Lower Passaic River Restoration Project

Final

Modeling Work Plan

In partnership with

September 2006

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Passaic River
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Project

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SECTION 1

INTRODUCTION

1.1 OVERVIEW OF ISSUES ON THE PASSAIC RIVER

The Lower Passaic River is the 17-mile tidal stretch of the Passaic River from the Dundee Dam to the river mouth at Newark Bay. During the 19th and 20th centuries, the Lower Passaic River became a focal point for the nation's industrial revolution. The urban and industrial development surrounding the river, combined with associated population growth, have resulted in poor water quality, contaminated sediments, bans on fish and shellfish consumption, lost wetlands and degraded habitat.

Numerous studies conducted by federal and state agencies have established that contaminated sediments and other hazardous chemical sources exist along the 17-mile tidal stretch of the Passaic River. Contaminants of concern include dioxin/furans, polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), pesticides and herbicide residues, and metals.

To restore the Lower Passaic River, a federal and state agency partnership has been formed, that includes the U.S. Environmental Protection Agency (EPA), U.S. Army Corps of Engineers (USACE), New Jersey Department of Transportation (NJDOT), National Oceanic and Atmospheric Administration (NOAA), U.S. Fish and Wildlife Service (USFWS) and New Jersey Department of Environmental Protection (NJDEP). The partner agencies are putting together a comprehensive plan that will improve water and sediment quality in the River, as well as restore degraded habitats along the River. The Lower Passaic River Restoration Project is being implemented jointly under the Comprehensive Environmental Response Compensation and Liability Act (CERCLA) and Water Resources Development Act (WRDA). The history of federal and state agency involvement that led to the implementation of this joint project is detailed in the project work plan (Malcolm Pirnie, 2005c).

The Lower Passaic River Restoration Project represents the umbrella under which the integrated effort of the partner agencies is taking place. HydroQual's part of the Project is the development and application of a suite of mathematical models (i.e., hydrodynamic, sediment transport, chemical fate and transport, bioaccumulation) of the Lower Passaic River to determine the relative significance of contaminant sources to water, sediments, and biota and to evaluate the effects of various remedial strategies on reducing environmental exposure end points, which in turn affect human health and ecological risks. The model suite being developed as part of the Project is, of course, only one element in a line of evidence that may be used to guide management and remediation decisions meant to restore the ecological health and function of the lower Passaic River. This report is a detailed modeling plan that HydroQual proposes to implement as part of the Lower Passaic River Restoration Project.

It is important to recognize, however, that the modeling work plan, presented herein, is a starting point for selecting, developing, and calibrating the required models and is not a final modeling report. As such, a final determination of the model grid, model assumptions, parameters, calibration data sets, etc. has not yet been performed, only the conceptual framework has been developed and is presented in this report. It is also important to recognize that an exhaustive data analysis has not as yet been performed. Readily available data have been obtained and undergone a preliminary analysis, in part to better understand the issues and water quality problems within the Lower Passaic River and in part to better identify areas where additional data are required. Efforts will be expended during the project to identify, obtain and utilize additional historical data sets as well as to utilize data sets being collected as part of the Lower Passaic River Restoration Project. As further data are collected and analyzed and as our understanding of the Lower Passaic River and its interactions with adjacent waterbodies improves, it may be necessary to modify elements of the work plan in order to develop the most technologically sound and defensible model of the lower Passaic River system.

1.2 PURPOSE AND OBJECTIVES OF THE LOWER PASSAIC MODELING STUDY

The modeling plan although not directly addressing WRDA related issues will be nonetheless a useful tool for answering specific questions that WRDA projects might pose. WRDA projects on the Passaic River are still in the making, including the final choice of the restoration sites, defining the criteria that will be used in designing these projects, and the type of restoration that will be implemented. Also, it is likely that these projects will have their own environmental investigation activities. However, it is anticipated that the main issues for the Passaic River will include the restoration of water quality, sediments and watershed drainage areas, and possibly nearby wetlands in the upper Newark Bay; the protection of river biota from contact with concentrations of multiple chemicals in the river sediments to help restore aquatic habitat; and the reduction and control of pollutants now entering the river from storm water runoff, outfalls, and atmospheric deposition to assist with restoration and to maintain the restored habitat. The modeling framework can help address facets of these issues, such as:

- The fate and transport of chemicals in the restoration site; time to recover, etc.
- The impact of a capping vs. dredging scenario.
- Navigational issues related to depth of channels with and without dredging
- Impact flooding (e.g., severe events)

Although the model could still answer some aspects related to raising the submerged, un-vegetated mudflats in the Passaic to create vegetated shallows (similar to pre-bulkhead conditions), the incorporation of restored vegetated shallows into riverfront developments for recreational, municipal and commercial uses, or the enhancement of degraded wetlands in the adjacent river systems to nurture expanded bird and fish populations, it is not explicitly designed to directly address those aspects.

There will certainly be other issues that will have to be addressed, once the WRDA project for the Passaic River is fully designed. Whether the modeling framework will be able to address all the issues will ultimately depend on the nature of the questions. Section 1.2 of the new Conceptual Site Model (CSM) explains how the CSM serves a role in WRDA. As such, the CSM process includes consideration of WRDA related components such as dredging, mudflats and habitats.

A number of "fundamental questions" were recently formulated (Malcolm Pirnie, 2005b) as part of the preparation of the Data Quality Objectives (DQOs) for the study the Lower Passaic River Restoration Project. Answers to those questions are meant to satisfy the CERCLA and WRDA requirements as well as the needs of a Natural Resource Damage Assessment (NRDA) under CERCLA. The "fundamental questions" are listed below.

1. If we take no action on the River, when will the COPCs and chemicals of potential ecological concern (COPECs) recover to acceptable concentrations?

2. What actions can we take on the River to significantly shorten the time required to achieve acceptable or interim risk-based concentrations for human and ecological receptors?

3. Are there contaminated sediments now buried that are likely to become exposed following a major flood, possibly resulting in an increase in contaminants within the fish/crab populations?

4. What actions can we take on the River to significantly improve the functionality of the Lower Passaic River watershed?

5. If the risk assessments for Newark Bay demonstrate unacceptable risks due to contaminant export from the Passaic River, will the plan proposed to achieve acceptable risks for Passaic River receptors significantly shorten the time required to achieve acceptable or interim risk-based concentrations for receptors in Newark Bay, or will additional actions be required on the Passaic River?[1]

6. What actions can we take on the River to significantly reduce the cost of dredged material management for the navigational dredging program?

[1] This question is shared with the RI/FS for the Newark Bay Study, since the actual benefits of such reduction will need to be jointly determined. A similar question to address the adequacy of any future Newark Bay plan toward achieving Passaic River goals may be included in the Newark Bay Study.

7. What actions can we take to restore injured resources and compensate the public for their lost use?

The main purpose of the modeling effort is, together with data analysis of Lower Passaic River sediment cores, to help answer those questions by developing and applying a hydrodynamic, sediment transport, a chemical fate and transport as well as a biological model to facilitate evaluation of sediment and water column contaminant fate and transport in the Lower Passaic River. The model will predict future concentrations of various COPCs in the study area under different management scenarios (e.g., dredging, monitored natural attenuation, capping, etc.). Specifically, the model will be used to:

- Establish the magnitudes and relative importance of specific contaminant sources to the 17-mile tidal reach of the Passaic River, including:
  - Upstream loads over the Dundee Dam,
  - Loads from tributaries and other point sources along the 17-mile tidal reach,
  - Re-mobilization of contaminants within the 17-mile tidal reach, and
  - Inputs from waterbodies tidally connected to the 17-mile tidal reach (including, for example, the effect of contaminant loadings from Newark Bay and its tributaries),
- Provide a tool to evaluate options to manage adverse ecological and human health risks caused by the transport and fate of the chemicals of concern within the system.
- Assess the impacts of sediment and chemical contaminant re-mobilization due to various remedial action alternatives that may be conducted within the 17-mile tidal reach of the Passaic River during the period of remediation, as well as during the recovery period.
- Assess sediment quality and contaminant levels if loadings are reduced or eliminated and the time frame for improvement under various remedial action alternatives.

The modeling portion of the Lower Passaic Restoration Study is designed not only to model the physical, chemical and biological processes occurring within the Passaic River, but it includes Hackensack River, Newark Bay and the adjacent tributaries. It will also determine the interaction that the system has with the surrounding waters of the Kill van Kull, the Arthur Kill, the Hudson River and the greater New York and New Jersey Harbor system.

The modeling plan presented in subsequent sections describes i) the basis for selecting the models to be used to meet the above goals and objectives, ii) the modeling framework *per se*, iii) the data needs, and iv) the road map towards calibrating and validating the models.

1.3 SITE PHYSICAL SETTING AND BACKGROUND

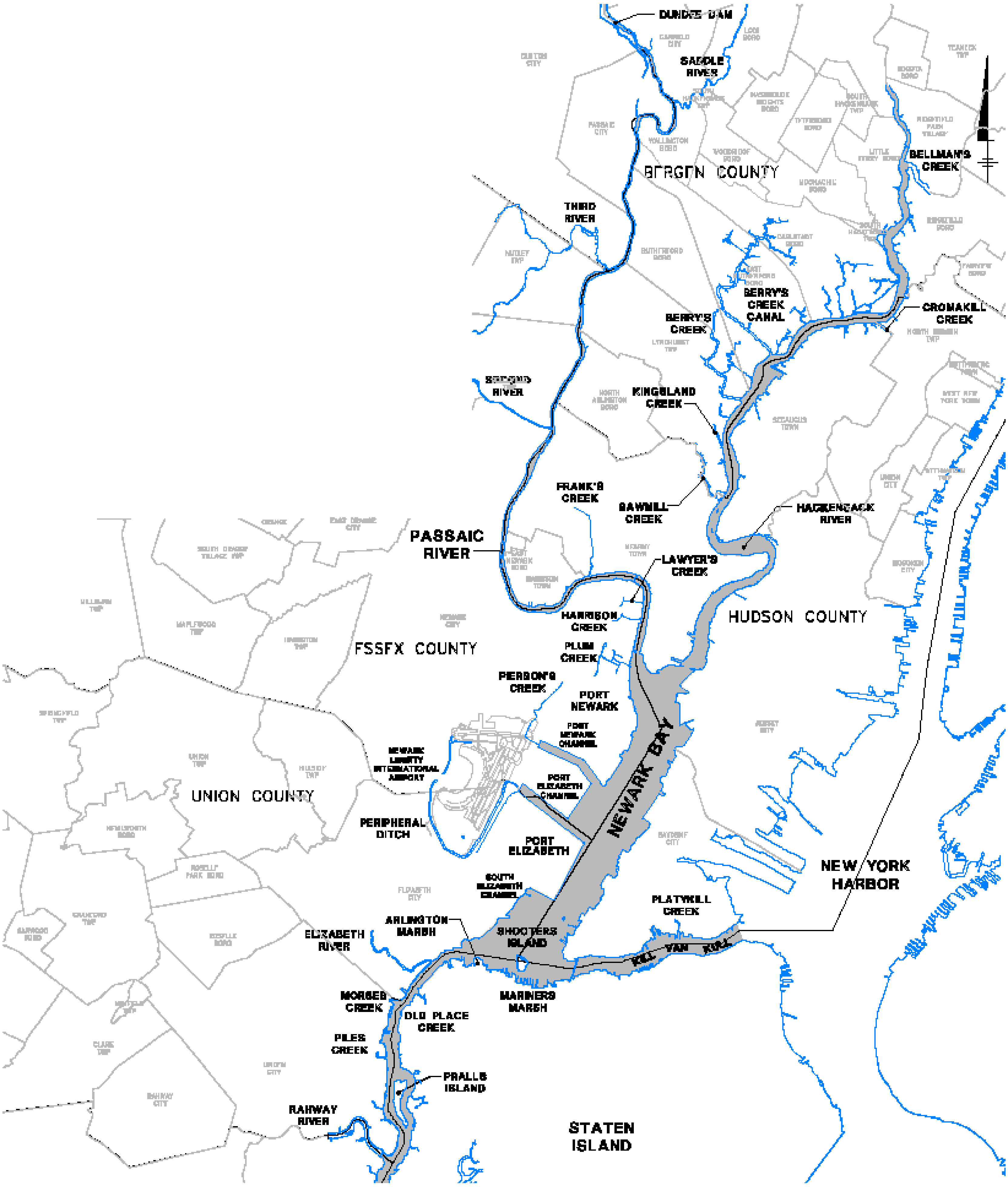
The Lower Passaic River Restoration Project Study Area encompasses the 17-mile tidal stretch of the Passaic River below the Dundee Dam, its tributaries and the surrounding watershed that hydrologically drains below the Dundee Dam. Because the Lower Passaic River is tidally connected to Newark Bay and the New York-New Jersey Harbor Estuary, the modeling domain will include the Lower Passaic River, Hackensack River, Newark Bay, Kill Van Kull and Arthur Kill (see Figure 1-1). Most of the freshwater originates from the upper portion of the Passaic River across the Dundee Dam. There are, however, three major tributaries to the Passaic River that bring additional fresh water river downstream of the Dundee Dam (Table 1-1).

Table 1-1. Mean and peak flows of the Passaic River and its three main tributaries (USGS Record)

	Average (cfs)	Peak Flow (cfs)
Passaic River	1,140 (110 years)	31,700 (10/10/1903)
Saddle River	100 (80 years)	5,330 (9/17/1999)
Third River	21 (20 years)	2,670 (9/16/1999)
Second River	18 (40 years)	6,500 (8/28/1971)

Four other tributaries, McDonald Brook, Frank Creek, Lawyer's Creek, and Plum Creek, have been identified historically as contributing freshwater inflow to the Lower Passaic River. However, these tributaries are now urbanized tributaries, having been bulk-headed, and receive freshwater inflows via discharges from combined sewer outfalls (CSOs) and storm water outfalls (SWOs). As such, estimates of freshwater inflow from the latter four tributaries will be accounted for via the use of an urban runoff model.

Figure 1-1. Passaic River, Hackensack River, Newark Bay, Kill Van Kull and Arthur Kill Study Area (Map adapted from TSI, 2004



The combined flow of the three major tributaries (Saddle River, Third River, and Second River) is estimated to represent less than 10% of the total flow at the mouth of the estuarine section of the river, which is influenced by semidiurnal tides reaching a mean tidal range of about 5 ft, 1.5 miles from Newark Bay (NOAA, 1972). As a result, density stratification is prevalent in the Lower Passaic River causing a distinct reversal of currents between top and bottom layers of the water column.

CSOs as well as SWOs also contribute to the inflow of freshwater in the Passaic. There are 109 inventoried CSOs, and an even larger number of SWOs in the Passaic River, Newark Bay, the Kills and lower section of the Hackensack River, as well as six (6) wastewater treatment plant (WWTP) outfalls distributed in Newark Bay, the Kills and the Hackensack River (TSI, 2004). It is noteworthy that no WWTPs are located on the Passaic River. As will be discussed in the modeling sections, these CSO, SWO and WWTP sources will need to be identified and assessed relative to their contribution to the load of contaminants entering the system.

It is also important to note that the lower section of the Hackensack River consists of vast area of tidal wetlands, the Meadowlands area. U.S. EPA's National Wetland Inventory identifies about 1,500 acres of the wetland area that are submerged with average tidal condition and that can be flooded during extreme flood conditions. Water storage that will occur in the marsh land during tidal cycling and after storm events is expected to have an effect on hydrodynamic transport through much of the Hackensack River and ultimately to the Passaic River study area. These processes of wetting and drying need to be explicitly considered in hydrodynamic model calculations as discussed in Section 2.

1.4 CHEMICALS OF POTENTIAL CONCERN (COPCs)

Federal and state agency studies show that Lower Passaic River sediments are contaminated with a number of hazardous substances. Although the list of contaminants to be modeled is not finalized yet, the list will likely include:

- Dioxin/Furan congeners
- PCB homologs and selected PCB congeners
- Selected PAH compounds
- Pesticides, such as DDT and chlordane
- Metals, including cadmium, zinc, nickel, copper, lead and mercury

The choice of the contaminants of concern for modeling purposes will depend on the needs of the human health and ecological risk assessments. Tables justifying data needs and use, as well as proposed analyses have been developed by HydroQual and the project team and are presented in the Quality Assurance Project Plan (QAPP) (Malcolm Pirnier, 2005b) and Field Sampling Plan (FSP) (Malcolm Pirnie, 2005a). The ongoing and planned data analyses will continue to update the CSM and this, along with the pathways analyses, will form the basis for the choice of COPCs that will be modeled.

Over the years, the Study Area has been the subject of a number of field sampling programs. There are a few hundred thousand historical data points generated by federal, state and private organizations, focusing mostly on sediment sampling (Malcolm Pirnie 2004). Most of that data were collected between 1991 and 1995 for EPA's Remedial Investigation (RI) study of the lower 6-mile stretch of the Passaic River.

An intrinsic part of data evaluation is to understand the physical characteristics of the sediments, the spatial and temporal distribution of the chemicals in the sediments, the spatial and temporal patterns in water column concentrations and in biota, as well as the hydrodynamic behavior likely to affect the stability of the sediments in terms of deposition and re-suspension. The bulk of the data analysis centered on establishing horizontal and vertical distribution of the contaminants in the surface and bottom sediments, as well as in the water column and biota. The analysis aimed at i) constructing a conceptual site model that guides the design of the hydrodynamic, sediment transport, fate and transport and bioaccumulation modeling framework, and ii) helping design a sampling program that supports data modeling needs as well as geochemical, risk assessment and engineering analyses.

A framework for conducting in-depth data analyses, both historical and planned is outlined in the CSM, which is Attachment A of the project work plan (Malcolm Pirnie, 2005c). The CSM lists data evaluations that have been completed to date and details the processes that will be used to incorporate future evaluations into the CSM. Such analyses will be an intrinsic part of the modeling effort and will include physical as well as geochemical data, including historical and 2005 radionuclide data from dated-sediment cores.

1.4.1 Data Evaluation: Sediments

For the initial assessment of sediment contamination, HydroQual has focused its historical data evaluation of the Passaic River on analyzing the EPA RI 1995 data, although other less populated databases are also available; these databases are listed and available on the www.ourPassaic.org project website (WP MPI, 2005). The RI consisted of 26 transects on a six-mile stretch on the Lower Passaic River, and each transect consisted of 3 stations: the left channel bed, the thalwag or deepest channel of the river, and the right channel bed. The average distance between transects was 375 meters. The transect interval was modified to avoid bridges and sewer outfalls; actual distances range from 250 to 2000 meters. The distance between samples laterally is 50 meters, on average, and varies from 25 to 150 meters. The EPA 1995 RI sampling plan established the depth for individual cores as the depth to the 1940 time period sediment. It should be noted, however, that the 1995 data set is already 10 years old; given the estimated rates of sediment deposition varying from up to five inches per year in the middle of the channel to no deposition or scour in the shoals (Malcolm Pirnie 2005c), it is possible that the 1995 surface layer is either buried 50 inches below the present-day surface or scoured away to be deposited elsewhere in the river or bay. The sediment samples were also analyzed for Dioxin/Furans (9 Species), PCBs (groups of 11 and 22 congeners), PAHs (higher and lower Molecular weight PAHs as well as 23 individual species), pesticides (DDT, DDE, chlordane), and metals (As, Ag, Cd, Cr, Cu, Hg, Ni, Pb, Zn). Since there are 26 transects of triple samples at the same river mile, the data were grouped into river miles, and averaged over transects.

Figure 1-2 illustrates the distribution of sampling locations laterally and horizontally in the Lower six miles of the Passaic. To better view and analyze this data, map-based spatial representations of some of the chemical concentrations were developed, in addition to two-dimensional plotting templates for rendering the data in the along-channel direction from the mouth of the Passaic River and along the six-mile stretch of the Lower Passaic River where most of the data has been collected. The chemicals of concern that have been presented are metals, PCB's, PAH's, Dioxins, DDT and metals in their dry weight form as well as organic carbon normalized form for the organic chemicals. Fish tissue samples that are available from the database have also been depicted along with their lipid-normalized counterparts.

Normalization of PCBs, dioxins, or PAHs to particulate organic carbon is not likely to change the overall observed trends, but would act to reduce some of the variability around the observed trend, particularly for contaminants with highest octanol-water partition coefficients, since hydrophobic organic contaminants preferentially sorb to organic carbon rather then suspended sediments. As a result, normalization helps achieve a greater central tendency in the data making trends easier to discern. Since hydrophobic organic contaminants preferentially sorb to organic carbon, normalizing by organic carbon rather than suspended sediment reduces the observed variability. An example of the effect of organic carbon normalization is shown below in the attached Figure 1-3.

In the following write-up, available data from each river mile transect will be shown along with NJDEP guidelines for sediment quality evaluations for the low (ER-L) and medium (ER-M) effects ranges for determining ecological risk. It should be noted, however, that the 1995 data set is already 10 years old; given the rates of solids deposition in the river (~1 to 5 inches/year), it is likely that in some areas, the 1995 surface layer is now buried 10 to 50 inches below present-day surface sediment. In addition, the section between RM 7 and RM 17 (Dundee Dam) has received much less attention than the lower six miles of the river. As a result, there is a large gap in the types (i.e., only few chemicals and biological tissues monitored), spatial (i.e., only few locations covered), and temporal (i.e., not often enough) distribution of information in the upper section of the Passaic. It is the intention of the Field Sampling Program (Malcolm Pirnie, 2005a) recently submitted by Malcolm Pirnie with assistance from Battelle and HydroQual to fill that and other data gaps in the study domain. Specific modeling data needs tables are provided in Attachment 1.2 in the QAPP.

The most pertinent plots are however presented and discussed as part of this modeling plan and the main observations about the spatial and temporal relationships that exist within the domain for the selected chemicals are presented below. The discussions are not intended to be exhaustive, but rather present a summary of the most important data features and availability.

Polychlorinated-Biphenyl's (PCBs). While the WP MPI 2005 data evaluation focused on results reported as Aroclors, this evaluation uses the EPA RI 1995 PCBs reported as 22 individual PCB congeners, and the total was estimated as the sum of these congeners; it is not however meant to be a surrogate for the real total. The 22 congeners, however, represented all ten homolog groups and comprised some of the co-planar congeners considered as the most toxic, or with dioxin-like characteristics. Figure 1-4 shows PCBs levels in surface sediments (0-15 cm) in the first 7 miles of the Lower Passaic. Average PCBs concentrations were calculated by averaging results of all samples at a river mile (RM). Examining the total PCB's (i.e., sum of 22 congener groups available in the database) reveals that at every river mile both the 23 ng/g ER-L and the 180 ng/g ER-M values are exceeded, whereas areas in the proximity of RM2, RM4 to RM4.5, RM5.2, RM6 and RM6.5 have the highest concentrations. The highest concentration was 2500 ng/g at approximately RM6. Had all congeners been reported – and not only 22 as is the case in this analysis – the guidelines would have been systematically exceeded in almost all surface sediment samples.

The data also show spatially varied PCB concentrations in the lateral direction and with depth. Large variations in PCB levels are observed along the same transect as illustrated in the five transects shown in the illustrative example in Figure 1-5. For example, for station 240A (using Tierra Solutions, Inc. (TSI) nomenclature), PCB levels are one order of magnitude higher than for station 242A, even though both stations are only 50 m apart. The same observation can be made with regard to stations 238A and 237A, which are even closer to each other. In addition, there is no clear pattern to the lateral distribution of PCB levels: concentrations vary independently of the proximity to the shore. Depth profiles of four transects are compared in Figure 1-6. In general, PCB levels peak at about two meters below the surface sediment, where concentrations (4 μg/g) could also be one order of magnitude higher than PCBs levels in surface sediments. However, as shown in Figure 1-6, PCBs are still detected at depths close or in excess of 4 meters (e.g., stations 242A, 234B, 237A), often at levels that are still above the ER-L and ER-M guidelines.

Figure 1-7 gives for each triplet samples per river mile the depth at which the highest levels of PCBs are detected. Most of the peaks (~40%) occur at depth between 1 and 2 meters while 25% occur in deeper sediments.

Not withstanding that some portions in the Passaic River are likely "erosional" areas, it is noteworthy that newly deposited solids have most probably covered a large portion of the 1995 "surface sediment layer". The proposed low and high resolution programs (Malcolm Pirnie, 2005a)

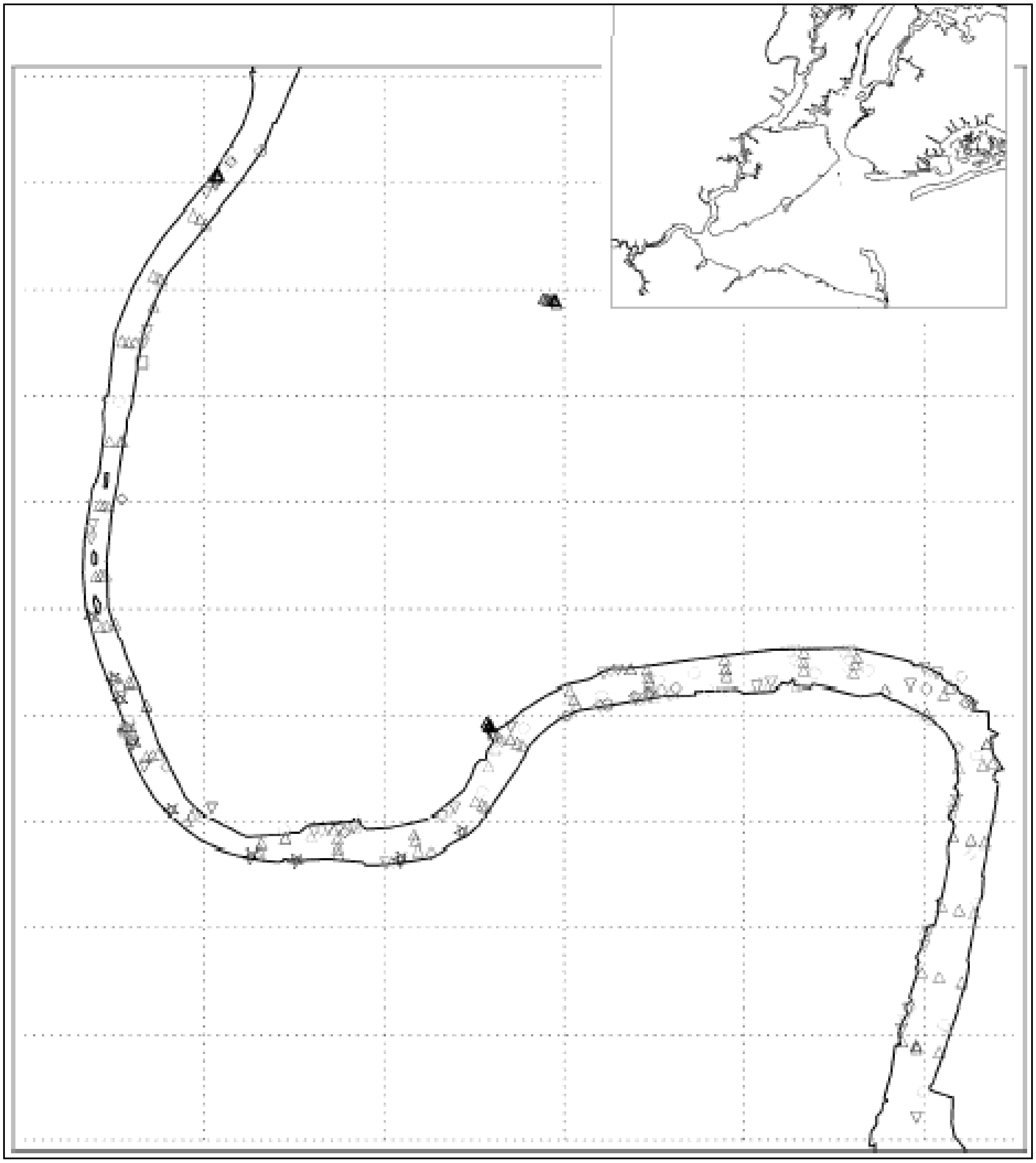


Figure 1-2. Distribution of sampling locations laterally and horizontally in the Lower six miles of the Passaic

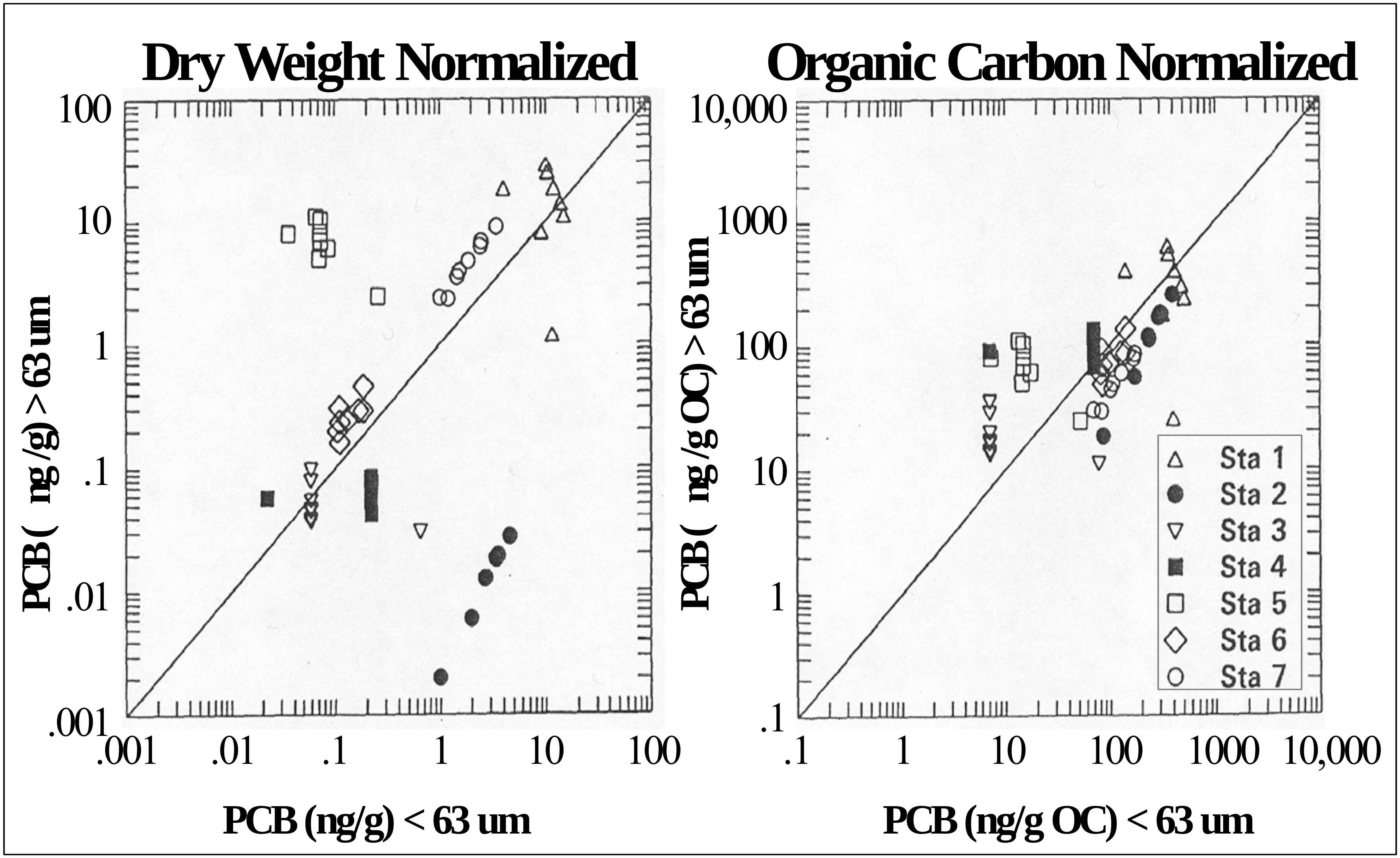


Figure 1-3. Dry Weight vs. Carbon Normalized PCB Concentrations.

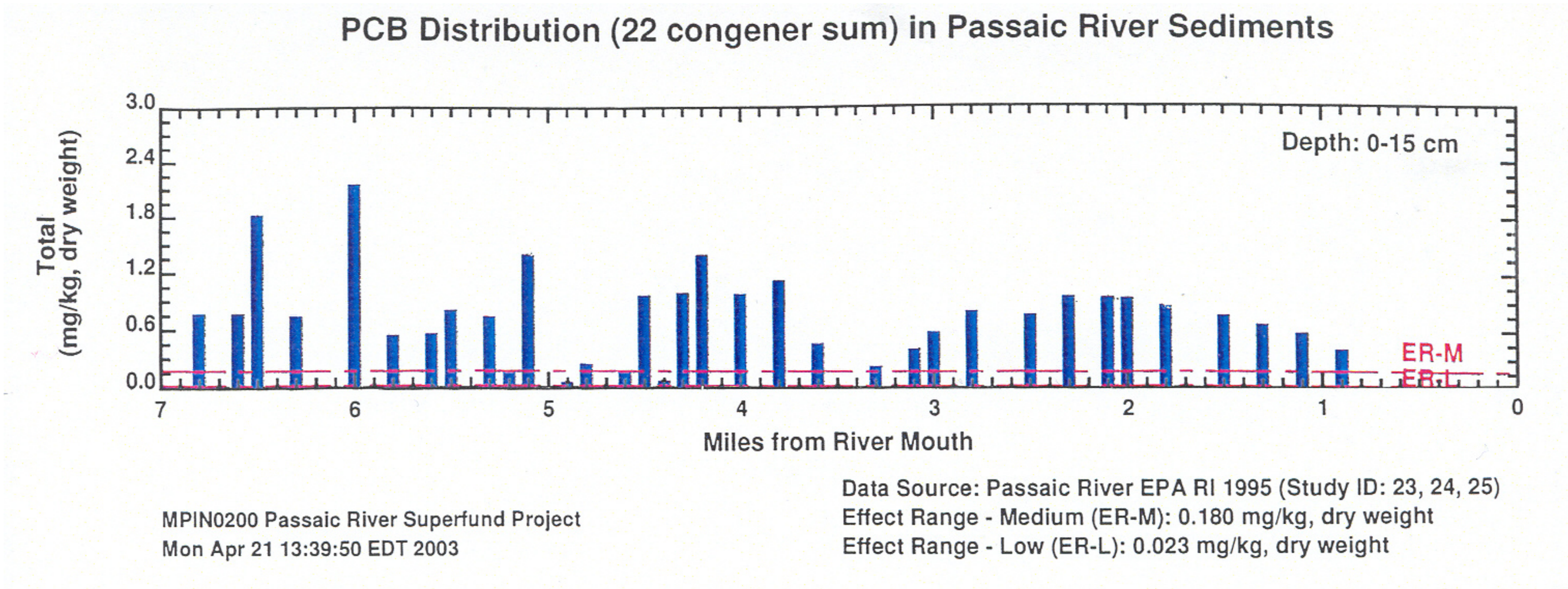


Figure 1-4. Spatial distribution of PCBs in the Lower Passaic River surface sediments.

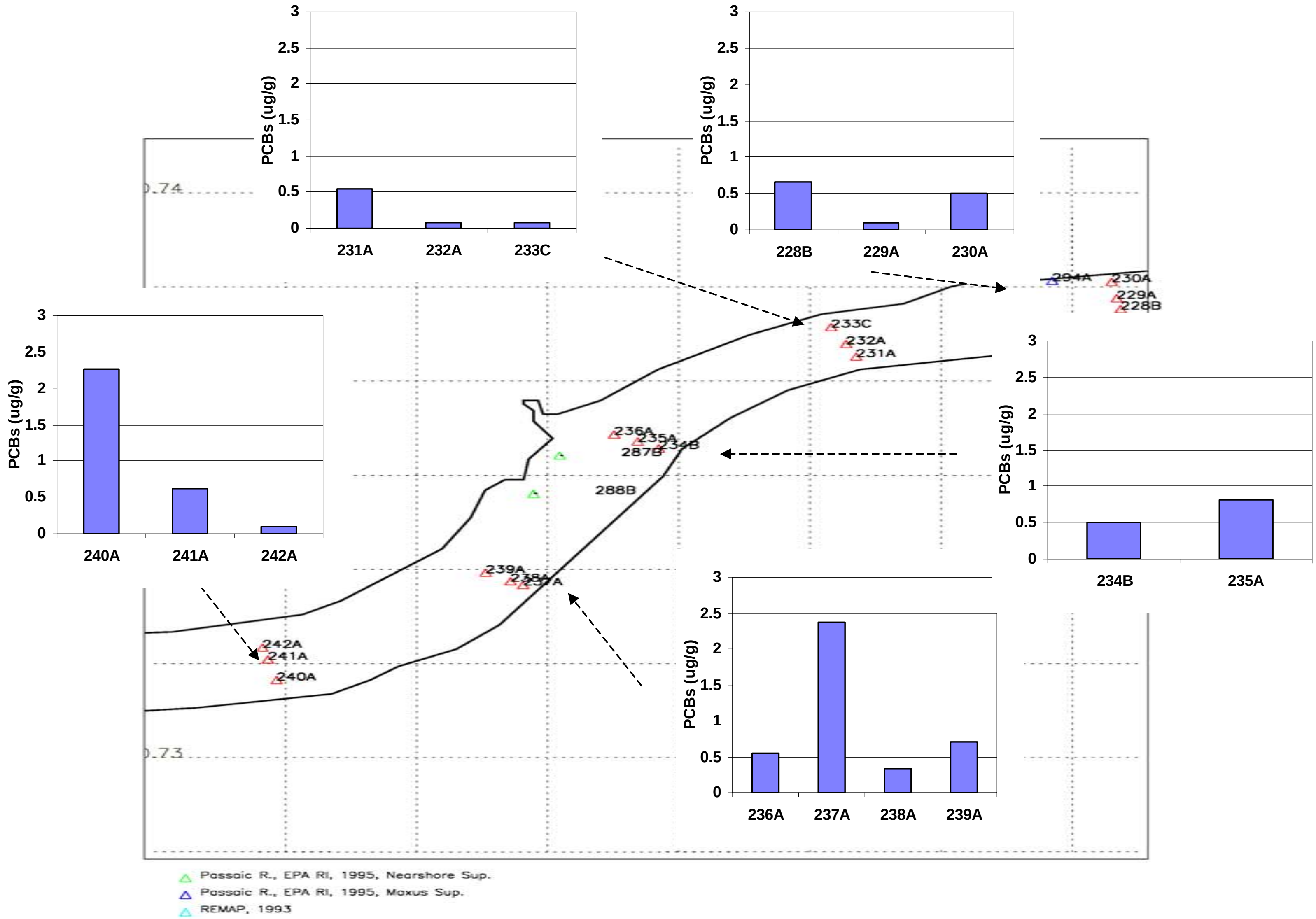


Figure 1-5. Lateral Distribution of PCBs at selected stations.

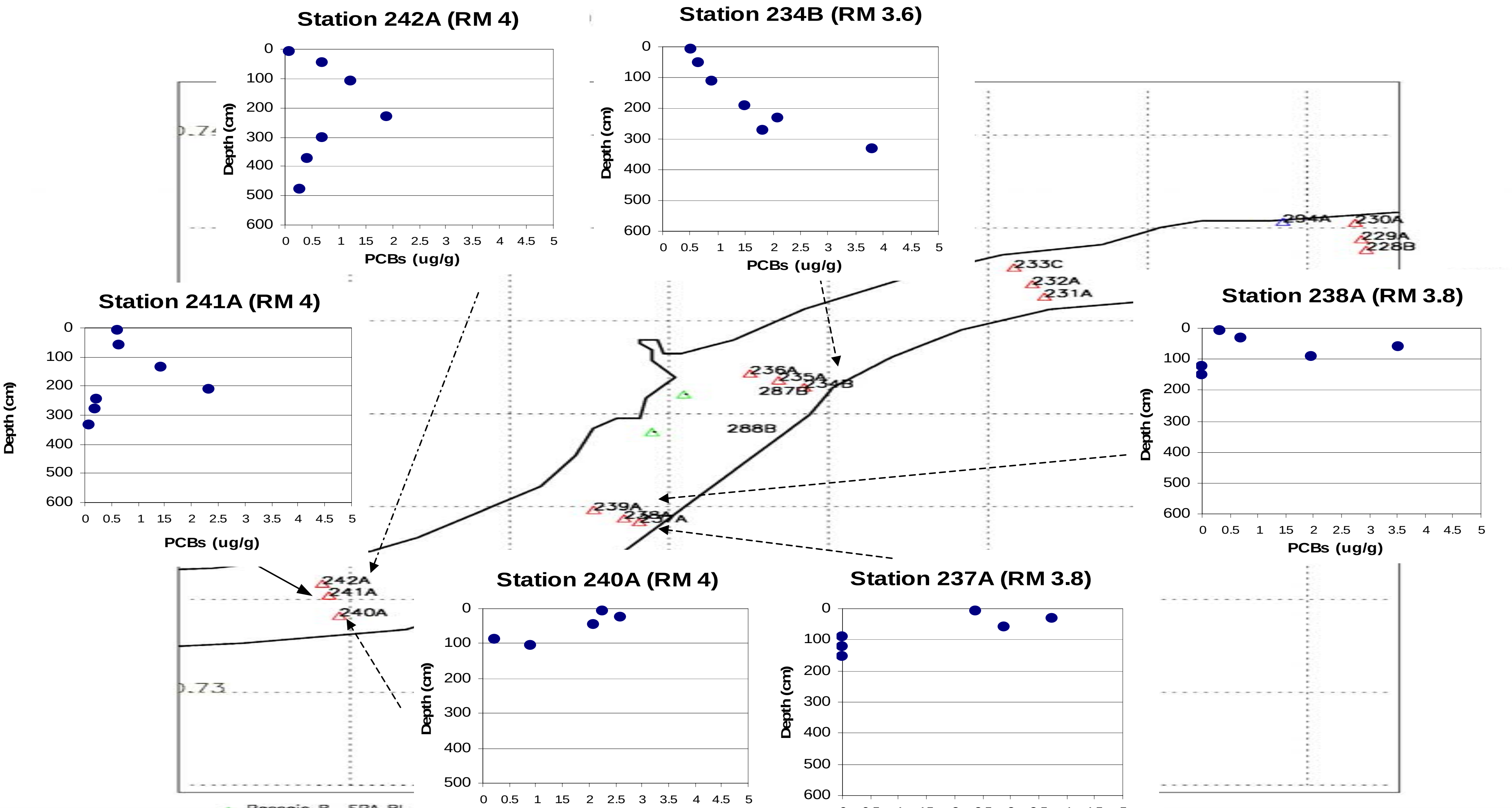


Figure 1-6. Depth profiles of PCBs at selected locations.

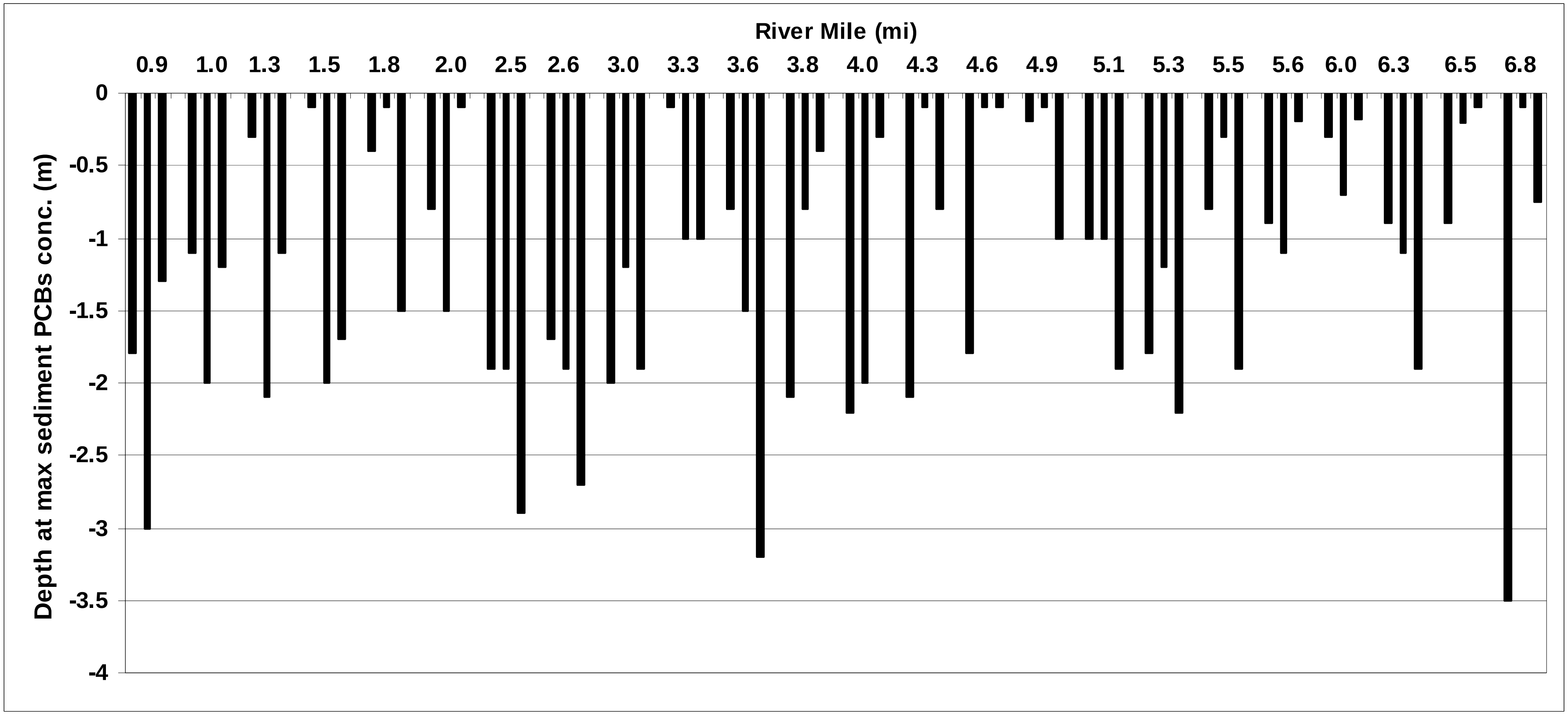


Figure 1-7. Depths at which the highest PCB concentration is detected as a function of River Mile in the Lower Passaic River

will be instrumental in tracking not only the depositional/erosional patterns in the Passaic, but also the fate of the contaminants as they are buried or re-suspended.

Dioxins: Although PCDD/F (dibenzo-p-dioxins and furans) consists of more than 200 compounds, only a fewer number of congeners are commonly analyzed. Those include OCDD (1,2,3,4,6,7,8,9-Octachlorodibenzo-p-dioxin), and 2,3,7,8-TCDD (2,3,7,8-Tetrachlorodibenzo-p-dioxin), both often considered as posing the highest risk. However, there are no NJDEP guidelines available for dioxin-related ecological risk assessment.

Seventeen (17) congeners were analyzed for the 1995 EPA RI program (Table 1-2). Longitudinal and depth profiles for TCDD and OCDD in the Passaic River sediments are presented in Figure 1-8 and Figure 1-9. The levels of TCDD in surface sediments (i.e., between 0 and 15 cm) are relatively low; the mean concentration for the lower 6-mile stretch is about 0.80 ng/g and the highest levels are found between RM2 and RM4, where concentrations reach 13.5 ng/g (station 224A at RM2.5). It is noteworthy that only averages of triplet samples at each river mile, not individual stations, are shown in Figure 1-8 and Figure 1-9. Although there are few measurements preformed in 1993 on top 2-cm sediment slices from Raritan Bay, Jamaica Bay and Newark Bay, no such measurements are reported for Passaic River sediments. However, the greatest concentrations occur deep in the sediments as shown in the lower panels of Figure 1-8. About 100 to 134 cm below surface, TCDD levels are almost two orders of magnitude higher than the highest recorded concentration in the surface sediment (1,100 ng/g vs. 13.5 ng/g).

Table 1-2. List of dioxin congeners analyzed for the 1995 EPA RI program.

Dioxin Congeners Analyzed during the 1995 EPA RI
1,2,3,4,6,7,8-Heptachlorodibenzo-p-dioxin
1,2,3,4,7,8-Hexachlorodibenzo-p-dioxin
Dioxin (tagged), 13C-1,2,3,6,7,8-HXCDD
1,2,3,7,8-Pentachlorodibenzo- p-dioxin
1,2,3,7,8,9-Hexachlorodibenzo-p-dioxin
2,3,7,8-Tetrachlorodibenzo-p-dioxin (**TCDD**)
1,2,3,4,5,6,7,8-Octachlorodibenzo-p-dioxin
1,2,3,4,6,7,8-Heptachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzofuran
1,2,3,4,7,8,9-Heptachlorodibenzofuran
1,2,3,6,7,8-Heptachlorodibenzofuran
1,2,3,7,8-Pentachlorodibenzofuran
1,2,3,7,8,9-Hexachlorodibenzofuran
2,3,4,6,7,8-Hexachlorodibenzofuran
2,3,4,7,8-Pentachlorodibenzofuran
2,3,7,8- Tetrachlorodibenzofuran (TCDF)
1,2,3,4,5,6,7,8-Octachlorodibenzofuran (**OCDD**)

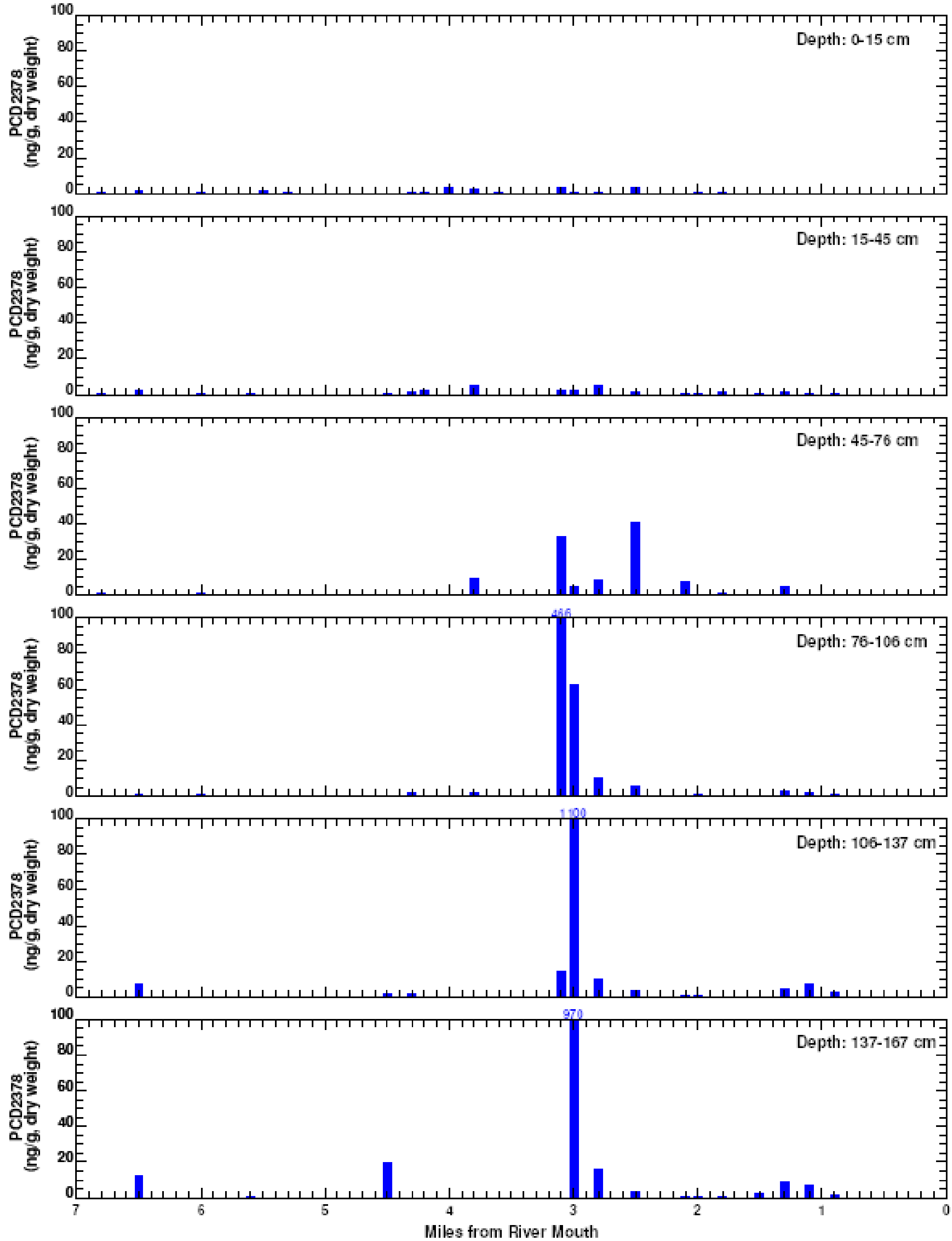


Figure 1-8. Longitudinal and depth profile of 2,3,7,8 TCDD in the Lower Passaic River sediments.

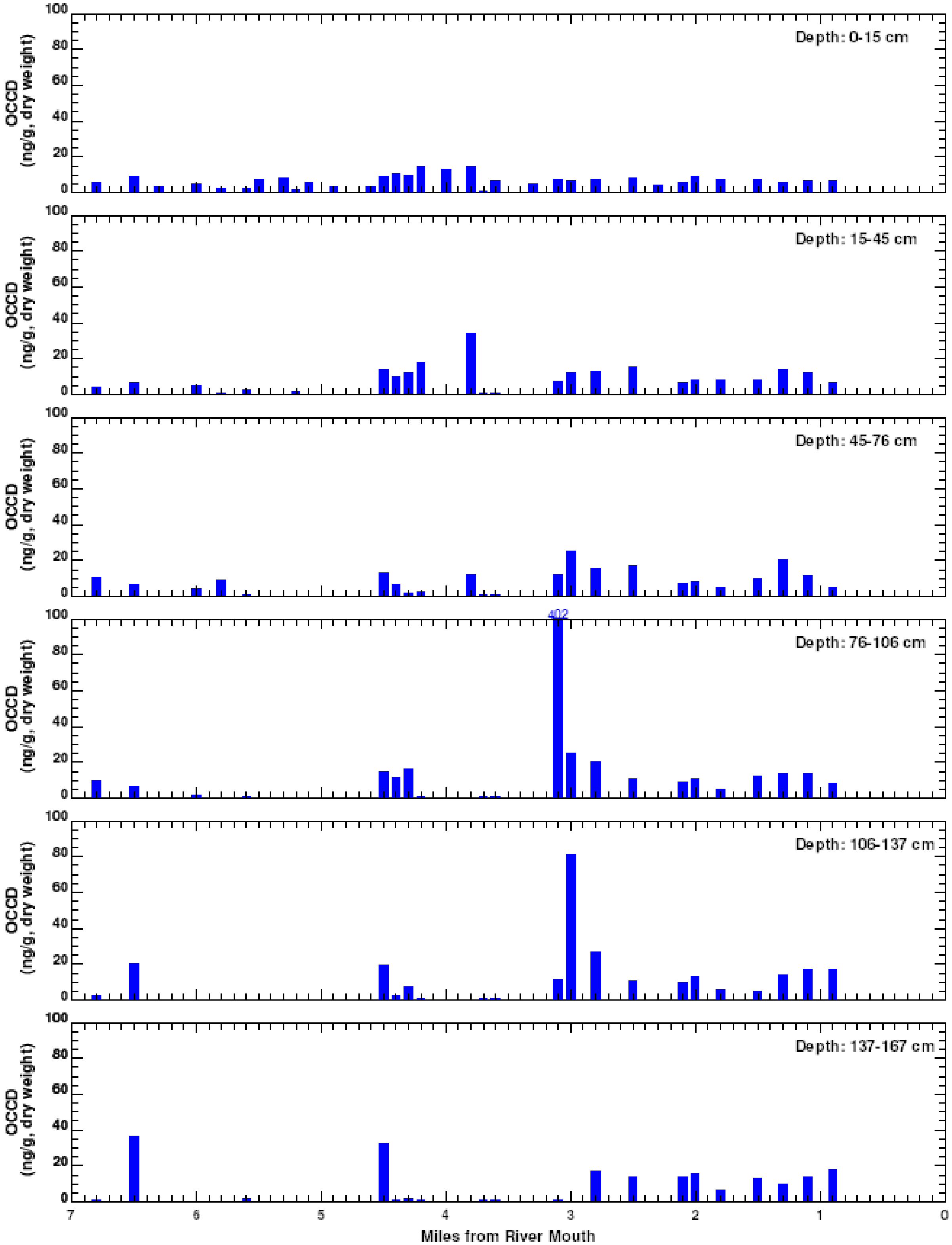


Figure 1-9. Longitudinal and depth profile of OCDD in the Lower Passaic River sediments.

OCDD are widely distributed throughout the lower 6 miles of the Passaic River (Figure 1-9). The highest levels in the surface sediment are also found between RM2 and RM4, where concentrations reach 22.6 ng/g (Station 239A at RM3.7). As with TCDD, the highest OCDD concentrations are found in deeper sediments. For instance, levels of OCDD reach a maximum concentration of 802 ng/g in sediment buried between 76 cm and 106 cm below the "1995 surface sediment" (Station 285A at RM3.1). As with PCBs, the vertical profiles of OCDD concentrations between river mile 2.8 and river mile 3.8 show that peak concentrations occur at depth between 1 and 2 meters below the surface sediment (peak concentration of 802 ng/g not shown), although in some instances OCDD is still detected at a depth of 4 m below the surface (Figure 1-10).

Polyaromatic Hydrocarbons (PAHs). The PAH's have 23 individual components within the 1995 EPA RI data set. Most of the components have no risk guidelines, as is the case with some PCBs congeners. For those species that have NJDEP guidelines (i.e., acenaphthene, benzo(a)pyrene, acenaphthylene, anthracene, Benzo(a)anthracene, benzo(g,h,i)pyrelene, benzo(k)fluoranthene, chrysene, dibenzo(a)anthracene, fluoranthene, fluroene, indeno(1,2,3-c,d)pyrene, 2, methylnaphthalene, phenanthrene, pyrene), the analysis reveals that much of the data are between the ER-L and ER-M, and in some cases, there are peaks that are orders of magnitude higher than the ER-M. Total PAH concentrations show at least two significant peaks in its along channel distribution. One occurs at approximately RM3.75 with the greatest concentration in the surface layer, and another near RM4.5 where the highest concentrations are found in the top 15 cm layer. Although initially decreasing with depth, PAHs concentrations increase in deeper layers with a sustained peak even past one and half meters below the sediment surface. Many of the PAHs show extremely high concentrations at all depths at RM4.5. Figure 1-11 shows an illustrative example of the spatial extent of chrysene concentrations in the Lower Passaic River sediments. Both surface and subsurface sediments contain chrysene levels that exceed the ER-L and ER-M guidelines, in particular between RM2 and RM5. As is the case for PCBs and dioxins, PAHs are likely candidates to be considered as COPCs.

Metals. For the metals that are available in the database, the spatial distribution and the 6-mile mean concentrations of cadmium, copper, lead, mercury, nickel and zinc in surface sediments of the Lower Passaic River are shown in Figure 1-12 and Figure 1-13. All six metals have elevated levels between RM3.5 and RM5, almost always above the ER-L and often in excess of the ER-M guidelines. Also, the average concentrations of all metals, except for cadmium, exceed the ER-M. It is noteworthy that mercury shows concentrations at 10 and 20 times the medium and low range of ecological effects.

However, in spite of the exceedances reported above, the application of Equilibrium Partitioning (EqP) approach (Di Toro et al., 1991) will help determine which metals should be on the COPCs list. The final list of metals on the COPC list will be determined based on the needs of the human health and ecological risk assessments. The determination of the final list needs also to

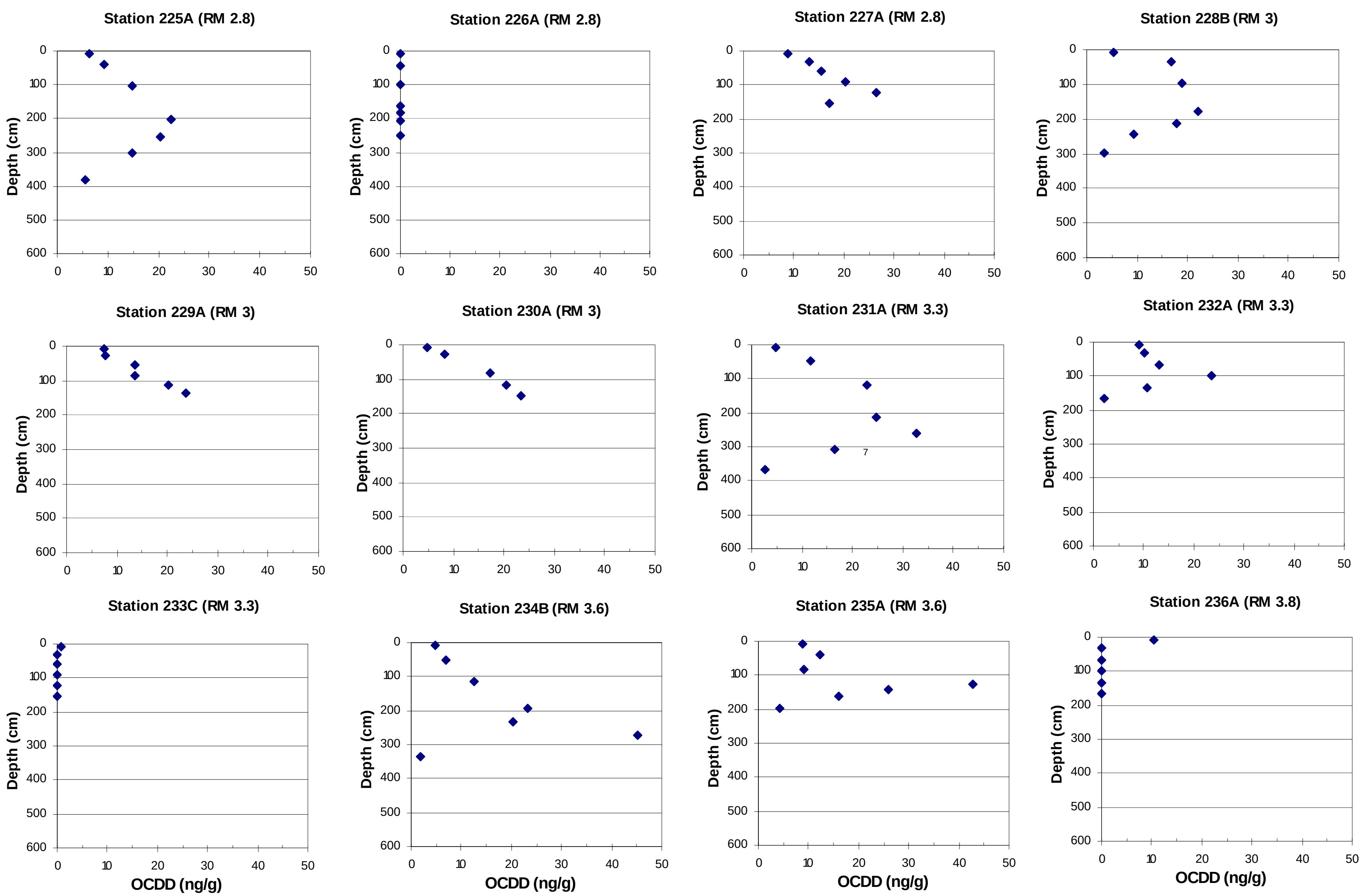


Figure 1-10. Vertical profiles of OCDD sediment concentrations between river mile 2.8 and 3.8 in the Passaic River (EPA RI, 1995).

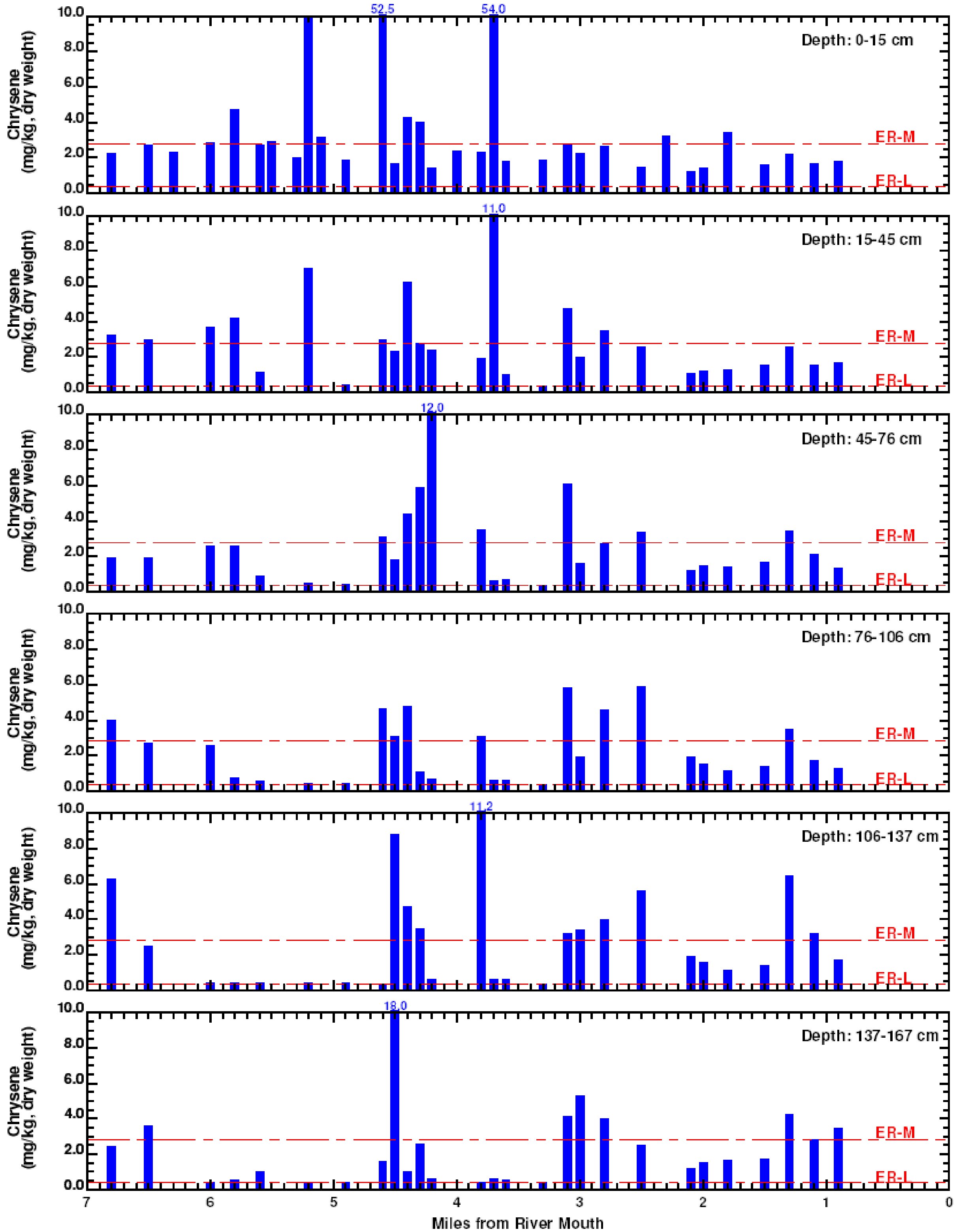


Figure 1-11. . Longitudinal and depth profiles of chrysene in the Lower Passaic River.

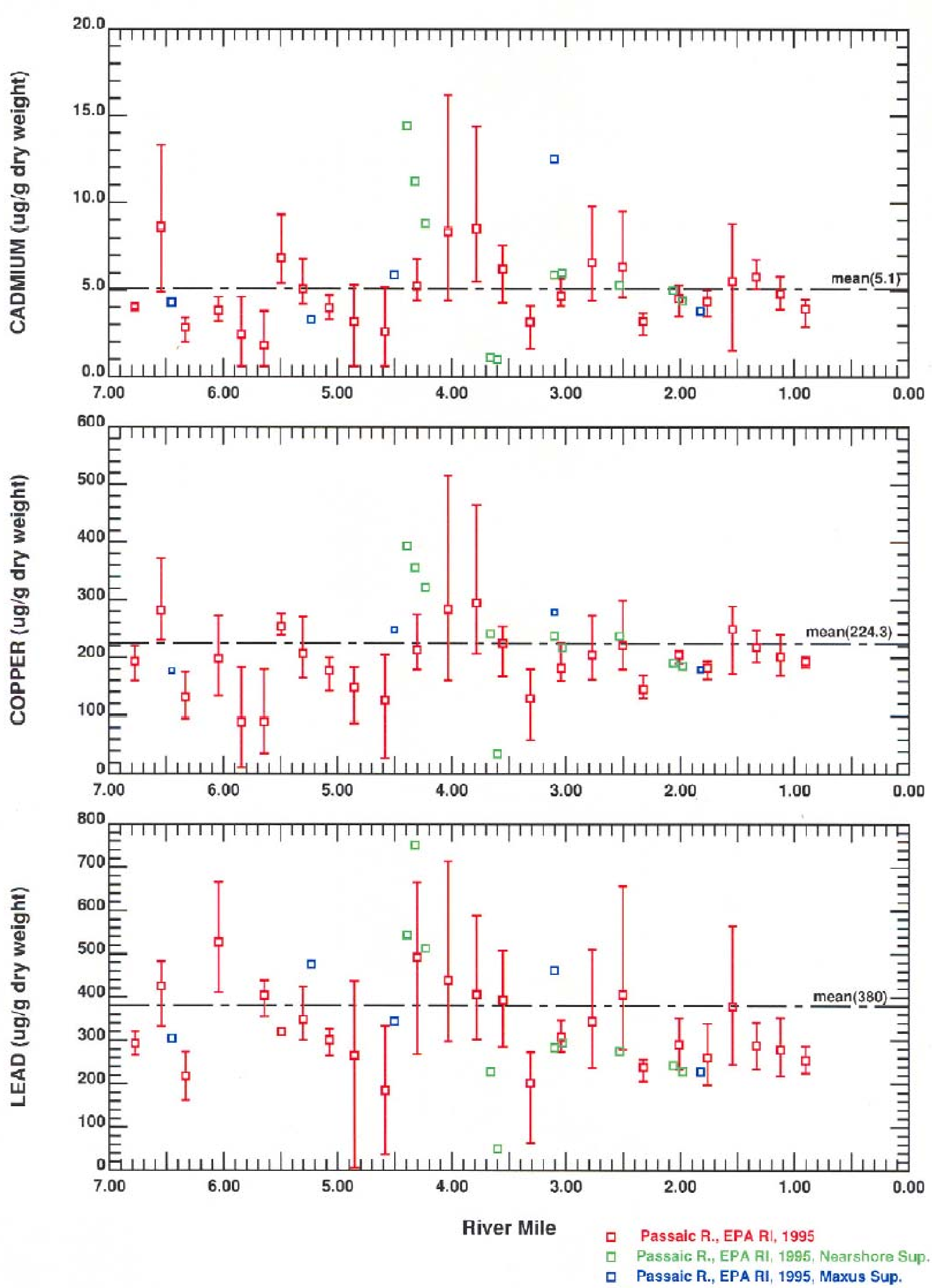


Figure 1-12. . Spatial distribution of cadmium, copper, and lead in the Lower Passaic River surface sediments.

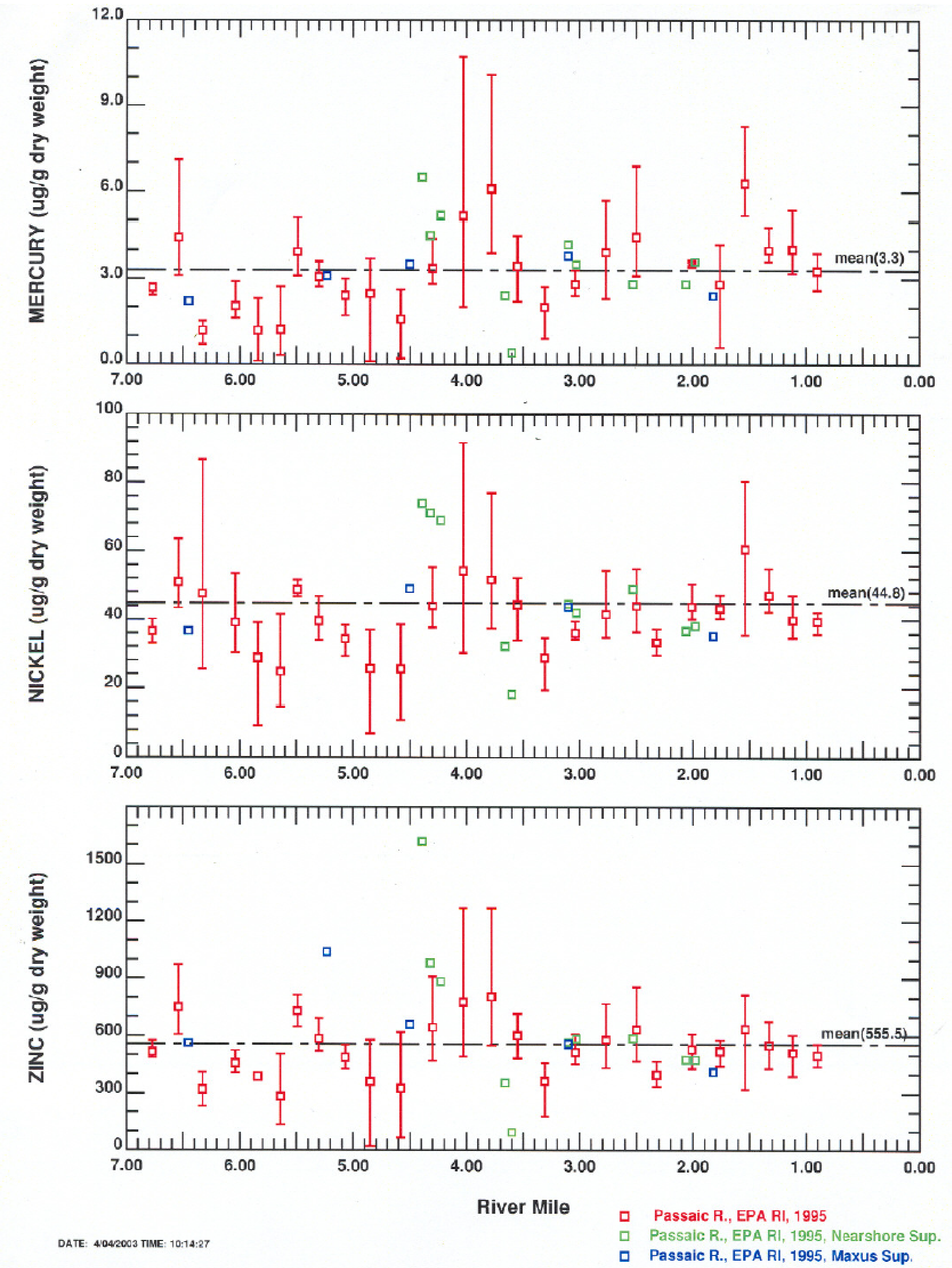


Figure 1-13. Spatial distribution of mercury, nickel and zinc in the Lower Passaic River surface sediments.

be reviewed for consistency with Battelle Pathway analysis report. One approach to determining metals toxicity requires measurements of the simultaneously extractable metals (SEM = sum of [Cd], [Cu], [Ni], [Pb], [Zn]) and acid volatile sulfide (AVS) in the sediments to evaluate the toxicity of the metals. In general, the sum of SEM must be less than the AVS for no toxicity to be present. Tierra Solutions collected AVS/SEM data as part of its 1999-2000 ecological sampling program. Until these data are analyzed, the significance of the high levels of metals encountered in the Passaic River sediments cannot be assessed. In addition, a final determination must be reached on the validity of the AVS/SEM approach as an acceptable method for evaluating metals toxicity.

Pesticides. NJDEP provides guidelines on dry weight and organic carbon bases for a number of agricultural chemicals detected in the Lower Passaic River sediment (Table 1-3). This analysis has focused on the contaminants of concern that the Contamination Assessment and Reduction Project (CARP) program has selected for modeling purposes (i.e., DDT, p,p'-DTT and chlordane). According to WP MPI 2005 data evaluation report, more than 100 million pounds of DDT and its by-products have been discharged in the Passaic River in the 1940s. The horizontal spatial plots reveal that total DDT and p-p'-DDT concentrations on dry weight basis in the surface sediments are highest near RM2 and RM3 and that concentrations all along the six-mile stretch of the Lower Passaic are elevated and often exceed the NJDEP standards (Figure 1-14).

The p-p'-DDT concentrations extend down into the sediment layers and reach a peak between at 0.5 and 0.75 m (Figure 1-15). Data for DDT is only available in the top 15 cm. However, the same DDT concentrations, once normalized to organic carbon, exceed the NJDEP guidelines (12 mg/Kg organic carbon) only on two occasions (Figure 1-14), whereas p-p'-DDT organic carbon normalized concentrations remain under the guidelines (70 mg/Kg organic carbon) in the surface sediment and exceed it only once in the deeper sediments (Figure 1-16).

Grain size distribution and total organic carbon (TOC). Grain size analysis conducted on the Passaic River sediment suggests that most of the particles are cohesive in nature. Using available data from the NOAA and TSI database, Figure 1-17 shows a spatial profile of percent fines measured in sediment samples collected in the Passaic River. The data clearly indicates the dominance of small size particles throughout the domain: only eight measurements out of 50 had less than 40% fine particles, whereas the rest contained between than 60% and 100%. The data also shows the absence of any clear spatial pattern between the lower and upper sections of the Passaic River. Fines are usually cohesive particles of less than 63 μm diameter, composed mainly of clay, silt and organic particles. It is noteworthy that total organic carbon measurements in the Passaic River sediments seem to correlate well with the class size distribution as shown in Figure 1-18. High total organic carbon content can be observed where fine-grained sediments are found. However, non-fine particles are still present and can affect sediment erosion rates.

Table 1-3. NJDEP Sediment Screening Guidelines.

