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

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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 resusupension. 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 µg/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.

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Acknowledgments

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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 µg/g wet weight in the tidal freshwater region to a low of 3 µg/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.

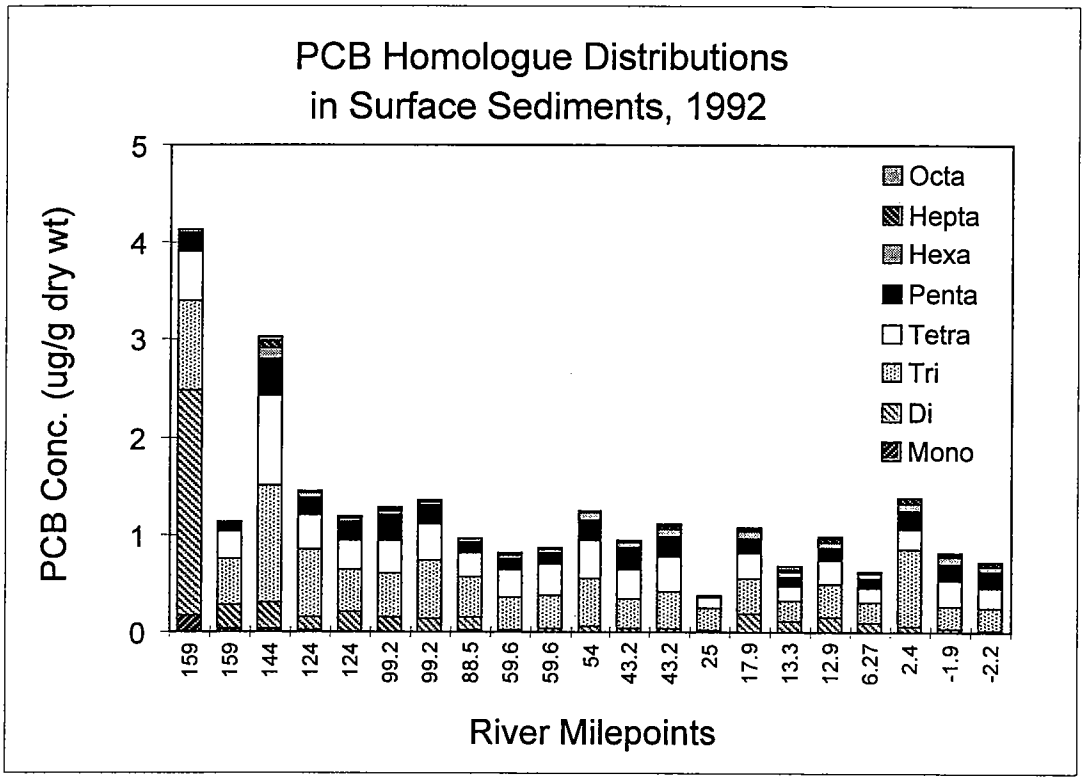


Figure 1-1. PCB concentrations and homologue distributions for surface sediments in the Hudson River Estuary (8/92 - 11/92). [Data from TAMS/Gradient (1995).]

