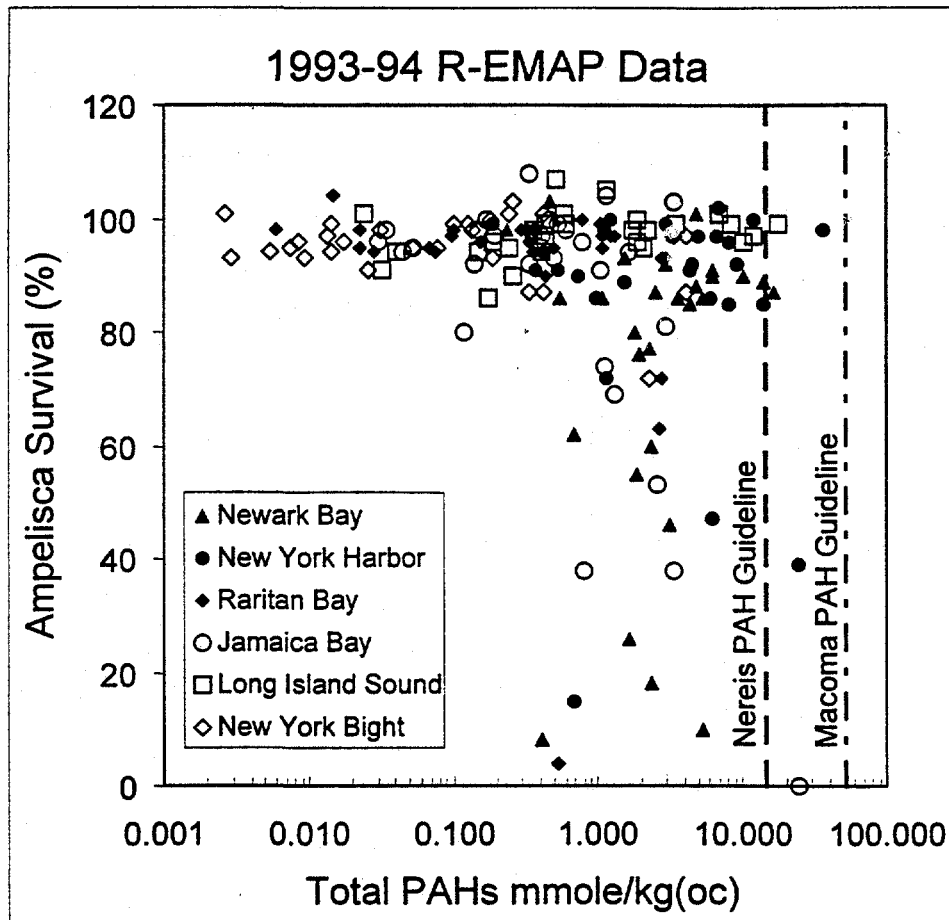


Figure 4-11. Comparison of *Ampelisca abita* survival rates to total PAH sediment contamination in New York-New Jersey Harbor. [Data from 1993-94 R-EMAP study (Adams et al. 1998).]

PCB Model Results

Chemical fate model calculations for PCBs, which were previously described in Chapters 2 and 3 of the report, were based on chemical loading from the Upper Hudson, the Mohawk River, New Jersey tributaries, wastewater treatment plants (WWTPs), combined sewer overflows (CSOs), stormwater outfalls (SWOs), and the atmosphere. Model results for total PCB concentrations in harbor sediments are shown in Figure 4-12 and compare reasonably well to average concentrations from the 1993-94 R-EMAP surface sediment data (Adams et al. 1998). Note that the large range in concentrations for the R-EMAP data is related to the random sampling design of the R-EMAP monitoring program which provides sediment samples from both high deposition zones with recently deposited PCBs and low deposition zones with aged deposits.

Model results for present (1999) PCB contamination levels in the harbor surface sediments are presented in the top panel of Figure 4-13. Since dredged material testing is based on homogenized sediments for the full project depth (typically given as one to several meters), average PCB concentrations for ten years (1989-99) of accumulated sediments are also included in the figure. As shown, PCB concentrations for the depth-averaged sediments are 30-100% greater than surface sediment concentrations due to higher PCB loadings in the 1980s and early 1990s. PCB concentrations in the dredged material test organisms, *Macoma nasuta* and *Nereis virens*, were calculated for the depth-averaged sediments as follows:

$$v_{\text{wet wt}} = f_{\text{lipid}} v_{\text{lipid}} = f_{\text{lipid}} \text{BSAF} \frac{\Gamma}{f_{\text{oc}}} \quad (4)$$

where $v_{\text{wet wt}}$ and v_{lipid} are the contaminant concentrations in the organism on a wet weight [$\mu\text{g/g(wet wt)}$] and lipid normalized basis [$\mu\text{g/g(lipid)}$]; f_{lipid} is the lipid fraction of the organism [$\text{g(lipid)}/\text{g(wet wt)}$]; BSAF is the biota-sediment-accumulation factor [$\text{g(oc)}/\text{g(lipid)}$] and is taken as 0.59 (based on Eq 23 and an average PCB $\log K_{\text{ow}}$ of 6); Γ is the contaminant concentration in sediment [$\mu\text{g/g(dry wt)}$]; and f_{oc} is the organic carbon fraction of the sediments [$\text{g(oc)}/\text{g(dry wt)}$]. Calculated PCB responses in macoma and nereis are given in the bottom panel of Figure 4-13, and show that average accumulations in the test organisms are below dredged material tissue guidelines for PCBs for all regions of the harbor. Based on the variability in PCB sediment concentrations (see Figure 4-8) however some exceedances in the PCB tissue guideline are likely, particularly for Newark Bay.

PAH Model Results

Chemical fate model calculations for PAHs were performed using the PCB homologue model and making appropriate adjustment for $\log K_{\text{ow}}$, and PAH loadings in the model input file. Because of large gaps in available data for PAH loadings, PAH

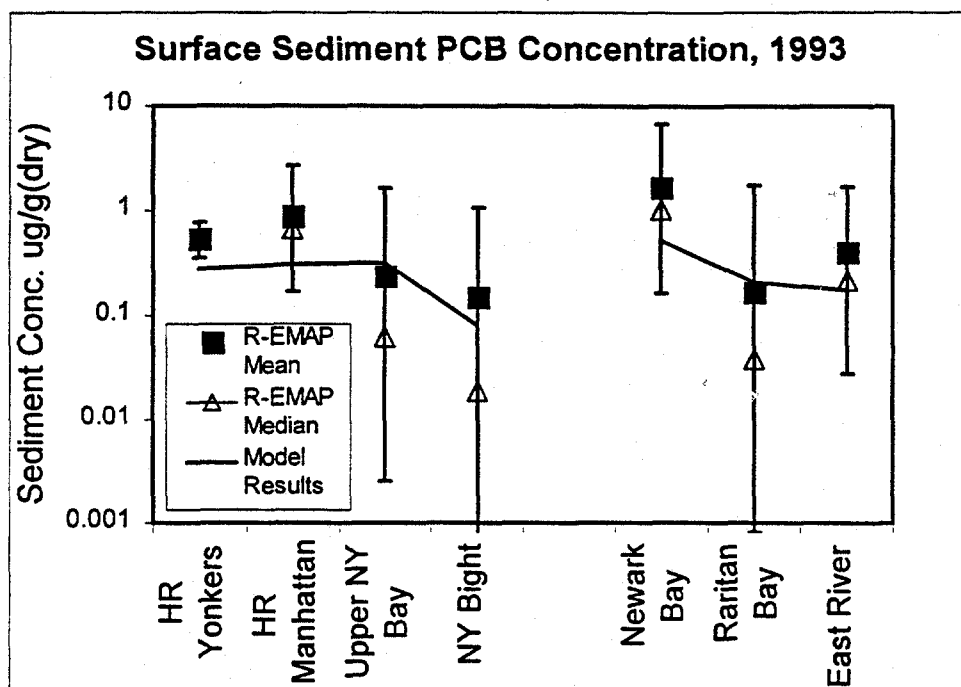


Figure 4-12. Model-field data comparison for 1993-94 total PCB surface sediment concentrations in New York-New Jersey Harbor. [Data from 1993-94 R-EMAP study (Adams et al. 1998) where total PCBs are taken as two times the summation of 20 congeners.]

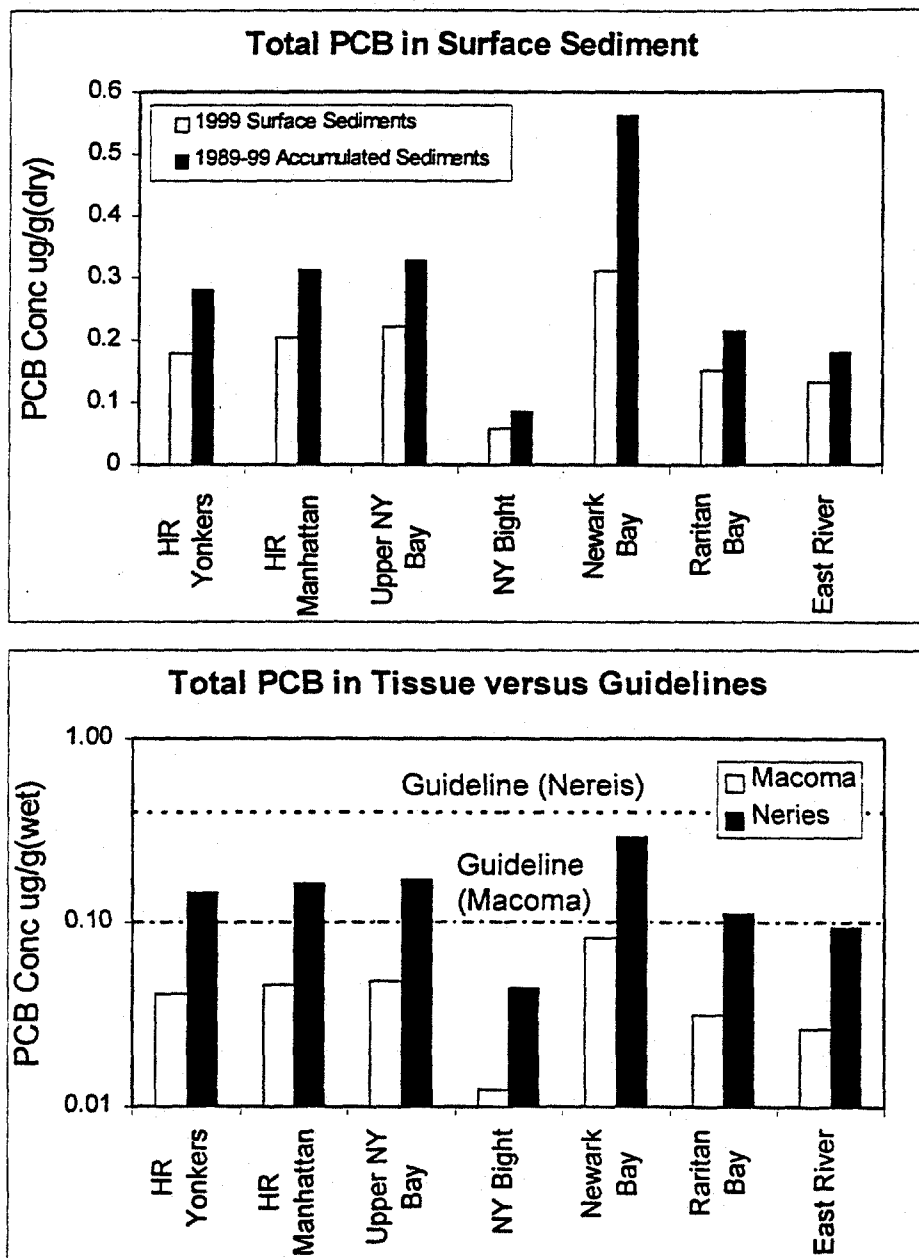


Figure 4-13. (Top Panel) Model results for 1999 PCB contamination levels in harbor surface sediments and for average PCB concentrations for ten years (1989-99) of accumulated sediments. (Bottom Panel) Model results for PCB accumulations in the dredged material test organisms (*Macoma nasuta* and *Nereis virens*) exposed to homogenized sediments from ten years (1989-99) of accumulation.

concentrations for WWTPs, CSOs, SWOs, tributaries, and the atmosphere were assigned based on values reported in the literature for other sites as follows: PAH concentrations for CSOs and WWTPs were assumed to be equal to wastewater influent and effluent concentrations that were measured for the Hamilton Harbor WWTP in Ontario, Canada (Melcer et al. 1995) (see Table 4-4 for concentrations). PAH concentrations for SWOs, industrial discharges, and tributaries were assumed to be equal to 0.1, 10, and 0.01 times the Hamilton Harbor WWTP influent concentrations. The assumed PAH concentrations were multiplied by appropriate flows from WWTPs, CSOs, SWOs, and tributaries to determine PAH loadings (see Chapter 2 for flows).

Atmospheric loadings of PAHs by wet deposition and back diffusion of PAH atmospheric gases were calculated in the same manner as PCB atmospheric loadings (see Chapter 2). For this computation, PAH rainwater and air concentrations were taken from measurements made west of Minneapolis/St. Paul, Minnesota by Franz and Eisenreich (Franz and Eisenreich 1998) in the winter of 1991-92. A listing of Henry's law constants (used in the atmospheric load computation) and $\log K_{ow}$ values (used in chemical fate model calculations) is given in Table 4-5. As in previous calculations for PCBs, the volatilization rate coefficient (k_v) was taken as 0.5 m/day, and as a first estimate, chemical degradation rates were assumed to be negligible.

Since there is such a large uncertainty in our specification of PAH loads and since no information was located on the time history of PAH loadings, model calculations focused on long-term sediment responses to constant PAH loads. Unit response matrices were therefore developed for 23 PAH compounds (see listing in Table 4-5). An example of the response matrices for pyrene ($\log K_{ow}=4.92$) and for benzo(a)pyrene ($\log K_{ow}=6.11$) are given in Table 4-6 and 4-7, respectively. Values within each matrix represent the sediment concentration response in segment "i" due to a unit load of one pound per day in segment "j". These values can readily be multiplied by direct loads into specific harbor segments or diffuse loads from the atmosphere to determine long-term sediment responses.

Values in the unit response matrices also provide a good indication of the transport and fate of PAHs in the harbor. For example, values in the Hudson River (segment #15) row of the matrix represent the response in Hudson River sediments to loads in different harbor segments. As expected, the largest concentration response in the Hudson River is due to direct loadings into the Hudson River. Values in the Hudson River (segment #15) column of the matrix represent the response in all segments due to a contaminant load into the Hudson River. Typically, we would expect the greatest concentration response due to loadings in the Hudson River to be in the Hudson River. In this case, however, the greatest concentration response is in Newark Bay (segment #24) sediments. This result is largely due to spatial variations of organic carbon in the harbor segments. For our example of loads into the Hudson River, dissolved contaminants are transported from the Hudson River into Newark Bay (by tidal dispersion) where they bind to high concentrations of POC. POC-bound contaminants are then transported to the sediments where they tend to remain due to the relatively low resuspension rate in Newark Bay.

Table 4-4. Chemical Concentrations Used in Estimating PAH Loadings from WWTPs, CSOs, SWO, Industrial Discharges, and Tributaries.⁽¹⁾

Chemical	WWTPs (ug/L) ⁽²⁾	CSOs (ug/L) ⁽³⁾	SWOs (ug/L) ⁽⁴⁾	Industry (ug/L) ⁽⁵⁾	Tributaries (ug/L) ⁽⁶⁾
1-Methylnaphthalene					
1-Methylphenanthrene					
2,3,5-Trimethylnaphthalene					
2,6-Dimethylnaphthalene					
2-Methylnaphthalene					
Acenaphthene		0.40	0.04	0.00	0.00
Acenaphthylene		0.50	0.05	0.00	0.01
Anthracene		0.80	0.08	0.00	0.01
Benzo(a)anthracene	0.10	3.60	0.36	1.00	0.04
Benzo(a)pyrene	0.10	4.30	0.43	1.00	0.04
Benzo(e)pyrene					
Benzo(b,k)fluoranthene	0.10	4.60	0.46	1.00	0.05
Benzo(g,h,i)perylene		2.60	0.26	0.00	0.03
Biphenyl					
Chrysene		3.20	0.32	0.00	0.03
Dibenz(a,h)anthracene		1.10	0.11	0.00	0.01
Fluoranthene		4.60	0.46	0.00	0.05
Fluorene		0.90	0.09	0.00	0.01
Indeno(1,2,3-C,D)pyrene		4.00	0.40	0.00	0.04
Naphthalene	0.20	5.10	0.51	2.00	0.05
Perylene					
Phenanthrene		3.60	0.36	0.00	0.04
Pyrene	0.10	3.70	0.37	1.00	0.04

⁽¹⁾ Blanks indicate no available estimate of concentration.

⁽²⁾ WWTP concentrations were assigned based on final effluent measurements for the Hamilton Woodward WWTP (Melcer et al. 1995).

⁽³⁾ CSO concentrations were assigned based on influent measurements for the Hamilton Woodward WWTP (Melcer et al. 1995).

⁽⁴⁾ SWO concentrations were assumed to be equal to 0.1 times CSO concentrations.

⁽⁵⁾ Concentrations in industrial discharges were assumed to be equal to 10 times WWTP concentrations.

⁽⁶⁾ Concentrations in tributaries were assumed to be equal to 0.01 times CSO concentrations.