Pesticides	**Lowest Effects Level (LEL) (mg/kg, dry weight)**	**Severe Effects Level (SEL) (mg/kg organic carbon, dry weight)**
Aldrin	0.002	8
Benzohexachloride (BHC)	0.003	12
a-BHC	0.006	10
b-BHC	0.005	21
y-BHC (Lindane)	0.003	1
Chlordane	0.007	6
DDT (Total)	0.007	12
Op+pp-DDT	0.008	71
pp-DDD	0.008	6
pp-DDE	0.005	19
Dieldrin	0.002	91
Endrin	0.003	130
Hexachlorobenzene (HCB)	0.020	24
Heptachlor epoxide	0.005	5
Mirex	0.007	130

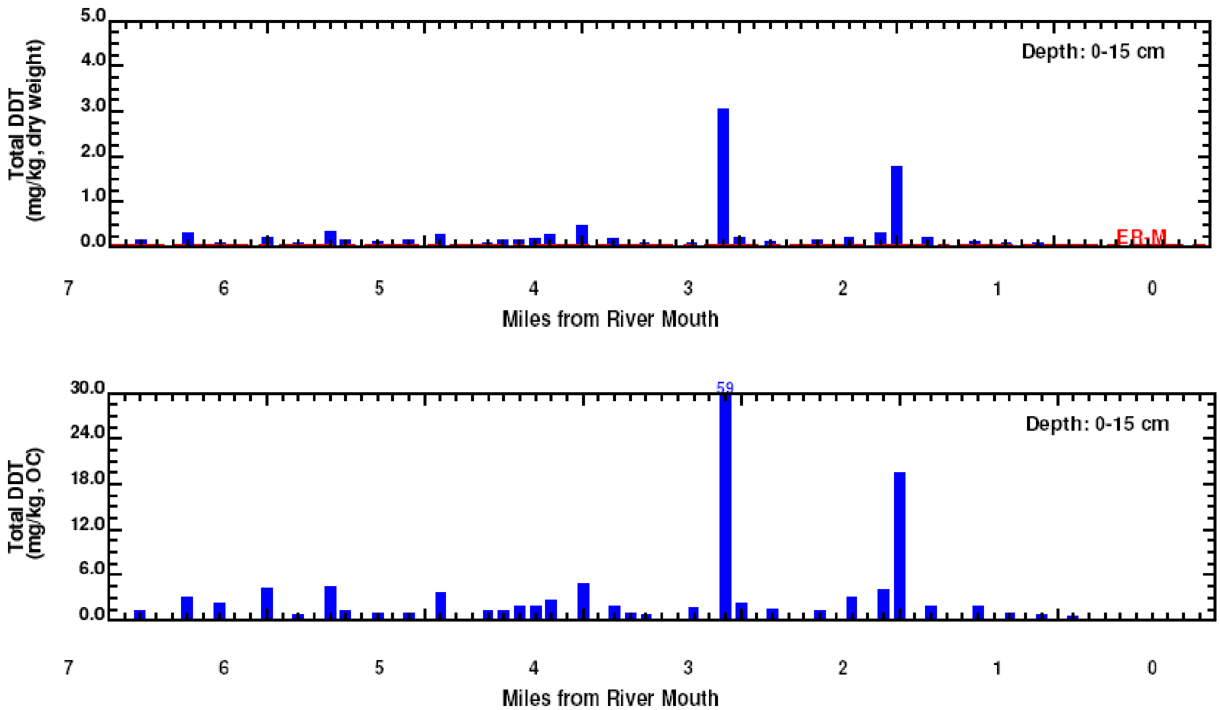


Figure 1-14. Spatial distribution of DDT on a dry weight (top panel) and organic carbon (lower panel) in the Lower Passaic River surface sediments.

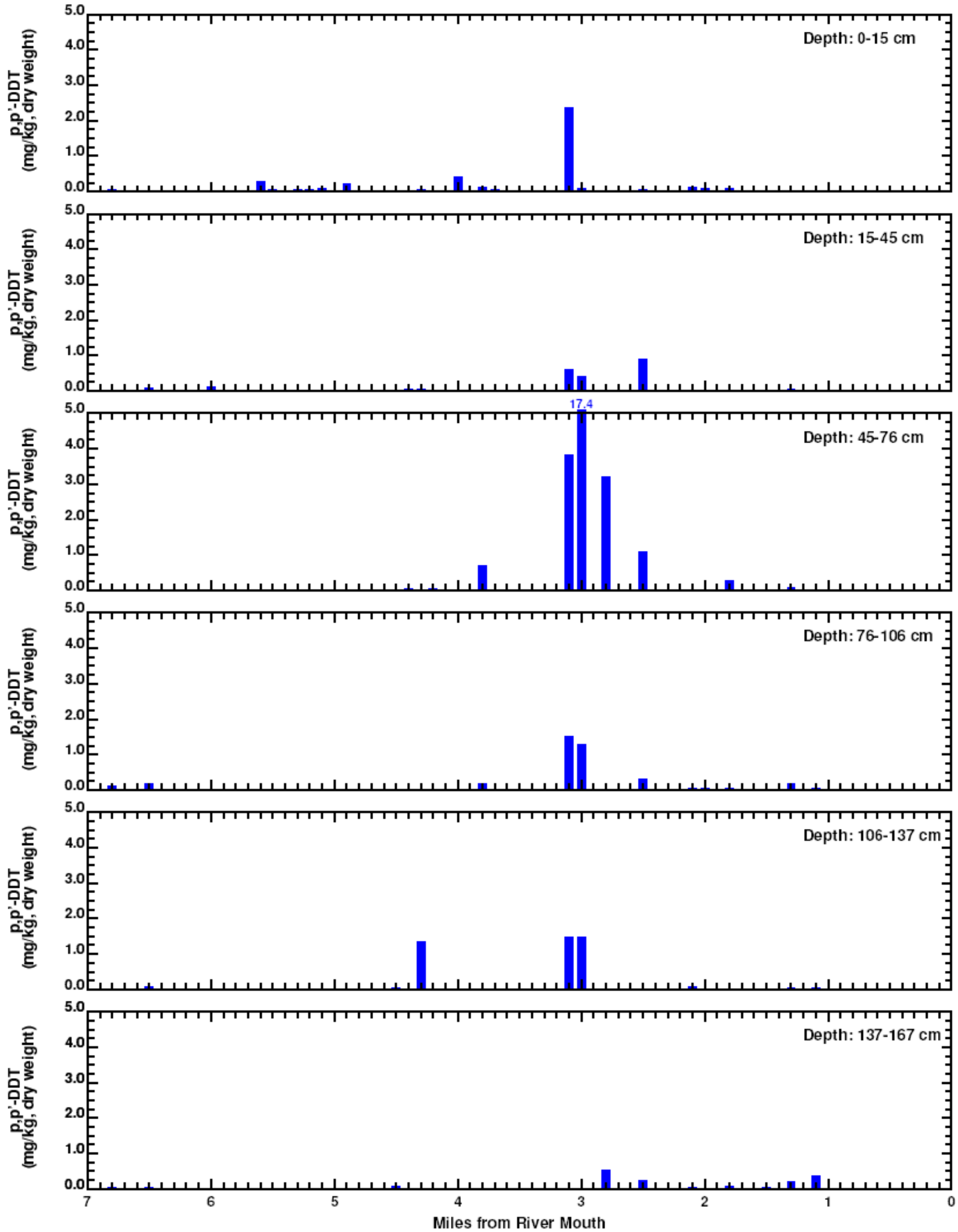


Figure 1-15. Spatial distribution of p-p'-DDT in the Lower Passaic River surface sediments (mg/kg dry weight).

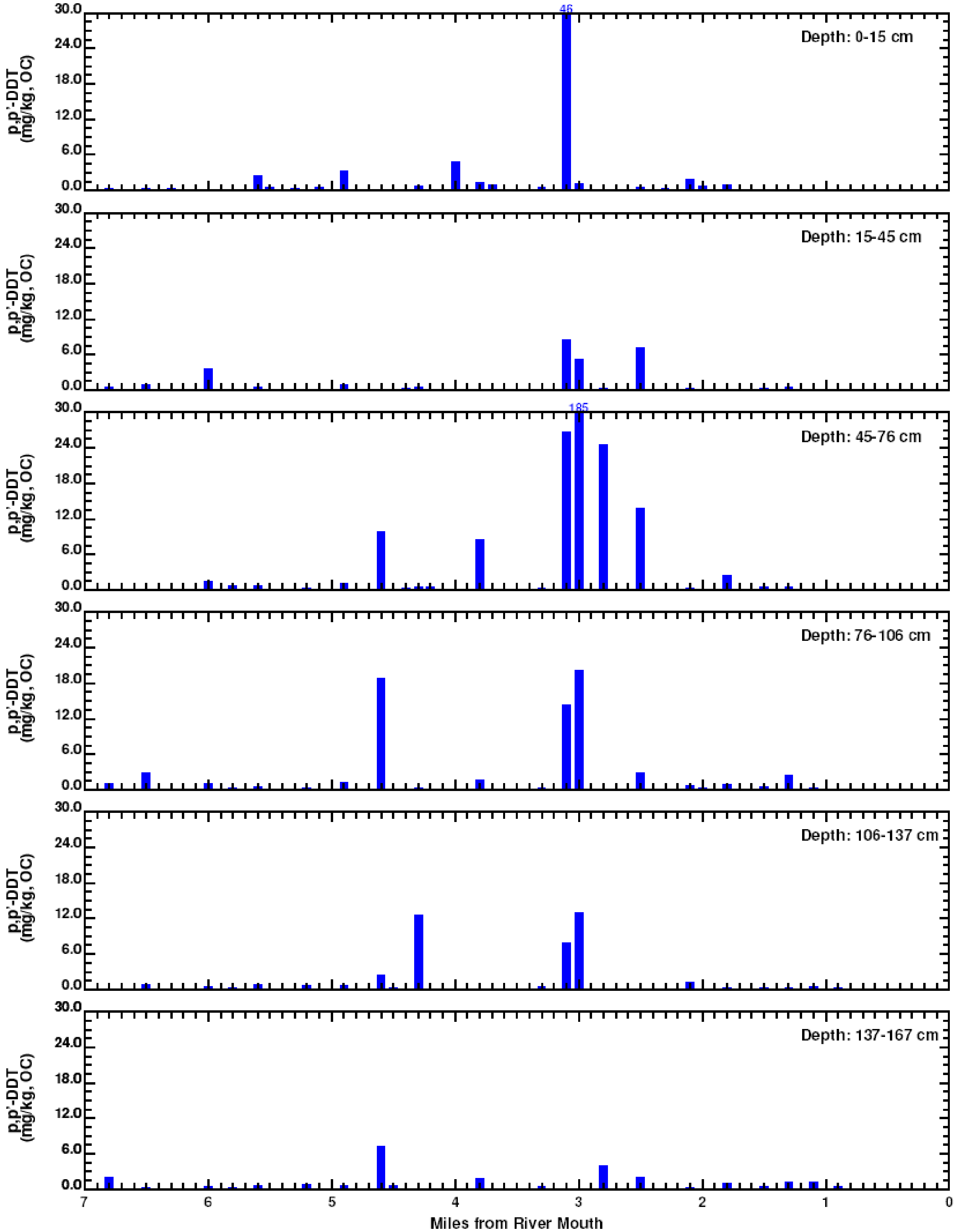


Figure 1-16. Spatial distribution of p-p'-DDT in the Lower Passaic River surface sediments (mg/kg organic carbon dry weight).

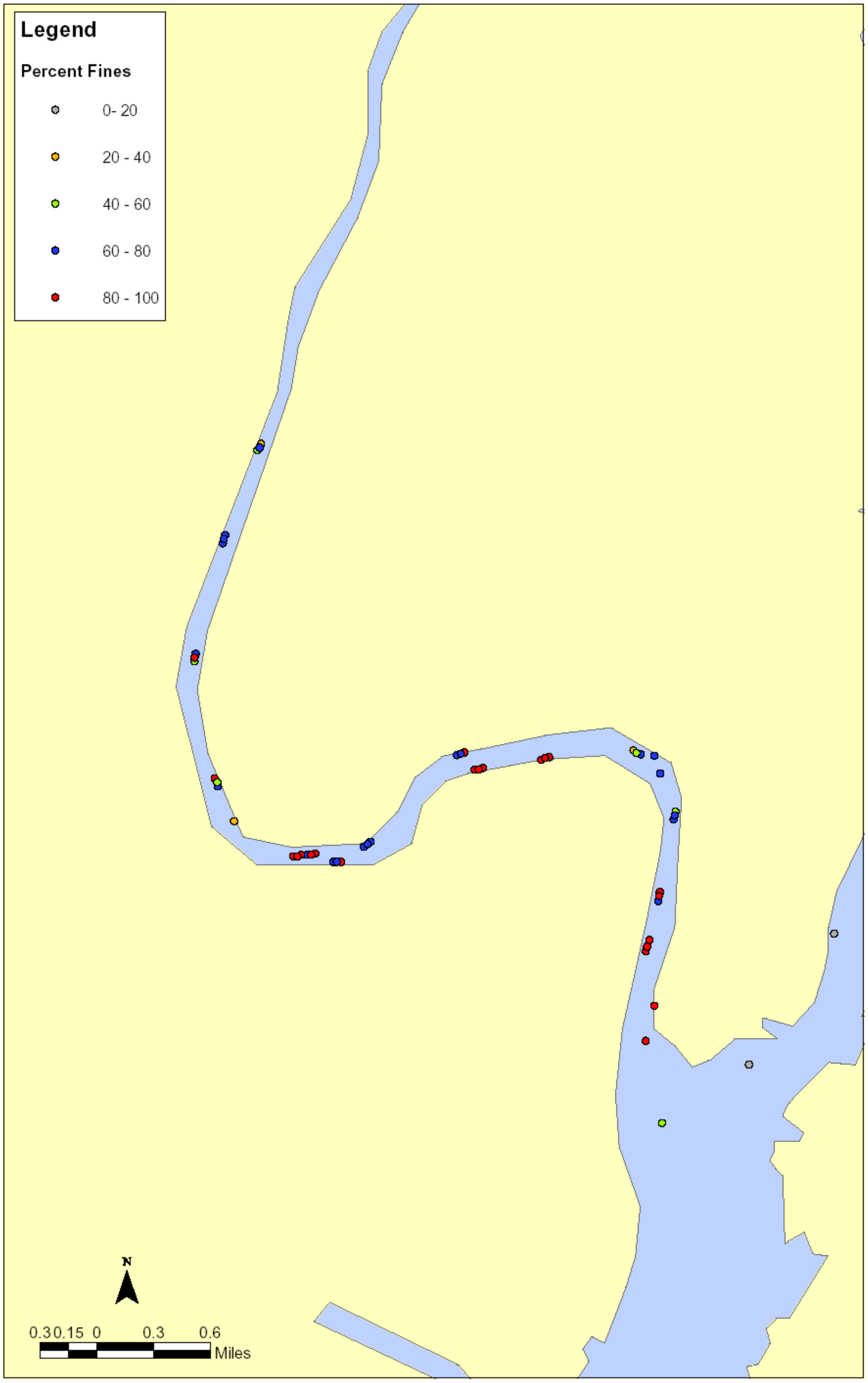


Figure 1-17. Distribution of percent fines in Passaic River sediment.

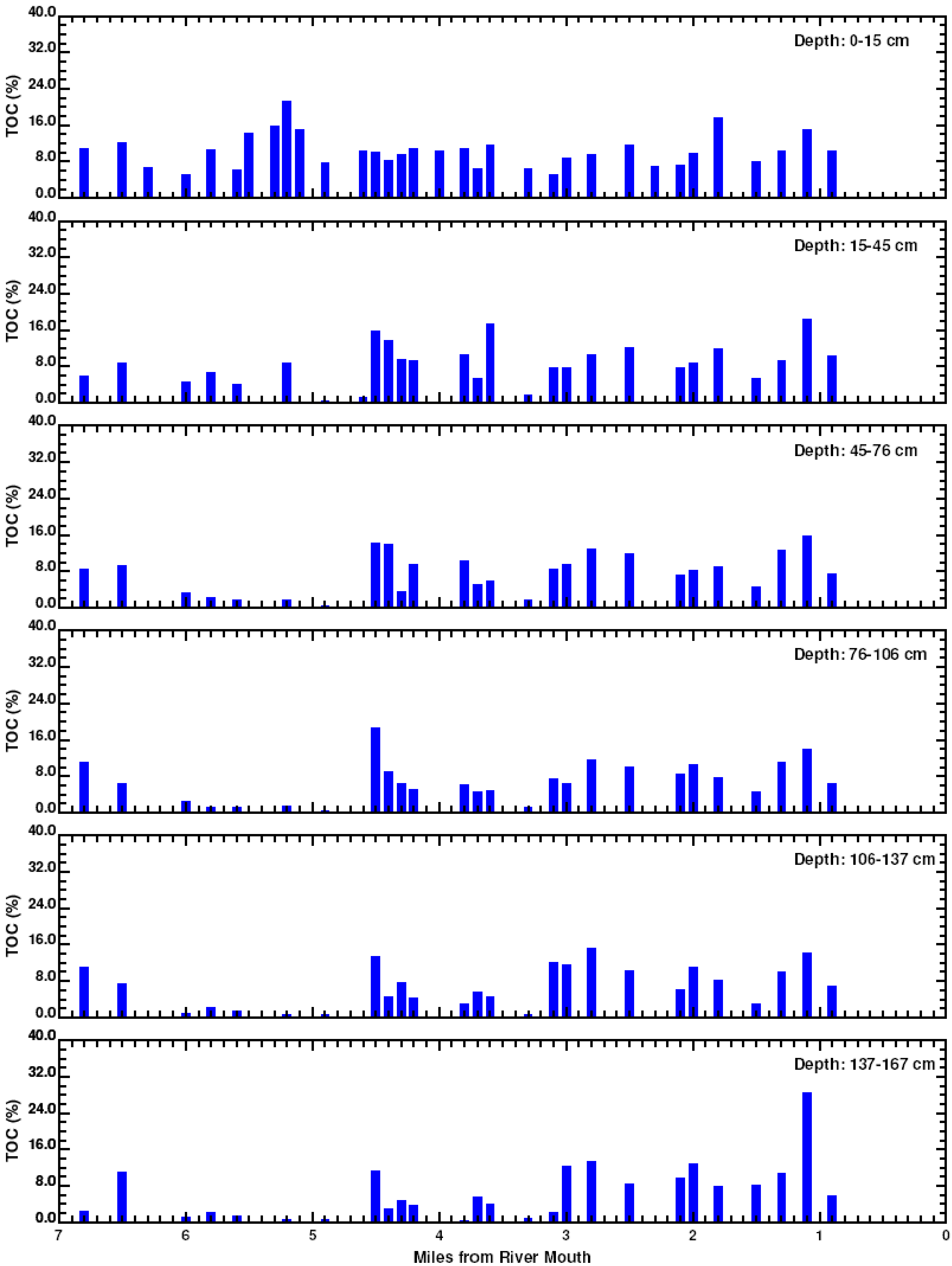


Figure 1-18. Spatial distribution of percent total organic carbon in Lower Passaic River sediments.

Particle Mixing (bioturbation). Biological information from marine systems suggests that macrofauna residing on surface sediments exert an influence on the fluxes of contaminants to and from the sediment (Di Toro, 2001). The feeding mode (i.e., tube feeding) and respiration processes result in the mixing of particles over a layer that could extend to few cm under the surface. As deposit feeders ingest sediment, they return it to a different location in the sediment. The mixing, also called bioturbation is an important process that affects the fluxes of contaminants to the water column. Literature values indicate that average depth of particle mixing as a function of sedimentation rate is about 10 cm as shown in Figure 1-19 (Boudreau, 1994). The Sediment Profile Imaging (SPI) survey of the Lower Passaic River completed in June 2004 (Germano and Associates, 2005) will be evaluated to obtain a site-specific refinement of the literature value, including potentially different values for fresh water and brackish reaches of the river.

1.4.2 Data Evaluation: Water Column

For the water column, HydroQual's initial data analysis relied on the field programs carried out under the CARP sampling programs which covered 27 pesticides, 209 PCB congeners, 17 dioxin/furan congeners, 3 metals, and 21 PAH compounds, and on the Regional Environmental Monitoring and Assessment Program (REMAP) analyte list that included 23 PAH compounds, six DDT/DDE/DDDs, 10 other chlorinated pesticides, 4 major and 12 trace elements, 20 PCB congeners, and 16 dioxin/furan congeners. Summaries of the available data are given in Table 1-4 (for CARP). In general, however, despite the large number of programs, water column data significantly lag behind the sediment data. As a result, any attempt to construct the contamination status of the river faces the uncertainty associated with temporal and spatial patterns that reflect sample variability due to other factors (e.g., time in the tidal cycle) rather than true patterns. In addition, most of the studies on the different environmental matrices were not conducted concurrently.

Table 1-4. Summary of available water column data for the Lower Passaic River Restoration Project.

PARAMETERS	STUDY NAME			
	NYSDEC	*NJ CARP Data*	*NJADN*	*NYDEP*
PCBs (homolog sums)	x	x	x	
Dioxins/furans	x	x	x	
Cd	x	x	x	
Hg	x	x	x	
PAHs	x	x	x	
Chlordane	x	x		
DDT and metabolites	x	x		
salinity		x (2000-01 and 2001-02 data)		
temperature		x (2000-01 and 2001-02 data)		
current		x (2000-01 and 2001-02 data)		
Pb		x		
POC	x	x		
DOC	x	x		
SS	x	x		

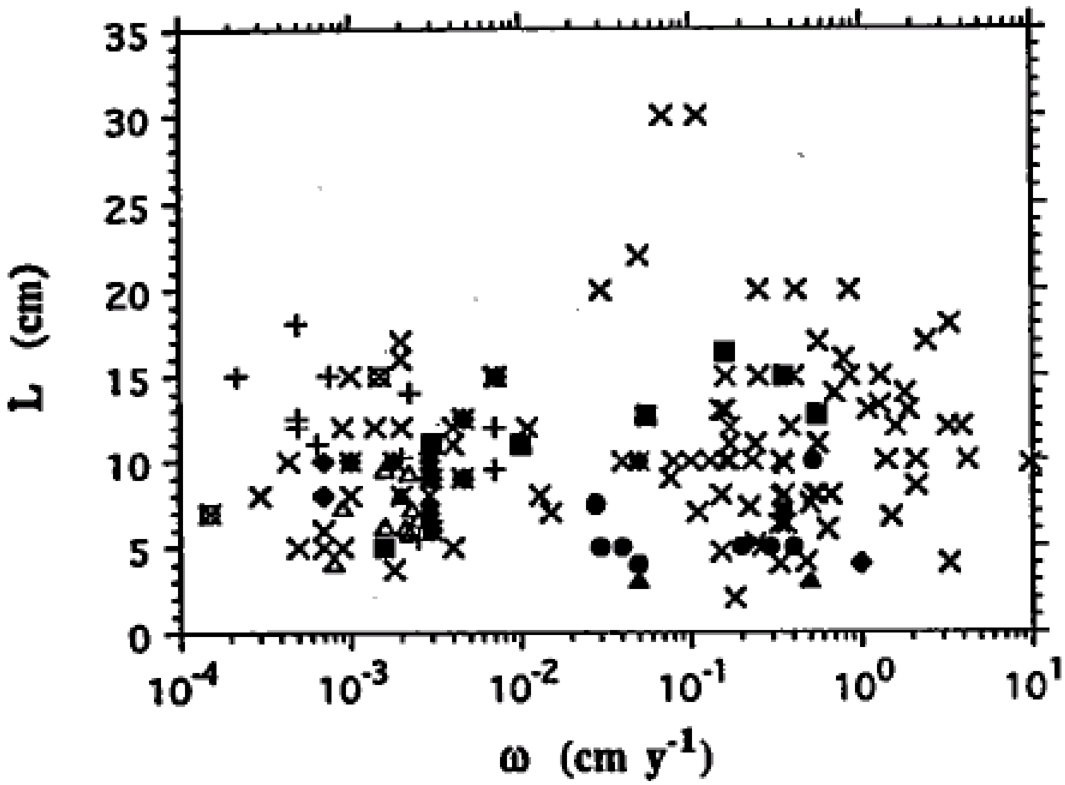


Figure 1-19. Depth of particle mixing as a function of sedimentation rate (Boudreau, 1994).

Figure 1-20 illustrates the scarcity of water column data for a typical contaminant (e.g., 2,3,7,8-TCDD). The figure presents concentrations versus distance along a transect through the Passaic River and Newark Bay. The panels present concentration of 2,3,7,8-TCDD in fish (white perch and mummichog) (top panel), water (middle panel) and sediments (bottom panel). There were only six water column sampling locations – three samples collected per location - from the Dundee Dam into Newark Bay, compared to 22 for the sediments. Although the water column concentrations appear to be highest in the lower 6-miles, the water samples were grab samples that distort any temporal pattern resulting from inter-tidal variations occurring in the estuarine and tidally-influenced sections of the Passaic. The uncertainty in the spatial pattern, coupled with the scarcity of data suggest that a better understanding of the behavior of the contaminants in the water column requires a detailed sampling program that accounts for inter-tidal variability and a variety of flow regimes.

The need for a more robust water sampling program, both spatially and temporally is further illustrated in the temporal representation of flow, total suspended solids and dissolved and particulate contaminant concentrations. Figure 1-21 shows all available paired data of TSS, particulate and total PCBs in the Lower Passaic River along with flow information between 1998 and 2002. For hydrophobic contaminants, such as PCBs, low TSS levels in the water column are usually associated with low particulate chemicals on the basis that the main source of the chemical in water is associated with the re-suspension of solids. High flow events on the other hand usually result in more solids being resuspended in the water column and higher levels of chemicals in the water. Although, TSS levels are not always well correlated with flow events, the PCB levels in the water column follow the TSS pattern reasonably well. More PCBs are measured in the water column when TSS levels are high. In addition, since most of the PCBs in the water column seem to be in the particulate form (i.e., particulate – open triangle - and total concentrations – closed circles - are almost identical) (Figure 1-21), this data suggest erosion and re-suspension are contributing to the load of chemicals in the water column. This plot does however show that a better temporal and spatial characterization of critical processes, such as erosion/resuspension and deposition is needed. Several yearly flow events that account for inter-tidal variability need to be captured to better characterize the physical (erosion/resuspension) and chemical (i.e., partitioning) processes taking place in the system. The same observations apply to all other COPCs: scarcity of data in the water column and limited temporal and spatial characterization of the contaminant distribution.

One interesting aspect of the Passaic River is the high productivity of its water, particularly during the early spring and summer. Levels of Chlorophyll-a, an indicator of algal biomass in the Passaic River, sometimes exceed 100 μg/L (Figure 1-22). There is a marked seasonality that suggests that algal production is important in the Passaic River. Similarly, POC measurements in the Passaic have been observed to exceed 10 mg/L, whereas DOC concentrations typically range between 4 and 6 mg/L. Because organic matter is an important component of the suspended sediment and because organic carbon concentrations are greatly influenced by nutrient cycles in the harbor and its adjoining waters, the interactions of inorganic and organic solids (e.g., through coagulation) need to be explicitly addressed in the Passaic River system. One way to address the interactions is to build the sediment transport calculations directly in the water quality model. The full sediment transport-organic carbon cycle calculation (ST-SWEM) will be described in Section 4.

1.5 LESSONS LEARNED

There are a number of observations that emerge from tasks performed prior to the development of the modeling plan. These tasks included data analysis – some of which is described above and will be discussed further in the next sections - as well as some preliminary mass balance calculations to i) establish a mass balance of the solids entering the Passaic River from the Dundee Dam, depositing in the river bottom, re-suspending and leaving the river, ii) help identify the major physical, chemical, and biological processes occurring in the Passaic River, and iii) determine the major inventories and fluxes of selected COPCs. These tasks are helpful in developing a conceptual site model (CSM) that guides the choice of the appropriate modeling framework, and in designing a field program that supports the implementation of the selected models. The first mass balance analysis will use a macro-scale solids approach based on the inputs of solids through Dundee Dam, tributaries, CSOs, storm water outfalls and the confluence with Hackensack and Newark Bay. This mass balance of solids will also analyze bathymetric changes and examine depth profiles of chemicals and radionucleides in dated-sediment cores collected over the years. Sedimentation rates estimated using bathymetric data would be reconciled with rates obtained from chemicals and radionuclide sediment profiles. The results of these simpler sets of mass balance calculations will be incorporated in the CSM for future iterations.

A number of lessons learned from the initial mass balance calculations are presented here.

Mass balance calculations were conducted for six (6) individual COPCs representing a range of hydrophobic chemicals, namely:

- 3,3',4,4'-tetrachlorobiphenyl (PCB77), a co-planar, dioxin-like tetrachlorobiphenyl, representative of the lighter molecular weight;
- 2,2',4,4',5,5'-hexachlorobiphenyl (PCB154), a hexchlorobiphenyl, representative of the heavier, more chlorinated PCBs;
- 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD), representative of the lighter molecular weight, less chlorinated dioxin congeners;
- Octachlorodibenzo-p-dioxin, OCDD, an octachlorodioxin with the maximum of eight chlorine substitutions. It is representative of the heavier molecular weight, more chlorinated dioxins.
- Pyrene, a four-ring compound, representative of lighter molecular weight PAHs.
- Benzo[a]pyrene, BAP a five-ring PAH compound, slightly heavier than Pyrene.

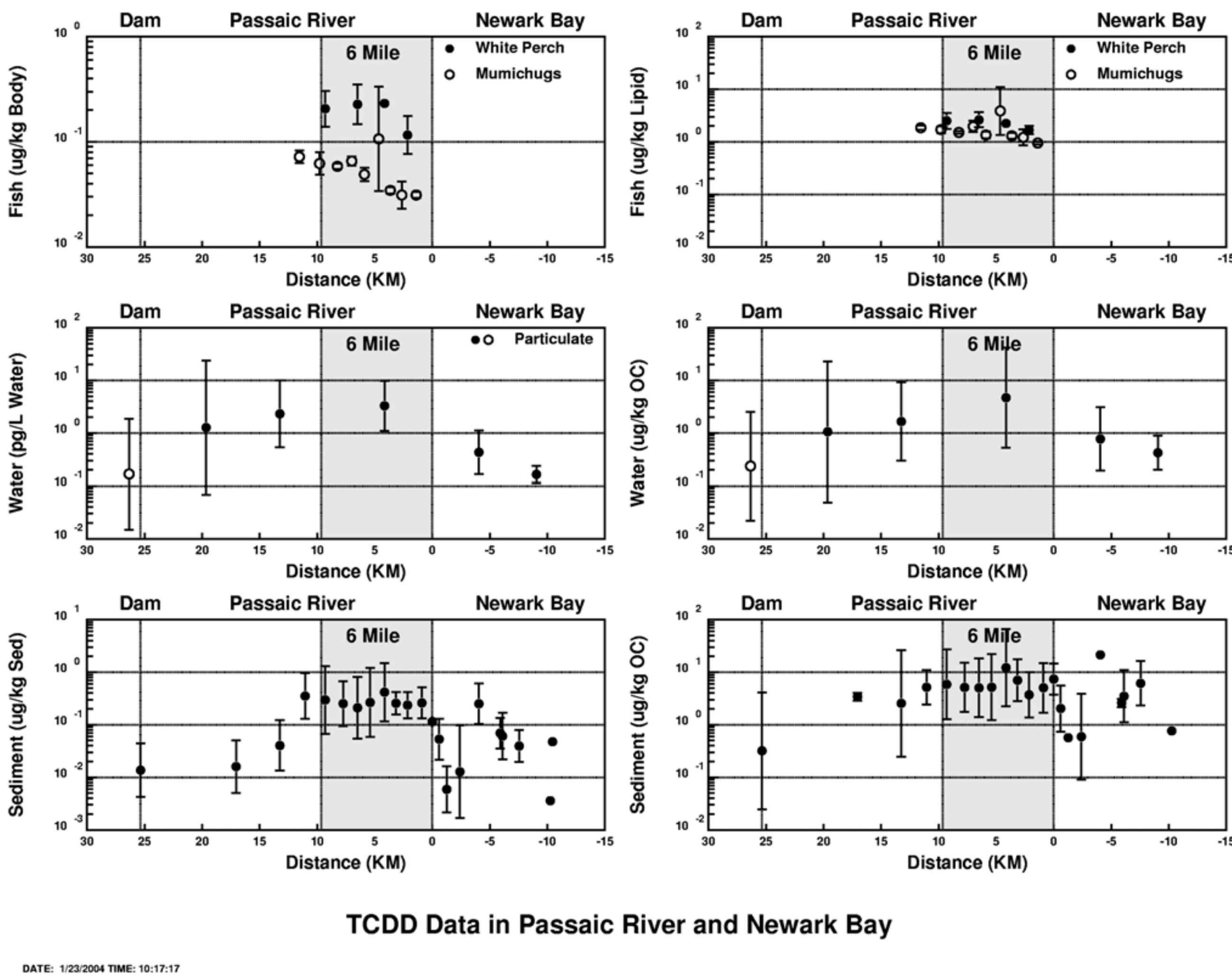


Figure 1-20. TCDD data in the Passaic River and Newark Bay

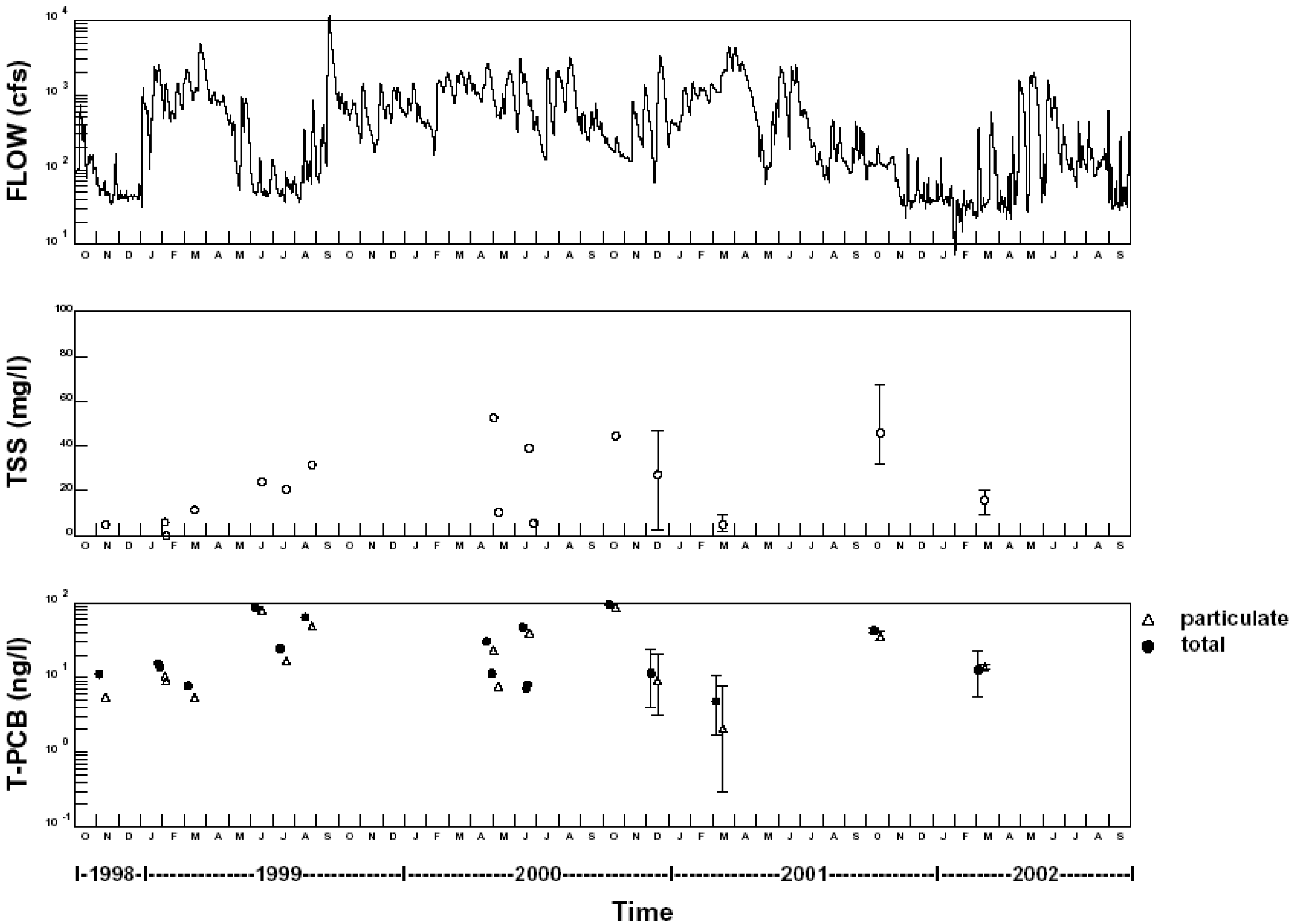


Figure 1-21. . Temporal variation of flow, TSS and Total PCBs in the Passaic River.

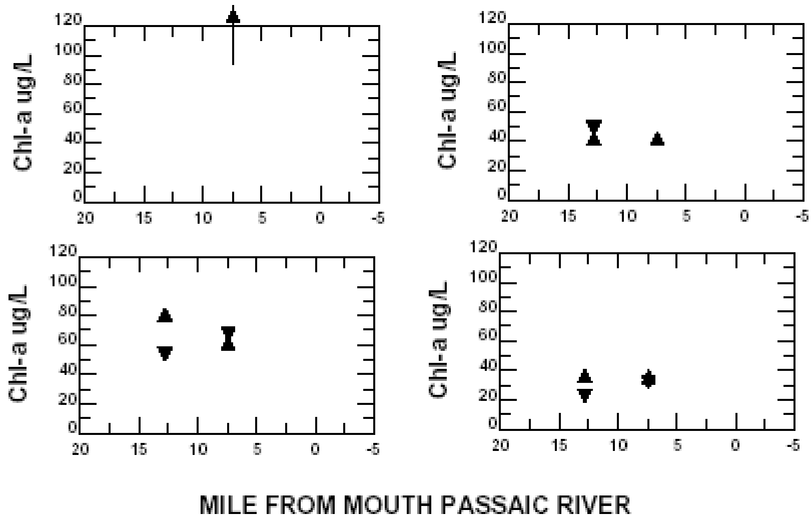


Figure 1-22. Spatial distribution of measured levels of chlorophyll a in the Passaic River during the months of August 1995 (upper left panel), September 1995 (lower left panel), March-April 1995 (upper right panel) and November 1994 (lower right panel) (Data from Battelle – CARP database, 2005).

The general modeling strategy was to perform steady-state mass balance simulations for present-day conditions. The computational framework for this analysis was based on the hydrodynamic component of the System-Wide Eutrophication Model (SWEM) developed for the New York City Department of Environmental Protection. The SWEM hydrodynamic model was used to transport chemicals within the Lower Passaic River study domain as well as other regions of the New York/New Jersey Harbor system. The chemicals were treated as conservative, which means that loss processes other than water column transport out of the study area (e.g. volatilization) were neglected. However, all major sources including release from the sediment bed by porewater diffusion and particle exchange (net of deposition and resuspension), point sources including CSOs, SWOs and WWTPs, and tributary inflows were included. Simplified approaches were adopted to estimate the porewater diffusion coefficient at 5 cm d^{-1} - consistent with the modeling work in the Hudson River Estuary by Farley et al, 1999 and to calculate the flux of chemical from the sediment bed by particle exchange based on the chemical concentration in the bed and an effective sediment flux velocity of 0.5 cm $year^{-1}$. Each of the source categories was simulated separately in the model, so that the contribution of each category to water column concentrations throughout the model domain could be visualized. The model computed water column concentrations from all source categories were added and compared to data.

These simulations were useful for a number of reasons. First, they illustrated the contribution of each source category to water column concentrations throughout the model domain. This helped identify the most important source categories, and was used to direct planning and data collection for this phase of the modeling. Second, when there were significant discrepancies between the modeled and measured concentrations, the simulations pointed to unknown or misrepresented sources or possibly questionable data.

Although the model was not in full agreement with the data in the Hackensack and Kill Van Kull, overall the modeled water column concentrations in the Passaic River were remarkably close to the data for the PCBs and dioxins. For PAHs, the model generally over predicted concentrations throughout the model domain. It is unclear whether the discrepancies are related to data quality, lack of adequate calibration or other processes that the model fails to account for. It is expected that the more elaborate modeling framework that will be used is likely to address these issues, notwithstanding that the mass balance calculations were preliminary in nature.

The preliminary mass balance analysis also showed that in the six-mile reach the relative contribution of source category varies significantly among the chemicals. For the PCBs, sediment flux was the most significant source category, while the remainder was split relatively evenly between point sources and tributaries. TCDD sediment flux was also responsible for most of the water column concentration, and a small amount came from tributaries. However, the source contribution for OCDD was significantly different from that of TCDD. The majority of OCDD came from tributaries and point sources while sediment flux contributed about 40%. These results should be interpreted very cautiously because of the simplifying assumptions used to run the model. As such, the absolute values of each source contribution are less important than the fact that reaches and tributaries outside the Passaic River *per se* contribute to the loads of contaminants, and as a consequence the model domain should be extended to include Newark Bay, the Hackensack and the Kulls.

The findings from the *preliminary mass balance calculations* and *site-specific data analysis* point to a number of observations that are pertinent to the development of the conceptual site model (CSM):

- The cross sectional features of the Passaic would require a lateral resolution in grid to resolve the main-channel and riverbank geometries and the physics of the rivers. Resolution of hydrodynamic structure in lateral grid resolution is important because it plays a key role in determining the ultimate transport of sediment and sediment-bound contaminants. To model such a hydrodynamic feature would require a three-dimensional resolution.
- There are recent indications that persistent winds of longer than one day from the east or west can cause flushing events that may disrupt the regular patterns of circulation in Newark Bay (Pence, 2004). The effects of wind waves on bottom shear stresses are likely to be important, particularly for the shallow areas of Newark Bay, are should be considered in hydrodynamic studies.
- Because water storage in the marshland during tidal cycling and after storm events are important processes that affect hydrodynamic transport through much of the Hackensack River and ultimately to the Passaic River study area, the processes of wetting and drying need to be explicitly considered in hydrodynamic model calculations. In addition, the additional drag due to marsh vegetation will need to be considered.
- The variability of the water column data are significant, with ranges of one order of magnitude not uncommon. To explain this variability will require time-variable modeling. This will require further and better characterization of the solids and pollutant loadings to the system.
- Tidal energies may be sufficient to cause resuspension and re-deposition of sediment over the tidal cycle. In addition, because contaminants seem to be mobilized with the suspended loads in the water column, erosion and re-suspension from the sediment bed seem to be occurring. As a result these important processes need to be considered in the modeling framework. A sediment transport model that can adequately characterize the short-term (i.e. tidal deposition and resuspension) and long-term (i.e. net sediment accumulation) temporal patterns, and fine-scale (bank vs. channel) and large-scale (i.e. turbidity maximum) features of the system should be developed.
- The mass balance analysis showed that chemical flux from the sediment bed is a significant contributor to water column concentrations. The characterization of this source is limited by the data available to define spatial gradients in chemical concentrations. Therefore,

sampling to support further modeling should include high-resolution sediment sampling throughout the model domain.

- Major sources of sediment to Passaic River section are suspended sediment inputs from above Dundee Dam since most of the freshwater originates from the upper portion of the Passaic River (above the USGS gauging Station of Little Falls) across the Dundee Dam.
- Although the available data on grain size distribution in the Passaic River sediment show the dominance of cohesive particles (<63 um), non-cohesive particles are nonetheless present in the system and could affect erosion rates as the smaller particles erode and a surface-armoring layer is left in place. As a consequence, to better represent the behavior of both cohesive and non-cohesive particles, the sediment transport model must account for at least two grain-size classes (further discussion is provided in the sediment transport modeling section of this report).
- Because of the elevated levels of contaminants with depth, the Passaic River seems to be an accumulation zone for sediments. Historical bathymetric analysis suggests that deposition rates vary between 1 and 5 inches/year depending on locations.
- Because hydrophobic organic compounds sorb to organic carbon, which is greatly influenced by nutrient cycles in the harbor and its adjoining waters, eutrophication processes need to be explicitly considered. In addition, because of the interactions or organic and inorganic solids (e.g., through coagulation), the sediment transport model will also have to explicitly consider these interactions.
- There is a potential of groundwater migration of contaminants. Although there is limited information regarding the flux of contaminants, chemical loading from groundwater might need to be considered on the basis of available or new data. An assessment of the groundwater contribution (via sampling) is planned as part of the CSM iterative development; initially an estimate of groundwater discharge and water balance will be based on base flow separation conducted upstream of Dundee Dam.

1.6 CONCEPTUAL SITE MODEL (CSM)

A CSM includes the relevant hydrodynamic, sediment and contaminant transport, fate, and biotic processes that are significant within the study area. Conceptual models are usually based on fundamental scientific principles and processes and on an in-depth analysis of available site-specific data. The discussions presented in this (above) and next sections identify those physical, chemical and biological processes that need to be considered in developing the modeling framework required to answer the goals and objectives of the Lower Passaic River Restoration Project. The CSM is, therefore, based on the main findings from the data analysis and the knowledge acquired by HydroQual from the implementation of the Contaminant Assessment Reduction Program (CARP) of the New York - New Jersey Harbor Estuary Program (HEP). Detailed analysis of the CSM components are provided in separate sections of this report.

The model framework for the Lower Passaic River Restoration Project includes model components for hydrodynamics, sediment transport and organic carbon cycling, toxic fate and transport, and bioaccumulation as shown in Figure 1-23. The model will be run with a fine grid resolution (described in Section 2) to capture spatial detail of the transport, fate and bioaccumulation processes within the project domain. For computational efficiency, the overall modeling calculations will be decoupled and performed in four successive model calculations as described below.

Hydrodynamic model calculations will first be performed to determine intra-tidal transport and bottom shear stresses throughout the model domain. This information will be passed forward to a sediment transport/organic carbon cycling model to determine the movement of inorganic particles and organic carbon between the overlying water and the bed. In the case of sediment transport we know from an evaluation of field data that changes in channel morphology have occurred. This will need to be accounted for in the hydrodynamic model by permitting feedback from the sediment transport model to the hydrodynamic model (it is envisioned that this will be performed in the simulation on a yearly basis). Information from the hydrodynamic and sediment transport/organic carbon cycling models will be passed forward to a chemical fate and transport model, and will be used along with descriptions of contaminant partitioning to organic carbon and other contaminant processes (e.g., volatilization, degradation, etc.) to determine contaminant concentrations in the overlying water and sediment. Finally, contaminant concentrations in the water column and sediment will be used in bioaccumulation and toxicity calculations.

Operationally, the hydrodynamic model would be run for a year. The hydrodynamic outputs would be passed to the sediment transport/organic carbon production model. The sediment transport/organic carbon production model would be run for the same year. Changes in bathymetry calculated by the sediment transport/organic carbon production model at the end of the year due to deposition and erosion would be passed to the hydrodynamic model for the next year of simulation. This procedure would be repeated multiple times to include each year of simulation. It is noted that in addition to changes in bathymetry calculated by the sediment transport/organic carbon production model, bathymetry changes associated with Harbor-deepening related dredging would also be incorporated into the hydrodynamic model at annual intervals.

In years with big events (e.g., 1984), it may be necessary to update the bathymetry more frequently than once per year. Depending on the change in bed elevation due to erosion or deposition, then the model would be stopped and the bathymetry would be updated.

The specific models that will be used are shown in Figure 1-24 and are discussed along with additional rationale for their selection in the following sections. A summary of processes included in the various models is outlined in Table 1-5. Model descriptions for these processes will be periodically reviewed throughout the project to ensure that the most up-to-date descriptions of the processes are included. Model calibration and skill assessment for the hydrodynamic and sediment transport/organic carbon cycling models will be performed for water years (October-September)

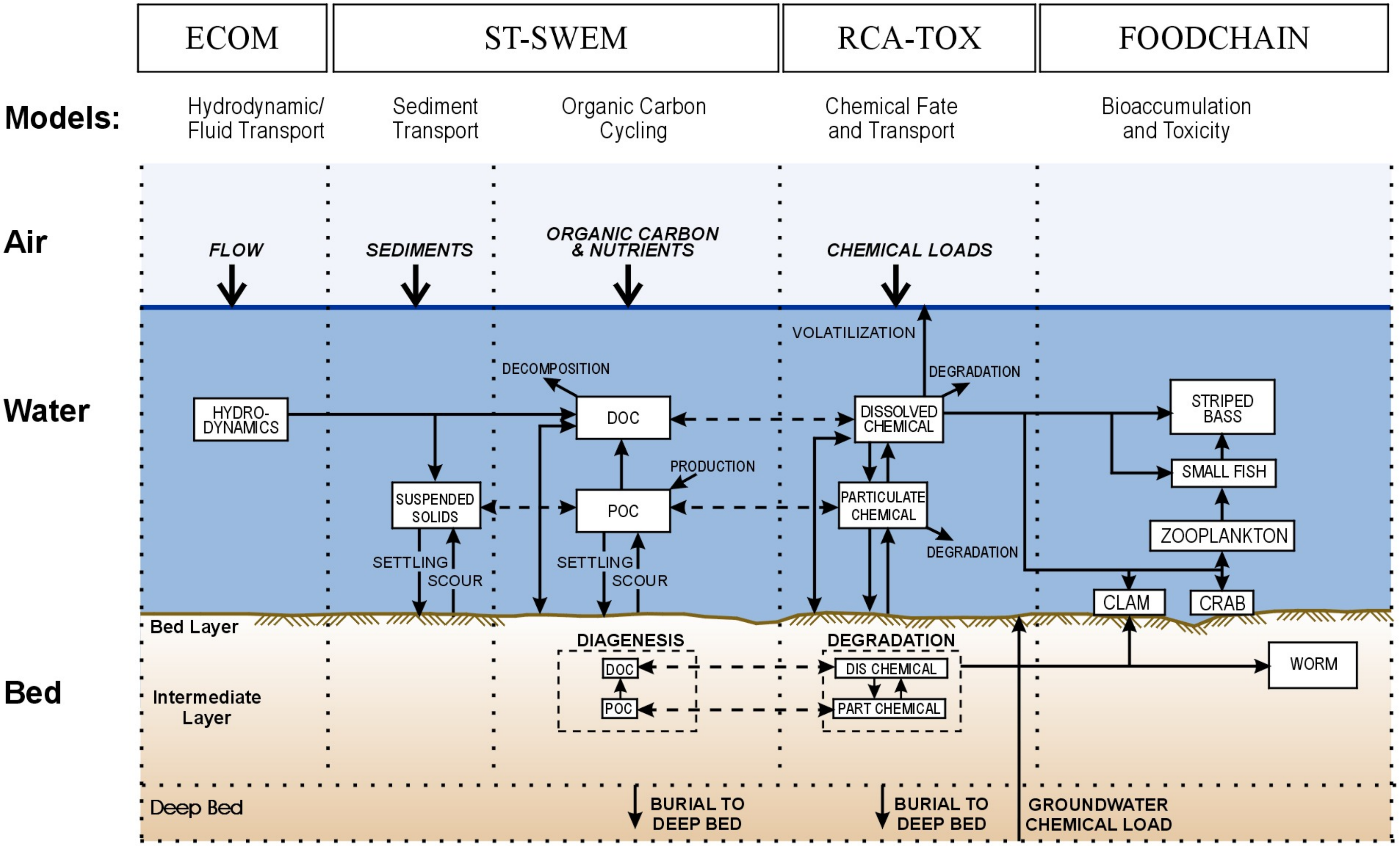


Figure 1-23. Schematic of the Modeling Framework Processes and their Interaction

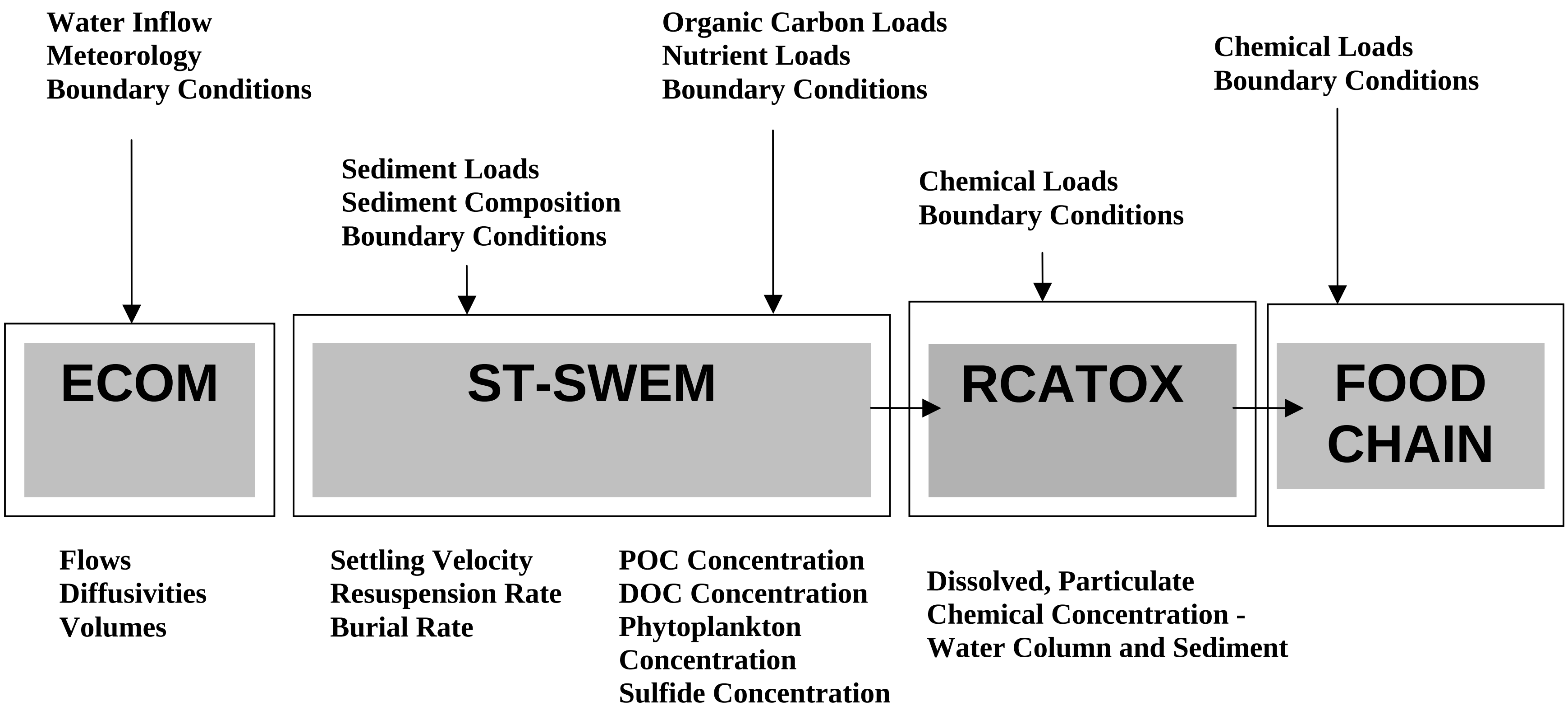


Figure 1-24. Modeling Framework: Model Sequence, Input and Output.

Table 1-5. Modeling Framework Processes

Models	**Processes**
Hydrodynamics	Water surface level and currents
	Thermal balance/heat transfer
	Density (salinity/temperature) driven flow
	Flow resistance
	Wind waves
	Bottom shear stress
Organic Carbon Production and Sediment Transport	Tributary, STP, CSO, SWO, and landside loadings
	Coagulation and settling
	Deposition/burial of solids and carbon
	Solids and carbon resuspension
	Bioturbation
	Productivity and respiration Sediment diagenesis and nutrient recycle
Toxics Fate and Transport	Tributary, STP, CSO, SWO and landside loadings Atmospheric loading
	Sorption/desorption, particulate chemical resuspension, settling and burial
	Porewater diffusion
	Volatilization
	Chemical transformations
Bioaccumulation	Gill transfer
	Dietary uptake/trophic transfer
	Assimilation
	Elimination
	Growth and Migration

described in the hydrodynamic section of this report. Chemical fate and bioaccumulation model calibration for the contaminants of concern will be performed for recent conditions (1995-2006). Based on the availability of information on historical contaminant loads, a time-variable model calculations may also be performed as a model hindcast for select contaminants to ensure that time constants in the model are properly calibrated. It is currently anticipated that a long-term hindcast computation will be performed as part of the sediment transport model calibration. Based on these evaluations, an overall assessment of the model will be conducted, and component load analyses and model projections under various scenarios will be performed. Details of model calibration, assessment, load analyses and projections are given later in the report.

1.7 SCHEDULE

The scheduling of the modeling task is given as a PDF file in Appendix A. This schedule is a working draft that will be modified as the project moves forward, based on stakeholder input.

1.8 CARP MODELING FRAMEWORK

Section 1.6 described the CSM envisioned for use on the Lower Passaic River Restoration Project. The CSM included models for hydrodynamics, sediment transport, carbon production, chemical fate and transport, and bioaccumulation. While there are a number of individual computer codes (i.e., hydrodynamics versus sediment transport versus fate and transport) available with which to construct an integrated modeling package for use on the Passaic River, there is not a single computer code that contains all of the necessary components identified in the CSM. However, HydroQual is currently in the process of completing development of an integrated modeling system for the New York - New Jersey HEP CARP study. Rather than develop a brand new modeling system for the Passaic River project, HydroQual proposes to utilize the CARP modeling system as the starting platform for construction of the Lower Passaic River model. While it is recognized that additional refinements (i.e., improved grid resolution in the Lower Passaic River, Hackensack River, and Newark Bay, Meadowlands wetlands system, possible changes to sediment transport formulations, etc.) may be necessary for the Passaic River system, HydroQual believes that adopting the CARP framework as the initial platform will benefit the USEPA and USACE from both a cost and time perspective by reducing labor efforts and timelines needed: to develop and verify linkages between the various component models and training/familiarization required to run the component models if other computer codes were to be chosen; to develop pollutant loadings estimates; to develop a calibration/validation of the carbon-production model, etc.

Another issue that was considered in developing this work plan and the recommendation to utilize the computer codes employed in the CARP modeling framework, is the need to provide boundary conditions for all of the component models. One way of constructing the Lower Passaic River model is to have a high-resolution grid that just encompasses the Lower Passaic River, the Hackensack River, Newark Bay, and the Arthur and Kill van Kills. If this option is chosen it will be necessary to provide boundary conditions for each of the component models. Of course, this information could be provided from the CARP model. However, one must develop a degree of confidence that one has selected the proper computational domain, so that the boundary locations are located far enough away from the influence of internal loadings and processes. Another approach that could be used in constructing the Passaic River model is to start with the SWEM/CARP computational grid, but to refine the grid in the area of interest. Proceeding this way would eliminate the concern over the boundary condition issue, since the SWEM/CARP boundaries are so far away from the area of interest. However, a potential concern for following this approach is the potential computational burden required to run the model. The time required to run the model may be so onerous that it becomes difficult to calibrate the model or infeasible to perform long term projection runs. HydroQual has recently developed an approach that may solve this latter problem. Essentially in this approach, the RCA water quality modeling code, which is the basis for the carbon-production model and the fate and transport model, has been modified so that (1) water cells in various parts of the domain can be turned off and on via a simple change in model input, and (2) "boundaries" for the resulting grid are obtained from a previous run. In other words, one must perform a model run that encompasses the entire model domain, saving computed concentrations at key model cells that will represent the boundaries of the sub-domain in the subsequent model run. In the second and subsequent runs, only a subset of the entire domain will be executed by "turning-off" undesired water cells, i.e., performing model computations only in the water cells contained in the area of interest. During the execution of the sub-domain, boundary conditions are obtained as appropriate from a data file generated during the initial model run. While the initial model run may be computationally intensive, subsequent runs will be completed in a more reasonable time, since only a portion of the model grid is being executed. If after a number of calibration runs are performed, it is found necessary to update or modify the boundary conditions of the sub-domain, the entire model domain can be re-run and a new "boundary condition" file can be generated. We believe that this is a reasonable approach for the Passaic River project, which provides a reasonable trade-off between the boundary condition issue and the long run time issue. We also believe that the RCA computer code is the only code that provides for this approach.

1.9 APPROACH TO UNCERTAINTY ANALYSIS

An analysis will be performed to assess the uncertainty of model predictions given a characterization of the uncertainty of model input parameter values. While a variety of ways have been proposed for conducting uncertainty analyses for fate and transport models (e.g., Monte Carlo analysis, probabilistic modeling, response surface models), no single approach has yet been identified that is generally accepted for use by the scientific and regulatory communities. In selecting an approach to be used, it is necessary to consider the computational requirements of the model, as this can place a practical constraint on the viability of the alternative approaches that are available for use. This consideration is particularly important in the case of this modeling effort, where the computational burden is likely to be substantial and the actual time that is required to complete a multi-year simulation is expected to be long (time scale of days to weeks). As an example, a Monte Carlo approach is commonly used for uncertainty analyses. However, this approach requires that a large number of model simulations be completed. This may not be feasible, given the number and duration (real time) of the fate and transport model simulations that would need to be performed. An alternative method that is more likely to be viable is to complete a limited number of model simulations and to use these results to develop frequency distributions of model outputs. These distributions provide a characterization of the uncertainty in output due to uncertainty in the inputs, but for a relatively small number of simulations. The distribution-free Kolmogorov-Smirnov (KS) confidence limits of the empirical cumulative distributions of the model output (i.e., the exposure levels) are then evaluated (see USACE and USEPA, 2006). These confidence limits are analogous to the confidence limits about a single point estimate, but in this instance the KS limits provide bounds for the overall statistical distribution rather than for a single point (Ferson et al., 2005). The KS confidence limits of these frequency distributions are then used to characterize the exposure levels that are input to the Monte Carlo analysis that is performed with food chain model. The food chain model, which runs relatively rapidly in comparison to the fate and transport model, is much more amenable for use with Monte Carlo techniques.

Another approach that could be explored is to use a response surface model (RSM) (USACE and USEPA, 2006). This approach makes use of a limited number of sets of model output that were obtained by the perturbaion of the values of key model input parameters. The perturbations of the 'n' model inputs are made within prescribed limits that are defined on the basis of what is understood to be the uncertainty of the values of these inputs. A multi-dimensional representation (i.e., a simplified regression) of the numerical model results, in the form of a linear function (or non-linear function, requiring additional runs) of the model input parameters, the n-dimensional RSM, is then developed. This n-space representation of the model output may then be used to rapidly synthesize approximations of the model output that would be obtained if a large number of model simulations had actually been performed. The results are then treated in the same way as are results from a Monte Carlo analysis. Selection of the general approach that will be adopted for use in the uncertainty analysis must necessarily await the results of ongoing analyses, including more specific information on the actual model run times.

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SECTION 2

HYDRODYNAMIC MODEL

2.1 INTRODUCTION

The Passaic River along with the Hackensack River and Newark Bay is one of the most complex estuarine systems in the United States. The system is connected to two tidal straits, named Kill van Kull and Arthur Kill. These straits connect Newark Bay and the Passaic and Hackensack Rivers with Upper New York Bay and Raritan Bay, through which tides, originating in the Atlantic Ocean, enter the system (Figure 2-1). The bathymetry of the Passaic-Hackensack-Newark Bay system is characterized by deep shipping channels along the center of both the Arthur Kill and Kill van Kull, as well as the west side of Newark Bay through the center of both Lower Passaic and Hackensack Rivers, with shallower side banks. The U.S. Army Corps of Engineers (USACE) maintains the navigability of the channels in order to support the New York-New Jersey Port operations. The ship channels, maintained by the USACE to facilitate the movement of container ships in and out of the Newark Bay, added additional complexity to the dynamics of the system. The ship channels are relatively deep (13m-15m) with respect to the near-shore depths, causing a significant variability in depths across the channels. Figure 2-2 shows the cross sections of different parts of the system. It illustrates the relatively deep shipping channels in the rivers and the Newark Bay. The average depth of the shipping channel in the Arthur Kill is about 11 meters MSL, while the average shipping channel depth in the Kill van Kull and Newark Bay are 13 meters MSL. These channels play an important role in transporting saline water from the ocean in to the system.

The hydrodynamics of the Passaic-Hackensack-Newark Bay system is predominantly controlled by three forcing mechanisms, freshwater flows (buoyancy sources), tides and winds. Two major sources of freshwater inflows, the Passaic and Hackensack Rivers, contribute to the salinity gradients in the system. By far, the largest freshwater contribution is from the Passaic River. Figure 2-3 illustrates 21 years of flows measured at Little Falls on the Passaic River and at the Oradell Dam on the Hackensack River. The long-term daily average flows measured at Dundee Dam are about 29 m^3/sec (1,000 cfs) and the maximum flows during this 21-year period were approximately 500 m^3/sec (18,000cfs) in April, 1984. In contrast the average flow in the Hackensack River is only 1.6 m^3/s (56 cfs) and a maximum measured flow of approximately 158 m^3/s (5,500 cfs) in September 1999 during Hurricane Floyd. The salinity dynamics in the system are mostly controlled by the freshwater flows from the Passaic and Hackensack Rivers and the saltier ocean waters that enter the system through Kill van Kull and Arthur Kill (Chant, 2002). Generally, the salinity front stays within the Lower Passaic and Hackensack Rivers but may be pushed into Newark Bay during extreme high flows. Salinity is, in general, higher during the time of low freshwater flow and is also

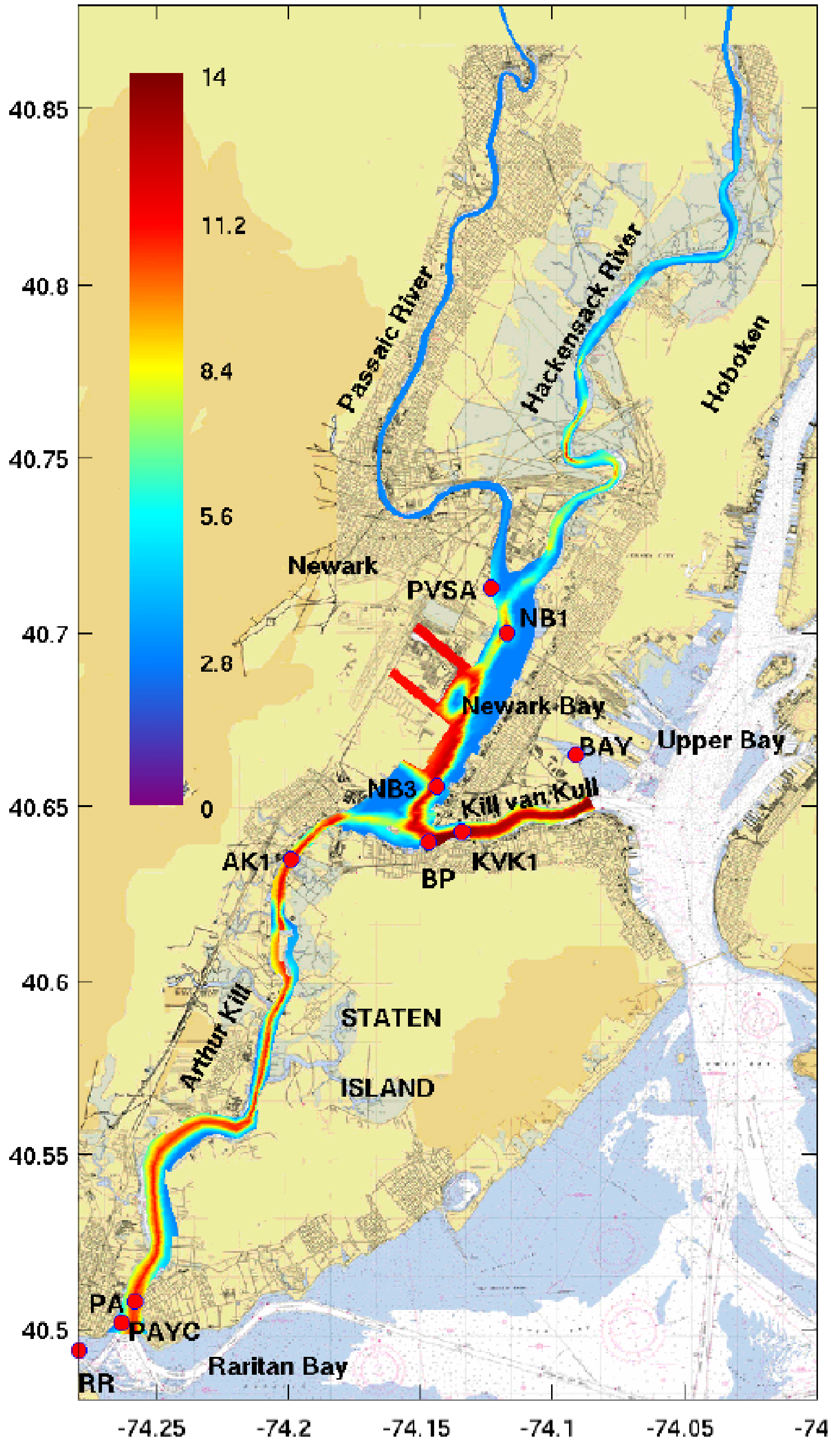

Figure 2-1. Bathymetry of study area (in meters). (Pence, 2004)

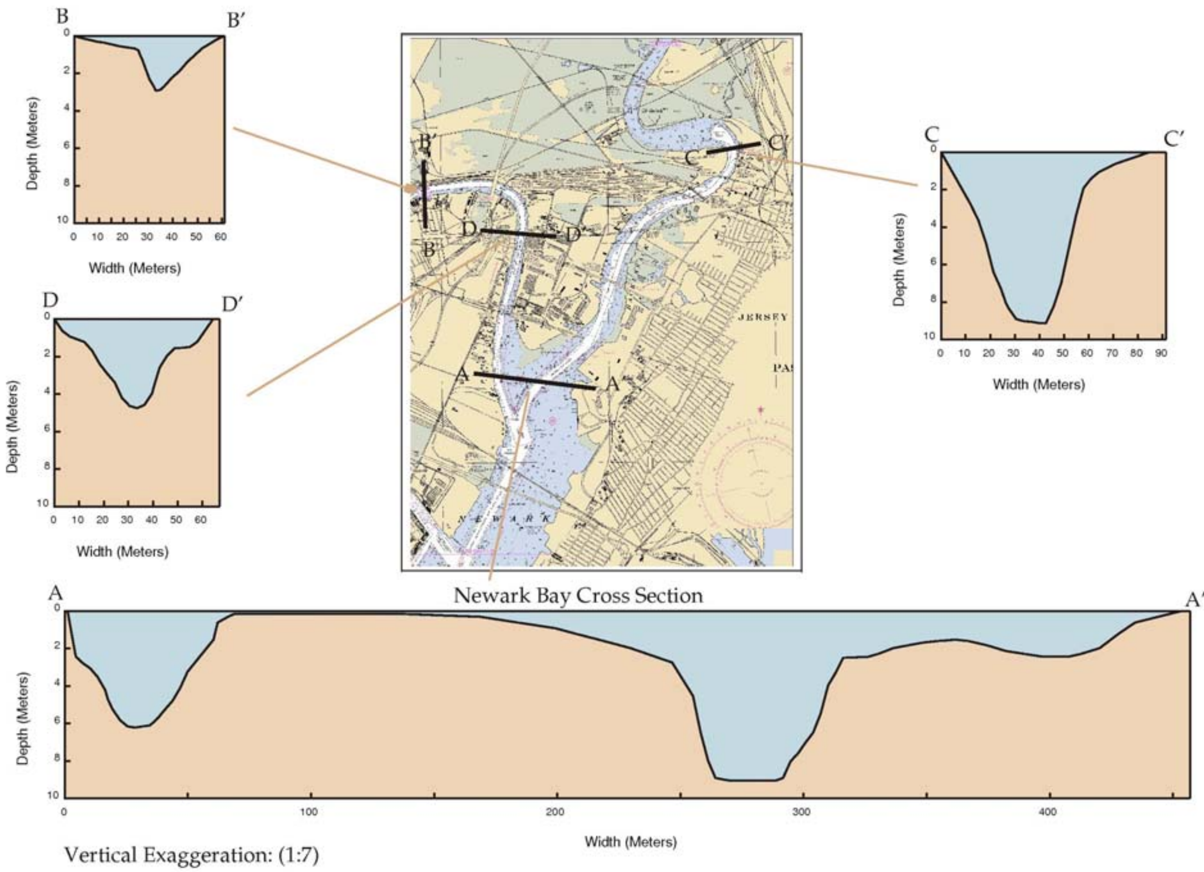

Figure 2-2. Cross-sectional view of bathymetry in Lower Passaic, Hackensack, and Newark Bay region

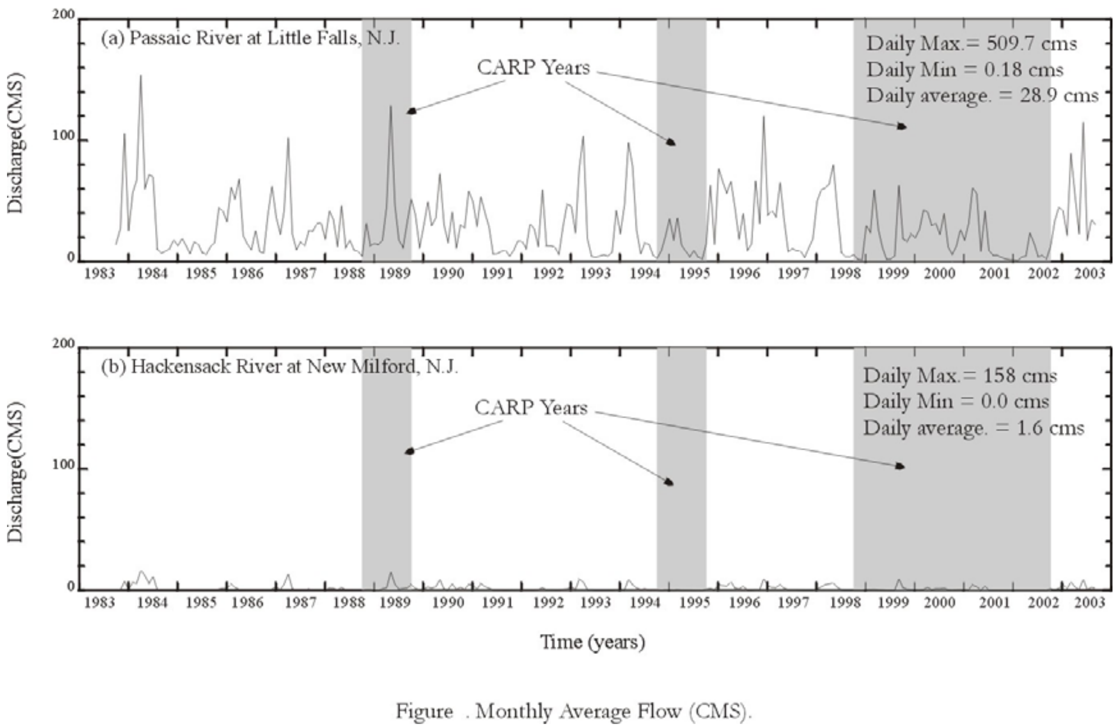

Figure 2-3. Monthly average flows of Passaic and Hackensack from 1983 through 2003

more uniform both vertically and horizontally throughout the system than during the time of high freshwater flow. Freshwater flows emanating from the Passaic River stay along the western edge of Newark Bay, creating a cross channel salinity gradients (Pence 2004). Deeper shipping channels in the system appear to act as conveyances of denser and saltier ocean water to upper Newark Bay and to the Lower Passaic and Hackensack Rivers.

Tidal influence has significant importance within the Passaic-Hackensack-Newark Bay estuarine system. A harmonic analysis of tidal elevation data measured at Bergen Point, which is at the entrance to the Newark Bay, suggests that the semi-diurnal constituents (M2 and S2) dominate the system. A spectral analysis of the tidal elevations also indicated that maximum variance occurred at an interval of approximately 12.4 hours, suggesting a dominant semi-diurnal tidal signal. The resultant tidal harmonic constituents are provided in Table 2-1. These constituents lead to a spring-neap tidal cycle with a period of approximately 13.5 days (Figure 2-4).

Table 2-1. Characteristics of Principal Tidal Constituents in Newark Bay

Constituents	Period (Hrs)	Amplitude (ft)	Phase (deg)
O1	25.82	0.175	107.11
K1	23.93	0.332	108.63
M2	12.42	2.391	233.70
S2	12.00	0.464	263.78
N2	12.66	0.523	220.40

Tidal currents in Newark Bay and in the Passaic and Hackensack Rivers are found to be moderate, with amplitude of 50 cm/sec. Most of the time, the surface and bottom tidal currents are of same magnitude and in phase. However, during high-flow periods the surface currents, directed towards the ocean (ebb currents), become much stronger than the bottom currents, indicating a presence of strong vertical shear (Pence 2004). Figure 2-4 illustrates surface and bottom currents during high flow season. During high freshwater flow, classical two-layer estuarine circulation is observed, with surface currents flowing seaward and bottom currents flowing upstream. The net flow along the side banks is downstream, with an increased magnitude under higher freshwater flow conditions.

Strong and persistent wind events in Newark Bay can have a strong effect on the circulation in the estuary, and in some extreme cases can disrupt the normal pattern of estuarine circulation. Modeling analysis (Pence 2004) suggests that strong winds from the west will flush water and water borne constituents from Newark Bay out through the Kill van Kull, with weaker flow in through the

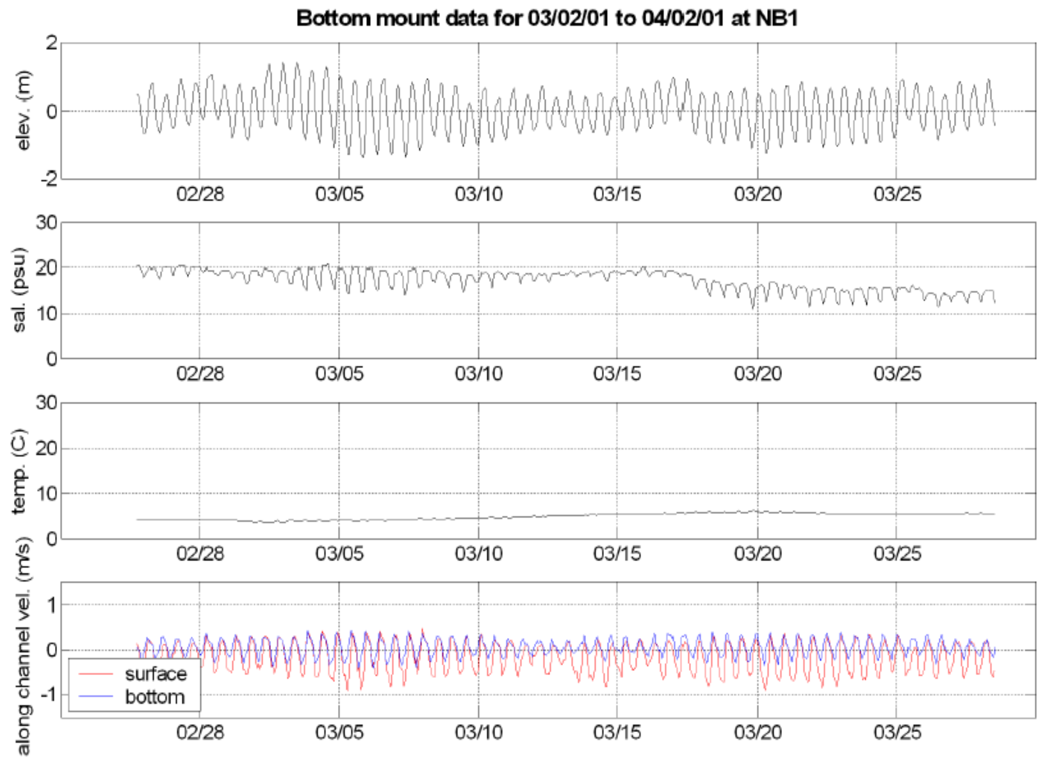

Figure 2-4. Surface elevation (top panel), salinity and temperature (middle panels), and surface and bottom currents measured at the head of Newark Bay from Feb-March, 2003 (Pence, 2004)

Arthur Kill. Model computations indicate that this flow pattern changes direction when strong winds blow from the east.