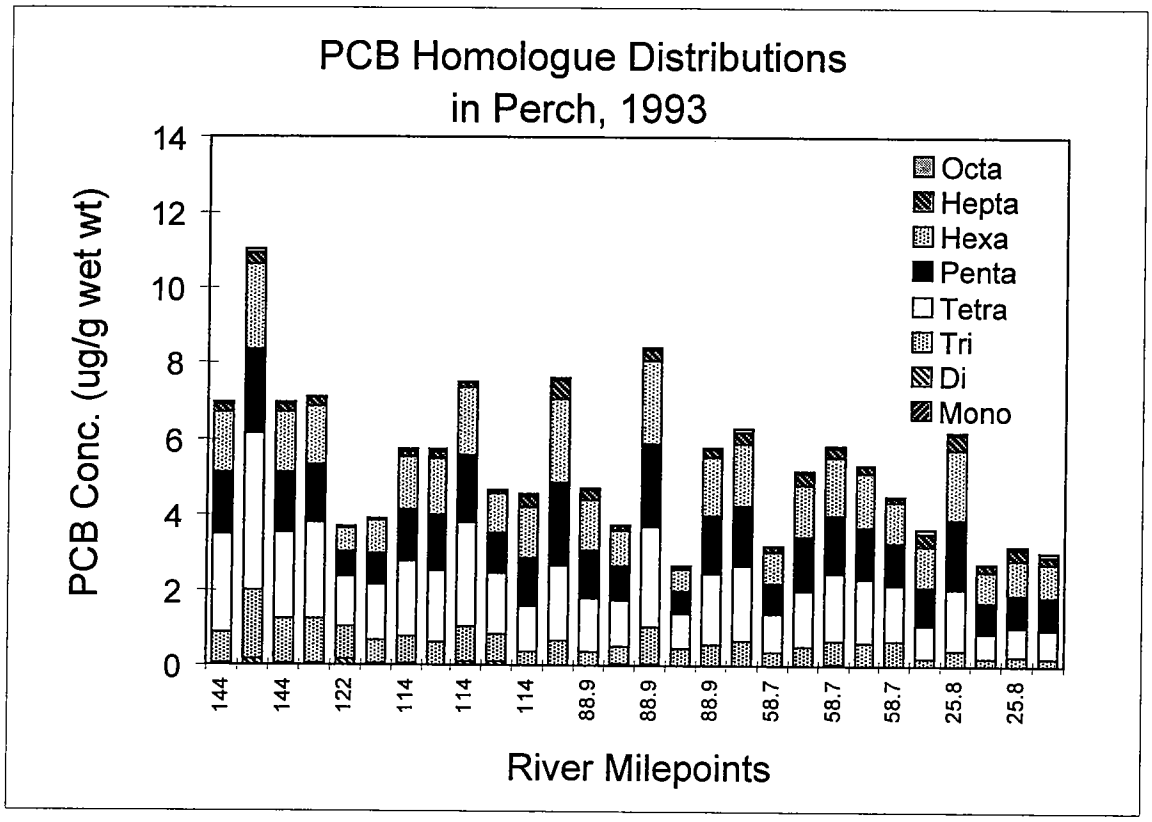


Figure 1-2. PCB concentrations and homologue distributions for white and yellow perch in the Hudson River Estuary (8/93). [Data from TAMS/Gradient (1995).]