Table 4-5. Octanol-Water Partitioning Coefficients (K_{ow}) and Henry's Law Constants (K_H) for PAH Compounds.

Chemical	log $K_{ow}^{(1)}$	MW	K_H (Pa- m ³ /mol)	Source of K_H Value (from Mackay et al (Mackay et al. 1992)
1-Methylnaphthalene	3.84	142.2	45	rec, Mackay & Shiu '81
1-Methylphenanthrene	5.04	192.26	N/A	
2,3,5-Trimethylnaphthalene	4.86	170.26	N/A N/L	
2,6-Dimethylnaphthalene	4.37	156.23	6.53	calc, Eastcott '88
2-Methylnaphthalene	3.86	142.2	40.52	calc, Mackay & Shiu '81
Acenaphthene	4.01	154.21	24	rec, Mackay & Shiu '81
Acenaphthylene	3.22	152.2	11.4	wetted wall col, Fendinger & Glotfelty '90
Anthracene	4.53	178.2	6	rec, Mackay & Shiu '81
Benzo(a)anthracene	5.67	228.29	0.092	15C, calc, Baker & Eisenreich '90
Benzo(a)pyrene	6.11	252.31	0.009	15C, calc, Baker & Eisenreich '90
Benzo(e)pyrene	6.14	252.32	N/A	
Benzo(b,k)flouranthene	6.27	252.32	0.054/0.111	15C, calc, Baker & Eisenreich '90
Benzo(g,h,i)perylene	6.51	276.34	0.001	15C, calc, Baker & Eisenreich '90
Biphenyl		154	28	calc, Mackay & Shiu '81
Chrysene	5.71	228.29	0.1064	calc-P/C, Mabey et al.'82
Dibenz(a,h)anthracene	6.71	278.35	0.0074	calc-P/C, Mabey et al.'82
Flouranthene	5.08	202.26	220	calc, Mackay & Shiu '81
Flourene	4.21	166.2	8.5	rec, Mackay & Shiu '81
Indeno(1,2,3-C,D)pyrene	6.72	276.34	N/A N/L	
Naphthalene	3.36	128.19	43	rec, Mackay & Shiu '81
Perylene	6.14	252.31	0.44	calc-P/C, Riederer '90
Phenanthrene	4.57	178.2	4	rec, Mackay & Shiu '81
Pyrene	4.92	202.26	1.1-1.2	batch-strip, rec, Mackay & Shiu '81

⁽¹⁾ log Kow values from SPARC (Karickhoff et al. 1991)

Table 4-6. Unit Response Matrix for Pyrene (log Kow = 4.92).

ug/kg (ppb) response segment "I"	unit lb/day load in segment "j"						Atmospheric load of 1 ug/m ² day
	10	15	16	24	25	26	
10	27.60	0.02	0.00	0.00	0.00	0.00	5.96
11	17.40	0.09	0.01	0.01	0.01	0.01	4.89
12	11.50	0.26	0.03	0.02	0.02	0.03	4.97
13	8.50	0.95	0.10	0.09	0.08	0.11	5.60
14	7.45	3.58	0.37	0.33	0.32	0.43	6.31
15	8.62	24.00	2.49	2.23	2.13	2.86	12.40
16	8.16	23.30	27.80	24.90	23.80	24.30	61.50
17	0.88	2.63	3.04	3.05	2.71	2.66	38.20
24	13.60	41.00	49.40	87.30	48.30	43.20	114.00
25	5.50	16.20	19.50	20.20	26.90	17.00	47.70
26	2.86	8.65	9.02	8.08	7.72	22.90	23.50

Table 4-7. Unit Response Matrix for Benzo(a)pyrene (log Kow = 6.11).

ug/kg (ppb) response segment "i"	unit lb/day load in segment "j"						Atmospheric load of 1 ug/m ² day
	10	15	16	24	25	26	
10	177.00	0.18	0.02	0.01	0.01	0.02	47.80
11	136.00	0.77	0.07	0.06	0.06	0.08	45.80
12	108.00	2.32	0.21	0.18	0.17	0.24	50.20
13	91.30	7.71	0.69	0.59	0.57	0.80	56.70
14	83.20	25.80	2.36	2.01	1.97	2.73	60.70
15	81.00	141.00	13.40	11.40	11.20	15.30	86.40
16	42.60	75.20	91.00	77.80	76.80	77.10	201.00
17	9.96	18.30	18.80	19.30	17.20	16.00	187.00
24	44.00	81.00	98.90	180.00	96.20	83.80	228.00
25	31.10	54.50	66.10	66.10	93.80	56.00	165.00
26	27.30	48.10	49.90	42.60	42.00	133.00	137.00

Calculated responses for pyrene and benzo(a)pyrene in harbor sediments which were determined from unit response matrices and estimated PAH loads) are given in Figure 4-14. As shown, maximum sediment concentrations for both pyrene and benzo(a)pyrene occur Newark Bay and are due to direct loadings of contaminant into the bay, and as explained above, to the effective trapping of contaminants from other areas in the harbor in Newark Bay. A large variation in sediment response concentrations in the harbor segments is also noted for pyrene (Figure 4-14) and for other low K_{ow} compounds (not shown). This behavior is also indicated in the unit response matrices, and in part, is explained by higher dissolved fractions for low K_{ow} compounds which leads to a more rapid loss of chemical by volatilization near the source area. For both pyrene and benzo(a)pyrene, approximately 50% of the calculated contaminant responses in sediments is associated with CSO loads.

Comparison of model results to R-EMAP field data is given for pyrene and benzo(a)pyrene in Figure 4-15. Overall, the model results provide a reasonable description of the field data considering the many assumptions that were used in assigning contaminant loads. Results for pyrene ($\log K_{ow}=4.92$) however are low, particularly in the Hudson River (segment #14 and #15) and in the East River (segment #26), suggesting that not all sources of pyrene were accounted for in the model calculations. Model results for other low K_{ow} compounds show similar behavior. A summary of model results for both low K_{ow} and high K_{ow} PAHs is given in the top panel of Figure 4-16. As shown, model results provide a reasonable description of the R-EMAP field data for the high K_{ow} compounds (with calculated/observed concentration ratios of approximately one), but tend to underestimate sediment concentrations for the lower K_{ow} compounds. This discrepancy in model results and field data (which is shown more clearly by geometric means of the calculated/observed concentration ratios in the bottom panel of Figure 4-16) may be associated with our specification of PAH partitioning to POC and DOC, slow desorption kinetics, or improper specification of PAH loads. Further examination of these issues will be considered after site-specific PAH loading data become available for New York-New Jersey Harbor.

In this modeling analysis, sediment contaminant concentrations were determined for fifteen PAH compounds (see listing of compounds with input concentrations in Table 4-4). Total PAH concentrations in sediments were then determined from the summation of the fifteen PAH compounds and are shown in the top panel of Figure 4-17. PAH concentrations in the dredged material test organisms, *Macoma nasuta* and *Nereis virens*, were determined using Eq 25 with a BSAF value of 0.65 (based on Eq 23 and an average PAH $\log K_{ow}$ of 5) (bottom panel of Figure 4-17). As shown, average PAH concentrations in macoma and nereis determined from the model calculation fall well below the PAH tissue concentration guideline of 40 $\mu\text{g/g}$ (wet wt). Final conclusions for PAH contamination of dredged material however can not be made until site-specific PAH loading data for New York-New Jersey Harbor are considered and the model is properly validated, particularly for low K_{ow} PAH compounds.

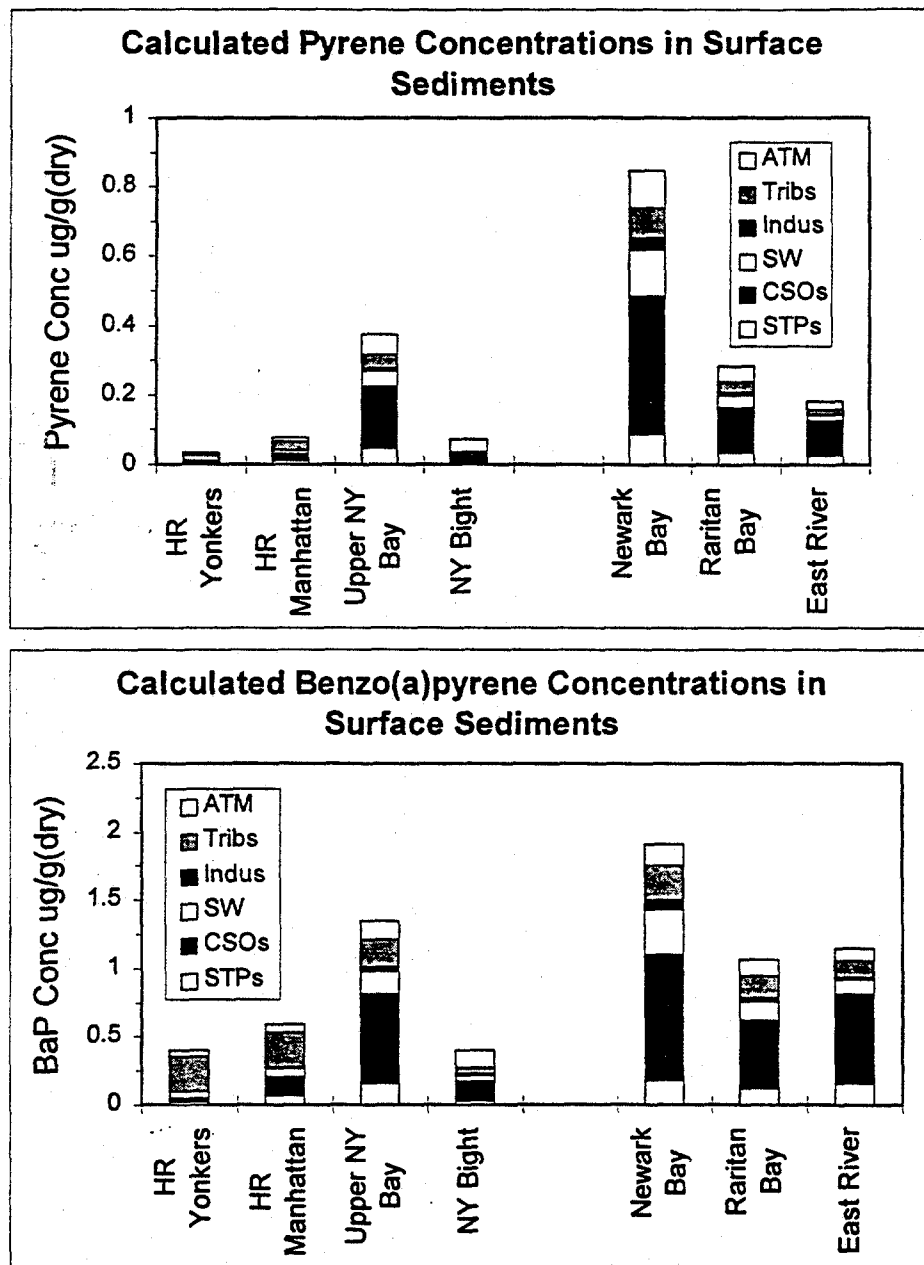


Figure 4-14. Calculated responses for pyrene and benzo(a)pyrene in New York-New Jersey Harbor sediments (as determined from unit response matrices and estimated PAH loads).

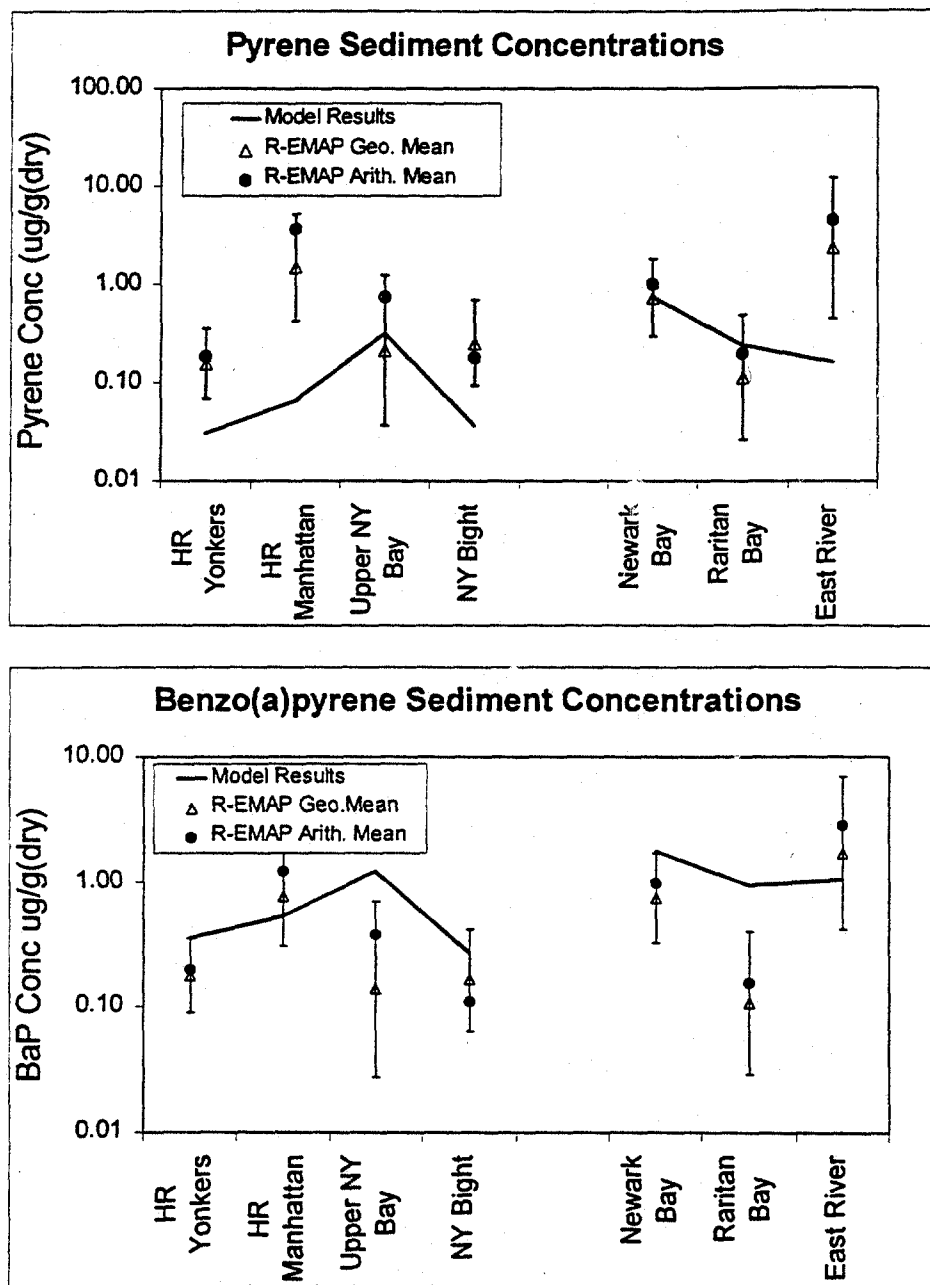


Figure 4-15. Model-field data comparisons for pyrene and benzo(a)pyrene sediment concentrations in New York-New Jersey Harbor. [Data from 1993-94 R-EMAP study (Adams et al. 1998).]

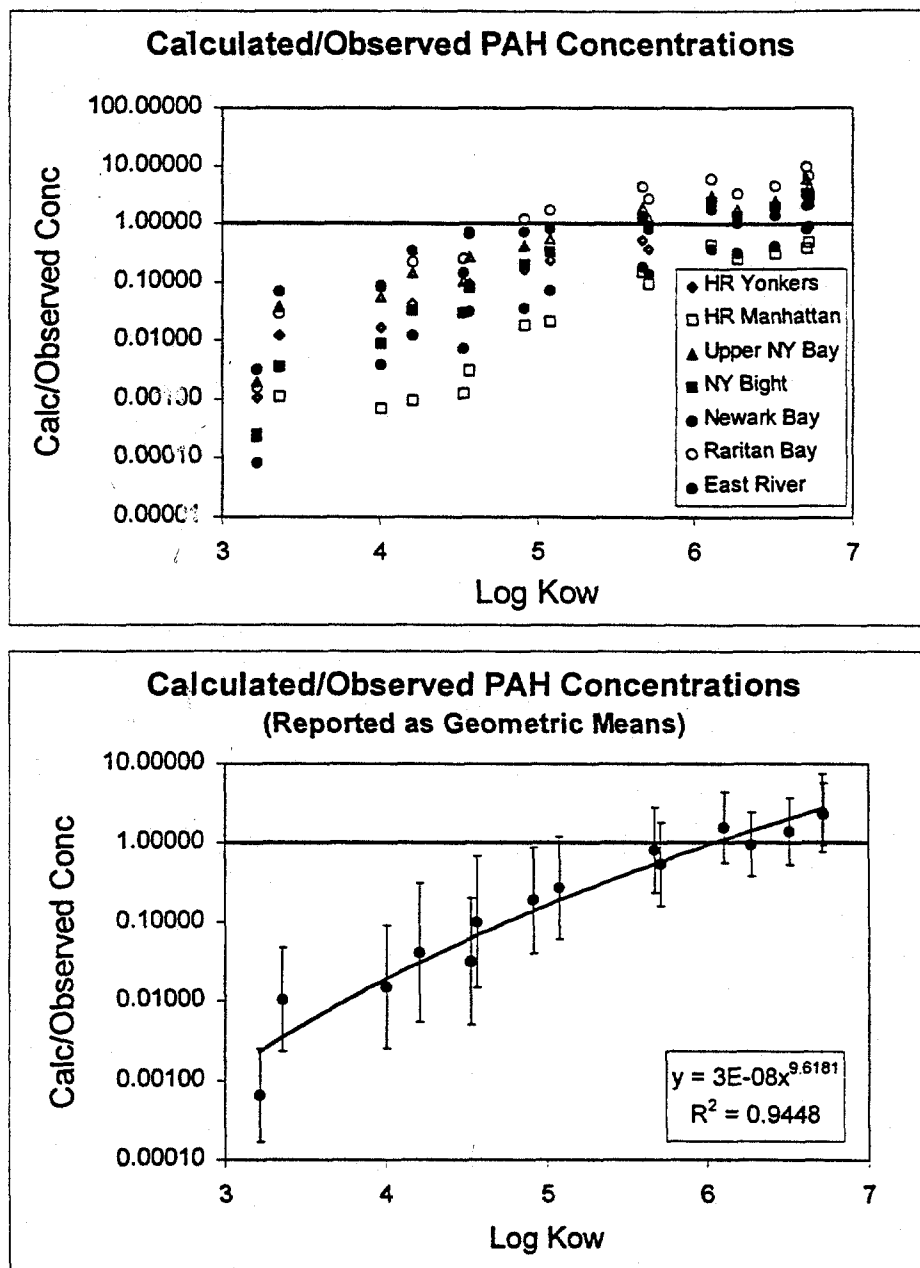


Figure 4-16. Summary of model-field data comparisons for 15 PAH compounds in New York-New Jersey Harbor sediments as a function of log K_{ow} . [Data from 1993-94 R-EMAP study (Adams et al. 1998).]