2.2 RATIONALE FOR A THREE-DIMENSIONAL MODELING FRAMEWORK

The purpose of hydrodynamic modeling is to develop a time-dependent, three-dimensional description of transport through the Passaic River study area, which includes Newark Bay and the Hackensack River. Modeling the hydrodynamics of the Passaic-Hackensack-Newark Bay system is essential to predict the movement of and concentrations of various chemicals of concern within the study area under different management scenarios (e.g., dredging, monitored natural attenuation, capping, etc.).

Previous hydrodynamic modeling studies of the Passaic River were performed as part of larger regional studies for eutrophication and toxic contamination for New York-New Jersey Harbor and adjoining waters. Previous modeling efforts, however, are not adequate in describing transport in the Passaic River study area. The grid resolution in the SWEM and CARP studies is not sufficient to describe bathymetric features (e.g., shipping channels versus tidal shoals) in the Passaic and Hackensack Rivers and Newark Bay sections of the model. However, some improvements were made to better represent the Passaic, Hackensack and Raritan Rivers in a subsequent modeling effort by HydroQual (2002). In this study cross-sectional areas and bathymetric representation of the New Jersey tributaries was refined and additional readjustment and reconfiguration of hydrodynamic calibration parameters were made. The hydrodynamic calibration parameters were adjusted to better parameterize small-scale physics not resolved by the initial SWEM grid especially in lateral direction.

However, it is important to note that the cross sectional features of the Passaic, the Hackensack Rivers and Newark Bay coupled with dredged ship channels, as shown in Figure 2-1 and Figure 2-2, requires additional lateral resolution in the computational grid to resolve the main-channel and river bank geometries and the physics of the rivers. Resolution of hydrodynamic structure in the lateral direction is important because it plays a key role in determining more realistic bottom shear stresses, which are important in the ultimate transport of sediment and sediment-bound contaminants.

Historical salinity data indicates that the salt can travel upstream about to 10 miles from the mouth of Passaic River (Figure 2-5) during low river inflows. However, in the Hackensack River, salt can penetrate about 15 miles from the river mouth (Figure 2-6).

Hydrodynamics in the Passaic River system are further complicated by the presence of large intertidal marshes on the Hackensack River. The lower section of the Hackensack River consists of vast area of tidal wetlands, the Meadowlands area. U.S. EPA's National Wetland Inventory identifies about 1,500 acres of the wetland area is submerged with average tidal condition. And it also identifies much of the same area can be flooded during extreme flood conditions. The wetting

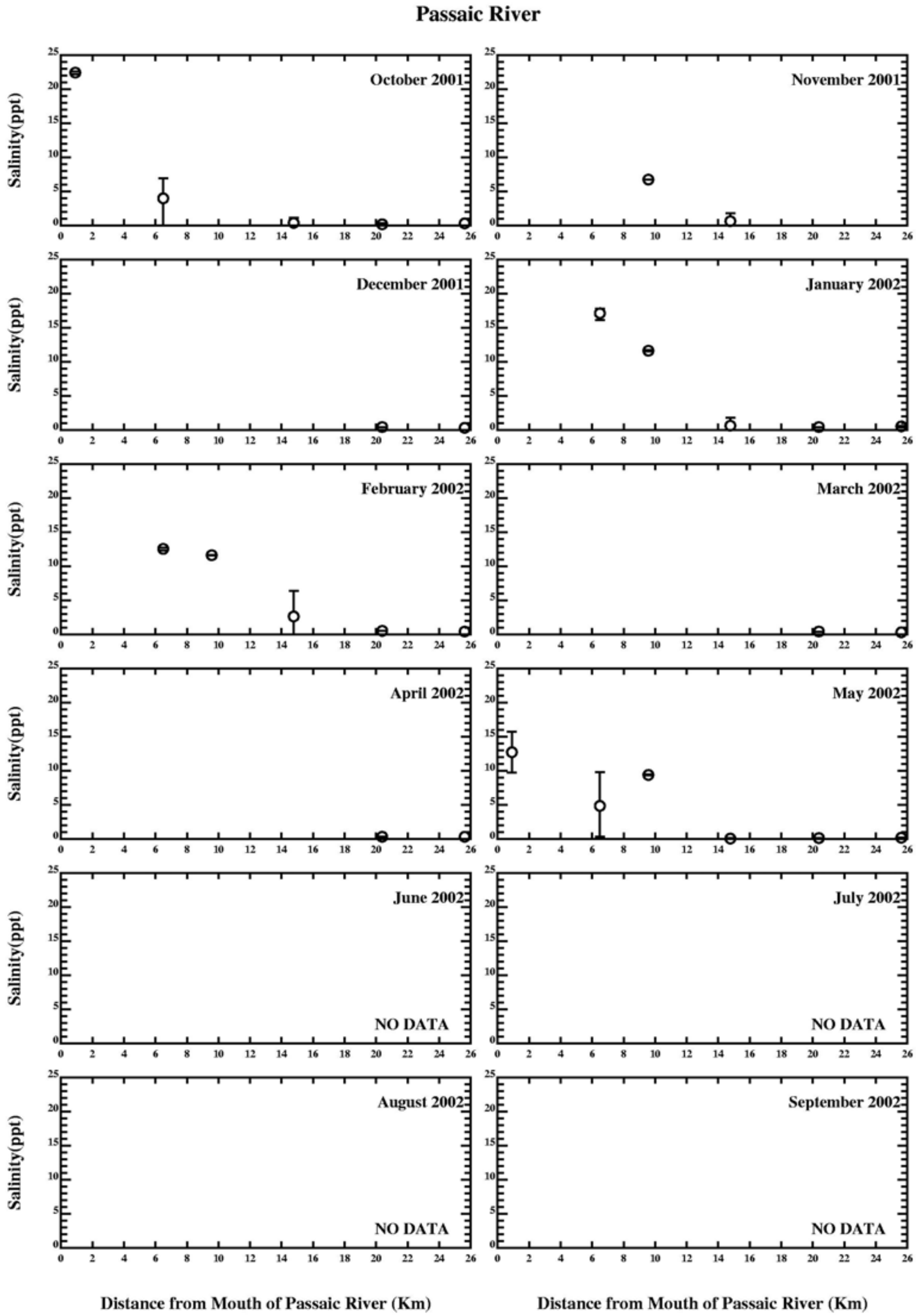

Figure 2-5. Longitudinal salinity distribution in Passaic River, Water Year 2002

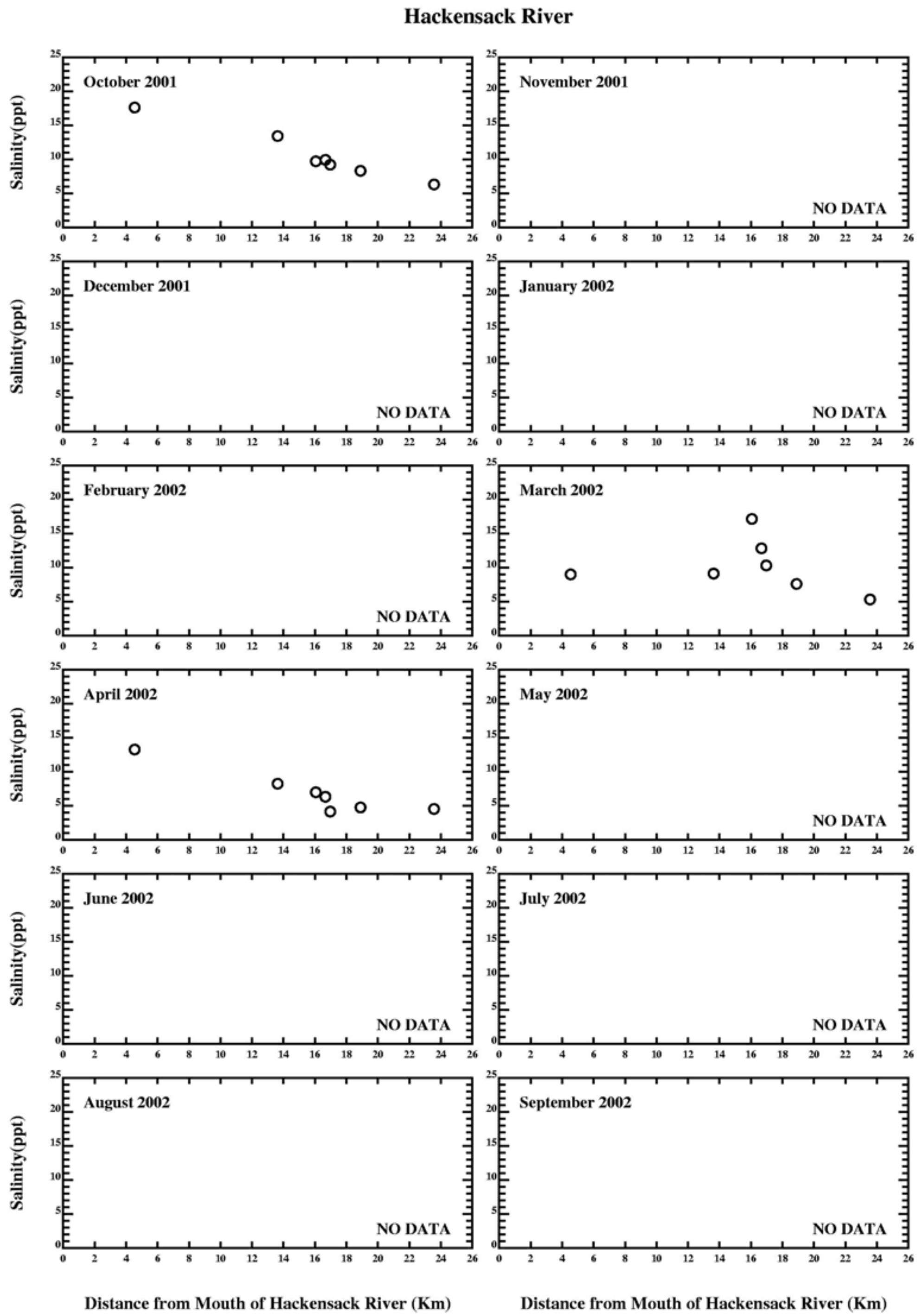

Figure 2-6. Longitudinal salinity distribution in Hackensack River, Water Year 2002

and drying of marshland in the Meadowlands was not included in the SWEM, CARP or New Jersey Tributaries Modeling evaluations. These marshes can provide significant water storage over a tidal cycle, and therefore, may alter the movement of water up the Hackensack and Passaic Rivers. Initial hydrodynamic calibration efforts of the SWEM/CARP modeled suffered by not accounting for the storage volumes represented by these intertidal marsh lands, i.e., salinity was not well reproduced in the Hackensack River and portions of the Passaic River. Representation of these physical areas and their wetting/drying needs to be explicitly considered in hydrodynamic model calculations.

In addition, wind waves and their effects on bottom shear were not incorporated in the SWEM and CARP hydrodynamic modeling calculations. However, the effects of wind waves on bottom shear stresses are likely to be important, particularly for the shallow areas of Newark Bay, and should be considered in hydrodynamic studies.

2.3 MODEL GRID

Complex estuarine systems with irregular coastlines and bathymetric features, such as the Passaic-Hackensack-Newark Bay system, often pose a significant challenge to modelers seeking solutions when resolution of microscale physics (order of meters to tens of meters) becomes dynamically important. For a credible scientific analysis, however, one must have a high-resolution representation of the model domain in order to resolve the coastline and bathymetry of the system, as well as other important physical, chemical and biological processes and their evolution within the system. The major challenge, however, comes from a computational perspective, even with the fastest and largest computers available to-date balancing desired spatial resolution with reasonable computational burden or "run-times" necessary to complete a model simulation. Thus, in order to provide an effective management tool, a balance must be struck between properly representing the system and its constituents while providing tractable solution times necessary to perform model calibration/validation, sensitivity analyses, and production runs.

The model domain will encompass the Passaic River, the Hackensack River, Newark Bay, their tributaries, and portions of the Arthur Kill and Kill van Kull as well as extending to include a portion of New York harbor and Raritan Bay as necessary to avoid boundary effects that will contaminate the model in the region of interest. The upstream extent in the Passaic River will be the Dundee dam, which also happens to be the limit of tidal influences within the river. The upstream extent of the Hackensack River will be the Oradell Dam. The model domain will also encompass the Hackensack River wetlands (the Meadowlands), which will be represented by model cells in the flood plain that wet and dry depending on the tidal elevation and the volume of flow within the Hackensack River free-flowing channel. Figure 2-7 illustrates a conceptual grid design of the wetting and drying tidal flats in the Meadowlands area.

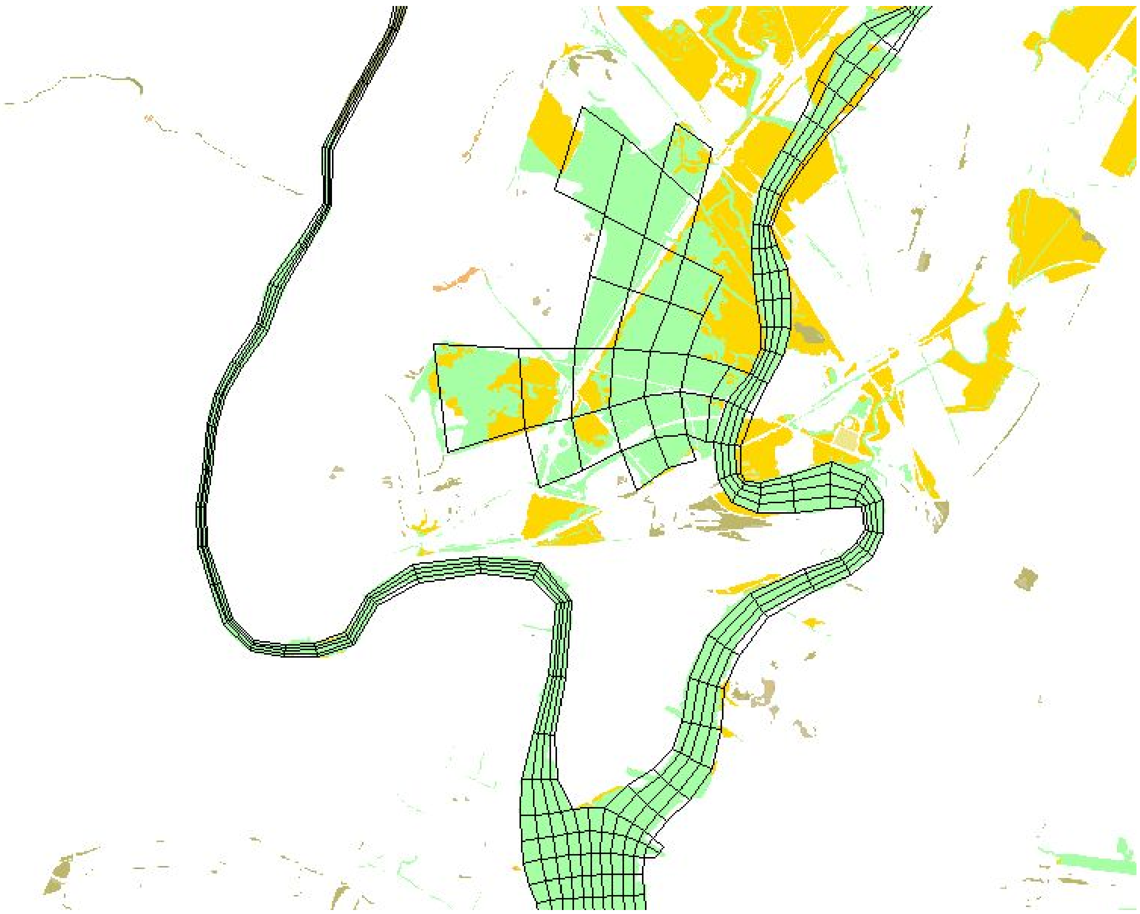

Figure 2-7. Conceptual design of tidal wetland of the Hackensack River

It is envisioned that the model grid will be designed with approximately three to four cells across the main channels of the Passaic and Hackensack Rivers and one cell across in the tributaries. The upstream extent of the model grid within tributaries will be decided based on the local flow conditions and availability of bathymetry. The upstream extent of the model grid will be the Dundee and Oradell dams on the Passaic and Hackensack Rivers, respectively. The model grid will allow for several cells across and along the length of Newark Bay, which will allow proper resolution of the approaches and dredged shipping channels and shallow areas within the bay. These high-resolution grid cells will be joined to existing SWEM/CARP model grid in Kill van Kull and Arthur Kill, in the east and south, respectively. Figure 2-7 depicts the proposed grid resolutions in the Passaic and Hackensack and Newark Bay system.

2.3.1 Horizontal Resolution

An orthogonal curvilinear grid will be designed to represent the horizontal computational grid system. This type of grid design allows for a variable level of horizontal resolution. For example, the grid can have smaller grid boxes, or high resolution, in areas where relatively high exchanges of contaminants are suspected or in regions of rapidly varying bathymetry, such as in and around the dredged channels. Less important areas, such as further out into the NY Harbor, can be represented with larger grid boxes, or less resolution. Additionally, a proper grid design strategy resulting in a more efficient model can decrease the necessary computer resources.

2.3.2 Vertical Resolution

For this study, we propose employing 10 vertical layers within the model domain. Blumberg, et al. (1999) and Warner, et al. (2005) showed that 10 layers were important to and sufficient for resolving the salinity and temperature stratification within the NY-NJ Harbor system. The sigma-level representation in the vertical has the additional advantage of resolving shallower areas with increased resolution compared to offshore, which is important because suspended sediment within the system will tend to accumulate in the nearshore areas during inter-event periods and be rapidly re-suspended from these areas during large events.

Currently, the planned development of a wetting/drying protocol is to enable the hydrodynamic model to properly account for the sponge-like effect that the Hackensack Meadowlands play in attenuating upstream water movement in the Hackensack River during flood tides. In the original development of the SWEM hydrodynamic model (used as the computational basis for the CARP model), the Hackensack Meadowlands were not included and as a consequence, the SWEM model was not able to fully reproduce the transport features of the mid- and upper-Hackensack River. Therefore, it is proposed for this study to include the Meadowlands areas in the model. However, HydroQual's current hydrodynamic model does not permit mixing of grids (for the open waters and tributaries) with 10 sigma layers with a grid (for the Meadowlands) that is vertically integrated. We believe that we can develop an approach that permits inclusion of the

Meadowlands, for the purposes of water storage, in the hydrodynamic model and still meet required stability requirements.

Currently there is no plan to address wetting/drying issues in the tributaries themselves, i.e., the possibility that the wetted perimeter extends into normally dry upland areas during flood events. Rather the assumption will be that all waters will be maintained within the confines of the main channels of the tributaries, even during flood events.

2.4 HYDRODYNAMIC MODELING FRAMEWORK

The hydrodynamic model will be based on HydroQual's in-house Estuarine, Coastal and Ocean Model (ECOM). The model simulates the spatial and temporal variation of water levels and currents, which advect and disperse contaminants through out the system, as well as the salinity and temperature fields as they vary with tide, wind, heating from solar and atmospheric radiation and freshwater inflows. ECOM will provide the capability of simulating events where water from the main channels can overtop the riverbank and flow into the floodplain, which is an important consideration, especially in wetlands areas of the Meadowlands adjacent to the Hackensack River. The model has been applied in a wide variety of domains from rivers and lakes to marine harbors and embayment and across wide coastal regions. ECOM is also a fundamental part of the System-wide Eutrophication Model (SWEM) for the greater New York Harbor, Long Island Sound, Hudson and East Rivers, and extending out to the New York Bight (Blumberg et al., 1999). The hydrodynamic model also incorporates a wave model (GLERL-WAVE) describing the effect of wind waves on the water surface and the wave effect on the bottom shear stress (Schwab, et al., 1984; Donelan, 1977). The latter will be an important consideration to the sediment transport model. A detailed description of ECOM in the form of a peer reviewed journal article (Blumberg et al., 1999) is provided in Appendix B.

The heat energy content in Passaic River is primarily governed by the surface heat exchanges. Measurements of heat fluxes are very difficult and costly to make and are often parameterized to obtain the fluxes, using the commonly available meteorological and atmospheric data. The processes that control the heat exchange between the water and atmosphere are well documented (Ahsan and Blumberg, 1999; Adams et al., 1981; Edinger et al. 1974). All of these works relied mostly on the bulk formulas to evaluate the components of the heat budget. Estimation of net heat fluxes requires a great deal of judgment in choosing the bulk formulas, which are dependent on many uncertain atmospheric parameters like cloud cover, humidity, and temperature. Four major heat flux components, such as short wave solar radiations, longwave atmospheric radiations, sensible heat, and latent heat fluxes have been incorporated in ECOM modeling framework. The formulations are largely based on the works of Ahsan and Blumberg (1999), Adams et al. (1981) and Cole and Buchak (1995). Appendix C provides a detailed description of the heat flux components incorporated in the ECOM framework.

Sediment heat flux could be an important process in shallow waters especially in wetlands and tidal flats. In shallow waters, incoming solar radiation often penetrates through the water column and heats bottom sediment. The flux of heat energy between sediment and water affects the distribution of water temperature in shallow waters. Sediment heat flux formulations of shallow water suggested by Tsay et al. (1992) will be incorporated in ECOM as necessary. After it is incorporated in ECOM, sensitivity tests will be conducted to determine if the formulation needs to be activated during model simulations.

2.5 ECOM MODEL INPUT

The hydrodynamic model requires a description of physical conditions over the region of interest. These include the bottom bathymetry within the rivers, the floodplains and out into Newark Bay and including the Kills. Bathymetric data will be used to guide the spatial resolution appropriate for the Passaic River model grid as discussed previously. The model will also use data describing the inflow of water via upstream boundaries, tributaries and overland flow as well as downstream near the open boundaries where tidal variation and the results of the larger CARP grid are to be applied. Finally, meteorological conditions, available from local area airports, will be assembled for driving the model.

In addition to initial conditions, the hydrodynamic transport model requires forcing that varies over time and space. Estuarine models typically require sources of freshwater resulting from inflowing rivers and stream as well as overland flow. Tides also affect the current, salt and temperature distribution within the system and these conditions are passed to the model through the open boundary at the open sea portion of the domain and the water surface. Water level variation, salinity and temperature distributions are input from data or larger scale models. Atmospheric inputs such as well as wind speed and direction, air temperature, relative humidity, cloud cover, atmospheric pressure and solar radiation are input through the water surface.

2.5.1 Fresh Water Inflows

Time series of water inflow are required to specify the upstream boundary conditions at the Dundee Dam (Passaic River) and the Oradell Dam (Hackensack River). The time series will be obtained, where available, from USGS gage records. Tributaries and overland flow sources will also be obtained or estimated and input to the model. In a similar way, adjacent to the more urbanized reaches of the domain, additional volume sources of water from CSOs, stormwater overflows and WWTPs will be obtained via the use of urban runoff models previously or currently under development by HydroQual for different municipalities that discharge to the Passaic River/Newark Bay system and reduced for input to the model.

2.5.2 Boundary Forcing

Boundary forcing will be achieved through specification at the open boundary. The values of water surface elevation, and temperature and salinity profiles will be developed from available data. The same data protocol developed for SWEM/CARP model (HydroQual, 2001) will be applied for this study, i.e., utilization of NOAA's World Ocean Atlas database for the temperature and salinity boundary conditions and Global Tidal Prediction Program (Egbert et. al. 1994) for the tides. Currently it is planned to embed a modified version of the high resolution Pence model of the Passaic River, Hackensack River, and Newark Bay system directly into the CARP model domain. If, however, this leads to unacceptable run times then the modified Pence model will be run in stand-alone mode. If this is the case, then boundary conditions (water elevation, salinity, and temperature) for the Pence model will be obtained from the CARP model.

It is proposed that estimates of stream flow for ungauged tributaries be performed as follows: multiply the ratio of the drainage area for the ungauged tributary to the gauged tributary time the ratio of the percent impervious of the ungauged drainage area to the percent impervious of the gauged drainage area times the tributary flow of the gauged tributary. Estimates of the drainage area and percent imperviousness are available from landside runoff models previously constructed by HydroQual.

2.5.3 Meteorological Data

The heating and cooling within the water body is provided by a heat flux calculation which accounts for solar radiation, air temperature near the water, humidity, cloud cover in addition to the temperature of the inflowing water upstream and the flux of temperature through the downstream boundary from the ocean. The wind speed and direction, available from local airports in the region, also affects the heating and cooling of the water body, but also significantly affects the current direction and level of turbulence in the system. Finally, with the wave model included in the calculations, the wind speed and direction will affect the wave climate and thus the sediment transport within the system. Hourly observations made at regional airports (i.e., Newark International, J.F.K, La Guardia) and offshore buoys, which are maintained by NOAA, will be used for specification of meteorological forcing.

2.6 MODEL CALIBRATION

2.6.1 Calibration and Validation Strategy

Calibrating a model is an iterative procedure whereby model parameters are evaluated and refined by comparing model results to observed data. Model validation is an extension to the calibration process to insure that a calibrated model will represent variables and conditions that the model must reproduce over longer time periods. Though related, the two procedures can be separated into two processes where some of the available data are used to calibrate and remaining data are used to validate. The calibration data might be in a period where there is a particularly high quality and/or high density of data, whereas validation data, which must be statistically independent to the data used for calibration, may be less dense and extend over a different period.

An extensive hydrographic data set was collected in the New Jersey tributary system during a field program conducted in support of SWEM calibration in 1994 and 1995 (HydroQual, 2001). Vertical casts of temperature and salinity were measured during the surveys. Figure 2-8 illustrates the location of these data stations. TSI also collected ADCP current data in 1995, which will be assessed in model calibration for the 1995 period. Additional survey data, conducted during the New York City DEP Harbor survey program, are also available in Kill van Kull and Arthur Kill (Figure 2-8). Field survey data conducted during 1988 and 1989 period, shown in Figure 2-8 is very limited and only surface salinity data are measured near the Raritan Bay area. These data are supplemented by the NJDEP coastal Monitoring Survey data as shown in Figure 2-8. As part of the New Jersey component of the Contaminant Assessment and Reduction Program (CARP) for the New York/New Jersey Harbor Estuary, hydrodynamic data were collected at various locations within Newark Bay system from 2000 to 2002. These data were collected using four different methods, namely, permanent tide gages, bottom mounts, stationary vessel profiling and vessel transect sampling. Sampling was performed under various tidal, freshwater and meteorological conditions. Figure 2-9 shows an inventory of the hydrodynamic data collected during the 2000-2002 field program. Final model validation will consider field data collected during 2004-2005 by Chant and MPI. These data from the 1988-89, 1994-95, 2000-02 and the 2004-2005 field programs will form the basis of the development of the hydrodynamic model in the present modeling effort.

For the present study, emphasis is placed on the calibration year 1994-95 since this period possesses a comprehensive database and was used for SWEM calibration. The 1988-89 database is not as extensive as the 1994-95 database and is particularly lacking in the New Jersey tributaries. Although the 1988-89 database is sufficient for validation purposes in the Raritan Bay, it does not provide for a robust model skill assessment in the Passaic, Hackensack Rivers and Newark Bay. However, this data set can be used to validate the boundary conditions to be set for the present model. It is envisioned that the data collected during the 2000 to 2002 period would form a good basis for the validation of the model.

2.6.2 Model Performance Measures and Skill Assessment

HydroQual has extensive experience in the calibration of hydrodynamic models and believes that it follows a logical and balanced approach to determining skill assessment of such models. However, given the importance of the lower Passaic River and Newark Bay in this project, particular attention will be focused on calibration of stage, flow (or currents), salinity, and temperature in this portion of the model domain. The skill assessment will be performed, given available data, on all four forms of the physical data (stage, flow, salinity, and temperature).

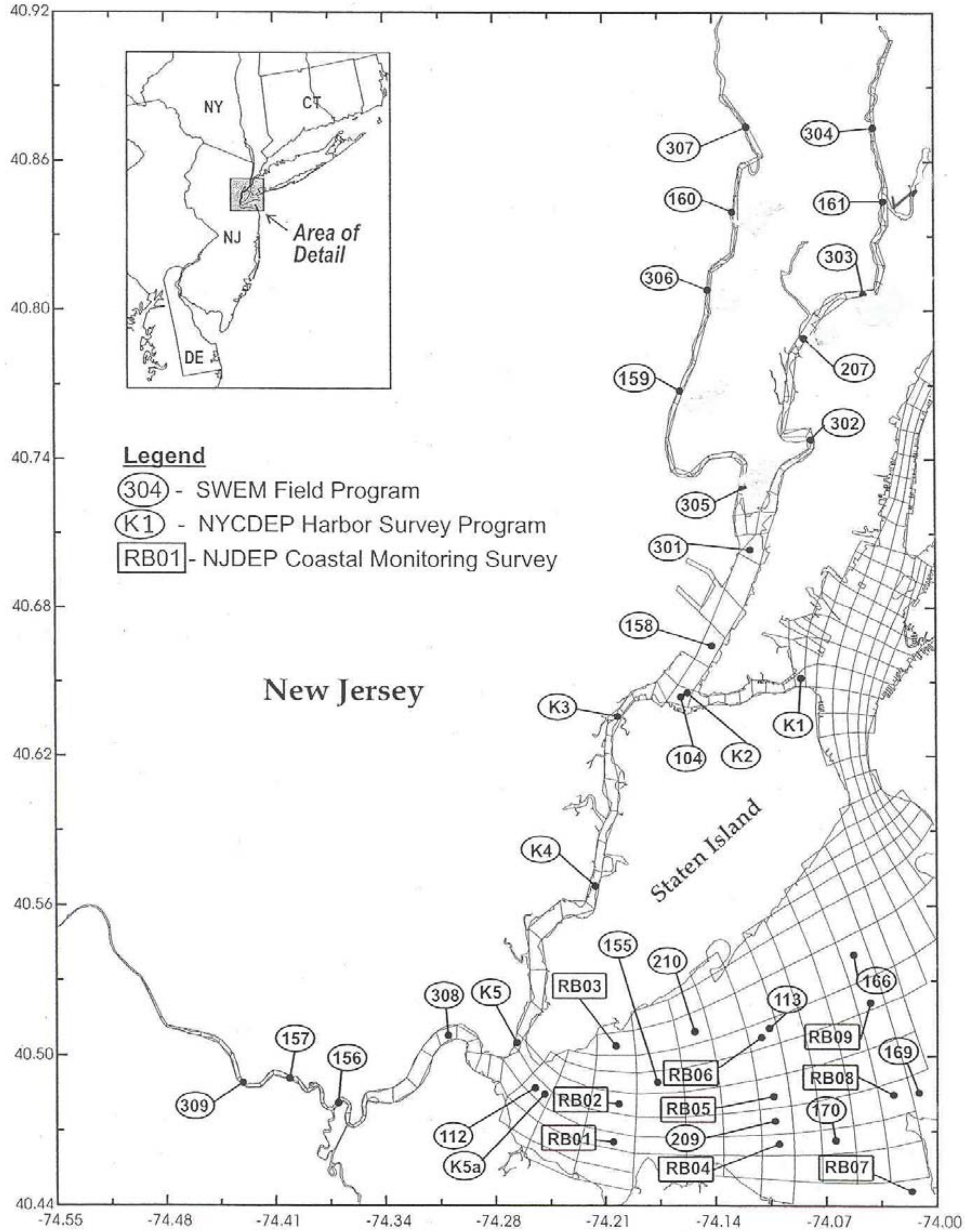

Figure 2-8. Historical field sampling locations in the study area

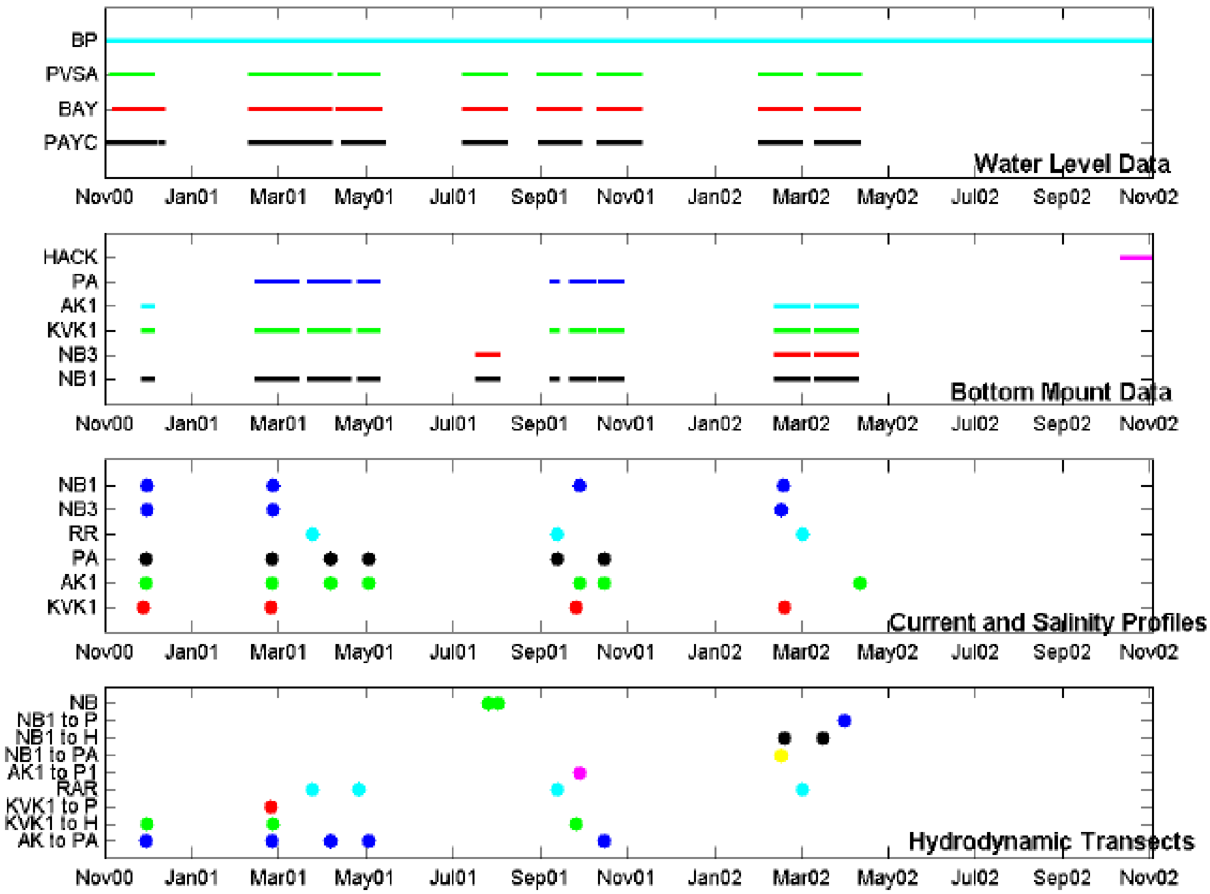

Figure 2-9. Timelines of field surveys for CARP study from 2000 to 2002. Locations of the stationary mooring are shown in Figure 2-1. (Pence, 2004)

Model performance measures provide a quantitative summary of model performance that can be factored into the assessment of whether the model results are adequate to support the decisions required to address the study objectives. Although no consensus on the model performance criteria has been established in the past or present literature, a number of "basic truths" can be established for the Passaic River study:

- Models are approximations of reality; they cannot precisely represent natural systems.
- There is no single, accepted statistic or test that determines whether or not a model is validated. Both graphical comparisons and statistical tests are required in model calibration and validation.
- Models cannot be expected to be more accurate than the sampling and statistical error (i.e. confidence intervals) in the input and observed data

All these "basic truths" will be considered in the development of appropriate procedures for quality assurance of the ECOM model to be used in the present study. Despite a lack of consensus on how they should be evaluated, in practice, the models elsewhere are being applied and their results are being used for assessment and regulatory purposes. A "weight of evidence" approach is most widely used and accepted when models are examined and judged for acceptance for these purposes. Based on the weight-of-evidence concept, derived from the truths, the following principles will be developed for the present modeling analysis:

- Because models are approximations of natural systems, exact duplication of observed data are not a performance criterion.
- The model validation process will measure the ability of the model to simulate measured values. As the project CSM (presented in Attachment A of the Work Plan) (Malcolm Pirnie 2005c) is updated with measured values from field sampling efforts, the numerical model will need to be able to explain the findings of the CSM.
- No single procedure or statistic is widely accepted as measuring, nor capable of establishing, acceptable model performance; thus numerous graphical comparisons and statistical tests will be adopted to provide sufficient evidence upon which to base a decision of model acceptance or rejection.
- Model and observed data comparisons must recognize, either quantitatively or qualitatively, the inherent error and uncertainty in both the model and observations.

The following graphical and statistical procedures will be used for the hydrodynamic model performance evaluation:

- Time series plots of observed and simulated results for stages and flows,
- Observed versus simulated scatter plots, with a best–fit linear regression line and correlation coefficient displayed for water levels, currents, temperature and salinity, and

- Error statistics, including mean error, absolute mean error, relative error, relative bias and standard error of estimate.
- An extensive analysis of the model data comparison and model validation will be performed to judge the adequacy of the model calibration and validation.

2.6.3 Model Sensitivity Analysis

Collection of meaningful datasets describing the physical processes of the Passaic River system is expensive and requires a great deal of efforts and resources. These data are required to force and validate the model and therefore reliable and meaningful datasets are vital to the success of the modeling analysis. One of the most important issues, which has been the focus of the present study, is to develop a mechanism that provides a measure of uncertainty in the modeling prediction. A highly sensitive parameter that is known with greater certainty may have much less effect on the uncertainty of model results than a much less sensitive parameter with high degree of uncertainty.

A simple approach will be adopted to analyze the model predicted percent changes in various hydrodynamic parameters such as water levels, currents, temperature, salinity and fluxes due to uncertainty in model input parameters (basic variables). This analysis has two objectives:

- To perform a check on the model framework and structure by evaluating if the changes in model results are reasonable, given the magnitude of the change in the model input and the processes affected by the input parameter.
- To determine the relative response of the model results to the perturbation in various model input parameters (basic variables).

The analysis to be performed in this study will provide insight to the model performance in terms of key parameters and the overall uncertainty of model prediction. The methodology determines the sensitivity of model results in terms of the percent changes in model predictions due to a perturbation introduced in each basic variable i.e. the input to the model. The base case for this analysis is one that provides the highest degree of model calibration using the best-known model forcing functions. A series of model simulations will be performed allowing perturbations in the basic variables. The basic variables considered are bottom roughness, freshwater flows, open boundary conditions such as mean water level, temperature and salinity and atmospheric forcing functions, such as wind speed and direction. The model results due to changes in basic variables will then be summarized and presented in a quantitative measure in a tabular form.

2.7 LINKAGE TO OTHER MODELS

i. Hydrodynamic model results will be generated for the ten water years from 1995 through 2004 in order to create a basis for long-term hydrodynamic condition of the study area.

ii. In addition, simulation of the year 1984, during which an extreme high flow event occurred, will be performed.

iii. For each water year simulation, ECOM will provide transport information to the sediment transport/organic carbon model (ST-SWEM) and to the contaminant fate and transport model (RCATOX) including time varying, preferably in hourly interval, volume exchange rates (fluxes), dispersion coefficients in three dimensions, and surface water elevations and bottom stresses induced by bottom currents and wave action.

iv. The linkages between ECOM and RCA (the computational framework) for ST-SWEM and RCATOX) have been verified to work. This is accomplished by including salinity as a state-variable in ST-SWEM and/or RCATOX and comparing the ST-SWEM/RCATOX computed salinity versus that computed by ECOM. During the initial development of RCA's coupling to ECOM, we have shown that RCA has been able to exactly match ECOM on a time-step by time-step basis.

SECTION 3

SEDIMENT TRANSPORT

3.1 INTRODUCTION

The development of a sediment transport model depends on achieving an understanding of how the mass of solids moves, deposits, resuspends and redistributes in the Lower Passaic River Restoration Project domain.

In general, the sediment transport within the Lower Passaic River is dominated by the suspended sediment transport coming from sources outside the domain (i.e., upstream Passaic River across Dundee Dam, tributaries, CSOs and storm waters) and from internal sources (i.e., resuspension and deposition of particles). The mass of solids entering the system is evident in the volumes of suspended solids observed in the water column under normal and storm events conditions. The accurate determination of the solid load is a critical element of the model, since it influences the net deposition of suspended particles. In addition to providing an estimate of the solids loading into the Passaic River and an analysis of the suspended solids data, this section gives a detailed description of the sediment transport formulations, the modeling approach, the data needs and the calibration/validation methods proposed for the Lower Passaic River Restoration Project.

3.2 SUSPENDED SEDIMENT LOADINGS

Suspended sediment loading estimates were developed for the CARP program using HydroQual's Normalized Sediment Load (NSL) approach. NSL is a non-dimensional loading function with predictive capabilities, and takes into account the observed behavior of rivers (i.e., a large fraction of the annual sediment load occurs during a relatively small number of events). It calculates daily suspended sediment loadings normalized by mean daily sediment discharge under non-flood conditions as function of the daily flow rate normalized by the long term mean flow rate drainage basin characteristics, and a stochastic term which accounts for variability. A description of the methodology is provided in Appendix D. Figure 3-1 compares for six water years, estimates of sediment loadings into the Passaic River using the NSL approach with the New Jersey USGS rating curve approach. The Figure shows that suspended sediment loadings are in general agreement. The annual sediment load varies between 20,000 tons/year and 35,000 tons/year. It should be noted that the tributary characteristics applied for determining loading using NSL were derived from Little Falls, NJ (location of the USGS station). For the purpose of the Lower Passaic River Restoration Project, more accurate measurements of data required to generate loading estimates are planned upstream and downstream of the Dundee Dam. It is also estimated that solid loads from about 109 CSOs, and a larger number of storm water outfalls constitute between 10 and 20% of the load into the Passaic (HydroQual 1999c). Sampling details for estimating current loads for tributaries, CSOs and storm water are provided in the Field Sampling Program document recently prepared by

Malcolm Pirnie with assistance from HydroQual and Battelle. It is expected that much of the uncertainty that exists in the definition of the sediment load at Dundee Dam, will be addressed by a one-year sampling program that will be performed by the USGS. This program will include sediment sampling within events and also continuous measurement of turbidity for estimating daily sediment loads. It is believed that this extensive sampling program will reduce upstream Passaic River sediment load uncertainty to an acceptable level.

3.3 SUSPENDED SEDIMENT CONCENTRATIONS

The most comprehensive total suspended solids (TSS) data set comes from the 1995-1996 Sediment Mobility Testing Program (TSI, 2004); measurements were conducted in July 1995 (i.e., semi- or hourly samples for seven or 10 consecutive days at different depths) and in April and May 1996 (semi- or hourly samples for 12 consecutives days in each month). A very limited data set was also collected in 1999 as part of the USACE Drift Removal Monitoring Program. Very recent information is available from the work conducted by Rutgers University on the Lower Passaic River. This information has not yet been fully released, but will be available in the near future. A brief analysis of TSI's 1995-1996 dataset is presented below.

The TSI program was carried out along a transect that extends from RM7.2 down to the mouth of the Passaic River. Although the data presented in Figure 3-2 represent an average over a 9 month period, a distinct TSS concentration gradient with depth can be observed: concentrations measured at the surface were always much lower than those observed in the bottom 15 ft, and were usually less by 100 mg/L. Concentrations as high as 4.5 g/L were also measured during the sampling period. These levels, however, were short-lived pulses. Recent information from Rutgers field investigators (Dr. Bob Chant) suggest that during April 2005, as peak flows reached 12,000 cfs pushing the freshwater front into Newark Bay, TSS levels were likely to vary between 300 mg/L and 800 mg/L. Dr. Chant expected the TSS level to reach 1000 mg/L near the salt front.

The shape of the tidal profile seems also to reflect the influence of the tidal currents that causes resuspension of sediments in the water column. There seem to be enough magnitude in those currents to resuspend solids. Similar trends were observed in Chesapeake Bay where during maximum tidal flow, solids - as well as zooplankton materials - were resuspended from the bottom sediment (Roman et al, 2001). In addition, the spatial distribution of TSS in the Lower Passaic River measured on depth-integrated samples collected during the program shows that, despite the scatter, TSS concentrations are higher upstream of the river, decrease with distance downstream, before peaking again near the mouth of the river at its confluence with Newark Bay (Figure 3-3). The decrease likely reflects solids deposition in the river, whereas the increase at the mouth is probably associated with higher solids resuspension in shallower areas as a result of high flood tidal currents and/or effects of wind-induced waves. In any case, the analysis of the TSS data, the dominance of cohesive particles in the Lower Passaic River (section 1.4.1.), the high sedimentation rates (section 3.9.3), as well as the lessons learned from the CARP project, point to the need of a sediment transport model that– as explained below – accounts for the suspended load, including resuspension and deposition of solids and carbon, upstream and tributary loading, bed armoring, flocculation and settling, and bioturbation.

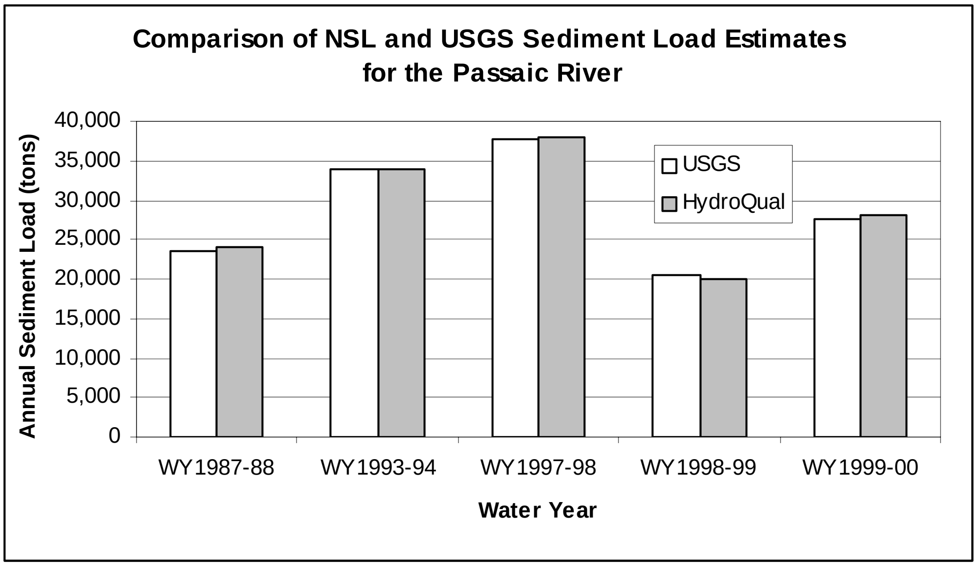

Figure 3-1. Comparison of NSL and USGS sediment load estimates for the Passaic River

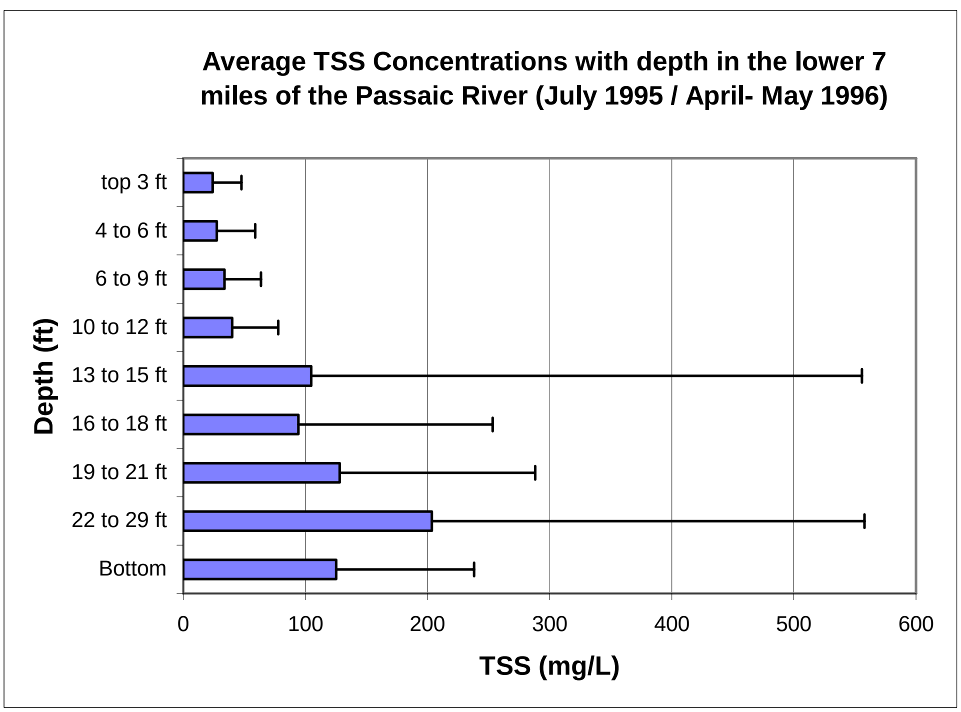

Figure 3-2. Mean and Standard Deviation TSS concentrations with depth in the Lower Passaic River.

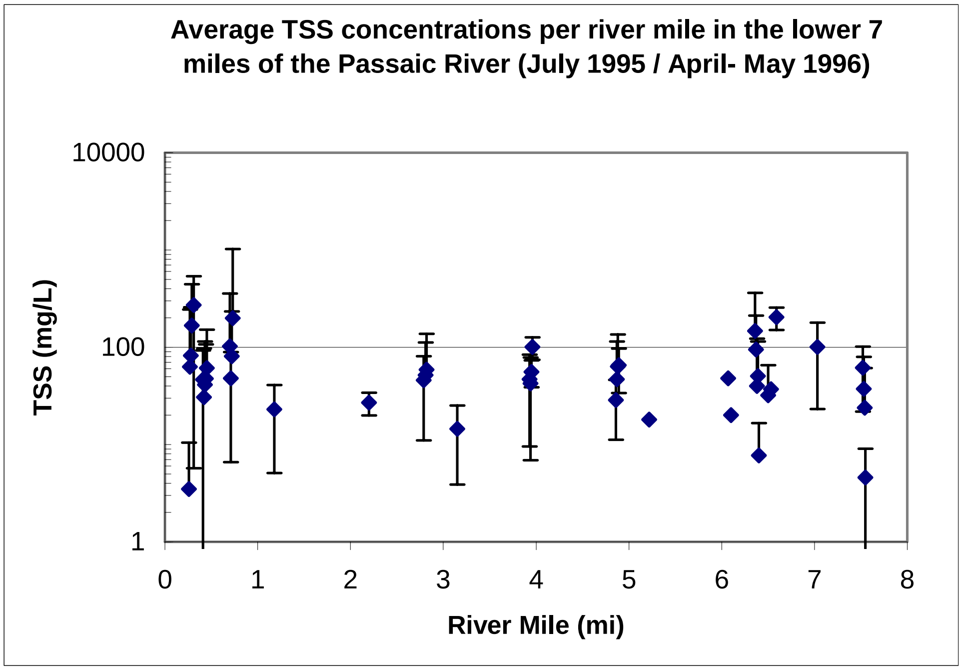

Figure 3-3. Spatial distribution of TSS concentrations in the Lower Passaic River (mean, max, min). Note the logarithmic scale.

3.4 PURPOSE

The purpose of sediment transport modeling is to establish how sediment moves through the Passaic River study area, how sediment is deposited in certain areas, and how sediments are mobilized and redistributed by tidal currents and large flow events.

Previous sediment transport modeling of the Passaic River study has been performed as part of larger regional studies for toxic contamination (Farley, personal communication) for New York-New Jersey Harbor and adjoining waters.

Because organic matter is an important component of the suspended sediment and because organic carbon concentrations are greatly influenced by nutrient cycles in the harbor and its adjoining waters, the sediment transport calculations were built directly into SWEM. This allows the interactions of inorganic and organic solids (e.g., through coagulation) to be considered explicitly in the model calculations. The full sediment transport-organic carbon cycle calculation (ST-SWEM) will be described in section 4.

Preliminary results (Farley, personal communication) from the CARP sediment transport model indicate:

1. Newark Bay-Passaic River section of the harbor typically serves as an accumulation zone for sediment,

2. Major sources of sediment to Newark Bay-Passaic River section are suspended sediment inputs from above Dundee Dam and bottom water transport of sediment from New York Harbor,

3. Tidal energies may be sufficient to cause resuspension and re-deposition of sediment over the tidal cycle,

4. Bottom water transport tends to move resuspended sediments toward zones of bottom water convergence,

5. Major mobilization of sediments is not expected to occur for flows during the six CARP years (given years and maximum Passaic River freshwater flow).

Additional modeling studies are needed:

1. to examine the effects of finer scale grid resolution (with better definition of bottom bathymetry) on bottom shear stresses and the potential for sediment resuspension,

2. to examine the effects of finer scale resolution on bottom water transport and movement of resuspended sediment,

3. to test various formulations for characterizing settling behavior,

4. to implement the SEDZLJ sediment transport model algorithms for describing resuspension and bed behavior (e.g., consolidation, armoring),

5. to use measured data to develop site-specific coefficients generated from Sedflume and Gust field experiments to describe sediment transport in the Passaic River study area.

Further, at the request of the Technical Advisory Committee (TAC), HydroQual was directed to proceed with incorporating SEDZLJ kinetics into the ST-SWEM and ECOMSED frameworks applied previously for the Harbor under CARP. The SEDZLJ work will proceed in parallel with our consideration of model formulations as described in Section 3.5

3.5 MODEL FORMULATION

3.5.1 Settling Formulation

For non-cohesive particles, settling velocities can be described reasonably well as a function of particle density and diameter (e.g., using Stokes law). For cohesive particles, particles are continually assembled (and disassembled into flocs by coagulation and disaggregation processes. This results in significant changes in the effective diameter and density of flocs and in their rates of settling through the water column. Detailed model calculations for the coagulation and settling of cohesive particles have been developed over the years (Valioulis 1983; Farley and Morel, 1986). These models consider multiple particle size classes and describe the transfer of particle mass between size classes by particle collision rates (Smoluchowski 1916, 1917) and collision efficiency functions. Extension of this framework to include disaggregation processes is discussed in Lick and Lick (1988).

Although multiple size class models are possible, their applications are limited in large water quality studies where the additional dimension of particle size adds to the computational burden of the calculation, and potential fast aggregation/disaggregation kinetics leads to the requirement for extremely small time steps and/or the use of implicit solvers. As an alternative, many water quality models are based on the specification of apparent settling velocities to describe overall mass removal rates of cohesive sediment from the water column. A glaring limitation of this approach is that apparent settling velocities may vary significantly with changing conditions in the water column (e.g., solids concentrations, shearing rates). In addition, apparent settling velocities are difficult to assigned with any degree of confidence. For example, settling velocities that have been used in various modeling studies of the harbor and its adjoining waters have varied over several orders of magnitude from 10^{-1} m/day (Dortch et al. 1999) to values approaching 10^{2} m/day (Geyer et al. 2001).

Various power law functions have also been proposed to describe the overall rates of coagulation, disaggregation and settling. A preliminary list of functions is given in Table 3-1.

Although the proposed functions for apparent settling velocities have certain similarities, there is no generally accepted formulation for describing the mass removal rate of cohesive sediment. An evaluation of the settling velocity formulations will be performed as part of our studies. Specific tasks will include:

1. Literature review to provide a more complete list of proposed formulations and to assess the methods and data used in development of the approach.

2. Comparison of proposed formulations as a function of cohesive particle concentrations, fluid shear, etc.

3. Comparison of proposed formulations to field observations for the Passaic River study area. (Note that in this analysis care will be taken to distinguish between floc settling velocities and overall mass removal rates).

4. Selection of one or possibly two formulations for testing in the sediment transport model (see below).

Table 3-1. Simplified Functions to Describe Coagulation, Disaggregation, Settling of Cohesive Solids

$w_s = a \cdot SS^{4/3}$	Krone (1962)	Based largely on field observations
$w_s = a \cdot SS^m$	Many other investigators (see Mikkelsen and Pejrup, 2000)	Based largely on field and laboratory observations
$w_s = -B \cdot SS \cdot h$	Morel and Schiff (1980); Hunt (1982)	Based on laboratory studies
$w_s = \left[-B_{ds} \cdot SS^{1.3} - B_{sh} \cdot SS^{0.9} - B_b \cdot SS^{0.3}\right] \cdot h$	Farley and Morel (1986)	Based on multiple particle size class modeling results and laboratory observations
$w_s = a \cdot (SS \cdot G)^m$	Manning and Dyer (1999)	Based on laboratory studies
$w_s = a \cdot SS \cdot G^{-0.5}$	Winterwerp and Van Kesteren (2004)	Based on model application
$w_s = 80 + (.268d^{1.56} - 80)e^{-.07G}$ $d = 9.0(SS\,G)^{-0.56}$ (freshwater)	Lick et al. (2005)	Based on laboratory experiments.

SS = suspended solids
h = depth
d = diameter
B = coagulation-settling rate coefficient
B_b = coagulation-settling rate coefficient associated with Brownian motion
B_G = coagulation-settling rate coefficient associated with fluid shear
B_{ds} = coagulation-settling rate coefficient associated with differential settling
G = fluid shear
Note: Mass removal rates for Morel and Schiff (1980), Hunt (1982) and Farley and Morel (1986) were converted to apparent settling velocities by dividing the expressions by SS and multiplying the expressions by h.

3.5.2 Erosion, Deposition, and Sediment Bed Formulations

An almost inevitable consequence of the complexity of fine sediment erosion processes has been the adoption of multiple formulations for interpreting data and modeling erosion. Perhaps the most commonly used formulation is simple linear erosion with a constant critical stress, in one of two forms:

$$E = M\left(\frac{\tau_b}{\tau_c} - 1\right) \tag{3.1a}$$

$$E = M'(\tau_b - \tau_c) \tag{3.1b}$$

(Ariathurai and Krone, 1976; Lang et al., 1989; Sanford and Halka, 1993; Sanford and Maa, 2001; Van Ledden, 2002; Winterwerp and Van Kesteren, 2004), where E is erosion rate, M (or M') is a constant of proportionality, τ_b is the applied bottom shear stress, and τ_c is the critical stress for erosion. This is quite similar to the most commonly used expression for mobilization of non-cohesive sediments (Harris and Wiberg, 2001; McLean, 1985), and it has been adapted recently for use with depth-varying critical stresses (Sanford and Maa, 2001). Also common are power law expressions, with or without a critical stress:

$$E = M(\tau_b - \tau_c)^n \tag{3.2}$$

(Lavelle et al. 1984; Lick 1982; Maa et al., 1998; Roberts et al., 1998), where n is an empirically derived exponent.

The sediment transport model SEDZLJ (Jones and Lick 2001) will be incorporated into ECOM for Passaic sediment transport modeling. We will utilize the default erosion formulations in SEDZLJ to the extent possible. These are generally of the form of eq. 3.2, fit directly to site-specific erosion testing data. SEDZLJ models 2 types of sediment bed: in-place sediments whose erosion rates are characterized based on direct erosion testing of cores collected in situ, and new sediments deposited on top of the in-place sediments.

Erosion of in-place sediments will be modeled based on spatial interpolation of the in situ erosion testing data (see below). In Situ erosion measurements collected with a Sedflume (for deep erosion) and a Gust microcosm (for surficial erosion of fine sediment) will yield depth dependant profiles of erosion rates as a function of shear stress, critical shear stress for erosion, $\tau_c(z)$, location, bulk density, and sediment grain size. These data will be used to define the initial conditions of the sediment bed. Using the measured data, erosion will be allowed to proceed only if $\tau_b > \tau_c$.

Different sequences of erosion, deposition, consolidation, and bed armoring can lead to potentially large variability in surface sediment erodibility. *In situ* measurements of sediment erodibility provide snapshots of the condition of the bed at one point in time, but for the most part cannot address short-term temporal variability without the incorporation of a mechanistic sediment bed model. We will address this limitation by implementing bed mechanisms based on the SEDZLJ model framework. Erosion of pre-existing or in-place sediments will be modeled using applied bottom shear stresses, as computed by the hydrodynamic model, and in situ microcosm and Sedflume erosion tests as described above. Erosion of newly deposited sediments will be modeled based on laboratory erosion tests of consolidating sediment slurries, which will be used to derive changes in sediment erodibility due to consolidation. Consolidation will be modeled by allowing the erosion characteristics of newly deposited sediment layers to adjust through time towards an equilibrium state, from which they may be further perturbed by transport events. The equilibrium state for newly deposited sediments, modeled as a function of depth and sand/mud mixture, will be based on laboratory microcosm and Sedflume erosion tests of consolidating Passaic sediment slurries.

The deposition rate is written as:

$$D = pw_s c_{dep} \tag{3.3}$$

Where w_s is the settling speed of the sediment particles just above the bed (predicted by the floc model), cdep is a reference suspended sediment concentration just above the bed, and p is the probability of deposition. Traditionally, the probability formulation attributed to (Krone, 1962) is employed in cohesive sediment transport models:

$$p = (1 - \frac{\tau_b}{\tau_d}) \tag{3.4}$$

where τ_d is the critical stress for deposition such that no deposition occurs for $\tau_b > \tau_d$. Probabilistic versions of Equation 3.4 have also been proposed (Parthenaides 1992) and implemented (e.g., in the present version of HydroQual's ECOMSED). SEDZLJ uses a probability of deposition of the form of eq. 3.4 for cohesive sediments and a probabilistic expression for non-cohesive sediments. However, (Sanford and Halka, 1993) showed that p=1 frequently describes natural scale erosion-deposition cycles better than equation (3.4) and standard non-cohesive bedload transport models assume that p=1. In addition, (Winterwerp and Van Kesteren, 2004) show that the flume experiments that formed the original basis for equation (3.4) may be equally well described with p=1, as long as newly deposited sediment is allowed to consolidate and become resistant to future erosion.

For the Passaic sediment transport model, we will begin with the default deposition probability formulations in SEDZLJ, but will test and compare deposition formulations both with and without (p=1) a critical stress for deposition if it appears that the default SEDZLJ formulations are not performing satisfactorily. In addition, preliminary work has indicated that a bed model including the potential for either resuspension or consolidation of newly deposited sediments, depending on the stress time history and the deposition rate (e.g., as described by (Winterwerp and Van Kesteren, 2004) may remove many of the distinctions between the two deposition modes. Since we plan to adopt such a model for deposited layers, it may be that the exact formulation of the probability of deposition is not critical.

3.6 SEDIMENT TRANSPORT MODELING APPROACH

The sediment transport component of the modeling framework will be calibrated using multiple lines of evidence, including comparisons between computed and measured water column suspended sediment, and comparisons of spatial patterns of computed sedimentation rates with estimates developed from analyses of bathymetric data and analyses of radionuclide tracers in cores. Parameters in the settling and resuspension formulations, discussed in the preceding section, will be evaluated on the basis of site-specific data that will be collected in 2005 and 2006. These data collection efforts will provide a more complete dataset for water column suspended sediment than is presently available, and will, therefore be an important part of the sediment transport model calibration. Model simulations for the period 1995 – 2006 will further test the parameterization of the sediment transport model. This time period was selected because a substantial amount of sediment-contaminant data were collected in 1995 and will be used to assign initial conditions for the model simulations. The most rigorous test of the sediment transport model will be conducted as part of hindcast simulation for cesium (^{137}Cs), through the evaluation of spatial patterns in sedimentation rates computed over approximately half of a century.

3.7 SEDIMENT TRANSPORT MODEL INPUTS

The sediment transport model requires several types of inputs, including representation of particle size distributions by a limited number of size classes, initial conditions, boundary conditions, point source loads, parameters for the resuspension, settling, and deposition formulations, and advective and dispersive transport information.

3.7.1 Determination of Non-cohesive Size Classes

The sediment transport model will include two broad groups of solids: cohesive and non-cohesive. Non-cohesive sediment is generally inorganic and predominantly composed of sand-size quartz grains, whereas cohesive sediment is composed of a mixture of clay, silt, and organic particles. Non-cohesive sediment particles are generally larger in diameter and the particles are easily separable, whereas cohesive sediment particles are small, tend to be flat or plate-like, and often possess a non-uniform static charge that allows the particles to stick together as aggregates of hundreds or thousands of particles.

Detailed grain size analysis will be available from high resolution cores collected in selected locations in 2005-06. Unless analysis of this data indicates otherwise, one cohesive class will be used to represent disaggregated particles in the size range less than 63 um. A flocculation model (see above) will be used to predict the transport and settling characteristics of this fine sediment. An analysis of sediment particle size distribution data in the size range greater than 63 um will be performed to determine the number of non-cohesive solids classes that will be included in the model and the effective particle diameters for each class. The number of non-cohesive grain size classes will be selected by considering a balance between the effect on model run-time, the ability of the model to reasonably depict bed armoring and predict the transport characteristics of eroded sediments. The breakpoint between non-cohesive size classes will correspond to sieve sizes used in planned and historical analyzes of the sediment samples. The effective diameters used in the model to represent non-cohesive sediment within these size ranges will be evaluated by considering three different methods:

- Based on the median diameter (d50) of particles with each size class.
- Based on settling velocities associated with measured non-cohesive particle sizes.

- Based on critical shear velocities associated with measured non-cohesive particle sizes.

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3.7.2 Initial Conditions for the Different Sediment Size Classes

Initial conditions in this context refer to the concentrations of each state variable (each cohesive and non-cohesive solids class) in each layer of the water column and sediment-bed in each model grid element, at the start of the simulation. Initial conditions will be based on data collected in 1995, or as close to 1995 as practical, depending on the spatial coverage of data for bed properties (i.e. bulk or dry density, porosity and particle size distribution). In grid cells corresponding to areas where multiple samples were collected, averages of data will be used as initial conditions. The common situation is that the grid resolution will be finer than the spacing of the sampling locations, in which case interpolation of the data will be required to provide estimates of concentrations in grid cells that do not contain a sampling location. Potential interpolation schemes include, but are not limited to, inverse distance weighting, spline surfaces, kriging, and triangular irregular networks (TIN).

Initial conditions in the water column are less important than initial conditions in the sediment, because water column concentrations will be flushed from the model domain much more quickly than initial conditions in the sediment bed. Based on information that has been compiled to date, suspended solids/sediment data are not available for assigning initial conditions for water column solids, and therefore, water column initial conditions will be assigned at "typical concentrations" indicated more recent data. This data gap is not expected to have a substantial, or lasting effect on the computations over the course of the 11-12 year simulation.

3.7.3 Boundary Conditions

Suspended sediment inputs to the model domain must be specified at each location where inflows are specified in the hydrodynamic model. Time series of suspended sediments flowing into the upstream boundaries of the model at Dundee Dam (Passaic River) and Oradell Dam (Hackensack River) and from tributaries, including Saddle River, Third River, Second River, Lawyer Creek, and Frank's Creek, will be specified. Data collection planned for 2005 and 2006 will provide a basis for describing time variable model inputs for only a portion of the 1995-2006 simulation period. Data analyses will be performed to develop a basis for describing suspended sediment inputs to the model domain for periods for which data do not exist.

Automated sampling of suspended sediment will be conducted at the upstream boundary on the Passaic River at Dundee Dam. Data from this sampling program will be used to develop a suspended-sediment rating curve, using the normalized sediment loading (NSL) technique (HydroQual, 1996, reproduced in Appendix D) that was applied to data from the Passaic River at Little Falls, and the Hackensack River at New Milford as part of the CARP project. The distribution of total suspended sediment among the model's cohesive and non-cohesive solids classes will be based on variations in composition measured during high flow events, when non-cohesive solids could be carried into the model domain because of increased turbulence upstream of the model boundary. Particle size distributions of suspended sediment will be determined for samples analyzed with a Malvern Mastersizer. Specification of boundary conditions for the remainder of the tributaries represented in the hydrodynamic model will be based on data collected in the "Tributary and Fixed Transect Water Column Sampling" program. Data collected in this program will be evaluated to determine if the NSL technique can be applied to the more limited data for these smaller tributaries.

3.7.4 CSO Sources

Specification of time variable suspended sediment inputs from combined sewer overflows will be based on data collected in the "CSO Sampling" program. These data will be analyzed to determine if time-variable relationships between solids concentrations and precipitation can be developed, or if an event-mean concentration approach is more reasonable. The variability in the available data and the relative magnitude of the measured loadings from CSOs and other sources will be considered in developing the final approach for representing solids loadings from CSOs.

3.7.5 Erosion Characteristics

Cohesive sediment erosion is highly site-specific, requiring measurements to define parameters in formulations used to describe erosion rates as a function of shear stress exerted on the sediment-bed. Erosion rates depend on the relative magnitude of the shear strength of the sediment and the shear stress exerted on the sediment surface. Bulk density, particle size distribution, mineralogy, organic content, pore water salinity, amount of gas, oxidation or other chemical reactions, and consolidation time can affect the shear strength of the sediment. Two devices will be used to measure erosion rates of sediments in the Passaic River:

(1) a Gust Microcosm will be used to evaluate erosion from the surficial sediment (<5 mm). Gust Microcosm field experiments will be conducted to test for changes in surficial sediment erosion characteristics over the range of 0-0.4 Pa applied shear stress. These erosion tests, which involve increasing shear stress through approximately eight levels, with each level of constant stress lasting approximately 20 minutes, will be performed according to protocols described in detail in Sanford and Maa (2001).

(2) Sedflume will be used to measure erosion throughout the depth of a sediment core. The erosion experiments will be conducted in the field on cores collected from 15 locations in the river. Sediment cores will be collected using box corers for these experiments. During the Sedflume erosion tests, small amounts of sediment will be removed at different depths in the core and used to determine other bulk properties of the sediment, including water content, grain size (using a Malvern Mastersizer) and organic content (Roberts, et al., 1998). Sedflume experiments will be conducted on sediment cores to determine erosion rates as a

function of depth and shear stress. This flume can measure erosion rates of sediments at high shear stresses (up to stresses on the order of 20 Pa) and with depth (down to a meter or more). Therefore, Sedflume measures sediment erosion at shear stresses ranging from normal flow to flood conditions and with depth below the sediment/water interface. Protocols for conducting Sedflume experiments are described in McNeil, et al. (1996). For a better interpretation of the Sedflume data, the use of a density profiler will also be considered (depending on instrument availability and funding) in order to obtain density as a function of depth in each core with a very fine resolution (~ 1cm). In any case, a thorough literature review on similar studies will be conducted.

Because of the sparsity of in-place erosion testing data and the known heterogeneity of Passaic bottom sediments, a wide array of data types will be used to spatially interpolate erosion characteristics from the erosion tests to the model grid cells. We will combine shear stress distributions predicted by the hydrodynamic model for major flow and storm events, observed grain size and roughness distributions from a compilation of various sources (including high resolution side scan), detailed bathymetric maps, and maps of depositional thickness to place each of the core locations in context and facilitate the spatial distribution of observed in-place erosion characteristics.

In addition to erosion tests on cores collected in situ at selected sites in the Passaic, erosion experiments will be run with both devices in the laboratory to characterize changing erosion characteristics over time due to consolidation after deposition. Surficial sediments will be collected at depositional sites in the lower Passaic, transported to appropriate laboratory test sites, slurried to a uniform high water content, poured into test cores, and subjected to erosion testing at fixed intervals to determine the time and depth course of developing erosion resistance. The results of these experiments will be used to develop estimates of the parameters in the erosion formulation for deposited sediments, as discussed in the preceding section.

3.7.6 Sediment Settling/Flocculation

Settling velocities of non-cohesive particles are predicted from the diameter and specific gravity of the particles (van Rijn, 1984). Cohesive particles may be differentiated from non-cohesive particles by the fact that they are subject to interparticle forces that allow the cohesive material to be subject to aggregation (flocculation) resulting from electrostatic or organic binding forces and collisions between particles. Collisions occur due to three primary processes: Brownian motion, fluid shear, and differential settling. Continued aggregation results in larger-sized aggregates (flocs) that can be characterized by higher porosity, increased irregularity and fragility, and higher settling rates (Krone, 1962).

Suspended particle sizes and settling velocities will be estimated in situ through use of a laser in-situ scattering and transmissometry (LISST) instrument system in combination with an optical backscatter sensor (OBS) and direct estimates of suspended sediment mass. These devices have been used to determine concentrations and fall velocities of estuarine particle populations in Chesapeake Bay, with details described in Fugate and Friedrichs (2002) and Sanford et al. (2005). In any case, a literature review will be conducted on previously conducted flocculation studies, in addition to conducting settling field measurement and, if possible, laboratory experiments such as the disk and Couette flocculators The results of these experiments will be analyzed to calibrate estimates of suspended particle size and settling speed described in the preceding section.

3.7.7 Bed Layering and Mixing

The sediment bed will be modeled as a series of vertical layers of variable composition in which all sediment properties will be tracked. The thickness of the vertical layering scheme for the sediment-bed will be determined from vertical gradients of erosion characteristics and a radionuclide tracer (^{7}Be), which will be measured in high-resolution cores. As described above, there will be two types of sediment layers: deposited layers in which erosion characteristics adjust through time towards an experimentally determined equilibrium state, and between which mixing due to bioturbation may occur; and in-place sediment layers with their erosion characteristics and compositions fixed at observed conditions. Existing algorithms in SEDZLJ allow for development of new depositional layers and mass-conserving exchange between layers. A simplified consolidation algorithm based on laboratory tests and bed mixing due to bioturbation will be added for the Passaic model. It is worth noting that the use of a density profiler on collected sediment cores will help investigate the effects of bioturbation near the sediment surface interface with a 1.0 mm resolution.

3.8 MODEL OUTPUT

The output of the sediment transport model includes

- water column concentrations of cohesive solids in each model grid cell,
- water column concentrations of each non-cohesive solids class in each model grid cell,
- sediment-bed concentrations of cohesive solids in each model grid cell,
- sediment-bed concentrations of each non-cohesive solids class in each model grid cell,
- net-depositional flux of solids to the bed,
- erosion rates of cohesive and each non-cohesive solids class.

These concentrations are computed on the time-scale of seconds because of stability limitations on the hydrodynamic time step. Model results can be saved as averages over longer time periods (e.g. an hour, week or month) depending on the type of model-data comparison of interest. The relative concentration of each of the solids classes in the sediment bed can result in changes in the bed-armoring condition. The calculated deposition of solids at every grid cell is saved in terms of an areal-flux rate ($M/L^2/T$) and a bed elevation change (L/T). Calculated erosion rates are passed through the model linkage to the resuspension of particulate organic carbon and sediment-bound contaminants.

3.9 MODEL CALIBRATION

The calibration period for the sediment transport component of the modeling framework will extend from 1995 through the period of sampling in 2005 – 2006. The calibration approach will be to use the period of increased data density in 2005 and 2006 to evaluate individual processes (e.g. intra-tidal resuspension, settling, and high flow induced resuspension). Concurrent longer simulations for the 1995-2006 period will be performed to evaluate the effect of the parameterizations of settling and resuspension on sediment-bed conditions over this longer period. By screening model parameterization on intra-tidal and high-flow event time-scales, combinations of parameters that fail to reproduce the high-resolution data will not have to be tested in decadal scale simulations. It is recognized that extreme high flow events may be important to delivering large sediment loads to the Lower Passaic River as well as potentially resuspending/eroding the riverbed. The model will be exercised for high flow events such as Hurricane Floyd (1999) and the Passaic River floods of 1984.

3.9.1 Calibration Strategy

Parameters that will be adjusted as part of the calibration process include terms in the settling and resuspension formulations for cohesive solids. As discussed above, alternate settling formulations will be investigated. The range in parameter adjustments will be constrained by values reported in the literature and analysis of the data from site-specific studies that will be conducted in 2005 – 2006 (e.g. Gust microcosm and Sedflume erosion measurements, settling tube experiments). Experimental results will be evaluated to determine if temporal or spatial variability in particular parameters (e.g. critical shear stress for erosion) would be appropriate. The calibration strategy will be to keep parameters temporally and spatially constant unless there is evidence to support temporal/spatial variations. Varying model parameters in space and/or time to improve model-data comparisons does not necessarily improve the predictive power, and therefore, utility of the model to contribute to management decisions.

The focus of the calibration efforts will start in the upstream portions of the model domain, in an attempt to deliver reasonable suspended sediment loads to downstream locations. As part of the initial efforts, gross mass balances will be performed to assess the trapping efficiency in different parts of the model domain that would be required to reproduce estimated sediment accumulation rates, given the solids loadings from the various sources included in the model.

Multiple lines of evidence will be used in the calibration process, including comparisons between computed and measured water column suspended sediment and comparisons of spatial patterns of computed sedimentation rates with estimates developed from analyses of bathymetric data and analyses of radionuclide tracers in cores.