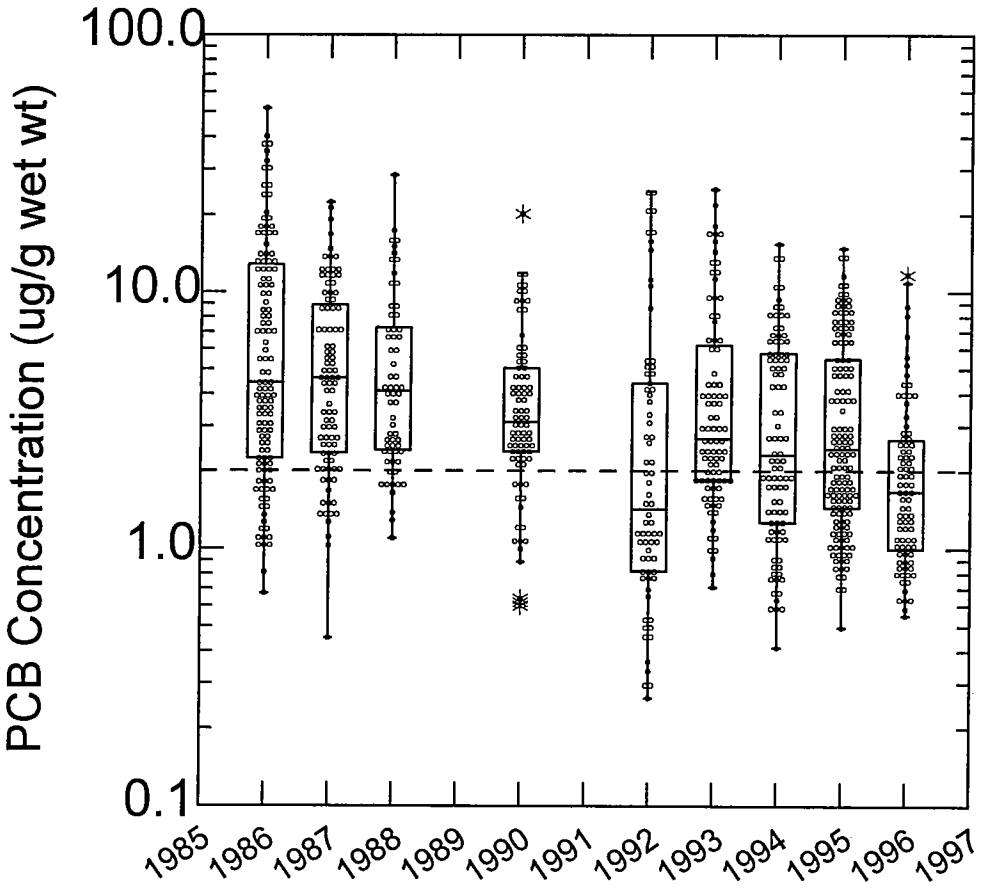


Figure 1-3. 1986-1997 time history of total PCBs in striped bass from the tidal freshwater Hudson. [Data from NYS DEC as reported in TAMS/Gradient (1995).]

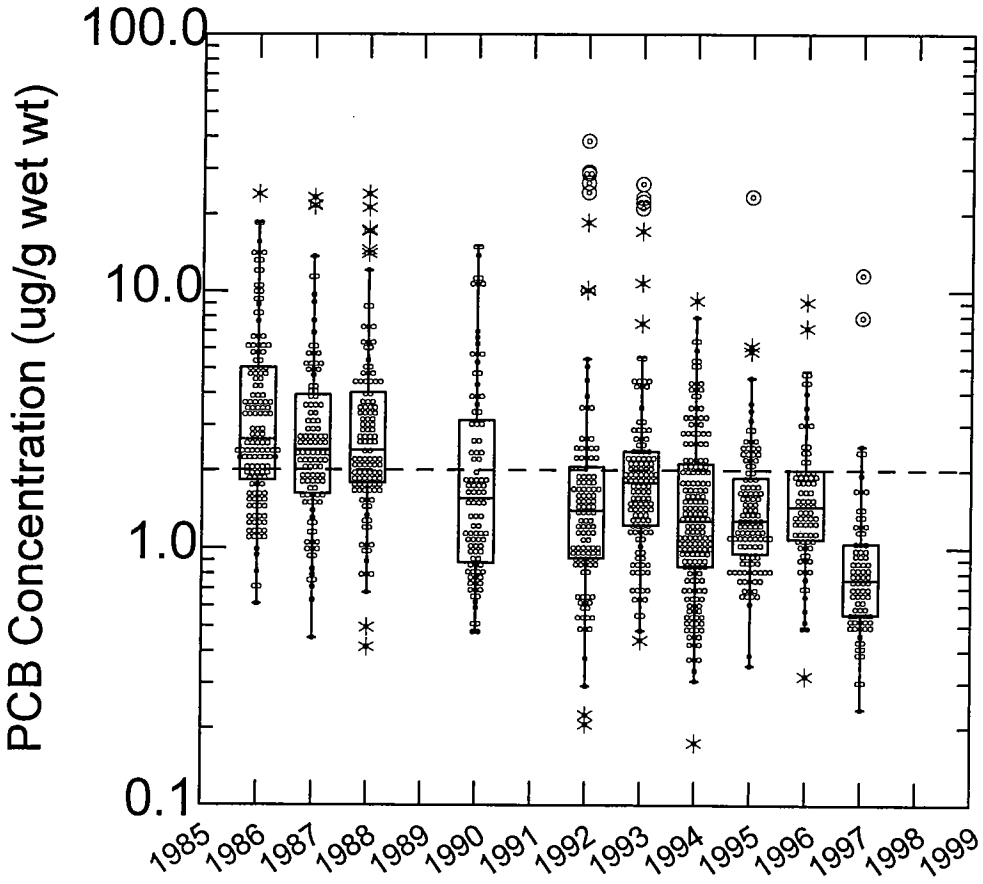


Figure 1-4. 1986-1997 time history of total PCBs in striped bass from the Hudson mid estuary. [Data from NYS DEC as reported in TAMS/Gradient (1995).]

