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PHASE 1 REPORT - REVIEW COPY
INTERIM CHARACTERIZATION AND EVALUATION

HUDSON RIVER PCB REASSESSMENT RI/FS

EPA WORK ASSIGNMENT NO. 013-2N84

AUGUST 1991

Region II

ALTERNATIVE REMEDIAL CONTRACTING STRATEGY (ARCS)
FOR
HAZARDOUS WASTE REMEDIAL SERVICES

EPA Contract No. 68-S9-2001

VOLUME 1
(BOOK 1 OF 2)

TAMS CONSULTANTS, Inc.
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ABBREVIATED CONTENTS*

* SEPARATE, DETAILED TABLES OF CONTENTS PRECEDE THE INTRODUCTION, PARTS A, B AND C, TABLES, FIGURES AND PLATES.

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EXECUTIVE SUMMARY
PHASE 1 REPORT
INTERIM CHARACTERIZATION AND EVALUATION
HUDSON RIVER PCB REASSESSMENT RI/FS

BACKGROUND

For approximately 30 years, two General Electric (GE) facilities, one in Fort Edward and the other in Hudson Falls, NY, used polychlorinated biphenyls (PCBs) to make electrical capacitors. GE discontinued the use of PCBs in 1977, when they ceased to be manufactured and sold in the United States. From 1957 through 1975, various sources have estimated that between 209,000 and 1.3 million pounds of PCBs were discharged from these facilities into the Upper Hudson River. Discharges resulted from washing PCB-containing capacitors and minor spills.

The PCBs discharged to the river tended to adhere to sediments and subsequently accumulated with the sediments as they settled in the impounded pool behind the former Fort Edward Dam. Because of its deteriorating condition, the dam was removed in 1973. During subsequent spring floods, PCB-contaminated sediments were scoured and released downstream.

In 1976, the New York State Department of Environmental Conservation (NYSDEC) issued a ban on fishing in the Upper Hudson River, from Hudson Falls downstream to the Federal Dam in Troy, because of the potential risk posed by consumption of PCB-contaminated fish. The ban remains in effect today. A commercial fishing ban on the taking of striped bass in the Lower Hudson was also imposed by NYSDEC.

In 1984 the United States Environmental Protection Agency (USEPA) completed a Feasibility Study that investigated remedial alternatives, including dredging and upland containment of the contaminated sediments. Later that year USEPA issued a Record of Decision (ROD) for the Hudson River PCB Superfund Site. The ROD called for: 1) an interim No Action decision concerning river sediments; 2) in-place capping, containment and monitoring of remnant deposit (formerly impounded) sediments; and 3) a treatability study to evaluate the effectiveness of the Waterford Treatment Plant in removing PCBs from Hudson River water.

Since the ROD was signed, the in-place containment remedy for the remnant deposit has been virtually completed, and the treatability study of domestic water quality from the Waterford treatment facility concluded that the water supplied meets all current Federal and State standards.

In December 1989, USEPA announced that the No Action decision for the Hudson river sediments would be reassessed. This decision was based on several factors.

- *The Superfund Amendments and Reauthorization Act of 1986 (SARA) indicates a preference for remedies which "permanently and significantly reduce the volume, toxicity or mobility of the hazardous substance involved."*

- *USEPA policy calls for a periodic review at least every five years for as long as hazardous substances, pollutants, or contaminants that may pose a threat to human health or the environment remain at the site.*

- *Technological advances have been made in processes and techniques for treating and removing PCB-contaminated sediment.*

- *New York State Department of Environmental Conservation (NYSDEC) requested a reassessment of the No Action decision.*

The reassessment process consists of a an Interim Characterization and Evaluation which was previously identified in the Scope of Work as Preliminary Reassessment (Phase 1), Further Site Characterization and Analysis (Phase 2) and a Feasibility Study (Phase 3).

The Hudson River PCB Superfund site encompasses the Hudson River from Hudson Falls to the Battery in New York Harbor, a stretch of nearly 200 river miles. Upper Hudson refers to that 40-mile stretch of the river upstream of Federal Dam to Fort Edward. Lower Hudson refers to the portion of the river downstream of Federal Dam to the Battery.

PHASE 1 SCOPE AND OBJECTIVES

The Phase 1 Report is a comprehensive summary and evaluation of all available data for the site. It is based on a compilation of approximately 30,000 records of data on sediments, water, fish and aquatic insects, which are now entered into a computerized database. The purpose of compiling this data is to:

- *provide as accurate a picture as possible of current levels of PCBs in the river and changes in these levels since the 1970s;*
- *identify needs for additional data;*
- *allow a preliminary assessment of risks to human health and the environment posed by the PCBs in the river;*
- *make possible a preliminary assessment of potential remedies and treatment options for the PCB-contaminated sediments.*

It must be emphasized that the Phase I Report presents an interim evaluation only, based on currently available data. It is not intended to be a definitive characterization of the site or the risks associated with it nor to suggest any conclusions with respect to what remedies may be proposed at the end of the reassessment. During Phase 2, USEPA will complete characterization of the site. After completion of Phase 3, the Feasibility Study, USEPA will determine what remedies, if any, are appropriate.

MEASURING AND REPORTING PCBS IN ENVIRONMENTAL SAMPLES

An assessment of PCB contamination requires an understanding of their chemical complexity. Polychlorinated biphenyls are not a single chemical. They are a class of chemicals, containing from one to ten chlorine atoms per biphenyl molecule, yielding 209 possible molecular configurations. Laboratory analyses for PCBs are typically reported as Aroclor mixtures, referring to the manufacturer's trade name for PCB mixtures, with each Aroclor mixture containing a different amount of chlorine by weight. As chlorine content increases, the PCBs tend to be less soluble, more strongly adsorbed to sediments or bioaccumulated, and less likely to volatilize into the atmosphere.

Once released into the environment, environmental samples rarely contain PCBs that reflect the original Aroclor mixture. In addition to physical processes, biological dechlorination in sediments can alter the chlorine content so that it is different from the Aroclor mixture originally discharged.

Evaluation of existing PCB data is difficult, because different laboratories have used various methodologies to measure PCB mixtures. Chemical extraction procedures and analytical methods can also differ for different media. New laboratory methods are now available to resolve many uncertainties in PCB measurement. Uniform methods of PCB analysis in samples to be collected in future phases will improve understanding of the specific types of PCBs that remain in sediments, water, fish and air.

LOWER HUDSON

The data available regarding PCB contamination in the Lower Hudson are more limited than data for the Upper Hudson, which has been the subject of intensive monitoring programs to evaluate PCB contamination.

Physical site characteristics, including basin characteristics, hydrology, water quality and aquatic resources were reviewed in Phase 1. The review of aquatic resources, relying upon published studies, demonstrates the presence of a diverse aquatic ecosystem.

The total loading of PCBs to the Lower Hudson was historically dominated by inputs from the Upper Hudson, but has also been influenced by other sources of contamination. These sources include sewage effluent discharges, combined storm/sewer outfalls, stormwater outfalls, industrial discharges, atmospheric deposition, landfill leachates and tributaries below Federal Dam. Contributions of these additional PCB sources to the Lower Hudson, a drainage area of 5,285 square miles, are difficult to estimate, because they are poorly identified and quantified. Various researchers have estimated, however, that these sources currently contribute PCB loads on the same order of magnitude as the load from the Upper Hudson.

Data are available to document PCBs in sediment cores, surface water and fish in the Lower Hudson. Sediment core data indicate that maximum PCB deposition in the Lower Hudson occurred about 1973 and has since declined. A limited number of Lower Hudson water column measurements from 1978-1981 indicate that PCB levels declined from approximately 0.17 ug/l (micrograms per liter) in 1978 to approximately 0.07 ug/l in 1981.

NYSDEC has reported PCB measurements for fish from the Lower Hudson, mostly in the period of 1975-1988. The data show that PCB levels in striped bass, the dominant species monitored, have declined since 1978, with an apparent half-life of approximately five years. Recent measurements indicate that median PCB levels in striped bass are approximately 5 to 12 mg/kg (milligrams per kilogram) in the Upper Estuary (River Mile 91 to 153) and

approximately 2 to 3 mg/kg in the Lower Estuary (River Mile 12 to 76). Previous investigations have suggested that PCBs in striped bass of the Lower Hudson are dominated by the highly chlorinated PCB mixtures. This observation is of significant interest, because sediment data for the Lower Hudson suggest that there are sources of highly chlorinated PCB mixtures from the New York City metropolitan area. Further investigations will be needed to assess potential effects of remedial efforts to reduce PCBs in the Upper Hudson on PCB levels in the Lower Hudson.

Only a comparative and qualitative human health risk assessment was performed for the Lower Hudson during this phase. For the Lower Hudson, consumption of fish would likely lead to PCB exposures in human populations that are smaller than or comparable to those in the Upper Hudson, since PCB concentrations in water and fish from the Lower Hudson are less than those in the Upper Hudson. The assessment of risks in the Lower Hudson as a result of PCB loadings those from the Upper Hudson is complicated by the presence of multiple sources of PCBs within the Lower Hudson.

UPPER HUDSON

An understanding of how PCBs transfer, accumulate and dissipate in sediments, water, fish and other media is important to the characterization of existing conditions and an evaluation of the potential benefits of remedial actions in the Upper Hudson. Major items addressed in Phase 1 are:

- *potential for redistribution of PCBs in the river sediments;*
- *transfer of PCBs in the sediments to the water column and to fish; and*
- *human health and ecological risks from PCBs in the Hudson River site.*

These items will be further addressed in Phase 2 in order to determine appropriate remedial measures in Phase 3.

Sediment

Data on approximately 2,500 sediment samples are contained in the database, covering the two principal studies sponsored by NYSDEC in 1977-78 and 1984, as well as data from other studies by USEPA, GE and the Lamont-Doherty Observatory. The 1978 study covered approximately 40 river miles, while the 1984 study investigated PCBs in Thompson Island Pool, a five-mile stretch impounded by the Thompson Island Dam. The Thompson Island Dam is the first control structure in the river downstream of the GE plants. Several factors, listed below, hinder both detailed comparisons of results among surveys and analysis of PCB transfer from sediments to water and fish.

- *PCB measurements in sediments exhibit extreme variability over short distances.*
- *The shifting of river sediment deposits confounds comparison of sampling results at a given river location over time. Too few samples with adequate areal coverage have been taken to determine trends over time in PCB levels in sediment deposits.*
- *Because of different laboratory measurement techniques, different methods of reporting PCB concentrations and lack of sediment and water or sediment and fish data obtained at the same location, statistical relationships between PCBs in sediment and PCBs in water or fish have not been developed.*

The redistribution of PCBs in sediment is largely determined by sediment scour occurring during high river flow. The results of flood recurrence computed in this report show that previously published computations of the magnitude of the 100-year flood or other large floods

are overestimates. Thus, previous predictions of sediment scouring and PCB loadings will need to be re-examined based on the new estimates.

Water

The United States Geological Survey (USGS) results of monitoring PCBs in the water column from 1975-1989 were evaluated in Phase 1. Average total PCB concentrations in water during the late 1970s ranged from 0.2 to 0.8 ug/l (ppb) and have declined to approximately 0.03 to 0.05 ug/l in the late 1980s.

PCB mass transport in water is analyzed statistically to correct for sampling bias and account for the numerous data in which PCBs were not detected at the quantification limits set for the analyses. These findings are summarized below.

- *Although the greatest number of PCB measurements coincide with high flow periods, previous estimates have not addressed this bias in the sampling data. Previous PCB mass transport estimates are, thus, somewhat higher than those computed in this report.*

- *Mass transport of PCBs from the Upper Hudson at Waterford to the Lower Hudson has declined from approximately 3,000 to over 4,000 kg/year in the late 1970s to approximately 150 to 500 kg/yr in recent years.*

- *Since 1983, there has been little, if any, discernible difference in the mass load carried by the Hudson River from Rogers Island in Fort Edward to Waterford.*

This last observation is significant, because it suggests that in recent years there is little increase in PCB load in the Hudson River above that already in the river at Rogers Island. Whether the PCBs in the river at the Rogers Island monitoring station, which is immediately

downstream from the remnant deposit sediments, are derived from the remnant areas or other sources remains to be investigated further. PCB results for water samples taken in 1990, analyses of results for continued USGS water column monitoring and analyses of samples proposed for Phase 2 of this reassessment are expected to provide more definitive information.