3.9.2 Water Column Suspended Sediment

Historical water column suspended sediment data within the Lower Passaic River Restoration Project model domain is fairly limited. The majority of water column suspended sediment data that will be used for calibration will be collected in 2005 and 2006, specifically to support the model calibration effort. These sampling efforts (FSP MPI, 2005a), will include:

- Shipboard Surveys with Hydrodynamic Data Collection Program
- Fixed Transect Water Column Sampling Program
- High Flow/Storm Sampling

3.9.3 Sedimentation Rate Comparisons

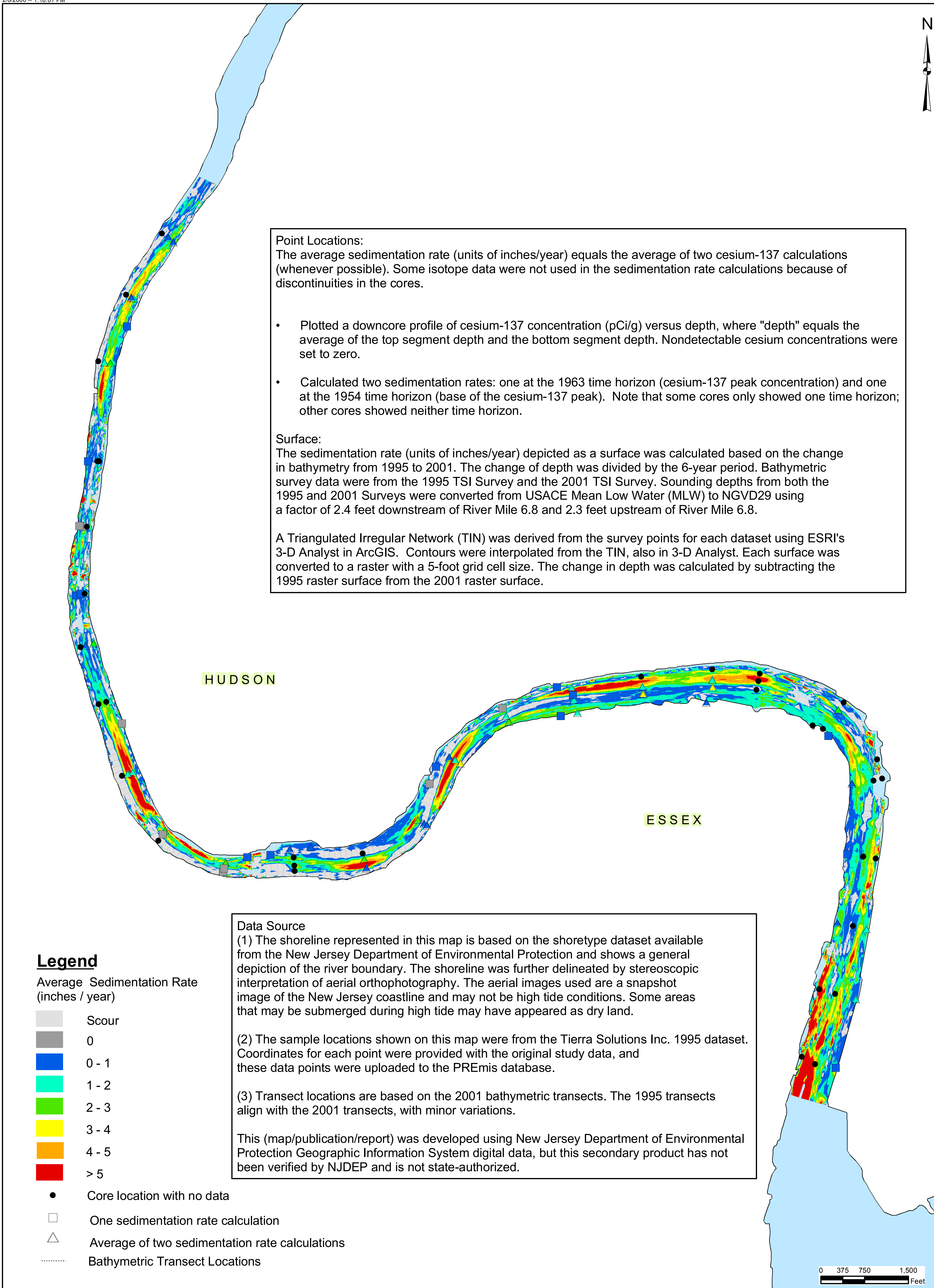
Spatial patterns in sedimentation rates have been estimated from an analysis of two bathymetric surveys, conducted in 1995 and 2001 by TSI. Point estimates of sedimentation rates, calculated from vertical profiles of radionuclides in cores collected in 1995 are also shown on Figure 3-4. Estimated sedimentation rates of more than 3 inches per year (7.6 cm/yr) in many areas correspond to accumulations of 3 feet (0.9 m) over the 1995-2006 simulation period. It is envisioned that accumulation rates of this magnitude will require that the simulation be executed as a series of shorter simulations to allow the bathymetry to be re-initialized to reflect the deposition that has been calculated.

3.10 SEDIMENT TRANSPORT MODEL SKILL ASSESSMENT

Graphical and statistical procedures will be adopted for evaluating the performance of the sediment transport model. Graphical comparisons will include:

- Time series plots of measured and computed suspended sediment concentrations - These plots will include data from a variety of sampling program components. For locations where intra-tidal suspended sediment data are available, time series on that time scale will be developed, in addition to inter-annual time-scales. Separate comparisons will be made for data collected from different depths in the water column.
- Spatial profiles of measured and computed suspended sediment concentrations along longitudinal transects (upstream to downstream) through the model domain - Data for these comparisons will come from the fixed station water-column sampling program, since that program will provide data at locations throughout the model domain on a consistent time frame. Separate comparisons will be made for data collected from different depths in the water column and data collected at different times.
- Probability plots of computed and measured suspended sediment concentrations for individual sampling locations - Separate comparisons will be made for data collected from different depths in the water column. These plots will include data from various sampling programs.

Map Document: (S:\Projects\PASSAIC\MapDocuments\4553001-CERCLA\Sedimentation_Bathymetry_Cesium_137_Dec_2005.mxd)
2/8/2006 -- 1:18:01 PM



Point Locations:
The average sedimentation rate (units of inches/year) equals the average of two cesium-137 calculations (whenever possible). Some isotope data were not used in the sedimentation rate calculations because of discontinuities in the cores.

- Plotted a downcore profile of cesium-137 concentration (pCi/g) versus depth, where "depth" equals the average of the top segment depth and the bottom segment depth. Nondetectable cesium concentrations were set to zero.
- Calculated two sedimentation rates: one at the 1963 time horizon (cesium-137 peak concentration) and one at the 1954 time horizon (base of the cesium-137 peak). Note that some cores only showed one time horizon; other cores showed neither time horizon.

Surface:
The sedimentation rate (units of inches/year) depicted as a surface was calculated based on the change in bathymetry from 1995 to 2001. The change of depth was divided by the 6-year period. Bathymetric survey data were from the 1995 TSI Survey and the 2001 TSI Survey. Sounding depths from both the 1995 and 2001 Surveys were converted from USACE Mean Low Water (MLW) to NGVD29 using a factor of 2.4 feet downstream of River Mile 6.8 and 2.3 feet upstream of River Mile 6.8.

A Triangulated Irregular Network (TIN) was derived from the survey points for each dataset using ESRI's 3-D Analyst in ArcGIS. Contours were interpolated from the TIN, also in 3-D Analyst. Each surface was converted to a raster with a 5-foot grid cell size. The change in depth was calculated by subtracting the 1995 raster surface from the 2001 raster surface.

Legend

Average Sedimentation Rate (inches / year)

- Scour
- 0
- 0 - 1
- 1 - 2
- 2 - 3
- 3 - 4
- 4 - 5
- > 5
- ● Core location with no data
- □ One sedimentation rate calculation
- △ Average of two sedimentation rate calculations
- ······· Bathymetric Transect Locations

Data Source
(1) The shoreline represented in this map is based on the shoretype dataset available from the New Jersey Department of Environmental Protection and shows a general depiction of the river boundary. The shoreline was further delineated by stereoscopic interpretation of aerial orthophotography. The aerial images used are a snapshot image of the New Jersey coastline and may not be high tide conditions. Some areas that may be submerged during high tide may have appeared as dry land.

(2) The sample locations shown on this map were from the Tierra Solutions Inc. 1995 dataset. Coordinates for each point were provided with the original study data, and these data points were uploaded to the PREmis database.

(3) Transect locations are based on the 2001 bathymetric transects. The 1995 transects align with the 2001 transects, with minor variations.

This (map/publication/report) was developed using New Jersey Department of Environmental Protection Geographic Information System digital data, but this secondary product has not been verified by NJDEP and is not state-authorized.

US Army Corps of Engineers

CESIUM-137 AND BATHYMETRY-BASED SEDIMENTATION RATES
River Mile 1 to River Mile 7

Lower Passaic River Restoration Project

FIGURE 3-4

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- Time-series plots of sediment-bed composition (fraction in each solids class) - even if data available for comparison are limited, temporal changes in computed composition will be evaluated qualitatively for consistency with the CSM.
- Spatial patterns in sediment-bed composition at different points in the simulation- these comparisons may also be limited to a qualitative assessment of consistency relative to the CSM.
- Temporal comparisons of sediment accumulation rates – Computed sediment accumulation rate in specific locations will be compared to estimates derived from bathymetric surveys and analyses of radionuclide profiles in sediment cores. Locations where similar sedimentation rate estimates were obtained from multiple methods (i.e. bathymetric changes, ^{137}Cs, ^{210}Pb) will be given more weight than locations where alternate estimates are more variable.
- Spatial patterns of sedimentation rates – Comparisons between the patterns indicated on and computed sedimentation rates will be used to assess the overall performance of the sediment transport model.
- Cross-plots of computed versus measured concentrations – These graphical displays will be developed and regression analyses will be used to compute a best–fit line and coefficient of determination (r^2) for each comparison. The coefficient of determination provides an indication of the fraction of the variance in the data that is explained by the model. The slope and intercept of the regression provide a means of assessing bias in the model performance.
- Analysis of residuals – Residuals (difference between model and data) and relative residuals (residual divided by data) will be plotted versus independent variables (e.g. river flow, stage, time of year) to assess bias associated with hydrodynamic forcing or seasonal patterns.

In addition to the graphical comparisons discussed above, which rely on qualitative judgment based on the modeler's understanding of the physical, chemical and biological characteristics of the system, the weight-of-evidence approach to model performance evaluation will be supported by quantitative metrics. Ultimately, the goal of model calibration and validation is "not to curve fit model to data, but to describe the behavior of the data with a modeling framework of the principal mechanisms relevant to the problem" (Thomann, 1982).

There are number of measures that can be used to quantitatively assess model goodness of fit. Many of these measures are described in detail along with a good discussion of overall model verification assessments in a number of journal papers (Thomann, 1982; Reckhow, et al., 1990). The following metrics will be evaluated for the sediment transport model:

- Model Bias: $= \overline{Y} - \overline{X}$;
- Relative Model Bias: $= \dfrac{\overline{Y} - \overline{X}}{\overline{X}}$;
- Mean Absolute Error: $= \dfrac{1}{n}\sum_{i=1}^{n}\left|Y_i - X_i\right|$;
- Median of Relative Error: $= \dfrac{(Y_i - X_i)}{X_i}$;
- where: Y = model, X = data, $\overline{Y}$ = average of model, $\overline{X}$ = average of data.

Graphical and quantitative comparisons will be developed for the 1995-2006 calibration period, as well as for results computed in the half-century hindcast for ^{137}Cs.

3.11 SENSITIVITY ANALYSES FOR THE SEDIMENT TRANSPORT MODEL

Sensitivity and uncertainty analyses are included in many modeling analyses to evaluate changes in model results in response to changes in model inputs. Sensitivity and uncertainty analyses serve different functions in the overall modeling process. Sensitivity analyses are used to quantify the magnitude of the response of model results to a change in a model input. By evaluating the relative response of model results to variations in input parameters, a sensitivity analysis can provide guidance for allocating resources for supplemental data collection intended to help refine model inputs. By understanding which parameters produce the greatest change in model results, efforts can be directed at those parameters that need to be assigned more accurately. Additional information, such as literature or previous experience, imposes important constraints on reasonable ranges used in the sensitivity analysis for a particular parameter.

Unlike the systematic changes in model inputs considered in a sensitivity analysis, uncertainty analyses generally try to consider how wide a range in model inputs might be reasonable and the effect that not knowing the precise value has on the results of the study. The results of an uncertainty can contribute to the evaluation the effectiveness of potential remedial actions. Formal uncertainty analyses, involving techniques such as Monte Carlo analyses, can require hundreds of runs, which is not practical given the long simulation periods required in contaminated sediment assessments. As an alternative, sensitivity analyses will be performed to accomplish the different objectives often split between sensitivity and uncertainty analyses. Variations in sediment transport model inputs that will be included in the sensitivity analyses will include:

- boundary conditions,
- parameters in the erosion formulation,
- parameters in the settling formulation.

An additional form of model sensitivity analysis that is planned is an analysis of the potential uncertainty in the specified solids loadings. The loading sensitivity will be conducted by varying boundary loadings, one at a time, and calculating the calibrated model's response to the loading change from an individual source. The model results can be stored in a spreadsheet-based unit response matrix that will enable users to scale the loadings and see estimate concentrations in the receiving water and sediments based on desired loading changes. The particular loading sources for which unit response will be calculated will be identified in consultation with USEPA, USACE and Malcolm Pirnie, Inc.

3.12 LINKAGE TO ORGANIC CARBON MODEL

Because particulate organic carbon (POC) and inorganic solids interact through coagulation processes in the water column and sediment bed processes of burial, armoring etc., the sediment transport calculation will be incorporated into the organic model calculation. Whereas, the sediment transport model provides the general features of solids distribution, it is the sediment transport-organic carbon model (ST-SWEM) in which sediment transport processes are directly incorporated into an entrophication model, that will provide at the same time carbon cycling and solids transport. As explained earlier, a preliminary calibration, based on inorganic solids, which represent 90 to 98% of the solids, will be performed first on the sediment transport model *per se*, before a final calibration is conducted using the sediment transport-organic-carbon model. A full description of the ST-SWEM is presented in the following section.

SECTION 4

SEDIMENT TRANSPORT-ORGANIC CARBON PRODUCTION MODELING

4.1 INTRODUCTION

Data collected as part of the SWEM calibration effort show that the water column of the Passaic River is very productive, particularly during the early spring and summer. Identifications of algal species throughout the Harbor-Bight-Sound complex indicate that a diatom bloom typically occurs in the early spring and that a bloom of green flagellates occurs in the summer. Absent Passaic River specific species identifications to the contrary, it is likely that the algal blooms in the Passaic River exhibit a similar seasonal functional group structure. Measurements of chlorophyll-a, an indicator of algal biomass, in the Passaic River, sometimes exceed 100 ug/L. Similarly, POC measurements in the Passaic have been observed to exceed 10 mg/L. DOC concentrations typically range between 4 and 6 mg/L. The nutrients that fuel algal growth in the Passaic River enter from a variety of sources (waters coming over the Dundee Dam and from the Saddle River, CSO and overland runoff, tidal exchange with other portions of the estuary, etc.). Algal growth in the Passaic River is generally not nutrient (i.e., N, P, Si) limited, but appears to be largely controlled by light and residence time in the photic zone. There have been, however, some observations in the late summer of dissolved silicate silica concentrations that approach levels limiting to algal growth. It is possible, however, that under future conditions algal growth may become nutrient limited as the USEPA Harbor Estuary Program is currently developing a nutrient TMDL in order to achieve compliance with water quality standards in New York/New Jersey Harbor and its tributary waters. This Total Maximum Daily Load (TMDL) may result in levels of point source and non-point source nutrient reductions, which may ultimately result in nutrient-control of algal growth in the Passaic River.

A screening level data analysis will be performed to demonstrate whether organic matter production and die-off and diagenesis in the sediment have significant effects on the fate of COPC to justify the need for a complex Organic Carbon production model.

4.2 SEDIMENT TRANSPORT-ORGANIC CARBON PRODUCTION MODEL PURPOSE AND OVERVIEW

The purpose of the sediment transport-organic carbon production model recommended for the Passaic River is to establish how organic carbon is being produced, removed and transported through the Passaic River. This is important because hydrophobic organic contaminants such as PCBs, dioxin/furans, pesticides and PAHs bind not to sediment per se but rather bind to POC and to a lesser extent DOC. Therefore, the fate and transport of organic carbon are important to understand the fate and transport of these hydrophobic chemicals. An organic carbon production and sediment diagenesis model of the Passaic River and contiguous waterways will also provide information on redox conditions, sulfate reduction rates, and sulfide concentrations which are critical in evaluating the fate and transport of mercury and the production of methyl mercury in sediments.

Previous organic carbon production modeling of the Passaic River has been performed as part of larger regional projects. These projects addressed nutrient management issues and toxic contamination. Both prior applications of organic carbon production modeling originate from the calibrated, validated, and peer-reviewed eutrophication model developed by HydroQual as part of the System-Wide Eutrophication Model (SWEM). SWEM has been used extensively by the New York City Department of Environmental Protection (NYCDEP) and the EPA NY/NJ Harbor Estuary Program (HEP). Since SWEM is the predecessor model to the suspended sediment transport/organic carbon production model planned for the Passaic River, some of the features of SWEM that will not be detailed in other sections of this modeling plan are described below.

4.2.1 System-Wide Eutrophication Model (SWEM) Background

SWEM was calibrated and validated against observed water and sediment quality data collected during two full annual cycles, the 12-month periods from October 1, 1994 to September 30, 1995 and from October 1, 1988 to September 30, 1989. The development, calibration, and validation of the SWEM eutrophication model are described in detail in a series of technical reports prepared by HydroQual for NYCDEP. Full citations for these reports are listed in the references section of this report (HydroQual, 1999a, b, c, d, e, f).

The peer-review process for SWEM development and application included both oversight by several modeling evaluation groups (MEGs), publication in a peer reviewed edited compilation (Miller et al., in press), and numerous technical presentations at national meetings of several professional societies. The sediment nutrient flux portion of SWEM has also been described previously (DiToro, 2001). A MEG, comprised of six members from the academic and modeling communities, was convened in 1994 by EPA HEP. This MEG met on three occasions and provided comprehensive review of the development of the SWEM and the supporting field program as well as the initial calibration of the model in the Harbor portion of the model domain. In 1997, a second MEG was convened by EPA HEP that consisted of four members. This MEG met on four occasions and provided comprehensive review of the calibration/validation of SWEM over the entire spatial domain. A third MEG was convened by the joint EPA HEP and Long Island Sound Study Nutrient Work Groups in 1999. This MEG met on four occasions and provided detailed review of the final model calibration/validation. In all three cases, the MEGs also evaluated the SWEM hydrodynamic model and the combined suitability of the hydrodynamic and water quality models for application to address nutrient management actions.

Prior to applying SWEM for CARP, additional enhancement of the SWEM calibration in the New Jersey tributaries was performed by HydroQual under oversight by New Jersey Department of

Environmental Protection (NJDEP) staff. Enhancement to SWEM in the New Jersey tributaries completed in July 2002 included refinements to loadings, vertical mixing coefficients, benthic filtration rates, nitrification rates, vertical light extinction coefficients, and temperature effects on algal growth. The enhancements both improved the overall level of calibration and/or made SWEM more defensible. The enhancements also included refinements to model grid geometry and several hydrodynamic parameters. A detailed description of this work appears in a technical report prepared by HydroQual for NJDEP that is available to Passaic River managing agencies and Technical Advisory Committee upon request. A full citation for this report is listed in the references section of this report (HydroQual, 2002).

During CARP, SWEM was upgraded to ST-SWEM. Specifically sediment transport calculations were incorporated directly into the organic carbon production model and as a result additional state variables were added to SWEM. A similar approach is planned for the Passaic River model; however, the sediment transport calculations incorporated in ST-SWEM for the Passaic River will be more sophisticated than those used for CARP as described above in modeling work plan section 2. It is necessary to incorporate sediment transport calculations within the organic carbon production model so that coagulation/settling processes which involve inorganic, organic, and living solids simultaneously may be properly accounted for.

The water quality model source code underlying both the CARP and SWEM applications that will be used for the Passaic River carbon and contaminant models is Row Column AESOP (RCA). RCA originates from the Water Analysis Simulation Program (WASP) developed by Hydroscience (HydroQual's predecessor firm) in the 1970's. RCA code has been used to develop numerous models outside of the NY/NJ Harbor region. The code has been constantly refined and upgraded to include both more realistic representations of the chemical and biological processes associated with eutrophication, and more robust numerical solution techniques. The code has evolved to include the capacity to interface directly with the outputs of hydrodynamic transport models. Since the early 1990's, HydroQual has maintained a users manual for the RCA code. An updated version of the users manual recently completed by HydroQual is available to the Passaic River managing agencies and Technical Advisory Committee upon request and includes a detailed description of the basic equations of the model, characteristics of the model, characteristics of the computer code, and descriptions of input and output files associated with the code.

Although the prior applications of the organic carbon production model in the Passaic River (i.e., SWEM and CARP) were successful in meeting their programmatic objectives, we believe additional refinements of the model beyond an upgrade of it sediment transport formulations will be required for purposes of the Passaic River Superfund Study. Specifically, our objectives for the Passaic River organic carbon production model are:

- to examine the effects of finer scale grid resolution on nutrient cycling, organic carbon distributions, oxygen concentrations, and sulfate reduction rates, and

- to develop more site-specific information on spatial and temporal distributions of organic carbon, algae, dissolved oxygen, sulfate reduction rates and sulfide concentrations for the water column and sediments in the Passaic River.

4.3 ORGANIC CARBON PRODUTION/SEDIMENT TRANSPORT MODEL FORMULATION

HydroQual's original approach in modeling sediment transport and organic carbon in NY/NJ Harbor (including the Passaic River) for CARP involved linking a sediment transport model (ECOMSED) to an organic carbon cycling model (SWEM). Some difficulties, however, were encountered in implementing this approach. First, sediment transport results that were passed forward from ECOMSED to SWEM caused mass conservation problems in SWEM (this was in part caused by the time-averaging scheme used in ECOMSED to pass information concerning settling and resuspension rates forward to RCA). Second, decoupling of sediment transport and organic carbon cycling in the proposed approach did not allow explicit consideration of interactions between inorganic and organic solids through coagulation processes. Therefore, a modified approach was developed by directly incorporating sediment transport into SWEM.

It is recommended that a similar strategy be repeated for the Passaic River model to avoid the issues described above associated with having separate sediment transport and carbon models. The Sediment-Transport version of SWEM (ST-SWEM), which will be used for the Passaic River modeling, includes both sediment transport and organic carbon cycling in the same framework. The sediment transport equations which will be incorporated into ST-SWEM for the Passaic River application have been described above in Section 3 of this modeling work plan. The organic carbon production equations incorporated into ST-SWEM are described below.

Like sediment transport, organic carbon transport in the Passaic River Superfund Study domain is dependent upon hydrodynamic flows, turbulent diffusion, settling, resuspension, and bed consolidation processes. In addition, the autochthonous production of organic carbon within the Passaic River Superfund Study domain is dependent upon availability of light and nutrients and residence time of algae in the photic zone.

The original SWEM included 24 state variables in the water column, which are described in detail in technical reports, prepared by HydroQual on SWEM (see references section) and are more briefly noted here (Table 4-1). As was noted earlier, while nutrients do not currently appear to limit phytoplankton growth in the Passaic River, they do appear to limit algal growth in other regions of the study domain and, therefore, may play a role in determining the concentrations of dissolved organic carbon and detrital particulate organic carbon influencing hydrophobic chemicals in the

Passaic River. Furthermore, as also mentioned earlier, the NY/NJ HEP is developing a nutrient TMDL that may result in a reduction of point source and non-point source nutrients, which may result in greater nutrient limitation on the study domain and more specifically the Passaic River. For this reason, the carbon production model to be used for the Passaic River needs to consider various nutrient forms (N, P, Si). In addition, the current state-of-the-science (ex., the USACE water quality model of eutrophication in Chesapeake Bay) considers both labile and refractory forms of particulate organic matter (C, N, P). This is important from the perspective of the sediment diagenesis/nutrient flux sub-model. The end result is the 24 state-variables listed in Table 4-1. The original SWEM also included a sediment nutrient flux sub-model, which contains state-variables in the sediment bed to account for diagenesis (including sulfate reduction and methanogenesis) and exchanges of nutrients and organic matter with the water column.

Table 4-1. 24 Water Column State Variables Included in SWEM

Salinity	ammonia nitrogen
winter phytoplankton carbon	nitrate and nitrite nitrogen
summer phytoplankton carbon	biogenic silica
refractory particulate organic phosphorus	available silica
labile particulate organic phosphorus	refractory particulate organic carbon
refractory dissolved organic phosphorus	labile particulate organic carbon
labile dissolved organic phosphorus	refractory dissolved organic carbon
dissolved inorganic phosphorus (DIP)	labile dissolved organic carbon
refractory particulate organic nitrogen	reactive dissolved organic carbon
labile particulate organic nitrogen	algal exudate dissolved organic carbon
refractory dissolved organic nitrogen	equivalents of aqueous dissolved oxygen demand (i.e., H_2S and CH_4)
labile dissolved organic nitrogen	dissolved oxygen

Note: Inert fractions of nutrients and organic carbon were not included in the SWEM water column because they do not contribute to the dissolved oxygen balance. These fractions were included in the SWEM sediment because they comprise a large portion of sediment concentrations. For purposes of ST-SWEM which considers resuspension, these fractions have been added to the water column. Inert material is continually resuspended to the water column and serves as an important sorbent phase for contaminants.

Figure 4-1 is a simplified diagrammatic representation of the principal eutrophication kinetics and water column-sediment interactions included in the original SWEM. The kinetics shown in Figure 4-1 have been described in detail (HydroQual 1999 a through f). Brief descriptions of the key features of primary production and sediment nutrient flux kinetics as shown in Figure 4-1 are presented.

4.3.1 Algal Growth

Phytoplankton growth in NY/NJ Harbor and Long Island Sound has been modeled for two functional groups or assemblages: winter diatoms and summer flagellates. Absent information to the contrary, it is likely that the phytoplankton of the Passaic River may also be characterized as two assemblages. The reason phytoplankton are considered as assemblages rather than as individual species is that at any particular time of the year there are literally tens of individual algal species present within the water column of the study domain. It is currently beyond the state-of-the-science in eutrophication modeling to include state-variables for each algal species since the growth rate of an individual population of phytoplankton in a natural environment is a complicated function of the species present and their differing reactions to solar radiation, temperature, and the balance between nutrient requirements and nutrient availability. This type of information is generally not known for many of the algal species present within New York/New Jersey Harbor waters.

4.3.2 Nutrient and Organic Carbon Cycling

Five forms of phosphorus, six forms of nitrogen, two forms of silica and six forms of organic carbon are included in the nutrient and carbon formulations in the original SWEM (for ST-SWEM, which will be used for the Passaic River Superfund Study, additional forms are included) as schematically shown on Figure 4-1. Inorganic phosphorus is utilized by phytoplankton for growth and is returned to various organic and inorganic forms via respiration and predation. A fraction of the phosphorus released during phytoplankton respiration and predation is in the inorganic form and is readily available for uptake by other viable phytoplankton. The remaining fraction is released in the dissolved and particulate organic forms. The organic phosphorus must undergo a mineralization or bacterial decomposition into inorganic phosphorous before it can be used by other viable phytoplankton.

During algal respiration and death, a fraction of the algal cellular nitrogen is returned to the inorganic pool in the form of ammonia. The remaining fraction is recycled to the dissolved and particulate organic nitrogen pools. Organic nitrogen undergoes a bacterial decomposition, the end product of which is ammonia. Ammonia nitrogen, in the presence of nitrifying bacteria and oxygen, is converted to nitrite nitrogen and subsequently nitrate nitrogen (nitrification). Both ammonia and nitrate are available for uptake and use in cell growth by phytoplankton; however, for physiological reasons, the preferred form is ammonia.

Two silica forms are considered. Available silica is dissolved and is utilized by diatoms during growth for their cell structure. Unavailable or particulate biogenic silica is produced from diatom respiration and diatom grazing by zooplankton. Particulate biogenic silica undergoes mineralization to available silica or settles to the sediment from the water column.

Pools of dissolved and particulate organic carbon are established on the basis of timescale for oxidation or decomposition. Zooplankton consume algae and take up and redistribute algal carbon to the organic carbon pools via grazing, assimilation, respiration, and excretion. Since zooplankton is not directly included in the SWEM kinetics, the redistribution of algal carbon to the organic carbon pools by zooplankton is simulated by empirical distribution coefficients. An

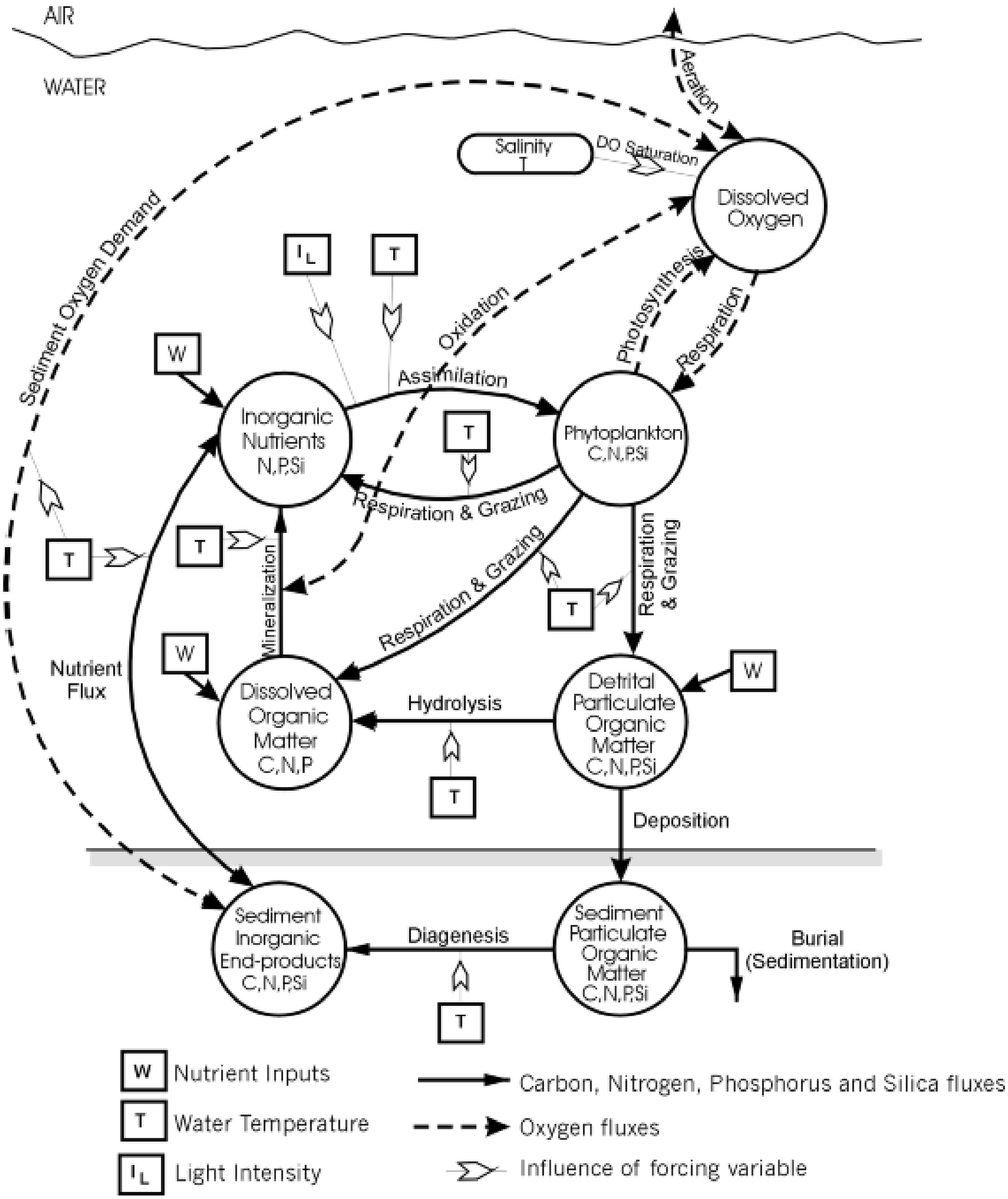


Figure 4-1. Principal kinetics and water column-sediment interactions for organic carbon production and sediment nutrient fluxes included in SWEM

additional term, representing the excretion of dissolved organic carbon by phytoplankton during photosynthesis, is included in SWEM. This algal exudate is very reactive. The decomposition of organic carbon is assumed to be temperature and bacterial biomass-mediated. Since bacterial biomass is not directly included within the SWEM framework, phytoplankton biomass is used as a surrogate variable. An additional loss mechanism of particulate organic matter is that due to filtration by benthic bivalves. This loss is handled in SWEM kinetics by increasing the deposition of non-algal particulate organic carbon from the water column to the sediment.

Although the number of dissolved and particulate pools of organic matter (including organic carbon) may appear greater than necessary for the purposes of modeling the fate and transport of hydrophobic contaminants, we believe it is easier to implement the SWEM model as it is currently calibrated rather than start over with a new modeling framework. In addition, recognizing the various reactivity pools of organic matter is essential to the framework incorporated in the sediment diagenesis model/nutrient flux sub-model, output of which is key to modeling rates of mercury methylation.

4.3.3 Dissolved Oxygen Balance

The dissolved oxygen balance includes both sources and sinks. Algal growth provides two of the sources: the production of dissolved oxygen from photosynthetic carbon fixation and an additional source of oxygen from algal growth when nitrate rather than ammonia is utilized. Atmospheric reaeration may be another source of dissolved oxygen, if the concentration of water column oxygen is less than dissolved oxygen saturation. Sinks of dissolved oxygen include algal respiration, nitrification, the oxidation of carbonaceous material, and sediment oxygen demand. Sediment oxygen demand (SOD) is the quantity of oxygen transferred from the water column to the sediment bed that is necessary to satisfy the oxygen requirements of bacteria in the sediment as they decompose previously deposited organic matter.

4.3.4 Sediment Dynamics

The mass balance equations of the SWEM sediment kinetics account for changes in particulate organic matter (carbon, nitrogen, phosphorus, and silica) in the sediments due to deposition from the overlying water column, sedimentation, and decay or diagenesis. The decay of particulate organic matter follows first-order kinetics as described by Berner (1971, 1974, and 1980) and is often referred to as the G model. The end products of diagenesis or decay of the particulate organic matter include ammonia nitrogen, dissolved inorganic phosphorus and dissolved inorganic silica. These end products can undergo additional biological, chemical, and physical processing within the sediment layer such as nitrification, sorption, and exchange with the overlying water column. Of particular importance to the overlying water column is the calculation of sediment oxygen demand, SOD. A more complete development of the SWEM sediment diagenesis model theory is presented elsewhere (Di Toro and Fitzpatrick 1993; Di Toro 2001). The sediment kinetics state variables include: temperature, labile particulate organic phosphorus (POP), refractory POP, slow refractory POP, labile particulate organic nitrogen (PON), refractory PON, slow refractory PON, labile particulate organic carbon (POC), refractory POC, slow refractory POC, biogenic silica, ammonia nitrogen, nitrate nitrogen, inorganic phosphorus, dissolved silica, and hydrogen sulfide. The latter variable considers sulfate reduction that we believe will be important to determining rates of mercury methylation.

4.3.5 Incorporating Sediment Transport into SWEM

The original SWEM model did not fully consider resuspension and erosion processes for particulate organic matter on a time variable basis (i.e., these were accounted for through net deposition and constant burial rates and were calibrated against exerted oxygen demand and water and sediment concentrations of the particulate organic matter).

For purposes of the CARP sediment transport/organic carbon production sub-model (and planned for the Passaic River model), settling, resuspension and burial of particulate organic carbon, nitrogen and phosphorus are determined as part of sediment transport calculations. Specifically, calculated settling rates are applied to both inorganic and organic particulate matter. In this approach, it is assumed that inorganic and organic particulate matter aggregate in the water column and are removed at similar rates as floc settle. Settling velocities for algae, however, are set independently due to their low rates of aggregation, i.e., low collision efficiencies. Time-variable resuspension and burial rates of bed material are also applied equally to inorganic and organic matter. With the addition of inorganic sediment and sediment transport to the original SWEM, ST-SWEM is the single model that was used for CARP to simulate both suspended sediments and organic carbon and will be used for the Passaic River model.

The ST-SWEM kinetics now include seven rather than six organic carbon variables in the water column to accommodate a detailed consideration of resuspension and erosion processes. The seven organic carbon state variables considered in ST-SWEM include: reactive dissolved organic (ReDOC), labile dissolved (LDOC), refractory dissolved (RDOC), labile particulate (LPOC), refractory particulate (RPOC), inert particulate organic carbon (IPOC), and dissolved algal exudate (ExDOC). Reactive, labile, refractory, and inert distinctions are based upon the time scale of oxidation or decomposition. Reactive organic carbon decomposes on a time scale of days to a week or two; labile organic carbon decomposes on the time scale of several weeks to a month or two; refractory organic carbon decomposes on the order of months to a year. Reactive and labile organic carbon decompose primarily in the water column or else rapidly in the sediments. Refractory organic carbon decomposes much more slowly, almost entirely in the sediments. Inert particulate organic carbon dominates the carbon present in sediments.

Table 4-2 presents the reaction rate terms for each of the organic carbon pools considered in the ST-SWEM framework. An additional loss mechanism of particulate organic matter is that due to filtration by benthic bivalves. This loss is handled in the model kinetics by increasing the deposition of non-algal particulate organic carbon from the water column to the sediment. Table 4-3 presents a summary overview of the organic carbon pools considered in ST-SWEM.

Table 4-2. Organic Carbon Reaction Equations

Labile Particulate Organic Carbon (LPOC)

$$LPOC = f_{LPOC} \cdot k_{grz}(T) \cdot P_c - k_{5,7}\theta_{5,7}^{T-20} \cdot LPOC.\frac{P_c + ReDOC + ExDOC}{K_{mP_c} + P_c + ReDOC + ExDOC}$$

$$-\frac{v_5}{H} \cdot POC + \frac{r_5}{H_{sed}} \cdot G_{1C}$$

(Note: Last term above applies only to layer 10)

Refractory Particulate Organic Carbon (RPOC)

$$RPOC = f_{RPOC} \cdot k_{grz}(T) \cdot P_c - \frac{v6}{H} \cdot RPOC - k_{6,8}\theta_{6,8}^{T-20} \cdot RPOC.\frac{P_c + ReDOC + ExDOC}{K_{mP_c} + P_c + ReDOC + ExDOC}$$

$$+\frac{r_6}{H_{sed}} \cdot G_{2C}$$

(Note: Last term above applies only to layer 10)

Inert Particulate Organic Carbon (IPOC)

$$IPOC = \frac{v_7}{H} \cdot IPOC + \frac{r_7}{H_{sed}} \cdot G_{3G}$$

(Note: Last term above applies only to layer 10)

Inert Particulate Organic Carbon (IPOC)

$$IPOC = -\frac{v_7}{H} \cdot IPOC + \frac{r_7}{H_{sed}} \cdot G3C$$

(Note: Last term above applies only to layer 10)

Labile Dissolve Organic Carbon (LDOC)

$$LDOC = f_{LDOC} \cdot k_{grz}(T) \cdot P_c + k_{5,7}\theta_{5,7}^{T-20} \cdot LPOC \cdot \frac{P_c + ReDOC + ExDOC}{K_{mP_c} + P_c + ReDOC + ExDOC}$$

$$- k_{7,0}\theta_{7,0}^{T-20} \cdot LDOC \cdot \frac{LDOC}{K_{mLDOC} + LDOC} \cdot \frac{DO}{k_{DO} + DO} \cdot \frac{P_c + ReDOC + ExDOC}{K_{mP_c} + P_c + ReDOC + ExDOC}$$

$$- \frac{5}{4} \cdot \frac{12}{14} \cdot K_{DN}\theta_{DN} \cdot NO_x \cdot \frac{K_{NOX}}{K_{NOX} + DO} \cdot \frac{LDOC}{K_{mLDOC} + LDOC}$$

Refractory Dissolved Organic Carbon (RDOC)

$$RDOC = f_{RDOC} \cdot k_{grz}(T) \cdot P_c - k_{8.0}\theta_{8,0}^{T-20} \cdot RDOC \cdot \frac{P_c + ReDOC + ExDOC}{K_{mP_c} + P_c} \cdot \frac{DO}{K_{DO} + DO}$$

$$+ k_{6,8}\theta_{6,8}^{T-20} \cdot RPOC \cdot \frac{P_c + ReDOC + ExDOC}{K_{mP_c} + P_c + ReDOC + ExDOC}$$

Reactive Dissolved Organic Carbon (ReDOC)

$$REDOC = - k_{9,0}\theta_{9,0}^{T-20} \cdot ReDOC \cdot \frac{ReDOC}{K_{mLDOC} + ReDOC} \cdot \frac{DO}{K_{DO} + DO} \cdot \frac{P_c + ReDOC + ExDOC}{K_{mP_c} + P_c + ReDOC + ExDOC}$$

Algal Exudate Dissolved Organic Carbon (ExDOC)

$$ExDOC = f_{ExPP} \cdot G_P \cdot P_c$$

$$- k_{10,0}\theta_{10,0}^{T-20} \cdot ExDOC \cdot \frac{ExDOC}{K_{mLDOC} + ExDOC} \cdot \frac{DO}{K_{DO} + DO} \cdot \frac{P_c + ReDOC + ExDOC}{K_{mP_c} + P_c + ReDOC + ExDOC}$$

Table 4-2 - Organic Carbon Reaction Equations (Continued)

Description	Notation	Units
Phytoplankton Biomass	P_c	mgC/L
Specific Phytoplankton Growth Rate	G_p	day^{-1}
Half Saturation Constant for Phytoplankton Limitation	K_{mPc}	mgC/L
Half Saturation Constant for LDOC	K_{mLDOC}	mgC/L
Fraction of Grazed Organic Carbon Recycled to:		
the LPOC pool	f_{LPOC}	
the RPOC pool	f_{RPOC}	
the IPOC pool	f_{IPOC}	
the LDOC pool	f_{LDOC}	
the RDOC pool	f_{RDOC}	
Fraction of Primary Productivity Going to the Algal Exudate DOC pool	f_{Expp}	
Hydrolysis Rate for RPOC	$k_{6,8}$	day^{-1}
Temperature Coefficient	$\theta_{6,8}$	
Hydrolysis Rate for LPOC	$k_{5,7}$	day^{-1}
Temperature Coefficient	$\theta_{5,7}$	
Settling Rate of LPOC	v_5	m/day
Settling Rate of RPOC	v_6	m/day
Settling Rate of IPOC	v_7	m/day
Resuspension Rate of G1C	r_5	m/day
Resuspension Rate of G2C	r_6	m/day
Resuspension Rate of G3C	r_7	m/day
Water Column Segment Depth	H	m

Table 4-2 - Organic Carbon Reaction Equations (Continued)

Description	Notation	Units
Sediment Segment Depth	H_{SED}	m
Oxidation Rate of LDOC	$k_{7,0}$	day^{-1}
Temperature Coefficient	$\theta_{7,0}$	
Oxidation Rate of RDOC	$k_{8,0}$	day^{-1}
Temperature Coefficient	$\theta_{8,0}$	
Oxidation Rate of ReDOC	$k_{9,0}$	day^{-1}
Temperature Coefficient	$\theta_{9,0}$	
Oxidation Rate of ExDOC	$k_{10,0}$	day^{-1}
Temperature Coefficient	$\theta_{10,0}$	
Half Saturation for Oxygen Limitation	K_{DO}	mgO_2/L
Dissolved Oxygen	DO	mgO_2/L
Denitrification Rate	K_{DN}	day^{-1}
Temperature Coefficient	θ_{DN}	
Nitrate + Nitrite	NO_X	mg N/L
Half Saturation Constant for Denitrification	K_{NOX}	mgO_2/L

Table 4-3. Organic Carbon Forms Included in ST-SWEM

		WATER COLUMN		SEDIMENT BED	
PHASE	POOL	SOURCES	SINKS	SOURCES	SINKS
Living Algae	Diatoms	external sources growth	settling respiration zooplankton grazing benthic filtration	NA	NA
	Greens	external sources growth	settling respiration zooplankton grazing benthic filtration	NA	NA
POC	Inert G3	resuspension grazed algae	settling benthic filtration	settling benthic filtration 15% of dead algae/POM deposition	resuspension burial mineralization/diagenesis
	Refractory G2	external loadings grazed algae resuspension	hydrolysis to DOC settling benthic filtration	settling benthic filtration 20% of dead algae/POM deposition	resuspension burial mineralization/diagenesis
	Labile G1	external loadings grazed algae resuspension	hydrolysis to DOC settling benthic filtration	settling benthic filtration 65% of dead algae/POM deposition	resuspension burial mineralization/diagenesis
DOC	Refractory	external loadings grazed algae from refractory POC	oxidation	NA	NA
	Labile I	external loadings grazed algae from labile POC	oxidation denitrification	NA	NA
	Labile II	external loadings	oxidation	NA	NA
	Exudate	algal exudation	oxidation	NA	NA

Table 4-4. Sediment Sub-Model Coefficients

Description	Notation	Units
<u>Physical Related</u>		
Water column-sediment layer temperature diffusion coefficient	D	cm^2/sec
depth of active sediment layer	H_2	cm
deposition velocity at 20° C for:	v_{dep}	
phytoplankton		m/day
non-phytoplankton POM	w_s	m/day
sedimentation velocity	v_{sed}	cm/yr
resuspension velocity	W_r	cm/yr
<u>Diagenesis Related</u>		
G1 diagenesis decay rate at 20°C	k_{diag1}	day^{-1}
temperature correction coefficient	θ_1	
G2 diagenesis decay rate at 20°C	k_{diag2}	day^{-1}
temperature correction coefficient	θ_2	
G3 diagenesis decay rate at 20°C	k_{diag3}	day^{-1}
temperature correction coefficient	θ_3	

	Labile	Refractory	Slow Refractory
G-Model Fraction Splits			
<u>Phosphorus</u>			
phytoplankton group 1	0.65	0.20	0.15
phytoplankton group 2	0.65	0.20	0.15
<u>Nitrogen</u>			
phytoplankton group 1	0.65	0.25	0.10
phytoplankton group 2	0.65	0.25	0.10
<u>Carbon</u>			
phytoplankton group 1	0.65	0.20	0.15
phytoplankton group 2	0.65	0.20	0.15

4.4 ORGANIC CARBON PRODUCTION MODEL INPUTS

This section provides a review and summary of the principal inputs of nutrients and oxygen demanding material to NY/NJ Harbor, Long Island Sound and the New York Bight required for the Passaic River Superfund Study model. These inputs are comprised of:

- municipal WPCP and industrial discharges,
- fall-line tributary loadings,
- combined sewer overflow (CSO) loadings,
- nonpoint source loadings from rainfall runoff (SW),
- atmospheric loadings falling directly on the water surface.

The databases, methodologies, and variability associated with each of these inputs are discussed in the following paragraphs. In general, ST-SWEM requires loadings of dissolved and particulate forms of nitrogen, phosphorus, silica, and carbon as well as reactivity classes. Loadings have historically been estimated on a monthly average basis for WPCPs and direct atmospheric deposition. Loadings for fall-line tributary inputs, CSOs, and SW in prior applications were estimated on an hourly to daily basis. WPCPs and tributary inputs, which are concentrated in the Harbor and Sound portions of the domain have for other applications represented the bulk of the loading. Our experience has been that atmospheric loadings are large but are distributed over the broad expanse of the NY Bight.

4.4.1 Fall-Line Tributary Nutrient Inputs

Fall-line tributary inputs represent loadings of water quality constituents, which are delivered from upland watersheds to the tidal Harbor/Sound/Bight system. These up-basin loadings result from ground water inflows, surface land runoff, direct atmospheric deposition to upland waters, and wastewater discharges to upland streams. ST-SWEM considers the delivery of these loads to the Harbor/Sound/Bight system via the following streams: Hudson River, Hackensack River, Passaic River, Saddle River, Raritan River, South River, Normans Kill, Moordener Kill, Esopus Creek, Rondout Creek, Wappinger Creek, Croton River, Sawmill River, Bronx River, Navesink and Shrewsbury Rivers, Catskill Creek, Norwalk River, Housatonic and Naugatuck Rivers, Quinnipiac River, Connecticut River, Thames River, Manasquan River, Metedeconk River, Toms River, Mullica River, Tuckahoe River, Great Egg River, and Westecunk Creek. To minimize model simulation time, in the water quality sub-model of previous versions of SWEM, the Hudson River, Normans Kill, Moordener Kill, Esopus Creek, Rondout Creek, and Catskill Creek discharge volumes were summed and assigned as a single input, the Hudson River near Poughkeepsie, New York. For purposes of CARP and ST-SWEM where a detailed understanding of the Hudson River above Poughkeepsie is of importance for several of the contaminants of concern, the Hudson River, Normans Kill, Moordener Kill, Esopus Creek, Rondout Creek, and Catskill Creek discharge volumes were all handled as separate tributary inputs. It is anticipated that for purposes of the Passaic River model, the calculations in the Hudson River will be truncated at Poughkeepsie. For CARP, the discharge inputs from the Rahway and Elizabeth Rivers are included as part of the runoff model described subsequently. For purposes of the Passaic River Superfund study, the Rahway and Elizabeth Rivers may also be considered as tributary inputs as well as the Frank's Creek, Lawyers Creek, the Second River, and the Third River.

To assign the fall-line tributary inputs in ST-SWEM, both discharge and quality have to be specified. Discharge data were compiled from USGS surface water records for New York, New Jersey, and Connecticut on a daily basis as part of the development of the CARP hydrodynamic sub-model. Tributary concentration data for individual water quality constituents collected during the 1994-95 monitoring program in support of the original SWEM for nine tributaries were used to assign concentrations for the fall-line tributary inputs on a monthly average basis. For the tributaries not monitored, concentration values were assigned based on the measured rivers. For purposes of the Passaic River, it is assumed that USGS discharge records will continue to be available. As for nutrient related concentration data, where/when available, data collected at tributary headwaters as part of the Passaic River Superfund Study monitoring program will be used to supplement previous nutrient concentration loading estimates.

4.4.2 WPCP Nutrient Inputs

Major municipal and industrial WPCPs discharging into the Harbor/Bight/Sound system are included in ST-SWEM. For each facility both discharge flow and individual constituent concentrations are specified as mass loadings (kg/day) on a monthly average basis. The mass loadings assigned in ST-SWEM were developed primarily from 1994-95 discharge monitoring report (DMR) data obtained from the USEPA Permit Compliance System (PCS). The DMR data were supplemented with data collected at the municipalities during the monitoring program conducted in support of SWEM in 1994-95. Specifically, DMR's do not contain effluent organic carbon or all of the nutrient forms required for SWEM input.

WPCP monitoring data collected during the SWEM 1994-95 field program were used to develop correlations between effluent BOD5 reported on DMR's and effluent DOC and POC both on a plant specific and average plant basis. For example, the following regressions were developed for converting BOD5 data reported on DMR's to DOC and POC required by ST-SWEM:

$$POC = 4.68 + 0.31(BOD5)$$

$$DOC = 9.98 + 0.26(BOD5)$$

where POC, DOC, and BOD5 are in mg/l.

For purposes of the Passaic River Superfund Study, updated DMR's will be obtained from EPA's Permit Compliance System and used to revise WPCP loadings for the years selected for modeling.

4.4.3 Combined Sewer Overflow (CSO) and Storm Water Runoff (SW) Nutrient Loadings

For CARP, CSO and SW volumes were generated on an hourly to daily basis using RRMP, a rainfall-runoff model developed for the New York City 208 Study by Hydroscience (Di Toro et al., 1978) and available calibrated Storm Water Management Models (SWMM) for various jurisdictions. These models calculate discharges for 268 land parcels in the NY/NJ Harbor Estuary area given: the hourly rainfall from regional airports or from local rain gauges, the drainage area of the parcel, land use, and the runoff flow captured by WPCPs if applicable. RRMP distinguishes between seven land use categories within each land parcel including: low density residential, middle density residential, high-density residential, commercial, industrial, parks and cemeteries, and large institutions. Each land use category has characteristic runoff coefficients. RRMP and SWMM simulations were performed for a unit rainfall which was then scaled according to the actual rainfall record for each of the 6 water years considered for CARP: 1988-89, 1994-95, 1998-99, 1999-2000, 2000-01, and 2001-02. For areas of Long Island Sound beyond the domains of RRMP and SWMM but within the CARP model domain, runoff loadings were assigned based on runoff loads developed during the Long Island Sound Study.

CSO and SWO nutrient concentrations were assigned using data collected during the SWEM 1994-95 monitoring program. Due to the highly variable nature of CSO and SW quality and the limited fraction of the total possible locations sampled, log mean concentrations of the data were used.

The log mean concentrations assigned for CSO and SW in the CARP model are tabulated below in Table 4-5.

It is intended that the CARP approach for generating CSO and SW nutrient loadings for ST-SWEM will be repeated. The appropriate rainfall records will be obtained for the specific years included in the Passaic River Superfund model. As additional SWMM models become available to HydroQual through efforts to upgrade the landside models for EPA's HEP TMDL program, they will be incorporated in the landside loading generation effort for the Passaic River Superfund Study.

4.4.4 Atmospheric Nutrient Inputs

Deposition of nitrogen, silica, phosphorus, and carbon resulting from direct precipitation to surface waters and dry fall are included in ST-SWEM as atmospheric inputs. Estimates of these loadings are based on atmospheric deposition data collected during the SWEM 1994-95 monitoring program, for the 1988-89 Long Island Sound Study, and by the University of Connecticut in the early 1990's.

Table 4-5. Concentrations Assigned to CSO and SW for ST-SWEM Calibration		
	CSO	SW
Phosphorus		
POP	0.697 mg P/l	0.090 mg P/l
DOP	0.130 mg P/l	0.019 mg P/l
DIP	0.596 mg P/l	0.084 mg P/l
Nitrogen		
PON	3.02 mg N/l	0.372 mg N/l
DON	1.63 mg N/l	0.404 mg N/l
NH_4	4.44 mg N/l	0.236 mg M/l
$NO_2 + NO_3$	0.492 mg N/l	0.765 mg N/l
Silica		
DSi	1.71 mg Si/l	1.77 mg Si/l
Carbon		
POC	41.5 mg C/l	7.32 mg C/l
DOC	18.7 mg C/l	8.81 mg C/l
Oxygen		
DO	3.8 mg O_2/l	6.33 mg O_2/l

The SWEM monitoring program included ten stations over the period November 1994 through June 1995. Concentrations in precipitation of DOC, PO_4, NH_4, NO_2 + NO_3, SiO_4, DON, and DOP were measured. Due to the limited temporal and spatial scope of the monitoring program, data from all ten stations were combined and analyzed by constituent. For each month for each analyte, a maximum likelihood estimate (MLE) was calculated and combined with precipitation data to assign a monthly average load on a mass per square meter per day basis. For months during which concentrations were not measured, a MLE was calculated from the data for the entire monitoring program for each analyte.

The USGS collected atmospheric wet deposition data at four sites in the LISS area between August 1988 and December 1989, Greenwich, Connecticut; Old Field, New York; Clinton, Connecticut; and Block Island, Rhode Island. Constituent concentrations reported include total dissolved nitrogen, ammonia nitrogen, nitrate nitrogen, total dissolved phosphorus, dissolved inorganic phosphorus, and total organic carbon. Dissolved organic phosphorus was estimated as the difference between total dissolved phosphorus and dissolved inorganic phosphorus, while dissolved organic nitrogen was estimated by subtracting ammonia and nitrate nitrogen concentrations from the total dissolved nitrogen concentration, an approximation which assumes that nitrite nitrogen concentrations were negligible. Due to the wide variation in the data and the limited scope of the sampling program, data from all four stations were combined. The data for each constituent were found to be log normally distributed. Where measurements fell below detection limits, such as with TDP and PO4, assumption of a log normal distribution below the detection limit enabled estimation of the actual concentration distribution. The most likely estimate (MLE) concentration for each constituent was calculated.

From 1991 to 1993, wetfall and dryfall data were collected weekly by the University of Connecticut at Storrs at two stations (Miller et al., 1993). These stations, Sherwood Island State Park in Westport CT and Hammonasset State Park in Madison Ct, are located close to Long Island Sound. Analytes included sulfate, sulfur dioxide, ammonia, nitrate, nitric acid vapor, total dissolved nitrogen and total phosphate. Based on the dryfall measurements collected at the two 1991-1993 stations, daily dryfall loadings of NH_4-N (981 lbs/day) and NO_2+NO_3-N (7184 lbs/day) were input at a constant rate for all months of model simulation.

Absent any new or more site specific information for the Passaic River, the approach will be repeated. A possible source of atmospheric nutrient deposition data are the New Jersey Atmospheric Deposition Network (NJADN) based at Rutgers University. We believe they have information on nitrogen deposition to the NY/NJ Harbor for the late 1990's/early 2000's timeframe.

4.4.5 Reactivity Data for Nutrient Loadings

ST-SWEM is a carbon-based model as opposed to a BOD based model. ST- SWEM incorporates seven forms of organic carbon. Similarly, ST-SWEM incorporates five forms each of organic phosphorus and organic nitrogen. Data collected during the SWEM 1994-95 monitoring program and available on WPCP DMR's do not provide guidance for the specification of the various organic forms which are reactivity and phase (particulate or dissolved) dependent. The reactivity classes of organic carbon include, in order of decreasing reactivity, are reactive, labile, refractory, and inert. The reactivity classes of organic phosphorus and nitrogen include labile, refractory, and inert. The reactivity classes are distinguished from one another by the relative rates of decomposition. Reactive organic matter decomposes rapidly, on the order of days. Labile organic matter decomposes on the order of weeks, while refractory and inert organic matter decomposes on the order of several months to years or longer.

Splits for the reactive classes of organic loadings were assigned in ST-SWEM for previous applications in the Harbor on the basis of an analysis of data collected in the spring and summer of 1994 for the Interstate Sanitation Commission. The Interstate Sanitation Commission reactivity study included monitoring at sites that represent the major inputs of nutrients to the system. The major sources included: 21 WPCPs, 6 CSOs, 4 SW sites, and 4 tributaries. Each source was sampled twice. The reactivity samples were incubated for 50 days and sub-samples were taken at 10-day intervals for most analytes. Analytes measured during the reactivity study include: POC, DOC, PON, DON, POP, DOP, NH_4, $N0_3$+NO_2, dissolved reactive PO_4, DSi, biogenic or particulate Si, and BOD.

Once the loadings for ST-SWEM are developed for the Passaic River Superfund Study, actual ST-SWEM simulations will be performed and model and data comparisons will be made to assess the level of calibration/validation achieved by an individual ST-SWEM simulation. The skill assessment for the nutrient portion of ST-SWEM using the loading data described above is the subject of next section of this modeling work plan document.

4.5 MODEL CALIBRATION

This project will build upon the enhancement of the System-Wide Eutrophication Model (SWEM) that was performed for New Jersey Department of Environmental protection in July 2002. The New Jersey tributaries component of the SWEM model was calibrated against 1994-95 data with additional comparison of model results to limited 1988-89 data. Although the New Jersey tributaries component of the original SWEM model was improved, it was acknowledged that model grid resolution and data limitations prevented a satisfactory model calibration for the New Jersey tributaries. In particular, the laterally averaged segmentation in the Passaic and Hackensack River limited the ability of the hydrodynamic model to capture secondary currents and small-scale bathymetric features. The calibration of the chlorophyll a and organic carbon components of the water quality model were significantly limited by adequate light extinction data.

To provide a well-calibrated eutrophication model of the Passaic River-Newark Bay watershed, it is anticipated that chlorophyll-a, organic carbon, nutrient and light data may need to be collected by the Lower Passaic River Restoration Project and Newark Bay Study. It is proposed to conduct 4 to 8 spatial surveys of this region with a total of 15 to 20 stations distributed between the Passaic River, Hackensack River, Newark Bay, the Arthur Kill and Kill van Kull. The surveys would be conducted concurrently with the sampling for the chemicals of potential concern. The eutrophication model will be calibrated against chlorophyll-a, organic carbon (soluble and particulate), the nitrogen series (organic, ammonia, and nitrite plus nitrate) phosphorus (organic and phosphate), silica (dissolved and biogenic), BOD, and dissolved oxygen. The primary data set for model calibration will be this new data; however, model results will be compared to the 1994-95 data and possibly other comprehensive data collected by other agencies.

The goodness of model calibration will be assessed by a combination of graphical displays and statistical analyses. Both model and data will be plotted along spatial transects for each of the surveys. In addition, temporal plots of model and data for all measured constituents will be developed for each of the 15 to 20 sampling stations. Other graphical comparisons will include computed concentration versus measured concentration for each constituent with stations grouped by geographical regions (Passaic River, Hackensack River, Arthur Kill and Kill van Kull). An additional graphical comparison will compare the probability distributions of data and model for each water quality constituent for these same geographical regions. Statistical comparisons may include mean error, mean absolute error, and relative mean error.

4.6 LINKAGE TO CONTAMINANT FATE AND TRANSPORT MODEL

In general, the Passaic River combined sediment transport/organic carbon production sub-model will calculate concentrations of suspended sediment, particulate organic carbon, and dissolved organic carbon over time and in longitudinal/lateral space in ten vertical layers of the water column and in the sediment bed in a 10 cm active layer including aerobic and anaerobic zones and in an anaerobic archive (a.k.a., archival stack). The archival stack is dynamically computed and depends on a balance between the rate of deposition of organic matter from the water column and the rate of resuspension of organic matter from the sediment bed. The carbon is type identified based on its reactivity.

Correct calculation of suspended sediment and organic carbon concentrations and vertical transport rates of carbon, in particular, is needed for calculating concentrations of contaminants bound to particles. Information calculated by the sediment transport/organic carbon production sub-model and specifically passed to the contaminant fate and transport sub-model is described below in greater detail.

4.6.1 General Information Passed to Carp Fate and Transport Sub-Models

The sediment transport/organic carbon production sub-model produces a relatively large output file (e.g., approximately 7.4 gigabytes per year for the CARP model) specifically for the contaminant fate and transport sub-models. The output file includes as time histories in three dimensions of the calculated (i.e., one hour average) water column phytoplankton settling rate and phytoplankton biomass; refractory, labile, and inert particulate organic carbon concentrations; refractory and labile dissolved organic carbon concentrations; average light intensity; settling rate for particulate organic carbon; and hydrogen sulfide concentrations. For the sediment bed, the output file includes as time histories in two dimensions the calculated (i.e., one hour average) diffusive and particle mixing rates; resuspension and burial rates for the active sediment bed; erosion rates from the sediment bed archival stack to the active sediment bed; rates of change of the depth of the active sediment bed; concentrations of G_1 (labile), G_2 (refractory), and G_3 (inert) carbon in the active sediment bed; depths and rates of change in depth of the archival stack sediment bed; and concentrations of G_1, G_2, and G_3 carbon in the archival stack sediment bed. This output file is read by the CARP contaminant fate and transport sub-models for both hydrophobic organic chemicals (HOCs) and metals. The linkages between the carbon-production sub-model and the contaminant fate and transport model have been already developed and verified to be working.

4.6.2 Additional Information Passed to the Metals Fate and Transport Sub-Model

In addition to the information passed from the sediment transport/organic carbon production sub-model to the contaminant fate and transport sub-models for HOCs and metals via the output file described above, supplemental outputs from the sediment nutrient flux model, including sediment bed concentrations of hydrogen sulfide and sulfate and sulfate reduction rates required for use in mercury methylation computations, are passed as two-dimensional time histories specifically to the metals fate and transport sub-model.

SECTION 5

DEVELOP CONTAMINANT FATE AND TRANSPORT MODEL

5.1 CONTAMINANT FATE AND TRANSPORT MODEL PURPOSE

The purpose of the contaminant fate and transport model is to develop a tool that permits an understanding of the fate and transport of contaminants within the Passaic River, as well as the export to or import from Newark Bay and other portions of the NY/NJ Harbor Estuary. An important feature of the contaminant fate and transport model is its predictive capabilities. It is being developed for purposes of relating future conditions resulting from specific management and remedial actions identified by Lower Passaic River Restoration Project to expected contaminant levels in receiving waters and sediments over time in the future. The remedial actions identified by Lower Passaic River Restoration Project are intended to achieve reduced risk to human health and other ecological receptors.

HydroQual will develop a contaminant fate and transport model for contaminants of concern identified for Lower Passaic River Restoration Project. The contaminant fate and transport model will be compatible with and reliant upon the hydrodynamic, sediment transport, and organic carbon production models developed for the project. The contaminant fate and transport model will be calibrated to contaminant measurements collected in the water column and sediments of the Passaic River and contiguous waterways. The contaminant fate and transport model will be analogous in structure to contaminant fate and transport model used for CARP but will take advantage of higher grid resolution and more refined hydrodynamics, sediment transport and organic carbon production calculations. Once calibrated, the contaminant fate and transport model will be used to drive foodchain calculations (described subsequently in Section 6). Specific features of the planned contaminant fate and transport model for Lower Passaic River Restoration Project are described in detail below.

5.2 IDENTIFY CONTAMINANTS OF CONCERN

Sediments underlying the Passaic River and the adjacent waters of the NY/NJ Harbor Estuary are contaminated with a wide variety of hydrophobic organic chemicals (HOCs) and metals. Large-scale sampling programs such as R-EMAP and CARP have attempted to represent the broad spectrum of contaminants present. The CARP analyte list, for example, includes 27 pesticides, 209 PCB congeners, 17 dioxin/furan congeners, 3 metals, and 21 PAH compounds. For purposes of CARP modeling, the analyte list was reduced to include: several major chlordane compounds, six DDT/DDE/DDD's, 10 PCB homologs, several individual PCB congeners which exhibit dioxin-like toxicity, 17 dioxin/furan congeners, 2 metals, and a subset of the 21 PAH compounds. The R-EMAP analyte list includes 23 PAH compounds, six DDT/DDE/DDDs, 10 other chlorinated pesticides, 4 major and 12 trace elements, 20 PCB congeners, and 16 dioxin/furan congeners. As reported by Chaky 2003, major historical production/release of contaminants adjacent to the Passaic River is known to include 2,3,7,8-TCDD, DDT, hexachlorobenzene, and lindane/ low γ-BHC (i.e., an isomeric form of benzene hexachloride).

Lower Passaic River Restoration Project contaminants of concern will be defined by EPA and partner agencies, based on the needs of the risk assessments and WRDA goals. However, finalizing the list of contaminants of concern to be modeled for the Passaic River may also involve several technical decisions based on the state-of-the-science. For example, regarding chlordane related contaminants, there are dramatic transformations between chlordane compounds in sediments (typically dominated by α and ϒ chlordane; similar to technical chlordane) and fish (dominated by the nonachlors and oxychlordane). Since TAC members have indicated that details of the transformations are poorly understood, modeling the chlordane related contaminants might require using empirical relationships based on site-specific data. Further, a fact sheet on chlordane available from Cornell University's Pesticide Management Education program (PMEP) indicates that available data are insufficient to fully assess the environmental fate of chlordane. Our cursory knowledge of the chlordane related contaminants indicates that, with the exception of oxychlordane, the chlordane related contaminants have similar octanol-water partition coefficients and Henry's constants. There are similar technical issues associated with selection of other contaminants.

An important decision regarding the contaminants of concern relates to PCBs. It is more technically defensible to model PCBs as homologs rather than as a total sum given the differences in behavior in the environment of the congeners between homolog groups. Although we plan on modeling homologs, it is essential that PCB congeners be measured for two reasons. If there is evidence that dechlorination of PCB congeners is occurring in the Passaic River or contiguous areas, a consideration of PCB measurements at the congener level may become necessary. Further, for toxicity reasons, it might be necessary to consider the coplanar PCB's as individual chemicals.

5.3 SELECT CONTAMINANT FATE AND TRANSPORT MODEL KINETICS AND FORMULATIONS

As described in greater detail below, the kinetic structure of the Lower Passaic River Restoration Project contaminant fate and transport model will be patterned after the CARP contaminant fate and transport model; however, there will be several decision points for upgrading and/or expanding the CARP kinetics.

5.3.1 HOC Water Column Kinetics

Partitioning, chemical transformations, and matrix (i.e., air, water, and sediment) transfers of HOCs will be modeled.

5.3.1.1 Partitioning

Partitioning of HOCs among freely dissolved, DOC-bound, and particulate phases will be described by equilibrium relationships for both DOC and POC. The equilibrium relationships may be governed by contaminant specific octanol-water partition coefficients or by specifying field derived partition coefficients specific to the Passaic River Superfund Study area. Partitioning of contaminant, on an equilibrium basis, between freely dissolved, DOC-bound, and particulate phases (i.e., three phase partitioning) is an important and complex element of the Passaic contaminant fate and transport model. It is acknowledged that there are other approaches to modeling contaminant phase partitioning (e.g., two phase, reversible, phosphorus dependent). The following discussion addresses some of the advantages/disadvantages of these alternative approaches.

Simply considering partitioning between dissolved and particulate phases (i.e., two phase partitioning), while sufficient to explain observed measurements of dissolved and particulate contaminants, would not be adequate (i.e., would be an over estimate) for purposes of addressing bioavailability of the dissolved contaminant phase.

In an estuarine system such as the Passaic River where particles experience frequent and repeated tidal resuspension, it is unlikely that a consideration of adsorption-desorption processes (i.e., reversible partitioning) would yield dramatically different non-instantaneous results (i.e., within a factor of 2 or less) than equilibrium partitioning yields. Some of Joel Baker's recent experiments (presented in a Hudson River Foundation seminar on December 8, 2004) in which Upper Hudson River bed sediments are resuspended multiple times in succession with a diminished increase in release of contaminant over successive events, support this conclusion. Given the data requirements to model adsorption-desorption processes and the magnitude of other uncertainties within the contaminant fate and transport model, we do not believe it is prudent to incorporate adsorption-desorption kinetics into the Passaic River contaminant fate and transport model framework. One exception to this might be for selected PAH chemicals following the work of Shor et al. 2003 involving rates of PAH desorption from sediments collected from Piles Creek in the Arthur Kill and Newtown Creek in the East River. As warranted, adsorption-desorption based partitioning kinetics may be added to the model kinetics. If it becomes necessary to add desorption kinetics to the Lower Passaic River Restoration Project model, the work of Kosson et al. (2000) will be reviewed.

Partitioning of metals to phosphorus, one example of a mineral species, has been considered with a total active metal (TAM) state variable by HydroQual on other projects (e.g. Onondaga Lake). Should data support that a consideration of mineral partitioning in the Passaic River is necessary, appropriate kinetics can be incorporated into the model, a data set indicative of mineral partitioning in the Passaic River or Harbor has not yet been identified.

A challenge in modeling the partitioning behavior of the contaminants of concern relates to measurement limitations. For example, XAD columns used as part of the CARP monitoring capture the freely dissolved portion of the dissolved contaminant plus some unspecified fraction of the DOC complexed portion of the dissolved contaminant. Researchers at the State University of New York (SUNY) at Syracuse are attempting to quantify how much of the DOC-complexed dissolved contaminant is actually captured by XAD as a function of several factors such as flow rate, column age, etc.

5.3.1.2 Chemical Transformations

Modeled chemical transformations will include as necessary hydrolysis, photolysis, biodegradation, and oxidation. The kinetics available from CARP includes neutral, acid, and alkaline hydrolysis. Biodegradation in water, on suspended sediments, and on various organic carbon forms may be calculated based on specified bacterial densities. Tying biodegradation rates to bacterial populations could be done as part of the estimation of model input parameters if site-specific measurements of bacterial populations are available. Modeling biodegradation rates explicitly in the numerical model becomes a necessary alternative when direct measures of bacterial populations are not available and surrogate indicators of bacterial biomass, e.g., locations where there is a labile organic carbon present must be relied upon.

5.3.1.3 Matrix Transfers

Transfer of contaminants across the air-water and sediment-water interfaces will be included in the calculations. Contaminant transfer across the air-water interface will be considered in two ways: as an independent external loading (i.e., wet and dry deposition plus forward diffusion gas exchange) and a dynamic back diffusion gas exchange which is dependent upon water column concentrations (i.e., liquid film control) of freely dissolved chemical. For contaminants with low Henry's constants, gas exchange rate coefficients may be calculated based on two-film theory (Schwarzenbach et al., 1993). Checks on the transfer across the air-water interface will be performed using the results of gas exchange rates as determined from sulfur hexafluoride and helium-3 tracer studies, which may be available for the Passaic River and contiguous waterways. One example is the work of Clark et al. 1994 performed on the tidal freshwater Hudson.

Section 5.3.4 below describes the approach for transfer of contaminants across the sediment-water interface.

5.3.2 Metals Water Column Kinetics

The kinetic structure planned for metals in the water column in general is analogous in several regards (i.e., transfers across the air-water interface, carbon partitioning, and chemical transformations) to that planned for HOCs. They will be modeled in the same manner as in the CARP model. For metals, sulfide will also be accounted for in the partitioning formulation. Specifically for mercury, several mercury forms (i.e., dissolved and particulate methyl mercury, dissolved and particulate Hg (II), elemental mercury, and inorganic mercury complexes) will be included as modeled state variables. The inorganic mercury complexes modeled will include mercury and methylmercury hydroxides, chlorides, carbonates, sulfates, and sulfides. The mercury chemical transformations that will be taken into account include methylation, demethylation, photodegradation of methylmercury, photoreduction of ionic mercury, and volatilization of dissolved gaseous mercury.

Modeling of mercury cycling includes abiotic and biotic kinetic processes, as well as chemical speciation. There are four major mercury transformations that will be addressed in this modeling analysis: volatilization, aquatic speciation, methylation, and demethylation. Photochemistry is an additional process affecting mercury cycling that will be evaluated, but may not be included if it is concluded that it is not significant.

The mercury fate sub-model will include three mercury valence states: divalent, methyl, and elemental mercury. From these three state variables, the model will use equilibrium speciation to calculate various additional forms of mercury, and will include major inorganic complexes as well as binding to dissolved and particulate forms of natural organic matter (NOM). Similar to HOCs, we are assuming that metal speciation will occur on an equilibrium basis because of the frequency with which repeated tidal resuspension occurs in this estuary (Sanford et al. 1991, Sanford 1994). This would not be true of a riverine system where resuspension events are more sporadic.

5.3.3 HOC Sediment Kinetics

Similar to the water column, modeled sediment kinetics for HOCs will include partitioning, chemical transformations, and matrix (i.e., water and sediment) transfers. Partitioning of HOCs among freely dissolved, DOC-bound, and particulate phases will be described by equilibrium relationships for both DOC and POC. The equilibrium relationships will be governed by contaminant specific octanol-water partition coefficients. Important sediment processes that will be modeled include diffusive exchange of dissolved contaminants between sediment pore water and the water column, sediment layering, and mixing processes of particle-bound contaminants. The sediment mixing processes ultimately determine the response time that contaminant concentrations in surface sediments and the overlying water column experience as a consequence of future changes in external loadings and other management or remediation actions. Modeling the sediment mixing processes correctly is a critical component of the Lower Passaic River Restoration Project model.

Mixing of pore water with overlying water and between sediment layers may include the effects of "hydrodynamic pumping" of water through sediment bed forms (Elliott, 1990) and/or "bioirrigation" due to the activity of sediment organisms (Boudreau, 1994; Schluter et al., 2000). For estimating rates of diffusive exchange of dissolved contaminant, including benthic enhancement, we will take advantage of the pore water diffusion coefficients, D_d, used in two site specific models: the calibrated/validated System Wide Eutrophication Model (SWEM) which is the basis of the CARP organic carbon production model and the Thomann-Farley PCB model. D_d as taken from SWEM is approximately two to three times higher than molecular diffusion. D_d in SWEM was calibrated against pore water ammonia concentrations, which are highly dependent upon pore water diffusion. Of particular importance for the Lower Passaic River Restoration Project model will be the influence of the pore water diffusion coefficients on the contaminants with low octanol-water partition coefficients such as the low molecular weight PAH compounds.

As applied in CARP, the D_d term is not varied or corrected for molecular weights across contaminants. The rationale for this approach is that the biological effects are the more predominant component driving pore water mixing. This approach will be revisited for the Passaic River application. The model code will include the ability to specify contaminant specific (i.e., molecular weight corrected) mixing rates between sediment pore water and overlying waters and between sediment layers. The model code will also include the ability to specify different rates of pore water mixing over sediment depth in the event there is evidence to support a reduction in biological activity over depth. At this time, pore water advection modeling is not planned.

Fluxes of dissolved contaminants, both HOCs and to a lesser extent metals, from the pore water to the overlying water column occur almost entirely as a DOC complexes. A noted weakness of the CARP model kinetic formulation of diffusive exchange is that the CARP organic carbon production model does not explicitly calculate pore water DOC concentrations. The CARP model accounts for this by using an assigned concentration of pore water DOC, which is not included in the overall carbon balance. As part of the development of the Lower Passaic River Restoration Project model, the pore water and sediment flux studies conducted by Burdige and Zheng (1998) and Burdige et al. (1999) will be evaluated. Depending upon the outcome of the review, the CARP POC mineralization kinetics may be expanded to include DOC as a step in organic carbon mineralization with inclusion of DOC sorption to iron oxyhydroxides in the sediment aerobic layer to properly constrain DOC fluxes. In any case, the evaluation of dissolved fluxes from the sediment will be a component of the overall calibration of the model. First, the sediment settling and resuspension fluxes will be calibrated, in the context of the sediment transport model, with regard to water column TSS levels and long-term sedimentation rate information. The fluxes of sorbed chemical between the water and sediment will be directly tied to the particulate fluxes. The sediment bed model will include the capability to evaluate total dissolved chemical concentrations in pore water. While the approach will differ for organic chemicals and metals, in either case it is the total dissolved concentration in pore water that will control the gradient that drives the diffusive flux. The mass transfer coefficient will set the magnitude of diffusive fluxes between the water and sediment pore water. Upon inspection of initial model data comparisons (water column and pore water concentrations, if available), a decision will be made with regard to the need to refine this approach to account for other factors that may have an effect on diffusive fluxes - including the details of how the dissolved pore water concentration is computed.

Transfer of particle-bound contaminants across the sediment-water interface and between sediment layers is due to bioturbation (Aller, 1988). To model bioturbation, the approach used in the CARP model. Bioturbation varies seasonally (Balzer, 1996), is proportional to the biomass of the macrobenthos inhabiting the sediment (Matisoff, 1982) and is influenced by temperature (Gerino et al., 1998). Benthic biomass will not be modeled directly, but rather it will be assumed that benthic biomass is proportional to the concentration of labile organic carbon in the sediment which will be calculated by the organic carbon production model and passed to the contaminant fate and transport model. The direct relationship between sediment labile organic carbon and benthic biomass, and the basis of our assumption, is that the flux of labile organic carbon deposited to the sediment, D_p, is the food source for the macrobenthos and the source of labile organic carbon in the sediment.

The recent work of Barabas et al. (2004), which concludes that reductive dechlorination and formation of 2,3,7,8-TCDD at the expense of 1,2,3,7,8-PeCDD is an element of contaminant fate in Passaic River sediments, will be considered. Part of our effort in developing the HOC sediment kinetics for the Lower Passaic River Restoration Project model will include an assessment of whether or not artificial, i.e., caused by "numerical dispersion", mixing of contaminants between sediment layers is occurring in the Lower Passaic River Restoration Project model. Numerical dispersion was found to occur in a model of the Fox River in areas experiencing alternating periods of erosion and deposition. Problems of numerical mixing may be attenuated by following the approach of Limno-Tech (1998).