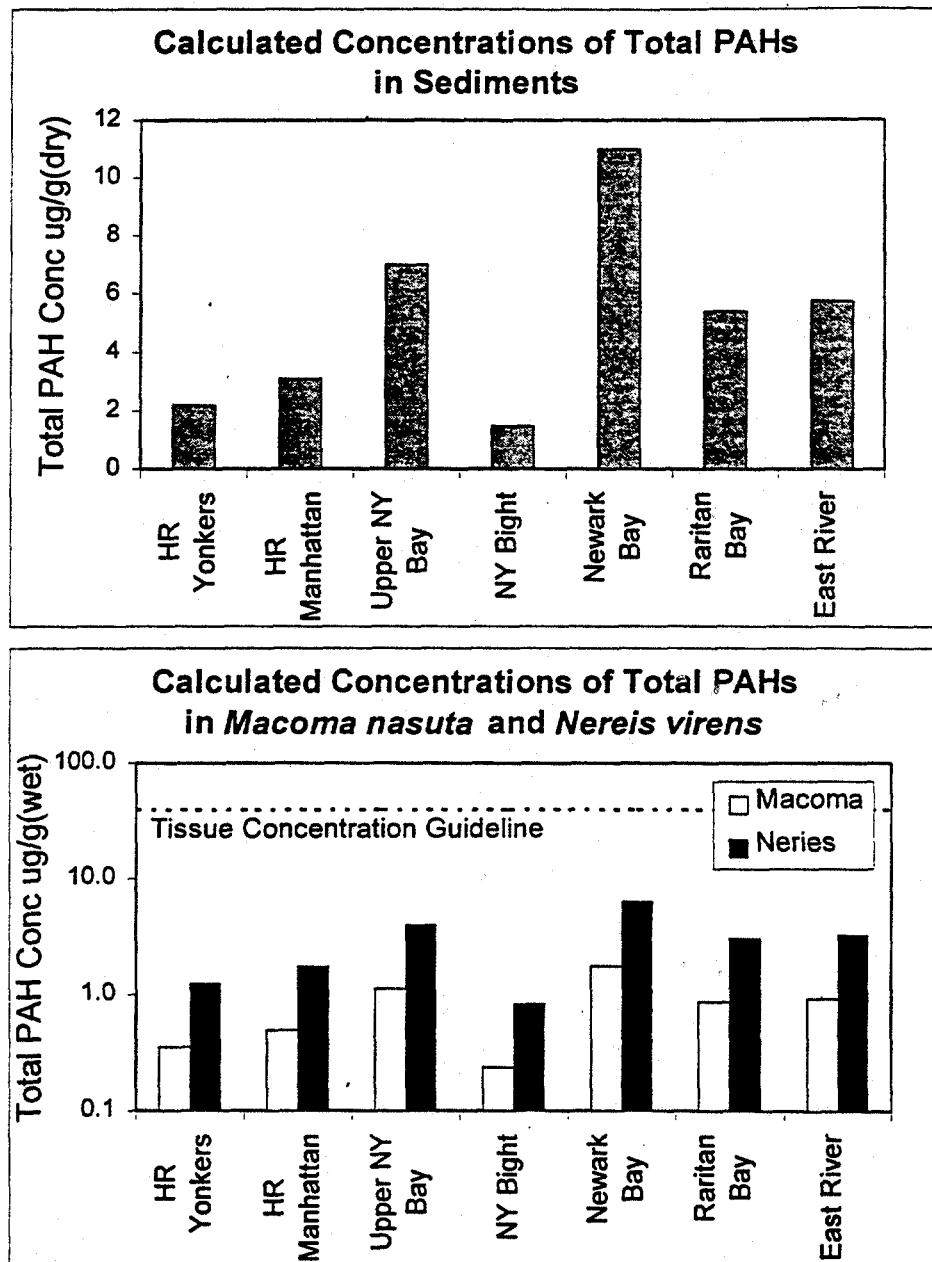


Figure 4-17. Model results for total PAH sediment concentrations (given as the summation of 15 PAH compounds), and for total PAH accumulation in dredged material test organisms (*Macoma nasuta* and *Nereis virens*).

Dioxin Model Results

Model calculations were performed for the most toxic dioxin congener, 2,3,7,8-TCDD. The principal source of TCDD contamination in the harbor is attributed to past discharges from an industrial site in Newark, NJ where chlorinated phenols were produced (Bopp et al. 1991). Continuing discharges of TCDD from CSOs and SWOs in the Newark area have also been noted (Burton 1998). Chemical fate model calculations for TCDD were therefore performed by first considering the long-term response of harbor sediments due to TCDD loads into the Lower Passaic/Newark Bay/Kills (segment #24). The response matrix for a unit load of TCDD into segment #24 is given in Table 4-8 and shows approximately an order of magnitude decrease in TCDD sediment concentration responses from Newark Bay to the New York Bight Apex. This result is comparable with observed TCDD concentration ranges given in Figure 4-18, and is consistent with the Newark area being the principal source of TCDD to New York-New Jersey Harbor.

Table 4-8. Unit Response Matrix for TCDD Load into the Lower Passaic/Newark Bay/The Kills (segment #24).	
ng/kg (pptr) Response in Segment 'i'	Unit g/day Load into Segment #24
10 (Hudson RM 56.8)	0.05
11 (Hudson RM 46.8)	0.22
12 (Hudson RM 36.8)	0.68
13 (Hudson RM 26.8)	2.20
14 (Hudson RM 16.8)	7.36
15 (Hudson RM 6.8)	38.36
16 (NY Harbor)	203.48
17 (NY Bight)	51.37
24 (Newark Bay)	438.71
25 (Raritan Bay)	167.55
26 (East River)	122.80

The long-term response of harbor sediments to continuing CSO and SWO discharges of TCDD into the Lower Passaic/Newark Bay/Kills (segment #24) is given by the solid line in Figure 4-18. For this calculation, average TCDD concentrations in CSOs and SWOs were taken as 5.8 pg/L based on November 1997 data (Burton 1998). Results

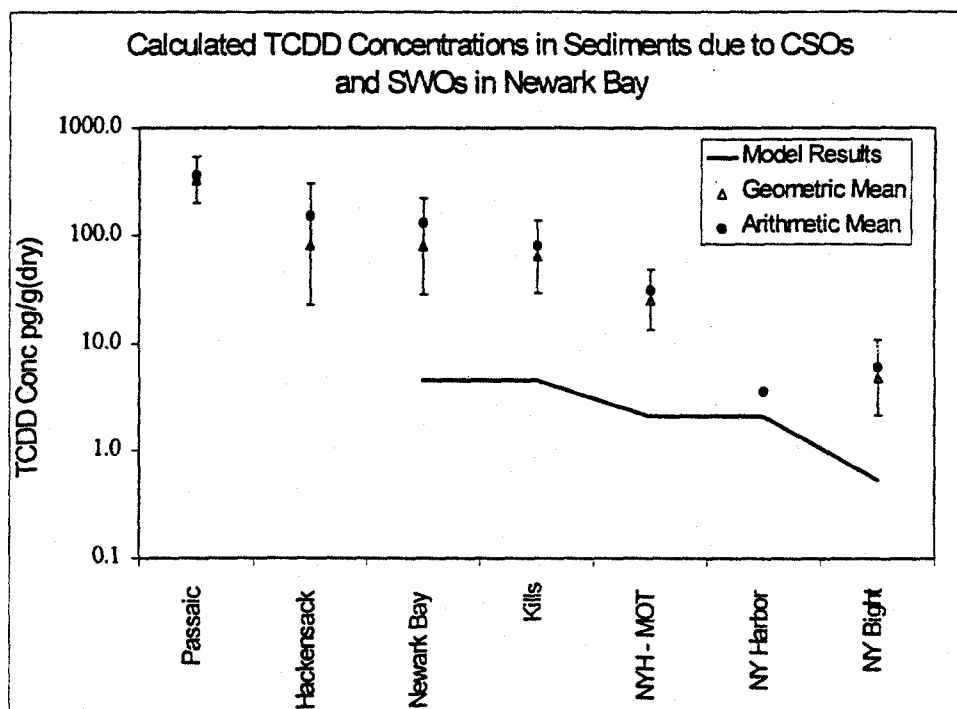


Figure 4-18. Model-field data comparison for TCDD sediment contamination in New York-New Jersey Harbor due to continuous TCDD discharges from CSOs and SWOs into the Lower Passaic and Newark Bay. [Data from Long et al. (1995) for the Passaic, Hackensack, Newark Bay, Arthur Kill, and NY-NJ Harbor; data from Army Corps of Engineers, New York Region permit applications for NY-NJ Harbor Military Ocean Terminal; data from Battelle (1996) for the NY Bight Mud Dump Site.]

of this calculation are shown to be an order of magnitude lower than observed concentrations, and suggest that present TCDD discharges from CSOs and SWOs are not the primary cause of TCDD contamination in harbor sediments. (The effects of present CSO and SWO loads on water column concentrations and TCDD accumulation in fish were not addressed in this preliminary evaluation.) Preliminary model calculations to assess the effects of historically-contaminated sediments from the Lower Passaic River and Newark Bay on present TCDD contamination levels are given below.

A thirty year time-variable model simulation was performed for the 1965-95 time period. For the first run, initial conditions were set at 2,000 pg/g(dry wt) in sediments for the Lower Passaic/Newark Bay/The Kills (segment #24) based on dated sediment core results from Bopp et al (Bopp et al. 1991), and at zero in all other model segments. Results show a reasonable decline in TCDD concentrations in surface sediments of Newark Bay with time (dashed line in top panel of Figure 4-19). The 1992 projections for TCDD concentrations in harbor sediments (dashed line in bottom panel of Figure 4-19) show a large drop off in TCDD sediment concentrations between The Kills and New York Harbor. The calculated TCDD concentration in New York Harbor is much lower than measured TCDD concentrations at the New York Harbor Military Ocean Terminal but is consistent with the 1992 NOAA New York Harbor sample (Long et al. 1995). The differences in model results (and the 1992 NOAA sample) and the Military Ocean Terminal samples is likely due to the close proximity of the Ocean Terminal to the Kill van Kull.

As an upper boundary of TCDD contamination, a second model run was performed keeping the initial conditions in the Lower Passaic/Newark Bay/The Kills (segment #24) at 2,000 pg/g(dry wt) and adding initial conditions of 100 pg/g(dry wt) in New York Harbor (segment #16) and Raritan Bay (segment #25). For this simulation, model results (as given by the solid line in Figure 4-19) are in reasonable agreement with field observations for the New York Harbor Military Ocean Terminal.

Results from the first model simulation are replotted in the top panel of Figure 4-20. These values (which as discussed above are considered as a lower boundary of TCDD contamination) were used to calculate TCDD accumulations in *Macoma nasuta* and *Nereis virens*. Eq 25 was used for this calculation with a BSAF value of 0.54 (based on Eq 23 and a log K_{ow} of 7 for TCDD). The results (given in the bottom panel of Figure 4-20) show that dredged material in Newark Bay would be classified as Category 3 (and not acceptable for ocean disposal) based on average TCDD accumulations in nereis. Sediments for other locations in the harbor region would be at or near the tissue concentration guidelines for Category 2 dredged material.

These preliminary findings suggest that past discharges of TCDD into the Lower Passaic and Newark Bay may have contributed to TCDD sediment contamination beyond Newark Bay. TCDD concentrations in sediments however are slowly decreasing to acceptable levels. Further evaluation of sediment data, particularly for the New York Harbor segment, should be performed before reaching any final conclusions on the extent of TCDD contamination.

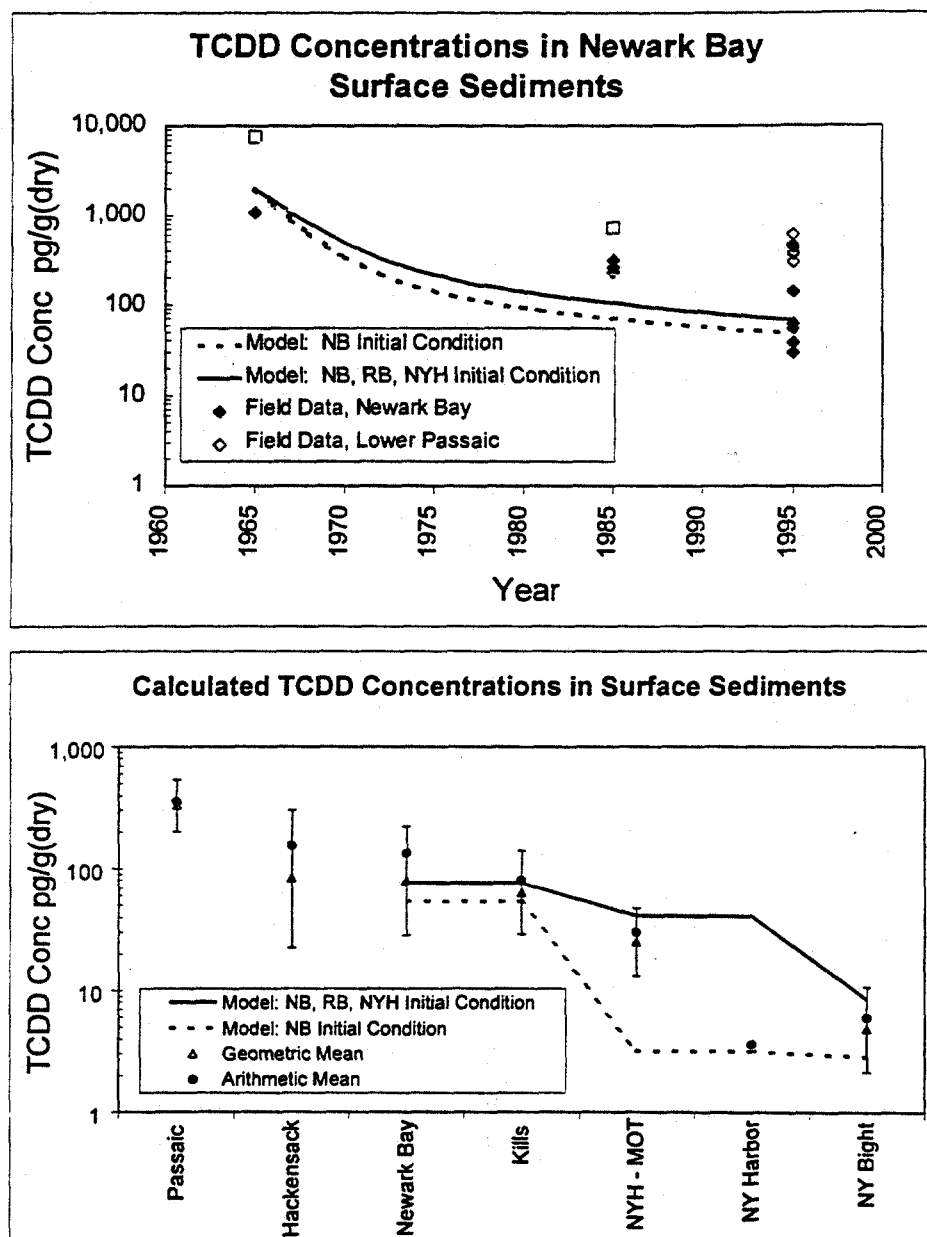


Figure 4-19. Model-field data comparison for 1992 TCDD surface sediment concentrations in New York-New Jersey Harbor due to 1965 initial TCDD sediment concentrations (see text). [Data from Long et al. (1995) for the Passaic, Hackensack, Newark Bay, Arthur Kill, and NY-NJ Harbor; data from Army Corps of Engineers, New York Region permit applications for NY-NJ Harbor Military Ocean Terminal; data from Battelle (1996) for the NY Bight Mud Dump Site.]