Fish

Just as PCBs in the water column have declined since 1977, detailed analyses of trends over time indicate a similar decline for PCBs in fish. NYSDEC has reported analyses of PCB levels in fish sampled in the Upper Hudson since 1975. The dominant species sampled were largemouth bass, brown bullhead, pumpkinseed and carp. Median PCB levels in fish have declined from levels ranging from 3 to 143 mg/kg measured in the late 1970s to current levels ranging from 1 to 30 mg/kg. PCB levels in fish have also declined at downstream sample locations. Statistical analyses presented in this Phase 1 report reveal the following current conditions and trends.

- *The current upper-bound, 95 percent confidence limit of the average PCB level for all fish sampled in the Upper Hudson from 1986 to 1988 is approximately 12 mg/kg.*

- *Lower chlorinated PCBs in fish exhibit a half-life of approximately three to four years, whereas the higher chlorinated PCBs appear to be declining at a much slower rate and exhibit half-lives of 7 to 40 years, depending on fish species.*

- *A very strong linear correlation between PCBs in fish tissue and PCBs in the water column is apparent. The concentration of PCBs in fish tissue, based on lipid (fatty) content of fish, is on the order of one million times greater than the PCB concentration in the water column. This ratio is referred to as the*

bioaccumulation factor. Insufficient data are available to relate levels of PCBs in fish to those in sediment.

Additional fish samples were collected by the NYSDEC in 1990 and the analyses from these samples will be included in subsequent phases.

Sediment Transport Modeling

Development and calibration of hydraulic and sediment transport models have been initiated as part of the Phase 1 work. Because PCBs in the river are bound primarily to sediments, scour of sediments is a crucial mechanism to the movement of PCBs. A mathematical model provides one tool to predict potential scour and redeposition of sediments containing PCBs. A basic modeling framework has been developed in conjunction with the analysis of available data in order to determine the type and extent of modeling that may later be appropriate and feasible.

Preliminary Human Health Risk Assessment

Human exposure to PCBs in the environment or in the workplace generally does not result in any immediate or acute toxicity, but such exposures are of a public health concern because of the persistence of PCBs in the environment, their potential to bioaccumulate in animal and human tissues and their potential for chronic toxicity. Occupational exposures to relatively high concentrations of PCBs have resulted in effects on liver-function as well as effects on the skin. Emerging evidence indicates that PCBs may also be related to other toxic effects in humans, such as developmental or neurological effects. USEPA has classified PCBs as probable human carcinogens, based on the induction of cancers in laboratory animals.

A preliminary human health risk assessment for the Upper Hudson was performed during Phase 1. USEPA considered it important to establish at an early stage the working assumptions

for the risk assessment in order to allow the public sufficient opportunity to review and comment upon these assumptions. This preliminary health risk assessment will be modified in the future as new data become available, or if new information regarding the toxicity of PCBs is accepted by USEPA through a scientific review process before completion of the Reassessment.

Quantitative health risks associated with PCB exposure and uptake were calculated for pathways with adequate data. The exposure pathways quantified in the risk assessment are:

- *consumption of fish;*
- *drinking of river water; and*
- *incidental contact with PCBs in water and sediments during recreational activities associated with the river.*

Based on available data, there appear to be unacceptable potential cancer and non-cancer risks associated with regular ingestion of fish from the Upper Hudson River. This risk is based upon the assumption that local residents catch and consume fish from the Upper Hudson. However, as mentioned above, because of the potential risk posed by PCB-contaminated fish, NYSDEC issued a fishing ban in 1976. The ban remains in effect today. Recent surveys of angler activity by NYSDEC and fish consumption rates from NYSDEC and USEPA were utilized, but no specific survey of the population in question has been conducted to confirm these assumptions.

The risks associated with exposure as a result of drinking river water and recreational contact were estimated to be within the acceptable range.

Other potential exposure pathways, such as inhalation of contaminated air, consumption of local crops or dairy products and ingestion of breast milk by infants, are discussed, but not quantified. Although some available data indicate low levels of PCBs in plants and air in the vicinity of Fort Edward and the Hudson river, these data are insufficient to perform a quantitative risk assessment. Furthermore, the contribution of PCBs from other sources cannot be determined.

Interim Ecological Risk Assessment

The interim ecological risk assessment relies upon available data for selected indicator species in the Upper Hudson River ecosystem. Data are currently insufficient to justify a quantitative ecological risk assessment. Although thousands of samples have been collected of sediments, water and fish in the Upper Hudson River over a period of years, there are little data that relate PCB levels in the River to demonstrated harm to fish or other organisms. Only one published report was found regarding abnormal cell growth in fish from the contaminated stretch of the Upper Hudson. Although the researchers conclude that the observed abnormalities may be attributable to hazardous organics, additional study would be required to establish a causal relationship and to identify the chemical or chemicals responsible.

The concentration of PCBs measured in sediment, water, insect larvae and fish, and estimated in fish-eating birds and mammals, have been compared to published guidelines and toxicity values for PCBs. Recent PCB levels in water exceed freshwater Ambient Water Quality Criteria for the protection of aquatic life by two to five-fold. Additional evaluations will be necessary to assess species health and determine levels of ecological risk.

FEASIBILITY STUDY

Potential remedial technologies and processes capable of treating river sediments are identified in this initial step toward a final Feasibility Study. These technologies and processes are: containment; natural PCB degradation in sediments; removal; disposal; and treatment, including physical, chemical, thermal and biological treatment techniques. An initial screening of technologies is presented. No particular technology nor the possibility that no remedy may be warranted is eliminated from further consideration. Preliminary approaches to remedial options and recommendations for treatability studies are discussed.

PHASE 2

Based on the understanding reached in the Phase 1 process, field sampling and additional data evaluation are necessary in Phase 2 to provide improved understanding of PCB levels and transfer mechanisms among sediments, water, air and biota.

In order for the Hudson River PCB Oversight Committee (HROC) and the participants in the Community Interaction Program (CIP) to provide input into the Phase 2 work, these groups will be allowed sufficient time to evaluate the Phase 1 Report. Therefore, a full Phase 2 Work Plan will not be developed until after comments are received on the Phase 1 Report. There are some data, however, that USEPA believes should be collected in Fall 1991, because the data will be needed to guide subsequent sampling activities and allow the project schedule to be maintained. These priority sampling activities, which will be conducted this Fall, are described in the Phase 2A Sampling Plan. Activities included in this plan are geophysical surveys and water column monitoring in the Upper Hudson, as well as sediment corings in both the Upper and Lower Hudson.

When the Phase 2 Work Plan is issued for review, it will include a Phase 2B Sampling Plan and the description of other activities necessary to complete characterization of the site, as well as a summary of the Phase 2A activities. HROC and CIP participants will be allowed to review and comment on this Plan prior to initiation of this second sampling effort.

INTRODUCTION

PHASE 1 REPORT

INTERIM CHARACTERIZATION AND EVALUATION

HUDSON RIVER PCB REASSESSMENT RI/FS

CONTENTS

PLATES

Plates Located in Volume 1 (2 of 2)

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I. INTRODUCTION

I.1 Purpose of Phase 1 Report

This Phase 1 report provides an interim characterization and evaluation of the Hudson River PCB Superfund site. It summarizes the results of the first phase in a three-phase Reassessment Remedial Investigation/Feasibility Study (RI/FS) to reassess the 1984 No Action decision of the United States Environmental Protection Agency (USEPA) concerning sediments contaminated with polychlorinated biphenyls (PCBs) in the Upper Hudson River.

In December 1990, USEPA issued a Scope of Work for reassessing the No Action decision for the Hudson River PCB site. The scope of work identified three phases:

- Phase 1 - Preliminary Reassessment
- Phase 2 - Further Site Characterization and Analysis
- Phase 3 - Feasibility Study

This report presents the results of Phase 1 only. The Phase 1 report contains a compendium of background material, discussion of findings where findings could be made and preliminary assessments of risks.

The material presented here is not intended to characterize the site definitively nor to draw final conclusions. The principal reason for submitting this interim report early in the reassessment process is to offer sufficient background material to the USEPA, other concerned agencies, government officials, the Hudson River Oversight Committee, the Science and Technical Committee, the Steering Committee, liaison groups and the public, so that these parties can reach informed judgments concerning the technical direction and focus of the project through the various phases.

I.2 Purpose of Reassessment RI/FS

In December 1989, USEPA, Region II announced that it would conduct a reassessment of its September 24, 1984 No Action decision concerning the sediments contaminated with PCBs in the Upper Hudson River.

USEPA decided to reassess the No Action decision, based on the following events that have occurred since 1984.

- With the Superfund Amendments and Reauthorization Act of 1986 (SARA) came the indication that preferred remedies were those which "permanently and significantly reduce the volume, toxicity or mobility of the hazardous substance involved."
- USEPA policy is to perform periodic review for both pre-and post-SARA RODs at least every five years for as long as hazardous substances, pollutants, or contaminants that may pose a threat to human health or the environment remain at the site.
- Technological advances have been made in processes and techniques for treating and removing PCB-contaminated sediment.
- New York State Department of Environmental Conservation (NYSDEC) requested a reassessment of the No Action decision.

The reassessment is being performed for the PCBs contained within the river-bottom sediments of the Upper Hudson between Hudson Falls and Federal Dam in Troy, New York (see Plate I.2-1). The Superfund site itself, however, extends to the Battery in New York Harbor (see Plate I.2-2).

The reassessment also evaluates the threat of PCBs entering the river from the remnant area (see Plate I.2-3) and assesses environmental impact on the Lower Hudson. Previously dredged PCB-contaminated sediments contained in upland disposal facilities will not be addressed in this study.

I.3 Site History

I.3.1 Prior to 1980

PCBs were manufactured by Monsanto Corporation between 1927 and 1977 and were distributed under the generic name Aroclor. Two General Electric Company (GE) capacitor manufacturing plants located in Fort Edward and Hudson Falls, New York began to use PCBs in 1946. In-plant sources of PCB discharges have been characterized as both minor spills and effluent from washing capacitor cans, with the latter being the major source. Capacitor cans were flood-filled with dielectric fluid and then washed with detergent and water to remove excess material. Contaminated wash water was discharged directly to the river. This practice was discontinued about 1973 (Brown, Jr. *et al.*, 1984).

During a 30-year period from 1946 to 1977, PCBs were discharged into the Upper Hudson River from the two GE plants. Discharged PCBs adhered to the sediments in the bottom of the river and accumulated in areas behind the Fort Edward Dam. When the dam was removed in 1973 because of its deteriorating condition, PCB-contaminated sediments were released downstream, particularly during large spring floods in 1976 and 1983.

PCBs have been associated with a variety of adverse health effects. Studies performed on rats, mice and monkeys have revealed that various kinds of toxic effects are associated with PCBs, such as liver damage, reproduction effects, skin disturbances and cancer.

The first report of PCB contamination in the Hudson River was published in 1970. In 1971, NYSDEC added PCBs to their statewide analyses of pesticide residues in fish, although no results were released publicly until 1975. After USEPA investigations in 1974 of PCB contamination in the Fort Edward area, NYSDEC intensified its PCB sampling program. In 1976, following the 1975-1976 sampling effort, NYSDEC banned all fishing on the Upper Hudson River, from Albany north through Fort Edward. The commercial striped bass fishery in the Lower Hudson was also closed at the same time. The bans are still in effect today. In addition,

the presence of PCBs restricted dredging activities. The New York State Department of Transportation (NYSDOT) had periodically dredged the river, which is prone to sediment buildup, in order to maintain a minimum depth to accommodate river traffic. According to NYSDOT, no channel maintenance dredging has occurred from 1984 to the present (1991).