Finally, to reduce the computational burden and Lower Passaic River Restoration Project model simulation times, the HOC kinetics will take advantage of an archival stack (Limno-Tech, 1998). An archival stack is a well-established modeling technique whereby contaminant concentrations in deeper sediment layers are held constant (i.e., effectively removed from the model calculations) for most simulation time steps unless there is a major storm event or dredging operation that would alter the concentrations in the deeper sediment layers. During such events, the contaminant concentrations in the deeper sediment layers are included in the model calculations. The archival stack can be handled as a reservoir of uniform contaminant concentration or it may be configured to include multiple layers of varying concentrations. For purposes of the Lower Passaic River Restoration Project model, multiple layers will be used in the archival stack and the historical pattern of sediment contaminant concentration with depth shall be tracked. The use of the archival stack in this manner requires that detailed measurements of contaminants at a high spatial resolution in deep sediments are available for purposes of specifying the initial contaminant concentrations in the archival stack.

5.3.4 Metals Sediment Kinetics

All of the sediment mixing and layering processes and numerical considerations described above in Section 5.3.3 for HOCs in sediments will be applied to the Lower Passaic River Restoration Project sediment kinetics for metals. The Lower Passaic River Restoration Project sediment kinetics for metals, similar to the CARP model, will also take into account the role of sulfides in binding metals to particles as well as sediment processes unique to mercury.

In the absence of methylation, mercury would not be bioavailable at the very low concentrations that need to be considered for methylmercury; therefore, the correct formulation of methylation kinetics will be a critical component of the Lower Passaic River Restoration Project model. Fortunately, the Lower Passaic River Restoration Project model can take advantage of the CARP mercury model. The CARP mercury model was developed in consultation with national mercury experts, Robert Mason and William Fitzgerald. It is known that mercury methylation is directly related to the rate of sulfate reduction in the sediment (King et al., 1999) and that the sulfide concentration in pore water affects the methylation rate as well (Benoit et al., 1999). The SWEM-based CARP organic carbon production model computes both the rate of sulfate reduction (i.e., it is the critical step in the generation of sediment oxygen demand) and the sulfide concentrations in pore water and the water column. The calculated rates of sulfate reduction and sulfide concentrations will be used to drive the Lower Passaic River Restoration Project mercury model. The mercury demethylation process will also be modeled following the CARP model. Initial demethylation kinetic constants and rate coefficients will be based on the ACME data collected by Marvin-DiPasquale and Oremland (1998) in the sediments and soils of the Florida Everglades. The relatively high concentrations used in the ACME data have likely biased those demethylation rates lower than may be appropriate under field conditions, and experience in the CARP modeling for mercury have supported the use of somewhat higher demethylation rates.

5.4 DEVELOP CONTAMINANT LOADINGS AND OTHER MODEL INPUTS

Section 5.3 above describes in detail the kinetic processes that will be modeled and the features planned for the Lower Passaic River Restoration Project contaminant fate and transport model. The application of the planned model requires that external loadings of contaminants and other model forcing functions be specified. The development of the required inputs for the Lower Passaic River Restoration Project contaminant fate and transport model is described in detail in the following sub-sections.

5.4.1 Specify Contaminant Loadings

Major sources of external contaminant loadings that need to be incorporated into the Lower Passaic River Restoration Project Contaminant Fate and Transport model include: tributary headwaters or heads of tide (HOTs), sewage treatment plants (STPs), combined sewer overflows (CSOs), stormwater runoff (SWR) from the land, and direct deposition from the atmosphere to the water surface of the model domain. Protocols for the generation of these loadings established during the development of the CARP contaminant fate and transport model will be applied for the development of these loadings for the Lower Passaic River Restoration Project contaminant fate and transport model. The contaminant load generation protocols planned for the Lower Passaic River Restoration Project model are described in detail below on a loading source type specific basis.

A common feature to all loadings types is the need to specify both the flow (i.e., volume per time) and the contaminant concentration (mass per volume) components associated with each individual loading. The required flow component for the contaminant loadings is identical to the freshwater flows inputted to the hydrodynamic model (see Section 2). These flows were also used to generate the loadings required for the suspended sediment transport/organic carbon production model (see Sections 3 and 4). The contaminant concentration component of each loading will be developed based on available measurements. In certain cases, as described below, the contaminant loadings will be developed using three components: flow, organic carbon concentration, and contaminant concentration on a carbon normalized basis. Table 5.1 shows the protocols developed for the CARP model for quantifying the contaminant and sediment sources included into the model.

Table 5-1. Sources and Flows Used for Quantification of Contaminants and Solids Loadings in CARP Model

SOURCE	FLOW	SS	POC	CONTAMINANT
Tributary	USGS daily	NSL	NPL	CARP data medians for dissolved and POC normalized particulate
STP	Hourly to monthly DMR and plant records	DMR's	SWEM 1994-95 data; DMRs	Plant specific CARP total data medians
CSO	NOAA NCDC hourly precipitation and landside model	CARP data median; historical CSO program data	SWEM 1994-95 data	CARP total data medians
SW Runoff	NOAA NCDC hoursly precipitation and landside model	CARP data median; historical CSO program data	SWEM 1994-95 data	CARP total data medians; separated by urban and rural
Atmospheric Deposition	NA	NA	NA	Developed by NJADN for CARP
Landfill Leachate	NYCDOS estimates	NA	NA	CARP measurements

5.4.1.1 Tributary Headwater Contaminant Loadings

Following protocols developed for the CARP model, contaminant loadings from tributary headwaters will be specified as Lower Passaic River Restoration Project model input on a daily basis, using median dissolved and median POC normalized contaminant concentrations. Median dissolved and median POC normalized contaminant concentrations were used for CARP to better account for the observed variability in HOT contaminant measurements. The median concentrations will be calculated from measurements obtained during the Lower Passaic River Restoration Project sampling program. It is anticipated that Lower Passaic River Restoration Project HOT contaminant sampling will include the Passaic River at the Dundee Dam, the Saddle River, the Third River, the Second River, the Hackensack River at the Oradell Dam and Berry's Creek. For other NY/NJ Harbor Rivers (e.g., the Elizabeth River, the Rahway River, etc.), CARP HOT sampling may be relied upon. Median POC normalized contaminated concentrations from measure tributaries will be used to estimate contaminant concentrations from smaller unmeasured tributaries.

Under CARP, two protocols were established for developing daily contaminant loads from observed median dissolved and POC-normalized particulate contaminant concentrations. These methods are based on the availability of POC data for the tributary and are described below.

For tributaries with sufficient POC data, USGS gage flow data and POC loading estimates will be used to evaluate daily contaminant loads as follows:

$$\text{Load} = \text{Flow} \times \text{Dissolved Concentration} + \text{POC Load} \times \text{POC Normalized Particulate Concentration}$$

In this approach, POC loading estimates will be determined using the Normalized POC Loading Function (NPL). NPL is analogous to the Normalized Sediment Load Function (NSL). Site-specific applications of NPL will be developed based on the availability of USGS historical records of POC. In cases where the available data support it, development of relationships for both non-flood (i.e., flow rate less than or equal to twice the mean flow rate) and flood condition for each river will be developed. Otherwise, a single relationship will be applied under both non-flood and flood conditions.

For tributaries where sufficient POC data are not available, a slight revision will made to the above method for calculating loads. For these tributaries, POC loading estimates will be determined from NSL-generated sediment loads multiplied by an estimate for the fraction organic carbon (foc) on suspended sediment. The foc values used in this evaluation will be determined from generic relationships between POC and suspended sediment as measured by USGS for rivers within the Lower Passaic River Restoration Project model domain. The final equation for evaluating contaminant loads from these tributaries is expressed in terms of USGS gage flow data, NSL-estimated sediment loads, and fraction organic carbon estimates as follows:

$$\text{Load} = \text{Flow} \times \text{Dissolved Concentration} + f_{oc} \times \text{SS load} \times \text{POC Normalized Particulate Concentration}$$

5.4.1.2 STP Contaminant Load Estimates

Effluents of the major NY/NJ Harbor Estuary region STPs were sampled for contaminant concentrations as part of CARP. CARP sampling frequencies at the individual plants range between two and eight times with most being sampled three to four times. For each STP sampled by CARP, median contaminant concentrations were identified for each contaminant. These concentrations were paired with time varying flow records (i.e., from monthly DMRs or more detailed NYCDEP records) at each STP to produce time-variable loadings for use in the model. A decision was made not to vary STP effluent contaminant concentrations for purposes of CARP model input due to the temporally sparse (i.e., 3 to 4 or fewer points in most cases) data collected for each STP. A similar approach is planned for the Lower Passaic River Restoration Project model. It is recommended, however, that effluents of key facilities within the Lower Passaic River Restoration Project model domain (i.e., Bergen County, Secaucus, North Bergen, Linden Roselle, Joint Meeting, and Rahway) be sampled for contaminant concentrations again. It might be possible to express time variable effluent concentrations for key facilities sampled under Lower Passaic River Restoration Project and/or to represent these facilities in the Lower Passaic River Restoration Project model with more detailed flow records than were used for CARP.

Under CARP, for purposes of assigning effluent contaminant concentrations to unmeasured plants, the CARP STP effluent data for each state were screened to eliminate facilities with elevated effluent concentrations potentially attributable to industrial dischargers in their headworks. From each state, a median across measured effluents at all plants, which were not screened out, was identified for each contaminant. These median contaminant concentration values were assigned to unmeasured plants in the CARP model. Depending upon the expanse of the final Lower Passaic River Restoration Project model domain (i.e. if there are any unmeasured STPs within the Lower Passaic River Restoration Project model domain), a similar approach for unmeasured STPs will be adopted for the Lower Passaic River Restoration Project model.

5.4.1.3 CSO Contaminant Loadings

It is anticipated that CSO contaminant concentration data will be collected under the Lower Passaic River Restoration Project, to supplement what was already collected under the CARP sampling program and the TSI sampling program. These data will be used for assigning CSO concentrations in the Lower Passaic River Restoration Project model following a protocol established for the CARP model.

For the CARP model, CSO contaminant concentration data collected under CARP by both states were pooled to calculate natural logarithmic mean concentrations for each contaminant. The CARP representative median contaminant concentrations for CSOs as described above were combined with flows varying on an hourly basis to develop hourly loading estimates for more than 700 CSO outfall locations aggregated to the level of CARP model grid cell resolution (304 locations in the model with stormwater). The hourly flows, also used to drive the hydrodynamic model, were generated for each of six water years using detailed landside-loading models (i.e., SWMM and RRMP) developed previously by HydroQual. It should be noted that while the flow component of the individual CSO loadings is well established, based on the hourly outputs of calibrated sewershed models, the contaminant concentration component of CSO loadings, either for individual outfalls or for all the outfall considered is not as well defined. If the population of available CSO contaminant concentration measurements is considered a log normal probability distribution, the measurement which is most likely to occur at any time within an individual CSO whether sampled or not – but assumed similar, is the 50th percentile value, or natural logarithmic mean concentration. Use of the natural logarithmic mean concentration is consistent with a maximum likelihood estimator approach (MLE) for spared or censored data sets.

For purposes of the Lower Passaic River Restoration Project model, HydroQual's landside loading models will be upgraded. The landside loading models will be upgraded to incorporate, as they become available, new SWMM models developed by various jurisdictions in New Jersey under a legal requirement imposed by NJDEP. The most significant of these for Lower Passaic River Restoration Project will be the SWMM models developed by the Passaic Valley Sewerage Commissioners and the City of Newark. The landside models will be run for all Lower Passaic River Restoration Project water years.

New Jersey CSOs sampled under CARP include: Ivy Street, Christie Street, Court Street, Livingston and Front Streets, West Side Road, Elm Street, Anderson Street, and Rahway outfall 003. It is anticipated that Tierra Solutions will collected 3 to 4 samples from each of several drainage areas of the 17 mile stretch of the Passaic River. It is also anticipated that Tierra Solutions will implement its obligation to sample Newark Bay CSOs concurrently with the required Passaic River CSO sampling.

5.4.1.4 Stormwater Runoff Contaminant Loadings

Representative stormwater runoff concentrations assigned to contaminants in the CARP model are based on limited concentration measurements made by New York (at 2 locations) and New Jersey (at 5 locations) CARP investigators. For each contaminant, logarithmic concentration means were calculated and assigned to all stormwater outfall locations. These representative concentration estimates were paired with hourly flows generated from detailed landside models for each water year. There are probably more than 1000 stormwater outfalls to the estuary that were aggregated to the level of CARP model grid cell resolution (304 locations in the model with CSOs). A similar approach is planned for the Lower Passaic River Restoration Project model.

The limited CARP stormwater contaminant concentration samples available suggest a high degree of variability across the seven sampling locations for each contaminant; however, there is not sufficient information available to incorporate this variability into the specified loadings. Organic carbon normalizing the data did not help to reduce variability. The CARP stormwater contaminant sampling appears to have been biased toward the most urbanized areas of the Harbor. Supplemental CARP monitoring of runoff is planned for this summer and will focus on the less urbanized portions of the Harbor drainage area. Given the relatively large volumes associated with stormwater runoff, stormwater runoff will be an important loading source for Lower Passaic River Restoration Project to consider in the design of its sampling program. The CARP stormwater sampling locations in New Jersey include: the Newark Airport Peripheral Ditch, Blanchard Street on the Passaic River, CCI, Smith Marina, and Henley Road on the Hackensack River.

The detailed quantification of highly time variable contaminant loads from stormwater runoff and CSO's would require extensive sampling during an event and sampling of many events. This would be quite expensive and possibly an ineffective use of the available field sampling budget. The sampling strategy for this project is to perform sampling of stormwater runoff and some CSO's as described in this section. However if subsequent loading analyses or modeling analyses indicate these time variable loads are potentially significant but inadequately characterized, further targeted sampling will be recommended to reduce the uncertainty associated with their estimate.

5.4.1.5 Atmospheric Deposition Contaminant Loadings

Atmospheric deposition loadings applied in the CARP model were calculated based on data provided by the New Jersey Atmospheric Deposition Network (NJADN). The NJADN data were collected by researchers from Rutgers and Princeton Universities with support from the Hudson River Foundation, New Jersey Sea Grant, and New Jersey Department of Environmental Protection. Up to four (4) NJADN stations were identified for application to CARP model input:

1. Liberty State Park - Applied to Harbor core (i.e., Hudson River below Haverstraw Bay, Upper Bay, Newark Bay, Arthur Kill and Kill van Kull, East River, Harlem River, Jamaica Bay).

2. Sandy Hook - Applied to open water areas (i.e., Lower Bay and New York Bight, Raritan Bay, Long Island Sound)

3. New Brunswick - Applied to urban tributary areas (i.e., Hackensack, Passaic, and Raritan Rivers)

4. Chester - Applied to northern less urbanized areas (i.e., Hudson River above Haverstraw Bay).

For the case of PCB homologs, fluxes at each of the four stations including gas, particle, and precipitation were available from NJADN and were applied directly to the CARP model. For the case of mercury and cadmium, gas, particle, and precipitation flux data were available from NJADN on a harbor-wide basis that was applied to the entire CARP model domain. These fluxes are 0.080 mg m-2 yr-1 for cadmium and 0.0067 mg m-2 yr-1 for mercury. For dioxin/furan congeners, NJADN did not calculate fluxes, but provided gas and particle concentration measurements for the Liberty State Park, Sandy Hook, and New Brunswick stations. HydroQual followed NJADN protocols (Totten et al., in press) to develop the concentration measurements into fluxes. New Brunswick data were applied to both urban and northern less urbanized tributary areas since Chester data were not available for dioxin/furan congeners.

Atmospheric deposition loadings to the Lower Passaic River Restoration Project model will take advantage of the CARP loading generation protocol and NJADN data as well as any additional atmospheric deposition data collected for Lower Passaic River Restoration Project.

5.4.2 Specify Harbor Boundary Conditions

The specification of Harbor boundary conditions of the Lower Passaic River Restoration Project model will be dependent upon which of two approaches (described previously, see Section 1) is selected for the Lower Passaic River Restoration Project model computational grid. If the Lower Passaic River Restoration Project model computational grid were fully nested within the CARP model grid, the Lower Passaic River Restoration Project model would simply use the same open ocean boundaries contaminant conditions used in the CARP model which are based on CARP sampling data collected in the New York Bight. If the Lower Passaic River Restoration Project model is developed as a stand alone model, driven by the CARP model, contaminant concentrations at the boundaries of the Lower Passaic River Restoration Project model (likely to be at the western end of the Kill van Kull and the southern end of the Arthur Kill) will need to be specified based on collocated CARP model calculations and/or data.

The challenge of specifying boundary contaminant concentrations for the Lower Passaic River Restoration Project model if implemented on a stand-alone basis is that the boundary conditions themselves are likely to be controlled by loading sources and processes occurring in the Passaic River and Newark Bay. This does not present a problem for model calibration, but is problematic for model projection purposes. One would have to know *a priori* what fraction of the Lower Passaic River Restoration Project model boundary is due to Harbor conditions (i.e., is not impacted by inflows) and what fraction of the Lower Passaic River Restoration Project model boundary is due to Passaic River and Newark Bay conditions (i.e., is impacted by inflows). Thus, boundary conditions are therefore likely to be altered because of any remediation/restoration activities in the Passaic River or Newark Bay. HydroQual has faced this challenge before on other high resolution models developed for localized areas of the NY/NJ Harbor estuary. In these situations, a reflection coefficient (alpha boundary) technique or concurrent execution of a larger regional model such as SWEM for driving the boundaries of the high-resolution model have been implemented. A full description of the alpha boundary technique, as it was used in a Long Island Sound modeling analysis, is given in Appendix E. These techniques may be used for setting the contaminant boundary conditions of the Lower Passaic River Restoration Project model. The

CARP model could serve as the larger regional model that would drive Lower Passaic River Restoration Project model boundary conditions.

5.4.3 Perform Contaminant Loading Initial Dilution Simulations

Once contaminant loadings and open boundary conditions have been established for the Lower Passaic River Restoration Project model, the Lower Passaic River Restoration Project model will be used to perform initial dilution simulations. Initial dilution simulations were performed with the CARP model. Initial dilution simulations consist of running the present day loadings of contaminants to the system as conservative (i.e., subject to hydrodynamic transport only, no phase partitioning or other kinetic processes) tracers. Care most be taken to run the initial dilution simulations sufficiently long enough (i.e., based on CARP experience one year of "spin-up" prior to a year for consideration was sufficient in Harbor core areas. Outlying areas such as the Sound and the Bight might require more time) to reach an equilibrium condition. There are several purposes for performing initial dilution simulations:

Initial dilution simulation results when compared to ambient water column data serve as an initial check on the agreement or consistency between the assigned loadings (which are based on the loading measurements) and the measured ambient concentrations in the water column.

Comparing initial dilution simulation contaminant concentration results to ambient water column contaminant concentration data might help point out areas of the model domain where historical contamination in sediments is potentially acting as an active source of contaminants to the water column or where a present day loading has not been accounted for.

Initial dilution results serve as an excellent basis to build the model calibration upon in a stepwise fashion. There is the opportunity to gain insights and understandings into controlling processes along the way. A logical sequence for the Lower Passaic River Restoration Project model would be to repeat the initial dilution simulations as calibration runs turning on in a stepwise progression: partitioning, volatilization, other kinetic processes, etc, and ultimately including sediment bed initial contaminant concentration conditions.

5.4.4 Specify Initial Conditions for Contaminant Concentrations in the Sediment

For the Lower Passaic River Restoration Project model, initial contaminant concentrations in the sediment over depth may be taken from a variety of sources including CARP model calculations; high resolution sediment cores and surficial sediment grab samples collected for Lower Passaic River Restoration Project; and sediment cores and grab samples collected historically as part of other programs and initiatives. It is unlikely, however, that any of theses data sources will be as highly resolved as the Lower Passaic River Restoration Project model computational grid. Some degree of data interpolation may be necessary to assign unique sediment initial conditions at the level of Lower Passaic River Restoration Project model longitudinal/lateral/vertical grid resolution.

5.5 DEVELOP CONTAMINANT FATE AND TRANSPORT MODEL

The development of the Lower Passaic River Restoration Project contaminant fate and transport model involves the incorporation of the kinetics described above in Section 5.3 with the loadings and other model forcing described above in Section 5.4 into the FORTRAN-based RCATOX water quality model framework developed by HydroQual. The RCATOX model framework provides the necessary linkages with outputs from the hydrodynamic and sediment transport/organic carbon production models. In principal, the development of the Lower Passaic River Restoration Project contaminant fate and transport model should be a simple exercise in that it has been successfully accomplished on numerous other projects including CARP. Some of the application-specific challenges that may be faced in the development of the Lower Passaic River Restoration Project contaminant fate and transport model include:

- implementing numerical strategies to reduce model simulation times,
- relocating the open boundaries,
- forcing the open boundaries with CARP model outputs and reflection coefficients,
- determining the thickness of sediment bed computational layers, and
- accounting for the influence of the Hackensack Meadowlands on the behavior of the contaminants.

While some steps will necessarily be taken to overcome these challenges in the development of the computational grid and the hydrodynamic and sediment transport/organic carbon production models as described above in Sections 2, 3, and 4, it is unlikely that these location specific challenges can be fully resolved until the calibration of the contaminant fate and transport model is in progress. As a result of the development of the contaminant fate and transport model, there may be a need to revisit prior work on the computational grid and the hydrodynamic and sediment transport/organic carbon production models.

5.6 CALIBRATE CONTAMINANT FATE AND TRANSPORT MODEL

The Lower Passaic River Restoration Project Contaminant Fate and Transport model calibration for HOCs will be based primarily on the ability of the model to reproduce measured concentrations (historical and current) of dioxin/furan congeners and coplanar PCB congeners in water and sediments as available from the literature, CARP and the Lower Passaic River Restoration Project 2005 - 2006 sampling program. The calibrations for other HOCs will involve changing only contaminant specific model coefficients (e.g., partition coefficients, Henry's constants, etc.). Values assigned to physical and biological model coefficients (e.g., particle mixing rates) for the calibration of the dioxin/furan congeners will remain unchanged for other contaminant calibrations including metals. This calibration approach is similar to that of CARP (i.e., the CARP contaminant fate and transport model calibration is based most heavily on PCB homologs and dioxin/furan congeners) and is based on the assumption that the loading data set for the Lower Passaic River Restoration

Project contaminant fate and transport model will be strongest for the dioxin/furan congeners. Additional calibration for other chemical contaminants will be conducted after a more thorough investigation of loading sources, supplemented by additional sampling of input sources (if conducted) is performed. Initial estimates for model calibration parameters will be taken from the CARP contaminant fate and transport model calibration.

A further check on the calibration of the Lower Passaic River Restoration Project contaminant fate and transport model will be its ability to drive the bioaccumulation and food chain model described subsequently in Section 6. Additional aspects of the Lower Passaic River Restoration Project contaminant fate and transport model calibration are discussed below in Sections 5.8, 5.9 and 5.10.

5.7 PERFORM CONTAMINANT FATE AND TRANSPORT MODEL SKILL ASSESSMENT

The contaminant fate and transport model skill assessment will involve comparisons of the calculated contaminant concentrations in water and sediments to measured data. The purpose of the skill assessment is to assess how well the model compares to data other than the calibration data set. Ideally, there should be an independent, fully synoptic set of measured ambient and loading conditions that mimics the calibration data set to be used for purposes of a full model validation. Absent having the full model validation data set, a model skill assessment is a good test of model robustness. The Lower Passaic River Restoration Project contaminant fate and transport model skill assessment will include:

For the key contaminants of concern (e.g., the dioxin/furan congeners), comparisons of model results to water column and sediment data collected apart from the Lower Passaic River Restoration Project 2005-06 sampling program data that were used as the calibration data set. Skill Assessment data sets may include CARP, REMAP, and Tierra Solutions Ecological Sampling Plan Surficial Sediment data.

For the secondary contaminants of concern (e.g. contaminants for which the calibration loadings data set was weak and calibration parameters were inferred from the calibrations of other contaminants), the calibration data set could be used for skill assessment.

Model and data comparisons for the water column and sediment will include analysis of results over time at individual locations, over depth at individual locations, along spatial transect and along lateral transects. Where practical, model and data comparisons will also be made using regional probability diagrams.

HydroQual will use the results of model skill assessment to characterize model uncertainty. For example if the comparison of model and data indicates that there is a 25% relative error, this same relative error of 25% can be imposed on the results of model projections representing various remedial actions. The results of this uncertainty in calculating contaminant concentrations in water and sediment can be carried forward to the risk assessment to determine if the desired ecological and human health risk targets are achieved considering this model uncertainty. This approach to evaluating model uncertainty is more meaningful than a common approach of varying model inputs by arbitrary percentages and categorizing the results of this analysis as model uncertainty. The analysis of the charge in model response to changes in model coefficients, sometime referred to as sensitivity analyses, indicates the important of model coefficients in determining contaminant concentration, but not model uncertainty.

5.8 PERFORM CONTAMINANT FATE AND TRANSPORT MODEL HINDCAST VERIFICATION

The model calibration and skill assessment procedures described above in Sections 5.6 and 5.7 are a test of the ability of the model to reproduce the calibration conditions or other representative measured conditions. The calibration procedure does not however test the predictive capability of the model, the ability of the model to forecast future conditions over time. It is planned to perform a hindcast verification to demonstrate the long-term predictive capability of the Lower Passaic River Restoration Project model. The hindcast verification will demonstrate whether the model accurately represents the interactions between the water column and the sediment over a long time horizon.

For the hindcast, the Lower Passaic River Restoration Project contaminant fate and transport model will be run for a thirty or forty year time period in the past to determine if the model correctly calculates current conditions. For example, for the CARP model, a hindcast verification involving PCB homologs, 2,3,7,8-TCDD, and cesium from 1965 through to the present is planned. It is not clear at this time which contaminants will be selected for the Lower Passaic River Restoration Project model hindcast verification although cesium is recommended. Cesium is recommended because its historical loadings are relatively well known and originate from a limited number of sources. The chemicals 2,3,7,8-TCDD and DDT are also recommended because they have historical sources located on the Passaic River.

The challenge of performing hindcast verification is reconstructing the historical record of loadings. Dated sediment cores, emissions records, and production records may be useful for this purpose. Since it is not feasible to run the hydrodynamics and sediment transport/organic carbon production models for the 30 or 40 actual years included in the hindcast, the available years of hydrodynamics and sediment transport/organic carbon production for Lower Passaic River Restoration Project will likely be sequenced to mimic the variability in conditions that may have occurred over the hindcast period.

5.9 PERFORM CONTAMINANT FATE AND TRANSPORT MODEL SENSITIVITIES

The calibrated Lower Passaic River Restoration Project contaminant fate and transport model will be further tested to evaluate how sensitive the model calculations are to individual model input parameters. The sensitivities will be performed on a contaminant specific basis since the properties of a given contaminant are likely to influence a contaminant's sensitivity to a given model input. It is anticipated that at least two key contaminants will be selected for sensitivity testing. The parameters for which sensitivities may be performed include: the particle mixing coefficient and the depth of the well-mixed sediment bed layer, the critical sheer stress which ultimately determines net burial and resuspension rates the diffusive exchange between the sediment bed and the water column, partitioning of the dissolved fractions of HOCs to DOC, and contaminant degradation/ dechlorination

For each of these parameters, factor of two changes in the value assigned in the model calibration will be evaluated by performing long-term simulations. Effectively, the planned sensitivity work effort may involve 200 years (i.e., 2 contaminants x 5 input parameters x 2 variations x 10 simulation years) of model simulations.

An additional form of contaminant fate and transport model sensitivity analysis that is planned is an analysis of the potential uncertainty in the specified contaminant loadings. The loading sensitivity can be achieved by shutting off key loadings in the model one at a time and calculating the calibrated model's response to that specific loading. The model results can be stored in a spreadsheet-based unit response matrix that will enable users to scale the loadings and see instantaneously resultant concentrations in the receiving water and sediments based on desired loading changes. Contaminant/loading source (e.g., 2,3,7,8 TCDD coming over the Dundee Dam) unit responses will be performed to produce a loading sensitivity analysis. The contaminant/loading sources for which unit response will be calculated will be identified in consultation with EPA.

An option that will be considered for the sensitivity analyses is coordinating the effort with the completion of the bioaccumulation model so that results of sensitivity runs will include potential contaminant concentration changes in biota as well as in water and sediments.

5.10 LINKAGES TO BIOACCUMULATION MODEL

The bioaccumulation model will take advantage of the outputs generated by the hydrodynamic, sediment transport, organic carbon production, and contaminant fate and transport models. Of critical importance for the bioaccumulation model is the calculation of contaminant concentrations in various media specifically bioavailable to a given organism based on its feeding preferences and other uptake/exposure mechanisms. For example, for exposures involving the water column and/or pore water, the calculation of freely dissolved contaminant concentration from the contaminant fate and transport model is of greater relevance than the total dissolved or DOC complexed dissolved contaminant concentration. Specifically for mercury, predominantly methylmercury and, to a much lesser extent, inorganic mercury are known to bioaccumulate. Both methylmercury and inorganic mercury calculations will be passed forward to the bioaccumulation model.

SECTION 6

BIOACCUMULATION

6.1 INTRODUCTION

Chemicals in water and sediment may present direct toxicological effects to fish and other aquatic organisms in the Passaic River study area. In addition, chemicals may be transferred from the water and sediment to lower trophic organisms, and through the food web to higher-level organisms. The potential for chemicals to be transferred through the food web and bioaccumulate in higher trophic level organisms is a major concern because of toxicological effects to higher organisms and because of exposure to humans through the consumption of contaminated seafood.

As a follow-up to our chemical transport and fate modeling work, we will address bioaccumulation of chemicals in the aquatic food web. In addition to a detailed evaluation of the field data, this investigation will involve the application of a bioaccumulation model. The purpose of bioaccumulation modeling will be to establish how contaminants are being transferred through the food web and how body burdens are expected to change in response to changes in contaminant concentrations in the water column and sediments.

6.2 BACKGROUND

As part of the Passaic River preliminary mass balance study, chemical body burden data were compiled for select fish species (white perch and mummichugs) and compared to chemical concentrations in the water and sediment. Comparative plots for two dioxins (2,3,7,8-TCDD and OCDD), two PCBs (BZ#77 and BZ#153), and two PAHs (pyrene and benzo-a-pyrene) are shown in Figures 6-1 through 6-6. In the plots, fish concentrations are presented on both a wet weight (µg/kg wet weight) and lipid normalized (µg/kg lipid) basis, total water concentrations are given in pg/L, suspended solids concentrations are given on an organic carbon normalized (µg/kg OC) basis, and sediment concentrations are presented on both a dry weight (µg/kg OC) and organic carbon (µg/kg OC) basis. Presented results show:

1. Chemical concentrations in fish in water, sediment and fish are elevated relative to background levels and are a cause for concern.
2. TCDD levels (as shown most clearly suspended solids concentrations) are highest in the lower six-mile stretch of the Passaic River, with lower concentrations in the upper stretch of the Passaic and in Newark Bay.
3. PCB concentrations also show higher concentrations in the lower stretch of the Passaic River, but the trend is not as dramatic as for TCDD.
4. By comparison, OCDD, pyrene and benzo-a-pyrene show less variation with location.

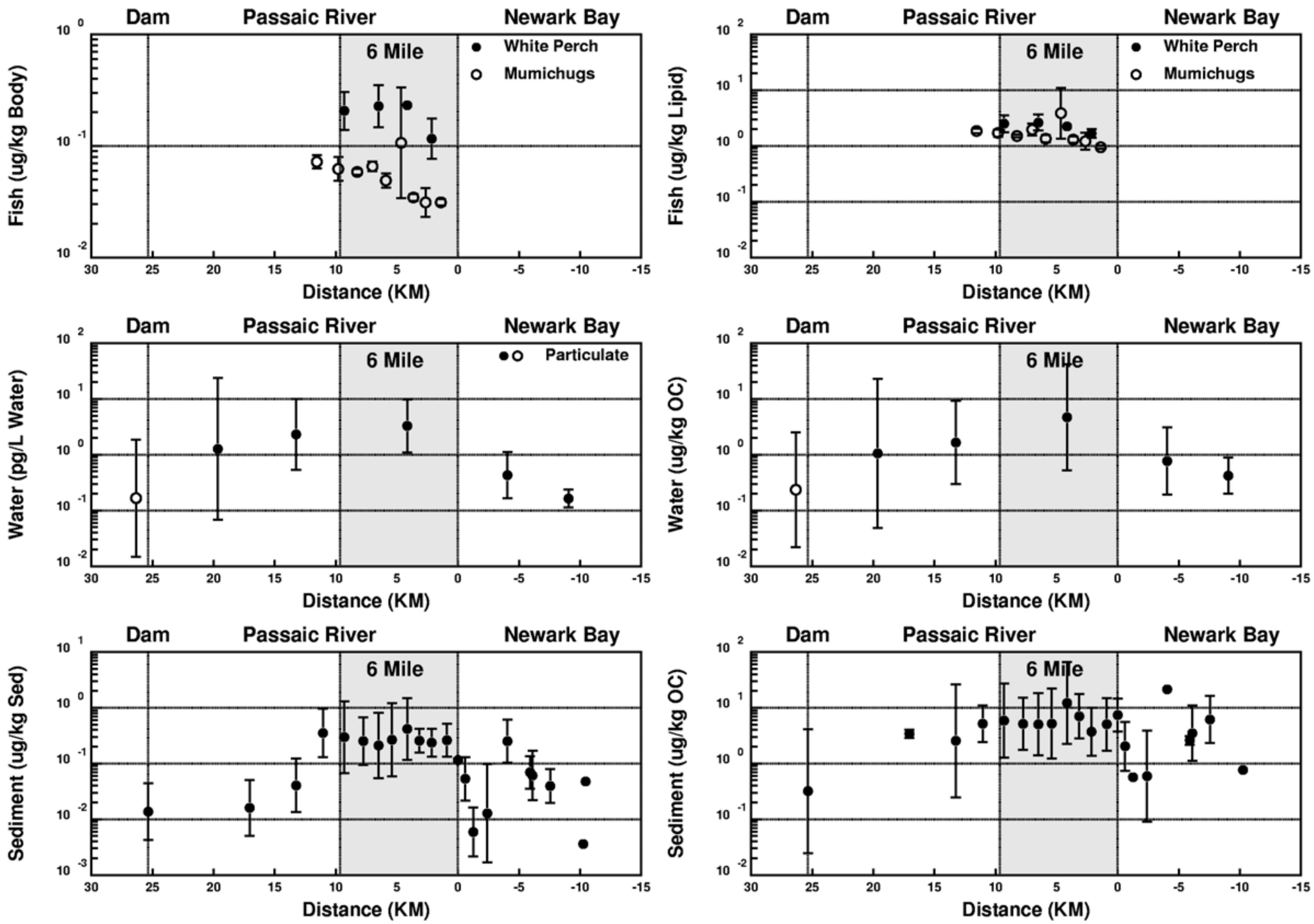


Figure 6-1. TCDD Data in Passaic River and Newark Bay

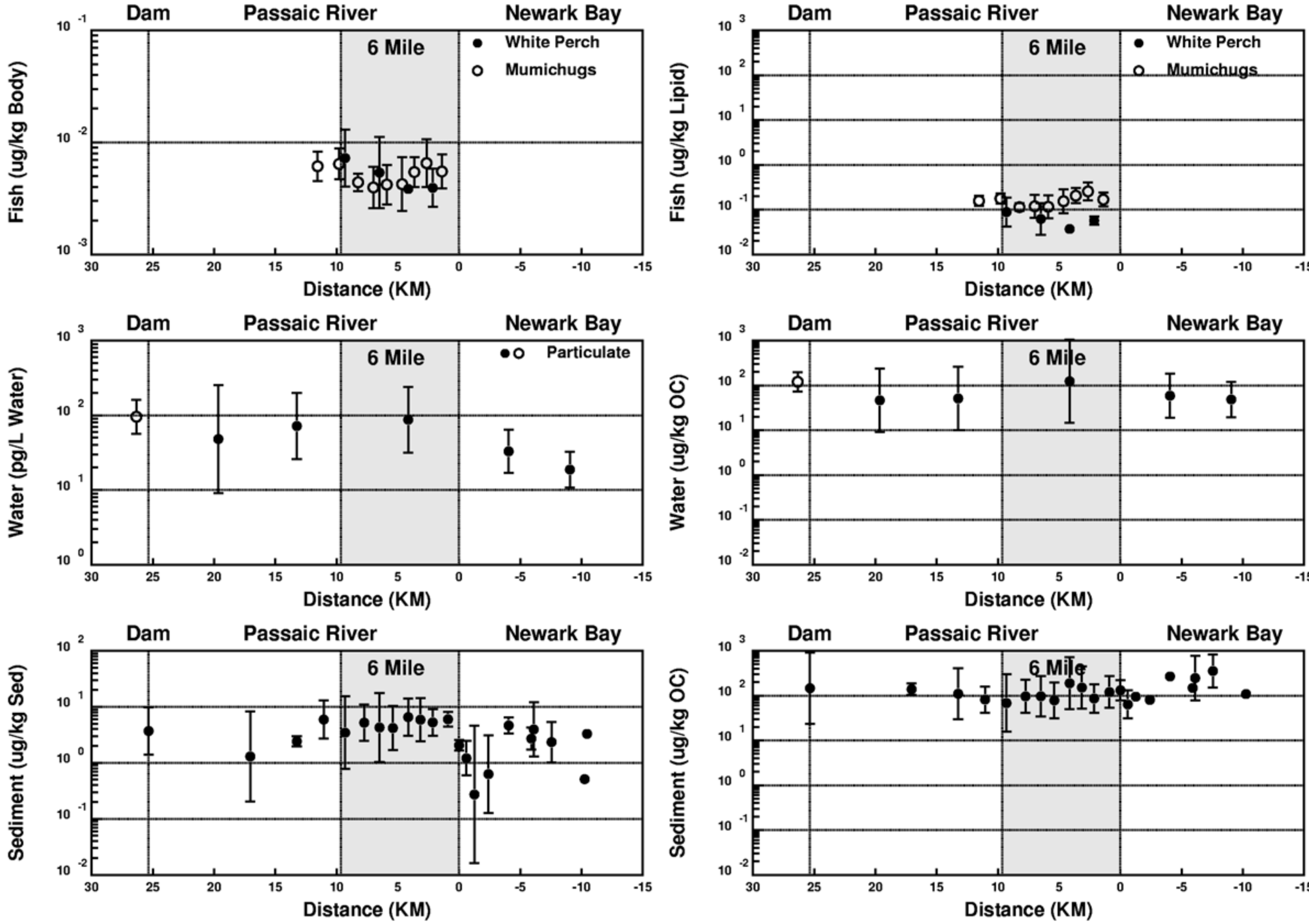


Figure 6-2. OCDD Data in Passaic River and Newark Bay

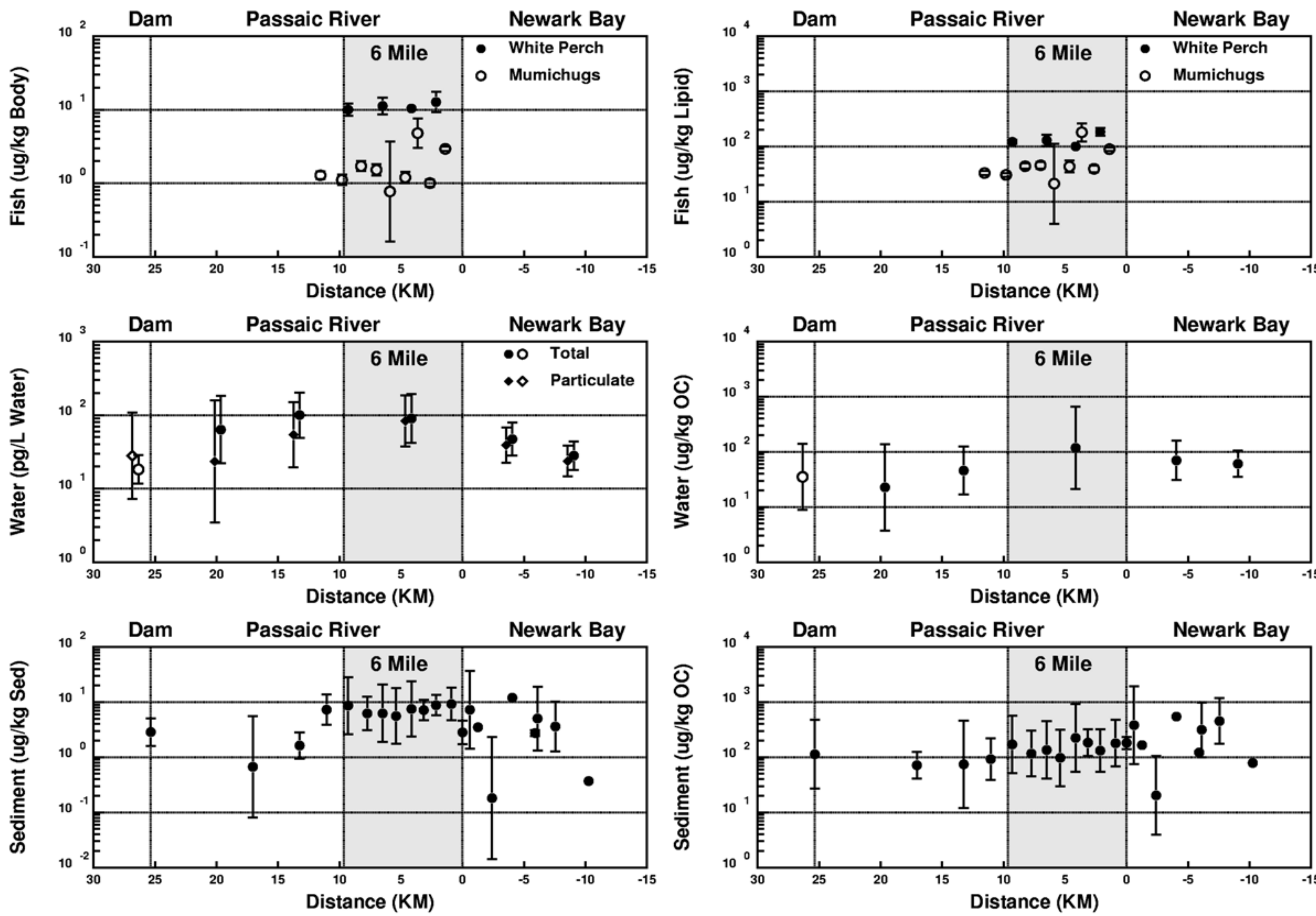


Figure 6-3. PCB77 Data in Passaic River and Newark Bay

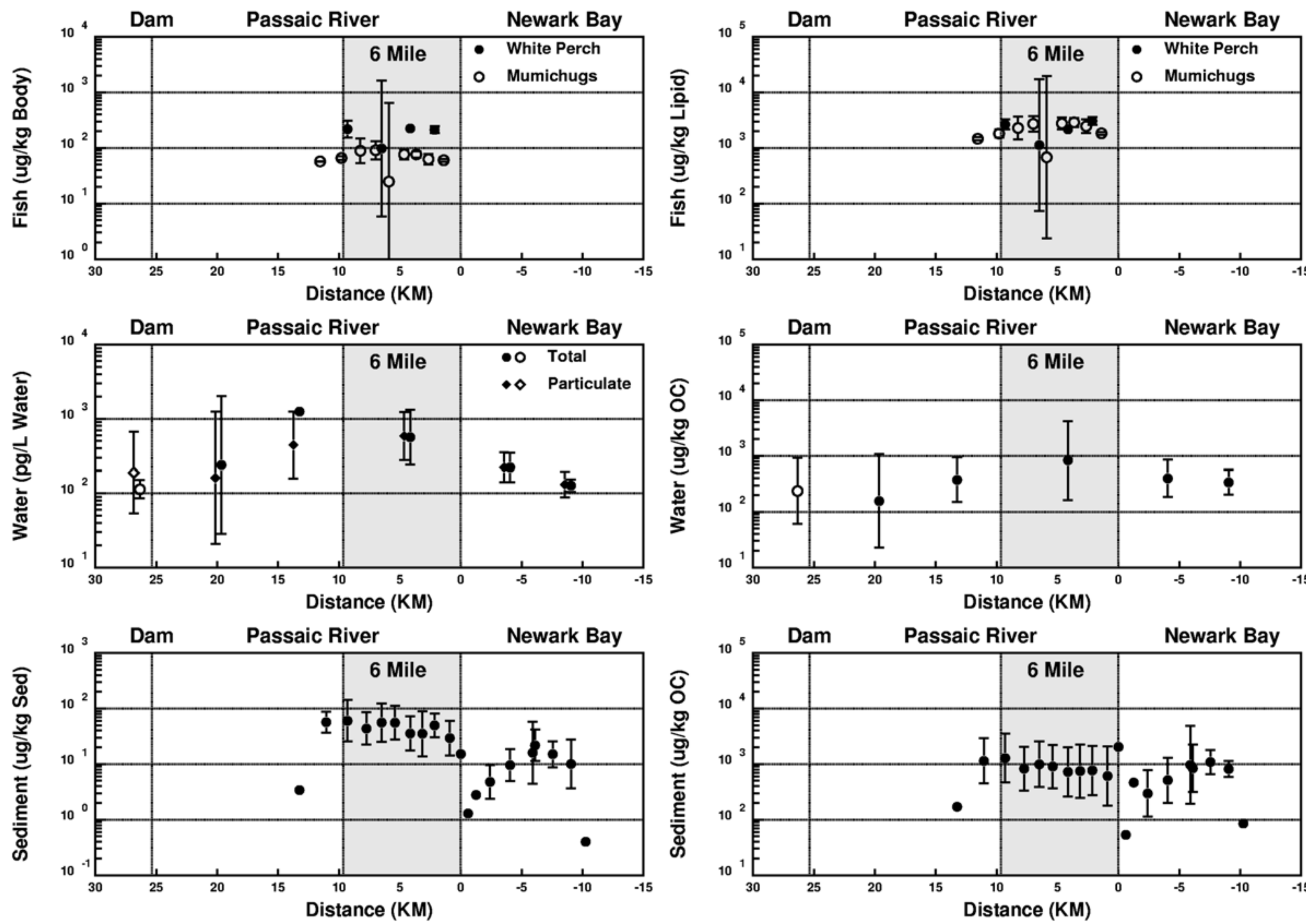


Figure 6-4. PCB153 Data in Passaic River and Newark Bay

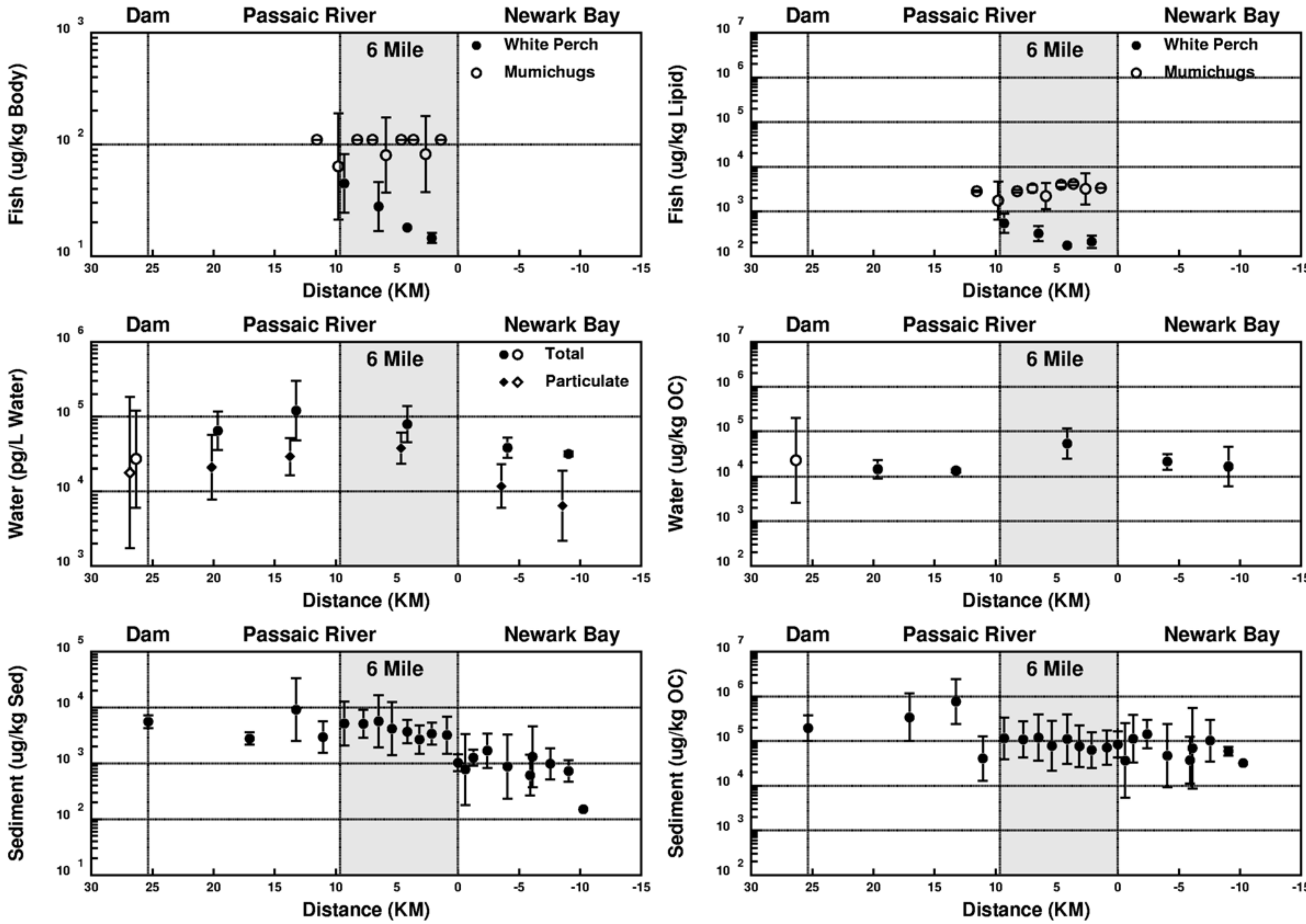


Figure 6-5. Pyrene Data in Passaic River and Newark Bay

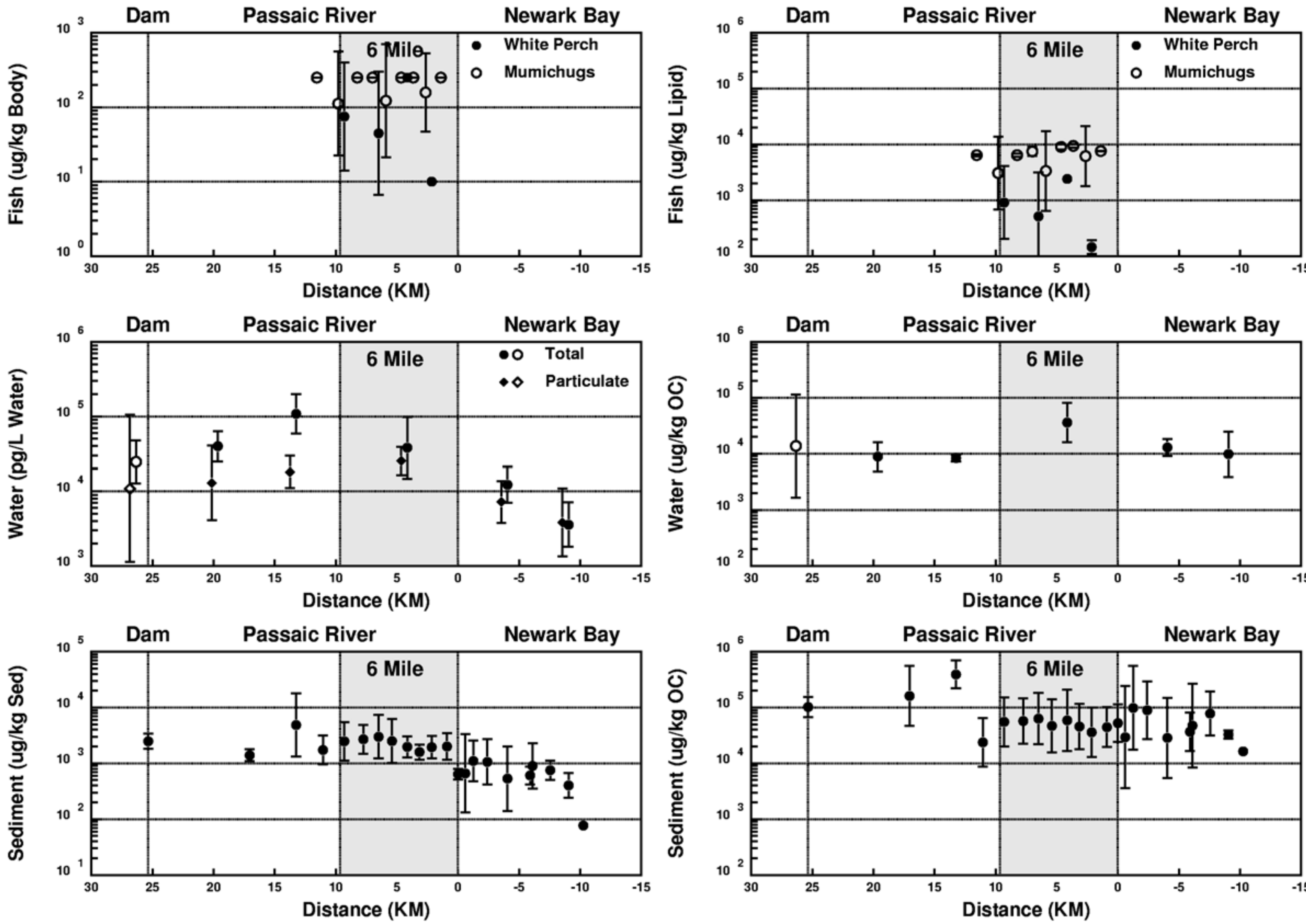


Figure 6-6. BAP Data in Passaic River and Newark Bay

5. Chemical concentrations on suspended solids are not appreciably different than concentrations on surface sediments in the lower six-mile stretch of the Passaic River.

In addition to magnitude and patterns in chemical concentrations, Biota-Sediment-Accumulation Factors (BSAFs), which are given by the ratio of lipid normalized concentrations in fish to organic carbon normalized concentrations in sediment, were also calculated for mummichugs in the lower six mile stretch (Table 6-1).

Table 6-1. Observed BSAFs for Mummichugs in the Lower Six Mile Stretch of the Passaic River

	Pyrene	B(a)P	BZ#77	BZ#153	TCDD	OCDD
log K_{ow}	4.9	6.11	6.36	6.92	7.0	8.6
BSAF (kg OC/kg lipid)	0.04	0.13	0.3	3.0	0.25	0.0015

As shown, the observed BSAFs suggest that chemical accumulations in mummichog are related to log K_{ow} values. A similar pattern for BSAFs in harbor worms has also been reported (Farley et al. 2004). The statement is not meant to imply that a linear correlation between BSAF and log Kow. Rather, the statement together with Table 6-1 is used to describe an observed trend which shows that BSAFs tend to increase to some maximum value and then decrease over a range of Kow values. This phenomenon has been attributed to an increasing importance of dietary exposure and subsequently a decreasing chemical assimilation efficiency for higher Kow compounds (e.g., see Thomann, et al., (1992a). Beyond octanol-water partitioning, other effects of chemical structure (e.g., on metabolism and/or diffusion of compounds through membranes) may also be important in determining BSAF behavior. Preliminary modeling studies have been performed to examine possible explanations of this apparent trend in BSAFs and are discussed later in this section. (Note that the particularly high BSAF value for BZ#153 relative to other chemicals and other PCB congeners (not shown), which may be related to the arrangement of chlorines on BZ#153 and its effect in greatly inhibiting bacterial degradation and metabolism by higher trophic organisms, will need further investigation.)

Based on the preliminary evaluations, bioaccumulation model evaluations are needed: (1) to provide a more detailed understanding of chemical accumulation in the Passaic River food web; (2) to test bioaccumulation model calculations against additional field data; (3) to evaluate the link between current contaminant discharges and in-place sediment contamination and levels in the biota; and (4) to evaluate the response of the biota to changes in the contaminant concentrations in the water column and in sediments.

6.3 BIOACCUMULATION MODEL FORMULATION

The accumulation of toxic chemicals into aquatic organisms is typically 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 the contaminant within the organism. For phytoplankton and possibly lower trophic species, direct uptake from the water is described by diffusion of the contaminant through cell membranes. For fish and other higher trophic organisms, diffusion (e.g. through gill membranes or dermal layers) and food ingestion may both play important roles.

Several bioaccumulation models (Thomann et al, 1984, 1992a, 1992b; Gobas, 1993; Park, 1998; Barber et al, 1991) have been developed over the past fifteen or twenty years to describe the processes of contaminant uptake, depuration, and transformation in aquatic organisms and contaminant transfers through aquatic food webs. Overall, the models are similar in their construct and reflect a cross-fertilization of ideas among investigators (see comparison of Thomann and Gobas models in Burkhard, 1998). Further details of the modeling approach, which has largely been developed for hydrophobic organic chemicals (HOCs), are described below.

General Equation for Bioaccumulation: Model equations for the uptake and release of contaminants are often written in terms of μg contaminant per g organisms (ν) where organism weight is expressed in terms of wet weight or lipid content (Thomann et al., 1992a). The general form of bioaccumulation equations is given below:

$$\frac{dv_i}{dt} = k_{ui}C_d - k_{bi}v_i + \sum \alpha_{ij}I_{ij}v_j - \left[k_e + k_m + k_g\right]v_i \qquad (6\text{-}1)$$

where ν_i is the concentration of the chemical in organism i (μg contaminant/g organism i), t is time, k_{ui} is the diffusive uptake rate of dissolved contaminant from the water and into the organism (L/g organism i/day), C_d is the freely-dissolved contaminant concentration (μg contaminant/L) typically does not include complexed forms of the contaminant, k_{bi} is the back diffusive transfer rate of contaminant from the organism and into the water (1/day), α_{ij} is the efficiency of organism i to assimilate contaminant from feeding on organism j (unitless), I_{ij} is the consumption rate of organism i on organism j (g prey/g predator/day), k_e is the excretion/egestion rate coefficient for contaminant removal from organism i (1/day), k_m is the metabolic transformation rate coefficient for contaminant in organism i (1/day), and k_g is the growth rate coefficient (1/day) and is included to account for the reduction in ν_i due to the increase in the size of the organism.

If contaminant transfer from the water phase is the dominant uptake mechanism (which is an appropriate assumption for phytoplankton and macrophytes), the steady-state solution of equation 6-1 is given in terms of a bioconcentration factor (BCF):

$$BCF_i = \frac{v_i}{C_d} = \frac{k_{ui}}{k_{bi} + k_e + k_m + k_g} \tag{6-2}$$

where the BCF_i is the ratio of ν_i/C_d for uptake of contaminant from the water phase. If removal of the contaminant by excretion/egestion and metabolic transformation are negligible and the growth of the organism is small compared to back diffusion of contaminant from the organism and into the water (k_{bi}), then BCF is equal to:

$$BCF_i = \frac{k_{ui}}{k_{bi}} \tag{6-3}$$

where the ratio of k_{ui} over k_{bi} is related to the affinity of the chemical to partition into the organism. For hydrophobic organic chemicals (HOCs), bioconcentration may be related to chemical fugacity or octanol-water partitioning. For metals, bioconcentration may be related to binding of metal to specific chemical functional groups in the organism (e.g., the metal-binding protein, metallothionein). For strongly bound chemicals, the back diffusion of contaminant from the organism into the water (k_{bi}) will tend to be small and growth of the organism (k_g) will likely serve as the primary mechanism for reducing chemical concentrations in the organism (Thomann *et al.*, 1992b).

For higher trophic organisms, food ingestion is also expected to be an important uptake route. At steady state, the solution to equation1 is given in terms of a bioaccumulation factor (BAF):

$$BAF_i = \frac{v_i}{C_d} = BCF_i + \frac{\sum \alpha_{ij} I_{ij} BAF_j}{k_{bi} + k_{ei} + k_{mi} + k_{gi}} \tag{6-4}$$

where the BAF_j is the ratio of ν_i/C_d for uptake of contaminant from both the water phase and food ingestion and is dependent on BAFs of lower trophic levels.

In similar fashion, steady-state bioaccumulation of contaminant in organisms may also be expressed in terms of the biota-sediment-accumulation factor (BSAF):

$$BSAF_i = \frac{v_i}{r_s} = \frac{BCF_i}{K_{sw\ i}} + \frac{\sum \alpha_{ij} I_{ij} BSAF_j}{k_{bi} + k_{ei} + k_{mi} + k_{gi}} \tag{6-5}$$

where the $BSAF_j$ is again a measure of uptake of contaminant from both the water phase and food ingestion but is expressed in terms of the contaminant concentration in the sediment (r_s) in µg/g sediment; and K_{sw} is the sediment-water partition coefficient (typically in units of mL/g).

For HOCs, field observations for fish indicate that BAF is about four times greater than BCF values (Connolly and Thomann, 1992). This indicates that higher trophic organisms are not in equilibrium with the dissolved contaminant concentrations. Since no evidence exist for active transport of HOCs into organisms, Gobas *et al.* (1993) and others have hypothesized that the digestion and absorption of food in the gastrointestinal tract (GIT) of higher organisms causes the fugacity (or activity) of the contaminant in the unconsumed food to increase. Passive diffusion of contaminant from the unconsumed food and through the GIT membrane then is believed to result in a higher accumulation of the contaminant in higher trophic organisms. This results in biomagnification of HOCs as contaminated food is passed through the food chain.

For metals, food ingestion can also be a significant pathway for accumulation in aquatic organisms (Thomann et al, 1995; Fisher and Wang, 1998). This is particularly true for metals such as zinc, cadmium, copper, and mercury, which induce the production of the metal-binding protein, metallothionein, and as a result, enhance transfer of metal across the gut wall (Thomann et al, 1995). Once accumulated by organisms, metals are typically bound strongly to protein or sulfur groups and are less likely to be transferred to higher trophic levels (Fisher and Wang, 1998).

<u>Food Web Models</u>: Several food chain models (Thomann and Connolly, 1984; Thomann *et al.*, 1991; Connolly, 1991; Thomann *et al.*, 1992a,b; Gobas, 1993) have been proposed to evaluate the bioaccumulation of contaminant in fish from feeding on lower trophic organisms. For example, a generic food chain model proposed by Thomann *et al.* (1992a,b) is presented in Figure 6-7. Five interactive biological compartments are considered, together with the particulate and freely-dissolved contaminant concentrations in the water column and in sediments. In these types of models, the contaminant concentration in phytoplankton is often considered to be in equilibrium with dissolved contaminant concentrations (as described by the equilibrium relationship given in Equation 6-2). The accumulation of contaminant in higher trophic organisms is dependent on both diffusive transfer (e.g. through gills) and feeding as described in Equation 1. Here, zooplankton obtain their food from the ingestion of phytoplankton, benthic invertebrates obtain contaminant through the ingestion of contaminated sediment particles and/or from phytoplankton and detrital matter at the sediment-water interface, forage fish feed on zooplankton and benthic invertebrates, and piscivorous fish feed primarily forage fish.

For specific model applications, feeding patterns, ingestion rates (I_{ij}), growth rates (k_g), and egestion rates (k_e) are determined from bioenergetic models of energy flows through food chains and/or from stomach content, fish growth, and fecal matter production data. Because age may play an important role in describing feeding patterns and in determining the accumulation of contaminant, a further breakdown in age classes may be required (e.g. see the schematic of model compartments for age-dependent accumulation of PCBs in striped bass for the Hudson River (Thomann *et al.,* 1991) presented in Figure 6-8). Other model parameters for diffusive uptake and backward diffusive transfer (k_{ui} and k_{bi}), assimilation efficiencies (α_{ij}), and the metabolic rate coefficients (k_m) are usually taken from previous laboratory studies or are determined from model calibration of field data.

6.4 PREVIOUS APPLICATIONS

6.4.1 Application to the Hudson River Striped Bass Food Chain

A time-variable, age-dependent striped bass food chain model was previously developed for the Hudson River Estuary by Thomann et al. (1989; 1991), and later applied by Farley et al. (2005) in a subsequent study of the estuary. The model includes a five component, water-column food chain that consists of phytoplankton, zooplankton, small fish, seven age classes of perch, and seventeen age classes of striped bass (Figure 6-7). In applying the model to the Lower Hudson, PCB homologue concentrations in water and phytoplankton are taken directly from the transport and fate model calculations. Phytoplankton are preyed upon by a zooplankton compartment, the

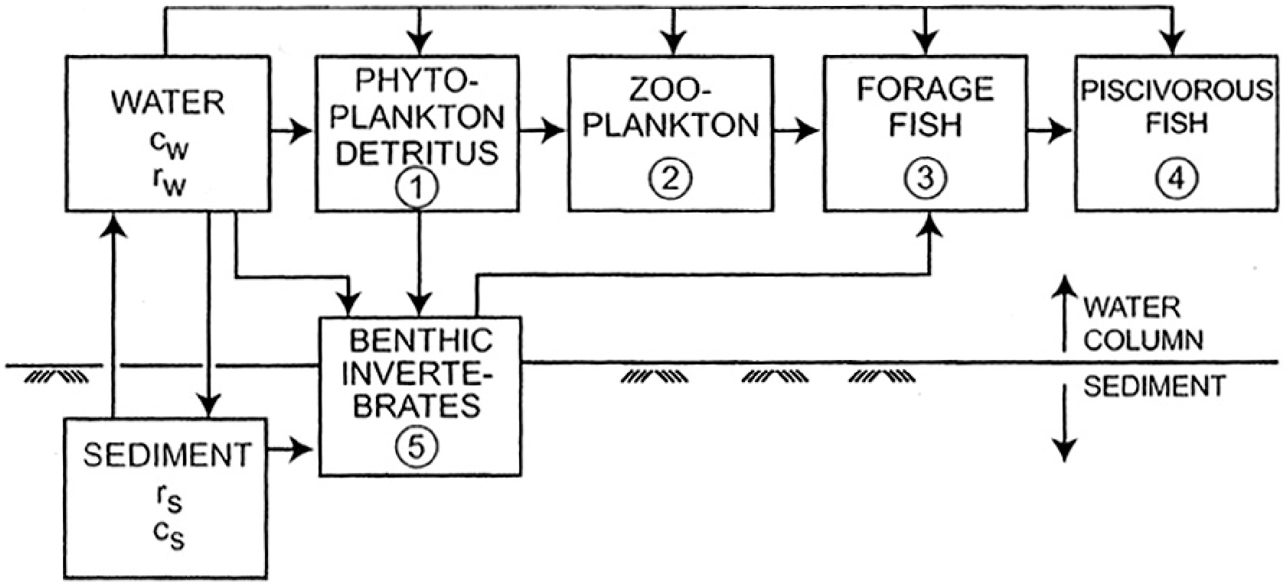


Figure 6-7. Generic Food Web Model (Thomann et. al., 1992).

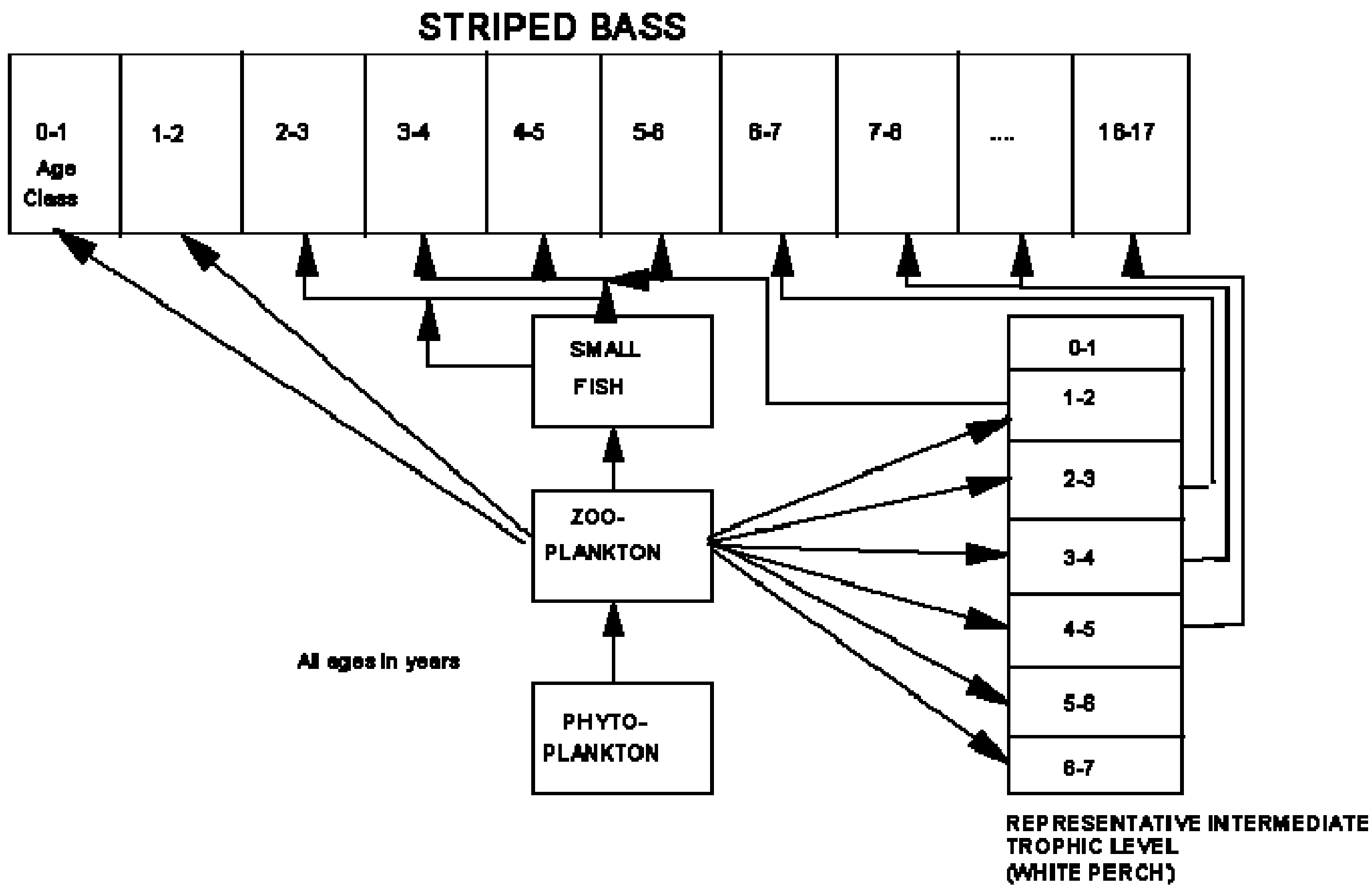


Figure 6-8. Age-dependent Striped Bass Food Chain

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 age 0-1 tomcod and herring. White 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.

Growth rates were determined from results of 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 (1982) for white perch; and from the age-weight data of Setzler et al. (1980) and Young (1988) for striped bass. Details of age-dependent weights and growth rates are given in Thomann et al. (1989) and are summarized in Farley et al. (1999).

Respiration rates for zooplankton, small fish, white perch, and striped bass were estimated using formulations given in Thomann and Connolly (1984) and Connolly and Tonelli (1985). Details of respiration rates, along with lipid content, dry weight fractions, and food assimilation efficiency, are given in Farley et al. (1999). These values are used with the gill transfer efficiency (β), chemical assimilation efficiency from food (α) and PCB homologue-specific parameters for K_{ow}, to calculate gill uptake rates ($k_u = \beta \cdot R_{oxygen}/C_{oxygen}$), back-diffusion rates ($k_b = k_u/(f_{lipid}\ K_{ow})$), and food ingestion rates ($I = (R+k_g)/a$). log K_{ow} values were previously given as 5.0, 5.6, 6.0, 6.45, and 6.85 for di- through hexa-CB. The chemical assimilation efficiency (α) was set equal to the food assimilation efficiency (a) of 0.3 for zooplankton and 0.8 for fish. Gill transfer efficiency (β) was the only remaining parameter and was adjusted in calibrating model results to observed PCB homologue concentrations in white perch. This value was then used for all fish species throughout the Lower Hudson, New York Harbor, Long Island Sound, and New York Bight.

In bioaccumulation calculations, migration of striped bass added a further complication in specifying time-dependent exposure concentrations. Migration patterns used in the calculations were assigned based on Waldman (1988; 1990) and are described in Thomann et al. (1989; 1991). These are summarized as follows: Striped bass are born on May 15th of each year and the yearlings are assumed to remain in the mid estuary (as defined by Km 30 to 126; RM 18.5 to 78.5). The 2-5 year old striped bass are considered to migrate from the mid estuary into New York 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 15th and return to the mid estuary around April 15th to spawn. They remain in the mid estuary until the middle of July.

This information was used in conjunction was used with freely-dissolved and phytoplankton-bound PCB homologue exposure concentrations from the transport and fate model calculations in bioaccumulation model calculations for zooplankton, small fish, white perch and striped bass. Since little or no data were available for PCB accumulation in zooplankton and small fish, testing of the model was performed by comparing model results to observed PCB homologue concentrations in white perch. All parameters for this evaluation were previously specified except for the gill transfer efficiency coefficient (β), which was adjusted to 0.25 for simulation results presented below.

A good comparison of model results to observations was obtained for di- through hexa-CB concentrations in white perch at Km 239 (RM 148.5) (see Figure 6-9 for di-, tri- and penta-CB comparisons) and Km 191 (RM 118.5) (not shown). Di-CB accumulations in perch are quite low (ca. 5 g g-1(lipid)) and appear to rapidly adjust to large variations in PCB exposure concentration in this portion of the river (see Farley et al., 2005 for details). In contrast, accumulations of higher chlorinated homologues in perch are greater (ranging from 10 to 60 g g^{-1}(lipid)). This is largely due to increased hydrophobicity (as represented by the increased K_{ow} value) of the higher chlorinated homologues that favor their accumulation in the lipid of fish. Accumulation of the more-chlorinated homologues by perch show a clear increase in the early 1990s (corresponding to increased PCB loads from the Upper Hudson). Higher frequency variations that are apparent for dissolved PCB concentrations and for di-CB in perch (Figure 6-9), however, are largely attenuated. This is due to the relatively slow rates (of several months or more) for the accumulation and loss of more chlorinated homologues by perch.

Calculated PCB homologue concentrations in white perch further downstream in the mid estuary at Km 94 (RM 58.5) (not shown) also compared well to observed data. At this location, PCB responses in perch exhibit a slow decline, largely in response to the slow decline in dissolved exposure concentrations (see Farley et al., 2005 for details). The resulting concentrations of PCBs in perch at Km 94 (RM 58.5) decreased from a high of 5 g g^{-1}(wet weight) in 1987 to approximately 1 g g^{-1}(wet weight) at the end of our simulation period in 2002. Perch in this portion of the river are particularly important as a food source for striped bass.

PCB accumulation in striped bass however is further complicated by fish migration behavior. This is best illustrated by examining the accumulation of tri- and penta-CB in a striped bass cohort born in 1987. As shown in Figure 6-10, the 1987 cohort quickly accumulates PCBs during the first two years of life in the mid estuary (solid lines in Figure 6-10). As the cohort ages, fish begin to migrate from the mid estuary into the New York Bight (open triangles), and for older fish, the Atlantic Ocean (open circles). During their time out of the estuary, striped bass feed on less contaminated prey and their stored PCB concentrations are reduced by depuration and growth dilution. Each year, as striped bass migrate back into the estuary, their PCB concentrations increase as fish again feed on more contaminated prey.

Differences in homologue behavior are presented in Figure 6-10. As shown, there is a significant loss of tri-CB from striped bass during their migration to less contaminated waters. This

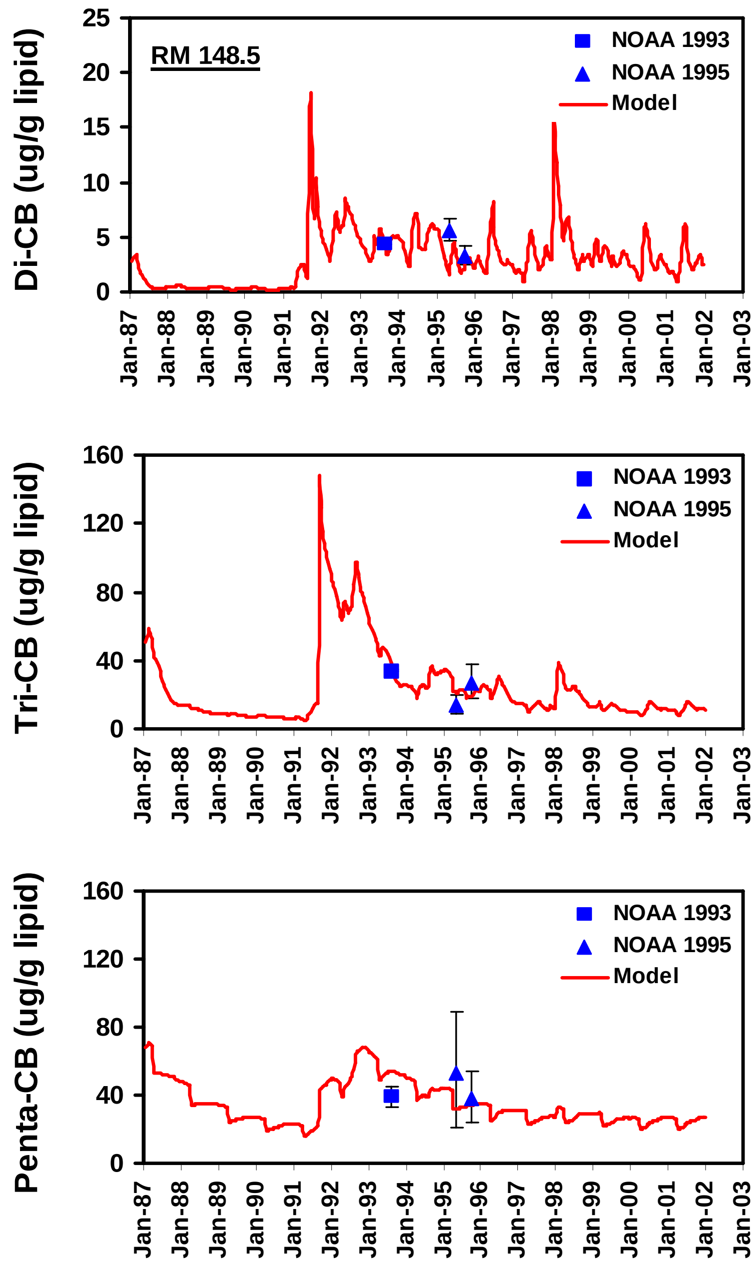


Figure 6-9. Comparison of PCB model results to observations in white perch.