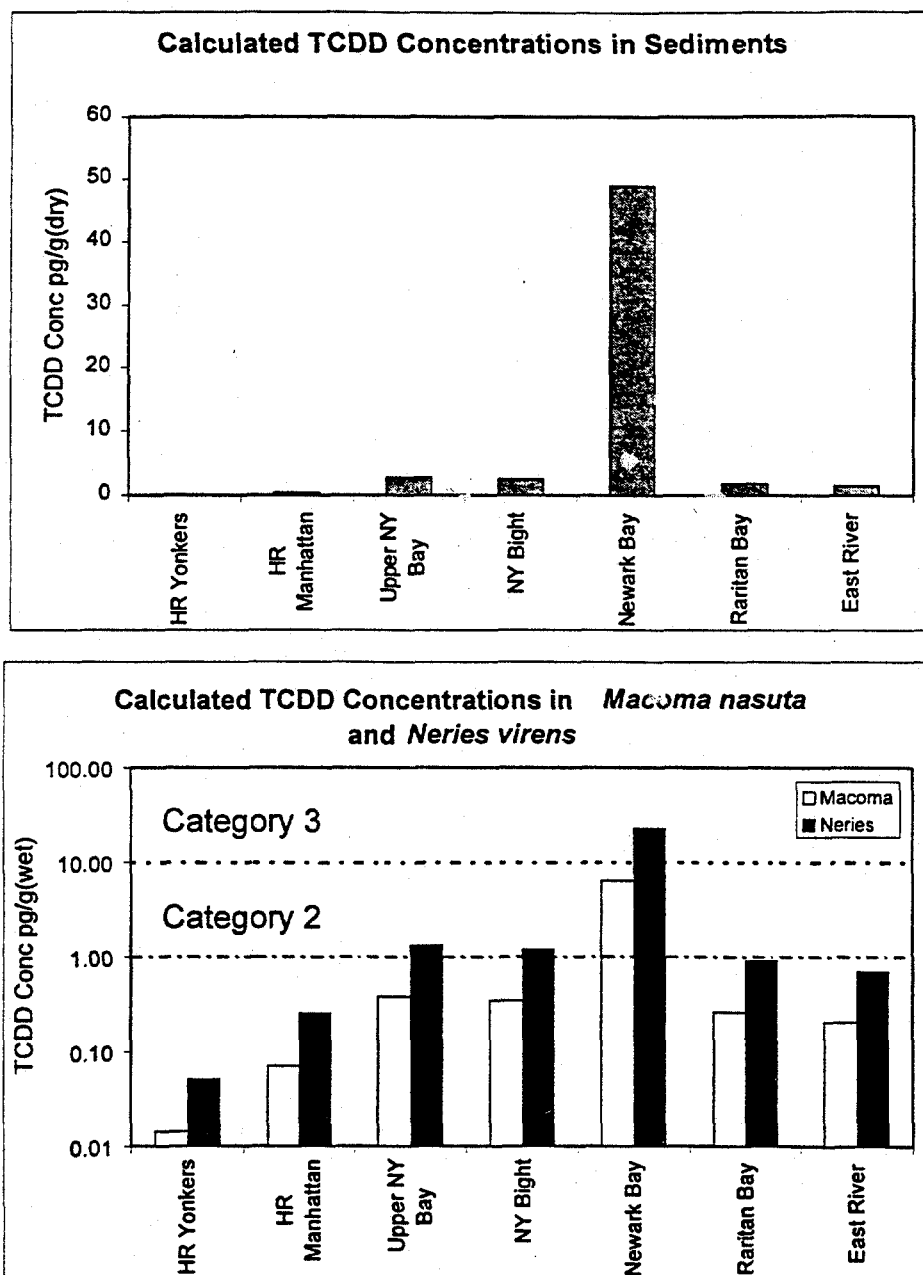


Figure 4-20. Model results for 1995 TCDD surface sediment concentrations in New York-New Jersey Harbor, and for TCDD accumulation in dredged material test organisms (*Macoma nasuta* and *Nereis virens*) due to a 1965 initial TCDD concentration of 2,000 pg/g(dry) in the Lower Passaic and Newark Bay sediments.

Summary and Conclusions

PCBs, PAHs, and dioxins are three of the contaminants considered as potential causative agents in the contamination of dredged material in New York-New Jersey Harbor. In our studies, a quantitative modeling approach was developed to relate PCB, PAH, and dioxin contaminant loadings to specific dredged material endpoints (including the bioaccumulation of PCBs, PAHs, and dioxins in the test organisms *Macoma nasuta* and *Nereis virens*). The modeling approach includes a chemical fate calculation (to predict contaminant concentrations in harbor sediments) and a bioaccumulation calculation (where contaminant sediment concentrations are related to tissue concentrations using chemical-specific BSAFs). In this approach, calculated contaminant concentrations in macoma and nereis are then compared directly to PCB, PAH, and dioxin tissue concentration guidelines for dredged material for New York-New Jersey Harbor. Key findings in model development included the following:

1. BSAFs for PCBs (that were derived from Army Corps of Engineers permit data) and for dioxins (that were derived from field data from the mud dump site) corresponded reasonably well to the BSAF free energy relationship given by DiToro and McGrath (DiToro and McGrath in prep). This relationship, which does not include the effects of chemical metabolism, overpredicted BSAFs for PAHs. The higher BSAF estimates from the relationship of DiToro and McGrath (DiToro and McGrath in prep) however were used in our analysis to account for narcotic effects of both the PAH parent compounds and their metabolites.
2. Preliminary assessment of New York-New Jersey Harbor dredged material was made based on available sediment data. Results showed that approximately 17% of the harbor sediments would likely cause an exceedance of PCB tissue concentrations. For TCDD, sediment samples from many of the harbor regions would likely cause an exceedance of Category 1 guidelines, and sediment samples from Newark Bay and from New York Harbor (near the Marine Ocean Terminal) would likely fail to meet requirements for Category 2 dredged material. Although 16% of the sediment samples exhibited reduced *amperisca* survival rates, exceedance of present PAH tissue concentration guidelines would be expected in only a few sediment samples. This suggests that the present narcosis-based guideline is too high or that other causative agents of *amperisca* toxicity are present.
3. Model results showed that average PCB accumulations in macoma and nereis approached dredged material tissue concentration guidelines for Newark Bay. Average PCB accumulations in macoma and nereis were calculated to be two or three times lower than the tissue concentration guidelines for the other regions of the harbor. These results are consistent with our preliminary assessment of field data.
4. PAH model calculations were performed using assigned WWTP, CSO, SWO, tributary, and atmospheric loads from reported concentrations for other sites. Model

results for PAH sediment concentrations corresponded reasonably well to field data for high K_{ow} compounds, but underpredicted PAH sediment concentrations for low K_{ow} compounds. The discrepancy in model results and field data for low K_{ow} compounds may be associated with our specification of PAH partitioning to POC and DOC, slow desorption kinetics, or improper specification of PAH loads. Further examination of these issues will be considered after site-specific PAH loading data become available for New York-New Jersey Harbor.

5. Model results for total PAH concentrations in macoma and nereis show that average PAH concentrations for the various harbor regions are below PAH tissue concentration guidelines. These results are consistent with our preliminary assessment of field data.
6. Long-term response of harbor sediments to continuing TCDD discharges from CSOs and SWOs into the Lower Passaic and Newark Bay are an order of magnitude below observed concentrations and suggest that present TCDD discharges are not the primary cause of TCDD contamination in harbor sediments. Time-variable simulations with: (1) an initial TCDD concentration specified in the Lower Passaic, Newark Bay, and The Kills sediments (segment #24); and (2) initial TCDD concentrations in the Lower Passaic, Newark Bay, and The Kills (segment #24), New York Harbor (segment #16), and Raritan Bay (segment #25) sediments, corresponded more closely to the field data. Further evaluation of sediment data, particularly for the New York Harbor segment, should be performed before reaching any final conclusions on the extent of TCDD contamination.

References

References

- Abramowicz, D.A. (1990). "Aerobic and anaerobic biodegradation of PCBs: A review." *Crit. Rev. Biotechnol.*, 10(3), 241-251.
- Abramowicz, D.A., Brennan, M.J., Dort, H.M.V., and Gallagher, E.L. (1993). "Factors influencing the rate of polychlorinated biphenyl dechlorination in Hudson River sediments." *Environ. Sci. Technol.*, 27(6), 1125-1131.
- Adams, D.A., O'Connor, J.S., and Weisberg, S.B. (1998). "Sediment Quality of the NY/NJ Harbor System: An Investigation under the Regional Environmental Monitoring and Assessment Program." *EPA/902-R-98-001*.
- Ambrose, R.B., Wool, T.A., Connolly, J.P., and Schnaz, R.W. (1988). "WASP4: A hydrodynamic and water quality model - model theory, user's manual and programmer's guide." *EPA/600/3-87/039*, U.S. EPA Environmental Research Laboratory, Athens, GA.
- Ashley, J.T.F., Secor, D.H., Zlokovitz, E.R., Wales, S.Q., and Baker, J.E. (submitted). "Linking habitat use of Hudson River striped bass to accumulation of PCB congeners." *Environ. Sci. Technol.*
- Bath, D.W., and O'Connor, J.M. (1982). "The biology of the white perch, *Morone americana*, in the Hudson River Estuary." *Fish. Bull.*, 80(3), 599-610.
- Battelle. (1996). "Sediment Survey of the Mud Dump Site and Environs." , Battelle Ocean Sciences, Duxbury MA.
- Battelle. (1995). "Laboratory Analytical Services on Wastewater Samples: Dry Weather Events I and II; Wet Weather Events I and II." *New Jersey Harbor Dischargers Group (C.N. 94-1-NJHDG)*, Battelle, Ocean Sciences, Duxbury, MA.
- Battelle. (1995b). "Analysis of WPCP Influent and Effluent Samples for PCB and Dioxin: Dry Weather Events I and II." *New York City Department of Environmental Protection (P.N. 927-ADM)*, Battelle, Ocean Sciences, Duxbury, MA.
- Bedard, D.L., Haberl, M.L., May, R.J., and Brennan, M.J. (1987a). "Evidence for novel mechanisms of polychlorinated biphenyl metabolism in *Alcaligenes eutrophus* H850." *Appl. Environ. Microbiol.*, 53(5), 1103-1112.

- Bedard, D.L., Unterman, R., Bopp, L.H., Brennan, M.J., Haberl, M.L., and Johnson, C. (1986). "Rapid assay for screening and characterizing microorganisms for the ability to degrade polychlorinated biphenyls." *Appl. Environ. Microbiol.*, 51(4), 761-768.
- Bedard, D.L., Wagner, R.E., Brennan, M.J., Haberl, M.L., and J.F. Brown, J. (1987b). "Extensive degradation of Aroclors and environmentally transformed polychlorinated biphenyls by *Alcaligenes eutrophus* H850." *Appl. Environ. Microbiol.*, 53(5), 1094-1102.
- Bopp, R.F., Gross, M.L., Tong, H., Simpson, H.J., Monson, S.J., Deck, B.L., and Moser, F.C. (1991). "A major incident of dioxin contamination: sediments of New Jersey estuaries." *Environ. Sci. Technol.*(25), 951-956.
- Bopp, R.F. (1979). "The geochemistry of polychlorinated biphenyls in the Hudson River," PH.D. dissertation, Columbia University, New York.
- Broman, D., Naf, C., Lundbergh, I., and Zebuhr, Y. (1990). "An in-situ study on the distribution, biotransformation and flux of polycyclic hydrocarbons (PAHs) in an aquatic food chain (seston-mytilus edulis L.-somateria mollissima L.) from Baltic: An ecotoxicological perspective." *Environ. Toxicol. Chem.*(9), 429-442.
- Brownawell, B.J. (1997). Personal communication.
- Brunciak, P., Lavorgna, C., Nelson, E., and Eisenreich, S. "PCBs in the New Jersey urban and coastal atmosphere." *Annual Meeting of the International Association of Great Lakes Research*, Hamilton, Ontario.
- Burton, S. (1998). "September 18, 1997 - November 1, 1997 Passaic River Outfall Water Sampling." , Personal Communication to Sharon Jaffess, Diamond Alkali Project Manager, Passaic River Study Area, USEPA, Region II.
- Butcher, J.B., Garvey, E.A., and Bierman, V.J.J. (1998). "Equilibrium partitioning of PCB congeners in the water column: Field measurements from the Hudson River." *Chemosphere*, 36(15), 3149-3166.
- Chen, I. (1995). "New York/New Jersey Harbor PCBs Loading, A Review of the PCB Report Submitted by New York City Department of Environmental Protection." *Draft*, US Environmental Protection Agency, Region 2, Water Management Division, New York, NY.
- Chillrud, S.N. (1996). "Transport and Fate of Particle Associated Contaminants in the Hudson River Basin," PhD, Columbia University.

- Clark, J.F., Schlosser, P., Stute, M., and Simipson, H.J. (1996). "SF6-3He tracer release experiment: A new method of determining longitudinal dispersion coefficients in large rivers." *Environ. Sci. Technol.*, 30, 1527-1532.
- Coch, N.K., and Bokuniewicz, H. (1986). "Sedimentation in the Hudson system: the Hudson River and contiguous waterways." *Northeastern Geol.*, 8(3), 94-170.
- Cook, P.M., Erickson, R.J., Spehar, R.L., Bradbury, S.P., and Ankley, G.T. (1993). "Interim Report on Data and Methods for Assessment of 2,3,7,8-Tetrachlorodibenzo-p-dioxin Risks to Aquatic Life and Associated Wildlife." , prepared U.S. Environmental Protection Agency.
- Connolly, J.P. (1991a). "Application of a Food Chain Model to Polychlorinated Biphenyl Contamination of the Lobster and Winter Flounder Food Chains in New Bedford Harbor." *Environ. Sci. Technol.*, 25(4), 760-770.
- Connolly, J.P. (1991b). "Users Guide for WASTOX2, Version 2.51." , Manhattan College, Riverdale NY.
- Connolly, J.P. (1997). Personal communication.
- Connolly, J.P., and Thomann, R.V. (1985). "WASTOX, A Framework for Modeling the Fate of Toxic Chemicals in Aquatic Environments: Part 2. Food Chain." , Manhattan College, Bronx, NY.
- Connolly, J.P., and Tonelli, R. (1985). "Modelling Kepone in the striped bass food chain of the James River Estuary." *Estuar., Coast. Shelf Sci.*, 20, 349-366.
- Di Toro, D.M., McGrath, J.A., and Hansen, D. J. (1999). "Technical basis for narcosis chemicals and PAH criteria. I. Water and tissue." *in prep.*
- DiToro, D.M., and McGrath, J.A. (1999). "Technical basis for narcosis chemicals and PAH criteria. II. Mixtures and sediments." *in prep.*
- DiToro, D.M. (1985). "A particle interaction model of reversible organic chemical sorption." *Chemosphere*, 14(10), 1503-1538.
- Driscoll, S.K., and McElroy, A.E. (1996). "Bioaccumulation and metabolism of benzo[a]pyrene in three species of polchaete worms." *J. Great Lakes Res.*, 15(8), 1401-1410.

- Dunnivant, F.M., Elzerman, A.W., Jurs, P.C., and Hasan, M.N. (1992). "Quantitative structure-property relationships for aqueous solubilities and Henry's law constants of polychlorinated biphenyls." *Environ. Sci. Technol.*, 26(8), 1567-1573.
- Erickson, R.J., and McKim, J.M. (1990). "A model for exchange of organic chemicals at fish gills: Flow and diffusion limitations." *Aquat. Toxicol.*, 18, 175-198.
- Farrington, J.W., Wakeham, S.G., Livramento, J.B., Tripp, B.W., and Teal, J.M. (1986). "Ultraviolet fluorescence analyses and hydrocarbons in New York Bight polychaetes: Ultraviolet fluorescence analyses and gas chromatography/gas chromatography-mass spectrometry analyses." *Environ. Sci. Technol.*(20), 69-72.
- Farrow, D.R.G., Arnold, F.D., Lombardi, M.L., Main, M.B., and Eichelberger, P.D. (1986). "The National Coastal Pollutant Discharge Inventory: Estimates for Long Island Sound." for Strategic Assessment Branch, Ocean Assessments Division, Office of Oceanography and Marine Assessment, NOD, NOAA; 40 pp.
- Franz, T.P., and Eisenreich, S.J. (1998). "Snow scavenging of polychlorinated biphenyls and polycyclic aromatic hydrocarbons in Minnesota." *Environ. Sci. Technol.*(32), 1771-1778.
- Gailani, J., Ziegler, C.K., and Lick, W. (1991). "Transport of suspended solids in the lower Fox River." *J. Great Lakes Res.*, 17(4), 479-494.
- Gardinier, M.N., and Hoff, T.B. (1982). "Diet of Striped Bass in Hudson River Estuary." *New York Fish Game Jour.*, 29(2), 152-165.
- Gardiner, W.W., Barrows, E.S., Antrim, L.D., Gruendell, B.D., Word, J.Q., and Tokos, J.J.S. (1996). "Evaluation of dredged material proposed for ocean disposal from Hudson River, New York." *PNNL-11342*, Pacific Northwest National Laboratory, Sequim WA.
- Garvey, E.A., and Hunt, C. (1997). "Long term fate of PCBs in the Hudson River sediments." *Clearwaters*, 27(2), 17-22.
- Harbor Estuary Program (1996). "Summary of the Comprehensive Conservation & Management Plan of March 1996."
- Hawker, D.W., and Connell, D.W. (1988). "Octanol-water partition coefficients of polychlorinated biphenyl congeners." *Environ. Sci. Technol.*, 22, 383-387.