I.3.2 Post 1980

In the 1980s, site activities diverged in two directions. One direction was pursued by the NYSDEC Project Sponsor Group (PSG) and included the Hudson River PCB Reclamation Demonstration Project, which in 1989 became the more comprehensive Project Action Plan. The second direction was pursued by USEPA and included the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) and Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) or Superfund process. Only the direction pursued by USEPA under the NCP/CERCLA process is discussed below. The PSG activities are described in detail in the Community Relations Plan (December 1990.) for this Reassessment.

The 1984 Record of Decision (ROD) stated that a technologically feasible, cost-effective, remedial response to the river sediments was not available that would reliably and effectively mitigate and minimize damage to public health, welfare and the environment. This decision was based on the results of the NUS Feasibility Study (FS) dated April 1984. At that time it was deemed more appropriate to address the sediments in connection with the Hudson River PCB Reclamation Demonstration Project being pursued by the PSG. There were several reasons for this decision: (1) the modeling and sampling data collected at that time indicated a decreasing threat to public health and the environment; (2) the reliability and effectiveness of extant dredging technologies were subject to considerable uncertainty; (3) the estimated high cost of dredging and disposal were considered likely to rule out such options, based on fund balancing considerations, especially given the moderate degree of risk reduction which might have been achievable. The ROD stated that this decision was to be reassessed in the future if, during the interim evaluation period, the reliability and applicability of *in situ* or other treatment methods were to be

demonstrated or if techniques for dredging of contaminated sediment were further developed.

The 1984 ROD addressed five remnant deposits (see Plate I.2-3) behind the Old Fort Edward Dam and the river sediments. The ROD reflected USEPA's decision to perform in-place containment, or capping, of the remnants, stabilization of the associated riverbanks and revegetation of the areas. As stated in the ROD (USEPA, 1984): "The appropriateness of further remedial action for these sites will be reexamined, if EPA decides at a later date to take additional action with respect to sediments in the river." The construction of the remnant caps is essentially complete. No in-place containment was required at one of the remnant deposits (site 1).

The 1984 ROD also included performance of a treatability study to evaluate the effectiveness of the Town of Waterford's treatment plant in removing PCBs from Hudson River water. The Town of Waterford is located 40 miles south of the remnant deposits and was selected for evaluation, because it is the northernmost community downstream that receives its water supply directly from the Hudson River. Findings indicated that PCB levels in the water supplied by the Waterford Water Works did in fact meet standards for public water supplies.

I.4 Guide to Phase 1 Report

I.4.1 Relationship to Phase 1 Work Plan

The Phase 1 Work Plan (January 1991) detailed five main tasks to be completed during Phase 1.

- Task 1 - Site Characterization and Data Synthesis
- Task 2 - Evaluation of Fish and Food Chain PCB Bioaccumulation
- Task 3 - PCB Transport Model

- Task 4 - Baseline Risk Assessments
- Task 5 - Remedial Technology Assessment

Rather than report results of Phase 1 by these Work Plan tasks, the information gathered and the findings of this phase are presented here in a format that is more consistent with the CERCLA RI/FS reporting process. This format will be compatible with the final product of this project, which will be a Reassessment RI/FS report.

Phase 1 was originally called the Preliminary Reassessment. As a consequence of discussions at the Hudson River Oversight Committee (HROC) meeting of April 4, 1991, the term "preliminary reassessment" was considered potentially misleading and a new designation for Phase 1 was considered appropriate. In order to stress the fact that additional characterization and evaluation of the site are required prior to a final characterization and prior to any decision on reassessment, the title Interim Characterization and Evaluation Report has been chosen.

I.4.2 Organization of Phase 1 Report

This Phase 1 report is Volume 1 of the overall three-phase study. In order to accommodate the amount of material presented in Phase 1, two books have been prepared. Book 1 of Volume 1 contains all text, divided as Parts A, B and C. Book 2 of Volume 1 contains all tables, figures, and plates.

A table of contents introduces each Part (A, B and C) in Book 1. Each Part is paginated by section and page number within that section. For example, page B.6-7 is page number seven (B.6-7) within Section B.6 (B.6-7). Tables, figures and plates referenced in the Introduction (I) and Parts A, B and C are designated so that they can be easily referenced to a report section. For instance, Table B.3-9 indicates that the table was referenced in Section B.3 (B.3-9). The nine (B.3-9) further indicates that the table is the ninth table within that section.

Synopses intended to guide the reader in selecting areas of focus or interest are provided in Book 1. A synopsis of Part A and of Part C is located at the beginning of each part. Because of the length of Part B, separate synopses for Sections B.1 through B.7 are provided prior to each section.

The interim site characterizations of the Lower Hudson and Upper Hudson are presented in Book 1, Parts A and B, respectively. Initial emphasis has been placed on the Upper Hudson, because the basic project premise is to reassess options for remediation of the PCB-contaminated sediments of the Upper Hudson. This approach is not intended to diminish the importance of evaluating the impacts on human health and the environment of the Lower Hudson as a result of PCB contamination in the Upper Hudson. Such impacts will be pursued more emphatically in Phases 2 and 3.

The interim site characterization for the Lower Hudson (Part A) is based on a summary of available literature; particular attention is paid to sources of PCBs into the Lower Hudson and to the river's aquatic ecology. As no comprehensive data synthesis and evaluation were undertaken at this time, the reader should not expect new or previously unreported analyses. For those readers not familiar with the chemical aspects and structure of PCBs, an explanation is provided in Section A.2.

The interim site characterization for the Upper Hudson (Part B) is based on comprehensive collection, synthesis and evaluation of available data. The understanding or interpretation of the information and analyses presented for the Upper Hudson demands varying levels of technical knowledge. To assist the reader, an overview of the nature of Part B sections is provided here.

Sections B.1 and B.2, both general in nature, report on the physical site characteristics and the sources of PCB contamination in the Upper Hudson.

Section B.3 discusses the available data that were collected, reviewed and synthesized for sediments, water, fish, air, and plants. Parts of this section utilize statistical techniques, some of which will not be familiar to all

readers. The conclusion of Section B.3 discusses the adequacy of the data and presumes some knowledge of analytical chemistry.

Section B.4 analyzes the interrelationships among data sets and examines trends in the data over time in order to extrapolate trends in PCB levels and transport into the future. Although this section relies heavily upon statistical techniques, readers will gain new insights into the data, some of which contradict and/or extend existing knowledge.

Section B.5, which reports on sediment transport modeling, is highly mathematical and oriented to those readers who will wish to provide comment on or input to the modeling effort. Achieving that objective has dictated a mathematical presentation.

Finally, in Part B, Sections B.6 and B.7 provide preliminary human health and ecological risk assessments, utilizing information reported in the previous sections. All readers are reminded that the findings presented in these sections are preliminary since they are based on available information.

Part C contains Phase 1 of the Feasibility Study for the Upper Hudson. Presented here is a review of regulations, available technologies and processes to treat PCB sediments, some innovative technologies under development and the status of natural dechlorination or biodegradation of PCBs in sediments.

Following Parts A, B, and C in Book 1 are the list of references and a glossary. Readers are invited to suggest changes to the list of references and to recommend additional terms for future expansion of the glossary, as both of these will form the basis of an expanded reference list and glossary in subsequent documents.

SYNOPSIS

LOWER HUDSON CHARACTERIZATION

(Sections A.1 through A.4)

Part A provides an interim characterization and evaluation of Lower Hudson River characteristics pertinent to the Hudson River PCB reassessment. Presented here are physical site characteristics, sources of PCB contamination, the nature and extent of Lower Hudson PCB contamination and an overview of a published mathematical model by Thomann et al. (1989) on PCB dynamics in the Lower Hudson.

The discussion of physical site characteristics (A.1) contains information on basin characteristics, hydrology, water quality and aquatic resources. The description of basin characteristics (drainage areas and climate) covers both the Upper and Lower Hudson to establish a framework for the entire site. The discussions of hydrology and water quality for the Lower Hudson describe the physical/chemical factors that affect each. The review of aquatic resources, relying upon published studies, demonstrates a diverse aquatic ecosystem.

There are several sources of PCB contamination (A.2) in the Lower Hudson. PCB loadings to the Lower Hudson have occurred from the Upper Hudson, but also from sewage effluent discharges, tributary contributions, combined sewer/storm water and storm water outfalls, atmospheric deposition, landfill leachate and other sources within the New York City metropolitan area, all within the Lower Hudson Basin itself. These additional PCB sources are important to consider, since some have been estimated to contribute PCB inputs of similar magnitude to current loads from the Upper Hudson.

The nature and extent of PCB contamination is analyzed, using available data for sediments, water and fish (A.3). As demonstrated by dated sediment cores, maximum PCB deposition in the Lower Hudson occurred around 1973 and has decreased subsequently. Sediment cores also indicate that sediment influenced by New York City metropolitan area inputs has recently been accumulating higher PCB levels than those farther upstream. Although water column PCB measurements since 1981 are lacking, 1978-81 data show that PCB levels declined during that period. Studies indicate that PCB concentrations in striped bass have declined. For migrant/marine fish species and freshwater resident species, data are limited or dated.

A mathematical model of PCB dynamics in the Lower Hudson (Thomann et al., 1989) is examined (A.4). This model considers many aspects of mass transport, geochemistry and ecology and evaluates the time history of PCB inputs. The model indicates that PCB load to the Lower Hudson via the Upper Hudson had declined substantially since 1973. Various assumptions used in the model regarding mass transport estimates, geochemical processes and ecological parameters are discussed in order to provide perspective on its results.

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PART A

PHASE 1 REPORT

INTERIM CHARACTERIZATION AND EVALUATION

HUDSON RIVER PCB REASSESSMENT RI/FS

CONTENTS

PART A

CONTENTS
(continued)

PART A

CONTENTS
(continued)

PART A

CONTENTS
(continued)

TABLES

Tables Located in Volume 1 (2 of 2)

FIGURES

Figures Located in Volume 1 (2 of 2)

PART A

CONTENTS
(continued)

PLATES

Plates Located in Volume 1 (2 of 2)

PART A

CONTENTS
(continued)

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A. LOWER HUDSON CHARACTERIZATION

A.1 Physical Site Characteristics

A.1.1 Hudson River Basin Characteristics

A.1.1.1 Drainage Areas

The Hudson River Basin is discussed first to establish a framework for this discussion of the Lower Hudson (Part A) and subsequent discussion (see Part B) of the Upper Hudson.

The source of the Hudson River is at Lake Tear-of-the-Clouds, a two-acre pond located on Mount Marcy (Boyle, 1969) in the Adirondack Mountains in northern New York State. From the Adirondack headwaters, the Hudson flows in a southerly direction for approximately 315 river miles to the Battery in New York City (River Mile 0) at the southern tip of Manhattan Island. The Hudson River drainage basin encompasses an area of 13,390 square miles (Plate A.1-1) and has three distinct parts.

- The Upper Hudson River flows from Mount Marcy in the Adirondacks to the Federal Dam at Green Island, Troy, New York. The drainage area of this segment is approximately 4,640 square miles.
- The Mohawk River sub-basin originates in the southern Tug Hill Plateau and flows southeasterly to its confluence with the Hudson River north of Albany, New York. The drainage area of the Mohawk sub-basin is approximately 3,465 square miles.
- The Lower Hudson River flows 153.4 miles from the Federal Dam at Troy to the Battery. The drainage area of the Lower Hudson basin is approximately 5,285 square miles. This segment of the Hudson is tidal to the Federal Dam.

A.1.1.2 Climate

Because the Hudson River Basin occupies a substantial portion of New York State, nearly all of the state's climatic conditions occur within the basin. The climate of New York State is subject to air masses originating from three principal areas. Masses of cold, dry air frequently arrive from the northern interior of the continent. Prevailing winds from the south and southwest transport warm, humid air, modified by the Gulf of Mexico and adjacent subtropical waters. The third great air mass flows inland from the North Atlantic Ocean and produces cool, cloudy and damp weather. This maritime influence, important to the southernmost portion of the Hudson Basin, is secondary to that of the more prevalent air mass flow from the continent.

Nearly all storm and frontal systems moving eastward across the continent pass through or in close proximity to New York State. Storm systems often move northward along the Atlantic coast and influence weather and climate of the Lower Hudson Basin and Long Island; such systems can also influence weather conditions in the more northern portions of the Hudson Valley.