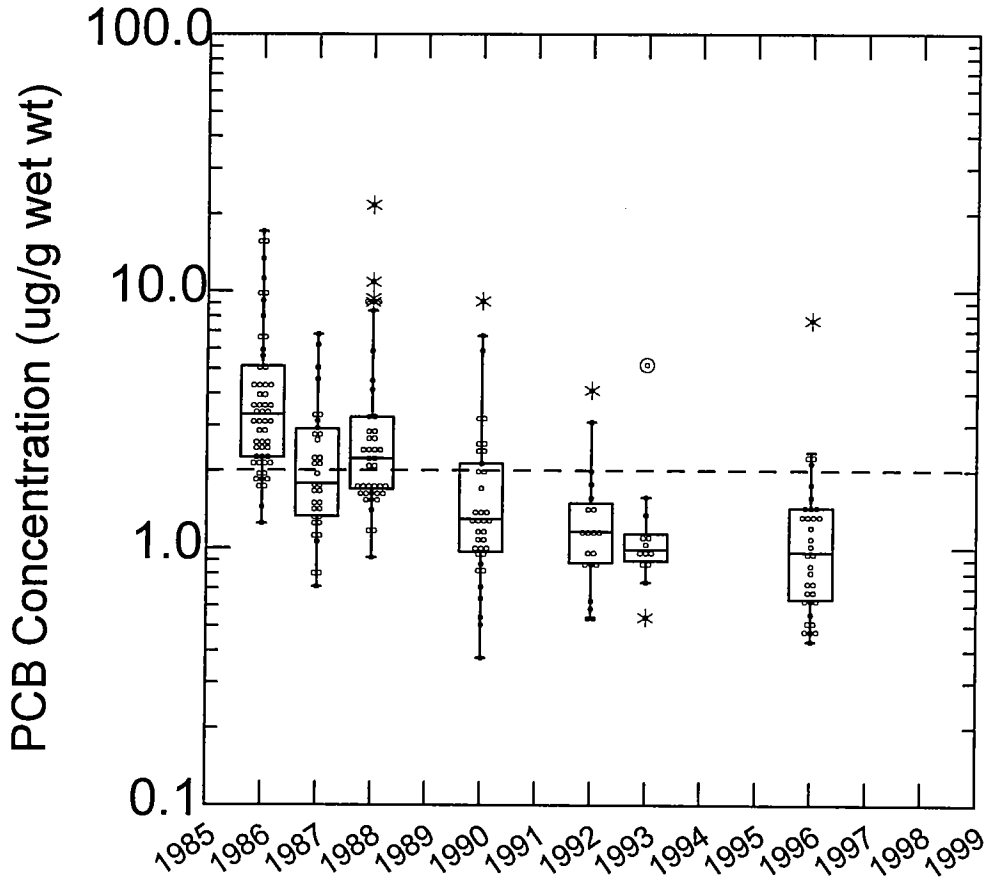


Figure 1-5. 1986-1997 time history of total PCBs in striped bass from New York-New Jersey Harbor. [Data from NYS DEC as reported in TAMS/Gradient (1995).]