- HydroQual, Inc. (1998). "Collapsed results of SWEM hydrodynamics and organic carbon." Technical Memorandum, HydroQual, Inc., Mahwah, NJ.
- HydroQual, Inc. (1996). "Water Quality Modeling Analysis of Hypoxia in Long Island Sound Using LIS 3.0." , HydroQual, Inc., Mahwah, NJ.
- HydroQual, Inc. (1991). "Task 7.1 Assessment of Pollutant Loadings to New York/New Jersey Harbor." *Job Number WOCL0302*, United States Environmental Protection Agency Marine and Wetlands Protection Branch, Region II.
- Hydroscience, Inc. (1978). "Seasonal Steady-State Modeling." *NYC 208 Task Report, 314 for Hazen & Sawyer, Engineers, and NYC DWR*, Westwood NJ.
- Johnson, B., Kim, K., Heath, R., Hsieh, B., and Butler, L. (1993). "Validation of a three-dimensional hydrodynamic model of Chesapeake Bay." *J. Hydraul. Engr.*, 119(1), 2-20.
- Karickhoff, S.W., McDaniel, V.K., Melton, C., Vellino, A.N., Nute, D.E., and Carreira, L.A. (1991). "Predicting chemical reactivity by computer." *Environ. Toxicol. Chem.*(10), 1405-1416.
- Karickhoff, S.W. (1981). "Semi-empirical estimation of sorption of hydrophobic pollutants on natural sediments and soils." *Chemosphere*, 10(8), 833-846.
- Limno-Tech, Inc., Menzie Cura & Associates, Inc., and The CADMUS Group, Inc. (1996). "Phase 2 Report - Review Copy Further Site Characterization and Analysis Volume 2B - Preliminary Model Calibration Report Hudson River PCBs Reassessment RI/FS." prepared for U.S. Environmental Protection Agency, Region II.
- Mackay, D., Shiu, W.Y., and Ma, K.C. (1992). *Illustrated Handbook of Physical-Chemical Properties and Environmental Fate for Organic Chemicals*, Lewis Publishers, Boca Raton FL.
- McGroddy, S.E., Read, L.B., Field, L.J., Severn, C.G., and Dexter, R.N. (1997). "Hudson River Congener-Specific Analysis: Data Summary and Analysis Report." , prepared for National Oceanographic and Atmospheric Administration, Seattle WA.
- McFarland, V.A. (1995). "Evaluation of field-generated accumulation factors for predicting the bioaccumulation potential of sediment-associated PAH compounds." *Technical Report D-95-2*, U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg MS.

- McKim, J., Schmieder, P., and Veith, G. (1985). "Absorption dynamics of organic chemical transport across trout gills as related to octanol-water partition coefficient." *Toxicol. Appl. Pharmacol.*, 77, 1-10.
- Melcer, H., Steel, P., and Bedford, W.K. (1995). "Removal of polycyclic aromatic hydrocarbons and heterocyclic nitrogen compounds in municipal treatment plant." *Wat. Environ. Res.*, 67(6), 926-934.
- Morrison, H.A., Gobas, F., Lazar, R., and Haffner, G.D. (1996). "Development and verification of a bioaccumulation model for organic contaminants in benthic invertebrates." *Environ. Sci. Technol.*, 30, 3377-3384.
- Mossman, D.J., Schnoor, J.L., and Stumm, W. (1988). "Predicting the effects of a pesticide release to the Rhine River." *J. Wat. Pollut. Control Fed.*, 60, 1806-1812.
- O'Brien & Gere. (1997). "Correction of Analytical Biases in the 1991-1997 GE Hudson River PCB Database." , prepared for General Electric Company, East Syracuse NY
- O'Connor, D.J. (1988a). "Models of toxic sorptive substances in freshwater systems. I: Basic equations." *J. Environ. Engr.*, 114, 507-531.
- O'Connor, D.J., and Dobbins, W.E. (1958). "Mechanisms of reaeration in natural streams." *Transactions ASCE*, 641, 123.
- O'Connor, D.J., Mueller, J.A., and Farley, K.J. (1983). "Distribution of Kepone in the James River Estuary." *J. Environ. Engr.*, 109, 397-413.
- O'Connor, D.J., Thomann, R.V., and Salas, H.J. (1977). "Water quality; MESA New York Bight atals monograph 27." 27, by Hydrosience for New York Sea Grant Institute, Albany, NY.
- O'Connor, J.M. (1984). "PCBs: Dietary dose and burdens in Striped Bass from the Hudson River." *Northeastern Environ. Sci.*, 3(3/4), 152-158.
- Oliver, B.G., and Niimi, A.J. (1988). "Trophodynamic Analysis of Polychlorinated Biphenyl Congeners and Other Chlorinated Hydrocarbons in the Lake Ontario Ecosystem." *Environ. Sci. Technol.*, 22(4), 388-397.
- Olsen, C.R., Larsen, I.L., Brewster, R.H., Cutshall, N.H., Bopp, R.F., and Simpson, H.J. (1984). "A geochemical assessment of sediment and contaminant distributions in the Hudson-Raritan Estuary." *NOS OMS 2*, NOAA, Rockville, MD.

- Pabst, D., and Tavoraro, J. (1998). "Review of Compliance with the Testing Requirements of 40 CFR 227.6 and 227.27, and Site Designation Provisions of 40 CFR 228.15 for the Passenger Ship Terminal, New York, New York."
- Parkerton, T.F. (1993). "Estimating Toxicokinetic Parameters for Modeling the Bioaccumulation of Non-ionic Organic Chemicals in Aquatic Organisms," PhD, Rutgers University, New Brunswick NJ.
- Phillips, P. (1998). Personal communication.
- Poje, G.V., Riordan, S.A., and O'Connor, J.M. (1988). "Food Habits of the Amphipod *Gammarus tigrinus* in the Hudson River and the Effects of Diet Upon its Growth and Reproduction." Fisheries Research in the Hudson River, C. L. Smith, ed., State University of New York Press, Albany, NY, 255-270.
- Quensen, J.F., Tiedje, J.M., and Boyd, S.A. (1988). "Reductive dechlorination of polychlorinated biphenyls by anaerobic microorganisms from sediments." *Science*, 242, 752-754.
- Rappe, C., and Buser, H.R. (1989). "Chemical and Physical Properties, Analytical Methods, Sources and Environmental Levels of Halogenated Dibenzodioxins and Dibenzofurans." Elsevier Science Publishers.
- Rhea, J., Connolly, J., and Haggard, J. (1997). "Hudson River PCBs: A 1990s Perspective." *Clearwaters*, 27(2), 24-28.
- Rhee, G.-Y., Bush, B., Bethoney, C.M., DeNucci, A., Oh, H.-M., and Sokol, R.C. (1993a). "Anaerobic dechlorination of aroclor 1242 as affected by some environmental conditions." *Environ. Toxicol. Chem.*, 12, 1033-1039.
- Rhee, G.-Y., Bush, B., Bethoney, C.M., DeNucci, A., Oh, H.-M., and Sokol, R.C. (1993b). "Reductive dechlorination of aroclor 1242 in anaerobic sediments: pattern, rate and concentration dependence." *Environ. Toxicol. Chem.*, 12, 1025-1032.
- Rhee, G.-Y., Sokol, R.C., Bush, B., and Bethoney, C.M. (1993c). "Long-term study of the anaerobic dechlorination of aroclor 1254 and without biphenyl enrichment." *Environ. Sci. Technol.*, 27(4), 714-719.
- Rossmann, L. (1998). Personal communication.
- Schwarzenbach, R.P., Gschwend, P.M., and Imboden, D.M. (1993). *Environmental Organic Chemistry*, John Wiley & Sons, Inc., New York.

- Secor, D.H., and Piccoli, P.M. (1996). "Age-and sex dependent migrations of the Hudson River Striped Bass population determined from otolith chemical microanalysis." *Estuaries*, 19(4), 778-793.
- Servos, M.R., Muir, D.C.G., and Webster, G.R.B. (1992). "Bioavailability of polychlorinated dibenzo-p-dioxins in lake enclosures." *Can. J. Fish. Aquat. Sci.*(49), 735-742.
- Setzler, E.M., Boynton, W.R., Wood, K.V., Zion, H.H., Lubbers, L., Mountford, N.K., Fere, P., Tucker, L., and Mihursky, J.A. (1980). "Synopsis of biological data on striped bass, *Morone saxatilis* (Waldbaum)." *NMFS Cir. 433, FAO Synopsis No. 121*, U.S. Dept. of Commerce, Rockville, MD.
- Suszkowski, D.J. (1998). Personal communication.
- Suszkowski, D.J. (1995). "Development of a dredged sediment contamination reduction plan for the New York/New Jersey Harbor Estuary." *Second Annual Marine and Estuarine Shallow Water Science Management Conference*, Atlantic City, N.J.
- Swaney, D.P., Sherman, D., and Howarth, R.W. (1998). "Modeling Water, Sediment, and Organic Carbon Discharges in the Hudson/Mohawk Basin: Coupling to Terrestrial Sources." , Cornell University, Ithaca NY.
- TAMS/Gradient. (1995). "Further Site Characterization and Analysis Database Report. Phase 2 Report - Review Copy." *EPA Contract No. 68-S9-2001*, U.S. Environmental Protection Agency, Region II.
- Thomann, R.V., and Connolly, J.P. (1984). "Model of PCB in the Lake Michigan Trout Food Chain." *Environ. Sci. Technol.*, 18(2), 65-71.
- Thomann, R.V., Connolly, J.P., and Parkerton, T.F. (1992a). "An equilibrium model of organic chemical accumulation in aquatic food webs with sediment interaction." *Environ. Toxicol. Chem.*, 11, 615-629.
- Thomann, R.V., Connolly, J.P., and Parkerton, T.F. (1992b). "Modeling accumulation of organic chemicals in aquatic food webs." *Chemical Dynamics in Fresh Water Ecosystems*, F.A.P.C. Gobas and J.A. McCorquodale, eds., Lewis Publishers, Chelsea, MI, 153-186.
- Thomann, R.V., Mueller, J.A., Winfield, R.P., and Huang, C.-R. (1991). "Model of fate and accumulation of PCB homologues in Hudson Estuary." *J. Environ. Engr.*, 117(2), 161-178.

- Thomann, R.V., Mueller, J.A., Winfield, R.P., and Huang, C.-R. (1989). "Mathematical Model of the Long-Term Behavior of PCBs in the Hudson River Estuary." *The Hudson River Foundation, Grant Numbers 007/87A/030, 011/88A/030*, Manhattan College, Riverdale, NY.
- Thomann, R.V., and Mueller, J. A. (1987). *Principles of Surface Water Quality Modeling and Control*, Harper & Row, New York.
- Thomann, R.V., and Salas, H.J. (1986). "Manual on Toxic Substances in Surface Waters." *CEPIS*, Pan American Health Organization, Lima, Peru.
- U.S. Army Corps of Engineers, New York District/U.S. Environmental Protection Agency, Region II. (1992). "Guidance for Performing Tests on Dredged Material Proposed for Ocean Disposal." *Draft*, U.S. Army Corps of Engineers, New York District, New York, NY.
- Waldman, J.R. (1988a). "1986 Hudson River Striped Bass tag recovery program." , Hudson River Foundation, New York, NY.
- Waldman, J.R. (1988b). Personal communication.
- Young, R.A. (1988). "A Report on Striped Bass in New York Marine Waters." , NYS Marine Fisheries, Stony Brook, NY.

Appendix A:
Model Parameters and Other Supporting Material

Table A-1 Geometric Characteristics of Model Segments								
Segment	Milepoints		Length (mi)	Avg. Width ^a (ft)	Avg. Depth (ft)	Surface Area (10 ⁷ ft ²)	Volume (10 ⁸ ft ³)	
	Upstream	Downstream					Water	Sediment ^b
Hudson River - Federal Dam @ Troy to the Battery								
1	153.5	143.5	10	620	16.5	3.3	5.4	0.011
2	143.5	133.5	10	1080	19.6	5.7	11.2	0.019
3	133.5	123.5	10	1650	15.5	10.5	16.3	0.034
4	123.5	113.5	10	2430	12.9	15.1	19.5	0.050
5	113.5	103.5	10	3090	15.0	19.8	29.7	0.065
6	103.5	93.5	10	3710	27.9	22.3	62.2	0.073
7	93.5	83.5	10	3990	33.3	22.6	75.2	0.074
8	83.5	73.5	10	2430	55.9	12.8	71.5	0.042
9	73.5	63.5	10	3370	45.2	18.2	82.2	0.060
10	63.5	53.5	10	5190	29.4	27.8	81.6	0.091
11	53.5	43.5	10	2140	82.9	11.3	93.7	0.037
12	43.5	33.5	10	9200	20.0	48.6	97.2	0.159
13	33.5	23.5	10	11950	17.7	63.1	111.5	0.207
14	23.5	13.5	10	4870	32.7	25.7	84.1	0.084
15	13.5	0	13.5	3820	35.1	27.2	95.4	0.089
Non-Hudson River Regions								
16	0	-15	Upper-Lower Bay		25.9	189.0	490.0	0.620
17	-15	-29.9	Apex		63.4	729.0	4620.0	2.390
18			NY Bight		79.6	1082.0	8610.0	3.550
19			NY Bight		91.5	3650.0	33400.0	11.980
20	-29.9	-52.4	NY Bight		105.0	1428.0	15000.0	4.690
21	-52.4	-86.2	NY Bight		155.0	4380.0	67900.0	14.400
22			NY Bight		67.3	843.0	5700.0	2.770
23			NY Bight		123.0	2730.0	33600.0	8.960
24			Kills + Newark Bay		18.9	36.0	68.0	0.118
25			Raritan Bay		17.3	223.0	385.0	0.732
26	0	16.2	East + Harlem River		32.3	36.5	118.0	0.120
27	16.2	41.1	Long Island Sound		46.6	395.0	1841.0	1.300
28	41.1	72.7	Long Island Sound		75.6	1120.0	8470.0	3.670
29	72.7	109.3	Long Island Sound		91.0	1397.0	12700.0	4.580
30	109.3	125.2	Long Island Sound		66.3	1310.0	8680.0	4.300
Note: ^a Excluding Coves, Side Channels, and Bays								
^b For 1 centimeter Depth of Sediment								

Table A-2 Seasonal Flows Across Model Interface (ft ³ /sec)		
Interface	Spring High Flow (March-April)	Summer-Fall-Winter Low Flow (May-February)
0-1	24096.5	10145.7
1-2	27164.1	10737.7
2-3	27751.7	10884.6
3-4	27711.2	10889.8
4-5	31657.5	11982.3
5-6	31600.1	11989.6
6-7	32469.3	12193.6
7-8	32420.3	12200.0
8-9	36779.1	13433.4
9-10	36782.3	13452.8
10-11	38557.6	13878.6
11-12	38569.3	13882.5
12-13	39994.9	14190.2
13-14	40238.5	14261.5
14-15	40696.2	14528.5
15-16	41168.4	15315.6
0-24	4498.1	1356.1
16-24	5498.1	2990.2
24-25	10721.5	4862.7
0-25	0.0	0.0
25-16	15235.4	6633.2
26-16	9611.9	8708.6
26-15	-8.5	413.4
17-16	62420.7	28807.5
18-17	135024.2	110097.8
19-18	1299522.8	1373524.9
17-22	94934.4	197107.2
20-17	167541.4	278400.8
0-23	6559070.3	5565576.3
21-20	416781.1	922815.5
21-19	769089.5	1199214.4
23-22	820303.2	421910.3
23-21	5738943.7	5143652.6
22-20	915260.6	619011.2
21-0	4577239.0	3018481.0
20-18	1164495.0	1263425.9
19-0	2075658.5	2571997.6
27-26	7313.3	7126.8
28-27	6059.5	6646.8
29-28	-4046.6	3386.2
29-30	53944.0	11276.7
30-0	63095.4	12537.7

Table A-3 Tidal Dispersion Characteristics				
Interface	Dispersion Coefficient (mi ² /day)	X-sect. area (10 ⁴ ft ²)	Length i (10 ³ ft)	Length j (10 ³ ft)
0-1	0.00	1.23	52.80	52.80
1-2	1.25	1.70	52.80	52.80
2-3	1.25	1.99	52.80	52.80
3-4	1.25	2.72	52.80	52.80
4-5	1.25	4.69	52.80	52.80
5-6	1.25	6.86	52.80	52.80
6-7	1.25	11.30	52.80	52.80
7-8	1.75	10.60	52.80	52.80
8-9	2.00	11.10	52.80	52.80
9-10	4.00	16.30	52.80	52.80
10-11	6.00	14.30	52.80	52.80
11-12	7.50	21.00	52.80	52.80
12-13	10.00	18.20	52.80	52.80
13-14	12.50	16.50	52.80	52.80
14-15	15.00	13.60	52.80	71.28
15-16	20.00	19.40	71.28	79.20
15-26	10.00	0.63	5.28	5.28
16-17	15.00	79.00	51.00	51.00
16-24	15.00	17.10	5.28	5.28
16-25	7.50	154.60	13.00	13.00
16-26	20.00	16.00	10.00	10.00
17-18	12.00	339.00	114.00	114.00
17-20	12.00	615.00	76.00	76.00
17-22	12.00	251.00	94.00	94.00
18-19	12.00	1150.00	118.00	118.00
18-20	10.00	1770.00	100.00	100.00
19-0	12.00	2880.00	177.00	177.00
19-21	10.00	2360.00	180.00	180.00
20-21	12.00	2080.00	119.00	119.00
20-22	10.00	1080.00	73.00	73.00
21-0	12.00	5870.00	178.00	178.00
21-23	10.00	2860.00	167.00	167.00
22-23	12.00	810.00	118.00	118.00
23-0	12.00	2820.00	180.00	180.00
24-25	15.00	7.05	6.00	6.00
26-27	10.00	20.60	77.00	77.00
27-28	10.00	272.00	124.00	124.00
28-29	10.00	761.00	141.00	141.00
29-30	10.00	435.00	100.00	100.00
30-0	10.00	780.00	80.00	80.00
0-24	0.00	4.12	50.00	50.00
0-25	0.00	3.84	80.00	80.00