Precipitation is variable and is influenced by topography and proximity to ocean and lake sources of moisture. Nevertheless, precipitation is quite uniformly distributed throughout the year within the basin as a whole, with the least amount generally occurring in the winter and the greatest amount occurring during the warm season. The annual precipitation throughout the Hudson River Basin varies from about 35 inches in the Albany area to more than 55 inches in the higher elevations of the Catskill and Adirondack Mountains. The Lower Hudson maximum average annual precipitation is approximately 46 inches.

Precipitation records collected at New York City (Central Park) since 1826 and at Albany since 1890 (Figure A.1-1) show that for the period 1826-1985, the average annual precipitation at New York City was 42.46 inches and for 1890-1985, the mean annual precipitation at Albany was 34.24 inches. There is wide variability in annual precipitation for the two areas as shown in this figure.

A five-year moving average for both New York and Albany data, also plotted on the figure, smooths out the year to year variability. From this plot, the correlation between trends of high and low precipitation for both New York and Albany (especially in the period 1950-1985) suggest a cyclic, rather than random, pattern to annual precipitation.

In a typical year, the highest streamflow throughout the basin occurs in the spring as a result of snowmelt and precipitation. Low streamflow generally occurs in the late summer, when evapo-transpiration effects are the greatest. Floods, however, can occur during any time of the year. They are generally the result of snowmelt/precipitation during the winter and spring season, hurricanes during the June to October period, or thunderstorms during the summer.

Floods resulting from snowmelt/precipitation or hurricanes generally affect larger areas and the larger streams. Thunderstorms with rains of high intensity over a small area produce the maximum discharge for a variety of smaller streams.

The Upper Hudson and Mohawk sub-basins contain many lakes and swamps, which significantly influence flood flow. In contrast to the Upper Hudson, floods in the Lower Hudson are not significantly affected by storage, but are related to the slope of the main channel.

Evaporative losses for the Lower Hudson River were calculated by Garvey (1990) to be about 570 cubic feet per second (cfs) or 16 m^3/s during July to 220 cfs (6.3 m^3/s) during October. Water loss due to evaporation was most significant in the summer, at about three percent of the mean annual freshwater flow. In all other seasons, evaporative loss was closer to one percent of the total flow.

A.1.2 Hydrology

A.1.2.1 Physical Characteristics

Since the main channel of the Lower Hudson runs fairly straight along a north/south axis, it permits a rather precise definition of specific river locations along a north-south axis. Locations in the river are usually specified as river miles (RM). For this study, the Lower Hudson is defined as that portion of the Hudson River from the Federal Dam at Troy (River Mile 153.4) to the Battery (River Mile 0) at the southern tip of Manhattan. River miles south of the Battery are denoted with a negative (-) value.

In contrast to the rather steep gradient of 5.0 meters (m)/mile north of Fort Edward and more moderate gradient (1.0 m/mile) south of Fort Edward to the Federal Dam at Troy, the Lower Hudson is considered to be a drowned river valley with a gradient of only 0.01 m/mile. With the exception of the Tappan Zee, Haverstraw and Newburgh Bays, the Lower Hudson has a narrow geometry of less than 0.9 miles in width (Moran and Limburg, 1986).

The navigational channel is 32-feet deep from The Battery to Albany and 14 feet deep from Albany to Troy. Although the Lower Hudson has an average depth of about 27 feet, Stedfast (1980) records a maximum depth of more than 200 feet in the vicinity of West Point as the river cuts through the Hudson Highlands. The total surface area of the Lower Hudson is about 129 square miles (Hammond, 1975). The total volume of water is approximately 0.74 cubic miles (Hammond, 1975) with the greatest volume recorded within the Haverstraw Bay region from River Mile 25-40 (Texas Instruments, 1977).

The Lower Hudson experiences two tidal cycles daily. The tidal range is about 4.5 feet at the mouth, 2.7 feet at West Point and about 4.7 feet at the head of tide at Troy. The increase in tidal range in the upper reaches of the Lower Hudson results from the constrictional effects of the diminishing cross-sectional area of the river.

A.1.2.2 Freshwater Flow and Tributary Inputs

The mean annual freshwater flow of 19,500 cfs (550 m^3/s) is fairly large in comparison to other rivers located in the northeastern United States. The freshwater flow, however, is still small in comparison to the daily tidal movements. According to the USGS, the maximum tidal current discharge at Poughkeepsie has been as high as 240,000 cfs (6900 m^3/s), which is more than an order of magnitude greater than the annual mean freshwater discharge past the Battery.

The runoff pattern for the Lower Hudson drainage basin normally contains a large seasonal signal in spite of the relatively constant precipitation rate throughout the year (USGS, 1986). This seasonal signal is a result of melting winter snow in early spring, producing a major increase in freshwater discharge. Flow at the Federal Dam can be characterized as having two basic regimes, a low steady flow of about 5300 to 7100 cfs (150 to 200 m^3/s) for nine months of the year and a large spring surge from March to May resulting from the melting of winter snow. Figure A.1-2 (from Hammond, 1975) illustrates flow over the Federal Dam at Troy for water year 1962. Superimposed on this flow is the seasonal signal from the lower tributaries to the estuary.

The relationship between the flow of a representative Lower Hudson tributary and the flow at the Federal Dam at Troy is illustrated in Figure A.1-3a. For water year 1986, the spring thaw in the lower basin, as indicated by the Wallkill River flow, is evident about one month before the thaw in the Upper Hudson. Exceptions to this pattern occur, as shown in Figure A.1-3b for water year 1984. Water year 1984 had high runoff rates occurring several times throughout the year; the flow rate patterns of the upper and lower basin were similar because of large storm systems and warmer temperatures, affecting the upper and lower parts of the drainage basin concurrently.

An interesting feature of the Lower Hudson River is the flow contributions of tributaries along the river axis (Figure A.1-4a and Table A.1-1). On average, approximately 50 to 70 percent of the freshwater that enters the Lower Hudson flows over the Federal Dam at Green Island. As a consequence of the topography of the Lower Hudson drainage basin, more than 75 percent of all additional tributary inputs, other than sewage, occur north of Poughkeepsie at River Mile 75 (Figure A.1-4b).

A.1.2.3 Circulation

The basic features of circulation within the Lower Hudson are well described as a quasi-two-layer system (Stommel, 1953; Pritchard, 1955, 1969). Seaward (and southward) flow of the surface layer is driven by the gradient in river surface height from the northern end of the Lower Hudson at Troy to The Narrows, located south of The Battery. This gradient is dependent upon the total freshwater flow. The net movement of the lower layer is upstream, northward from The Narrows, in response to the density gradient between the freshwater supply to the north and seawater, supplied to the Lower Hudson largely through The Narrows. The net result of these flows superimposed on the tidal surges is to mix the fresh and salt waters, creating a salt distribution intermediate between the vertical isohalines of a well-mixed system like the Thames of Great Britain and the nearly flat isohalines of a salt wedge estuary like the Mississippi delta (Deck, 1981). (Isohaline is defined as a line connecting points of equal salinity.)

Because of the narrow geometry of the Lower Hudson, the location of the northern edge of the saline intrusion (sometimes referred to as the salt front) becomes a sensitive indicator of the balance between the freshwater and seawater flows (Deck, 1981; Prandle, 1981). Typically the saline intrusion extends to just above the Hudson Highlands, around River Mile 55. During major spring runoff events, the salt front can be found below the George Washington Bridge at River Mile 12. During times of extreme drought, the salt front has been located as far north as River Mile 75, near Poughkeepsie.

A.1.3 Water Quality

A.1.3.1 Overview

Water quality parameters, such as dissolved oxygen, nutrients, turbidity, toxic chemicals, heavy metals and pathogens, affect aquatic resources as well as a variety of recreational activities in the Lower Hudson. The maintenance of acceptable water quality levels is important to the continued viability of the Lower Hudson River ecosystem.

Although overall water quality in the Hudson River has improved in recent years as a result of the construction of new sewage treatment facilities and the upgrading of older facilities, there are still many segments of the Lower Hudson with water quality problems. For example, a variety of heavy metals, including copper, lead, mercury, zinc, chromium and cadmium, have been found in the water column of the heavily industrialized New York metropolitan region (NYSDEC, 1990). In addition, concentrations of PCBs, cadmium, TCDD (dioxin) and TCDF (furan) in fish and shellfish may exceed levels considered safe for human consumption and are of concern, because they can be transferred throughout the aquatic food chain (NYSDEC, 1990).

The following excerpts illustrate the present general condition of the water quality within the Lower Hudson (NYSDEC, 1990):

> "Water quality in the River has improved steadily in recent years, especially in the Albany 'pool' due to the completion of high level secondary treatment plants serving the Albany County and Rensselaer County Sewer Districts. Surveys and monitoring in this area have shown relatively good water quality with respect to conventional pollutants."
>
> "Water quality in the mid-Hudson area is best, and it deteriorates in the last twenty miles above New York Harbor due to the huge population concentrations and resultant nonpoint, storm and wastewater discharges on both the New York and New Jersey sides of the river. A combined sewer

overflow study done for the City of Yonkers indicates that water quality standards violations occur in the vicinity of Irvington. Construction is currently underway to provide treatment of these discharges."

Specific water quality parameters are briefly discussed below. PCBs in the Lower Hudson are addressed at A.2 and A.3.

A.1.3.2 Salinity

It is generally recognized that the Lower Hudson River can be described as having four salinity zones: (1) limnetic or freshwater zone of less than 0.3 parts per thousand (< 0.3 ppt); (2) oligohaline from 0.3 to 5 ppt; (3) mesohaline from 5 to 18 ppt and (4) polyhaline from 18-30 ppt (Cooper *et al.*, 1988). Although the location of each of these zones varies depending on the magnitude of tidal and freshwater flow, a tidal freshwater region typically occurs above River Mile 50-55; an oligohaline zone extends from River Mile 25 to 50; and a mesohaline to polyhaline zone occurs below River Mile 25. Usually the polyhaline zone is limited to the extreme lower reaches of the Lower Hudson in the vicinity of Manhattan. During periods of pronounced drought, however, the polyhaline zone may extend beyond Manhattan (Ristich *et al.*, 1977).

A.1.3.3 Temperature

Mean monthly temperatures recorded by the USGS at Green Island indicate that freshwater entering the estuary ranged from a low of 0°C in January to a high of 29°C in July for the period 1970-1981 (Moran and Limburg, 1986; Cooper *et al.*, 1988). Lower Hudson water temperature patterns are influenced by freshwater discharge and ocean waters. For example, in the summer, ocean water that enters the Lower Hudson is considerably cooler than freshwater. This situation may result in temperature differences of as much as 11°C in the upper and lower reaches of the Lower Hudson (Abood *et al.*, 1976; Garvey, 1990). The average annual temperature of water in the Lower Hudson is 12.3° C (NOAA, 1982).

A.1.3.4 Dissolved Oxygen

Typical dissolved oxygen levels throughout the Lower Hudson are between 5 and 14 mg/l, depending on spatial and temporal constraints (Cooper *et al.*, 1988). Although there is considerable variability with season, the highest levels occur in the late winter to early spring (Moran and Limburg, 1986). Dissolved oxygen levels are generally undersaturated throughout much of the Lower Hudson during the summer. Supersaturated conditions may, however, occur in some shallow bays as a consequence of algae blooms (Cooper *et al.*, 1988). A survey by Garvey (1990) during the fall of 1985 indicates that percent dissolved oxygen saturation values decline from 105 percent at River Mile 153 (supersaturation resulting from photosynthesis) to 71.4 percent at River Mile 6.5. Increased sewage discharges from the New York City metropolitan region result in increased biological oxygen demand (BOD) and reduce the dissolved oxygen levels throughout much of the saline portion of the Lower Hudson compared to the more freshwater regions (Garvey, 1990).

A.1.3.5 Turbidity and pH

The main influence on turbidity in the Hudson River is the presence of silt/clay particles that are transported either by marine sources or by terrestrial runoff. Additional silt/clay particles may be resuspended within the Lower Hudson by erosion or scouring. Turbidity in the Lower Hudson is generally higher during periods of greatest discharge (Cooper *et al.*, 1988).