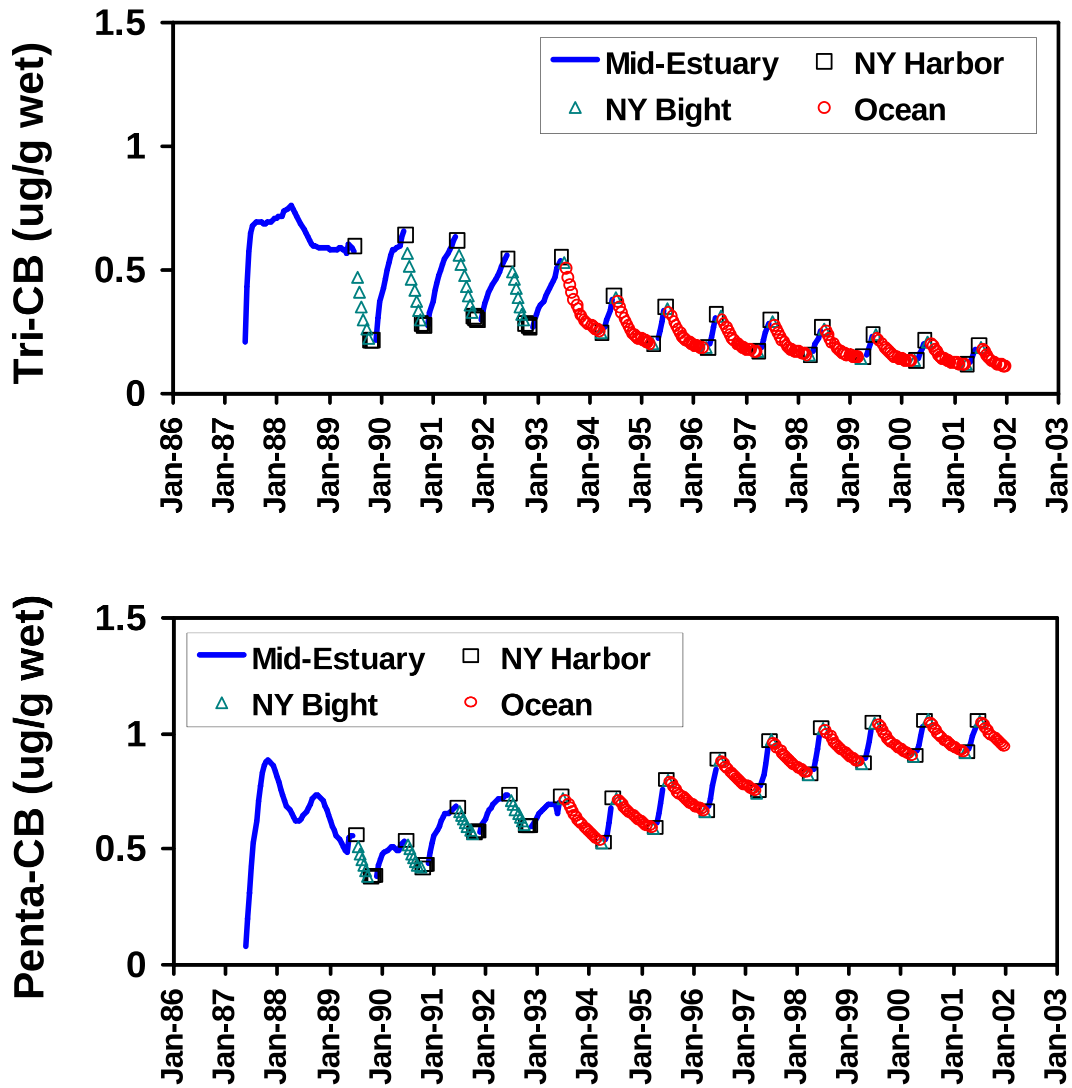


Figure 6-10. PCBs accumulation in striped bass.

is accompanied by a slow decline in tri-CB concentrations over many years. In contrast, penta-CB shows only moderate reductions in concentration during migration. Since the reduction in penta-CB is less than the accumulation of penta-CB by striped bass during their return to the mid estuary, a long-term buildup in penta-CB concentrations occurs over the years. Differences in homologues responses are related to their hydrophobicity (as measured by the log K_{ow}). In this case, penta-CB has a greater affinity to remain in fish lipids and its loss by depuration occurs at very slow rates. Reduction in penta-CB concentrations in striped bass is therefore slow and is largely controlled by growth dilution. This results in a slow decline of penta-CB during migration and ultimately leads to a long-term buildup of penta-CB over time. A shift in PCB homologue distributions to highly chlorinated homologues is therefore expected for older striped bass.

Lastly, comparison of PCB striped bass model simulation results and 1987-97 field data (TAMS/Gradient 1995) are shown for 2-5 year old striped bass in the mid estuary (Figure 6-11). Simulated 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 percentiles. For fall 1990 and fall 1992, average concentrations were recalculated after eliminating a few high outliers from the sample distributions (Farley et al. 1999). As shown in Figure 6-11, model results are consistent with average observed concentrations in striped bass, and show a slight increase from fall to spring as the young fish overwinter in the mid estuary. A slow decline in PCB concentrations in 2-5 year old striped bass is also determined with average concentrations of approximately 1 g g^{-1} (wet weight) at the end of the simulation period in 2002. Similar responses are obtained for PCB accumulations in older striped bass (not shown) with average concentrations of approximately 2 g g^{-1} (wet weight).

6.4.2 Application to New York-New Jersey Harbor Worm Data

As part of the Contaminant Assessment Reduction Project (CARP), we are currently evaluating the accumulation of PCBs, dioxins furans, and PAHs in harbor worms. The key calibration data set provided by CARP is the coincident measures of contaminants in worms and sediments of the Harbor collected at the request of the NJDOT OMR. Preliminary analysis of the PCB data were performed by calculating observed BSAFs (kg OC/kg lipid). Observed BSAF results show a clear homologue trend, with BSAF values increasing from di-CB to hexa- or hepta-CB and subsequently declining. In addition, differences in BSAF behavior is noted between the more contaminated, inner harbor and less contaminated, outer harbor sites.

Preliminary bioaccumulation model calculations were performed to explore this behavior. The model fit for the Sandy Hook data. In this calculation, the increase in the calculated BSAF from di-CB (log K_{ow} =5.1) to hexa-CB (log K_{ow} = 6.8) occurs due to biomagnification of the more chlorinated PCBs from ingestion of contaminated sediments. The subsequent decline in the calculated BSAF beyond hexa-CB is primarily due to a prescribed decrease in the chemical assimilation efficiency (α) for highly hydrophobic chemicals that has been reported in fish studies.

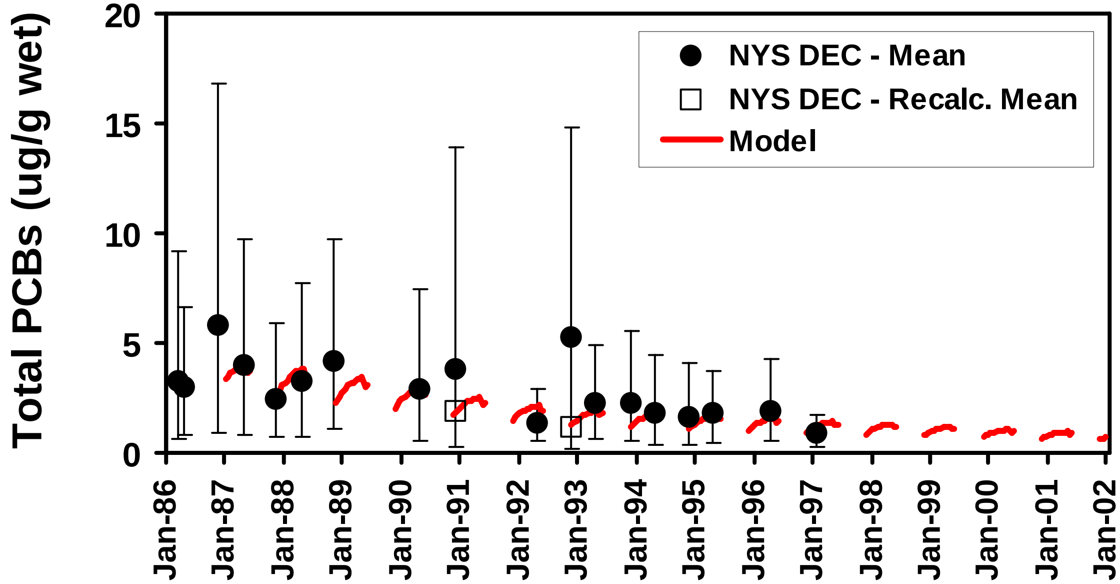


Figure 6-11. Comparison of PCBs model results to observation in striped bass.

The bioaccumulation model was also applied to the Newark Bay site and fit to the field data by adjusting the bioenergetic parameters (e.g., ventilation rates, growth rate, etc.). Although further studies will be required to fully understand this proposed difference in bioenergetics for inner and outer harbor sites, the presence of environmental stressors (e.g., low dissolved oxygen, narcotic responses to high PAH contamination) are offered as a possible explanation. Understanding the reasons for the difference responses at the inner and outer harbor sites however are likely to be critical in our evaluations for the Passaic River. For example, if the difference in chemical uptake at the inner and outer harbor sites is due to differences in the species of worms that inhabit the areas, then we would not expect bioenergetic parameters for the worms to change appreciably in time. If however differences in chemical uptake at the inner and outer harbor sites are due to current conditions of contamination, then higher BSAFs may be expected at the more contaminated, inner harbor site after remedial measures are enacted. Further work in this area is clearly needed.

6.5 PLAN FOR PASSAIC RIVER APPLICATION

6.5.1 Model Structure

Initially we will conduct steady-state model computations. If the temporal response of the fate model proves to be relatively slow, the steady-state assumption should be adequate to the needs of the risk assessment. The importance of time-variable behavior will also be assesses and if warranted (and this indeed may be important in assessing contaminant accumulations in migratory fish species) a fully time-variable, age-dependent bioaccumulation model for the Passaic River study area will be developed based on our previous work on the Hudson River and current work on New York-New Jersey Harbor CARP study. Governing equations for model calculations were discussed in detail previously in this section. The computer code for the bioaccumulation model is based on the generic Food Chain model (Connelly and Thomann, 1985; Connelly, 1991). This provides a flexible platform for modeling bioaccumulation in complex food webs and allows a full coupling of the benthic and pelagic food webs (e.g., see Figure 6-7) along with capabilities to model migrating fish species. Modifications to the modeling structure may be necessary (e.g., for metals) as discussed below.

6.5.2 Food Web Species

At a minimum, the bioaccumulation model for the Passaic application will include phytoplankton, one or two zooplankton species, one or two species representing an intermediate trophic level such as small/juvenile fish, one or two higher level resident fish species, one or two benthic sediment feeders (e.g., polychaete worms), one or two benthic water column feeders (e.g., mussels), and one or two migrating fish species. Final selection of the number and type of species to be considered in the model will be made after consultation with EPA, and will depend on: (1) species currently populating or likely to populate the Passaic River (e.g., after remedial action), (2) species that are representative of a larger group of organisms that play a vital role in the Passaic

River food web (e.g., a copepod species to represent smaller crustaceans), (3) availability of body burden data for species used in model calibration and model hindcasts, and (3) species that will ultimately be considered in risk assessment calculations.

In addition, early life stages such as eggs, roe, larvae, and fry, which are often more susceptible to contaminant concentrations than adult life stages, will be considered for inclusion in the bioaccumulation model. For the early life stages and for the lower trophic level organisms, we will investigate their time responses to changing exposure concentrations. For computational efficiency, if their time responses to changing exposure concentrations are sufficiently rapid, these species will be modeling using steady-state response calculations instead of the fully time-variable solutions that will be required for higher trophic species.

6.5.3 Chemicals of Concern

Chemicals that will be considered in bioaccumulation model calculations will be selected after consultation with EPA and will be based on (1) chemicals that are present in elevated concentrations in the study area, (2) chemicals that have been shown to accumulate in organisms, and (3) chemicals that present a specific ecological and/or human health risk.

6.5.4 Bioaccumulation Model Modifications

The Food Chain model has been developed for hydrophobic organic chemicals (HOCs) and is appropriate for modeling PCBs, dioxin/furans, PAHs, pesticides, etc. In this framework, each organism is treated as a single compartment with bulk transfer properties. This is based on the observation that HOCs have an equal affinity to lipid within the organism, which is independent of its location in a specific organ such as the liver, the kidney, the muscle, etc. This general behavior however does not apply to metals, which tend to concentrate in specific organs such as the liver and kidney (Thomann et al. 1994). A more detailed modeling approach will likely be required for metals. As a starting point, the multi-compartment pharmacokinetic (PB-PK) model of Thomann et al. (1997) will be evaluated, and if deemed appropriate, will be expanded into a full model of metal transfer through the food web.

In addition, seasonal and inter-annual variations in lipid content and growth have not been considered in previous bioaccumulation modeling of the Hudson River and New York-New Jersey Harbor. A decision to include these variations in the modeling framework will be made after a review of the available data and consultation with EPA.

6.5.5 Model Inputs

For HOCs, the bioaccumulation model will require time-variable exposure concentrations for freely-dissolved and particle-bound chemical for the water column and sediment. For metals, the free metal activity, the particle-bound concentrations, and for mercury, the methyl mercury concentrations in solution and on particles will be required. This information will be obtained directly from the chemical transport and fate model calculations for HOCs and metals. (For specific issues related to chemical distributions between freely-dissolved, DOC-bound, and particulate chemical concentrations of HOCs; metal speciation; and mercury methylation rates see the previous section on Chemical Fate Modeling.) Because of the large size of files that would be required to pass concentration results for each model time step, model results will be time averaged (e.g., over a 150 minute period) before being forwarded to the bioaccumulation model.

In addition to exposure concentrations, information on the Passaic River food web will need to be compiled and information on the feeding structure will need to be developed as part of the model input. Bioenergetic parameters (including growth rates, respiration rates, excretion rates), and information on lipid content and life cycles (including spawning, migration behavior, etc.) will also be required for each organism in the food web. For this purpose, a literature review will first be performed to determine what information is available based on field programs and previous modeling studies. A field sampling program will then be designed to collect missing information on the Passaic River food web. Specific issues that will need to be addressed in our review and subsequently in sampling program design include intraspecies variability in lipid content, growth rates, migration patterns, and chemical body burdens; intra-annual variation in lipid content and growth rates; and the potential effects of other environmental stressors on bioenergetic parameters (as previous discussed in our preliminary analysis of PCB body burdens in harbor worms.

6.5.6 Spatial Aggregation

The fine-scale spatial resolution of the Passaic model is largely driven by requirements of the hydrodynamic and sediment transport models. With the possible exception of benthic organisms that show minimal mobility, performing bioaccumulation model calculations on the fine-scale grid is not warranted. This is based on the expectation that the home range (or aerial feeding range) of resident fish will extend over many grid cells, that sharp gradients in chemical exposure concentrations are likely to occur only during short-term events, and that the kinetics for the higher trophic levels are not likely to be fast enough to rapidly response to short-term events. Information on the home range of fish, foraging areas, along with results of exposure concentrations from the chemical fate model will be gathered and reviewed to determine appropriate scales of spatial aggregation that are ecologically relevant and appropriate for bioaccumulation modeling. Based on these findings, chemical transport and fate model results for individual computational grid segments will be aggregated before being passed to the bioaccumulation model. For migratory fish, bioaccumulation calculations will likely be performed using a coarser spatial aggregation of model results for regions outside the Passaic River study area.

6.6 MODEL OUTPUT

Results from the bioaccumulation model will provide detailed information on chemical body burdens (e.g., μg chemical/kg wet weight or μg chemical/kg lipid) in all species as a function of space and time.

6.7 MODEL CALIBRATION AND HINDCAST

The Passaic River bioaccumulation model will be calibrated to "present conditions" using the most recent field data for chemical body burdens. For initial evaluations, bioenergetic parameters and feeding preferences for the food web will be specified based on literature values, field data and published correlations. Values for the bioaccumulation modeling coefficients (e.g., gill transfer efficiencies, chemical assimilation efficiencies, etc.) will also be set based on standard literature values. Sensitivity calculations to determine the range in predicted response for chemical body burdens will be performed for parameters with the highest uncertainty or variability (e.g., feeding preferences, lipid content, relative portions of pore water and overlying water for ventilation by benthic invertebrates, migration patterns, etc.). In addition, specific attention will be paid to the potential for metabolism of chemicals in the food web organisms. During data evaluation and model calibration specific attention will be paid to differences in the bioaccumulation behavior of chemicals that are not likely to be metabolized and those that are susceptible to metabolism. For this purpose, we will first consider chemicals that are not likely to be metabolized (e.g., BZ#153) in model calibration.

Results of these evaluations for all chemicals will be documented and will serve as a basis for the final calibration of the bioaccumulation model. Species that are most important in the risk assessment will be considered most heavily in this evaluation. Standard statistical measures will be used as a quantitative measure of the model calibration.

As a further test of the critical time constants in the model, a hindcast calculation will be performed. As discussed previously in the Chemical Fate modeling section, initial conditions will be specified for the early 1990s and the model will be run to present conditions. For the bioaccumulation portion of this test, particular attention will be given to depuration rates and migration behavior which are likely to play a key role in determining time responses for fish. Comparison of model results to all available field data will be made for the hindcast period.

Model and data comparison for the biological data will include analysis of results over time at individual locations, over depth at individual locations, along spatial transects, and along lateral transects. Where practical, model and data comparisons will also be made using regional probability diagrams. Some of the data displays will incorporate water column and sediment data as well as the biological data (e.g., carbon normalized sediment contaminant concentrations as compared to lipid normalized organism contaminant concentrations).

6.8 MODEL SENSITIVITY/UNCERTAINTY ANALYSIS

In addition to testing the sensitivity of certain model parameters to chemical responses in organisms as part of our calibration procedures as discussed above, we will conduct additional sensitivity calculations on select model parameters. These evaluations will be targeted at quantifying the uncertainties in the bioaccumulation model calculations as they apply to the final assessments of human health and ecological risk. As explained in Section 1.9, a number of model simulations will be performed to develop frequency distributions of the food chain model outputs. These distributions provide a characterization of the uncertainty in output due to uncertainty in the inputs, but for a relatively small number of simulations. The distribution-free Kolmogorov-Smirnov (KS) confidence limits of the empirical cumulative distributions of the model output (i.e., the exposure levels) are then evaluated (see USACE and USEPA, 2006). These confidence limits are analogous to the confidence limits about a single point estimate, but in this instance the KS limits provide bounds for the overall statistical distribution rather than for a single point (Ferson et al., 2005). The KS confidence limits of these frequency distributions are then used to characterize the exposure levels that are input to the Monte Carlo analysis that is performed with food chain model. The food chain model, which runs relatively rapidly in comparison to the fate and transport model, is much more amenable for use with Monte Carlo techniques. Specific details for the sensitivity/uncertainty analyses will be determined after initial assessments are made and in consultation with and with guidance from the EPA, the risk assessment team and appropriate members of the Technical Advisory Committee.

6.9 MODEL OUTPUT/LINKAGE TO RISK ASSESSMENT MODELS

As computational results become available from the food chain/bioaccumulation model, HydroQual will meet with the USEPA risk assessment team, as well as the risk assessment members of the project team to develop the specifications as to how they wish to see model outputs prepared. Possible outputs could include contaminant body burdens on a wet weight basis. Issues to be decided include a determination of the time and space scales for averaging results, selection of model scenarios for which outputs will be generated, etc.

SECTION 7

REFERENCES

Adams, E. E., D. R. F. Harleman, G. H. Jirka, K. D. Stolzenbach, 1981."Heat Disposal in the Water Environment", Ralph M. Parsons Laboratory for Water resources and Hydrodynamics, Department of Civil Engineering, MIT, Cambridge, MA 02139.

Ahsan, A.K.M, and Blumberg, A.F. 1999. Three-dimensional hydrothermal model of Onondaga Lake, New York. J. Hydr. Engrg. 125(9): 912-923.

Aller, R.C. 1988. Benthic fauna and bio-geochemical processes in marine sediments: The role of burrow structures. Nitrogen cycling in coastal marine environments. New York, NY, Wiley, Chichester. 301-340.

Ariathurai, R. and Krone, R.B., 1976. Finite Element Model for Cohesive Sediment Transport. Journal of Hydraulics Division, 102(HY3): 323-338.

Balzer, W., 1996. "Particle mixing processes of Chernobyl fallout in deep Norwegian Sea sediments: Evidence for seasonal effects." Geochim. Cosmochim. Acta 60: 3425-3433.

Barber, M. C., Suarez, L. A., and Lassiter, R. R. (1991). "Modelling bioaccumulation of organic pollutants in fish with an application to PCBs in Lake Ontario salmonids." Can. J. Fish. Aquat. Sci., 48, 318-337.

Bath, D. W. and J. M. O'Connor. 1982. The biology of the white perch, Morone americana, in the Hudson River Estuary. Fisheries Bulletin 80:599-610.

Benoit, J.M., C.C. Gilmour, and R.P. Mason, 1999. Sulfide Controls on Mercury Speciation and Bioavailability in Sediment Pore Waters. Environ. Sci. Technol. 33: 951-957.

Berner, R.A. 1971. Principles of Chemical Sedimentology. McGraw-Hill, New York.

Berner, R.A. 1977. Stoichiometric models for nutrient regeneration in anoxic sediments. Limnol.eanogr., 22(5): 781-786.

Berner, R.A. 1980. Early Diagenesis. A Theoretical Approach. Princeton University Press, Princeton, NJ.

Blumberg, A.F., L.A. Khan, and J.P. St. John. 1999. Three-dimensional Hydrodynamic Model of New York Harbor Region, Journal of Hydraulic Engineering, 125: 799-816.

Boudreau, B.P., 1994. Is burial velocity a master parameter for bioturbation?: Geochimica et Cosmochimica Acta, v. 58, p. 1243-1249.

Boudreau, B.P., 1996. A method-of-lines code for carbon and nutrient diagenesis in aquatic sediments. Computers Geosci. 22: 479-496.

Burdige, D.J., W.M. Berelson, et al., 1999. "Fluxes of dissolved organic carbon from California continental margin sediments." Geochim. Cosmochim. Acta.

Burdige, D.J. and S. Zheng, 1998. "The biogeochemical cycling of dissolved organic nitrogen in esturine sediments." Limnol. Oceanogr. 43: 1796-1813.

Burkhard, L. P. (1998). "Comparison of two models for predicting bioaccumulation of hydrophobic organic chemicals in a Great Lakes food web." Environ. Toxicol. Chem., 17(3), 383-393.

Chaky, D.A. 2003. Polychlorinated biphenyls, polychlorinated dibenzo-p-dioxins, and furans in the New York Metropolitan Area, Rensselaer Polytechnic Institute, Troy, New York.

Chant, R.J., 2002. Secondary flows in a region of flow curvature: relationship with tidal forcing and river discharge. J. of Geophysical Research 107,C9,3131.

Chartrand, 2003. A Geostatistical Assessment of Metals in Passaic River Sediments. Fields Group, USEPA Region 5.

Clark, J.F., R.H. Wanninkhof, P. Schlosser, and H.J. Simpson, 1994. Gas exchange rates in the tidal Hudson River using a dual tracer technique, Tellus, 46B:274-285.

Cole, T.M. and Buchak, E.M. 1995. "CE-QUAL-W2: a two-dimensional, laterally averaged,hydrodynamic and water quality model, version 2.0 user manual." Instruction Rep. EL-95-1, U.S. Army Corps of Engineers, Washington, D.C.

Connolly, J. P. (1991). "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. and R. Tonelli. 1985. Modeling Kepone in the striped bass food chain of the James River Estuary. Estuarine, Coastal and Shelf Sciences 20:349-366.

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 Thomann, R. V. (1992). "Modeling the Accumulation of Organic Chemicals in Aquatic Food Chains." Fate of Pesticides and Chemicals in the Environment, J. L. Schnoor, ed., John Wiley & Sons, Inc., New York, 385-406.

Di Toro D.M. 2001. Sediment flux modeling. Wiley Interscience Ed. P. 624.

Di Toro D.M., Fitzpatrick, J. 1993. Chasapeake Bay Sediment Flux Model. HydroQual, Inc., Mahwah, NJ. Prepared for the U.S. Army Corps of Engineer Waterways Experiment Station, Vickburg, MS. Contract Report EL-93-2.

Di Toro, D.M., C.S. Zarba, D.J. Hansen, W.J. Berry, R.C. Swartz, C.E. Cowan, S.P. Pavlou, H.E. Allen, N.A. Thomas and P.R. Paquin. 1991. Technical basis for establishing sediment quality criteria for nonionic organic chemicals by using equilibrium partitioning, Environmental Toxicology and Chemistry, Vol. 10, No. 12.

Di Toro, D.M., J.A. Mueller and M.J. Small. 1978. Rainfall-Runoff and Statistical Receiving Water Models, NYC 208 Task Report 225. Prepared by Hydroscience, Inc. for Hazen and Sawyer Engineers and New York City DWR, 271 pp. March 1978.

Donelan, M.A., 1977. A simple numerical model for wave and wind stress application. Report. National Water Research Institute, Burlington, Ontario, Canada.

Dortch, Q., M. L. Parsons, N. N. Rabalais and R. E. Turner .1999. What is the threat of harmful algal blooms in Louisiana coastal waters. Recent Research in Coastal Louisiana. Rozas, Nyman, Profittet al. Lafayette, Louisiana. 134-144.

Edinger, J.E., D.K. Brady and Greyer, J.C. 1974. "Heat exchange and transport in the environment." Rep. No. 14, Cooling Water Res. Project (RP-49), Electric Power Research Institute, Palo Alto, Calif.

Egbert, G.D., A.F. Bennett and M.G.G. Forman. 1994. TOPE/POSEIDON Tides Estimated using a Global Inverse Model, J. Geophy. Res., 99 ()C12), 24, 821-852.Elliott, A.H., 1990. Transport of solutes into and out streambeds, Report No. KH-R-52, W.M. Keck Laboratory of Hydraulics and Water Resources, California Institute of Technology, Pasadena, CA.

Farley, K.J. and Morel, F.M.M., 1986. Role of coagulation in the kinetics of sedimentation. Environmental Science and Technology, 20, 187-195.

Farley, K. J., R. V. Thomann, T. F. Cooney III, D. R. Damiani and J. R. Wands, 1999. An integrated model of organic chemical fate and bioaccumulation in the Hudson River Estuary. Final Report to the Hudson River Foundation. Manhattan College, Riverdale, NY.

Farley, K.J., Miller, R.L., Saha, S., Douglas, W.S. and DiToro, D.M. (2004). "Bioaccumulation of PCB Homologs in New York-New Jersey Harbor Worms." Presentation, SETAC 25th Annual Meeting, Portland OR, November 14-18, 2004.

Farley, K.J., Wands, J.R., Damiani, D.R, and Cooney, T.F., 2005. "Transport, Fate and Bioaccumulation of PCBs in the Lower Hudson River," in The Hudson River Ecosystem, J. Levington, editor, (In Press).

Ferson, S., J. Hajagos, D.S. Myers and W.T. Tucker, 2005. *Constructor: Synthesizing Information about Uncertain Variables*. SAND2005-3769, Sandia National Laboratories, Albuquerque, NM.

Fisher, N. S., and Wang, W.-X. (1998). "Trophic transfer of silver to marine herbivores: A review of recent studies." Environ. Toxicol. Chem., 17(4), 562-571.

Fugate, D.C. and Friedrichs, C.T., 2002. Determining concentration and fall velocity of estuarine particle populations using ADV, OBS and LISST, Continental Shelf Research, 22(11), 1867-1886.

Gardinier, M. N. and T. B. Hoff. 1982. Diet of striped bass in Hudson River Estuary. New York Fish Game Journal 29:152-165.

Gerino, M., R.C. Aller, et al. 1998. "Comparison of different tracers and methods used to quantify bioturbation during a spring bloom: 234-thorium, luminophores and chlorophyll a." Estuarine Coastal and Shelf Sci. 46: 531-547.

Germano and Associates, Inc. 2005. Sediment profile imaging survey of sediment and benthic habitat characteristics of the Lower Passaic River, June 2005. Prepared for Aqua Survey, Inc. August 2005.

Geyer, W.R., Woodruff, J.D., and Traykovski, P., 2001. Sediment transport and trapping in the Hudson River estuary. Estuaries, Vol. 24 No. 5, p. 670-679.

Gobas, F. A. P. C., Zhang, X., and Wells, R. (1993). "Gastrointestinal magnification: The mechanism of biomagnification and food chain accumulation of organic chemicals." Environ. Sci. Technol., 27(13), 2855-2863.

Harris, C.K. and Wiberg, P.L., 2001. A two-dimensional, time-dependent model of suspended sediment transport and bed reworking for continental shelves. Computers & Geosciences, 27(6): 675-690.

Hunt, J.R. 1982. Particle Dynamics in Seawater: Implications for Predicting the Fate of Discharged Particles, Environ. Sci., Technol, Vol. 16, No. 6.

HydroQual, Inc., 1999a. Newton Creek Water Pollution Control Project East River Water Quality Plan, Report to NYCDEP. Task 10.0 System-Wide Eutrophication Model (SWEM) Sub-task 10.1 Construct SWEM. Prepared under subcontract to Greeley and Hansen, New York, NY.

HydroQual, Inc., 1999b. Newton Creek Water Pollution Control Project East River Water Quality Plan. Task 10.0 System-Wide Eutrophication Model (SWEM), Sub-tasks 10.1-10.7 reports prepared under contract to Greeley and Hansen, New York, NY for the City of New York Department of Environmental Protection.

HydroQual, Inc., 1999c. Newton Creek Water Pollution Control Project East River Water Quality Plan, Report to NYCDEP. Task 10.0 System-Wide Eutrophication Model (SWEM) Sub-task 10.2 Obtain and Reduce Loading/Water Quality Data. Prepared under subcontract to Greeley and Hansen, New York, NY.

HydroQual, Inc., 1999d. Newton Creek Water Pollution Control Project East River Water Quality Plan, Report to NYCDEP. Task 10.0 System-Wide Eutrophication Model (SWEM) Sub-task 10.4 Calibrate SWEM Water Quality. Sub-task 10.6 Validate SWEM Water Quality. Prepared under subcontract to Greeley and Hansen, New York, NY.

HydroQual, Inc., 1999e. Newton Creek Water Pollution Control Project East River Water Quality Plan, Report to NYCDEP. Task 10.0 System-Wide Eutrophication Model (SWEM) Sub-task 10.5 Apply SWEM for Preliminary Facility Design Prepared under subcontract to Greeley and Hansen, New York, NY.

HydroQual, Inc., 1999f. Newton Creek Water Pollution Control Project East River Water Quality Plan, Report to NYCDEP. Task 10.0 System-Wide Eutrophication Model (SWEM) Sub-task 10.7 Final Facility Design. Prepared under subcontract to Greeley and Hansen, New York, NY.

HydroQual, Inc., 2001. Newtown Creek Water Pollution Control Project East River Water Quality Plan, Task 10.0 System-Wide Eutrophication Model (SWEM), Sub-task 10.6 Validate SWEM Hydrodynamics, Report to NYCDEP. Prepared under subcontract to Greeley and Hansen, New York, NY.

HydroQual, Inc., 2002. Calibration Enhancement of the System-Wide Eutrophication Model (SWEM) in the New Jersey Tributaries, Report to NJDEP. Final Technical Report April 23, 2001 through July 31, 2002. Prepared under subcontract to Passaic Valley Sewerage Commissioners, Newark, NJ.

Jones, C. and W. Lick. 2001. "Contaminant Flux Due to Sediment Erosion." Estuarine and Coastal Modeling. Proceedings of the Seventh International Conference St. Petersburg, Florida, November 5-7, 2001, American Society of Civil Engineers, ASCE 280-293.

King, J.K., F.M. Saunders, R.F. Lee, and R.A. Jahnke, 1999. Coupling Mercury Methylation Rates to Sulfate Reduction Rates in Marine Sediments. Environmental Toxicology and Chemistry, 18(7): 1362-1369.

Kosson, D.S., LM. Shor, et al., 2000. "Mass transfer limitations on bioavailability of PAHs from contaminated estuarine sediments." American Chemical Society, 220: 85-ENVR Part 1 August 20.

Krone, R.B., 1962. Flume Studies of the Transport of Sediment in Estuarial Shoaling Processes. University of California Hydraulic Engineering Laboratory and Sanitary Engineering Research Laboratory, Berkeley, CA, 110 pp.

Lang, G. et al., 1989. Data Interpretation and Numerical Modeling of the Mud and Suspended Sediment Experiment 1985. Journal of Geophysical Research, 94(C10): 14,381-14,393.

Lavelle, J.W., Mofjeld, H.O. and Baker, E.T., 1984. An In Situ Erosion Rate for a Fine-Grained Marine Sediment. Journal of Geophysical Research, 89(C4): 6543-6552.

Lick, W., 1982. The Transport of Contaminants in the Great Lakes. Annual Review of Earth and Planetary Science, 10: 327-353.

Lick, W. and J. Lick, 1988. On the aggregation and disaggregation of fine-grained sediments. J. Great Lakes Res., Vol. 14(4), pp. 514-523.

Lick, W., J. Gailani, C. Jones, E. Hayter, L. Burkhard, J. McNeil, 2005. The Transport of Sediments and Contaminants in Surface Waters. Short Course Notes, University of California, Santa Barbara, January 2005.

Limno-Tech, Inc., 1998. Fox River and Green Bay PCB Fate and Transport Model Evaluation: Technical Memorandum 4a - Alternate Sediment Bed Handling in IPX Lower Fox River Model. (Draft) Ann Arbor, MI, May 11.

Litten, S., 2003. Contaminant Assessment and Reduction Project - CARP - Water. Bureau of Water Assessment and Management, Division of Water, New York State Department of Environmental Conservation.

Maa, J.P.-Y., Sanford, L.P. and Halka, J.P., 1998. Sediment resuspension characteristics in Baltimore Harbor, Maryland. Marine Geology, 146(1-4): 137-145.

Malcolm Pirnie, Inc. 2004. Historical data evaluation, Lower Passaic River restoration project. May 2004.

Malcolm Pirnie, Inc. 2005a. Lower Passaic River Restoration Project Field Sampling Plan Volume 1. Prepared in conjunction with Battelle, Inc. and HydroQual, Inc. December 2005.

Malcolm Pirnie, Inc. 2005b. Quality Assurance Project Plan. Lower Passaic River Restoration Project. Prepared in conjunction with Battelle, Inc. and HydroQual, Inc. August 2005.

Malcolm Pirnie, Inc. 2005c. Lower Passaic River Restoration Project Work Plan. Prepared in conjunction with Battelle, Inc. and HydroQual, Inc. August 2005.

Manning, A.J. and Dyer, K.R., (1999). A laboratory examination of floc characteristics with regard to turbulent shearing. Marine Geology, 160: 147-170.

Marvin-DiPasquale, M. and R.S. Oremland, 1998. Bacterial Methylmercury Degradation in Florida Everglade Peat Sediment, *Environmental Sci. and Technol.*, 32: 2556-2563.

Matisoff, G., 1982. Mathematical Models of Bioturbation. Animal-Sediment Relations. The Biogenic Alteration of Sediments. New York, Plenum Press. 289-330.

McLean, S.R., 1985. Theoretical Modelling of Deep Ocean Sediment Transport. Marine Geology, 66(1-4): 243-265.

Mikkelsen, O.A., and M. Pejrup. 2000. In situ particle size spectra and density of particle aggregates in a dredging plume. Mar. Geol., 170: 443-459.

Miller, D.R., N.P. Nikolaidis, L.H. Yang, M.A. Geigert, I. Heitert and H.S. Chen. 1993. Technical Report on the Long Island Sound Atmospheric Deposition Project. University of Connecticut, for Connecticut Dept. of Environmental Protection. July 1, 1993.

Morel, F.M.M., S.L. Schiff, R.M. 1980. Parsons Laboratory, Report 259, Massachusetts Institute of Technology: Cambridge, MA.

NOAA. 1972. Tide Tables, High and Low Water Prediction, East Coast of North American and South America including Greenland, U.S. Dept. of Commerce, National Oceanic Survey, Rockville, Maryland.

O'Connor, J. M. 1984. PCBs: Dietary dose and burdens in Striped Bass from the Hudson River. Northeastern Environmental Science 3(3/4):152-158.

Park, R. A. (1998). "AQUATOX for Windows: A Modular Toxic Effects Model for Aquatic Ecosystems." Eco Modeling, Montgomery Village, MD.

Partheniades, E. 1992. "Esturine Sediment Dynamics and Shoaling Processes" in Handbook of Coastal and Ocean Engineering, Vol. 3, J. Herbich, ed, pp 985-1071, Gulf Publishing Co., Houston, Tx.

Pence, A.M., 2004. Dominant forces in an estuarine complex with multiple tributaries and free connections to the open ocean with application to sediment transport, a PhD dissertation submitted to the Faculty of the Stevens Institute of Technology, 91 p.

Poje, G. V., S. A. Riordan and J. M. O'Connor. 1988. Food habits of the amphipod Gammarus tigrinus in the Hudson River and the effects of diet upon its growth and reproduction. P. 255-270. In C. L. Smith (ed.), Fisheries Research in the Hudson River. State University of New York Press, Albany, NY.

Reckhow, K.H., J. T. Clements, and R.C. Dodd. 1990. Statistical evaluation of mechanistic water-quality models, Journal of Environmental Engineering, Vol. 116, No. 2.

Roberts, J., Jepsen, R., Gotthard, D. and Lick, W., 1998. Effects of particle size and bulk density on erosion of quartz particles. Journal of Hydraulic Engineering-Asce, 124(12): 1261-1267.

Roman, M.E., Holliday, V.D., and Sanford, L.P. 2001. Temporal ans spatial patterns of zooplankton in the Chesapeake Bay turbidity maximum. Marine Ecology Progress, 213: 215-227.

Sanford, L.P., 1994. Wave-Forced Resuspension of Upper Chesapeake Bay Muds. Estuaries/ 17(1B) 148-165.

Sanford, L.P., Chang M-L. 1997. The Bottom Boundary Condition for Suspended Sediment Deposition. Journal of Coastal Research Special Issue 25:3-17.

Sanford, L.P. and Halka, J.P., 1993. Assessing the paradigm of mutually exclusive erosion and deposition of mud, with examples from upper Chesapeake Bay. Marine Geology, 114(1-2): 37-57.

Sanford, L.P. and Maa, J.P.-Y., 2001. A unified erosion formulation for fine sediments. Marine Geology, 179(1-2): 9-23.

Sanford, L.P., W. Panageotou, and J.P. Halka, 1991. Tidal resuspension of sediments in Chesapeake Bay. Marine Geology. 97 (1/2) 87-103.

Sanford, L.P. 2005. Uncertainties in sediment erodibility estimates due to a lack of standards for experimental protocols and data interpretation. Proceedings of the Third International Conference on Remediation of Contaminated Sediments, New Orleans, LA, January 2005.

Schluter, M., E. Sauter, H.-P. Hansen, and E. Suess, 2000. Seasonal variations of bioirrigation in coastal sediments: Modeling of field data. Geochim. Cosmochim. Acta 64: 821-834.

Schwab, D.J., J.R. Bennett, P.C. Liu, and M.A. Donelan. 1984. Application of a simple numerical wave prediction model to Lake Erie, J. Geophys. Res., 89(C3), 3586-3592.

Shor, L.M., K.J. Rockne, G.L. Taghon, L.Y. Young and D.S. Kosson, 2003. Desorption kinetics for field-aged polycyclic aromatic hydrocarbons from sediments. Environ. Sci. Technol. 37, 1535-1544.

Schwarzenbach, Rene P., P.M. Gschwend and D.M. Imboden, 1993. Environmental Organic Chemistry, John Wiley & Sons, New York.

Setzler, E. M., W. R. Boynton, K. V. Wood, H. H. Zion, L. Lubbers, N. K. Mountford, P. Fere, L. Tucker and J. A. Mihursky. 1980. Synopsis of biological data on striped bass, Morone saxatilis (Waldbaum). NMFS Cir. 433, FAO Synopsis No. 121, U.S. Department of Commerce, Rockville, MD.

Smoluchowski, M., 1916. Drei Vortrage Uorue diffusion brownsiche bewegung und koagulation von kolloidteilchen, Physick Zeitschrift, Vol. 17, pp. 557.

Smoluchowski, M., 1917. Versuch einer mathematischen theorie der koagulationskinetic killoider losungen, Z. Phys. Chem., Vol. 92, pp. 129-168.

TAMS/Gradient. 1995. Further site characterization and analysis database report. Phase 2 Report. EPA Contract No. 68-S9-2001, U.S. Environmental Protection Agency, Region 2.

Thomann, R.V., 1982. Verification of water quality models, ASCE, Vol. 108, No. EE5.

Thomann, R.V. and D.M. Di Toro. 1983. Physico-chemical model of toxic Substances in the Great Lakes, J. Great Lakes Res., 9(4): 474-496.

Thomann, R. V. and J. P. Connolly. 1984. Model of PCB in the Lake Michigan Trout food chain. Environmental Science and Technology 18(2):65-71.

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., J. A. Mueller, R. P. Winfield and C. -R. Huang. 1989. Mathematical model of the long-term behavior of PCBs in the Hudson River Estuary. Final Report to the Hudson River Foundation, Grant Numbers 007/87A/030, 011/88A/030, Manhattan College, Riverdale, NY.

Thomann, R. V., J. A. Mueller, R. P. Winfield and C. -R. Huang. 1991. Model of fate and accumulation of PCB homologues in Hudson Estuary. Journal of Environmental Engineering 117(2):161-178.

Thomann, R. V., J. P. Connolly and T. F. Parkerton. 1992a. An equilibrium model of organic chemical accumulation in aquatic food webs with sediment interaction. Environmental Toxicology and Chemistry 11:615-629.

Thomann, R. V., J. P. Connolly and T. F. Parkerton. 1992b. Modeling accumulation of organic chemicals in aquatic food webs. p 153-186. In F. A. P. C. Gobas and J. A. McCorquodale (eds.), Chemical Dynamics in Fresh Water Ecosystems. Lewis Publishers, Chelsea, MI.

Thomann, R. V., Mahony, J. D., and Mueller, R. (1995). "Steady-state model of biota sediment accumulation factor for metals in two marine bivalves." Environ. Toxicol. Chem., 14(11), 1989-1998.

Thomann, R. V., Shkreli, F., and Harrison, S. (1997). "A pharmacokinetic model of cadmium in rainbow trout." Environ. Toxicol. Chem., 16(11), 2268-2274.

Thomann, R. V., Snyder, C. A., and Squibb, K. S. (1994). "Development of a pharmacokinetic model for chromium in the rate following subchronic exposure: 1. The importance of incorporating long-term storage compartment." Toxicol. Appl. Pharmacol., 128, 189-198.

Thomann, R.V., J.A. Mueller, R.P. Winfield and C.R. Huang (1991), "Model of the fate and accumulation of PCB homologues in Hudson estuary," ASCEJ. Environ. Engr., 117:161-177.

Tierra Solutions Inc. 2003. Executive Summary: Passaic River Study Area: Preliminary Findings.

Tierra Solutions Inc. 2004. Newark Bay Work Plan - Supplement.

Tsay, T.K., G.J. Ruggaber, S.W. Effler, C.T. Driscoll. 1992. Thermal stratification modeling of lakes with sediment heat flux. J. of Hydraul. Eng., 118: 407-419.

USACE and USEPA, March 2006. "Sensitivity and Uncertainty Analysis," in *Model Validation: Modeling Study of PCB Contamination in the Housatonic River*, Volume 1, Section 5.

Valioulis, I.A., 1983-03-14. Particle collisions and coalescence in fluids, pH.D dissertation Caltech.

Van Ledden, M., 2002. A Process-based Sand-Mud Model. In: Fine Sediment Dynamics in the Marine Environment. In: K.C. Winterwerp J.C. (Editor). Elsevier Science B.V., pp. 577-594.

Van Rijn, L.C. 1984. Sediment Transport, Part II Suspended Load Transport, ASCE, J. Hydr. Engr., 110: 1613-1639.

Waldman, J. R. 1988. 1986 Hudson River striped bass tag recovery program. Hudson River Foundation, New York, NY.

Waldman, J. R., D. J. Dunning, Q. E. Ross, and M. T. Mattson. 1990. Range dynamics of Hudson River striped bass along the Atlantic coast. Transactions of the American Fisheries Society 119:910-919.

Warner, J.C., W.R. Geyer, and J.A. Lerczak. 2005. Numerical modeling of an estuary; A comprehensive skill assessment: Journal of Geophysical Research, v. 110, C05001, doi: 10. 1029/2004JC002691.

Winterwerp, J.C. and Van Kesteren, W.G.M., 2004. Introduction to the Physics of Cohesive Sediment in the Marine Environment, 56. Elsevier B.V., Amsterdam, The Netherlands, 466 pp.

WP MPI 2005. Lower Passaic River Restoration Project Draft Work Plan. Prepared by Malcolm-Pernie

Young, R. A. 1988. A Report on striped bass in New York marine waters. New York State Marine Fisheries, Stony Brook, NY.

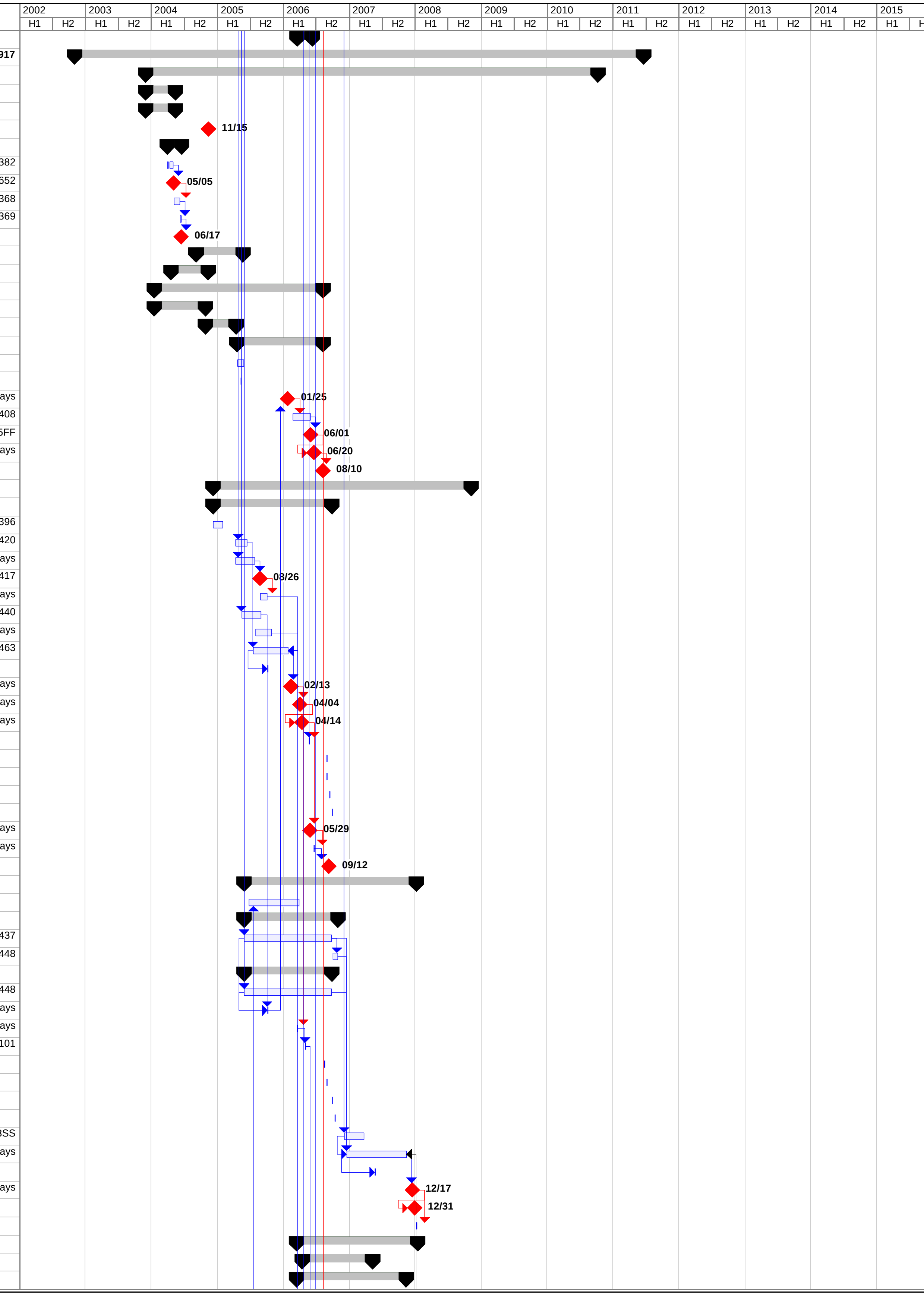
APPENDIX A

SCHEDULE

Date Updated: August 18, 2006

Lower Passaic River Restoration Project and Newark Bay RI/FS
Integrated Schedule
Working Draft

ID	Task Name	Duration	Responsibility	Current Start	Current Finish	Predecessors	Successors
346	**RTC/Final CIP**	**60 days**	**MPI**	**Mon 03/20/06**	**Mon 06/12/06**		
352	**Technical Studies and Investigations**	**2246 days**	**MPI, TAMS, BAT, HQI**	**Mon 11/04/02**	**Tue 06/21/11**		**917**
353	**Work Plan Preparation**	**1790 days**	**MPI, TAMS**	**Tue 12/02/03**	**Mon 10/11/10**		
354	**Agency Stakeholder Coordination/Scoping Meeting**	**119 days**	**EPA**	**Tue 12/02/03**	**Fri 05/14/04**		
355	**Agency Coordination and Scoping Meeting**	**119 days**	**Project Team**	**Tue 12/02/03**	**Fri 05/14/04**		
363	DESA Meeting	1 day	EPA, MPI, TAMS	Mon 11/15/04	Mon 11/15/04	523FS-7 days	
364	**Evaluation and Documentation of Surface Sediment Historical Data**	**57 days**	**MPI, BAT**	**Wed 03/31/04**	**Thu 06/17/04**		
365	Review of data/Preparation of Draft Technical Memorandum	20 days	MPI	Wed 03/31/04	Tue 05/04/04	947SS+10 days	366,382
366	Submit Draft Tech Memo to USEPA/USACE	1 day	MPI	Wed 05/05/04	Wed 05/05/04	365	367,652
367	Agency Review	25 days	USEPA/USACE	Thu 05/06/04	Wed 06/09/04	366	368
368	Preparation of Final Technical Memorandum	5 days	MPI	Thu 06/10/04	Wed 06/16/04	367	369
369	Submit Final Tech Memo to USEPA/USACE	1 day	MPI	Thu 06/17/04	Thu 06/17/04	368	
370	**Evaluation and Documentation of (Selected) Subsurface Sediment H**	**186 days**	**MPI, BAT**	**Tue 09/07/04**	**Tue 05/24/05**		
381	**Identify Draft DQOs/ARARs/PRGs**	**148 days**	**Agencies, MPI**	**Wed 04/21/04**	**Fri 11/12/04**		
386	**Modeling Plan**	**668 days**	**HQI**	**Tue 01/20/04**	**Thu 08/10/06**		
387	**Pre-Draft Modeling Plan/Discussion**	**202 days**	**HQI**	**Tue 01/20/04**	**Wed 10/27/04**		
392	**Revised Modeling Plan**	**121 days**	**HQI**	**Fri 10/29/04**	**Fri 04/15/05**		
403	**RTC/Final Modeling Plan**	**341 days**	**HQI**	**Thu 04/21/05**	**Thu 08/10/06**		
404	Review: Draft Modeling Plan (Agency/PRP)	28 days	Agencies, TAC, PRP	Thu 04/21/05	Mon 05/30/05	402FS+5 days	
405	Workgroup Meeting: Draft Modeling Plan	1 day	Project Team, TAC	Wed 05/11/05	Wed 05/11/05	402FS+17 days	
406	EPA submits collated comments to MPI/HQI	0 days	EPA	Wed 01/25/06	Wed 01/25/06	440FS+80 days	407FS+20 days
407	Prepare Final Modeling Plan	71 days	HQI	Thu 02/23/06	Thu 06/01/06	406FS+20 days	408
408	Submit Final Modeling Plan to EPA/USACE	0 days	HQI	Thu 06/01/06	Thu 06/01/06	407	409FS+13 days,1085FF
409	Final Comments to HQI	0 days	USEPA	Tue 06/20/06	Tue 06/20/06	408FS+13 days	410FS+37 days
410	Final Modeling Plan Approval & Posting to ourPassaic.org	0 days	EPA	Thu 08/10/06	Thu 08/10/06	409FS+37 days	
411	**Model Development & Calibration**	**1021 days**	**HQI**	**Fri 12/10/04**	**Fri 11/07/08**		
412	**Hydrodynamic Model (Not Fully Funded)**	**471 days**	**HQI**	**Fri 12/10/04**	**Fri 09/29/06**		
413	Background Document Review	40 days	HQI	Fri 12/10/04	Thu 02/03/05	393SS+30 days	396
414	Grid Design	50 days	HQI	Mon 04/11/05	Fri 06/17/05	192FS+6 days	420
415	Wetting/Drying Protocol	80 days	HQI	Mon 04/11/05	Fri 07/29/05	192FS+6 days	416FS+20 days
416	Deliver Hydrodynamic Model Code for Peer Testing	0 days	HQI	Fri 08/26/05	Fri 08/26/05	415FS+20 days	417
417	Peer Testing and Comment	28 days	USEPA	Mon 08/29/05	Wed 10/05/05	416	420FF+40 days
418	Multi-year Model Input	78 days	HQI	Mon 05/16/05	Wed 08/31/05	192FS+31 days	440
419	Bathymetric Data Review and Final Model Configuration	66 days	HQI, USACE	Mon 08/01/05	Mon 10/31/05		420FF+65 days
420	Hydrodynamic Transport Model Calibration	140 days	HQI	Tue 07/19/05	Mon 01/30/06	711FF+10 days,399FF,414,417FF+40 days,419FF+65 days	823FF,422FS+10 days,421SS+55 days,462,463
421	Workgroup Meeting - Hydrodynamic Model Mid-calibration	1 day	Agencies, TAC, PRP	Wed 10/05/05	Wed 10/05/05	420SS+55 days	
422	Submit Draft Hydrodynamic Calibration Report to MPI & Agencie	0 days	HQI	Mon 02/13/06	Mon 02/13/06	420FS+10 days	323FS+10 days,441FS+24 days,423FS+36 days
423	Agency Comments submitted to HQI and MPI	0 days	Agencies	Tue 04/04/06	Tue 04/04/06	422FS+36 days	424FS+8 days
424	Draft Hydro Calibration Report posted to ourPassaic.org	0 days	HQI, MPI	Fri 04/14/06	Fri 04/14/06	423FS+8 days	425FS+29 days,430FS+31 days
425	Workgroup Meeting - Draft Hydro Calibration Report	1 day	Agencies, TAC, PRP	Fri 05/26/06	Fri 05/26/06	196,424FS+29 days	
426	Implement Wind-wave model for Newark Bay	1 day	HQI	Thu 08/31/06	Thu 08/31/06		
427	Provide revised ECOMSED code/test bed to Earl Hayter for QA	1 day	HQI	Thu 08/31/06	Thu 08/31/06		
428	Establish 1984 Flood Condition	1 day	HQI	Fri 09/15/06	Fri 09/15/06		
429	Confirm revised hydrodynamic model against 2005 data set	1 day	HQI	Fri 09/29/06	Fri 09/29/06		
430	Stakeholder Comments on Draft Hydro Cal Report submitted to I	0 days	Stakeholders	Mon 05/29/06	Mon 05/29/06	424FS+31 days	431FS+10 days
431	USEPA Comment Meeting with HQI	1 day	USEPA, HQI	Tue 06/20/06	Tue 06/20/06	430FS+10 days	432FS+60 days
432	Submit Final Hydrodynamic Cal. Report (Date Approximate)	0 days	HQI	Tue 09/12/06	Tue 09/12/06	431FS+60 days	
433	**Sediment Transport Model (Not Fully Funded)**	**681 days**	**HQI**	**Wed 06/01/05**	**Wed 01/09/08**		
434	Review of Sedflume and Gust Experimental Data	200 days	HQI	Fri 06/24/05	Thu 03/30/06	704	
435	**Development of Bed Erosion/Consolidation Protocol**	**370 days**	**HQI**	**Wed 06/01/05**	**Tue 10/31/06**		
436	Develop bed consolidation protocol for newly deposited sol	348 days	HQI	Wed 06/01/05	Fri 09/29/06	192FS+43 days	440SS+90 days,448,437
437	Develop protocol for implementing morphology changes intc	22 days	HQI	Mon 10/02/06	Tue 10/31/06	436	448
438	**Development of Coagulation Protocol**	**348 days**	**HQI, TAC**	**Wed 06/01/05**	**Fri 09/29/06**		
439	Develop, apply, and confirm coagulation protocols	348 days	HQI	Wed 06/01/05	Fri 09/29/06	192FS+43 days	440SS+90 days,448
440	Modeling Workgroup Meeting - Hydrodynamic and Sediment Tra	1 day	Agencies, TAC, PRP	Wed 10/05/05	Wed 10/05/05	418,436SS+90 days,439SS+90 days	406FS+80 days
441	SEDZLJ Integration Meeting	2 days	HQI, Jones, Hayter, MPI	Mon 03/20/06	Tue 03/21/06	422FS+24 days	442FS+30 days
442	SEDZLJ Integration TAC Teleconference	1 day	EPA, ACE, HQI, MPI, TAC	Wed 05/03/06	Wed 05/03/06	441FS+30 days	1101
443	Implement SEDZLJ code within ECOMSED	1 day	HQI	Fri 08/18/06	Fri 08/18/06		
444	Provide revised ECOMSED code/test bed to Earl Hayter for QA	1 day	HQI	Thu 08/31/06	Thu 08/31/06		
445	Implement SEDZLJ code with RCA/CARP/LPR/NB fate and tran	1 day	HQI	Fri 09/29/06	Fri 09/29/06		
446	Provide RCA code/test bed to Earl Hayter for QA/QC	1 day	HQI	Fri 10/13/06	Fri 10/13/06		
447	Sediment Transport Inputs	80 days	HQI	Tue 12/05/06	Mon 03/26/07	192FS+43 days,708FF+30 days	448SS
448	Sediment Transport Model Calibration	239 days	HQI	Tue 12/19/06	Fri 11/16/07	714FF+60 days,447SS,709SS,436,437,439	450FS+21 days,449SS+112 days
449	Workgroup Meeting - Sediment Transport Model Mid-calibration	1 day	HQI	Thu 05/24/07	Thu 05/24/07	448SS+112 days	
450	Submit Draft Sediment Transport Calibration Report to MPI/EPA	0 days	HQI	Mon 12/17/07	Mon 12/17/07	448FS+21 days	452FS+16 days,451FS+10 days
451	Submit Draft Sediment Transport Calibration Report to Agencies	0 days	HQI	Mon 12/31/07	Mon 12/31/07	450FS+10 days	
452	Workgroup Meeting - Sediment Transport Calibration Report	1 day	Agencies, TAC, PRP	Wed 01/09/08	Wed 01/09/08	450FS+16 days	
453	**Fate and Transport Model (Subject to Avail. Funding)**	**480 days**	**HQI**	**Fri 03/17/06**	**Thu 01/17/08**		
454	**Organic Carbon Sub-model**	**280 days**	**HQI**	**Mon 04/17/06**	**Fri 05/11/07**		
459	**Contaminants of Concern Modeling**	**432 days**	**HQI**	**Fri 03/17/06**	**Mon 11/12/07**		

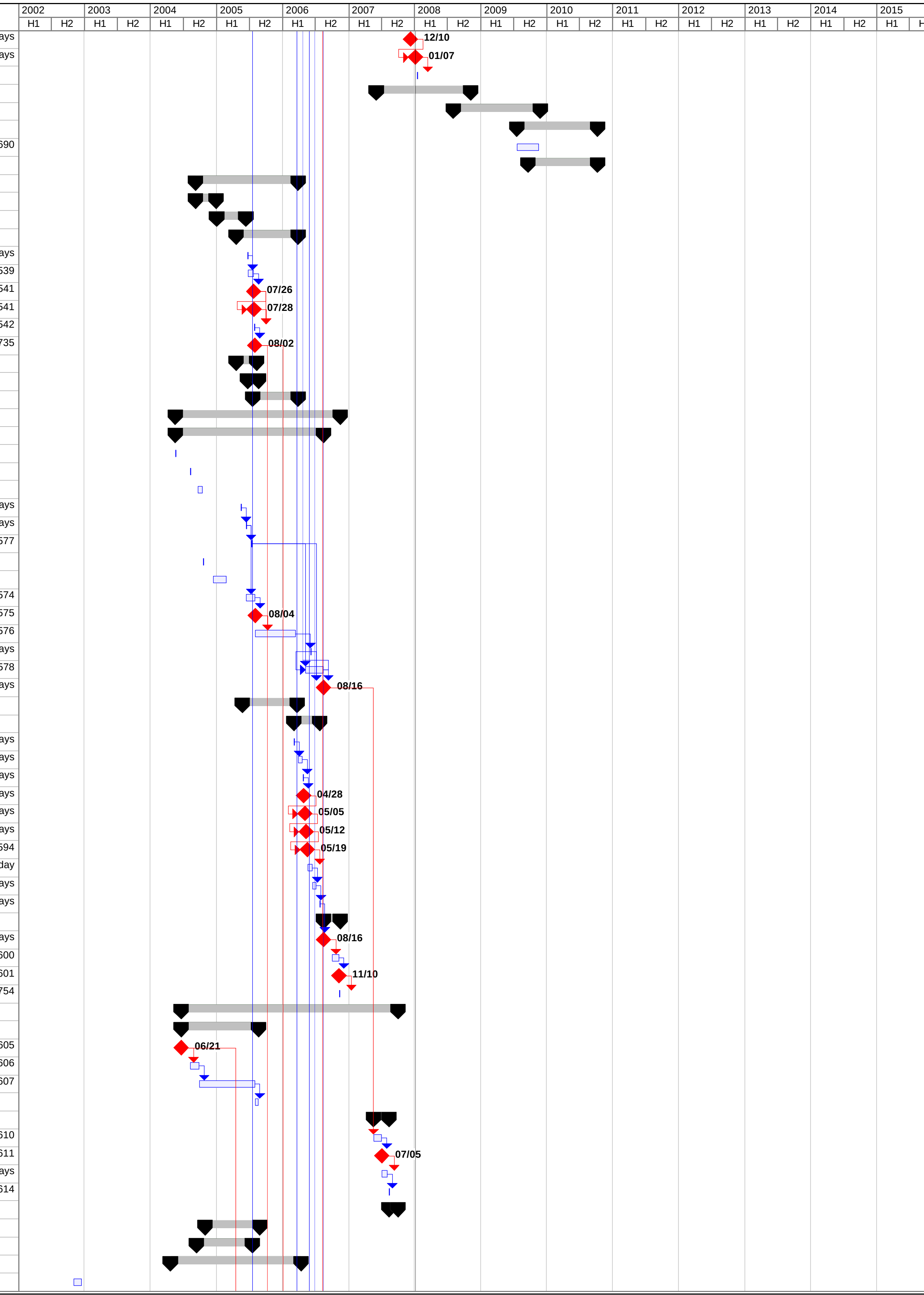


Overall Project Summary: WAD Level Summary: WO Level Summary: Task Summary: Task: Milestone:

Date Updated: August 18, 2006

Lower Passaic River Restoration Project and Newark Bay RI/FS
Integrated Schedule
Working Draft

ID	Task Name	Duration	Responsibility	Current Start	Current Finish	Predecessors	Successors
470	Submit Draft Fate and Transport Report to MPI/EPA	0 days	HQI	Mon 12/10/07	Mon 12/10/07	469FS+20 days	471FS+20 days
471	Submit Draft Fate and Transport Report to Agencies	0 days	HQI	Mon 01/07/08	Mon 01/07/08	470FS+20 days	472FS+7 days
472	Workgroup Meeting - Fate and Transport	1 day	Agencies, TAC, PRP	Thu 01/17/08	Thu 01/17/08	471FS+7 days	
473	**Food Chain Model (Subject to Avail. Funding)**	**374 days**	**HQI**	**Tue 06/05/07**	**Fri 11/07/08**		
481	**Model Calibration Report (Subject to Avail. Funding)**	**345 days**	**HQI**	**Mon 08/04/08**	**Fri 11/27/09**		
497	**Baseline Modeling (Subject to Avail. Funding)**	**319 days**	**HQI**	**Wed 07/22/09**	**Mon 10/11/10**		
498	Simulations: Baseline Modeling	88 days	HQI	Wed 07/22/09	Fri 11/20/09	493FS-44 days	500FS-44 days,670,690
499	**Baseline Model Reporting**	**275 days**	**HQI**	**Tue 09/22/09**	**Mon 10/11/10**		
517	**Field Sampling Plans/Work Plans: Volume 1 (Sediment and WQ)**	**405 days**	**MPI, BAT**	**Wed 09/08/04**	**Tue 03/28/06**		
518	**Pre-Draft WP/FSP Volume 1**	**83 days**	**MPI, BAT**	**Wed 09/08/04**	**Fri 12/31/04**		
527	**Draft WP/FSP Volume 1**	**115 days**	**MPI, BAT**	**Mon 01/03/05**	**Mon 06/13/05**		
536	**Final WP/FSP Volume 1**	**243 days**	**MPI, BAT**	**Fri 04/22/05**	**Tue 03/28/06**		
537	EPA submits collated comments to MPI	1 day	EPA	Thu 06/23/05	Thu 06/23/05	534FS+13 days	538,549,553FS+20 days
538	Prepare Pre-Final WP	23 days	MPI, BAT, HQI	Fri 06/24/05	Tue 07/26/05	537	539
539	Upload Pre-Final WP Text to PREmis (highlighted text)	0 days	MPI	Tue 07/26/05	Tue 07/26/05	538	540FS+2 days,541
540	Upload Pre-Final WP Plates to PREmis	0 days	MPI	Thu 07/28/05	Thu 07/28/05	539FS+2 days	541
541	USEPA and USACE Pre-Final WP Review and Comment Submi	3 days	EPA, MPI	Fri 07/29/05	Tue 08/02/05	540,539	542
542	Upload Revised Pre-Final WP to PREmis & ourPassaic.org	0 days	MPI	Tue 08/02/05	Tue 08/02/05	541	743,744,735
543	**FSP Volume 1: Water Column Sampling**	**81 days**	**MPI**	**Fri 04/22/05**	**Fri 08/12/05**		
548	**FSP Volume 1: High Resolution Coring**	**42 days**	**MPI**	**Fri 06/24/05**	**Mon 08/22/05**		
552	**FSP Volume 1: Low Resolution Coring**	**178 days**	**MPI**	**Fri 07/22/05**	**Tue 03/28/06**		
563	**Field Sampling Plan: Volume 2 (Ecological/Biological)**	**652 days**	**MPI, NY, TAMS, OMR**	**Thu 05/20/04**	**Fri 11/17/06**		
564	**WRDA Pre-Investigations**	**585 days**	**MPI, NY, TAMS, OMR**	**Thu 05/20/04**	**Wed 08/16/06**		
565	Restoration Workshop I: FSP Vol 2	1 day	MPI, NY, TAMS, OMR	Thu 05/20/04	Thu 05/20/04		
566	Restoration Workshop II: FSP Vol 2	1 day	MPI, NY, TAMS, OMR	Wed 08/11/04	Wed 08/11/04		
567	GIS Screening	21 days	NY	Mon 09/20/04	Mon 10/18/04		
568	Restoration Workshop III: FSP Vol 2	1 day	MPI, NY, TAMS, OMR	Wed 05/18/05	Wed 05/18/05		580FS+5 days,569FS+19 days
569	Workgroup Meeting: Restoration (Interim Mtg)	1 day	MPI, NY, TAMS, OMR	Wed 06/15/05	Wed 06/15/05	568FS+19 days	573,570FS+20 days
570	Workgroup Meeting: FSP Vol 2	1 day	MPI, NY, TAMS, OMR	Thu 07/14/05	Thu 07/14/05	569FS+20 days	578,577
571	Site Recon Round 1	3 days	MPI	Wed 10/20/04	Fri 10/22/04		
572	Site Recon Round 2	53 days	MPI	Tue 12/14/04	Thu 02/24/05		
573	Prepare Pre-Draft Site Selection and Screening Report	36 days	TAMS, MPI	Thu 06/16/05	Thu 08/04/05	569	574
574	Submit Pre-Draft Site Selection and Screening Report	0 days	TAMS, MPI	Thu 08/04/05	Thu 08/04/05	573	575
575	Review: Pre-Draft SS&SR	158 days	Agencies and Stakeholders	Fri 08/05/05	Tue 03/14/06	574	576
576	Workgroup Meeting: Restoration	1 day	MPI, NY, TAMS, OMR	Wed 06/07/06	Wed 06/07/06	575	577SS-20 days,578FS+10 days
577	Prepare Final Restoration Opportunities Report	71 days	TAMS, MPI	Wed 05/10/06	Wed 08/16/06	570,576SS-20 days	295FS+100 days,578
578	Submit Final Restoration Opportunities Report	0 days	TAMS, MPI	Wed 08/16/06	Wed 08/16/06	577,570,576FS+10 days	609FS+197 days
579	**Pre-Draft FSP Volume 2 (Ecological/Biological)**	**216 days**	**MPI, TAMS, BAT**	**Thu 05/26/05**	**Thu 03/23/06**		
586	**Draft FSP Volume 2 (Ecological/Biological)**	**101 days**	**MPI, TAMS, BAT**	**Wed 03/08/06**	**Wed 07/26/06**		
587	Project Team Kick-off Meeting	1 day	MPI, TAMS, BAT	Wed 03/08/06	Wed 03/08/06	584FS+18 days	588FS+15 days
588	Prepare DQOs for Biota Sampling	17 days	MPI, TAMS, BAT, HQI	Thu 03/30/06	Fri 04/21/06	587FS+15 days	589FS+2 days
589	DQO and Preliminary Sample Plan Meeting	1 day	MPI, TAMS, BAT, HQI	Wed 04/26/06	Wed 04/26/06	588FS+2 days	590FS+2 days
590	Submit DQOs to USEPA and BTAG for Review	0 days	MPI, TAMS, BAT	Fri 04/28/06	Fri 04/28/06	589FS+2 days	591FS+5 days
591	Draft FSP Vol 2 Elements due to MPI	0 days	MPI, TAMS, BAT	Fri 05/05/06	Fri 05/05/06	590FS+5 days	592FS+5 days
592	Draft FSP 2 Submitted to Consultant Team for Internal Review	0 days	MPI	Fri 05/12/06	Fri 05/12/06	591FS+5 days	593FS+5 days
593	Submit Draft FSP Vol 2 to EPA/USACE/BTAG	0 days	MPI	Fri 05/19/06	Fri 05/19/06	592FS+5 days	594
594	Final Check of Draft Language/Revisions if needed	19 days	EPA, MPI	Mon 05/22/06	Thu 06/15/06	593	595FS+1 day
595	Review of Draft FSP Vol 2: PRP	15 days	PRP	Mon 06/19/06	Fri 07/07/06	594FS+1 day	596FS+12 days
596	Draft FSP 2 Workgroup Meeting	1 day	Project Team	Wed 07/26/06	Wed 07/26/06	595FS+12 days	598FS+15 days
597	**Final FSP Volume 2 (subject to WRDA funding)**	**67 days**	**MPI, TAMS, BAT**	**Wed 08/16/06**	**Fri 11/17/06**		
598	Submit collated Draft FSP Vol 2 comments to MPI	0 days	Agencies	Wed 08/16/06	Wed 08/16/06	596FS+15 days	599FS+32 days
599	Prepare Final FSP Vol 2 (currently not funded)	30 days	MPI, TAMS, BAT	Mon 10/02/06	Fri 11/10/06	598FS+32 days	600
600	Submit Final FSP Vol 2 for EPA Legal Review	0 days	MPI, TAMS	Fri 11/10/06	Fri 11/10/06	599	601
601	Submit Final FSP Vol 2 For Public Release	5 days	Agencies	Mon 11/13/06	Fri 11/17/06	600	755,752,753,754
602	**Field Sampling Plan: Volume 3 (Geophysical & WRDA Activities)**	**857 days**	**MPI, TAMS, NY, GP**	**Mon 06/21/04**	**Tue 10/02/07**		
603	**Pre-Draft FSP Volume 3**	**307 days**		**Mon 06/21/04**	**Tue 08/23/05**		
604	Prepare Pre-Draft FSP Vol 3	36 days	MPI, TAMS, NY, GP	Mon 06/21/04	Mon 08/09/04		746FS+180 days,605
605	Submit Pre-Draft FSP Vol 3	36 days	MPI, TAMS, NY, GP	Tue 08/10/04	Tue 09/28/04	604	606
606	Review FSP Vol 3	220 days	Agencies	Wed 09/29/04	Tue 08/02/05	605	607
607	Prepare Revised Pre-Draft FSP Volume 3	15 days	MPI	Wed 08/03/05	Tue 08/23/05	606	
608	**Draft FSP Volume 3 (Subject to WRDA Funding)**	**62 days**	**MPI, TAMS, NY, GP**	**Mon 05/21/07**	**Tue 08/14/07**		
609	Prepare Draft FSP Vol 3	34 days	MPI, TAMS, NY, GP	Mon 05/21/07	Thu 07/05/07	578FS+197 days	610
610	Submit Draft FSP Vol 3	0 days	MPI, TAMS, NY, GP	Thu 07/05/07	Thu 07/05/07	609	611
611	Review: Draft FSP Vol 2 - Agency/PRP	22 days	Agencies, NE, PRP	Fri 07/06/07	Mon 08/06/07	610	612FS+5 days
612	Workgroup Meeting (Tentative): Draft FSP Vol 3	1 day	Agencies, PRP, MPI, TAMS	Tue 08/14/07	Tue 08/14/07	611FS+5 days	614
613	**Final FSP Volume 3**	**35 days**	**MPI, TAMS, NY, GP**	**Wed 08/15/07**	**Tue 10/02/07**		
616	**Quality Assurance Project Plan (QAPP)**	**217 days**	**MPI, BAT**	**Mon 11/01/04**	**Tue 08/30/05**		
637	**Health and Safety Plan (HASP)**	**221 days**	**MPI**	**Tue 09/14/04**	**Tue 07/19/05**		
651	**Preliminary Risk Assessment**	**517 days**	**BAT, MPI**	**Thu 04/22/04**	**Fri 04/14/06**		
661	Focused Feasibility Study Risk Assessment Report	33 days	BAT, MPI	Mon 11/04/02	Thu 12/19/02		



Overall Project Summary: WAD Level Summary: WO Level Summary: Task Summary: Task: Milestone:

APPENDIX B

ECOM HYDRODYNAMIC MODELING FRAMEWORK

The hydrodynamic model to be used in this study is a three-dimensional, time-dependent, estuarine and coastal circulation model (ECOM) developed by Blumberg and Mellor (1980 and 1987). The model incorporates the Mellor and Yamada (1982) level 2-1/2 turbulent closure model to provide a realistic parameterization of vertical mixing. A system of curvilinear coordinates is used in the horizontal direction, which allows for a smooth and accurate representation of variable shoreline geometry. In the vertical scale, the model uses a transformed coordinate system known as the σ-coordinate transformation to permit better representation of bottom topography. Water surface elevation, water velocity (in three dimensions), temperature and salinity; and water turbulence are calculated in response to weather conditions (wind and incident solar radiation), freshwater inflows and tides, temperature and salinity in open boundaries connected to the coastal waters.

The model solves a coupled system of differential, prognostic equations describing the conservation of mass, momentum, temperature, salinity, turbulence energy and turbulence macroscale. The governing equations for velocity U_i = (u, v, w), temperature (T), salinity (S), and x_i = (x,y,z) are as follows:

$$\frac{\partial U_i}{\partial x_i} = 0 \tag{1}$$

$$\frac{\partial}{\partial t}(U,V) + \frac{\partial}{\partial x_i}\left[U_i(u,v) + f(-v,u)\right]$$
$$= -\frac{1}{\rho_o}\left[\frac{\partial P}{\partial x}, \frac{\partial P}{\partial y}\right] + \frac{\partial}{\partial z}\left[K_M \frac{\partial}{\partial z}(u,v)\right] + \left(F_U, F_V\right) \tag{2}$$

$$\frac{\partial T}{\partial t} + \frac{\partial}{\partial x_i}\left(U_i T\right) = \frac{\partial}{\partial z}\left[K_H \frac{\partial T}{\partial z}\right] + F_T \tag{3}$$

$$\frac{\partial S}{\partial t} + \frac{\partial}{\partial x_i}\left(U_i S\right) = \frac{\partial}{\partial z}\left[K_H \frac{\partial S}{\partial z}\right] + F_S \tag{4}$$

The horizontal diffusion terms, (F_U, F_V), F_T and F_S, in equations 2 through 4 are calculated using a Smagorinsky (1963) horizontal diffusion formulation (Mellor and Blumberg, 1985). The hydrostatic approximation yields:

$$\frac{P}{\rho_o} = g(\eta - z) + \int_z^\eta g\,\frac{\rho' - \rho_o}{\rho_o}\,dz' \tag{5}$$

where P is pressure, z is water depth, η(x,y,t) is the free surface elevation, ρ_o is a reference density, and $\rho = \rho(T,S)$ is the density, which is a function of T and S, as defined by Fofonoff (1962).

The vertical mixing coefficients, K_M and K_H, in Equations (2) through (4) are obtained by appealing to a 2 ½ order turbulence closure scheme and are given by:

$$K_M = \hat{K}_M + \upsilon_M,\ K_H = \hat{K}_H + \upsilon_H \tag{6}$$

$$\hat{K}_M = q\ell S_M,\ \hat{K}_H = q\ell S_H \tag{7}$$

where $q^2/2$ is the turbulent kinetic energy, ℓ is a turbulence length scale, S_M and S_H are stability functions defined by solutions to algebraic equations given by Mellor and Yamada (1982) as modified by Galperin et al. (1988), and υ_M and υ_H are constants. The variables q^2 and l are determined from the following equations:

$$\frac{\partial q^2}{\partial t} + \frac{\partial (uq^2)}{\partial x} + \frac{\partial (vq^2)}{\partial y} + \frac{\partial (wq^2)}{\partial z} = \frac{\partial}{\partial z}\left[K_q \frac{\partial q^2}{\partial z}\right]$$

$$+2K_M\left[\left(\frac{\partial u}{\partial z}\right)^2 + \left(\frac{\partial v}{\partial z}\right)^2\right] + \frac{2g}{\rho_o} K_H \frac{\partial \rho}{\partial z} - 2\frac{q^3}{B_1 \ell} + F_q \tag{8}$$

$$\frac{\partial (q^2\ell)}{\partial t} + \frac{\partial (uq^2\ell)}{\partial x} + \frac{\partial (vq^2\ell)}{\partial y} + \frac{\partial (wq^2\ell)}{\partial z} = \frac{\partial}{\partial z}\left[K_q \frac{\partial (q^2\ell)}{\partial z}\right]$$

$$+E_1\ell\left\{K_M\left[\left(\frac{\partial u}{\partial z}\right)^2+\left(\frac{\partial v}{\partial z}\right)^2\right]+\frac{g}{\rho_o}K_H\frac{\partial \rho}{\partial z}\right\}-\frac{q^3}{B_1\tilde{\omega}}+F_\ell \tag{9}$$

where, $Kq = 0.2q\ell$, the eddy diffusion coefficient for turbulent kinetic energy; F_q and F_ℓ represent horizontal diffusion of the turbulent kinetic energy and turbulence length scale and are parameterized in a manner analogous to either Equation 6 or 7; $\tilde{\omega}$ is a wall proximity function defined as $\tilde{\omega} = 1 + E_2 (\ell/\kappa L)^2$, $(L)^{-1} = (\eta - z)^{-1} + (H + z)^{-1}$, κ is the von Karman constant, H is the water depth, η is the free surface elevation, and E_1, E_2 and B_1 are empirical constants set in the closure model.