Table A-4 Summary of Sediment Loadings Into Model Segments (Mtons/year)								
Segment	Rivers	CSO/SW	WWTP	Industry	Atmospheric	Productivity	Boundary Inputs	Total
1	461,506	0	0	0	6	581	0	462,094
2	15,852	0	0	0	10	1,004	0	16,867
3	15,852	0	0	0	19	1,850	0	17,721
4	15,852	0	0	0	27	2,661	0	18,540
5	15,852	0	0	0	35	3,489	0	19,376
6	15,852	0	0	0	40	3,929	0	19,821
7	15,852	0	0	0	40	3,982	0	19,875
8	15,852	0	0	0	23	2,255	0	18,130
9	15,852	0	0	0	32	3,207	0	19,092
10	15,852	5,575	0	0	49	4,899	0	26,375
11	15,852	0	0	0	20	1,991	0	17,863
12	15,852	0	0	0	86	8,564	0	24,502
13	15,852	0	0	0	112	11,119	0	27,083
14	15,852	0	0	0	46	4,528	0	20,426
Subtotal HR (1-14)	667,586	5,575	0	0	545	54,060	0	727,766
15	0	871	19,186	106	48	4,793	0	25,004
16	0	10,337	34,787	15	336	33,303	0	78,777
24	35,659	37,216	9,451	2,060	64	6,343	0	90,793
25	35,695	970	88	95	396	39,294	0	76,538
26	0	16,466	32,071	0	65	6,432	0	55,033
Subtotal NY-NJH	71,354	65,859	95,582	2,275	909	90,164	0	326,145
27	0	0	0	0	702	69,601	0	70,303
28	10,329	0	0	0	1,990	197,350	0	209,670
29	113,721	0	0	0	2,483	246,159	0	362,363
30	11,720	0	0	0	2,328	230,829	1,474	246,351
Subtotal LIS	135,770	0	0	0	7,503	743,940	1,474	888,687
17	0	0	0	0	1,295	128,454	0	129,749
18	0	0	0	0	1,923	190,654	0	192,577
19	0	0	0	0	6,486	643,150	1,476	651,113
20	0	0	0	0	2,538	251,621	0	254,159
21	0	0	0	0	7,783	771,780	2,992	782,556
22	0	0	0	0	1,498	148,541	0	150,039
23	0	0	0	0	4,851	481,041	14,464	500,356
Subtotal NY Bight	0	0	0	0	26,375	2,615,242	18,932	2,660,549
Total	874,710	71,434	95,582	2,275	35,332	3,503,406	20,407	4,603,147

Table A-5
Summary of New York-New Jersey Harbor Dredging (from Army Corps of Engineers Records (Suszkowski, personal communication))

Annual Summary of Harbor Dredging (MCY)										Excluded Dredged Material	
Year	Model Segments								Total	Entrance to	
	11	13	15	16	24	25	26	27		Ambrose Channel ^a	New Construction ^b
1981	0.000	0.000	0.445	0.468	0.929	0.221	0.248	0.019	2.331	0.000	0.000
1982	0.000	0.000	0.510	0.339	1.880	0.971	0.620	0.000	4.320	0.000	0.000
1983	0.000	0.000	0.270	0.763	1.804	0.121	1.205	0.000	4.163	0.000	0.000
1984	0.000	0.000	0.982	2.668	2.963	0.180	0.222	0.000	7.015	0.285	7.300
1985	0.000	0.000	0.424	2.631	0.376	1.827	0.345	0.000	5.603	0.000	0.000
1986	0.000	0.000	0.421	1.066	1.009	0.617	0.479	0.000	3.592	0.000	0.000
1987	0.159	0.000	0.614	0.443	0.829	0.743	0.081	0.000	2.869	1.185	4.054
1988	0.000	0.099	0.384	0.257	0.738	0.053	0.661	0.000	2.193	0.178	2.371
1989	0.000	0.000	0.483	0.871	0.459	0.048	0.188	0.000	2.050	3.941	5.991
1990	0.000	0.000	0.521	0.686	0.403	1.331	0.111	0.053	3.104	0.000	4.496
1991	0.000	0.000	0.444	0.339	0.042	0.628	0.523	0.037	2.013	0.000	0.000
Totals	0.159	0.099	5.499	10.532	11.431	6.742	4.682	0.109	39.253	5.589	24.212
Average Dredging of Fine-Grained Sediment (MCY)											
Years	11	13	15	16	24	25	26	27	Total		
1981 - 1991	0.014	0.009	0.500	0.957	1.039	0.613	0.426	0.010	3.568		
1987 - 1991	0.032	0.020	0.489	0.519	0.494	0.561	0.313	0.018	2.446		
^a - Dredged material from entrance to Ambrose Channel is assumed to be comprised of sands from the ocean and is not considered in estimating average annual dredging rate of fine grained material.											
^b - New construction not included in estimating average annual dredging rate of fine grained material.											
Note: For conversion from MCY to Mtons/year, see text.											

Table A-6 Summary of Annual Sediment and Dredged Material Accumulation Rates (Mtons/year) and Seasonal Distributions for Accumulation						
Dredging - Accumulation Summary						
Segment	Burial Rate (cm/year)		Sediment Accumulation (Mtons/yr)	Dredged Material Accumulation (Mtons/yr)	% Accumulation	
	Spring	10 Months			Spring	10 Months
1	5.320	0.573	3,828	20,132	65%	35%
2	1.242	0.134	6,612	7,103	65%	35%
3	0.854	0.092	12,179	8,233	65%	35%
4	0.623	0.067	17,515	7,693	65%	35%
5	0.409	0.044	22,967	5,033	65%	35%
6	0.238	0.026	25,867	1,138	65%	35%
7	0.267	0.029	26,214	1,923	65%	35%
8	0.195	0.021	14,847	0	65%	35%
9	0.195	0.021	21,111	0	65%	35%
10	0.195	0.021	32,246	0	65%	35%
11	0.462	0.050	13,107	3,589	65%	35%
12	0.195	0.021	56,373	0	65%	35%
13	0.261	0.028	73,192	4,938	65%	35%
14	0.195	0.021	29,810	0	65%	35%
Subtotal HR (1-14)			355,868	59,782		
15	2.736	0.669	31,550	121,660	45%	55%
16	0.532	0.130	219,227	129,081	45%	55%
24	2.120	0.518	41,758	122,866	45%	55%
25	0.377	0.092	25,867	139,427	45%	55%
26	1.253	0.306	4,234	77,770	45%	55%
Subtotal NY-NJH			322,636	590,804		
27	0.020	0.005	45,817	4,478	45%	55%
28	0.014	0.003	129,913	0	45%	55%
29	0.014	0.003	162,043	0	45%	55%
30	0.014	0.003	151,951	0	45%	55%
Subtotal LIS			489,724	4,478		
17	0.008	0.005	84,559	-513,034	25%	75%
18	0.008	0.005	125,505	0	25%	75%
19	0.008	0.005	423,376	0	25%	75%
20	0.008	0.005	165,639	0	25%	75%
21	0.008	0.005	508,051	0	25%	75%
22	0.008	0.005	97,782	0	25%	75%
23	0.008	0.005	316,662	0	25%	75%
Subtotal NYB			1,721,573	-513,034		
Totals			2,336,454	1,310,127		

Table A-7
Calculated Suspended Solids Concentration and Sediment Resuspension Rates

Seasonal Sediment Transport Summary

Segment	RM	Spring High Flow Period		10 Month Low Flow Period	
		Suspended Solids mg/L	Resuspension Rate cm/yr	Suspended Solids mg/L	Resuspension Rate cm/yr
1	148.5	70.9	10.45	19.4	3.74
2	138.5	70.6	14.47	19.5	4.21
3	128.5	71.5	15.06	19.9	4.34
4	118.5	63.4	13.49	18.6	4.06
5	108.5	64.5	13.94	19.1	4.21
6	98.5	64.2	14.05	19.6	4.33
7	88.5	65.4	14.28	20.1	4.45
8	78.5	59.5	13.04	19.1	4.22
9	68.5	60.8	13.34	19.7	4.37
10	58.5	60.0	13.16	20.0	4.44
11	48.5	60.7	13.05	20.3	4.46
12	38.5	59.3	13.00	20.0	4.44
13	28.5	58.2	12.69	19.0	4.20
14	18.5	55.1	12.06	16.2	3.58
15	8.5	37.3	5.57	8.6	1.24
16	-2	11.2	1.96	2.6	0.46
17	-25	3.1	0.69	1.0	0.21
18		1.6	0.35	1.0	0.22
19		1.3	0.28	1.0	0.21
20		1.4	0.29	0.9	0.20
21		0.8	0.16	0.7	0.16
22		1.2	0.25	0.9	0.19
23		0.6	0.12	0.6	0.13
24		10.4	0.20	2.4	0.01
25		10.3	1.91	2.5	0.45
26		10.5	1.08	3.4	0.44
27		8.5	1.87	7.8	1.72
28		7.9	1.75	7.4	1.64
29		7.1	1.56	6.2	1.38
30		3.7	0.81	3.2	0.71

Note: Settling velocity over all segments is 10 ft/day.

Table A-8					
POC, DOC, Suspended Solids, and Computed foc Values for the Two Month (March-April) Spring High Flow Period					
Low Flow Segment	RM	POC mg/L	DOC mg/L	m mg/L	foc
1	148.5	1.52	3.35	19.37	0.08
2	138.5	1.52	3.35	19.54	0.08
3	128.5	1.52	3.35	19.91	0.08
4	118.5	1.52	3.35	18.57	0.08
5	108.5	1.52	3.35	19.13	0.08
6	98.5	1.52	3.35	19.57	0.08
7	88.5	1.52	3.35	20.12	0.08
8	78.5	1.62	3.39	19.07	0.08
9	68.5	1.42	3.30	19.74	0.07
10	58.5	1.09	3.10	20.05	0.05
11	48.5	0.83	2.90	20.27	0.04
12	38.5	0.77	2.75	20.03	0.04
13	28.5	0.78	2.72	19.01	0.04
14	18.5	0.74	2.73	16.21	0.05
15	8.5	0.81	2.63	8.59	0.09
16	-2	1.73	2.66	2.64	0.61
17	-25	0.87	1.29	0.98	0.86
18		0.63	1.13	1.00	0.62
19		0.53	1.07	0.97	0.54
20		0.46	1.05	0.91	0.50
21		0.34	1.13	0.73	0.46
22		0.71	1.14	0.88	0.80
23		0.50	1.12	0.60	0.83
24		4.07	4.73	2.38	1.57
25		2.15	2.61	2.46	0.79
26		1.07	1.90	3.37	0.31
27		0.78	0.99	7.75	0.11
28		0.51	0.83	7.40	0.07
29		0.46	0.95	6.22	0.08
30		0.43	1.17	3.21	0.13

Since our solids balance is based on fixed (inorganic plus refractory organics) solids, foc values in excess of one are possible, particularly for areas of the harbor and the bight where concentrations of phytoplankton and sewage organic carbon are greater than fixed solids concentrations.

Note:

For segments #8 - 30, POC concentrations were determined directly from 1989 SWEM results collapsed onto our 30 segment water column grid.

For segments #1 - 7, POC and DOC concentrations were assumed to be the average of 1989 SWEM results for segments #8 and 9.

Table A-9
POC, DOC, Suspended Solids, and Computed foc Values For the Ten
Month (May-February) Summer-Fall-Winter Low Flow Period

High Flow Segment	RM	POC mg/L	DOC mg/L	m mg/L	foc
1	148.5	1.12	3.37	70.88	0.02
2	138.5	1.12	3.37	70.60	0.02
3	128.5	1.12	3.37	71.53	0.02
4	118.5	1.12	3.37	63.42	0.02
5	108.5	1.12	3.37	64.50	0.02
6	98.5	1.12	3.37	64.23	0.02
7	88.5	1.12	3.37	65.40	0.02
8	78.5	1.15	3.39	59.48	0.02
9	68.5	1.09	3.35	60.82	0.02
10	58.5	0.97	3.27	60.03	0.02
11	48.5	0.88	3.19	60.72	0.01
12	38.5	0.82	3.12	59.32	0.01
13	28.5	0.75	3.07	58.20	0.01
14	18.5	0.73	3.05	55.07	0.01
15	8.5	0.92	2.95	37.32	0.02
16	-2	1.91	2.74	11.19	0.16
17	-25	1.16	1.50	3.12	0.36
18		0.93	1.34	1.60	0.57
19		0.73	1.21	1.28	0.56
20		0.69	1.34	1.35	0.50
21		0.57	1.42	0.75	0.76
22		0.87	1.35	1.16	0.73
23		0.64	1.43	0.59	1.07
24		2.78	4.38	10.44	0.25
25		2.15	2.75	10.26	0.19
26		1.07	1.99	10.46	0.12
27		0.78	1.00	8.51	0.14
28		0.51	0.90	7.95	0.10
29		0.46	1.10	7.09	0.12
30		0.43	1.35	3.69	0.18

Since our solids balance is based on fixed (inorganic plus refractory organics) solids, foc values in excess of one are possible, particularly for areas of the harbor and the bight where concentrations of phytoplankton and sewage organic carbon are greater than fixed solids concentrations.

Note:

For segments #8 - 30, POC concentrations were determined directly from 1989 SWEM results collapsed onto our 30 segment water column grid.

For segments #1 - 7, POC and DOC concentrations were assumed to be the average of 1989 SWEM results for segments #8 and 9.

Table A-10 Summary of PCB Loads (lb/day) for the Mohawk River and the New Jersey Tributaries									
Mohawk River			Pounds/day						
	Month	Flow (cfs)	Di	Tri	Tetra	Penta	Hexa	Total (Di-Hexa)	Total
Segment 1	Oct-88	2349.00							
	Nov-88	7286.67							
	Dec-88	3800.97							
	Jan-89	2657.10							
	Feb-89	3698.57							
	Mar-89	5801.29							
	Apr-89	9733.00							
	May-89	10391.29							
	Jun-89	7654.67							
	Jul-89	2499.68							
	Aug-89	1739.58							
	Sep-89	2507.27							
Water Year Average		5009.92	0.0182	0.0750	0.0892	0.1112	0.0607	0.3542	0.3704
* Passaic & Hackensack Rivers			Pounds/day						
	Month	Flow (cfs)	Di	Tri	Tetra	Penta	Hexa	Total (Di-Hexa)	Total
Segment 24	Oct-88	529.80							
	Nov-88	2119.19							
	Dec-88	706.40							
	Jan-89	883.00							
	Feb-89	883.00							
	Mar-89	1236.19							
	Apr-89	2295.79							
	May-89	6180.97							
	Jun-89	2648.99							
	Jul-89	1236.19							
	Aug-89	1236.19							
	Sep-89	1589.39							
Water Year Average		1795.43	0.0065	0.0269	0.0320	0.0399	0.0218	0.1269	0.1328
* Raritan River			Pounds/day						
	Month	Flow (cfs)	Di	Tri	Tetra	Penta	Hexa	Total (Di-Hexa)	Total
Segment 25	Oct-88	353.20							
	Nov-88	2295.79							
	Dec-88	883.00							
	Jan-89	883.00							
	Feb-89	1412.79							
	Mar-89	1589.39							
	Apr-89	2119.19							
	May-89	4591.58							
	Jun-89	2825.59							
	Jul-89	1765.99							
	Aug-89	1059.60							
	Sep-89	2472.39							
Water Year Average		1854.29	0.0067	0.0277	0.0330	0.0412	0.0225	0.1311	0.1371
Note: * PCB homologue concentrations for the Passaic, Hackensack, and Raritan Rivers were assumed to be equal to the average of measured concentrations for the Mohawk River [Data from TAMS/Gradient, 1997]									

Table A-11
PCB Homologue Loads From the Fourteen New York City Wastewater Treatment Plants in lbs/day

NYC	WWTP	Flow (MGD)	Mono	Di	Tri	Tetra	Penta	Hexa	Hepta	Octa	Nona	Deca	Total
	Rockaway	26.50	9.23E-04	3.93E-04	1.80E-03	7.71E-04	1.24E-03	3.41E-04	8.43E-05	4.87E-05	6.33E-06	3.43E-04	5.95E-03
	North River	160.50	8.15E-03	6.58E-03	7.21E-03	4.97E-03	8.55E-03	5.53E-03	1.53E-03	6.23E-04	2.75E-05	4.23E-04	4.36E-02
	Coney Island	114.50	7.14E-04	3.05E-03	6.89E-03	2.33E-03	4.47E-03	2.26E-03	7.53E-04	5.52E-04	9.39E-05	1.15E-03	2.23E-02
	Wards Island	258.00	3.16E-03	8.95E-03	9.37E-03	3.99E-03	5.70E-03	2.13E-03	6.82E-04	3.97E-04	5.82E-05	1.47E-03	3.59E-02
	Red Hook	41.00	3.35E-04	9.13E-04	1.43E-03	1.01E-03	9.82E-04	6.70E-04	1.94E-04	1.14E-04	1.56E-04	1.52E-04	5.96E-03
	Oakwood Beach	26.50	3.63E-04	3.97E-04	7.41E-04	3.79E-04	3.11E-04	1.99E-04	2.90E-05	2.14E-05	8.00E-06	1.59E-04	2.61E-03
	Owls Head	124.00	1.85E-02	7.59E-03	7.43E-03	3.39E-03	4.97E-03	1.72E-03	4.33E-04	2.64E-04	4.98E-05	4.10E-04	4.47E-02
	Tallman Island	55.00	3.25E-03	1.12E-03	5.49E-03	1.51E-03	1.77E-03	8.75E-04	1.97E-04	8.65E-05	9.92E-06	1.97E-04	1.45E-02
	Hunts Point	148.00	1.15E-03	5.76E-03	5.38E-03	3.49E-03	5.02E-03	1.99E-03	4.46E-04	2.00E-04	4.47E-05	4.34E-04	2.39E-02
	Port Richmond	34.50	3.18E-03	1.58E-03	1.51E-03	1.03E-03	9.40E-04	3.79E-04	1.78E-04	4.96E-05	2.64E-06	1.44E-04	8.98E-03
	26th Ward	66.50	1.42E-04	1.08E-03	1.47E-03	9.73E-04	1.17E-03	5.13E-04	9.08E-05	5.01E-05	3.33E-05	1.65E-04	5.70E-03
	Jamaica	81.00	8.16E-04	9.57E-04	2.81E-03	2.06E-03	4.90E-03	1.91E-03	4.88E-04	2.23E-04	4.60E-05	6.14E-04	1.48E-02
	Newtown Creek	286.00	3.35E-02	1.82E-02	2.32E-02	1.25E-02	2.64E-02	9.97E-03	3.16E-03	5.57E-03	3.70E-03	2.46E-03	1.39E-01
	Bowery Bay	126.00	1.83E-03	2.91E-03	2.65E-03	3.28E-03	2.96E-03	1.54E-03	3.87E-04	1.99E-04	2.27E-05	8.47E-04	1.66E-02
	Totals	1548.00	7.59E-02	5.95E-02	7.74E-02	4.17E-02	6.93E-02	3.00E-02	8.65E-03	8.40E-03	4.26E-03	8.97E-03	3.84E-01

Notes: Loads = $Q \cdot \text{Avg. } C$ (Avg. C From avg. effluent concentration for two 1994 dry weather events - Batelle 1995)

Congener 180 experienced matrix interferences - treated as zero.