Increased turbidity generally decreases light transparency within the water column, which, in turn, limits the extent of the photic (light) zone. Investigations in water bodies such as the Hudson and Delaware River estuaries have demonstrated that light transparency is approximately one meter plus or minus 0.5 meters, depending upon the season (Cantelmo and Wahtola, 1989). Results of interpier light transmission studies along the west side of Manhattan and the Westside Highway Project Study (NJMSC, 1984; EEA, 1988) indicate that the photic zone is generally confined to the upper meter of the water column. These

results are typical for near shore, relatively turbid estuarine waters. This situation contributes to a large vertical attenuation of light and limits the photic zone to the upper 1-1.5 meters.

In the well-buffered Lower Hudson, very little spatial or temporal change in pH values occurs. Although historical records reviewed by Cooper *et al.* (1988) indicate that the pH in the Lower Hudson may vary between 6.4 and 8.2, most measurements were above 7.0 (Moran and Limburg, 1986). Recent data collected by Garvey (1990) throughout the entire Lower Hudson also confirms that the system is well-buffered with slight pH variations of 7.6 to 7.8.

A.1.3.6 Municipal Wastewater Discharges

There has been a steady increase in municipal wastewater discharges since 1952, but daily BOD has declined as a consequence of the construction of new sewage treatment facilities (Hetling, 1976; Moran and Limburg, 1986). Recent BOD loading data are not available for the entire Lower Hudson. Hetling (1976) and Moran and Limburg (1986) have calculated 1975 BOD loadings of 55.5 metric tons/day and 131.5 metric tons/day for River Miles 14-152 and River Miles 0-14, respectively.

A.1.3.7 Phosphates and Nitrates

It is well established that sewage is the major source of both phosphates and nitrates to the Lower Hudson (Deck, 1981; Moran and Limburg, 1986; Cooper *et al.*, 1988; Garvey, 1990). Biological uptake of nutrients, such as nitrates and phosphates by phytoplankton, has been shown to be insignificant compared to the total amount of nutrients available within the Lower Hudson (Deck, 1981; Moran and Limburg, 1986).

Concentrations of ortho-phosphate above the salt front are generally less than 95 µg/l and range between 190-620 µg/l below the salt front (Garvey, 1990). The large increase in ortho-phosphate below the salt front is a result of the

large inputs of sewage, which accounts for 73 percent of the total phosphorus sources to the Lower Hudson (Deck, 1981). Concentrations of phosphates, generally the highest during low freshwater flow conditions, typically occur during the late summer (Moran and Limburg; Cooper *et al.*, 1986).

The predominant form of nitrogen in the Lower Hudson is nitrate (Deck, 1981). The only exception occurs below River Mile 18, where ammonia from sewage is the dominant form of nitrogen (Deck and Bopp, 1984). Nitrate levels as nitrogen are typically the highest (560 µg/l) just south of the salt front and decline in the more saline reaches. The largest source of nitrates to the freshwater reaches of the Lower Hudson enters the system from the Upper Hudson (Moran and Limburg, 1986). The nitrates in the more saline portion of the Lower Hudson are governed generally by ammonia additions from sewage and urban runoff (Deck, 1981).

A.1.3.8 Classification and Use

The Hudson River, like other surface waters in New York State, is classified according to the intended "best use". The classification scheme for the Lower Hudson, illustrated in Plate A.1-2, takes into consideration river flow, water quality, condition of adjacent shorelines, and historic, present and future uses. Drinking water is classified as A; swimming as either B or SB; fish propagation and fishing as C; and secondary contact recreation (fishing and boating) as I.

The Hudson River is used as a source for public water supplies in sections of the river classified as A. There are nine Lower Hudson facilities that draw Hudson River water directly for consumption, including five communities and four institutions, camps or schools (Table A.1-2). In addition, a water intake located at Chelsea (River Mile 66) north of Beacon is used by New York City as an emergency water supply during severe periods of drought (NUS, 1984).

Hudson River water is also used for industrial and commercial purposes such as cooling, manufacturing processes, fire protection and hydroelectric and thermal power generation. An inventory of such facilities and plants can be found in reports for the Hudson River-Black River Regulating District (Malcolm Pirnie, 1984) and for the NYSDOT (December 1984).

A.1.4 Aquatic Resources in the Lower Hudson

A.1.4.1 Conceptual Framework

There have been many attempts to conceptualize the structure and function of stream and/or river ecosystems, including those by Hynes (1970), Cummins (1974), Whitton (1975), McIntire and Colby (1978), Cummins and Klug (1979), Moran and Limburg (1986) and Gladden *et al.* (1988). Collectively, these studies have generally emphasized four major categories of organic resources:

Primary Producers -- phytoplankton, periphyton and macrophytes;

Detritus -- particulate organic matter and associated microbial biomass;

Dissolved organic matter; and

Consumers -- microzooplankton, macrozooplankton, benthic invertebrates and fish.

Primary producers include the phytoplankton, periphyton and macrophytes. Phytoplankton are varied microscopic floating plants that have no power of locomotion and are spatially dispersed by river currents; periphyton are those algae attached to various substrates, including rocks (epilithic), silt (epipelic) or other plant species (epiphytic); and macrophytes are macroscopic forms of aquatic vegetation generally limited to shallow (<10-15 feet) water.

Detritus is any non-living particulate organic matter derived from the production of living populations of plants and animals. The primary source of detritus may vary, depending on the nature of the aquatic habitat. For example, in large bodies of water, most of the detritus is derived from aquatic plants; in small forest-covered streams, the terrestrial contribution from the surrounding watershed is dominant.

Dissolved organic matter is excreted by consumers, released from cells during feeding, released due to microbial transformations or released by primary producers. It includes amino acids, glucose, dissolved organic phosphorus and dissolved organic nitrogen.

Consumers include zooplankton, benthic invertebrates and fish. Additionally, invertebrates in the Hudson River can be categorized into major groups, depending on habitat preference and size. For example, the relatively small zooplankton, *i.e.*, microzooplankton, are in the 50-500 µm size range; the larger zooplankton, *i.e.*, macrozooplankton, are generally in the 500-2,000 µm size category. The benthos or bottom dwelling invertebrates may be further subdivided into the epifauna, those organisms primarily inhabiting the surficial sediments, and the infauna, living predominantly within the sediments. Fish populations in the Lower Hudson River can be broadly characterized as freshwater or euryhaline year-round residents and those that utilize the Hudson during spawning or feeding migrations.

The nutritional value of the four preceding categories of organic resources is determined by a number of factors, including the C/N (carbon to nitrogen) ratio, protein content and percentage of refractory (unavailable) nitrogen. Russell-Hunter (1970) has suggested that the lower limit of nutritional requirements for organic resource consumers is generally 16 percent protein (dry weight) and a C/N ratio of <17. In addition, Cummins (1979) notes that the C/N ratio of organic resources varies considerably and some sources with a C/N ratio <17 may still not be considered "quality" resources. For example, fine detritus (0.05 - 1.00 mm) may have a significant portion of refractory nitrogen, which is

resistant to microbial transformations and/or direct consumer utilization (Cummins and Klug 1979). The quality or nutritional value of food is especially important in temperate lotic freshwater systems, which are more likely to encounter seasonally high C/N ratios during peak spring detrital inputs from the surrounding watershed.

A number of studies have concluded that various autochthonous, primary production inputs (phytoplankton, periphyton and macrophytes) make important contributions to aquatic food webs (see reviews by Wetzel, 1975 and Mann, 1975). In some relatively fast flowing streams (>2.0 m/sec), periphyton (algae attached to various substrates) may be the major contributor to primary production (Mann 1975). As stream velocity decreases, however, phytoplankton and macrophyte production may dominate autotrophic processes (Taylor, 1971 and Wetzel, 1975).

Inputs provided by organic carbon from the surrounding terrestrial watershed, including particulate and dissolved organic matter (allochthonous inputs) are also important food chain constituents for maintenance of trophic structure (Hynes, 1970). Organic carbon sources within upstream reaches of various river systems have been shown to make significant contributions to the lower reaches. For example, the largest source of allochthonous carbon to the Lower Hudson (66,024 metric tons of carbon/yr) originates from the Upper Hudson (Gladden *et al.*, 1988). The following table provides some estimates of the organic carbon inputs to the Lower Hudson (Moran and Limburg, 1986 and Gladden *et al.*, 1988).

Source	Organic Carbon Inputs Metric Tons Carbon/yr	Percent
Phytoplankton	36,364	14.8
Macrophytes	5,364	2.2
Upper Watershed	66,024	26.8
Lower Watershed	43,254	17.6
Sewage	57,649	23.5
Marine	36,898	15.0
Total	245,553	100.0

The amount of allochthonous inputs from upper watershed areas to the lower reaches of many temperate rivers, including the Hudson, varies seasonally with discharge. The greatest discharge in the Upper Hudson occurs during the spring (Cooper *et al.*, 1988) and corresponds to the highest seasonal input of allochthonous carbon to the Lower Hudson (Gladden, per. comm.).

Although the relative importance of autochthonous versus allochthonous carbon sources has been debated in the scientific literature (Stephens 1967; Gladden *et al.*, 1988), this relationship may ultimately depend on the specific nature of the system being investigated. For example, Fisher and Likens (1973) showed that 99 percent of the energy input to a first order stream in New Hampshire is from the allochthonous inputs. In a classic study of energy flow in Silver Springs, Florida, Odum (1957) found that autochthonous production in the form of freshwater eel grass was predominant. Gladden *et al.* (1988) and Moran and Limburg (1986) calculated that approximately 65 to 83 percent of the total organic carbon in the Lower Hudson came, respectively, from allochthonous sources in the upper watershed and from sewage effluents or from marine sources in the lower watershed. Nevertheless, Gladden *et al.* (1988) speculated that much of the allochthonous carbon may be refractory and of limited or reduced nutritional value to the consumers, compared to autochthonous phytoplankton production.

Studies of small watersheds and larger rivers (Fisher and Likens, 1973; Fisher, 1977; Sedell *et al.*, 1973; Schaffer 1978) indicate that more organic carbon may be in a dissolved rather than particulate form. In a review of the Lower Hudson River ecosystem, Gladden *et al.* (1988) remark that dissolved forms of carbon must first be transformed into microbial biomass before they are incorporated by various consumers, including zooplankton and benthic invertebrates. Since there is an additional transformation step, less net energy may be transferred to the consumers by a dissolved organic carbon route than by a particulate route.

In most systems investigated to date, trophic pathways and the efficiency of organic carbon utilization are not clearly understood, because of the complexity of interactions, resource partitioning and changes in food preferences by many organisms during their life cycle. Particulate organic compounds (allochthonous and autochthonous) grazed by organisms in the water column or incorporated into the sediments may be utilized by zooplankton and benthic food resources (Hynes 1970; Gladden *et al.*, 1988). In many aquatic systems temporal and spatial dispersal patterns, generation times and population fluctuations of these lower trophic groups may, thus, exert a pronounced influence on the foraging success of juveniles and adult fish populations. As such, the invertebrates are important trophic links in the food web between organic carbon sources and fish populations.

A.1.4.2 Physical Constraints

Many studies of lotic systems have established that a number of physical/chemical parameters (dissolved oxygen, temperature, nature of the substrate, *etc.*) depend primarily on various hydrological features of the river basin. Energy transfers between trophic levels and biotic spatial and temporal patterns are inextricably linked to a variety of hydrodynamic factors responsible for shaping and maintaining the stream channel. Of all the hydrodynamic factors, stream current (velocity) is the most important physical factor regulating river biota. Many freshwater river studies have acknowledged that longitudinal distribution as well as species composition of primary producers and consumers are greatly influenced by velocity (for review see Whitton, 1975).

Velocity is determined by a number of elements, including the size, shape and roughness of the channel, load of suspended sediments and gradient. The gradient in the Upper Hudson from Fort Edward to the Federal Dam is approximately 1 m/mile (Sanders, 1982), which produces a slow to moderate velocity from 0.3 to 1.3 m/s (Simpson, 1974). Although the gradient in the Lower Hudson is 0.01 m/mile (Helsinger and Fiedman, 1982), the velocity is driven by tidal influences.