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

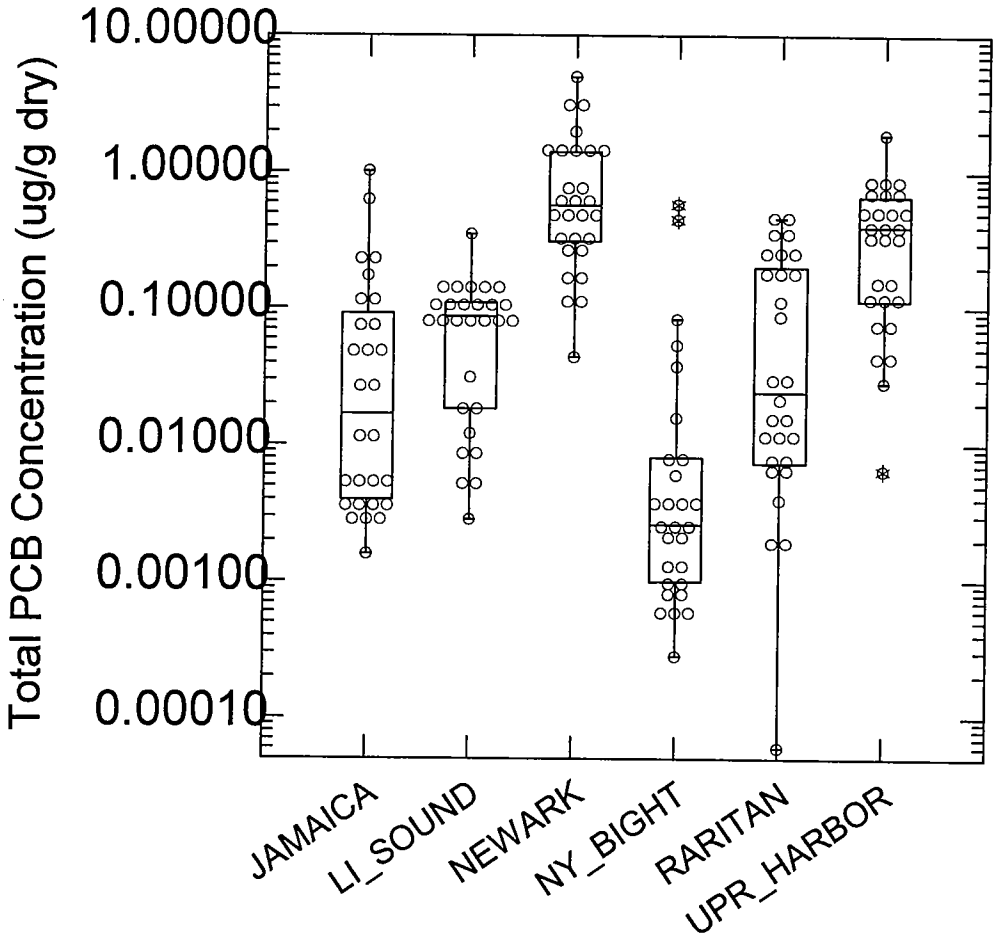


Figure 1-6. 1993-94 total PCB concentrations in surface sediments of New York-New Jersey Harbor. [Data from 1993-94 R-EMAP study (Adams et al. 1998).]