The sum of congeners in this study was 92.5% of all congeners, so loads were adjusted to take into account all congeners.

Table A-12
PCB Homologue Loads from the Twelve New Jersey Wastewater Treatment Plants in lbs/day

NJ	WWTP	Flow (MGD)	Mono	Di	Tri	Tetra	Penta	Hexa	Hepta	Octa	Nona	Deca	Total
	West New York	10.80	4.23E-05	2.82E-04	3.05E-04	2.70E-04	3.37E-04	2.40E-04	7.82E-05	4.08E-05	3.36E-06	5.89E-06	1.60E-03
	Rahway Valley	25.95	9.91E-04	3.37E-04	1.57E-03	1.21E-03	1.26E-03	6.78E-04	1.97E-04	6.54E-05	9.71E-06	8.14E-05	6.40E-03
	N. Bergen-Central	6.90	4.51E-04	2.27E-04	3.10E-04	4.08E-04	8.98E-04	5.73E-04	1.48E-04	2.45E-05	3.55E-06	8.15E-06	3.05E-03
	Joint Meeting	59.20	8.19E-04	3.90E-03	1.01E-02	3.56E-03	2.35E-03	1.04E-03	2.70E-04	7.61E-05	2.16E-05	8.14E-05	2.22E-02
	N. Bergen-Woodcliff	3.00	1.37E-04	1.17E-04	1.36E-04	9.88E-05	2.21E-04	1.04E-04	4.50E-05	7.84E-06	2.35E-06	3.77E-06	8.73E-04
	Linden Roselle	13.30	1.98E-03	3.42E-04	6.78E-04	9.74E-04	1.71E-03	1.20E-03	1.64E-04	5.56E-05	7.37E-06	2.02E-05	7.12E-03
	Passaic Valley	282.50	4.50E-02	1.92E-02	1.48E-02	9.02E-03	7.91E-03	5.28E-03	2.58E-03	4.84E-04	5.73E-05	1.72E-04	1.05E-01
	Secaucus	2.70	5.04E-05	6.97E-05	5.92E-05	5.41E-05	1.16E-04	5.46E-05	1.91E-05	5.96E-06	9.74E-07	2.60E-06	4.33E-04
	Tri City	11.80	3.60E-04	3.86E-04	4.20E-04	2.78E-04	3.82E-04	1.72E-04	1.01E-04	1.25E-05	1.81E-06	8.83E-06	2.12E-03
	Bergen County	68.80	1.62E-03	1.71E-03	2.07E-03	1.81E-03	3.38E-03	8.95E-04	1.96E-04	4.99E-05	8.06E-06	1.55E-05	1.18E-02
	Edgewater	3.45	1.20E-04	7.66E-05	9.43E-05	5.69E-05	8.28E-05	4.24E-05	1.34E-05	3.73E-06	7.47E-07	7.15E-07	4.91E-04
	Middlesex	115.00	0.00E+00	8.08E-03	3.85E-03	4.31E-03	4.30E-03	3.76E-03	6.11E-04	2.65E-04	8.92E-05	3.21E-04	2.56E-02
	Totals	603.40	5.16E-02	3.48E-02	3.44E-02	2.21E-02	2.29E-02	1.40E-02	4.42E-03	1.09E-03	2.06E-04	7.21E-04	1.86E-01

Notes: *Loads = Q * Avg. C (Avg. C From avg. effluent concentration for two 1994 dry weather events - Batelle 1995)*
 Congener 180 experienced matrix interferences - treated as zero.
 The sum of congeners in this study was 92.5% of all congeners, so loads were adjusted to take into account all congeners.

Table A-13
PCB Homologue Loads from the New York City and New Jersey CSOs in lbs/day

NYC	CSO	Flow (MGD)	Mono	Di	Tri	Tetra	Penta	Hexa	Hepta	Octa	Nona	Deca	Total
	Rockaway	5.36	1.10E-03	8.45E-04	3.96E-04	5.68E-04	4.27E-04	3.16E-04	9.16E-05	5.24E-05	9.08E-06	2.91E-05	3.83E-03
	North River	5.11	6.78E-04	9.93E-04	1.11E-03	1.13E-03	2.14E-03	3.90E-03	1.60E-03	6.47E-04	8.83E-05	8.49E-05	1.24E-02
	Coney Island	9.24	3.97E-03	3.43E-03	7.66E-04	1.78E-03	1.89E-03	1.27E-03	3.34E-04	2.53E-04	5.49E-05	5.95E-05	1.38E-02
	Wards Island	11.96	2.28E-03	2.35E-03	1.88E-03	3.36E-03	6.06E-03	4.46E-03	1.22E-03	6.89E-04	1.23E-04	8.19E-05	2.25E-02
	Red Hook	4.98	4.63E-04	5.63E-04	2.10E-04	8.73E-04	9.73E-04	1.09E-03	4.15E-04	2.34E-04	2.34E-05	5.17E-05	4.89E-03
	Oakwood Beach	0.00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
	Owls Head	15.90	1.52E-03	1.08E-03	1.38E-03	2.63E-03	5.11E-03	4.52E-03	1.21E-03	6.32E-04	1.09E-04	2.19E-03	2.04E-02
	Tallman Island	11.63	1.46E-03	1.36E-03	6.88E-04	1.50E-03	2.16E-03	2.37E-03	1.02E-03	4.12E-04	4.08E-05	9.80E-05	1.11E-02
	Hunts Point	20.75	3.33E-03	2.17E-03	2.47E-03	3.48E-03	2.77E-03	2.08E-03	5.98E-04	2.53E-04	3.90E-05	5.49E-05	1.72E-02
	Port Richmond	11.70	2.22E-03	2.80E-03	2.10E-03	2.29E-03	2.80E-03	2.51E-03	7.25E-04	3.25E-04	4.96E-05	3.68E-04	1.62E-02
	26th Ward	4.27	1.74E-03	1.14E-03	7.97E-04	1.67E-03	1.34E-03	8.98E-04	2.36E-04	9.81E-05	1.36E-05	2.76E-05	7.97E-03
	Jamaica	21.85	6.33E-03	7.29E-03	2.71E-03	6.36E-03	4.13E-03	1.90E-03	5.29E-04	2.69E-04	8.21E-05	3.85E-03	3.35E-02
	Newtown Creek	23.01	5.06E-03	2.30E-03	2.84E-03	5.32E-03	7.96E-03	6.21E-03	2.14E-03	1.18E-03	1.61E-04	9.05E-05	3.33E-02
	Bowery Bay	17.90	4.45E-03	2.69E-03	1.57E-03	3.14E-03	4.66E-03	4.33E-03	1.48E-03	8.24E-04	9.19E-05	1.47E-04	2.34E-02
	Totals	163.66	3.46E-02	2.90E-02	1.89E-02	3.41E-02	4.24E-02	3.59E-02	1.16E-02	5.87E-03	8.86E-04	7.13E-03	2.20E-01
NJ	CSO	Flow (MGD)	Mono	Di	Tri	Tetra	Penta	Hexa	Hepta	Octa	Nona	Deca	Total
	All Service Areas	61.30	7.81E-03	1.09E-02	1.69E-02	1.79E-02	1.60E-02	1.18E-02	4.77E-03	2.82E-03	5.29E-04	2.06E-04	8.97E-02

Notes: Flows From HydroQual, 1991 For All New York City WWTP Service Areas And Total of NJ WWTP Service Areas
 Loads = $Q \cdot \text{Avg. } C$ (Avg. C from avg influent concentration from two 1994 wet weather events - Battelle, 1995)
 Congener 180 experienced matrix interferences - treated as zero
 The sum of congeners in this study was 92.5% of all congeners, so loads were adjusted to take into account all congeners.

Table A-14 PCB Atmospheric From Wet Deposition in lbs/day							
Segment		Di	Tri	Tetra	Penta	Hexa	Total (Di-Hexa)
1		7.73E-06	4.74E-06	1.69E-05	1.97E-05	4.36E-06	5.34E-05
2		1.33E-05	8.18E-06	2.92E-05	3.41E-05	7.52E-06	9.23E-05
3		2.46E-05	1.51E-05	5.37E-05	6.28E-05	1.39E-05	1.70E-04
4		3.54E-05	2.17E-05	7.72E-05	9.03E-05	1.99E-05	2.44E-04
5		4.64E-05	2.84E-05	1.01E-04	1.18E-04	2.61E-05	3.21E-04
6		5.22E-05	3.20E-05	1.14E-04	1.33E-04	2.94E-05	3.61E-04
7		5.29E-05	3.24E-05	1.16E-04	1.35E-04	2.98E-05	3.66E-04
8		3.00E-05	1.84E-05	6.55E-05	7.65E-05	1.69E-05	2.07E-04
9		4.26E-05	2.61E-05	9.31E-05	1.09E-04	2.40E-05	2.95E-04
10		6.51E-05	3.99E-05	1.42E-04	1.66E-04	3.67E-05	4.50E-04
11		2.65E-05	1.62E-05	5.78E-05	6.76E-05	1.49E-05	1.83E-04
12		1.14E-04	6.98E-05	2.49E-04	2.91E-04	6.41E-05	7.87E-04
13		1.48E-04	9.06E-05	3.23E-04	3.77E-04	8.33E-05	1.02E-03
14		6.02E-05	3.69E-05	1.31E-04	1.54E-04	3.39E-05	4.16E-04
Subtotal	HR	7.18E-04	4.40E-04	1.57E-03	1.83E-03	4.05E-04	4.97E-03
15		6.37E-05	3.90E-05	1.39E-04	1.63E-04	3.59E-05	4.40E-04
16		4.43E-04	2.71E-04	9.67E-04	1.13E-03	2.49E-04	3.06E-03
24		8.43E-05	5.17E-05	1.84E-04	2.15E-04	4.75E-05	5.83E-04
25		5.22E-04	3.20E-04	1.14E-03	1.33E-03	2.94E-04	3.61E-03
26		8.55E-05	5.24E-05	1.87E-04	2.18E-04	4.82E-05	5.91E-04
Subtotal	NY/NJH	1.20E-03	7.34E-04	2.62E-03	3.06E-03	6.75E-04	8.28E-03
17		1.71E-03	1.05E-03	3.73E-03	4.36E-03	9.62E-04	1.18E-02
18		2.53E-03	1.55E-03	5.53E-03	6.47E-03	1.43E-03	1.75E-02
19		8.55E-03	5.24E-03	1.87E-02	2.18E-02	4.82E-03	5.91E-02
20		3.34E-03	2.05E-03	7.30E-03	8.54E-03	1.88E-03	2.31E-02
21		1.03E-02	6.29E-03	2.24E-02	2.62E-02	5.78E-03	7.09E-02
22		1.97E-03	1.21E-03	4.31E-03	5.04E-03	1.11E-03	1.36E-02
23		6.39E-03	3.92E-03	1.40E-02	1.63E-02	3.60E-03	4.42E-02
Subtotal	NYB	3.30E-02	2.03E-02	7.22E-02	8.44E-02	1.86E-02	2.28E-01
27		9.25E-04	5.67E-04	2.02E-03	2.36E-03	5.21E-04	6.39E-03
28		2.62E-03	1.61E-03	5.73E-03	6.70E-03	1.48E-03	1.81E-02
29		3.27E-03	2.01E-03	7.15E-03	8.35E-03	1.84E-03	2.26E-02
30		3.07E-03	1.88E-03	6.70E-03	7.83E-03	1.73E-03	2.12E-02
Subtotal	LIS	9.89E-03	6.06E-03	2.16E-02	2.52E-02	5.57E-03	6.84E-02
Totals		4.48E-02	2.75E-02	9.80E-02	1.15E-01	2.53E-02	3.10E-01
Note: Rainfall rate specified as 42.7 inches/year [Thomann et al, 1989]							

Table A-15 PCB Atmospheric Loading From Back Diffusion from the Atmosphere							
Segment		Di	Tri	Tetra	Penta	Hexa	Total (Di-Hexa)
1		3.27E-05	6.11E-05	8.34E-05	4.56E-05	2.32E-05	2.46E-04
2		5.65E-05	1.06E-04	1.44E-04	7.88E-05	4.01E-05	4.25E-04
3		1.04E-04	1.94E-04	2.65E-04	1.45E-04	7.39E-05	7.83E-04
4		1.50E-04	2.80E-04	3.82E-04	2.09E-04	1.06E-04	1.13E-03
5		1.96E-04	3.67E-04	5.00E-04	2.74E-04	1.39E-04	1.48E-03
6		2.21E-04	4.13E-04	5.64E-04	3.08E-04	1.57E-04	1.66E-03
7		2.24E-04	4.18E-04	5.71E-04	3.12E-04	1.59E-04	1.69E-03
8		1.27E-04	2.37E-04	3.23E-04	1.77E-04	9.01E-05	9.54E-04
9		1.80E-04	3.37E-04	4.60E-04	2.52E-04	1.28E-04	1.36E-03
10		2.75E-04	5.15E-04	7.03E-04	3.84E-04	1.96E-04	2.07E-03
11		1.12E-04	2.09E-04	2.86E-04	1.56E-04	7.96E-05	8.43E-04
12		4.82E-04	9.00E-04	1.23E-03	6.72E-04	3.42E-04	3.62E-03
13		6.25E-04	1.17E-03	1.59E-03	8.72E-04	4.44E-04	4.70E-03
14		2.55E-04	4.76E-04	6.50E-04	3.55E-04	1.81E-04	1.92E-03
Subtotal	HR	3.04E-03	5.68E-03	7.75E-03	4.24E-03	2.16E-03	2.29E-02
15		2.70E-04	5.04E-04	6.87E-04	3.76E-04	1.92E-04	2.03E-03
16		1.87E-03	3.50E-03	4.78E-03	2.61E-03	1.33E-03	1.41E-02
24		3.57E-04	6.66E-04	9.10E-04	4.98E-04	2.53E-04	2.68E-03
25		2.21E-03	4.13E-03	5.64E-03	3.08E-03	1.57E-03	1.66E-02
26		3.62E-04	6.76E-04	9.22E-04	5.05E-04	2.57E-04	2.72E-03
Subtotal	NY/NJH	5.07E-03	9.47E-03	1.29E-02	7.08E-03	3.60E-03	3.82E-02
17		7.22E-03	1.35E-02	1.84E-02	1.01E-02	5.13E-03	5.44E-02
18		1.07E-02	2.00E-02	2.73E-02	1.50E-02	7.62E-03	8.07E-02
19		3.62E-02	6.76E-02	9.22E-02	5.05E-02	2.57E-02	2.72E-01
20		1.42E-02	2.64E-02	3.61E-02	1.97E-02	1.01E-02	1.06E-01
21		4.34E-02	8.11E-02	1.11E-01	6.06E-02	3.08E-02	3.27E-01
22		8.35E-03	1.56E-02	2.13E-02	1.17E-02	5.94E-03	6.29E-02
23		2.71E-02	5.05E-02	6.90E-02	3.77E-02	1.92E-02	2.04E-01
Subtotal	NYB	1.47E-01	2.75E-01	3.75E-01	2.05E-01	1.05E-01	1.11E+00
27		3.91E-03	7.31E-03	9.98E-03	5.46E-03	2.78E-03	2.95E-02
28		1.11E-02	2.07E-02	2.83E-02	1.55E-02	7.89E-03	8.35E-02
29		1.38E-02	2.59E-02	3.53E-02	1.93E-02	9.84E-03	1.04E-01
30		1.30E-02	2.43E-02	3.31E-02	1.81E-02	9.22E-03	9.77E-02
Subtotal	LIS	4.18E-02	7.82E-02	1.07E-01	5.84E-02	2.97E-02	3.15E-01
Totals		1.97E-01	3.68E-01	5.02E-01	2.75E-01	1.40E-01	1.48E+00
Note: Dimensionless Henry's constant set equal to 4×10^{-3} . The value represents a low estimate of KH (based on congener-specific values reported in Dunnivant et al, [1992]) and is used in determining a conservative estimate of PCB atmospheric loadings from gas exchange.							