The maximum velocity in the Lower Hudson is estimated to range from 0.5 to 1.0 m/s (Garvey, per. comm.).

In large measure, stream velocity determines what type of bottom substrate may be present. For example, erosion of sand and gravel beds generally occurs at velocities >1.7 m/s, deposition of sand occurs at velocities of 0.3 to 1.2 m/s and silt particles normally settle out at velocities <0.3 m/s (Terrell and Perfetti, 1989). Both the physical nature (average grain size, silt/clay fraction, range of particle sizes, *etc.*) and chemical nature (rate of exchange of compounds and gases across the sediment/water interface, vertical gradients of Eh, interstitial oxygen, *etc.*) of the stream bed are major constituents controlling the distribution of many benthic organisms.

A.1.4.3 Trophic Components in the Lower Hudson

Primary Producers

It is well documented that phytoplankton are the dominant primary producers within the Lower Hudson (Moran and Limburg 1986; Gladden *et al.*, 1988). Although other groups of primary producers, such as the macrophytes, are believed to represent additional sources in estuarine ecosystems (Gladden *et al.*, 1988), productivity values are not generally available for the Lower Hudson.

Estimates of phytoplankton gross productivity range between 100-250 g carbon/m^2/year from River Mile 0 to River Mile 76 (Sirois and Fredrick, 1978). The highest productivity occurred during June and July and decreased dramatically in August (Sirois and Fredrick, 1978; Gladden *et al.*, 1988), as explained later. Analysis of spatial trends in the data of Sirois and Fredrick (1978) indicated that gross primary production during May through October, 1972 was the highest within the Tappan Zee and Croton-Haverstraw regions (approximately, River Miles 28-38) and declined farther upriver and farther downriver (Gladden *et al.*, 1988).

Gladden *et al.* (1988) point out that the reasons for the rather large spatial and temporal variations in phytoplankton productivity may be attributed to the variations in river physical characteristics, flow patterns and light intensity. Hudson River phytoplankton are light rather than nutrient limited (Heffner, 1973; O'Reilly *et al.*, 1976; Storm and Heffner, 1976). In addition, phytoplankton may tend to increase and concentrate in the low-flushing, shallow regions of the Lower Hudson, such as the vicinity of River Miles 28-38.

Weinstein (1977) and others (Fredrick *et al.*, 1976; Storm and Heffner 1976; McFadden *et al.*, 1978; Moran and Limburg, 1986) have indicated that phytoplankton species are distributed in the Lower Hudson along spatial and temporal gradients. For example, more marine phytoplankton dominate the lower reaches (River Mile <25) of the Hudson River, while more brackish water and freshwater species dominate the middle (River Mile 25-50) to upper reaches (River Mile >50), respectively. During the warmer summer months, phytoplankton species are dominated by the Chlorophyta (green algae) and Cyanophyta (blue-green algae), whereas from the late fall through spring, Bacillariophyceae (diatoms) are often the most numerous. Typical dominant phytoplankton found throughout the Lower Hudson (Moran and Limburg, 1986) are listed below.

Chlorophyta (Green algae)
- Chlorophyceae
  - *Pediastrum*
  - *Scenedesmus*
  - *Ankistrodesmus*

Chrysophyta
- Bacillariophyceae (Diatoms)
  - *Asterionella*
  - *Melosira*
  - *Cyclotella*
- Chrysophyceae (golden or yellow-brown algae)
  - *Chrysococcus*

Cyanophyta (Blue-green algae)
- *Anabaena*
- *Anacystis*

Pyrrhophyta (Dinoflagellates)
- *Ceratium*
- *Porocentrum*

In addition to the contribution of phytoplankton to primary production, periphyton and aquatic macrophytes may also represent a primary food source for a variety of consumers. There have been no studies of periphyton in the Lower Hudson, but a number of studies have described Hudson River macrophytes.

The following list of common macrophyte genera in the Lower Hudson has been compiled from a variety of sources (Kiviat, 1973; McFadden *et al.*, 1978; Moran and Limburg, 1986).

Genera	Common Name
Anacharis (Elodea)	Waterweed
Carex	Sedge
Cyperus	Sedge
Eleocharis	Spike Rush
Heteranthera	Water Star-Grass
*Lythrum**	Purple Loosestrife
*Myriophyllum**	Milfoil
Najas	Naid
Nuphar	Spatterdock
Peltandra	Arrow-arum
Phragmites	Common Reed
Pontedaria	Pickerelweed
Potamogeton	Pondweed
Sagittaria	Arrowhead
Scripus	Bulrush
Sparganium	Bur Reed
Spartina	Cord Grass
*Trapa**	Water-chestnut
Typha	Cattail
Vallisneria	Water Celery
Zizania	Wild Rice

* Introduced species of European or Asian origin.

The salinity regime mediates spatial patterns of macrophytes. For example, salt-tolerant plants such as *Spartina* are common in the lower and middle reaches of the Hudson River, but are replaced by *Peltandra, Nuphar, Typha, Vallisneria, Myriophyllum and Potamogeton* in the freshwater marshes in the upper reaches (McFadden *et al.*, 1978; Moran and Limburg, 1986). At times, some introduced

species such as water-chestnut and purple loosestrife form extensive monocultures within the Lower Hudson and displace a variety of more desirable native plants such as cattails, sedges and bulrushes (Moran and Limburg, 1986; Stalter, per. comm., 1991).

Since productivity estimates for Lower Hudson River macrophytes are not available, it is not possible to ascertain the exact contribution of macrophytes to the total organic carbon sources. Moran and Limburg (1986) estimate that the 7.7 square miles of marshlands in the Lower Hudson should produce about 5,364 metric tons of organic carbon annually. Given these estimates, the net contribution of macrophytes is less than three percent of total organic carbon sources.

Consumers (Invertebrates and Fish)

Invertebrates

Invertebrate consumers in the Hudson River estuary are a heterogeneous group of organisms that link estuarine organic carbon sources and fish populations. Patterns of invertebrate temporal and spatial dispersion, generation times and population fluctuations may exert a pronounced influence on foraging success of larvae, juveniles and adult fish populations in the Hudson River estuary (Gladden *et al.*, 1988).

Although the two general categories of invertebrates (zooplankton and benthos) can be further subdivided depending on habitat preference and size, *i.e.*, microzooplankton, macrozooplankton, benthic epifauna and benthic infauna, the following discussion focuses on dominant members of the two general categories in order to develop an overview of how invertebrates are spatially distributed throughout the Lower Hudson.

There have been numerous studies and reviews (Ristich *et al.*, 1977; Weinstein, 1977; Gladden *et al.*, 1988; Moran and Limburg, 1986;) of invertebrates throughout the Lower Hudson. Listed below are some of the dominant taxa of benthic (b) and zooplankton (z) invertebrates found in the Lower Hudson.

Aquatic Insects	
Diptera	[Chironomids (b)]
Rotifera	
Ploima	[*Keratella* (z), *Brachionus* (z)]
Mollusks	
Gastropoda	[*Valvata* (b/z), *Hydrobia* (b), *Nassarius* (b)]
Pelecypoda	[*Congeria* (b), *Mya* (b), *Macoma* (b)]
Crustaceans	
Amphipoda	[*Gammarus* (b), *Leptocheirus* (b)]
Cirripedia	[*Balanus* (b/z)]
Cladocera	[*Bosmina* (z), *Diaphanosoma* (z), *Moina* (z)]
Copepoda	[*Acartia* (z), *Eurytemora* (z), *Temora* (z)]
Decapoda	[*Crangon* (b), *Palaemonetes* (b)]
Isopoda	[*Cyathura* (b), *Edotea* (b)]
Mysidacea	[*Neomysis* (b)]
Annelids	
Oligochaeta	[*Limnodrilus* (b), *Nais* (b)]
Polychaetes	[*Scolecolepides* (b), *Nereis* (b), *Boccardia* (b)]

Zooplankton populations are dominated by copepods, cladocerans and rotifers. Distinct spatial and temporal patterns of zooplankton populations have been documented throughout the Lower Hudson. Typical of many temperate estuaries, zooplankton exhibit large seasonal increases in response to spring inputs of allochthonous carbon and phytoplankton blooms (Heinle, 1974 and Chervin, 1977; Gladden *et al.*, 1988). For example, seasonal zooplankton densities may vary over ten-fold from early June to November at Indian Point (River Mile 22) and Cementon (River Mile 108) as documented by Gladden *et al.* (1988). In addition, Weinstein (1977) has shown that as salinity increases more halophilic copepods such as *Acartia tonsa* may replace less salt-tolerant forms, such as *Eurytemora affinis*.

Unlike the zooplankton populations, benthic invertebrates in the Hudson River estuary exhibit much smaller population variations and have longer generation times than the zooplankton communities (Gladden *et al.*, 1988). Benthic invertebrates also have a more constant food supply, because most species consume a variety of food particle sizes. In addition, organic carbon concentrations are temporally more stable in sediments (Cantelmo, 1978) than in the water column (Schaffer, 1978).

Distinct spatial population patterns are, however, observed. The lower reaches (River Mile <25) support a typical marine assemblage of benthic invertebrates, including marine oligochaetes, polychaetes and Crustacea. The middle reaches (River Mile 25-50) have a mixture of freshwater and marine forms. The upper reaches (River Mile >50) are dominated typically by freshwater insects, snails, oligochaetes and clams. Greater than 75 percent of the total benthic populations in many regions may consist of mysids and amphipods. Halophilic *Neomysis americana* usually replaces more freshwater *Gammarus* species in the lower reaches of the Hudson River estuary (Gladden *et al.*, 1988; NYU Medical Center, 1978; PASNY, 1986).

Simpson *et al.* (1985) found that the freshwater macrobenthic fauna of the main channel of the Hudson River between Glenmont (River Mile 141.1) and New Hamburg (River Mile 67.4) consisted primarily of oligochaete worms, midge larvae, crustaceans, bivalves and gastropods. The most abundant species was the oligochaete worm, *Limnodrilus hoffmeisteri*, representing approximately 54-79 percent of the total individuals collected at all 16 stations sampled. Simpson *et al.* (1985) also found that the freshwater macrofauna in the Lower Hudson were "...more or less typical of a large lowland river" and consisted primarily of oligochaetes, midge larvae, crustaceans and bivalves. The most abundant taxa were the oligochaetes, which represented approximately 54-79 percent of the total macrofauna.

Benthic invertebrate populations in the middle reaches (River Mile 25-50) of the Lower Hudson River estuary are numerically dominated by oligochaetes, polychaetes, molluscs, and harpacticoid copepods. These taxa account for more than 70 percent of the benthos in many regions (Texas Instruments, 1976).

The benthic invertebrates in the lower reaches (< River Mile 25) of the Hudson are numerically dominated by oligochaetes and polychaetes. Although EEA (1988) identified 80 benthic taxa during an aquatic study at the Hudson River Center Site (River Mile 4.0) from April 1986 to March 1988, only two taxa accounted for 73 percent of the total benthic macrofauna. These two taxa (spionid polychaetes and oligochaete worms) were also found to be the most dominant organisms in other studies. For example, in ecological surveys conducted at Liberty State Park (River Mile 0), Harborside (River Mile 1.0) and in the vicinity of the West Side Highway study area (River Miles 1.8, 4.0 and 9.0), the same two taxa were found to dominate the benthic macrofauna community (Texas Instruments, 1976; LMS, 1980; Harborside, 1987; EEA, 1988).

Fish

Numerous studies and reports have been prepared that document the fisheries of the Hudson River estuary. Since the historical fish surveys conducted in 1934 and 1936 (Greeley, 1935, 1937) from Albany to the Tappan Zee Bridge, surveys conducted over the entire length of the estuary have largely concentrated on concerns over power plant impacts. McFadden *et al.* (1978) summarized those studies conducted from 1936-1975 and Barnthouse *et al.* (1988) focused mainly on surveys conducted from 1976-1980. Beebe and Savidge (1988) updated the 1978 summary and included data from historical and recent fisheries reports in order to compile an extensive list of fish fauna within various regions of the Lower Hudson River. Smith and Lake (1990) have recently documented all known Hudson River fish within the entire Hudson River basin and included notes on the distribution of each species and probable geographic origins.