The basic equations, 1 through 9, are transformed into a terrain following σ-coordinate system in the vertical scale and an orthogonal curvilinear coordinate system in the horizontal scale. The resulting equations are vertically integrated to extract barotropic variables; and a mode splitting technique is introduced such that the fast-moving, external barotropic modes and relatively much-slower internal baroclinic modes are calculated by prognostic equations with different time steps. Detailed solution techniques are described in Blumberg and Mellor (1987).

The skill of ECOM has been extensively assessed in many studies of estuarine and coastal ocean regions. Previous recent applications include Chesapeake Bay (Blumberg and Goodrich, 1990), Massachusetts Bay (Blumberg et al., 1993), Georges Bank (Chen and Beardsley, 1995), the Oregon Continental Shelf (Allen et al., 1995), and the Gulf Stream Region (Ezer and Mellor, 1992). In recently completed studies by Blumberg et al. (1999) in New York Harbor and Connolly et al. (1999) in Mamala Bay, the model was validated using an extensive data set involving water surface elevations, currents, temperature, salinities, and three indicator organisms and four pathogens associated with fecal contamination.

APPENDIX C

HEAT FLUX COMPUTATIONS IN ECOM FRAMEWORK

Shortwave solar radiation is the radiant energy, which passes directly from the sun to the earth. ECOM model uses the measured solar radiation provided by the users. When observed solar radiation is not available, the model computes the solar radiation based on the formulation provided by the Smithsonian Meteorological Tables (List 1958). Although more than half of the solar radiation that enters the water body can be absorbed within the top meter, the remaining fraction that penetrates can have a significant effect on the development of the thermal structure (Rosati and Miyakoda, 1988). It has been found that the model simulation is particularly very sensitive to various optical water types and turbidity due to sediments and other water borne organic and inorganic substances including phytoplankton. Therefore proper parameterization of downward irradiance is crucial for accurate predictions of upper water body thermal structure. ECOM model allows a portion of the solar radiation being absorbed in the upper model layers and the rest are penetrated through the water column exponentially using spatially variable extinction coefficient. The spatially variable extinction coefficient represents a water body that has non-uniform turbidity.

The net atmospheric longwave radiation at the surface is the result of two processes: the downward radiation from the atmosphere and the upward radiation emitted by the water surface. Atmospheric radiation depends primarily on the air temperature, humidity, and cloud cover. The magnitude of the atmospheric radiation largely depends on the moisture content of the air and constitutes the major component of heat exchange processes during night and cloudy conditions (Ahsan and Blumberg, 1999; Adams et al., 1981; and Edinger et al., 1974). The physics of the longwave radiation is simply a black body radiation. The computation of downflux considers the effects of changes in atmospheric temperature, humidity, cloud, aerosol distribution, carbon dioxide, and other atmospheric constituents. Like Ahsan and Blumberg (1999) and Adams et al., (1981), a Swinbank (1963) formulation has been used in ECOM model, which suggests that saturation vapor pressure (e_a) is strongly correlated with the air temperature (T_a) and evaluates the downflux as a function of T_a alone. The net atmospheric flux is given as

$$H_a = \varepsilon\sigma\left((9.37 \text{ x } 10^{-6} T_a^6)(1 + 0.17C^2) - T_s^4\right)$$

Here H_a = net longwave atmospheric radiations (Watt m^{-2})

ε = emissivity of the water body (0.97)

σ = Stefan-Boltzmann constant (5.67×10^{-8} Watt $m^{-2}K^{-4}$)

T_a = atmospheric temperature in ^oK

T_s = water temperature in ^oK

C = cloud fraction (0-1)

Swinbank's formulation is sometimes found more attractive when surface humidity observations are not as readily available as air temperatures. This may also be attractive when a meteorological station is too far from the lake and may not provide site representative relative humidity data.

Sensible heat flux can occur between the atmosphere and a water body through conduction. The direction of the heat flux may be in either way depending on the sense of the temperature differences between the air and the water body. It has been shown (Edinger et al., 1974) that the daily rate of heat conduction is about an order of magnitude less than other dominant processes. The flux of conduction heat, incorporated in ECOM framework, is parameterized using a bulk transfer formula with dependencies on wind speed as suggested by Ahsan and Blumberg (1999) and Edinger et al., (1974). The conduction heat flux is given as follows:

$$H_c = C_c\, f(W)\, (T_s - T_a)$$

where H_c = Sensible (conduction) heat fluxes Watt m^{-2}

C_c = Bowen's coefficient (0.62 mb/K)

$f(W)$ = wind speed function defined as $a_0 + a_1W + a_2W$ (Watt $m^{-2}\,mb^{-1}$)

T_s and T_a are water and air temperature respectively as defined earlier

The coefficients a_0, a_1 and a_2 are chosen based on Brady et al., (1969) and suggested by Ahsan and Blumberg (1999) and Edinger et al., (1974).

The evaporative or latent heat flux is related to the conductive heat fluxes by the Bowen ratio and can be given as a function of wind speed and the difference between the saturated water vapor pressure at the water surface temperature and the water vapor pressure in the overlying air (Ahsan and Blumberg (1999) and Edinger et al., 1974). The evaporative heat flux is given as follows:

$$H_e = f(W)\, (e_s - e_a)$$

where H_e = evaporative heat flux (Watt m^{-2}),

e_s = saturated vapor pressure at temperature T_s (mb),

e_a = air-vapor pressure at temperature T_a (mb).

Significant discrepancies in formulating wind speed function have been reported in the latter studies, suggesting a wide variety of opinions among researchers. Suggestions have been made, whether conduction processes will remain to a negligible molecular scale in absence of wind or other small scale processes such as conduction currents due to density instabilities may dominate. The latter concept gained significant favors due to the fact that density instabilities exist during conduction and evaporation from thermally loaded water surface or during night when air temperature may be less than the water temperature. Following Brady et al. (1969) and Edinger et al. (1974) a slightly conservative formulation has been adopted in the ECOM framework:

$$f\ (W) = 6.9 + 0.345\ W^2 \quad (Wm^{-2}\ \ mb^{-1})$$

Where W is wind speed in m/s measured at 7 m above the water surface. For both the sensible and evaporative heat flux computations the evaporative wind speed function f(W) is a somewhat uncertain parameter (Cole and Buchak, 1995). Various formulations of f(W) have been examined in Edinger et al.(1974). Cole and Buchak (1995) termed the wind speed in this function as "ventilation speed" rather than a vector velocity speed as used in the wind stress computations. This ventilation speed is somewhat lower than the actual wind speed measured in a distant land based meteorological station, which accounts for the sheltering and canopy effect by the surroundings of a water body. A wind shelter coefficient has been introduced by Cole and Buchak (1995) having a range of 0 to 1 depending on the shape and size of the water body. For the James River and Farrar Gut model a shelter coefficient of 0.5 has been used for both of the simulation year of 1998.

REFERENCES

Adams, E. E., D. R. F. Harleman, G. H. Jirka, K. D. Stolzenbach, 1981."Heat Disposal in the Water Environment", Ralph M. Parsons Laboratory for Water resources and Hydrodynamics, Department of Civil Engineering, MIT, Cambridge, MA 02139.

Ahsan, Q., A. F. Blumberg, 1999. "Three-Dimensional Hydrothermal Model of Onondaga Lake, New York", Journal of Hydraulic Engineering, Vol 125, No 9, September 1999.

Allen, J.S., P.A. Newberger and J. Federiuk, 1995. Upwelling Circulation on the Oregon Continental Shelf, J. Phys. Oceanogr., 35, 1843-1889

Blayo, E., T. Mailly, B. Barnier, P. Brasseur, C. Le Provost, J. M. Molines and J. Verron, Complementarity of ERS-1 and Topex/Poseidon altimeter data in evaluating the oceanic circulation: assimilation in a model of the North Atlantic, Journal of Geophysical Research, submitted, 1996.

Blumberg, A. F. and D. M. Goodrich, 1990. Modeling of Wind-Induced Destratification in Chesapeake Bay, Estuaries, 13(3), 236-249.

Blumberg, A.F. and L.A. Khan, and J.P. St. John, "Three Dimensional Hydrodynamic Model of New York Harbor Region", J. Hydraulic Engineering, 125, 799-816, 1999

Blumberg, A.F. and G.L. Mellor, "A Coastal Ocean Numerical Model," In: Mathematical Modelling of Estuarine Physics, Proceedings of an International Symposium, Hamburg, August 24-26, 1978. J. Sundermann and K.P. Holz, Eds., Springer-Verlag, Berlin, 1980.

Blumberg, A.F. and G.L. Mellor, "A Description of a Three-Dimensional Coastal Ocean Circulation Model," In: Three-Dimensional Coastal Ocean Models, N. Heaps, Ed., 1-16, American Geophys. Union, Washington, D.C., 1987.

Blumberg, A. F., R. P. Signell and H. L. Jenter, 1993. Modeling Transport Processes in the Coastal Ocean, J. Marine Env. Engr., 1, 3-52.

Brady, D. K, Brooks, A.' S., and Buske, N. L. (1969).' "Future use of Chesapeake Bay for cooling thermal discharges" *Water* Sci. *and Mgmt. Seminar Rep.,* Johns Hopkins University, Baltimore.

Chartrand, (2003). A Geostatistical Assessment of Metals in Passaic River Sediments. Fields Group, USEPA Region 5.

Chen, C., R.C. Beardsley, R. Limeburner, 1995. "A Numerical Study of Stratified Tidal Rectification over Finite-Amplitude Banks. Part II: Georges Bank, J. Phys. Oceanogr., 25, 2111 - 2128.

Cole, T. M., and Buchak, E. M. (1995). "CE-QUAL-W2: a two-dimensional, laterally averaged, hydrodynamic and water quality model, version 2.0 user manuaL" *Instruction Rep. EL-95-I,* U.S. Army Corps of Engineers, Washington, D.C.

Connolly,P.J., A.F. Blumberg, and J.D. Quadrini, 1999, Modeling Fate of Pathogenic Organisms in Coastal Waters of Oahu, Hawaii, J. Envir. Engrg., 125:398-406

Edinger, J. E., Brady, D. K., and Greyer, J. C. (1974). "Heat exchange and transport in the environment." *Rep. No.* 14. *Cooling Water Res. Project (RP-49),* Electric Power Research Institute, Palo Alto, Calif.

Ezer, T. and G.L. Mellor, 1992. "A Numerical Study of the Variability and the Separation of the Gulf Stream, Induced by Surface Atmosphere Forcing and Lateral Boundary Flows", J. Phys. Oceanogr., 22, 660-682.

Galperin, B., L. H. Kantha, S. Hassid and A. Rosati, 1988. A Quasi-Equilibrium Turbulent Energy Model for Geophysical Flows, J. Atmos. Sci., 45, 55-62.

HydroQual, Inc., 1999a. Newtown Creek Water Pollution Control Project, East River Water Quality Plan, Report to NYCDEP. Task 10.0 System-Wide Eutrophication Model (SWEM), Sub-task 10.1 Construct SWEM. Prepared under subcontract to Greeley and Hansen, New York, NY.

HydroQual, Inc., 1999b. Newtown Creek Water Pollution Control Project, East River Water Quality Plan, Task 10.0 System-Wide Eutrophication Model (SWEM), Sub-tasks 10.1-10.7 reports prepared under subcontract to Greeley and Hansen, New York, NY.

HydroQual, Inc., 1999c. Newtown Creek Water Pollution Control Project, East River Water Quality Plan, Report to NYCDEP. Task 10.0 System-Wide Eutrophication Model (SWEM), Sub-task 10.2 Obtain and Reduce Loading/Water Quality Data. Prepared under subcontract to Greeley and Hansen, New York, NY.

HydroQual, Inc., 1999d. Newtown Creek Water Pollution Control Project, East River Water Quality Plan, Report to NYCDEP. Task 10.0 System-Wide Eutrophication Model (SWEM), Sub-task 10.4 Calibrate SWEM Water Quality. Sub-task 10.6 Validate SWEM Water Quality. Prepared under subcontract to Greeley and Hansen, New York, NY.

HydroQual, Inc., 1999e. Newtown Creek Water Pollution Control Project, East River Water Quality Plan, Report to NYCDEP. Task 10.0 System-Wide Eutrophication Model (SWEM), Sub-task 10.5 Apply SWEM for Preliminary Facility Design. Prepared under subcontract to Greeley and Hansen, New York, NY.

HydroQual, Inc., 1999f. Newtown Creek Water Pollution Control Project, East River Water Quality Plan, Report to NYCDEP. Task 10.0 System-Wide Eutrophication Model (SWEM), Sub-task 10.7 Final Facility Design Prepared under subcontract to Greeley and Hansen, New York, NY.

HydroQual, Inc., 1999g. Newtown Creek Water Pollution Control Project, East River Water Quality Plan, Report to NYCDEP. Task 10.0 System-Wide Eutrophication Model (SWEM), Sub-task 10.6 Validate SWEM Hydrodynamics Prepared under subcontract to Greeley and Hansen, New York, NY.

HydroQual, Inc., 2002. "Newtown Creek Water Pollution Control Project, East River Water Quality Plan, Task 10.0 System Wide Eutrophication Model (SWEM), Validate SWEM Hydrodynamics", City of New York, Dept. of Env. Protection, New York, NY", Mahwah, New Jersey

HydroQual, Inc., 2002. Calibration Enhancement of the System-Wide Eutrophication Model (SWEM) in the New Jersey Tributaries, Report to NJDEP. Final Technical Report April 23, 2001 through July 31, 2002. prepared under subcontract to Passaic Valley Sewerage Commissioners, Newark, NJ.

List, R. J., (1958), "Smithsonian Meteorological Tables." Smithsonian Institutions, Washington, DC, 527 pp.

Mellor, G.L. and T. Yamada, "Development of a Turbulence Closure Model for Geophysical Fluid Problems," Rev. Geophys. Space Phys., 20, 851-875, 1982.

Mellor, G.L. and A.F. Blumberg, "Modeling Vertical and Horizontal Viscosity and the Sigma Coordinate System," Mon. Wea. Rev., 113, 1379-1383, 1985.

Pence, A.M., 2004, "Dominant Forces in an Estuarine Complex with Multiple Tributaries and Free Connections to the Open Ocean with Application to Sediment Transport", Ph.D Thesis, Stevens Institutes of Technology, Hoboken, NJ

Roman, M.R., D.V. Holliday, L.P. Sandford, 2001. "Temporal and Spatial Patterns of Zooplankton in the Chesapeake Bay Turbidity Maximum," *Marine Ecology Progress Series*, 213: 215-227.

Rosati, A., and Miyakoda, K. (1988). "A general circulation model for upper ocean simulation." *J. Phys. Oceanogr.,* 18, 1601-1626.

Schwab, D.J., J.R. Bennett, P.C. Liu, and M.A. Donelan, 1984. Application of a simple numerical wave prediction model to Lake Erie. J. Geophys. Res., 89(C3), 3586-3589.

Simon Litten (2003). Contaminant Assessment and Reduction Project – CARP – Water. Bureau of Water Assessment and Management, Division of Water,New York State Department of Environmental Conservation. August 2003.

Smagorinsky, J., "General Circulation Experiments with the Primitive Equations, I. The Basic Experiment," Mon. Weather Rev., 91, 99-164. 1963.

Swinbank, W. C. (1963). "Longwave radiation from clear skies." *Quarterly J. Royal Meteorol. Soc.,* 89, 339-348.

Thomann,R.V., J.P. Connolly, T.F. Parkerton, 1992. Modeling accumulation of organic chemcials in aquatic food webs. In: Gobas, F.A.P.C. and J.A. McCorquodale, Eds. Chemical dynamics in fresh water system. Boca Raton, FL. Lewis Publishers. 247p.

Tierra Solutions Inc. (2003). Executive Summary: Passaic River Study Area: Preliminary Findings.

Tsay, T., G.J. Ruggaber, S.W. Effler, and C.T. Driscoll, 1992, "Thermal Stratification Modeling of Lakes with Sediment Heat Flux, J. of Hydraulic Eng. 118, 407-419.

APPENDIX D

NSL

AN EMPIRICAL METHOD FOR ESTIMATING SUSPENDED SEDIMENT LOADS IN RIVERS

A.1 INTRODUCTION

Engineers and scientists studying riverine systems must frequently estimate suspended sediment loads. Two examples illustrate the importance of accurately determined sediment loads in rivers. First, net annual deposition is a primary factor controlling the long-term fate of hydrophobic organic chemicals, e.g., PCBs and dioxin, in rivers and the burial rate is greatly affected by the annual sediment load. Secondly, reservoir sedimentation is a problem for which accurately determining the total sediment load delivered to a reservoir over long time periods by the tributary river, or rivers, is critical for predicting changes in the storage capacity of a reservoir.

Other types of problems, in addition to the previous examples, are routinely encountered that require accurate hindcasts or forecasts of sediment discharge rates on seasonal or annual time scales. In many cases, the loading time history needs to be specified in addition to the total mass of suspended sediment discharged by a river over a particular period. This requirement means that the estimated sediment loading should reflect the observed behavior of rivers wherein a large fraction of the annual sediment load is transported during a relatively small number of high flow events, or floods, each year [Walling et al., 1992].

The difficulty of accurately measuring sediment loads in rivers, particularly during floods, is well known [Walling and Webb, 1981; Thomas, 1985; Ferguson, 1987; Walling et al., 1992]. Data collection problems are related to the importance of flood-period sediment discharge to the annual load. Suspended sediment sampling programs must be carefully designed if accurate loading data are to be obtained and particular emphasis must be placed on sediment loading during high flow events.

Even if accurate sediment discharge data are available for a river that is of interest for a specific study, these data are usually collected during a limited period of time. Investigators are frequently faced with the task of using a restricted sediment loading data set to predict the response of a river during periods when no data are available. A variety of procedures have been used to predict suspended sediment discharge based upon existing data [Ferguson, 1987; Parker and Troutman, 1989]. The most widely used approach is the sediment rating curve, which is a relation of the form

$$C = aQ^n \tag{A-1}$$

where C = suspended sediment concentration and Q = flow rate. The parameters in Equation (A-1), a and n, are determined from a log linear regression analysis of the available data. While Equation (A-1) does provide predictive capability of C in a river, and hence sediment load, the rating curve approach has been shown to usually under-predict sediment loads [Walling, 1977; Ferguson, 1987]. A method to correct for rating curve bias has been proposed [Ferguson, 1986] but subsequent evaluation of this correction procedure questioned its effectiveness [Walling and Webb, 1988].

While methods exist for predicting sediment loads in rivers that have available suspended sediment load data, situations are commonly encountered where little or no loading data has been collected for a river of interest. One possible method for estimating sediment loads in cases where little or no data exist is to use gross soil erosion estimates, e.g., tons/km^2-year, for the drainage basin under consideration. The amount of eroded sediment transported into the river is the product of the gross soil erosion and a constant, termed the delivery ratio. Delivery ratios depend upon a number of drainage basin characteristics, including size, topography and land use [Robinson, 1977; Dickinson et al., 1986]. However, this method can produce a high degree of uncertainty in predicted annual loads, especially if the gross soil erosion and delivery ratio are not well known for a particular riverine system.

The above discussion indicates a need for an improved methodology to predict sediment loads in rivers, on seasonal or annual time scales, during periods when very limited or no sediment discharge data are available. An attempt has been made in the current study to develop such a procedure. The next section presents an analysis of existing sediment discharge data from a variety of rivers in the eastern United States that results in the development of a non-dimensional sediment loading function. The predictive capabilities of this non-dimensional formulation are evaluated in the third and fourth sections. A summary of the proposed methodology, highlighting its advantages and limitations, concludes the paper.

A.2 DATA ANALYSIS AND MODEL DEVELOPMENT

The U.S. Geological Survey (USGS) collects sediment discharge data at numerous locations on rivers throughout the United States. The currently available sediment load data base consists of 1552 stations, with daily sediment discharge records ranging in length from 2 days to 45 years at these stations. Generally, daily sediment discharge at a particular station is determined from suspended sediment concentration and flow rate data. However, daily sediment load may be estimated on days when no suspended sediment concentration data are collected. This estimate is based upon flow rate, observed suspended sediment concentrations before and after the period of no data collection, and measured sediment discharges on days with similar flow rates. The analysis presented in this paper does not consider the possible errors or biases in the determination of sediment discharge values.

A typical sediment discharge analysis involves developing a sediment rating curve, e.g., Equation (A-1), for a particular river. If issues concerning the accuracy and precision of sediment rating curves are neglected, this approach could be applied to a large number of rivers using the USGS sediment loading and flow rate data. A major problem with this approach is that identifying general trends in the rating curves of rivers with different characteristics, e.g., drainage area, mean flow rate and mean sediment load, would be very difficult.

An attempt has been made in the present study to overcome this obstacle by normalizing both sediment discharge and flow rate and then examining the relationships between normalized sediment discharge and normalized flow rate for a wide range of rivers. For a given river, the daily average flow rate, Q_d, is normalized with respect to the long term mean flow rate, Q_m, yielding

$$Q_N = \frac{Q_d}{Q_m} \tag{A-2}$$

where Q_N = normalized daily average flow rate. This normalization was chosen because, generally, Q_m can be determined from available data.

A useful normalization of the daily sediment discharge, L_d, is less clear and various methods could be proposed. The quantity used here to normalize L_d is the mean daily sediment discharge under non-flood conditions, L_m, so that

$$L_N = \frac{L_d}{L_m} \tag{A-3}$$

where L_N = normalized sediment discharge.

Non-flood conditions are defined as all flows where the daily average flow rate is less than or equal to twice the mean flow rate, i.e., $Q_N \leq 2$. This criterion was chosen for two reasons. First, examination of rating curves for a number of rivers suggested that a transition in the rating curve generally occurs when $Q_N \cong 2$, i.e., the slope of the log-linear regression line changes. Second, an important goal of this analysis was to develop predictive capabilities of sediment loads in rivers using data that are available or relatively easy to measure. The importance and difficulty of accurately measuring sediment discharge during floods was mentioned earlier in this paper and obtaining this type of data is a significant challenge on any river. However, reliable estimates of L_d, and therefore L_m, under non-flood conditions for a particular river are usually easy to obtain from available data or from a non-flood sediment discharge study.

The normalization procedure was first applied to rivers with the longest record lengths because the USGS sediment discharge data base is quite large and applying the procedure to all rivers prohibitive. Twenty-five rivers were selected with periods of record that ranged from 33 to 45 years. The initial focus of the data analysis was flood flows, i.e., $Q_N > 2$, because of the significant contribution of flood discharge to the annual sediment load of a river. The normalized sediment discharge plots for $Q_N > 2$ were informative. First, graphs of the proposed normalizations were similar over a large range of river sizes, indicating that this type of analysis held promise for producing a predictive model. Second, the impact of load hysteresis during floods was discovered to vary significantly between different geographic regions. Generally, rivers in the western and midwestern United States have dramatically different sediment load trends during the rising and falling limbs of a flood hydrograph. In contrast, the daily sediment load of rivers in the eastern United States tends to exhibit minor hysteresis effects during floods.

The normalization analysis was then extended to non-flood flows and, similar to the flood event regime, discernible geographic differences were observed. An example of regional variability, in both flow regimes, is illustrated by three rivers from different areas (Figure A-1): Animas River, New Mexico; Iowa River, Iowa; and Roanoke River, Virginia. The sediment load data were binned into groups of equal size, e.g., same fraction of the total population, and the log mean $\pm$ two standard deviations of each data group then plotted on Figure A-1. Normalized sediment loads in the Roanoke River have lower variability than the rivers in New Mexico and Iowa, which was found to be typical of rivers in the eastern United States when compared to rivers in the midwestern or western regions. The log means of L_N during flood flows, i.e., $Q_N \geq 2$, are also higher for the Roanoke River, by about a factor of five, than the Animas and Iowa Rivers.

One of the motivating factors behind the development of the methodology presented in this paper was the authors' involvement in a contaminant fate and transport study on the Pawtuxet River in Rhode Island. The sediment transport model developed for the Pawtuxet River [Ziegler and Nisbet, 1994] was calibrated over a 789 day period, beginning in March 1992 and extending to May 1994. Sediment loading data were only

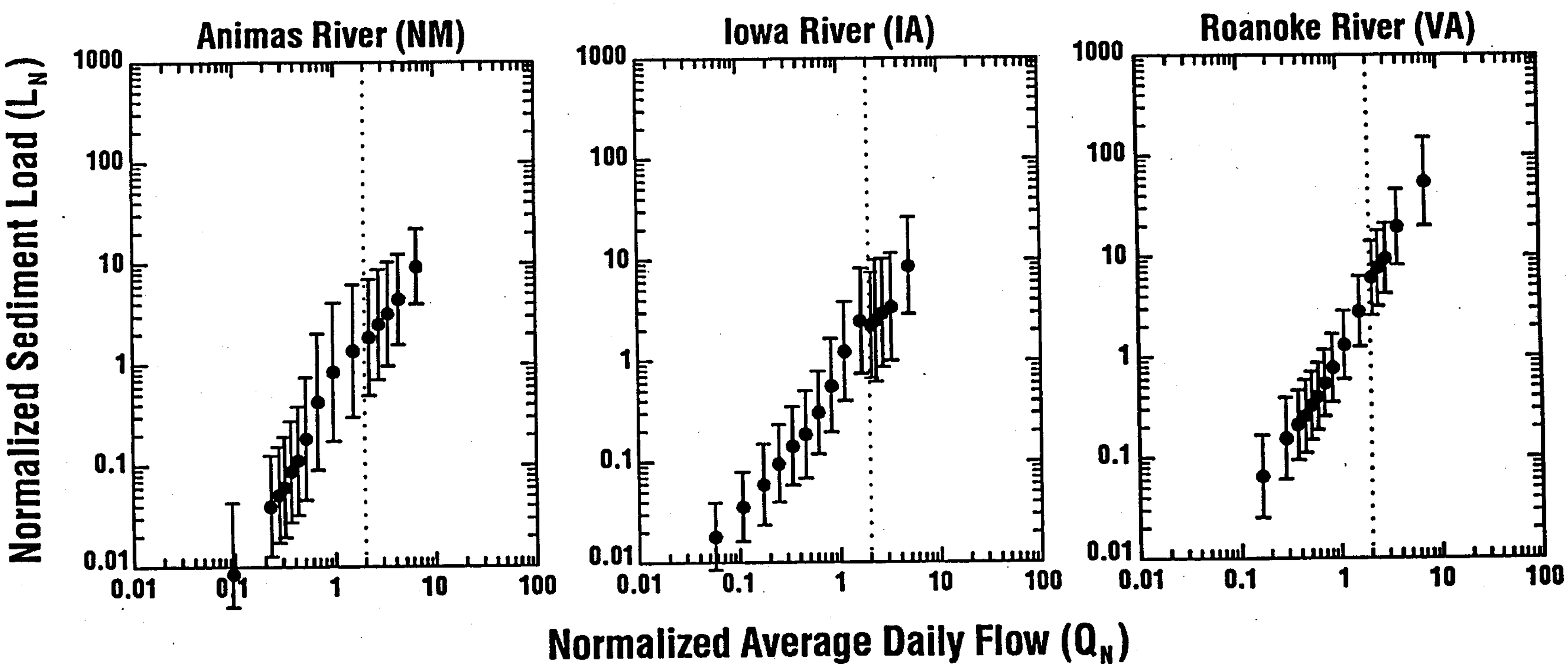


Figure A-1. Normalized sediment load plots for rivers in New Mexico, Iowa and Virginia. Log mean values, ± two standard deviations, of binned data are shown.

available for approximately 90 days during this period and an adequate sediment rating curve could not be developed from the data. A great need existed for estimating daily sediment loads for the 700 day period for which no data existed. The procedures described in this paper evolved from this lack of necessary sediment loading information for the Pawtuxet River sediment transport model. However, differences in the normalized sediment discharge plots of rivers from various geographic regions, particularly during floods, precluded development of a generic model for the entire United States. The focus of this study is thus limited to a region of the eastern United States, including the Pawtuxet River, that extends along the Atlantic seaboard from North Carolina to New England and westward to Ohio, see Figure A-2.

Twenty-nine rivers were selected from the region under consideration. The characteristics of these rivers are listed in Table A-1 and their locations are shown on Figure A-2. As can be seen in Table A-1, these rivers encompass a wide range of characteristics: drainage area, A, ranging from 2.4 to 62,400 km^2; mean flow rates from 0.048 to 980 m^3/s; and L_m ranging from 0.082 to 2440 tons/day. The mean non-flood sediment load, L_m, of each river was determined by averaging L_d on all days in the record for which $Q_N \leq 2$.

Normalized sediment discharge plots are presented on Figure A-3 for four rivers that span the range of river sizes included in the present analysis, from a small stream with A < 3 km^2 to a large river with A ~ 13,000 km^2. Normalized sediment load data are presented as log means with $\pm$ two standard deviations; L_N data were binned using 5% increments of the population along the Q_N axis. The solid lines on this figure are the result of separate log linear regressions of the low flow and high flow data; the log linear regressions were performed on all of the data, not on the log means of the binned data.

Several observations can be made about the plots on Figure A-3. First, a break or transition in the data is evident near $Q_N = 2$, which lends support to the choice of this normalized flow rate as a criterion for defining the non-flood regime. Second, the normalized sediment load plots are similar from one river to the next. Finally, the log

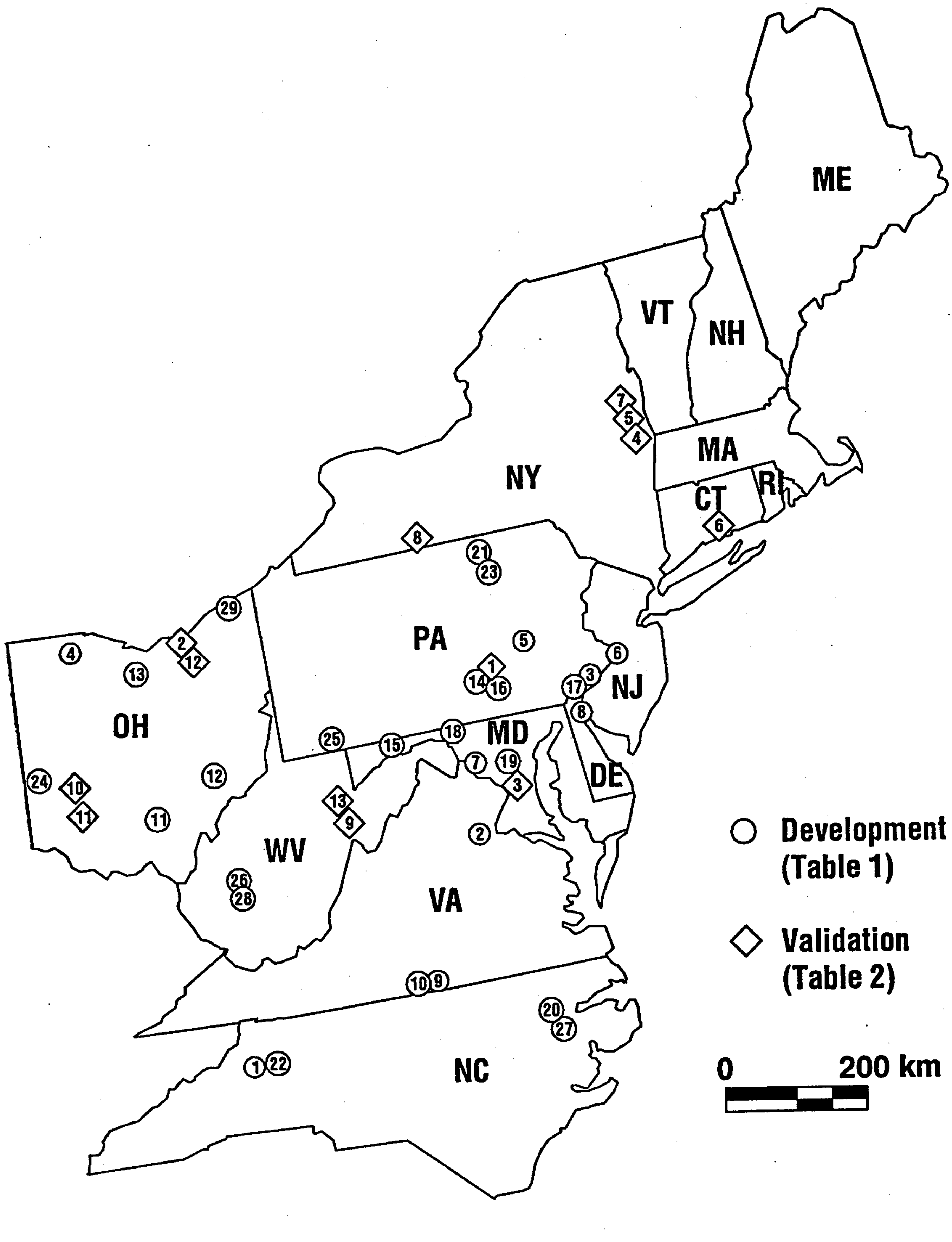


Figure A-2. Region in eastern United States considered in present analysis. Reference numbers for rivers are listed in Tables A-1 and A-2.

TABLE A-1. DRAINAGE BASIN CHARACTERISTICS OF RIVERS USED IN MODEL DEVELOPMENT				
Station Location (Figure A-2 reference number)	Record Length (years)	A (km^2)	Q_m (m^3/s)	Data-Based L_m (tons/day)
Yadkin R. at Yadkin College, NC (1)	42.8	5910	87	892
Rappahannock R. at Remingon, VA (2)	42.5	1610	19	35
Schuylkill R. at Manayunk, PA (3)	38.9	4740	78	124
Maumee R. at Waterville, OH (4)	43.5	16,400	146	419
Schuylkill R. at Berne, PA (5)	34.0	920	20	63
Delaware R. at Trenton, NJ (6)	32.6	6150	327	397
Potomac R. at Point of Rocks, MD (7)	33.0	25,000	276	2440
Brandywine C. at Wilmington, DE (8)	33.9	810	13	21
Roanoke R. at Randolph, VA (9)	27.8	7710	83	335
Dan R. at Paces, VA (10)	27.3	6610	79	563
Scioto r. at Higby, OH (11)	29.0	13,300	133	684
Muskingum R. at Dresden, OH (12)	22.0	15,530	165	592
Sandusky R. near Fremont, OH (13)	43.0	3240	29	89
Bixier Run near Loysville, PA (14)	17.4	39	0.42	0.34
NB Potomac R. near Cumberland, MD (15)	18.0	2270	38	69
Susquehanna R. at Harrisburg, PA (16)	19.1	62,400	980	1513
Brandywine Cr. at Chadds Ford, PA (17)	15.0	740	12	19
Conococheague Cr. at Fairview, MD (18)	14.0	1280	21	40
NWB Anacostia R. near Colesville, MD (19)	13.0	55	0.65	3.6
Tar R. at Tarboro, NC (20)	10.0	5660	62	137
Elk Run near Mainesburg, PA (21)	13.0	26	0.29	0.082
Third C. near Stony Point, NC (22)	12.3	13	0.18	0.36
Corey C. near Mainesburg, PA (23)	13.4	32	0.31	0.12
Stillwater R. at Pleasant Hill, OH (24)	12.0	1300	12	21
Stony Fork Trib. near Gibbon Glade, PA (25)	12.0	2.4	0.048	0.18
Coal R. at Tornado, WV (26)	11.8	2230	35	123
Chicod C. near Simpson, NC (27)	11.5	117	1.4	1.7
Little Coal R. at Danville, WV (28)	11.1	700	12	64
Grand R. near Painesville, OH (29)	11.0	1780	30	328

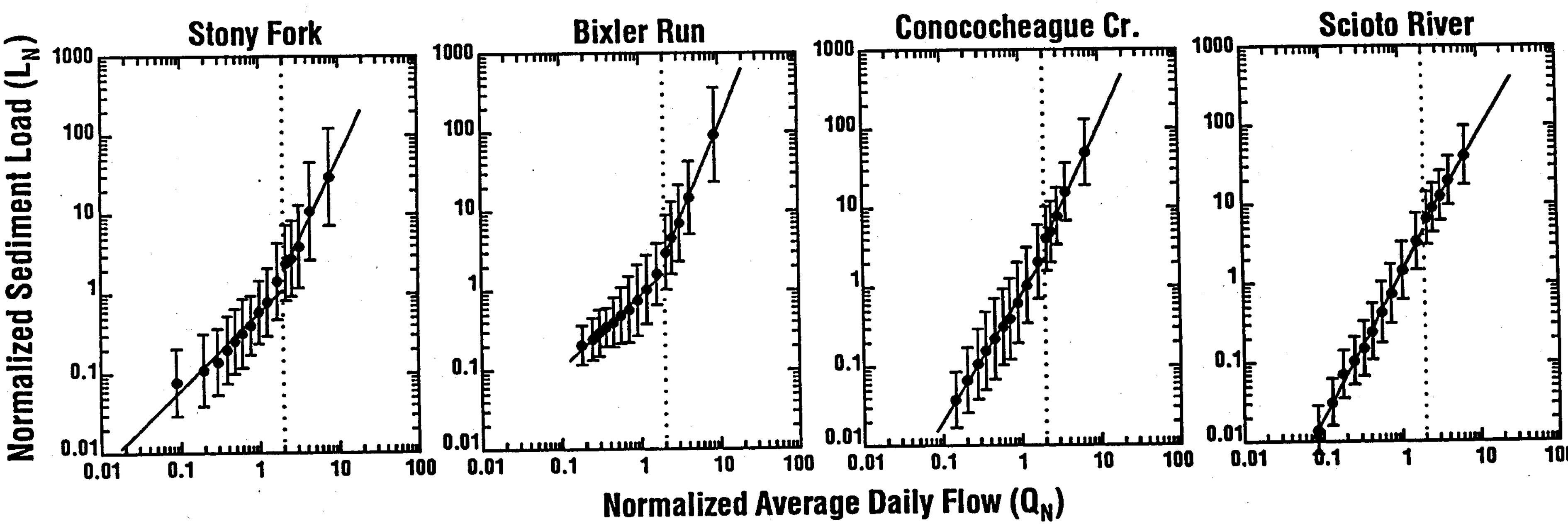


Figure A-3. Normalized sediment load plots for several rivers used in model development. Log mean values, $\pm$ two standard deviations, of binned data are shown. Solid lines indicate results of log linear regression analysis using the normalized data.

standard deviation of L_N is approximately constant with respect to Q_N, as can be seen on Figure A-3 from the relatively constant width of the standard deviation bars.

Sediment discharge hysteresis during floods was also examined for these twenty-nine rivers. Somewhat surprisingly, the difference between normalized sediment load curves developed for flows on the rising and falling limbs of flood hydrographs was minor (results not shown). Thus, stratifying flood flow regimes will not significantly improve daily sediment load predictions for rivers in the eastern United States.

These trends in the normalized sediment discharge plots indicate the possibility of developing a generalized function relating L_N to Q_N which would be applicable to rivers over a wide range of drainage basin sizes and mean flow rates. Such a generalized function has been developed and it is similar to a conventional sediment rating curve, i.e., Equation (A-1), except for three important differences. First, non-dimensionalizing the loading function produces a generalized expression that is applicable to many rivers. Second, variations in the sediment discharge characteristics among riverine systems are accounted for by making the parameters a and n in Equation (A-1) functions of river characteristics. Third, a stochastic component has been added to the non-dimensional version of Equation (A-1) to account for observed variability in sediment loads. Equation (A-1) is completely deterministic, meaning that a single suspended sediment concentration, or sediment discharge, corresponding to the median or geometric mean of the distribution, will be predicted at a specific flow rate. The deterministic method is not completely realistic because the sediment load at a particular flow rate can be highly variable. An important benefit of including the stochastic component, in addition to generating more realistic daily sediment loads, is that it improves the predictive capabilities of the method, as will be demonstrated later in this paper.

The modified form of Equation (A-1), expressed in log linear form, is

$$\log L_N = \log a + n \log Q_N + \delta S_L \qquad \text{(A-4)}$$

where log a and n are functions of drainage basin characteristics, S_L = standard deviation of the log estimate, and δ = normally distributed random number with mean of zero and standard deviation of one. Hereafter, Equation (A-4) will be referred to as the Normalized Sediment Load (NSL) function.

The motivation for including a stochastic component in the NSL function, i.e., δS_L, is to account for natural variability in L_N at a particular Q_N and to more accurately predict L_N. The tendency of sediment rating curves developed from log linear regression analysis to underestimate sediment loads in rivers has been recognized [Walling, 1977; Ferguson, 1986; Walling and Webb, 1988]. The typical development of a sediment rating curve ignores the variability not captured by log-linear regression. The resulting equation, e.g., Equation (A-1), predicts the median solids loading at any flow. The sediment rating curve under-predicts the mean load because the data are log-normally distributed. This bias is eliminated by including the estimate of residual variance in the log-linear form of the equation. The random nature of the NSL function will not make it possible to accurately predict sediment loads on short time scales, e.g., hourly or daily, however, it will increase the accuracy of predicted sediment loads over seasonal or annual time scales.

Parameter values in the NSL function, i.e., log a, n and S_L, were determined in the following manner. First, log linear regression was used to determine the best fit line for L_N as a function of Q_N for each of the twenty-nine rivers. To account for observed differences in the variation of L_N under non-flood and flood conditions, the flow regime was stratified prior to regression analysis, with $Q_N = 2$ being chosen as the break point between non-flood and flood flows. Thus, two best fit lines, one for $Q_N \leq 2$ and another for $Q_N > 2$, were determined for each river. The results of the regression analyses yielded values of log a, n and S_L, for $Q_N \leq 2$ and $Q_N > 2$, for each river.

Attempts were then made to develop generalized expressions for log a, n and S_L that were applicable over a wide range of river sizes. Correlations between the three parameters, log a, n and S_L, and five drainage basin characteristics, A, Q_m, L_m, Q_m/A and L_m/A, were examined using results from the regression analyses of the twenty-nine rivers. The analyses indicated that log a, n and S_L were not significantly correlated with many of the five drainage basin characteristics. This result suggested that mean values of log a, n and S_L, for the two flow regimes of $Q_N \leq 2$ and $Q_N > 2$, could be used when applying the NSL function. However, statistically significant correlation, even though it was relatively low, did exist between the NSL function parameters and either Q_m/A or A. Preliminary tests of the NSL function indicated that accounting for parameter variability with respect to Q_m/A or A, as opposed to using mean values, did improve the accuracy of Equation (A-4). The following relationships, stratified for $Q_N \leq 2$ and $Q_N > 2$, were determined from the correlation analyses

$$\log a = \begin{cases} 0.478 - 40.6 \dfrac{Q_m}{A} & , \quad Q_N \leq 2 \\ 0.714 - 54.5 \dfrac{Q_m}{A} & , \quad Q_N > 2 \end{cases} \tag{A-5}$$

$$n = \begin{cases} 0.794 + 0.205 \log A & , \quad Q_N \leq 2 \\ 1.18 + 69.3 \dfrac{Q_m}{A} & , \quad Q_N > 2 \end{cases} \tag{A-6}$$

$$S_L = \begin{cases} 0.40 & , \quad Q_N \leq 2 \\ 0.546 - 0.0572 \log A & , \quad Q_N > 2 \end{cases} \tag{A-7}$$

where A and Q_m have units of km^2 and m^3/s, respectively. The correlation plots corresponding to Equations (A-5) through (A-7) are presented on Figure A-4.

The predictive capability of the NSL function, utilizing Equations (A-5) through (A-7) to determine log a, n and S_L, was initially tested by applying these equations to the twenty-nine rivers used in the model development process. This check, while not a

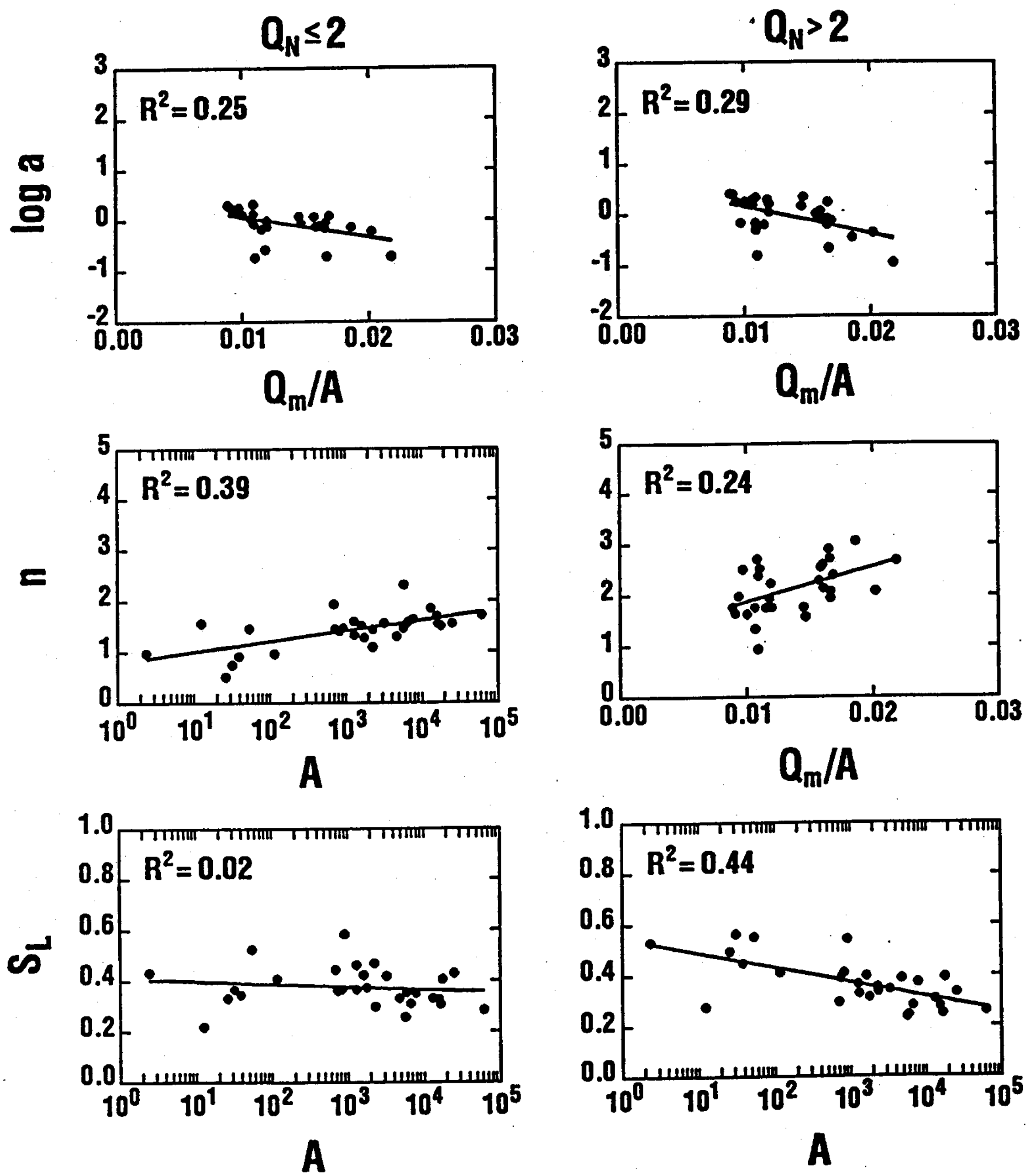


Figure A-4. Results of correlation analyses for determining log a, n and S_L values in Equation (A-4). Units of Q_m/A and A are $m^3/s\text{-}km^2$ and km^2, respectively.

validation of the NSL function approach to estimating sediment loads in rivers, was conducted to evaluate the accuracy of the proposed methodology with the calibration data set. Annual solids load was used as the basis of comparison. Normalized daily sediment loads, L_N, were predicted for the entire period of record for each of the twenty-nine rivers. Equation (A-3) was then used to calculate the daily sediment loads, i.e., $L_d = L_N L_m$, where L_m was determined for each river from the available data, see Table A-1. The predicted daily loads for each river were then summed on an annual basis. The resulting predicted annual sediment loads, for a total of 618 years, were next compared to measured annual loads, see Figure A-5a. These results are encouraging; the model demonstrates predictive capabilities for rivers with annual sediment loads ranging over five orders of magnitude. An error analysis was also conducted to quantify the accuracy of the NSL function. The relative error, i.e., (predicted - measured)/measured, for each of the 618 predicted annual loads was determined and the distribution of the errors is presented on Figure A-5b. The mean and median errors were 36% and -14%, respectively; 64% of the predicted annual loads were within a factor of two of the observed value.

A.3 VALIDATION OF NSL FUNCTION PREDICTIVE CAPABILITIES

The NSL function was expected to predict annual sediment loads for the rivers used to develop the model with a reasonable degree of accuracy and the above results indicate that this is the case. However, a necessary test of the model is its application to rivers not included in the calibration data set. Validation of the model was accomplished by predicting annual sediment loads for thirteen rivers from the same geographic region as the original twenty-nine rivers used to develop the model, see Figure A-2. These thirteen rivers span a wide range of drainage basin characteristics, see Table A-2, from a small stream with $A = 13$ km^2 and $Q_m = 0.43$ m^3/s to a large river with $A = 11{,}970$ km^2 and $Q_m = 231$ m^3/s.

Values of NSL function parameters, i.e., log a, n and S_L, for each the thirteen rivers were determined using river drainage basin characteristics, i.e., Q_m/A and A, in Equations (A-5) through (A-7). Use of the NSL function to predict sediment loads in these thirteen

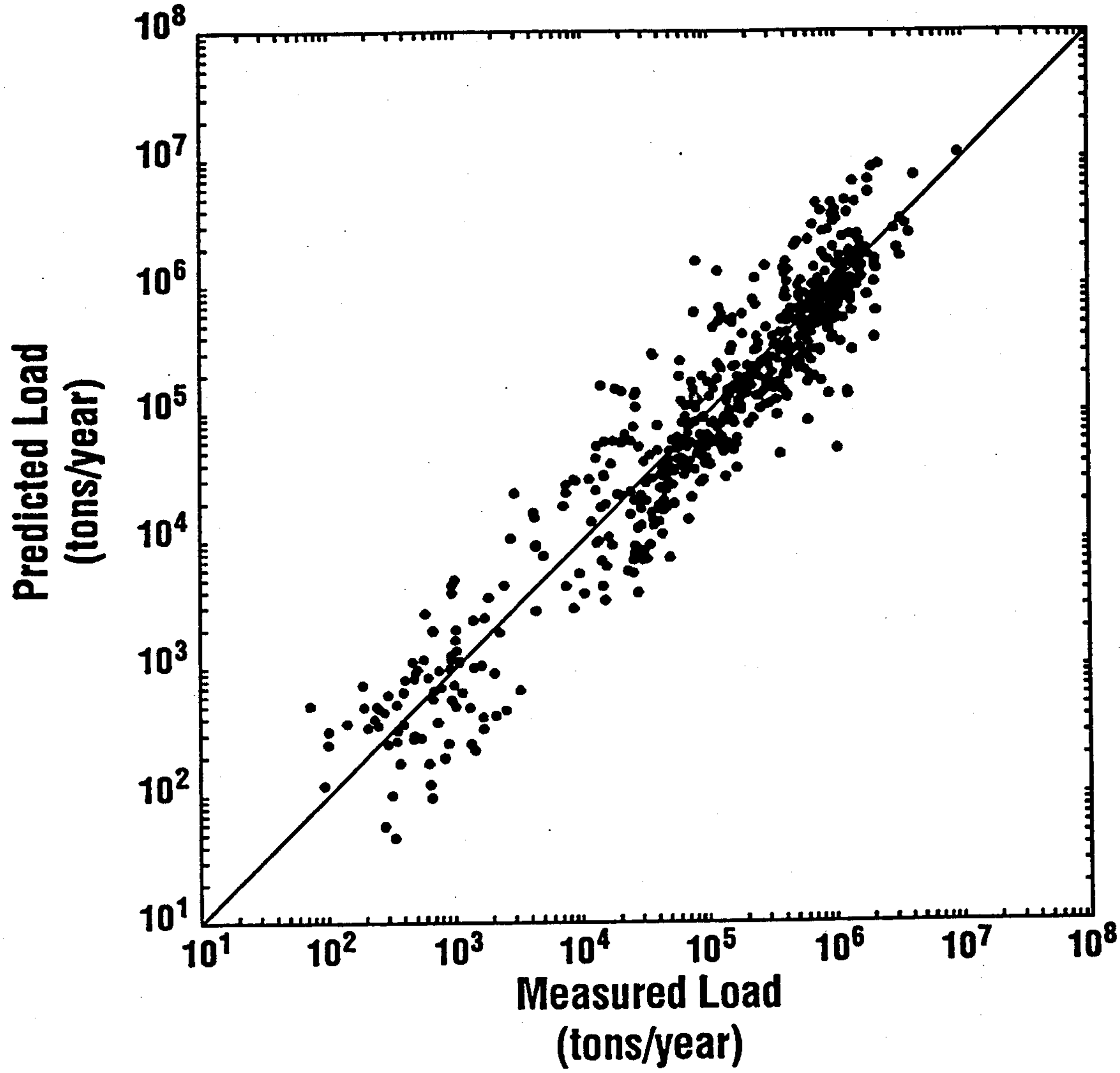


Figure A-5a. Results of NSL function application to 29 rivers used in model development: comparison of predicted and measured annual sediment loads.

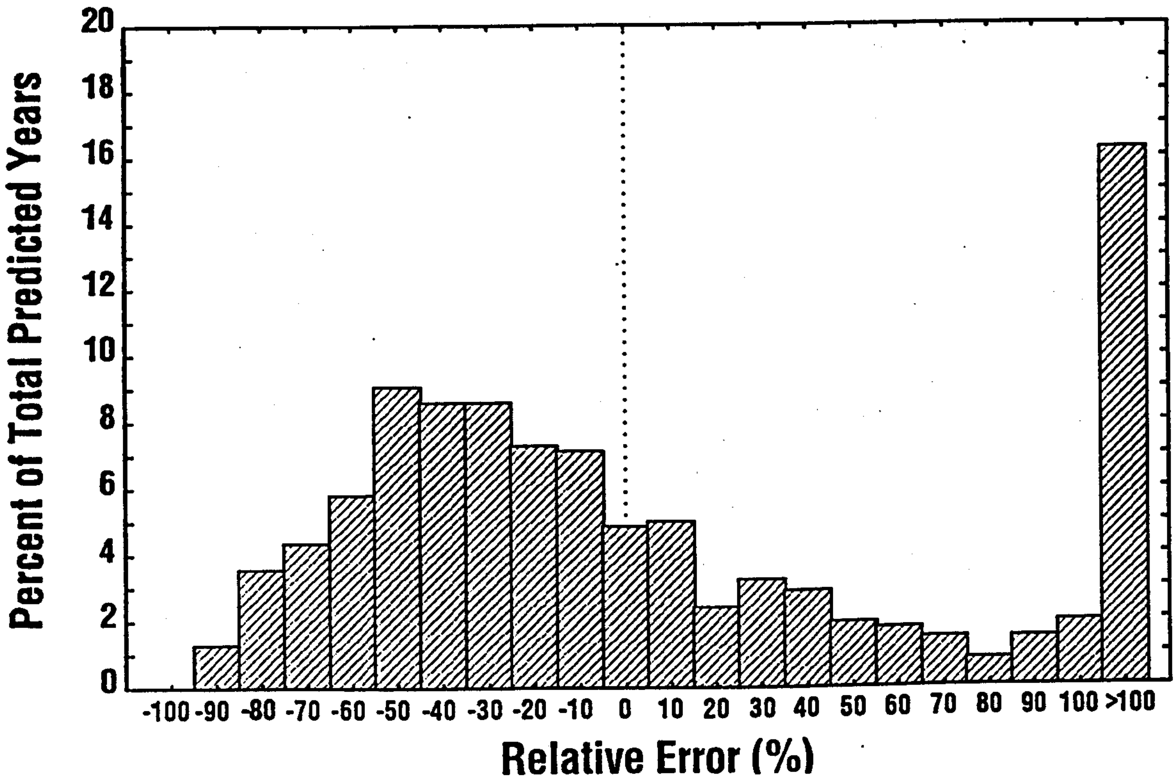


Figure A-5b. Results of NSL function application to 29 rivers used in model development: frequency distribution of relative errors.

TABLE A-2. DRAINAGE BASIN CHARACTERISTICS OF RIVERS USED IN MODEL VALIDATION					
Station Location (Figure A-2 reference number)	Record Length (years)	A (km^2)	Q_m (m^3/s)	Data-Based L_m (tons/day)	Estimated L_m (tons/day)
Juniata R. at Newport, PA (1)	40.0	8690	122	130	360
Cuyahoga R. at Independence, OH (2)	34.0	1830	23	135	63
NB Rock Cr. at Rockville, MD (3)	10.1	32	0.45	0.61	0.68
Mohawk R. at Cohoes, NY (4)	25.7	8940	165	168	360
Hudson R. at Waterford, NY (5)	8.0	11,970	231	191	520
Coginchaug R. at Middlefield, CT (6)	7.6	77	140	0.47	3.5
Hudson R. at Stillwater, NY (7)	8.5	9780	186	119	410
Tioga R. at Lindley, NY (8)	7.0	2000	23	48	70
Shavers Fork below Bowden, WV (9)	6.3	390	12	7.9	11
L. Miami R. near Oldtown, OH (10)	6.2	330	2.0	3.1	9.3
Todd F. near Roachester, OH (11)	6.1	570	6.0	23	17
Tinkers Cr. at Bedford, OH (12)	7.6	220	3.7	16	5.9
Taylor Run at Bowden, WV (13)	6.0	13	0.43	0.24	0.25

rivers also required determining L_m for each of the rivers. The mean daily sediment load under non-flood conditions, L_m, of a particular river was calculated using data from all days during which $Q_d \leq 2\ Q_m$, i.e., $Q_N \leq 2$. The resulting values of L_m are presented in Table A-2 for each of the thirteen rivers in the model validation.

A total of 149 annual sediment loads were predicted in the model validation. The comparison of predicted and observed annual sediment loads, that ranged over four orders of magnitude, demonstrates that the NSL function does yield predictions, on an annual time scale, that are relatively accurate, see Figure A-6a. The model is able to properly account for variations in drainage basin characteristics, e.g., A, Q_m and L_m, indicating that the normalizations used in the NSL function, L_N and Q_N, are physically relevant. A quantitative error analysis, where the relative error was calculated for each of the 149 predicted annual loads, yielded a mean error of 3% and a median error of -18%, see Figure A-6b, with 82% of the predicted annual loads being within a factor of two of the observed value.

To demonstrate the importance of the stochastic component in the NSL function, the validation calculations were repeated with the stochastic component in Equation (A-4) set to zero, i.e., $\delta S_L = 0$. As expected, the non-stochastic calculations under-predict the annual loads (compare Figure A-7a to Figure A-6a). The non-stochastic error distribution (Figure A-7b) has significantly more negative errors (under-predictions) than the error distribution resulting from application of the complete NSL function (Figure A-6b). The mean and median relative errors were -31% and -39%, respectively, for the non-stochastic predictions, and 64% of the predicted annual loads were within a factor of two of the measured annual load.

A.4 APPLICATION OF NSL FUNCTION WHEN L_m IS UNKNOWN

The previous applications of the NSL function assumed that L_m was known for each of the rivers; L_m was determined from available data in the above calculations. Frequently, sediment loading must be determined for a river that has very limited or no sediment

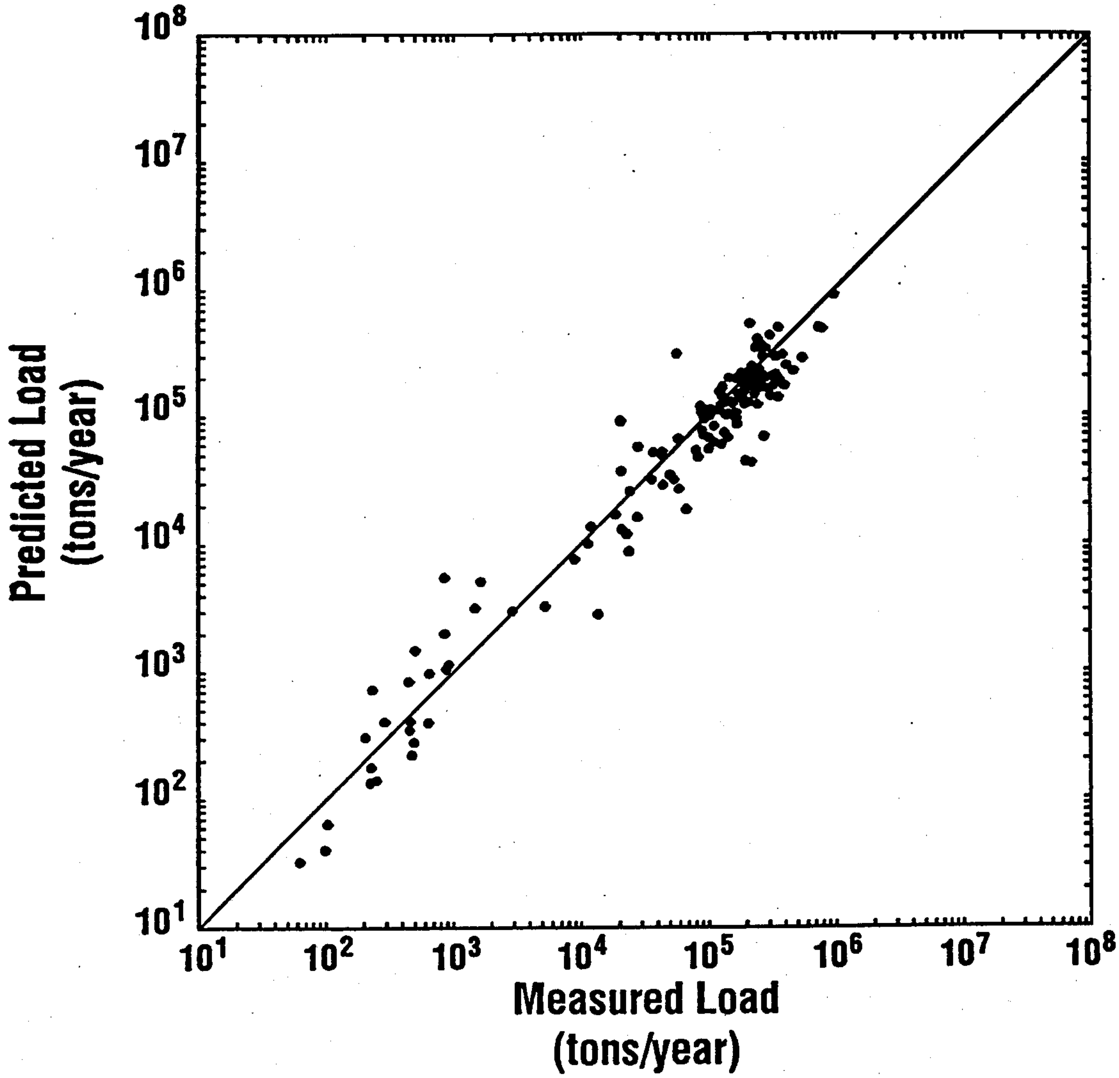


Figure A-6a. Results of NSL function application to 13 rivers used in model validation: comparison of predicted and measured annual sediment loads.

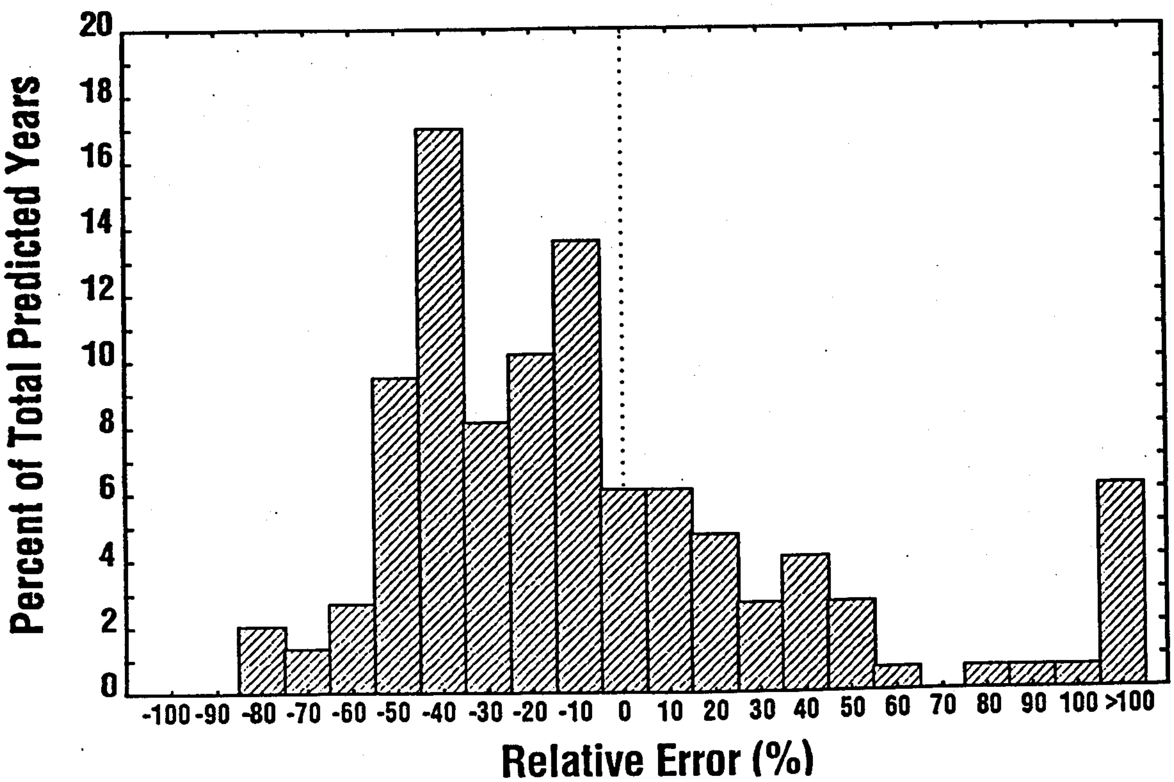


Figure A-6b. Results of NSL function application to 13 rivers used in model validation: frequency distribution of relative errors.

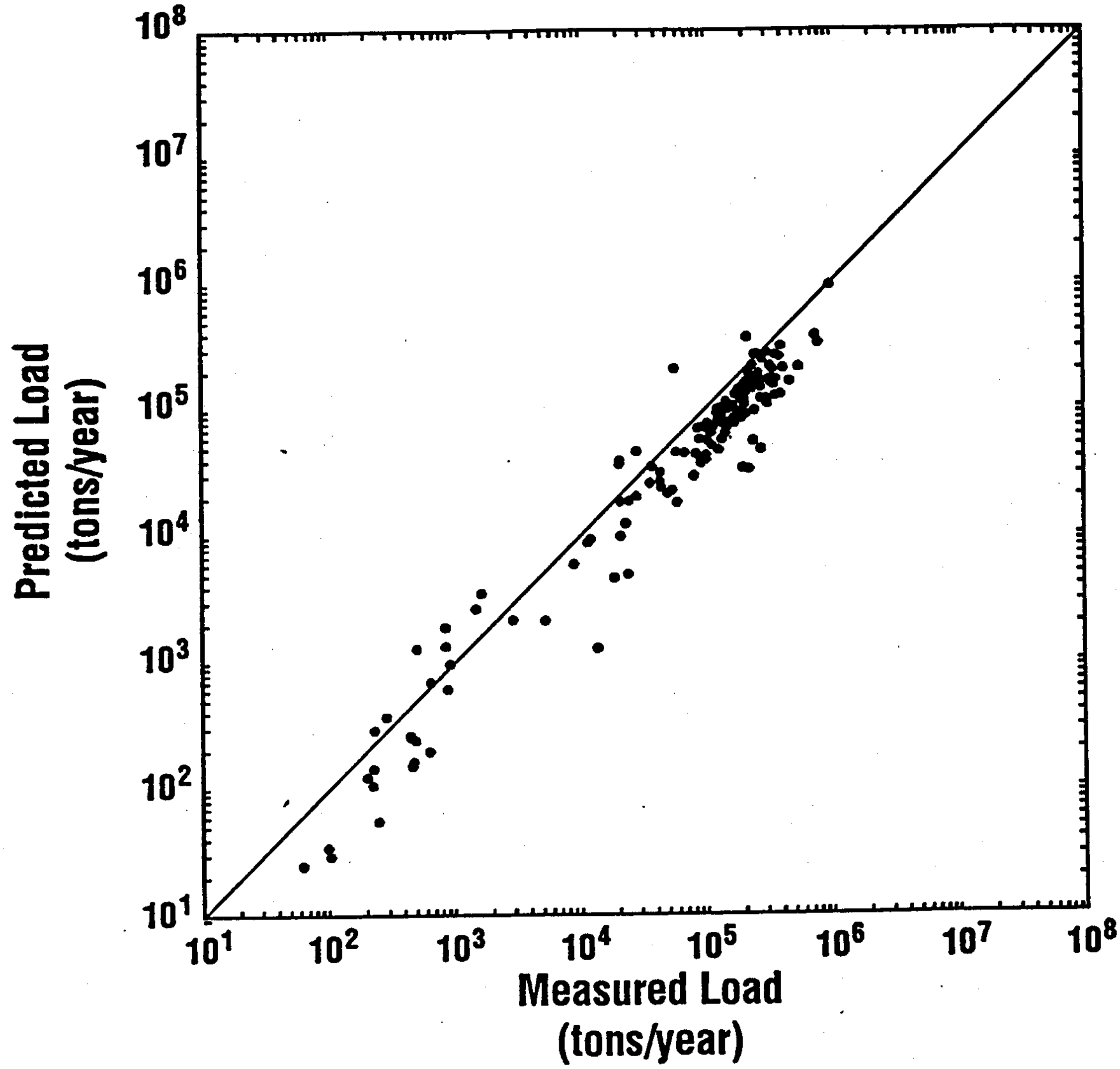


Figure A-7a. Results of NSL function application to 13 rivers used in model validation with stochastic component (δS_L) set to zero: comparison of predicted and measured annual sediment loads.

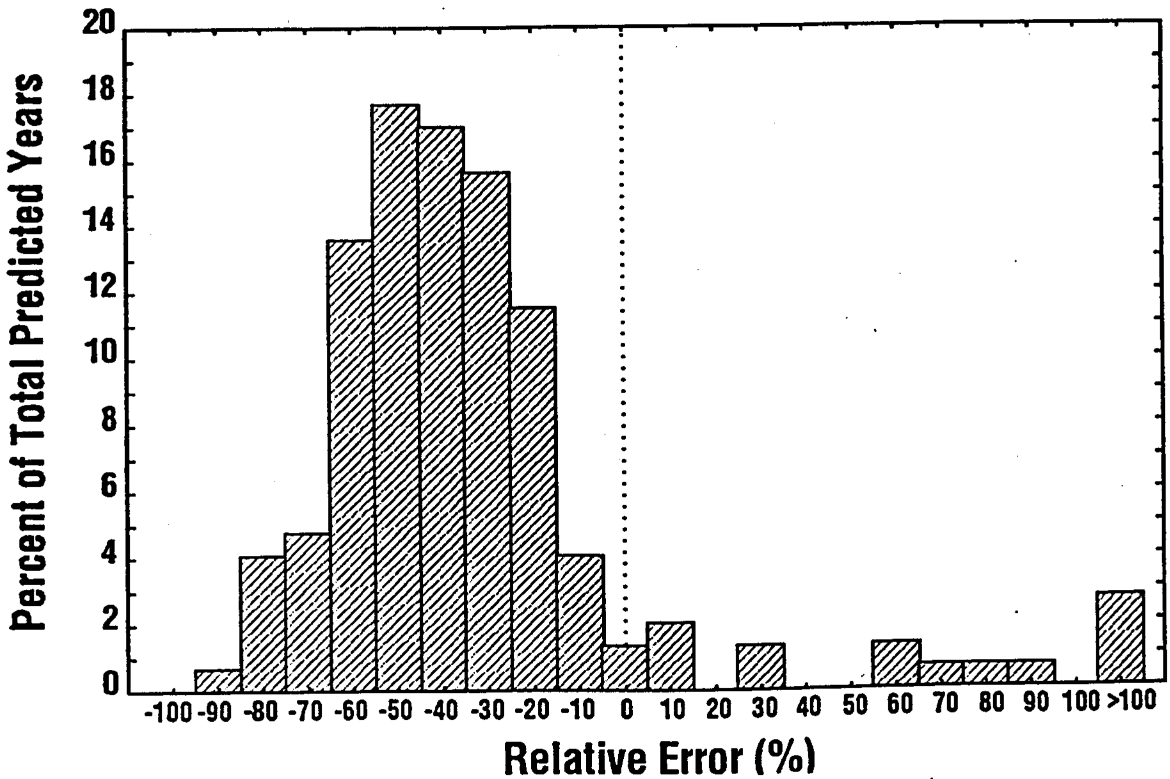


Figure A-7b. Results of NSL function application to 13 rivers used in model validation with stochastic component (δS_L) set to zero: frequency distribution of relative errors.

discharge data, making it extremely difficult to calculate L_m for that specific river. Without an estimate of L_m, the NSL function cannot be used to calculate sediment loads.

An approximate method for applying the NSL function to situations when L_m cannot be determined from data has been developed to overcome this problem. A correlation between L_m (tons/day) and drainage area, A (km^2), was found for the twenty-nine rivers used in the model development, see Figure A-8. Linear regression, in log space, of the data resulted in

$$L_m = 0.014 A^{1.12} \tag{A-8}$$

with 92% of the variation of L_m explained by A, i.e., $R^2 = 0.92$.

The validation calculations were repeated using Equation (A-8) to estimate L_m for each of the thirteen rivers prior to application of the NSL function. The estimated L_m values, listed in Table A-2, are generally much different than the data-based values. The estimates of L_m are all within a factor of eight of the actual value, with five of the thirteen rivers having estimated values within a factor of two of the data-based value. Model predictions based on estimated L_m values were not as good as when the data-based L_m values were used, see Figure A-9a. However, the predicted annual loads, based on L_m estimated using Equation (A-8), were not grossly inaccurate. The relative errors were more widely distributed, see Figure A-9b, with a mean of 74% and a median of 40%. The portion of the predicted annual loads that was within a factor of two of the actual load decreased to 51%.

A.5 CONCLUDING REMARKS

Analysis of sediment discharge data from rivers in the eastern United States indicated that a similarity relationship exists for a large size range of riverine systems when the daily sediment load, L_d, and daily mean flow rate, Q_d, are properly normalized. The quantities chosen to normalize L_d and Q_d were the mean daily sediment load under non-flood conditions, L_m, and the long-term mean flow rate, Q_m, respectively. This choice of

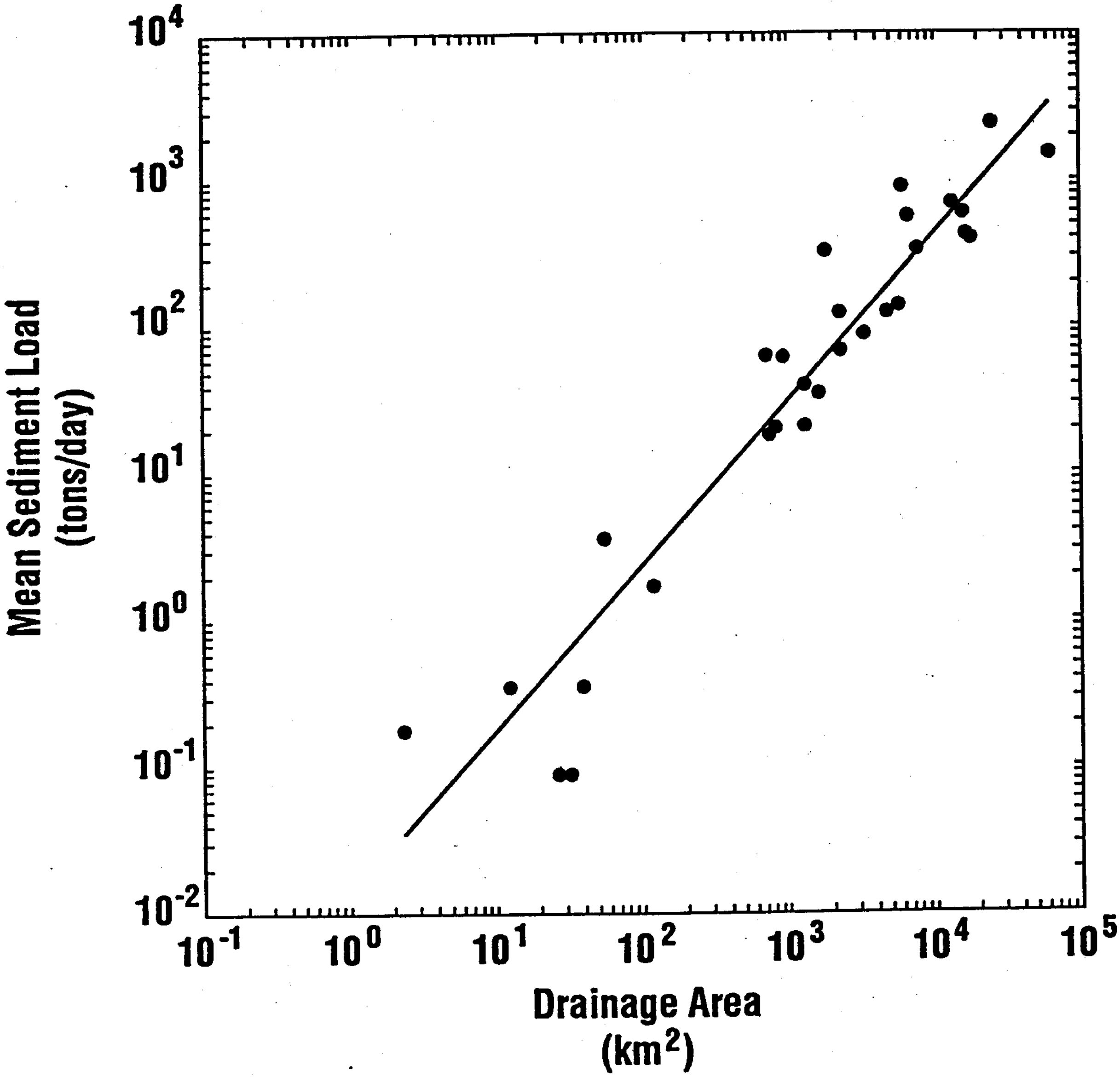


Figure A-8. Correlation between mean daily sediment load under non-flood conditions (L_m) and drainage area.