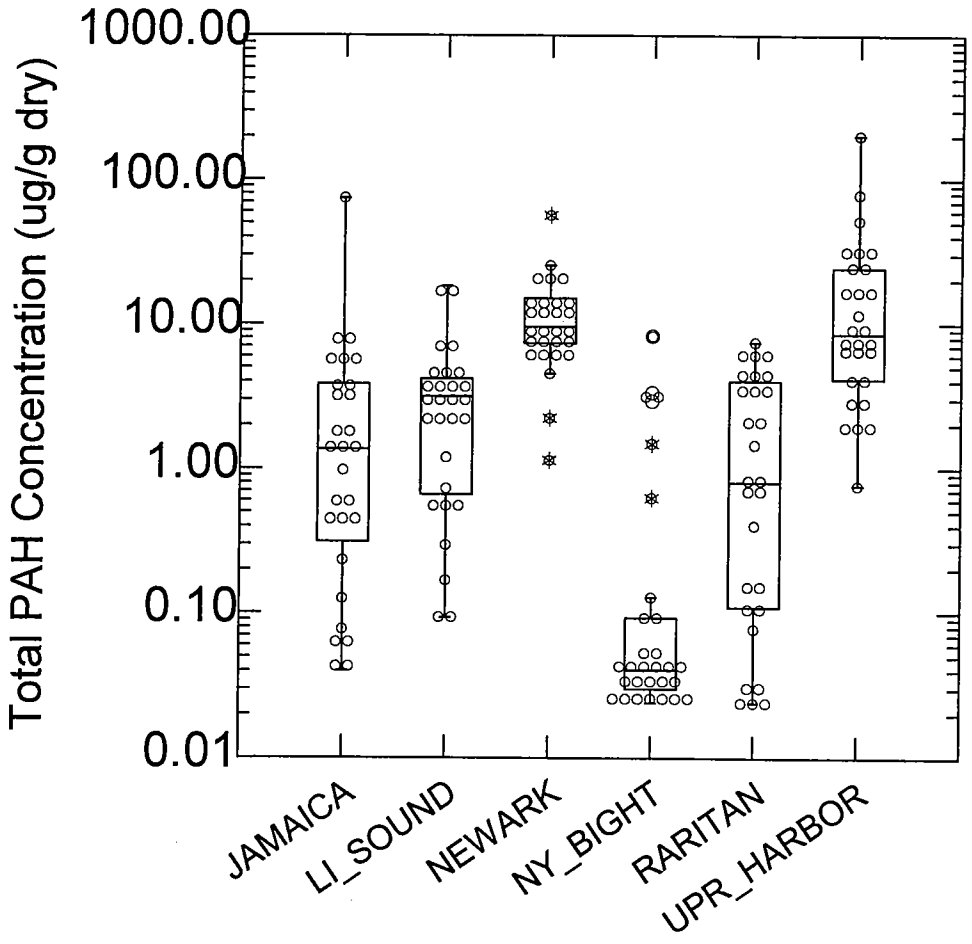
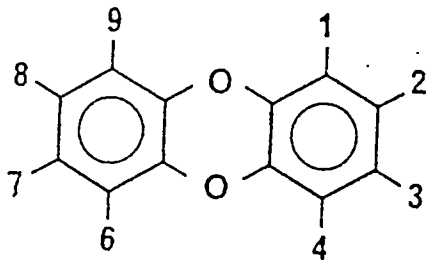


Figure 1-7. 1993-94 total PAH concentrations in surface sediments of New York-New Jersey Harbor. [Data from the 1993-94 R-EMAP study (Adams et al. 1998).]

concentrations for various regions around the harbor show large spatial variations with medians of 0.04 µg/g(dry wt) in the New York Bight and 10 µg/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-PeCCD	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