Major recent reviews by Moran and Limburg (1986) and Gladden *et al*. (1988) included fisheries components as part of their evaluation of the Lower Hudson River ecosystem. The following description of the overall fisheries of the Lower Hudson primarily utilizes information synthesized by Moran and Limburg (1986), Gladden *et al*. (1988) and Beebe and Savidge (1988).

The Lower Hudson River supports some 140 species of resident and migrant fish (Beebe and Savidge, 1988). Some 66 native, freshwater residents (Gilbert, 1980) and a variety of introduced freshwater species occur in the freshwater tidal areas with less than three parts per thousand salinity (< 0.3 ppt). The common species are listed below.

- Bluegill
- Brown bullhead
- Common carp
- Eastern silvery minnow
- Emerald shiner
- Golden shiner
- Goldfish
- Largemouth bass
- Pumpkinseed
- Redbreast sunfish
- Spottail shiner
- Tesselated darter
- White catfish
- White sucker
- Yellow perch

Those resident species consistently observed within oligohaline regions (0.3-5 ppt), mesohaline regions (5-18 ppt) or, at times, throughout the entire Lower Hudson include:

- Atlantic silverside
- Fourspine stickleback
- Hogchoker
- Longhorn sculpin
- Mummichog
- Shortnose sturgeon
- Tidewater silverside
- White perch

Many species living throughout the Lower Hudson are considered euryhaline and are adapted to the wide variations in salinity described at A.1.3.2.

The resident species occur mainly in the shore and the channel bottom zones and typically exploit the more stable benthic habitat (Gladden *et al.*, 1988). The adult freshwater residents, such as common carp, golden shiner, yellow perch or pumpkinseed, and euryhaline/marine residents, such as mummichog, hogchoker and longhorn sculpin, are broadly characterized as fish that feed at the lower end of the trophic level, *i.e.* omnivorous. Regions of peak abundance in the Hudson estuary shift from year to year (Gladden *et al.*, 1988).

The Lower Hudson supports abundant euryhaline and marine populations as suggested by Beebe and Savidge (1988):

> "...extensive marine and brackish-water fish faunas found in the Hudson are a result of the mid-Atlantic location of the estuary and its proximity to the Gulf Stream. Many marine species ride the Gulf Stream into coastal nurseries, including the Hudson river estuary. In addition, several tropical or pelagic species stray into the estuary during the summer (Smith, 1985), and deep-water species occasionally enter the estuary during winter. Individuals of these species have been collected frequently in the lower, more saline, portion of the estuary. .... Euryhaline species may be found throughout the estuary, often in large numbers in their preferred habitats.

Approximately eight fish species utilize the Lower Hudson River estuary as a migratory pathway for spawning activities (diadromous species), including:

Alewife	Atlantic tomcod
American eel	Blueback herring
American shad	Rainbow smelt
Atlantic sturgeon	Striped bass

Of these eight adult diadromous species, seven complete a part of their life cycle in the ocean and spawn in freshwater (anadromous). The American eel, the only catadromous species in the Lower Hudson, spawns in the Sargasso Sea; juveniles (elvers) enter the Lower Hudson and move upstream into the oligohaline to freshwater reaches (Boyce Thompson Institute, 1977).

Some migratory fish species that are not diadromous may periodically enter the Lower Hudson during seasonal feeding cycles in order to consume zooplankton or juvenile fish. Others are considered more permanent marine species. The seasonal and permanent marine species (Smith, 1990) include:

Atlantic menhaden	Fourbeard rockling
Bay anchovy	Northern pipefish
Bluefish	Weakfish

Gladden *et al.* (1988) found that most of the adult migrant species can be found in all habitat types but only for short periods of time. The anadromous species as well as many seasonal marine species are primarily carnivorous and feed on zooplankton and other fish species. The timing of the runs of various migrants, although flow and temperature dependent, is closely linked with the abundance of food for the newly hatched larval forms and juveniles of the species (Cushing, 1975).

The numbers of species and their respective abundance vary seasonally. The numbers tend to increase in the spring through mid-summer when they reach a maximum. Abundance then decreases through the fall and reaches a minimum in the winter. Gladden *et al.* (1988) and others (Barnthouse *et al.*, 1988) have examined various regions of the Hudson River estuary and attempted to correlate food preferences and depth (shore versus channel) in order to explain spatial distribution. Trophic, spatial and temporal partitioning by resident and migrant fish populations may reduce competition (Gladden *et al.*, 1988) and increases the likelihood of maintaining a diverse fisheries resource in the Lower Hudson.

The following list of common and scientific names of the dominant or abundant species of migratory and resident species has been compiled from Moran and Limburg (1986); Gladden *et al.* (1988); and Beebe and Savidge (1988).

Alewife	*Alosa pseudoharengus*
American eel	*Anguilla rostrata*
American shad	*Alosa sapidissima*
Atlantic menhaden	*Brevoorita tyrannus*
Atlantic silverside	*Menidia menidia*
Atlantic sturgeon	*Acipenser oxyrhynchus*
Atlantic tomcod	*Microgadus tomcod*
Banded killifish	*Fundulus diaphanus*
Bay anchovy	*Anchova mitchilli*
Black crappie	*Pomoxis nigromaculatus*
Blueback herring	*Alosa aestivalis*
Bluefish	*Pomatomus saltatrix*
Bluegill	*Lepomis macrochirus*
Brown bullhead	*Ictalurus nebulosus*
Common carp	*Cyprinus carpio*
Eastern silvery minnow	*Hybognathus regius*
Emerald shiner	*Notropis cornutus*
Fourbeard rockling	*Enchelyopus cimbrius*
Fourspine stickleback	*Apeltes quadracus*
Golden shiner	*Notemigonus crysoleucas*
Goldfish	*Carassius auratus*
Hogchoker	*Trinectes maculatus*
Largemouth bass	*Micropterus salmoides*
Longhorn sculpin	*Myoxocephalus octodecemspinosus*
Mummichog	*Fundulus heteroclitus*
Northern pipefish	*Syngnathus fuscus*
Pumpkinseed	*Lepomis gibbosus*
Rainbow smelt	*Osmerus mordax*
Redbreast sunfish	*Lepomis auritus*
Shortnose sturgeon	*Acipenser brevirostrum*
Spottail shiner	*Notropis hudsonius*
Striped bass	*Morone saxatilis*
Summer flounder	*Paralichthys dentatus*
Tesselated darter	*Etheostoma olmstedi*
Tidewater silverside	*Menidia peninsulae*
Weakfish	*Cynoscion regalis*
White catfish	*Ictalurus catus*
White perch	*Morone americana*
White sucker	*Catostomus commersoni*
Winter flounder	*Pseudopleuronectes americanus*
Yellow perch	*Perca flavescens*

Of the 140 species of fish in the Hudson (Beebe and Savidge, 1988), the striped bass has received a disproportionate amount of attention, because of its commercial and recreational value. In addition, most power plant impact studies have focused largely on striped bass populations and have seldom included an ecosystem approach to the Lower Hudson. Although the striped bass is an important species, the Lower Hudson is more than just a single species system. It has been shown to contain one of the most diverse fisheries found throughout Atlantic coastal systems (Beebe and Savidge, 1988).

Synthesis

Differences in the life history strategies and characteristics of zooplankton and benthic invertebrates may exert a major influence on the fish populations. The exploitation of invertebrates by Hudson River fish populations is well documented and leads to a pattern of trophic partitioning among the dominant fish species. For example, Gladden *et al.* (1988) has shown that migratory adult fish species predominantly feed on pelagic zooplankton and other fish, whereas, the resident adult fish species predominantly exploit the more stable benthic invertebrate populations. Since the shallow shore and shoal zone habitats generally support the most extensive resident fish populations in the Hudson River (Beebe and Savidge, 1988), the benthos in these regions is closely linked to resident fish populations with low risk/persistence strategies (Gladden, *et al.*, 1988).

In addition, Thurow (1974) has shown that the year class strength for some demersal fish may be linked to the availability of food and the foraging success of larval and juvenile fish. It has been proposed that the increased food demands by larval fish must coincide with increased populations of invertebrates to insure a strong year class (Cushing, 1975). Thus, the variability in fish numerical abundance and spawning success may be, in part, explained in terms of spatial and temporal variability of the invertebrate trophic base. Invertebrate consumer populations, in turn, are linked to the availability of energy supplied by other organic resources, including the primary producers, detritus and dissolved organic matter.

A.2 Sources of PCB Contamination

A.2.1 Description of PCBs

Polychlorinated biphenyls (PCBs) are a class of chlorinated, aromatic hydrocarbons. Each PCB consists of two connected rings of six carbons each (a biphenyl) to which one or more chlorine atoms are attached at any of 10 available sites (Figure A.2-1). Positions on the biphenyl molecule not filled with chlorine atoms have hydrogen atoms in their place. Mono- through deca-chlorinated biphenyls make up a homologous series in which each successive homologue group contains compounds with one more chlorine atom than the preceding group. Within each homologue group, the PCBs containing the same number of chlorine atoms but differing in their structural arrangement on the phenyl rings, are referred to as isomers. For example, 2-chlorobiphenyl and 4-chlorobiphenyl, both monochlorobiphenyls, are isomers of each other, each containing one chlorine atom per biphenyl molecule, yet differing in the position of chlorine substitution. Figure A.2-1 also lists the ten different PCB homologue groups and shows the number of possible isomers within each group. The isomers in all of the homologue groups are generically referred to as congeners. There are a total of 209 theoretically possible PCB congeners.

PCBs are produced commercially by the chlorination of biphenyl using ferric chloride or iodine as a catalyst. This produces a mixture of congeners, which is usually distilled to produce somewhat simpler mixtures with desired properties. The distilled mixtures have been marketed under various trade names including Aroclor (Monsanto, US), Clophen (Bayer, West Germany), Kanechlor (Kanegafuchi, Japan), Phenochlor (Caffaro, Italy), Pyralene (Prodelec, France) and Sovol (U.S.S.R.). In the US, each Aroclor product marketed was given a four digit numerical nomenclature; the first two digits (12) indicate the number of carbon atoms per molecule and the last two digits represent the approximate percent chlorine by weight in the mixture. (Aroclor 1242, for example, is a US-produced polychlorinated biphenyl mixture, containing approximately 42 percent chlorine by weight.) An exception to the nomenclature is Aroclor 1016. Table A.2-1 lists the weight percentages of different congener groups in various

Aroclor mixtures.

In the environment, PCBs have a high affinity for the organic carbon fraction of soil, sediments and suspended matter and a high tendency to accumulate in biota. PCBs (especially the lesser chlorinated congeners) also have a moderate tendency to volatilize from water into the atmosphere. PCB mixtures also have characteristics imparted by their individual PCB congeners, each possessing specific, unique physical and chemical properties and each behaving somewhat differently in the environment. The multiple congener composition of Aroclors is an important consideration in interpreting environmental data.

A.2.2 Lower Hudson PCB Loadings

Total loading of PCBs to the Lower Hudson has been historically dominated by inputs from the Upper Hudson. The TAMS/Gradient analysis of PCB loading from 1977 to 1989 (see B.4) indicates that approximately 15,000 kg of PCBs were released to the Lower Hudson over the Federal Dam at Troy during this period. This amount represents only a portion of the total historic loading of 178,000 kg (Thomann *et al.*, 1989) from the Upper Hudson to the Lower Hudson (see A.4). Based on the sediment measurements obtained by Bopp (1979), Bopp *et al.* (1983) and Bopp and Simpson (1989), the PCB loading from the Upper Hudson has been continuous from the early 1950s at levels comparable to and often greater than those measured over the period 1977 to 1989. As confirmed by sediment core PCB measurements, estimated maximum PCB loading occurred around 1973, the year the Fort Edward Dam was removed. PCB loads for that year were estimated to be on the order of 5,000 kg/yr or five times the annual loading in 1980.