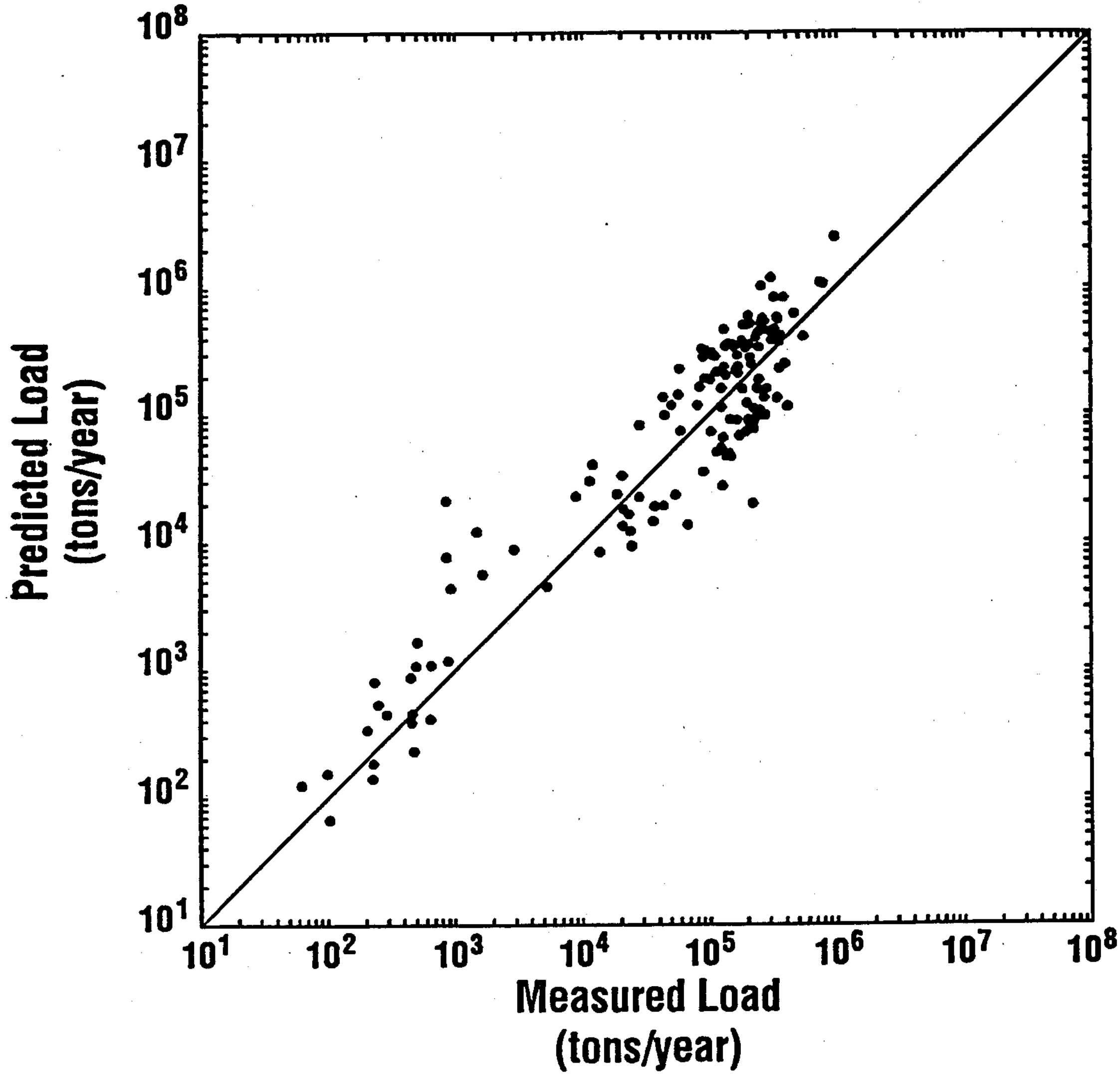


Figure 9a. Results of NSL function application to 13 rivers used in model validation with L_m predicted by Equation (A-8): comparison of predicted and measured annual sediment loads.

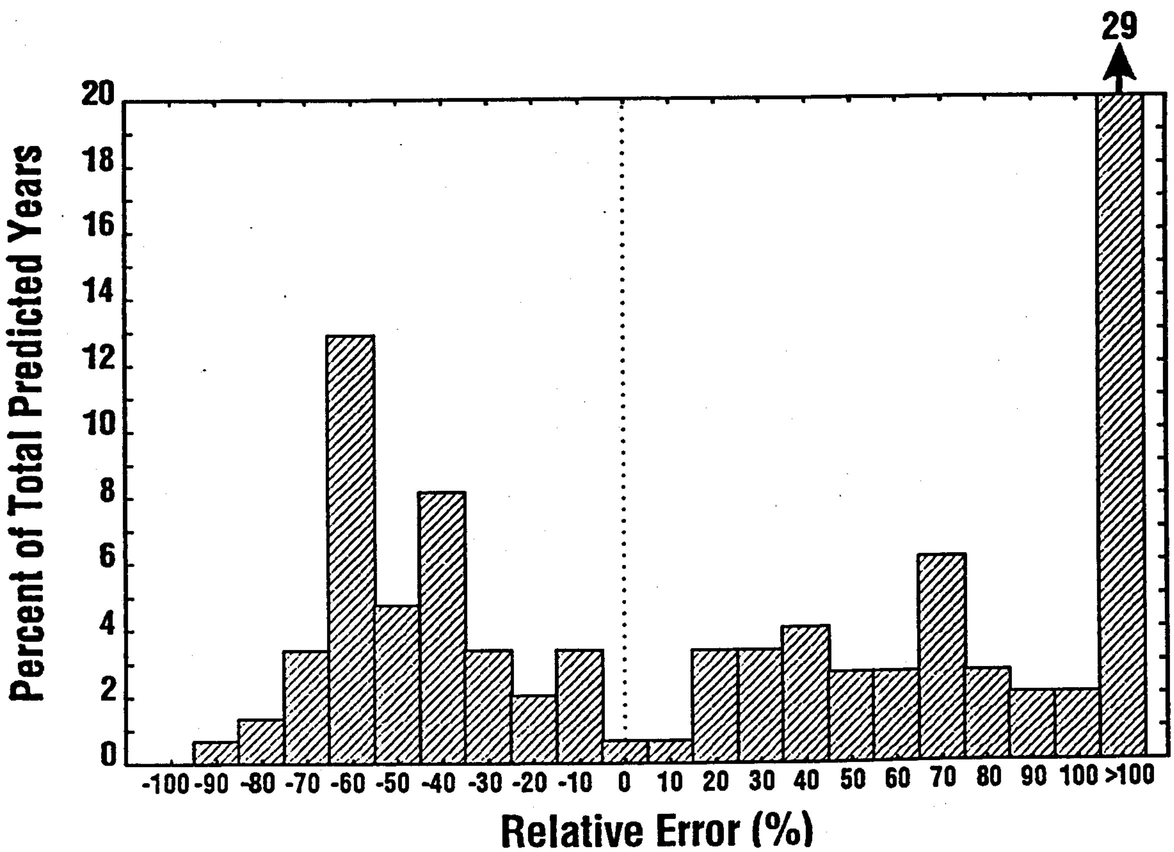


Figure 9b. Results of NSL function application to 13 rivers used in model validation with L_m predicted by Equation (A-8): frequency distribution of relative errors.

normalization, which is not unique, was chosen because L_m and Q_m can generally be determined for most riverine systems without much difficulty, either from existing data or from a relatively inexpensive field program.

This data analysis resulted in the development of a non-dimensional formulation, the NSL function, that is capable of predicting annual sediment loads in rivers located in the eastern United States with a reasonable degree of accuracy. The NSL function, as defined by Equations (A-4) through (A-7), is applicable to riverine systems, in the geographic region indicated on Figure A-2, that range over four orders of magnitude in size, with drainage areas of less than 3 km^2 to over 25,000 km^2. The proposed formulation, Equation (A-4), also includes a stochastic component that improves predictive capabilities and produces realistic variability in estimated daily sediment loads. As noted earlier, the NSL function depends upon knowledge of L_m, which may not be available for particular studies. An approximate method for estimating L_m, based upon drainage basin size, was presented that yields annual load predictions that have a higher degree of uncertainty but are still useful in situations when no sediment loading data are available for a particular river.

The NSL function, along with the parameters defined in Equations (A-5) through (A-7), has been shown to be a credible tool for predicting annual sediment loads in rivers. However, the limitations of this methodology must be acknowledged. First, the NSL function has only been shown to simulate sediment loads reasonably well on annual time scales. At the present time, this model may not be able to accurately predict riverine sediment discharge on short time scales, e.g., daily loads. Second, the NSL function parameters, Equations (A-5) through (A-7), were developed using data from rivers in the geographic region illustrated on Figure A-2. This model should not be applied to other regions because significant geographic differences in sediment discharge characteristics will require modification of the equations for log a, n and S_L. Continued work with the existing data base will hopefully result in the extension of the NSL function to other regions of the United States in the near future.

A.6 REFERENCES

Dickinson, W.T., Rudra, R.P. and Clark, D.J., 1986. A Delivery Ratio Approach for Seasonal Transport of Sediment, in: Drainage Basin Sediment Delivery, IAHS Pub. No. 159, pp. 237-252.

Ferguson, R.I., 1986. River Loads Underestimated by Rating Curves, Water Resour. Res., 22(1):74-76.

Ferguson, R.I., 1987. Accuracy and Precision of Methods for Estimating River Loads, Earth Surf. Proc. and Land., 12:95-104.

Parker, R.S. and Troutman, B.M., 1989. Frequency Distribution of Suspended Sediment Loads, Water Resour. Res., 25(7):1567-1574.

Robinson, A.R., 1977. Relationship Between Soil Erosion and Sediment Delivery, in: Erosion and Solid Matter Transport in Inland Waters, IAHS Pub. No. 122, pp. 159-167.

Thomas, R.B., 1985. Estimating Total Suspended Yield with Probability Sampling, Water Resour. Res., 21(9):1381-1388.

Walling, D.E., 1977. Assessing the Accuracy of Suspended Sediment Rating Curves for a Small Basin, Water Resour. Res., 13(3):531-538.

Walling, D.E. and Webb, B.W., 1981. The Reliability of Suspended Sediment Load Data, in: Erosion and Sediment Transport Measurement, IAHS Pub. No. 133, pp. 177-194.

Walling, D.E. and Webb, B.W., 1988. The Reliability of Rating Curve Estimates of Suspended Sediment Yield: Some Further Considerations, in: Sediment Budgets, IAHS Pub. No. 174, pp. 337-350.

Walling, D.E., Webb, B.W. and Woodward, J.C., 1992. Some Sampling Considerations in the Design of Effective Strategies for Monitoring Sediment-Associated Transport, in: Erosion and Sediment Transport Monitoring Programmes in River Basins, IAHS Pub. No. 210, pp. 183-190.

Ziegler, C.K. and Nisbet, B., 1994. Fine-Grained Sediment Transport in Pawtuxet River, Rhode Island, ASCE J. Hyd. Engr., 120(5):561-576.

APPENDIX E

ALPHA BOUNDARY

Management Committee
Long Island Sound Estuary Study
and New England Interstate
Water Pollution Control Commission

EVALUATION OF NUTRIENT MANAGEMENT SCENARIOS USING LISS 3.0

May 5, 1997

Project No: NENG0035

CONTENTS

FIGURES

FIGURES (Cont.)

Note: All figures exist in original full color form in the original job report file, stored with the library.

TABLES

Table Page

SECTION 1

INTRODUCTION

1.1 BACKGROUND

A coupled hydrodynamic/water quality model has been constructed for Long Island Sound. The details of the underlying mathematical frameworks employed in the hydrodynamic and water quality models as well as the details of their calibration to observed field data have been presented previously (HydroQual, 1996). The purpose of this report is to provide information and model results from a series of nutrient management scenarios evaluated using the calibrated water quality model of the Sound. Of particular interest to the water quality managers of Long Island Sound are the concentrations of dissolved oxygen that prevail during the late spring and summer months. In particular, the managers are concerned with the temporal and spatial extent of dissolved oxygen concentrations at various "threshold" levels thought to affect various life-form stages of finfish and dimersal fish within the Sound. Information concerning these threshold levels of dissolved oxygen are being developed by the USEPA Narragansett Research Laboratory.

This report focuses on the projected response of Long Island Sound to various nutrient loading scenarios. In the normal sense "nutrients" are usually taken to be nitrogen, phosphorus and, for diatomaceous phytoplankton, silica. However, given the paradigm, which appears to be supported by an analysis of Long Island Sound nutrient concentrations and ratios of inorganic nutrients, that in the marine environment nitrogen and not phosphorus is the nutrient that limits phytoplankton growth and given the fact that silica reductions, even if they could be accomplished, would only affect diatoms and not other phytoplankton forms, only nitrogen, and not phosphorus and silica, nutrient management was considered in this analysis. However, since discharges of organic carbon adversely affect concentrations of dissolved oxygen and since dissolved oxygen is of importance to the living marine resources within the Sound, the nutrient management scenarios evaluated in this analysis also considered organic carbon load reductions.

Loading sources considered in the analysis included: municipal and industrial waste water treatment plants (WWTPs), combined sewer overflows (CSOs), storm sewer overflows, fall-line tributary inputs, coastal runoff, atmospheric inputs directly impinging on the water surface of the Sound, and exchange with the New York Harbor complex and the Atlantic Ocean. In order to provide a basis or reference for judging the relative benefits or effects of various nutrient management alternates, two model simulations, against which the reduction alternatives could be compared, were performed. These model runs were for "baseline conditions" and for "natural conditions". By baseline conditions it is meant that waste water treatment plant loads were generated using 1988 and 1989 plant flows (as were used in the 1988/89 hydrodynamic and water quality model calibration efforts), but using 1991 WWTP effluent concentrations. The reason for using 1991 WWTP effluent concentrations was to take into account changes in plant operations and plant upgrades achieved between 1988/89 and 1991, the year chosen for baseline conditions. Table 1-1 provides a summary and comparison of the annually averaged WWTP effluent flows and nitrogen and carbon loads for the calibration and baseline conditions. (A more detailed breakdown of these flows and loads for each of the 45 WWTP facilities that discharge to the Sound is presented in Appendix A).

TABLE 1-1. ESTIMATED ANNUAL AVERAGE WWTP EFFLUENT FLOWS AND LOADINGS OF NITROGEN AND CARBON TO LONG ISLAND SOUND

	Effluent Flow (MGD)	Total Nitrogen (lbs/day)	Total Carbon (lbs/day)
Calibration			
New York	1,037	115,622	306,776
Connecticut	165	21,544	45,864
Total WWTP to Sound	1,202	137,166	352,640
Baseline Conditions			
New York	1,037	133,509	306,776
Connecticut	165	21,544	45,864
Total WWTP to Sound	1,202	155,053	352,640

The baseline condition, then, represents an increase of approximately 13 percent in total nitrogen loading to the Sound above the calibration period. The remaining loadings to the Sound, i.e., CSOs, storm sewers, tributaries, atmospheric, etc., remained the same for the calibration and baseline runs.

A model run was also made to attempt to define "natural" water quality conditions in the Sound that might be expected in the absence of anthropogenic or man-made inputs. This run was performed by eliminating anthropogenic inputs of nutrients and organic carbon. The results of this run, then, provide information as to what dissolved oxygen levels might have been before European settlement of the Americas occurred. Bottom water dissolved oxygen concentrations from this run reflect the interactions between "natural" or background nutrient levels, net transport between the Atlantic Ocean on the east, Long Island Sound, and what is now New York Harbor and the New York Bight Apex on the west, and seasonal density stratification of the water column.

1.2 NUTRIENT REDUCTION SCENARIOS

The Long Island Sound Study Office provided information necessary to perform four nutrient management scenarios. These included:

(1) baseline conditions + centrate addition
(2) phase II nutrient reductions
(3) phase III nutrient reductions
(4) limit of technology

Table 1-2 presents a summary of the nitrogen and carbon loadings to the Sound from each of the loading categories (i.e., WWTPs, CSOs, etc.) for each of the four scenarios considered. Since the period simulated in the calibration of the water quality model extended over an 18 month period and encompassed two springs of markedly differing rainfalls and tributary inflows, the following table presents tributary, CSO and atmospheric loadings averaged for April through August for the 1988 and 1989 hydrologies and rainfall, as well as averaged loadings for the eighteen month period.

As a result of court-ordered mandates to end ocean-disposal of WWTP sludge, New York City has had to provide for the dewatering of treatment plant sludge and landside disposal of the dewatered sludge materials. Supernatant waste water or centrate from this dewatering process is then returned to the WWTP for treatment. As a result of this procedure NYC WWTP effluent nitrogen concentrations have increased approximately 16 percent above 1991 levels (Table 1-2).

TABLE 1-2. MANAGEMENT SCENARIO NITROGEN AND CARBON LOADINGS

	NITROGEN (lbs/day)				
	Baseline	Baseline + Centrate	Phase II	Phase III	Limit of Technology
WWTPs	155,053	179,469	133,990	68,652	39,247
Tributaries					
1988-1989[1]	138,898	138,898	138,898	120,424	122,323
1988[2]	121,430	121,430	121,430	104,178	103,404
1989[3]	199,738	199,738	199,738	175,129	180,938
CSOs					
1988-1989[1]	4,351	4,351	4,351	1,806	3,051
1988[2]	3,845	3,845	3,845	1,957	2,697
1989[3]	5,934	5,934	5,934	2,462	4,161
Coastal Runoff	19,747	19,747	19,747	10,577	14,000
Atmospheric					
1988-1989[1]	34,762	34,762	34,762	34,762	30,539
1988[2]	28,742	28,742	28,742	28,742	25,139
1989[3]	47,055	47,055	47,055	47,055	41,399

	CARBON (lbs/day)				
	Baseline	Baseline + Centrate	Phase II	Phase III	Limit of Technology
WWTPs	352,640	352,640	283,026	134,510	116,227
Tributaries					
1988-1989[1]	541,010	541,010	541,010	532,357	524,399
1988[2]	509,130	509,130	509,130	499,590	490,779
1989[3]	835,232	835,232	835,232	824,161	813,777
CSO					
1988-1989[1]	60,559	60,559	60,559	44,510	25,585
1988[2]	53,883	53,883	53,883	39,580	22,798
1989[3]	82,241	82,241	82,241	60,452	34,687
Coastal Runoff	83,730	83,730	83,730	73,757	63,615
Atmospheric					
1988-1989[1]	50,149	50,149	50,149	50,149	50,149
1988[2]	38,716	38,716	38,716	38,716	38,716
1989[3]	73,221	73,221	73,221	73,221	73,221

[1] Average daily load for April 1988 through September 1989
[2] Average daily load April through August 1988
[3] Average daily load April through August 1989

The magnitude of the Phase II nitrogen and carbon reductions are those that were committed to in the Long Island Sound Study (LISS) Comprehensive Conservation and Management Plan (CCMP). These reductions are to be achieved through upgrades and improvements in removal efficiencies at the WWTPs. This management scenario results in reductions of 14 percent and 20 percent for nitrogen and carbon, respectively, below baseline conditions, respectively. Phase III reductions reflect a 58.5 percent reduction in enriched sources of nitrogen from the New York and Connecticut portions of the watershed; carbon reductions consistent with the in-basin nitrogen management practices were also included. This management scenario represents an overall reduction of 33 percent for nitrogen and 23 percent for carbon relative to baseline conditions. The limit of technology (LOT) scenario reflects maximum nitrogen and carbon reductions that can be achieved with todays waste water treatment technology and available best management practices (BMPs) for agricultural and urban runoff. The LOT run provides for reductions of 41 percent for nitrogen and 28 percent for carbon relative to baseline conditions.

A fifth nutrient scenario, known as the "resource-based" scenario, was also investigated. This model run estimated the additional levels of nitrogen load reduction necessary to achieve dissolved oxygen conditions that would minimize any negative impacts on aquatic resources. It was known from analysis of previous model results that the LOT reductions would not be sufficient in magnitude to achieve the dissolved oxygen goals being set for the Sound and that further nutrient reduction would be required.

A more detailed description of each of the reductions or increases in nutrient loadings associated with each management alternative investigated is provided in Appendix A.

1.3 METHODOLOGY FOR PERFORMING NUTRIENT REDUCTION SCENARIOS

The nutrient reduction scenarios or projection runs were made using the eighteen month hydrodynamic transport fields used in the LIS 3.0 water quality model calibration effort. The eighteen month period extended from April 1988 through September 1989. It was assumed that any nutrient reductions achieved in any of the scenarios evaluated would result from reductions in effluent concentrations and not from reductions in effluent flow. Therefore, it was implicitly assumed that there would not be any change in freshwater flow delivered to the Sound and hence the 1988/89 hydrodynamic fields could be used. In order to take into account the changes in water quality concentrations at the model boundaries at Block Island Sound, on the east, and at the Battery and Spyten Duyvill, on the west, that would result from the changes in nutrient loading for a specific scenario run, an ocean boundary submodel was utilized. The goal of the ocean boundary submodel is to separate out the effects of internal loadings on the water quality boundary concentrations. A brief description of this submodel and its calibration are now presented.

1.3.1 Mass Balance Around Open Boundary

Assume a boundary segment with a well-mixed volume exists between the end segments of the LIS 3.0 model and an infinite reservoir of constant water quality, ex., the continental shelf (Figure 1-1). This infinite reservoir of "constant" water quality, wherein internal loadings from Long Island Sound do not influence water quality, will be called the "un-impacted" zone. The concentrations of the various water quality parameters in the un-impacted zone are assumed to be independent of the concentrations and loadings in the Sound model domain. As stated above for the Block Island boundary, this un-impacted zone is the continental shelf, while for the Battery this un-impacted zone would be found somewhere within New York Harbor.

A mass balance for the volume representing the boundary segment can be developed as follows:

$$\frac{dC}{dt} = Q_u \bullet C_u - Q_b \bullet C_b - Q_f \bullet C_b + E'_{sh} \bullet (C_{sh} - C_b) \qquad (1\text{-}1)$$

where:

Q_u = outflow, i.e., the flow leaving the Sound and entering the boundary segment, (L^3/T)

C_u = concentration of a water quality constituent in the outflow, (M/L^3)

Q_b = inflow, i.e., the flow leaving the boundary segment entering into the Sound (L^3/T)

C_b = concentration in the inflow, (M/L^3)

Q_f = the flow (representing the net freshwater flow to the Sound) that exits the boundary segment and flows into the un-impacted zone, (L^3/T)

E'_{sh} = the bulk exchange between the boundary segment and the un-impacted zone, (L^3/T)

C_{sh} = concentration of a water quality constituent in the un-impacted zone, (M/L^3)

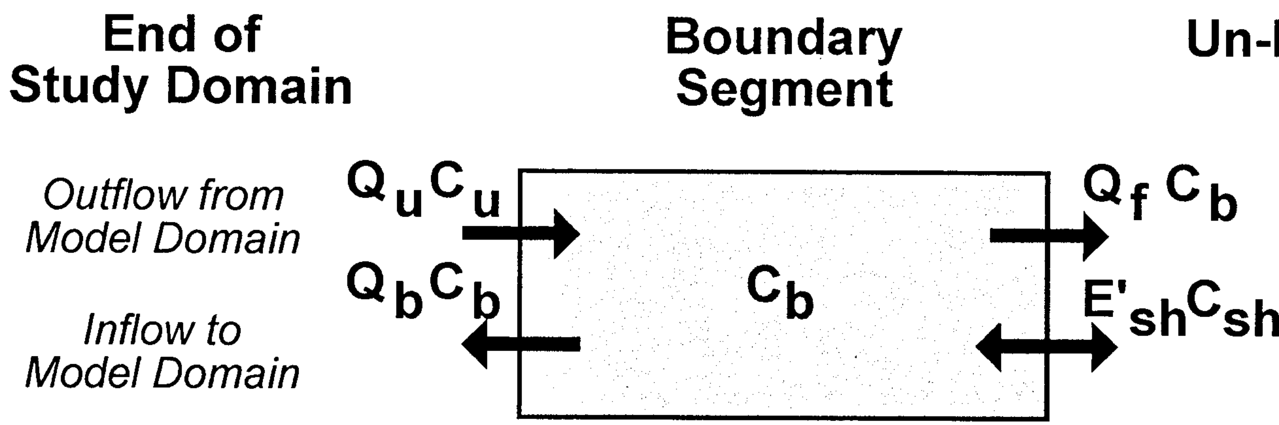


Figure 1-1 Diagram of Boundary Segment Submodel

In order to maintain flow continuity, Q_f must equal the difference between Q_u and Q_b, or $Q_f=Q_u-Q_b$. Assuming steady-state and rearranging terms, the concentration in inflow entering the Sound can be written:

$$C_b = \frac{Q_u}{Q_u + E'_{sh}} \bullet C_u + \frac{E'_{sh}}{Q_u + E'_{sh}} \bullet C_{sh} \tag{1-2}$$

Equation 1-2 indicates that the boundary concentration or the return concentration associated with the inflow is comprised of a mixture of the internal Long Island Sound water quality or outflow concentration and the concentration associated with the un-impacted zone. In other words, a fraction of the inflowing mass is made up of mass that previously exited the Sound. Defining α as per Equation (1-3),

$$\alpha = \frac{Q_u}{Q_u + E'_{sh}} \tag{1-3}$$

Equation 1-2 can be simplified to yield:

$$C_b = \alpha \bullet C_u + (1 - \alpha) \bullet C_{sh} \tag{1-4}$$

Note, as the value of α approaches 0, then the boundary condition is largely determined by the concentration associated with the un-impacted zone; as α approaches a value of 1, then the boundary concentration is largely determined by the internal concentrations of Long Island Sound. This latter condition is known as a reflecting boundary.

Once the reflection coefficient α and the concentration of a water quality parameter in the un-impacted zone, C_{sh}, can be determined, then, the boundary or inflow concentration simply becomes a function of the outflowing concentration. Thus it can reflect the effect of changes in internal loads on internal water quality concentrations and thus on the boundary conditions themselves. The evaluation of α and the estimation of the water quality concentrations in the un-impacted zone will be described in the following sections.

1.3.2 Block Island Sound Boundary

Although the above derivation of a reflecting boundary condition is presented using a simplified approach, in reality the boundaries of the LIS 3.0 model are considerably more complex. For example, the flows at the boundary are not constant in time but rather vary as a consequence of tidal action and the time-variable nature of freshwater entering the Sound from the Connecticut tributaries and WWTP discharges. In addition, the LIS 3.0 model employs a number of model cells to represent the boundary at Block Island Sound and also employs seven vertical layers. Therefore, the determination of α was slightly more complicated then presented above and involved a number of trial-and-error model runs to reach a final set of values for α. However, the general approach presented above and that is continued below can still be used to provide an overview of the boundary evaluation procedure to the interested reader.

Since the model boundary around Block Island Sound is open to the Atlantic Ocean, the reflection coefficient α may be determined from a salinity balance around Block Island Sound using Equation 1-4. Hence, after re-arranging terms α can be evaluated using Equation 1-5:

$$\alpha = \frac{S_b - S_{sh}}{S_u - S_{sh}} \tag{1-5}$$

where:

S_u	=	concentration of salinity just inside the eastern portion of Long Island Sound that enters the boundary segment (ppt)
S_b	=	concentration of salinity at the boundary interface of Long Island Sound (ppt)
S_{sh}	=	concentration of salinity in the un-impacted zone (ppt)

Due to data limitations it was necessary to use the monthly averaged salinities of two internal LIS 3.0 model cells, computed by the water quality model, as initial estimates of the outflow concentration, S_u, and the inflow concentration, S_b, in Equation 1-5. Next it was necessary to make an estimate of S_{sh}. Since the concentration of salinity was observed to increase as a function of distance from Block Island Sound towards the continental shelf break, as shown on Figure 1-2 (HydroQual, 1992), α also becomes a function of distance. Therefore, it was necessary to estimate a reasonable distance from the model boundary to the un-impacted zone. This distance was estimated in the following way. First, the outflow velocities, as computed by the hydrodynamic model during the 18 month simulation period, at the boundary cells were inspected to determine the maximum ebb velocity. Then this maximum ebb velocity of 1.1 m/sec was converted to the maximum distance a particle would travel during one-half of a tidal cycle. This resulted in a distance of approximately 15 miles. Using a distance of 15 miles and Figure 1-2, values of 32.2 and 32.8 ppt were used in Equation 1-5 as the salinity concentrations for the un-impacted zone for estimating α for the surface and bottom layers, respectively. These values of α, together with the un-impacted salinity concentrations of 32.2

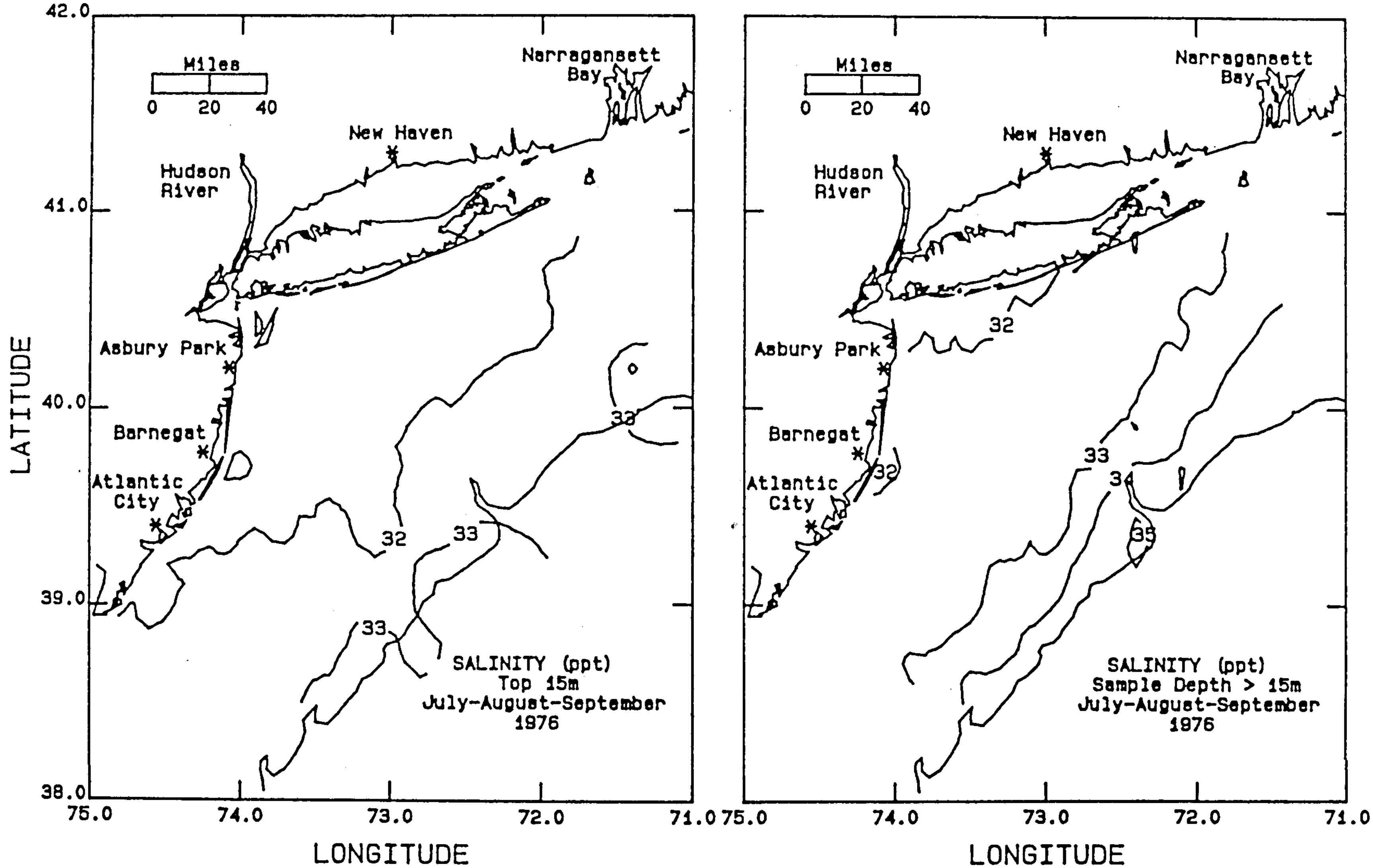


Figure 1-2 Distributions of Salinity in New York Bight, Summer 1976

and 32.8 were used in the water quality model to provide the eastern boundary conditions for the model on a time-variable basis. The model was run and the output was compared to the observed data and to the baseline calibration. Subsequently, adjustments were made to α until the model computations using reflecting boundary conditions were similar to those concentrations computed in the calibration using fixed boundary conditions. In this manner a final average α value of 0.75 was arrived at.

The next step in the boundary condition procedure was to estimate the concentrations of the other water quality constituents in the un-impacted zone. Initial estimates of the boundary concentrations for the other water quality constituents in the un-impacted zone were obtained by extrapolating the spatial gradients of these variables, as computed during the calibration, near the boundary. Again a series of trail-and-error runs were made, adjusting the concentrations of the water quality variables in the un-impacted zone until a favorable comparison to the calibration run was achieved. Figures 1-3 though 1-5 present the final concentrations of the water quality variables used for the Block Island Sound boundaries.

1.3.3 Battery Boundary

A similar analysis to that described above, was performed on the western boundary of LIS 3.0 at the Battery. However, due to the complexity of the New York Harbor system (Figure 1-6), high tidal velocities in the East River and the dilutional effects of freshwater flow from the Hudson River, the evaluation on α for the Battery boundary was more difficult. This difficulty was overcome, however, by making use of the fine-grid hydrodynamic/water quality model, HEM, that was being developed for New York City by HydroQual as part of the system-wide eutrophication modeling analysis. The method for determining α was through the use of a conservative tracer analysis using the HEM hydrodynamic circulation model. By using Equation 1-4 and by maintaining the concentration, C_{sh}, of dye in the un-impacted zone to be zero, α can be determined as follows:

$$\alpha = \frac{C_b}{C_u} \tag{1-6}$$

Equation 1-6 indicates that α is simply the ratio of dye concentrations in the inflow and outflow. A HEM run was made wherein 900 metric-tons of dye were released continuously at cell A (Figure 1-6) for 60 days. At cell B, which corresponds to the Battery boundary, the inflowing and outflowing dye concentrations were used to compute the α. Plots of dye concentrations at cells C and D were used to identify the potential edge of the un-impacted zone.

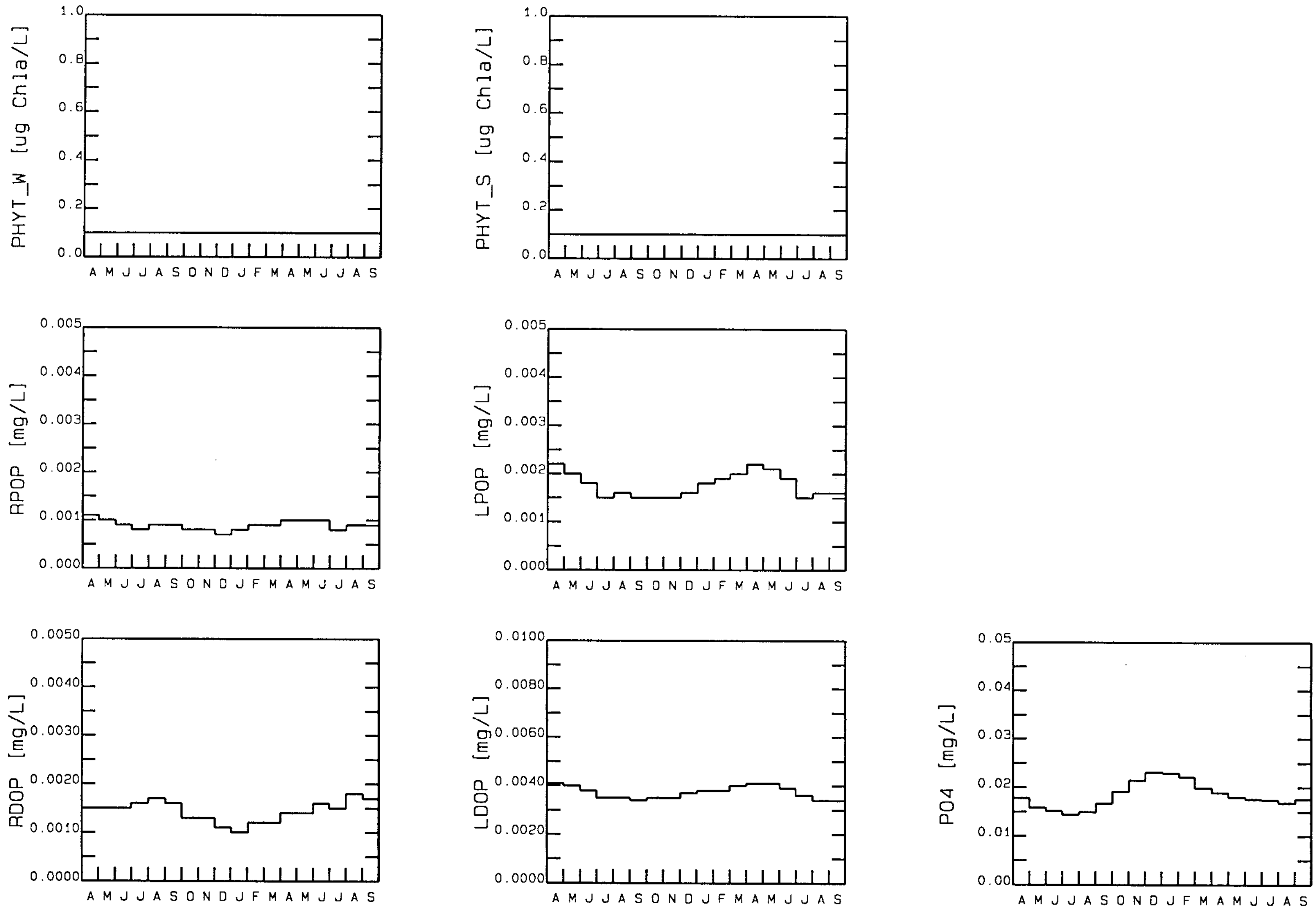


Figure 1-3 Temporal Profiles of Estimated "Un-Impacted" Concentrations of Winter and Summer Phytoplankton and Various Phosphorus Forms for the Block Island Boundary

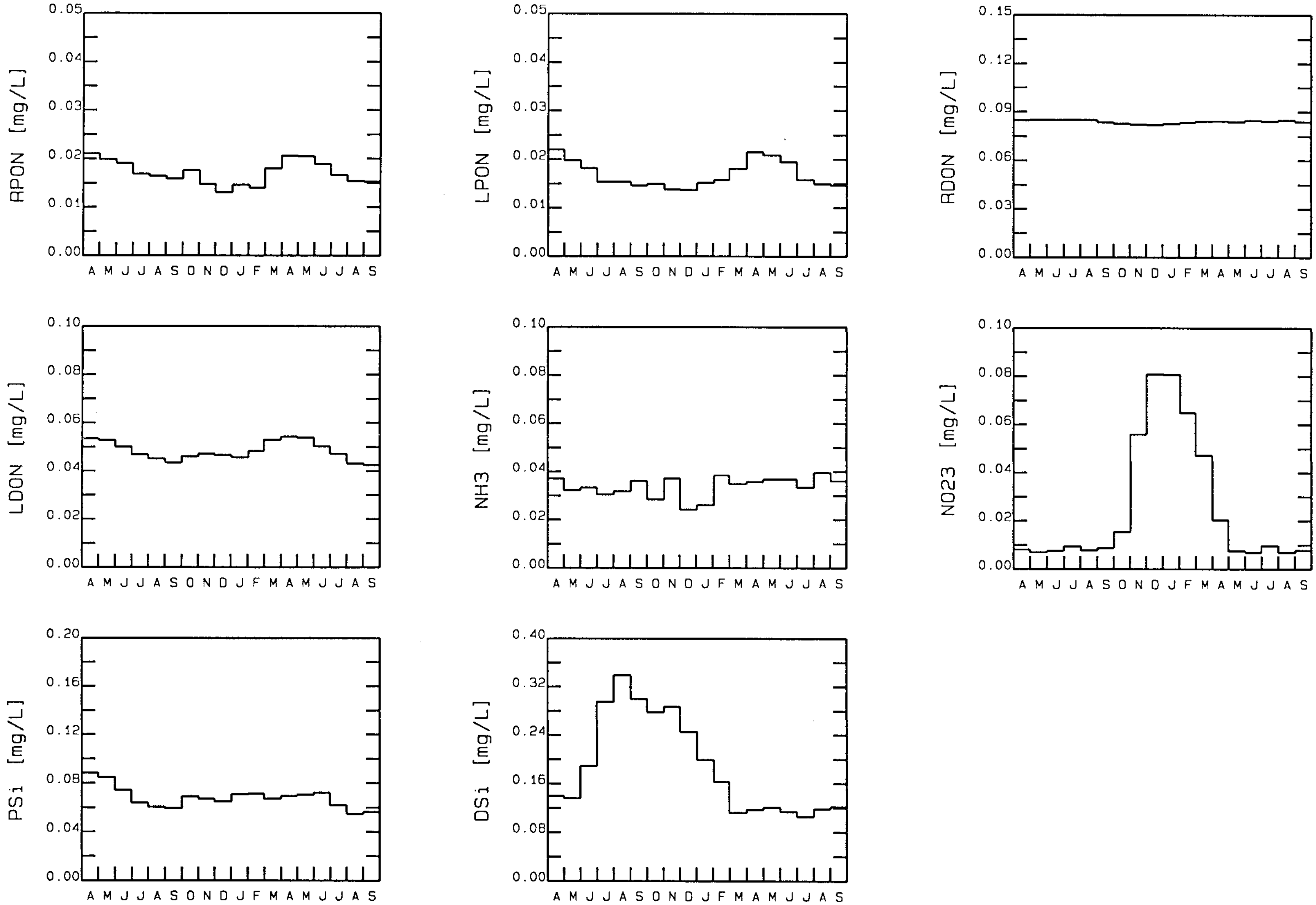


Figure 1-4 Temporal Profiles of Estimated "Un-Impacted" Concentrations of Various Nitrogen and Silica Forms for the Block Island Boundary

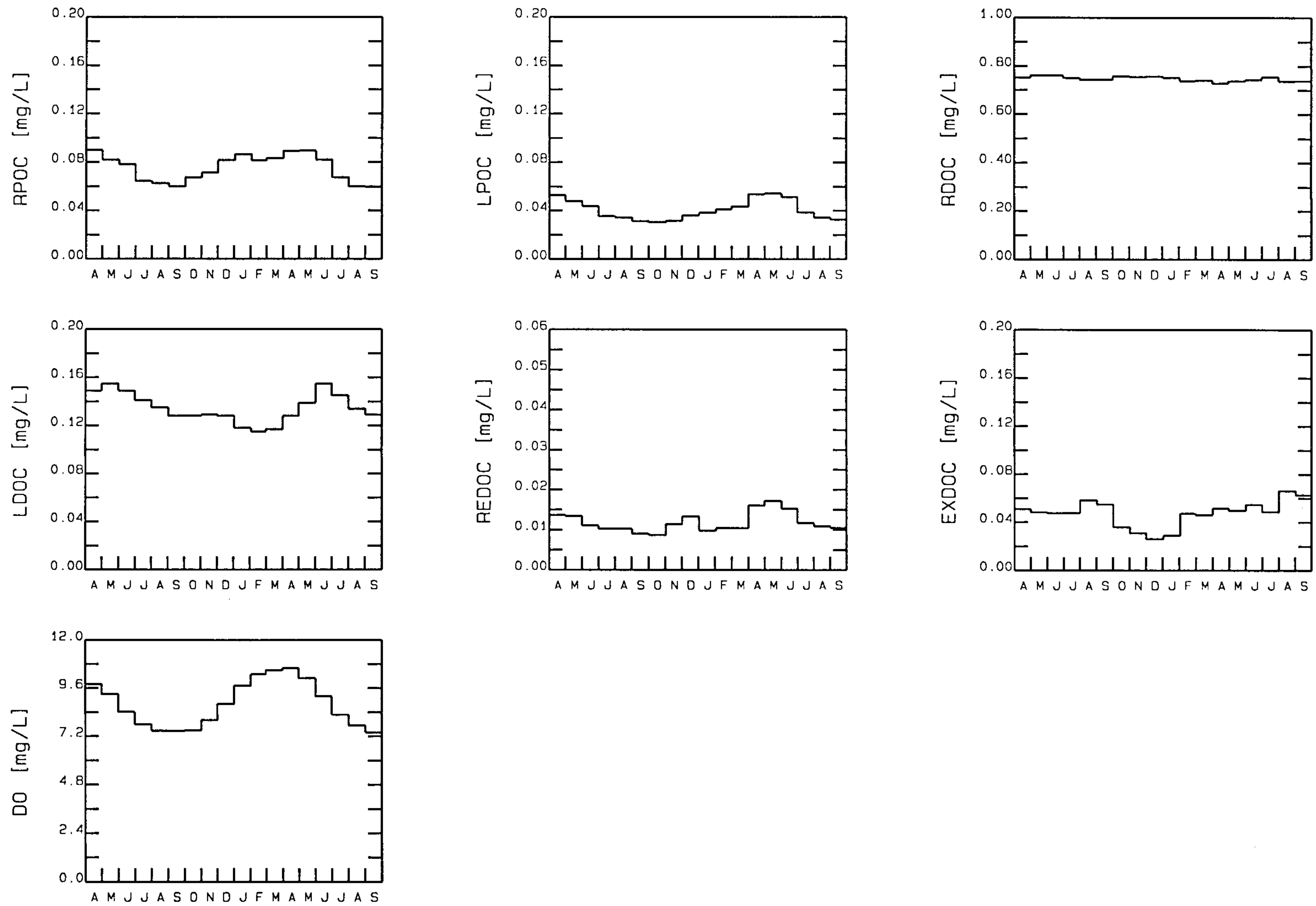


Figure 1-5 Temporal Profiles of Estimated "Un-Impacted" Concentrations of Various Organic Carbon Forms and Dissolved Oxygen for the Block Island Boundary

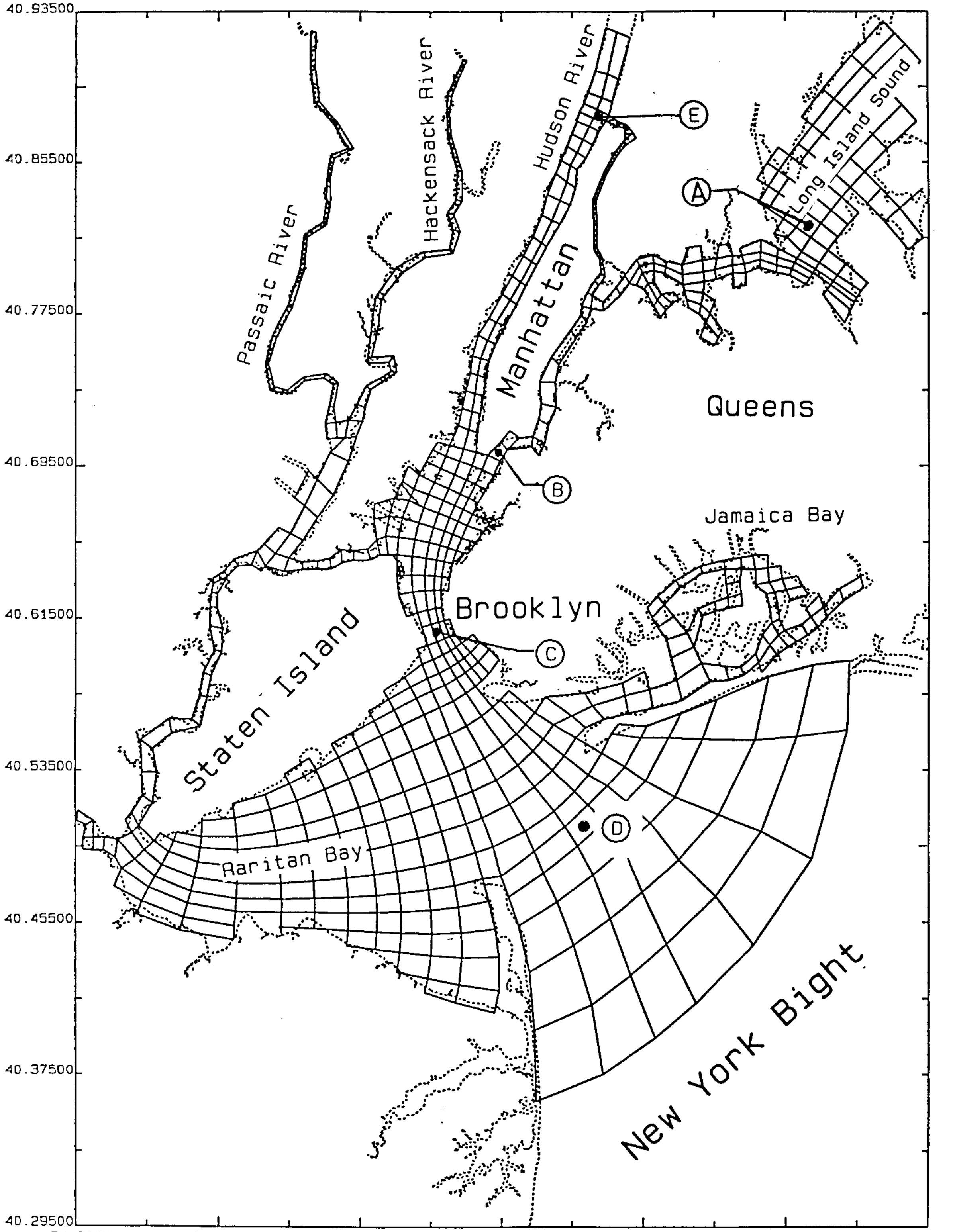


Figure 1-6 HEM Model Domain Showing Locations of Segments Used in Analysis of LIS 3.0 α for Battery Boundary

Figure 1-7 presents the dye distributions from this run, which indicate extremely small concentrations downstream of locations C and D. Thus the un-impacted zone is estimated to be located approximately 7 miles from the Battery. Figures 1-8 through 1-10 present the temporal distributions of α for layers 3, 6 and 9 of the HEM model, which correspond to the surface, middle and bottom layers in the LISS model, respectively. Averaging the information contained in Figures 1-8 through 1-10 results in an α of 0.85 for all three layers. This value of 0.85 implies that on average 85 percent of the mass exiting the lower East River on ebb tide is returned to the East River on flood tide.

The same procedures, described above for determining the concentrations of the various water quality constituents in the un-impacted zone, were employed in determining the Battery boundary. The concentrations of the various water quality constituents in the un-impacted zone, that were used for the projection runs, are present in Figures 1-11 through 1-13.

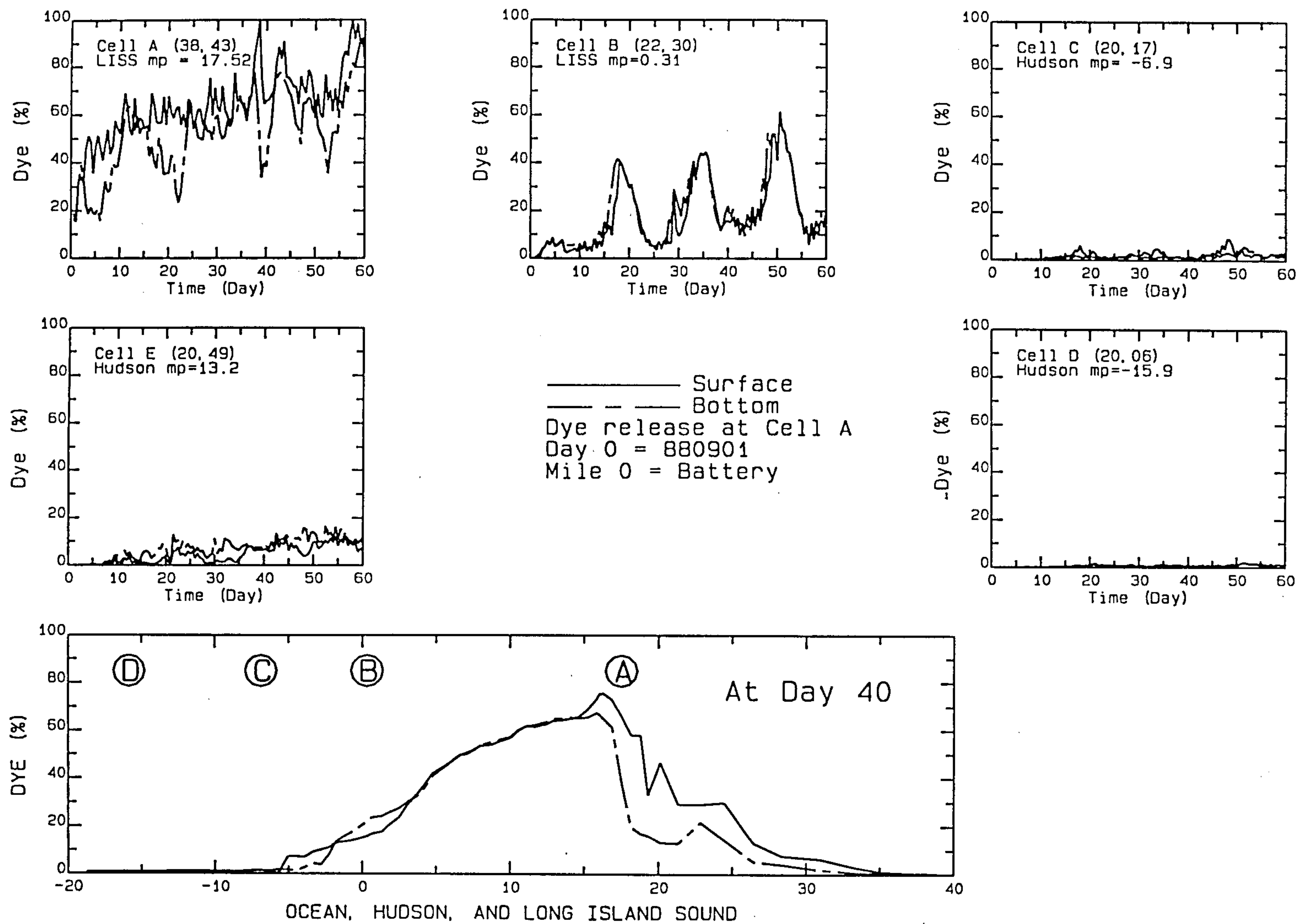


Figure 1-7 Results of Dye Study Using HEM Transport Model

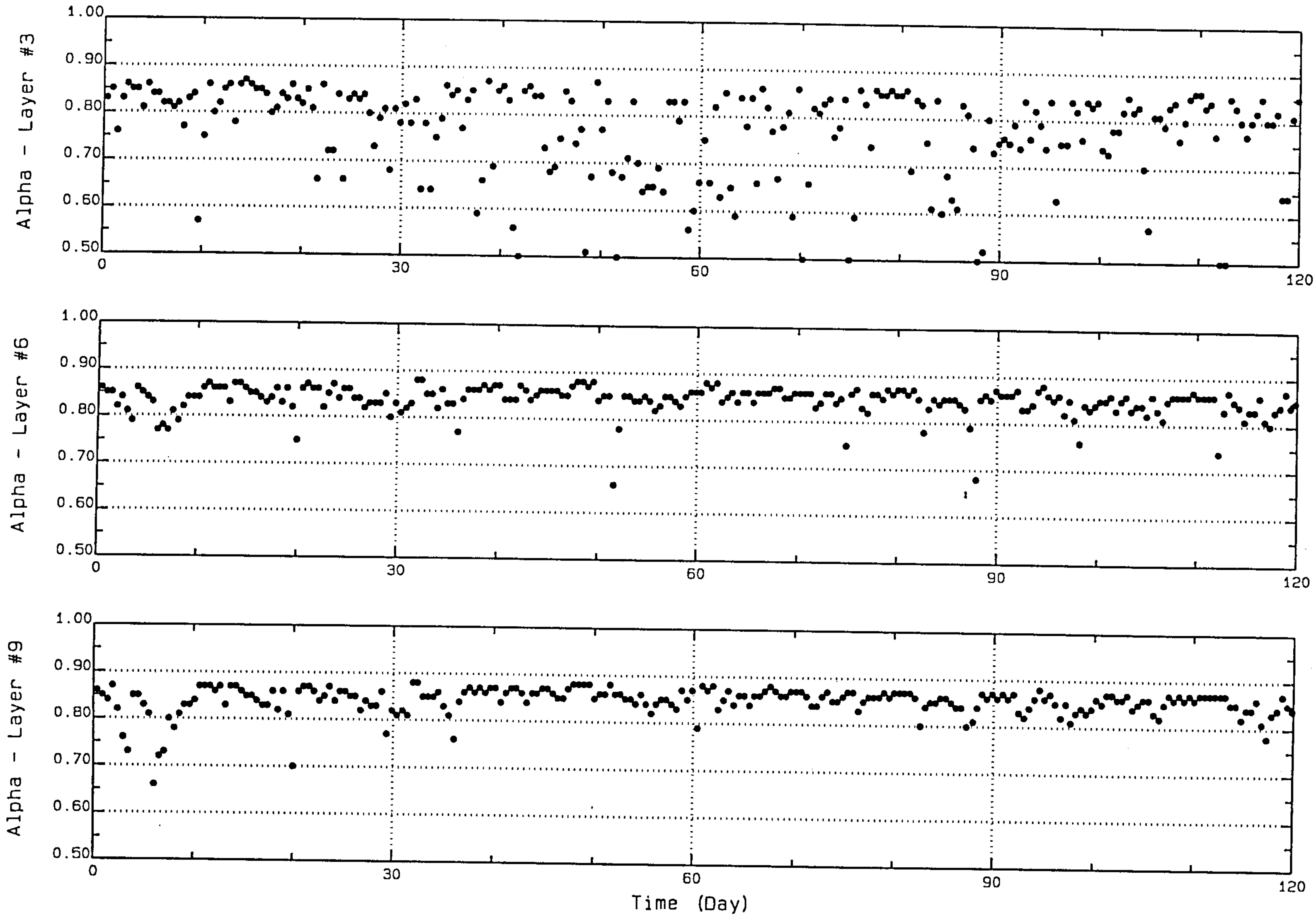


Figure 1-8 Plot of Battery Boundary α for Days 0-120

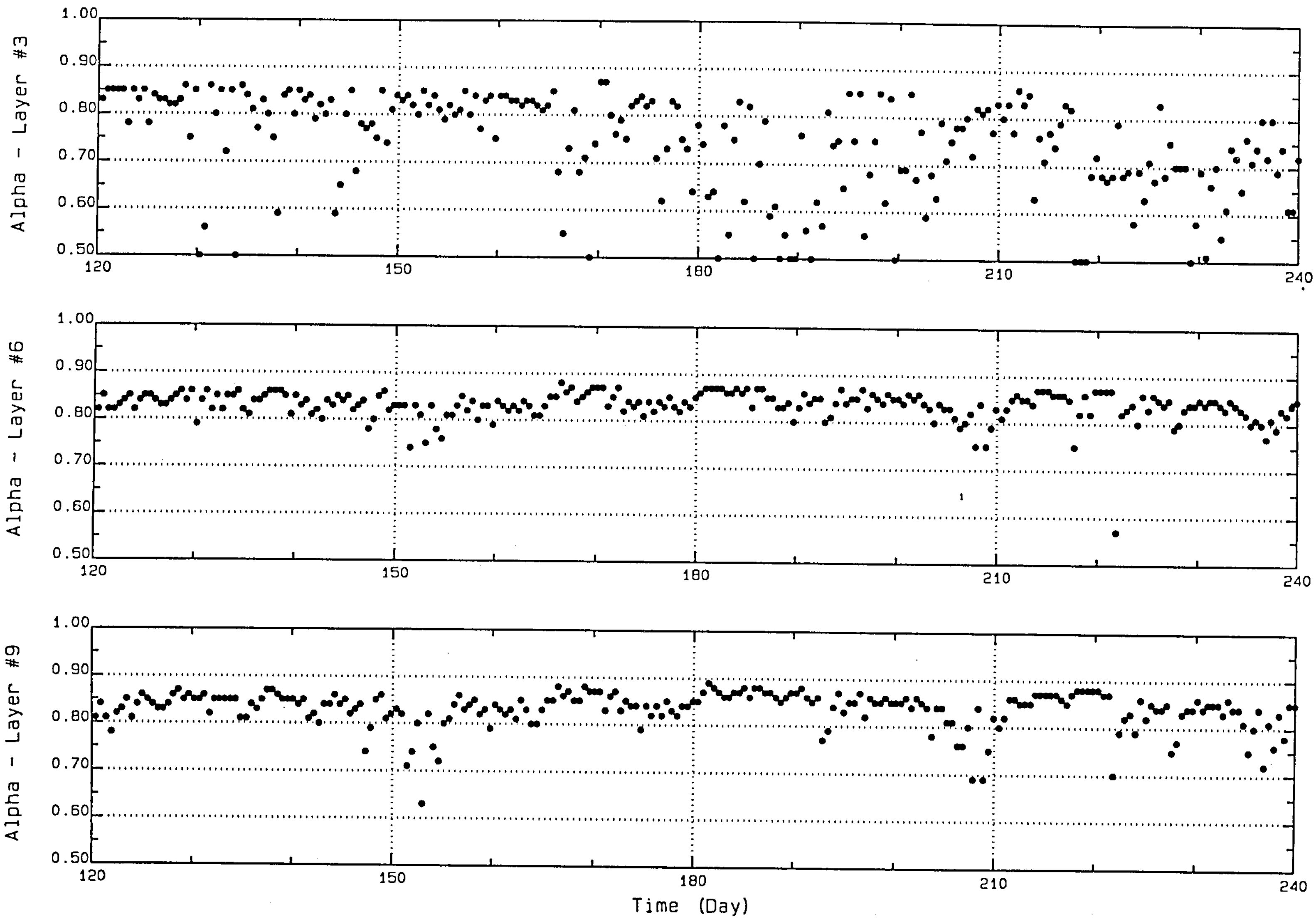


Figure 1-9 Plot of Battery Boundary α for Days 120-240

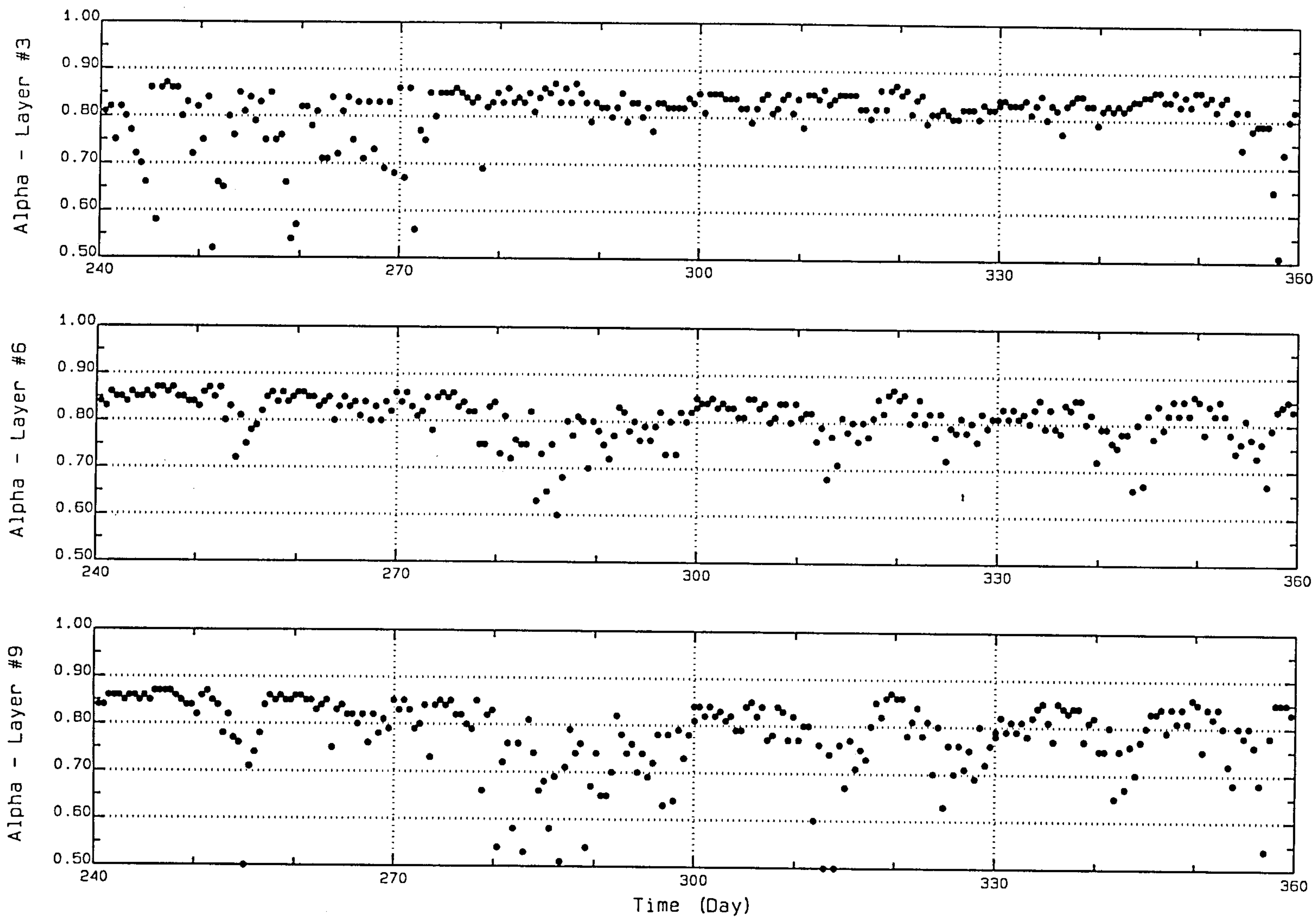


Figure 1-10 Plot of Battery Boundary α for Days 240-360

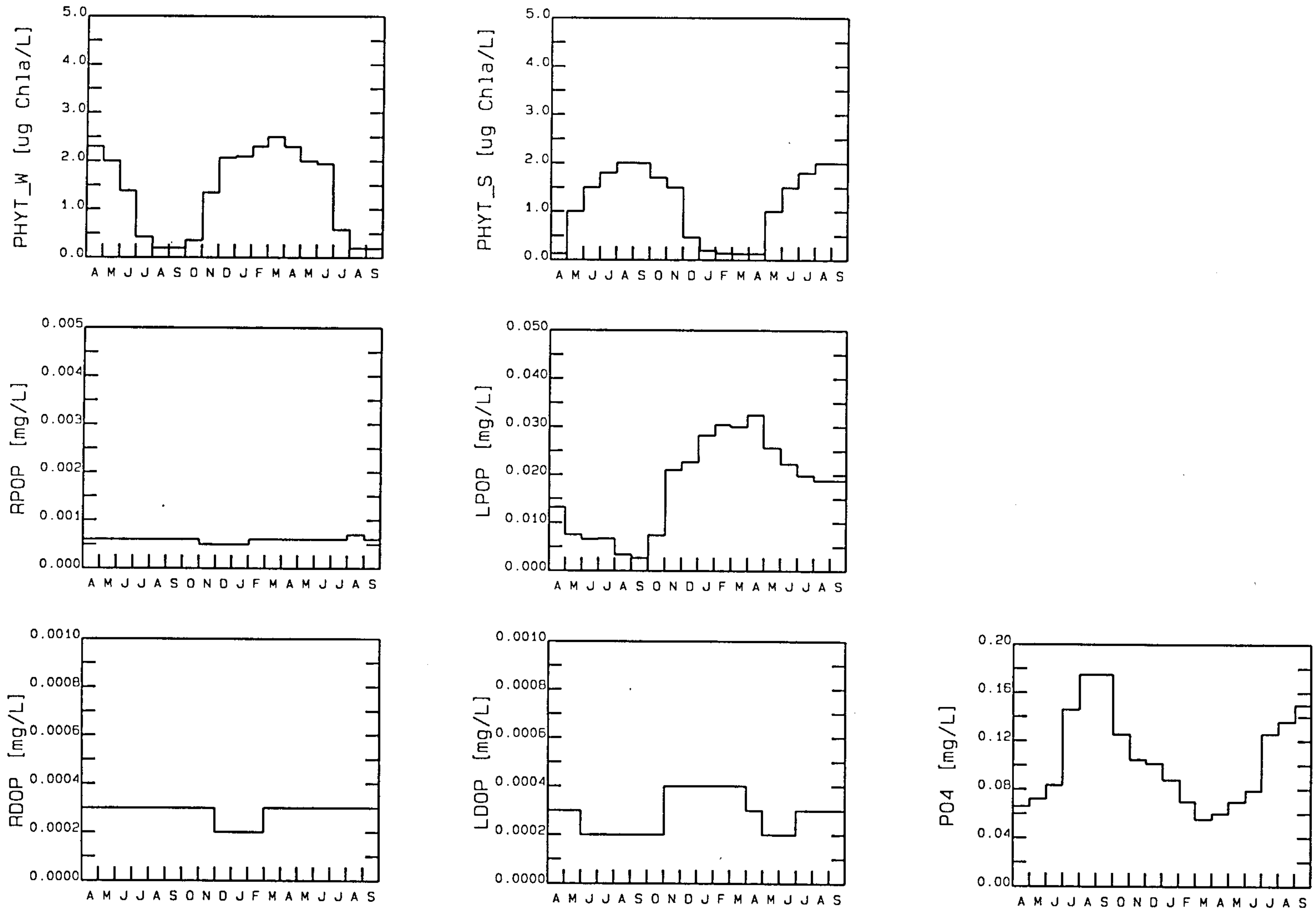


Figure 1-11 Temporal Profiles of Estimated "Un-Impacted" Concentrations of Winter and Summer Phytoplankton and Various Phosphorous Forms for the Battery Boundary

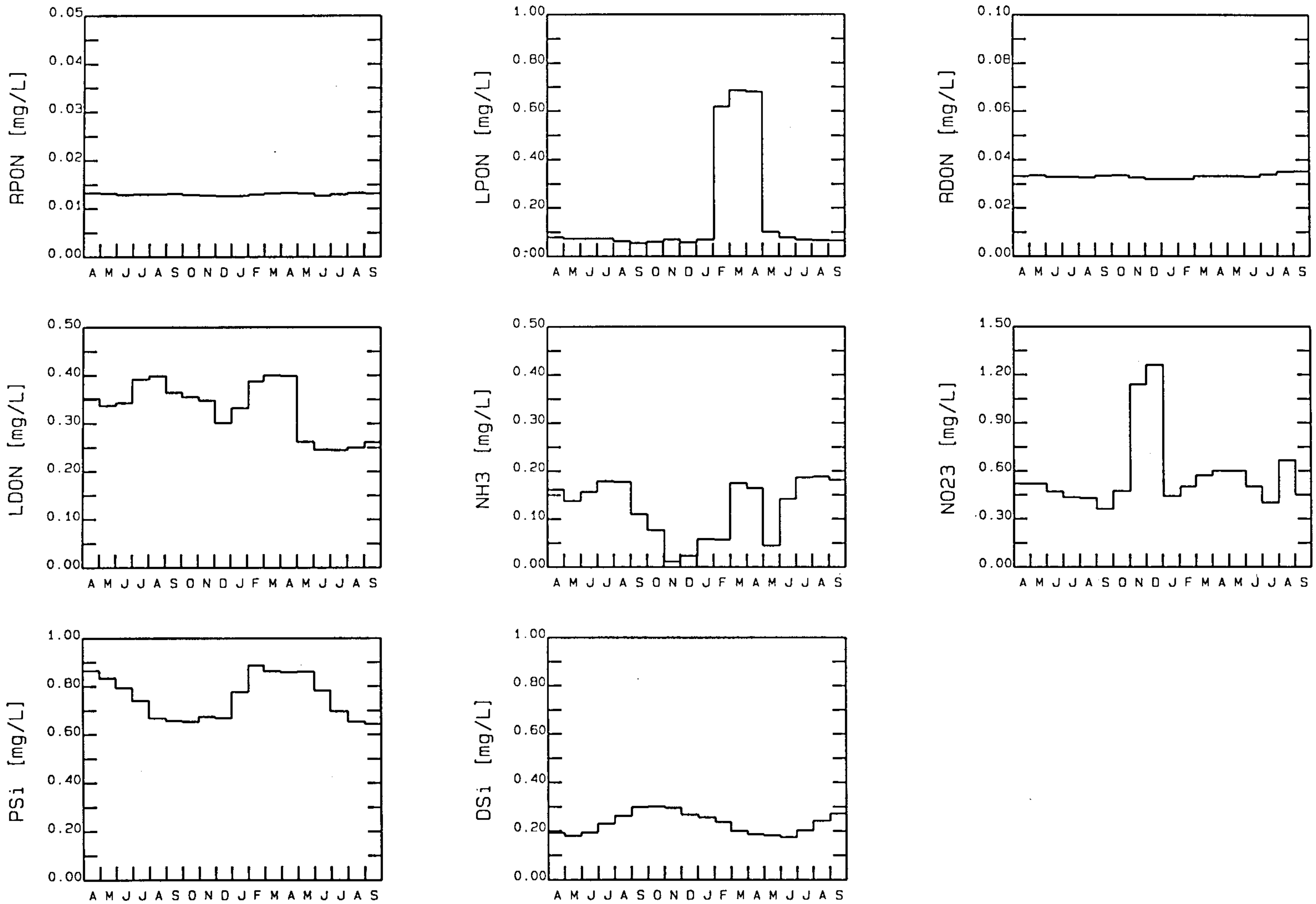


Figure 1-12 Temporal Profiles of Estimated "Un-Impacted" Concentrations of Various Nitrogen and Silica Forms for the Battery Boundary

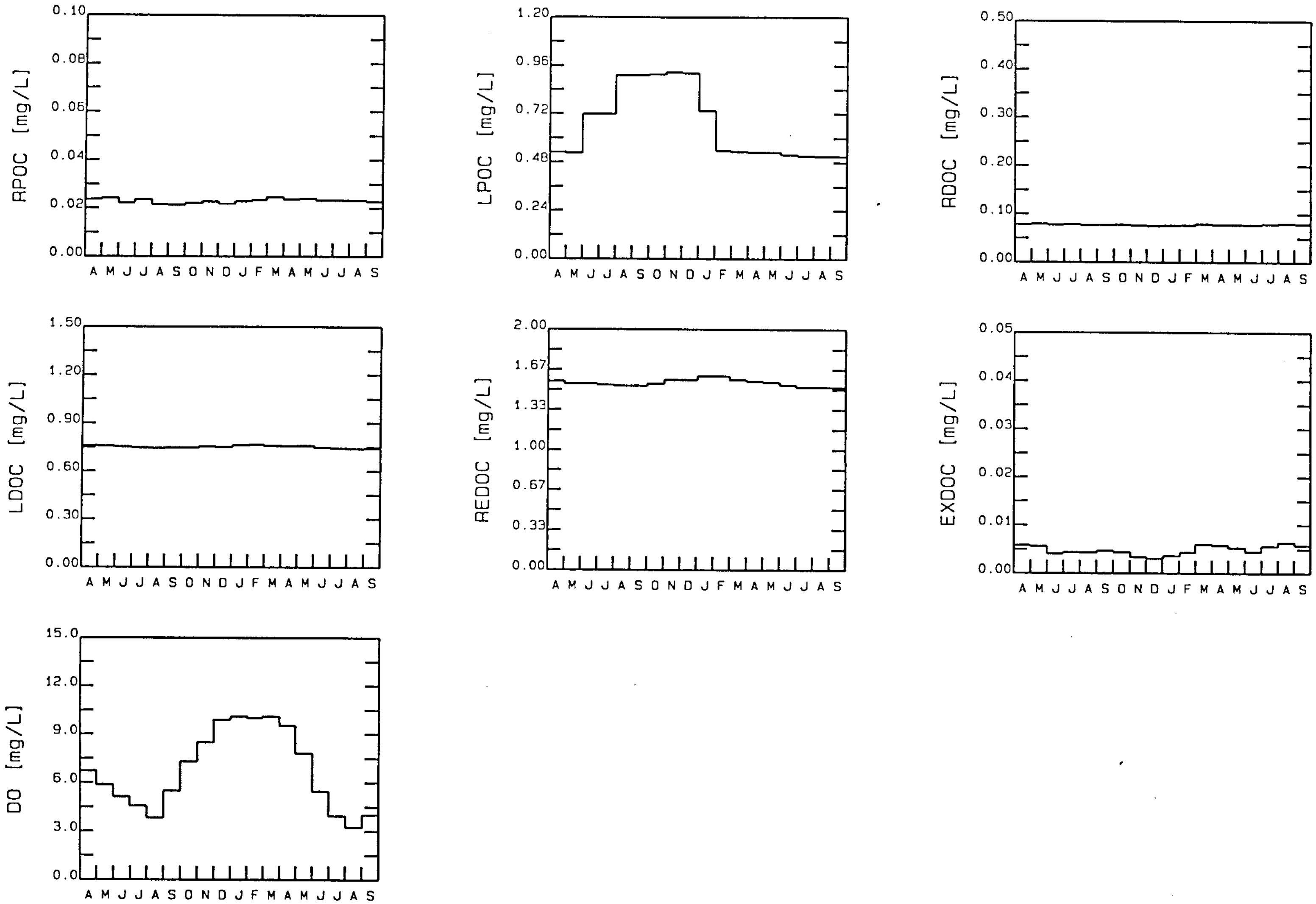


Figure 1-13 Temporal Profiles of Estimated "Un-Impacted" Concentrations of Various Organic Carbon Forms and Dissolved Oxygen for the Battery Boundary