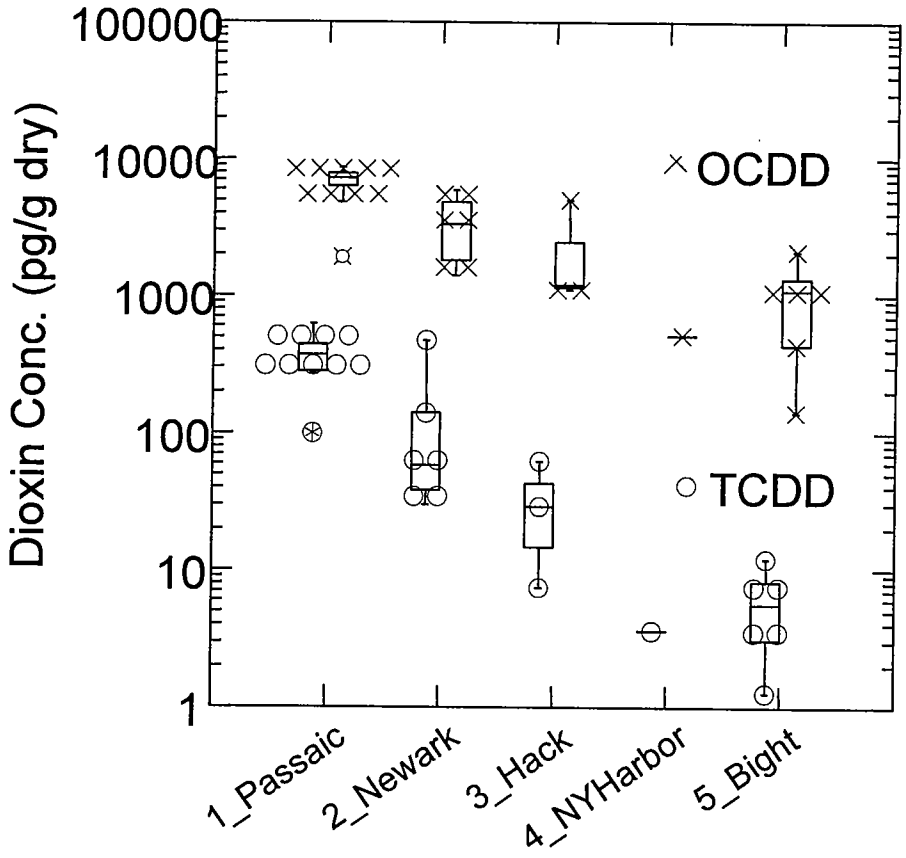


Figure 1-8. 2,3,7,8-TCDD and OCDD concentrations in sediments in New York-New Jersey Harbor. [Data from Long et al. (1995).]

(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

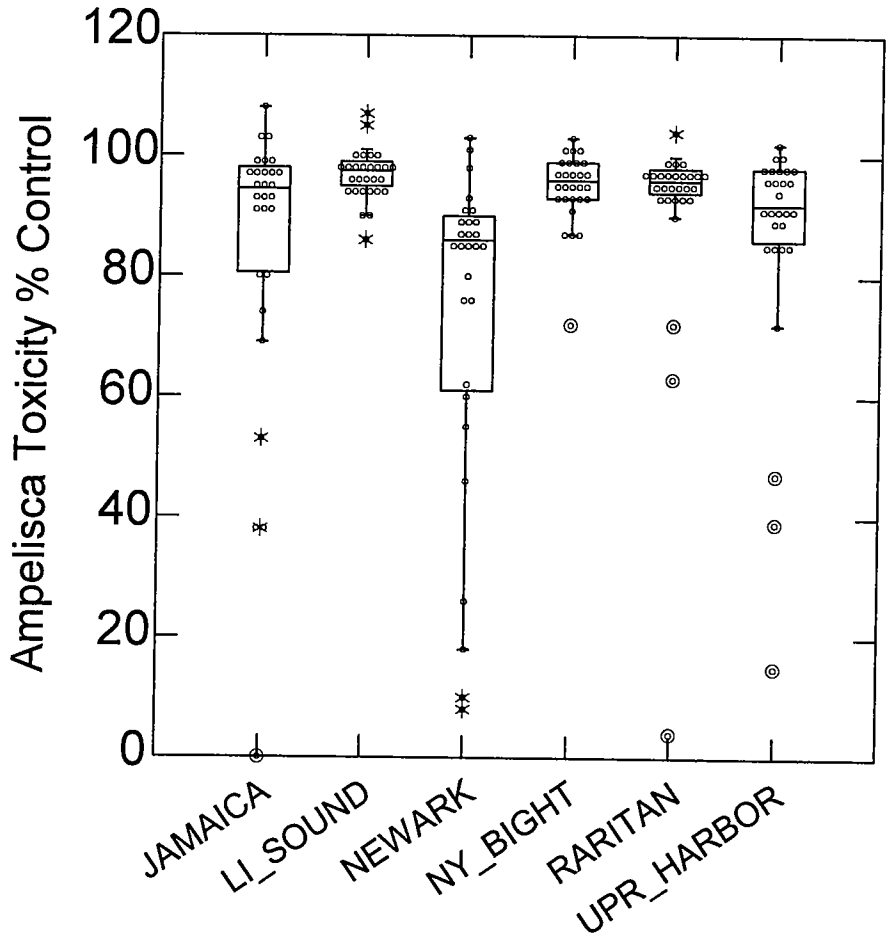


Figure 1-9. 1993-94 Ampelisca abita toxicity testing results for surface sediments in New York-New Jersey Harbor. [Data from 1993-94 R-EMAP study (Adams et al. 1998).]