Table A-16							
Total PCB Atmospheric Loading from Wet Deposition and Back-Diffusion from the Atmosphere							
Segment		Di	Tri	Tetra	Penta	Hexa	Total (Di-Hexa)
1		4.04E-05	6.58E-05	1.00E-04	6.54E-05	2.76E-05	2.99E-04
2		6.98E-05	1.14E-04	1.73E-04	1.13E-04	4.77E-05	5.17E-04
3		1.29E-04	2.09E-04	3.19E-04	2.08E-04	8.78E-05	9.53E-04
4		1.85E-04	3.01E-04	4.59E-04	2.99E-04	1.26E-04	1.37E-03
5		2.43E-04	3.95E-04	6.02E-04	3.92E-04	1.66E-04	1.80E-03
6		2.73E-04	4.45E-04	6.78E-04	4.42E-04	1.86E-04	2.02E-03
7		2.77E-04	4.51E-04	6.87E-04	4.48E-04	1.89E-04	2.05E-03
8		1.57E-04	2.55E-04	3.89E-04	2.54E-04	1.07E-04	1.16E-03
9		2.23E-04	3.63E-04	5.53E-04	3.60E-04	1.52E-04	1.65E-03
10		3.41E-04	5.55E-04	8.45E-04	5.51E-04	2.32E-04	2.52E-03
11		1.38E-04	2.25E-04	3.43E-04	2.24E-04	9.45E-05	1.03E-03
12		5.95E-04	9.69E-04	1.48E-03	9.63E-04	4.06E-04	4.41E-03
13		7.73E-04	1.26E-03	1.92E-03	1.25E-03	5.28E-04	5.73E-03
14		3.15E-04	5.13E-04	7.81E-04	5.09E-04	2.15E-04	2.33E-03
Subtotal	HR	3.76E-03	6.12E-03	9.32E-03	6.08E-03	2.57E-03	2.78E-02
15		3.33E-04	5.43E-04	8.27E-04	5.39E-04	2.27E-04	2.47E-03
16		2.32E-03	3.77E-03	5.74E-03	3.74E-03	1.58E-03	1.72E-02
24		4.41E-04	7.18E-04	1.09E-03	7.13E-04	3.01E-04	3.27E-03
25		2.73E-03	4.45E-03	6.78E-03	4.42E-03	1.86E-03	2.02E-02
26		4.47E-04	7.28E-04	1.11E-03	7.23E-04	3.05E-04	3.31E-03
Subtotal	NY-NJH	6.27E-03	1.02E-02	1.55E-02	1.01E-02	4.28E-03	4.64E-02
17		8.93E-03	1.45E-02	2.22E-02	1.44E-02	6.10E-03	6.62E-02
18		1.33E-02	2.16E-02	3.29E-02	2.14E-02	9.05E-03	9.82E-02
19		4.47E-02	7.28E-02	1.11E-01	7.23E-02	3.05E-02	3.31E-01
20		1.75E-02	2.85E-02	4.34E-02	2.83E-02	1.19E-02	1.30E-01
21		5.37E-02	8.74E-02	1.33E-01	8.67E-02	3.66E-02	3.97E-01
22		1.03E-02	1.68E-02	2.56E-02	1.67E-02	7.05E-03	7.65E-02
23		3.34E-02	5.45E-02	8.30E-02	5.41E-02	2.28E-02	2.48E-01
Subtotal	NYB	1.82E-01	2.96E-01	4.51E-01	2.94E-01	1.24E-01	1.35E+00
27		4.84E-03	7.88E-03	1.20E-02	7.82E-03	3.30E-03	3.58E-02
28		1.37E-02	2.23E-02	3.40E-02	2.22E-02	9.36E-03	1.02E-01
29		1.71E-02	2.79E-02	4.25E-02	2.77E-02	1.17E-02	1.27E-01
30		1.60E-02	2.61E-02	3.98E-02	2.59E-02	1.10E-02	1.19E-01
Subtotal	LIS	5.17E-02	8.42E-02	1.28E-01	8.36E-02	3.53E-02	3.83E-01
Totals		2.44E-01	3.97E-01	6.04E-01	3.94E-01	1.66E-01	1.80E+00

Table A-17
Surface Sediment PCB Initial Conditions

Segment	PCB Initial Conditions (mg C/L)				
	Di	Tri	Tetra	Penta	Hexa
31	0.036	0.529	0.470	0.167	0.031
32	0.036	0.529	0.470	0.167	0.031
33	0.036	0.529	0.470	0.167	0.031
34	0.036	0.529	0.470	0.167	0.031
35	0.036	0.529	0.470	0.167	0.031
36	0.036	0.529	0.470	0.167	0.031
37	0.082	0.397	0.163	0.084	0.022
38	0.082	0.397	0.163	0.084	0.022
39	0.082	0.397	0.163	0.084	0.022
40	0.082	0.397	0.163	0.084	0.022
41	0.082	0.397	0.163	0.084	0.022
42	0.008	0.084	0.070	0.068	0.028
43	0.008	0.084	0.070	0.068	0.028
44	0.008	0.084	0.070	0.068	0.028
45	0.015	0.292	0.060	0.049	0.018
46	0.019	0.090	0.086	0.064	0.027
47	0.005	0.022	0.022	0.016	0.007
48	0.005	0.022	0.022	0.016	0.007
49	0.005	0.022	0.022	0.016	0.007
50	0.005	0.022	0.022	0.016	0.007
51	0.005	0.022	0.022	0.016	0.007
52	0.005	0.022	0.022	0.016	0.007
53	0.005	0.022	0.022	0.016	0.007
54	0.074	0.360	0.345	0.257	0.109
55	0.074	0.360	0.345	0.257	0.109
56	0.074	0.360	0.345	0.257	0.109
57	0.005	0.022	0.022	0.016	0.007
58	0.005	0.022	0.022	0.016	0.007
59	0.005	0.022	0.022	0.016	0.007
60	0.005	0.022	0.022	0.016	0.007

Table A-18
Weights and Growth Rates for Zooplankton, Small Fish, White Perch, and
Striped Bass

Organism	Year Class	Weight ^a (g wet wt)	Growth Rate (1/day)
Zooplankton	-	1.04 (mg dry wt) ^b	0.10
Small Fish	-	10.	0.00631
White Perch	0-1	16.9	0.0125
	1-2	56.3	0.0033
	2-3	89.6	0.00127
	3-4	110.0	0.00056
	4-5	127.5	0.00040
	5-6	141.5	0.00028
	6-7	158.5	0.00031
Striped Bass	0-1	2.0	0.006302
	1-2	20.08	0.005464
	2-3	148.4	0.003825
	3-4	601.8	0.002185
	4-5	1,339.4	0.001366
	5-6	2,208.4	0.001366
	6-7	3,641.	0.000905
	7-8	5,072.	0.000518
	8-9	6,131.	0.000456
	9-10	7,247.	0.000408
	10-11	8,417.	0.000369
	11-12	9,637.	0.000337
	12-13	10,904.	0.000310
	13-14	12,217.	0.000287
	14-15	13,572.	0.000267
	15-16	14,969.	0.000250
	16-17	16,406.	0.000235
	17-18	17,881.	0.000221

^a Weights given for the beginning of the year class.

^b Zooplankton weight is given in mg dry weight; all other weights given in g wet weight.

Table A-19
Bioenergetic Parameters for Zooplankton, Small Fish, White Perch, and Striped Bass

	Zooplankton	Small Fish	White Perch	Striped Bass
Food Assimilation Efficiency (a)	0.3	0.8	0.8	0.8
Fraction lipid (f _{lipid}) ^a	0.06	0.06	0.04 (0.06)	0.07 (0.06)
Fraction dry weight (f _{dry wt})	0.2	0.25	0.25	0.25
Respiration Rate (R) (g wet wt resp/g wet wt-day)	$R = 0.01249 e^{pT}$	$R = 0.0047 e^{pT}$	$= \beta w^v e^{pT} e^v$ $u = \omega w^\delta e^{\phi T}$	$R = \beta w^v e^{pT} e^{vu}$ $u = \omega w^\delta e^{\phi T}$
Respiration Parameters: β	-	-	0.043	0.043
γ	-	-	-0.3	-0.3
ρ	0.06293	0.06293	0.03	0.03
ν	-	-	0.0176	0.0176
ω	-	-	1.19	1.19
δ	-	-	0.32	0.32
φ	-	-	0.0405	0.0405
Respiration Rate (R _{O₂}) ^b (g O ₂ /g wet wt-day)	$R_{O_2} = a_{oxygen-carbon} a_{carbon-dry wt} f_{dry wt} R$			

^a Revisions in f_{lipid} values for the recalibrated PCB homologue model are given in parentheses.

^b For respiration rates in oxygen equivalents, a_{oxygen-carbon} and a_{carbon-dry wt} are taken as 2.67 and 0.4, respectively.

Table A-20

1987 Initial PCB Concentrations in Zooplankton, Small Fish, Perch, and Striped Bass (ug/g(wet wt))

Region 1

	Di	Tri	Tetra	Penta	Hexa
Zoo	0.2254	0.6289	0.7571	0.3831	0.05615
small Fish	0.152	0.9326	1.541	0.7321	0.06845
Perch 1	0.105	0.6907	1.369	0.8063	0.09227
Perch 2	0.07739	0.6399	1.671	1.334	0.2012
Perch 3	0.7164	0.6241	1.769	1.638	0.3139
Perch 4	0.6952	0.6174	1.811	1.832	0.4375
Perch 5	0.6886	0.6141	1.821	1.919	0.5473
Perch 6	0.06837	0.6117	1.828	1.969	0.6453
Perch 7	0.06825	0.61	1.825	1.979	0.7173
STB	0	0	0	0	0

Region 2

	Di	Tri	Tetra	Penta	Hexa
Zoo	0.1095	0.3774	0.4876	0.2747	0.0419
small Fish	0.07388	0.5596	0.9926	0.525	0.05108
Perch 1	0.05103	0.41144	0.8817	0.5782	0.06886
Perch 2	0.03761	0.384	1.076	0.956	0.15
Perch 3	0.03482	0.3745	1.139	1.172	0.2327
Perch 4	0.03347	0.3704	1.166	1.309	0.3227
Perch 5	0.03347	0.3685	1.172	1.369	0.4018
Perch 6	0.03323	0.367	1.177	1.404	0.4705
Perch 7	0.03317	0.365	1.175	1.41	0.5189
STB	0	0	0	0	0

Region 3

	Di	Tri	Tetra	Penta	Hexa
Zoo	0.04407	0.1465	0.06	0.028	0.005
small Fish	0.02972	0.2172	0.15	0.05	0.005
Perch 1	0.02053	0.1608	0.15	0.05	0.0055
Perch 2	0.01513	0.149	0.15	0.1	0.02
Perch 3	0.01401	0.1453	0.15	0.1	0.025
Perch 4	0.01359	0.1438	0.15	0.15	0.035
Perch 5	0.01346	0.143	0.15	0.15	0.05
Perch 6	0.01337	0.1425	0.15	0.15	0.05
Perch 7	0.01334	0.1421	0.15	0.15	0.05
STB	0	0	0	0	0

Region 4

	Di	Tri	Tetra	Penta	Hexa
Zoo	0.06548	0.0227	0.0357	0.019	0.0031
small Fish	0.04416	0.0337	0.0727	0.0363	0.0037
Perch 1	0.0305	0.0249	0.0646	0.04	0.00509
Perch 2	0.02248	0.0231	0.0788	0.066	0.011
Perch 3	0.02081	0.0225	0.0853	0.0806	0.0164
Perch 4	0.0202	0.0223	0.0857	0.0896	0.0231
Perch 5	0.02	0.0222	0.086	0.0933	0.0282
Perch 6	0.01986	0.0221	0.0859	0.0955	0.0326
Perch 7	0.01983	0.022	0.0859	0.0957	0.0355
STB	0	0	0	0	0

Initial Concentration in Region 5 set equal to zero for all species.

Appendix B: Notation

Appendix B. Notation

The following symbols are used in this report:

a	=	food assimilation efficiency (dimensionless);
a_{DOC}	=	factor relating K_{DOC} to K_{ow} (dimensionless);
A_s	=	surface area (m^2);
BCF	=	bioconcentration factor (L/kg);
BSAF	=	biota-sediment-accumulation factor (kg(oc)/kg(lipid));
C	=	total (freely-dissolved plus DOC-bound plus particulate) chemical concentration ($\mu\text{g/L}$);
C_{dis}	=	freely-dissolved chemical concentration ($\mu\text{g/L}$);
$C_{\text{dis+DOC}}$	=	total dissolved (freely-dissolved plus DOC-bound) chemical concentration ($\mu\text{g/L}$);
C_{DOC}	=	DOC-bound chemical concentration ($\mu\text{g/L}$);
C_{O_2}	=	dissolved oxygen concentration in water (mg/L);
C_{part}	=	particulate chemical concentration ($\mu\text{g/L}$);
D_w	=	molecular diffusivity of PCBs in water (cm^2/sec);
E_{ij}	=	bulk dispersion coefficient for mixing between segments 'i' and 'j' (m^3/sec);
f_{lipid}	=	fraction of organism mass that is comprised of lipids ($\text{g lipid/g wet weight}$);
f_{oc}	=	fraction organic carbon for suspended solids and sediments ($\text{g organic carbon/g dry weight}$);
h	=	mean water depth (m);
K_d	=	equilibrium partition coefficient between particulate and freely-dissolved phases (L/kg);
K_{DOC}	=	equilibrium partition coefficient between DOC and freely-dissolved phases (L/kg);
K_H	=	Henry's constant ($\text{atm}\cdot\text{m}^3/\text{mol}$);
K_{ow}	=	octanol-water partition coefficient (L/kg);
k	=	chemical transformation rate coefficient ($1/\text{day}$);
k_b	=	back-diffusion rate coefficient across organism membrane or gill ($1/\text{day}$);
k_e	=	chemical excretion rate coefficient ($1/\text{day}$);
k_r'	=	porewater-water column exchange rate coefficient (m/day);
k_g	=	growth dilution rate coefficient ($1/\text{day}$);
k_m	=	chemical metabolic rate coefficient ($1/\text{day}$);
k_u	=	chemical uptake rate coefficient across organism membrane or gill (L/g(wet)/day);
k_v'	=	volatilization rate coefficient (m/day);
I_{kl}	=	consumption rate of organism 'k' feeding on organism 'l' ($\text{g(wet) prey/g(wet) predator/day}$);
MW	=	molecular weight (g/mol);
m	=	solids concentration (mg/L);

P_c	=	partial pressure of the chemical in the overlying air (atm);
Q_{ij}	=	flow rate from segment 'i' to segment 'j' (m^3/sec);
R	=	respiration rate in wet weight equivalents ($g(wet\ wt\ respired)/g(wet\ wt)/day$);
R_{O_2}	=	respiration rate in oxygen equivalents ($g(O_2\ respired)/g(wet\ wt)/day$);
t	=	time (days);
U	=	average river velocity (m/sec);
V	=	volume (m^3);
W_c	=	chemical input rate from external sources, e.g., tributaries, effluent discharges (kg/day);
W_m	=	solids input rate from external sources, e.g., tributaries (kg/day);
w_b	=	burial rate for solids (cm/yr);
w_s	=	solids settling velocity (m/day);
w_u	=	solids resuspension velocity (m/day);
$x_{5,95\%ile}$	=	the 5 and 95%ile concentrations;
Z	=	standard normal deviate (equal to -1.64 and +1.64 in computing the 5 and 95%iles);
α_{kl}	=	chemical assimilation efficiency for organism 'k' feeding on organism 'l' (dimensionless);
β	=	gill transfer efficiency (dimensionless);
δ_x	=	coefficient of variation (given as σ_x/μ_x);
Γ	=	solid phase chemical concentration ($\mu g/g(dry)$);
Γ_{oc}	=	organic carbon normalized solid phase chemical concentration ($\mu g/g(oc)$);
μ_x	=	arithmetic mean concentration;
$\mu_{ln\ x}$	=	log transformed mean concentration;
v	=	biota chemical concentration ($ug/g(wet)$);
$\bar{v}$	=	lipid-normalized biota chemical concentration ($ug/g(lipid)$);
ϕ	=	porosity (dimensionless);
σ_x	=	standard deviation;
$\sigma_{ln\ x}^2$	=	variance of the log transformed concentrations.

The following subscripts are used in this chapter:

i,j	=	segment descriptors for spatial segments;
k,l	=	descriptors for trophic levels in the food chain;
sed	=	surface (active) sediment layer;
deep sed	=	deep sediment layer underlying the surface sediment;
phyto	=	phytoplankton.