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.

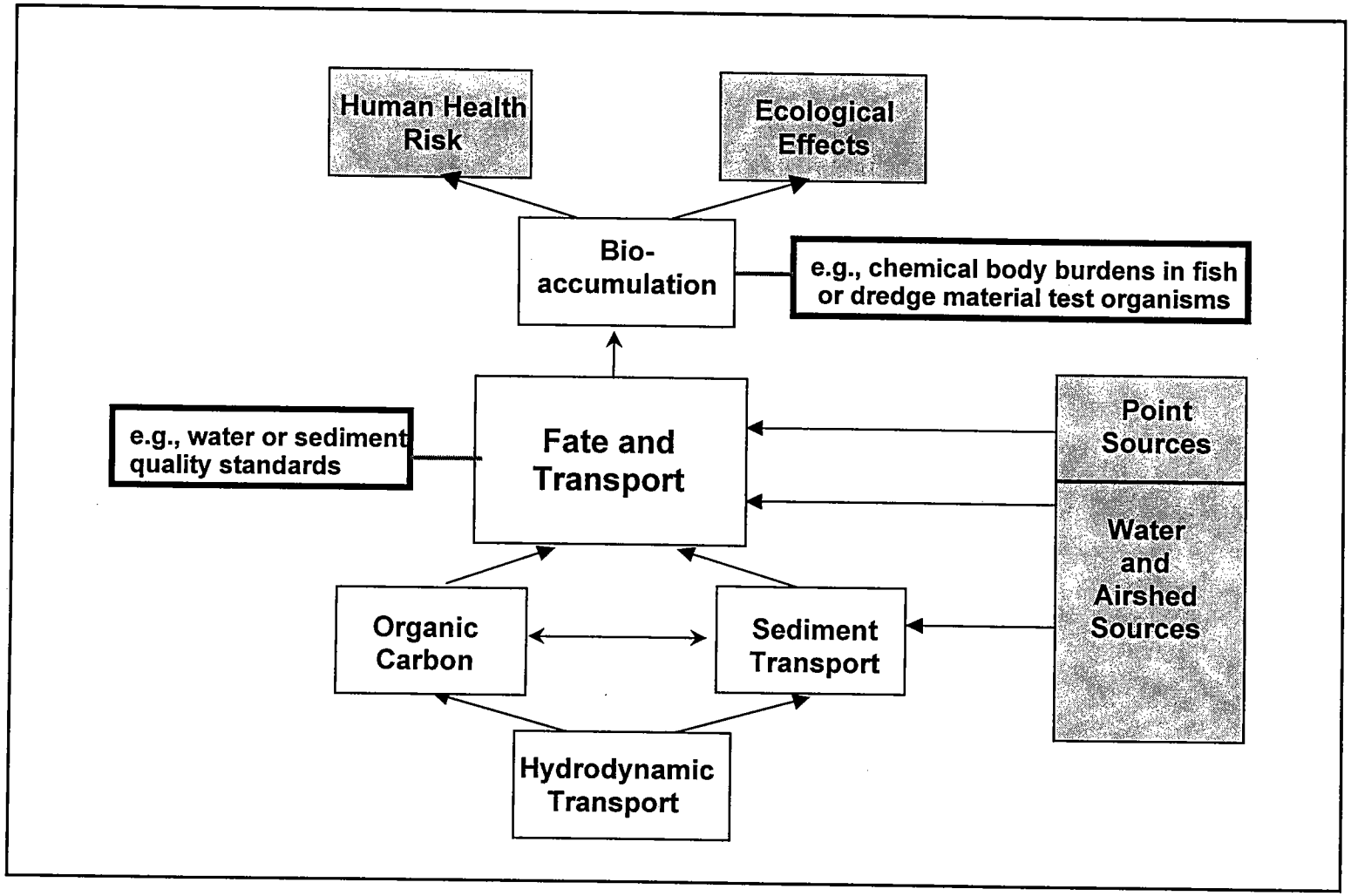


Figure 1-10. Components (or submodels) of the overall modeling approach.

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