A closer review of the loadings for the period 1977 to 1989 suggest a diminution of PCB releases. For example, 410 kg of PCBs have been estimated to be released over the Federal Dam at Troy in 1986 compared to 210 kg in 1989 (see B.4). Current estimates indicate that a substantial portion of the approximately 85,000 kg of PCBs estimated to remain in the sediments of the Lower Hudson can be attributed to the Upper Hudson loadings (Bopp and Simpson, 1989).

Although historically dominated by the Upper Hudson PCB loadings, total loading of PCBs to the Lower Hudson has also been a function of other sources. These sources have been identified and estimated by various investigators over the last 15 years and include: sewage effluent discharges, tributary inputs below the Federal Dam, combined sewer/storm water outfalls, storm water outfalls, atmospheric deposition and landfill leachates. These additional PCB sources are important to consider, since some may contribute inputs of similar magnitude to those sources currently originating upriver (Bopp and Simpson, 1984).

Precise data describing the exact quantity of PCB loadings from each of the preceding additional sources are generally not available. Most parameters, such as flow rate and PCB concentration, must be estimated in order to predict the total PCB loadings. The lack of data is, in part, a result of the dispersed nature of such sources. Table A.2-2 summarizes the range of estimates for these loads, based upon currently available information.

A.2.3 Sewage Effluent Discharges

Extensive data has been collected on sewage flow to the river by a number of agencies and at least two sets of measurements exist on PCB levels in New York City (NYC) metropolitan area effluent (MacLeod *et al.*, 1981 and Hydroqual, 1991 for NYCDEP).

The MacLeod data set consists of five separate measurements of flow-weighted composite samples of sewage effluent from three major treatment plants and several untreated outfalls. The measured total PCB levels averaged 0.3 and 0.97 µg/l for treated and raw sewage, respectively. Data collected by NYCDEP in 1989 from fourteen sewage treatment plants showed all non-detect levels for PCBs in both sewage influent and effluent. This analysis, on an Aroclor basis, had a detection limit of 0.33 µg/l. Based on the Hydroqual (1991) estimate of NYC metropolitan area sewage flow for 1989 of 1,750 mgd (76.6 m^3/s) and the MacLeod *et al.* (1981) estimates, the 1989 PCB loading would be 4.6 lb/day (2.1 kg/day). In addition, Thomann *et al.* (1989) estimate a somewhat lower sewage effluent concentration than either Hydroqual (1991) or MacLeod *et al.* (1981) and report

a PCB loading for 1980 conditions of 3 lb/day (1.4 kg/day). All of the above estimates would collectively yield a range of PCB loadings of approximately 3 lb/day (1.4 kg/day) to 4.6 lb/day (2.1 kg/day) for sewage effluent discharges.

A.2.4 Tributary Contributions

Estimates of PCB loadings from tributaries to the Lower Hudson can all be characterized as poor. Although flow and suspended matter measurements exist for most major tributaries, there are essentially no measurements of PCB concentrations in the tributary flow. Tributary PCB loadings to the Lower Hudson were estimated by Mueller *et al.* (1982) and Thomann *et al.* (1989), based on literature data and USGS flow and suspended matter measurements. PCB loadings for the Lower Hudson in 1980 estimated by Thomann *et al.* (1989) were 2.3 lb/day (1 kg/day), using a mean tributary PCB concentration of 0.05 µg/l. Based on sediment data collected for the Passaic, Raritan, Hackensack, Elizabeth and Rahway Rivers, Mueller *et al.* (1982) estimated that tributary concentrations were an order of magnitude lower. These estimates would collectively yield a range of PCB loadings of approximately 0.2 lb/day (1 kg/day) to 2.3 lb/day (1 kg/day) for the Lower Hudson tributaries.

A.2.5 Combined Sewer/Storm Water and Storm Water Outfalls

Combined sewer-storm water drainage systems in the NYC metropolitan area have long been a source of pollutants to the Lower Hudson. Overflow occurs after rainfall events and results in the release of diluted, untreated sewage directly to the river. In addition, effluent from storm water collection systems, draining residential and industrial areas, also reaches the Lower Hudson untreated. Estimates of flow via these pathways are based on modeling efforts with relatively little field data. Mueller *et al.* (1982) and Thomann *et al.* (1989) estimate respectively that storm water runoff and combined sewer outfalls contribute about 2 lb/day (1 kg/day) to 3 lb/day (1.4 kg/day).

A.2.6 Atmospheric Deposition

Atmospheric loading to the Lower Hudson can result from direct deposition of particle-bound PCBs, precipitation and gas exchange with the overlying air. Because the Lower Hudson has a substantial concentration of PCBs in the water column relative to the overlying air, gas exchange with the atmosphere results in a net loss of PCBs from the river (Bopp, 1983). Particle deposition and precipitation both can result in net PCB transfer to the river in spite of the gas exchange flux. These processes are considered here to the extent that they act on the river surface itself. PCB loads resulting from particle deposition and precipitation in the remainder of the Lower Hudson basin are included in the estimates of tributary input and storm water runoff.

Like most of the other sources of PCBs, there are few useful measurements of PCB deposition or precipitation levels. Mueller *et al.* (1982) estimate the total input to the Lower Hudson to be 0.2 to 2 lb/day (0.09 to 0.9 kg/day), based on a study of atmospheric PCB loadings in the Great Lakes by Galloway *et al.* (1980). Since the Mueller *et al.* (1982) analysis covers portions of the New York Bight and Long Island Sound as well as the Lower Hudson, those estimates should probably be reduced by a factor of two to four in order to adjust for surface area differences. Thomann *et al.* (1989) also estimated the atmospheric loading to the Lower Hudson and arrived at a figure of 0.5 lb/day (0.2 kg/day), based on a review of Mueller *et al.* (1982) and other data in the literature.

A.2.7 Landfill Leachates

Estimates of leachate loadings of PCBs to the Lower Hudson River from bordering landfills, examined by Mueller *et al.* (1982), are based on a minimal number of measurements and on a simple model of leachate transport. A total of 5,650 acres (23 km^2) of landfill area were included in the estimate of landfill leachate generation. A range of 0 to 1.3 lb/day (0 to 0.57 kg/day) was estimated as the PCB load from these landfills to the Lower Hudson, New York Bight and Long

Island Sound (Mueller *et al.*, 1982). Although the load to the Lower Hudson itself was not developed, it is expected that it would represent about half of the total landfill leachate load.

A.2.8 Other Sources of PCBs

There are five facilities with SPDES permits that may provide additional sources of PCBs, to the Lower Hudson (NYSDEC, March 7, 1991 list of facilities with SPDES permits). Four of these (Carlyle Piermont Corporation, IBM East Fishkill Facility, Norlite Corporation and Columbia Corporation) are currently permitted to discharge PCBs within the Lower Hudson River Basin, but not directly to the Lower Hudson River. The fifth permitted facility, Metro-North Commuter Railroad North Harmon Shops in Westchester County, discharges PCBs directly into the Lower Hudson River. Because estimates of flow are not available, the PCB loading to the Lower Hudson cannot be ascertained. However, according to the SPDES permit, the allowable daily average PCB concentration is 1.0 ppb (mg/l) with a daily maximum of 2.0 ppb.

There may be additional incidental releases of PCBs to the Lower Hudson as a result of accidental spills and illegal dumping activities. The extent and total PCB loading of these releases to the Lower Hudson River remain unknown.

A.3 Nature and Extent of Contamination

A.3.1 Sediments

Intensive studies of PCB geochemistry and transport in the Lower Hudson by investigators from Lamont-Doherty Geological Observatory began in the mid-1970s and have continued to the present day. As a part of those studies, measurements were made of PCB levels in sediments, riverine and estuarine suspended matter and dissolved constituents. In addition, accurate radionuclide measurement techniques were developed, which permitted the accurate dating of core samples taken from recent (post-1950) sediments.

The dating technique uses natural and anthropogenic radionuclides and it is possible to determine the year of deposition of a given layer of sediment within the sediment core (Olsen, 1979; Bopp, 1979; Bopp *et al.*, 1982). Since sediments deposited on the river bottom are derived from the sediments carried by the river, sediments deposited within a given area reflect the nature of the river sediment that has traveled past that area. By collecting a core of undisturbed river sediments and sectioning it into layers, which approximate an annual load of sediment accumulation, it is possible to analyze the annual load of sediments carried by the river and their associated contaminants.

On the basis of dated cores, investigators were able to examine the annual variations in sediment PCB levels and derive water-related PCB transports at locations throughout the Lower Hudson. Figure A.3-1 plots the year of deposition and total PCBs of dated cores at several Hudson River locations. River Mile 188.5 represents a location in the Thompson Island Pool in the Upper Hudson. River Mile 143.4 represents the Lower Hudson sediments in the Albany area. River Miles 88.6 and 91.8 are located near Kingston, New York. River Mile 53.8 is located in a cove near Cornwall-on-Hudson, New York. River Miles -1.65 and -1.7 are in upper New York Bay. River Miles 53.8, -1.65 and -1.7 are all located at or below the salt front and are subject to NYC metropolitan area impacts. Collectively, these plots illustrate the temporal trends of sediment PCB levels

from the mid 1950s to the late 1980s (Bopp, 1979; Bopp *et al.*, 1982; Bopp *et al.*, 1984 and Bopp and Simpson, 1989).

As indicated in Figure A.3-1, there is a gradual increase in the sediment PCB concentration from about 1954 to about 1970. This loading is largely attributed to General Electric PCB releases that escaped capture behind the dams of the Upper Hudson. In 1973, the removal of the Fort Edward Dam followed by major river flood events resulted in a substantial increase in the PCB loading to the Lower Hudson. As indicated by all the sediment cores, maximum PCB deposition occurred throughout the Lower Hudson around 1973. The decrease in sediment PCB levels from 1977 to the most recent measurement is attributed to the reworking, resuspension and gradual dispersion by the river of the sediments released after the Fort Edward Dam was removed. In addition, the discontinued use of PCBs by GE since 1977 may have also contributed to decreased PCB sediment levels.

The Lamont-Doherty studies also demonstrated that the sediments record the characteristics of the PCBs being transported in the Lower Hudson (Bopp *et al.*, 1982; Bopp *et al.*, 1984; Bopp and Simpson, 1989). The analysis of the cores showed that the nature of the PCBs stored in the sediments was not constant throughout the river and that an additional source of highly chlorinated PCBs must be located in the saline region of the Lower Hudson. Bopp and Simpson (1989) conclude that 1986 loadings of PCBs in the NYC metropolitan area sewage are comparable to the 1986 PCB loading from the Upper Hudson.

Figure A.3-2 shows the variation in some of the PCB homologues in two core locations, one above the salt front (River Miles 88.6 and 91.8) and one in upper New York Bay (River Miles -1.65 and -1.7). The cores taken at River Miles 88.6 and 91.8 are located well above the salt front and are free of any influence from NYC metropolitan area PCB loadings. The maximum values for the highly chlorinated homologues in these cores are attributed to the use of more highly chlorinated Aroclor mixtures in the early years of production at the General Electric facility. If the General Electric inputs were the only significant PCB releases to the Hudson River, then the PCB homologue variations with time should

be similar throughout the entire Hudson. When the sediments at River Miles -1.65 and -1.7 are examined, however, it is clear that the homologue variations with time are quite different from those at River Miles 88.6 and 91.8. The downriver cores show maximum values roughly 10 to 15 years later than those collected above the salt front. In addition, the absolute concentrations of these homologues are higher down river. Based on the preceding data, Bopp and Simpson (1989) conclude that an additional source or sources of highly chlorinated PCBs must be located in the lower portion of the Lower Hudson.

Figure A.3-3, an expanded view of the cores at River Miles 88.6 and -1.65, offers additional supporting evidence for the importance of the NYC metropolitan area as a source of PCBs (Bopp and Simpson, 1989). The results for the sediments at River Mile 88.6 show an exponential decrease in the sediment PCB concentration from 1973 to about 1986. The curve appears to be asymptotic to zero with the PCB concentration of annually deposited sediments decreasing by a factor of two every 3.5 years. This finding suggests that the annual loading of PCBs to this part of the river is decreasing at the same rate. The general decrease in sediment PCB concentrations with time is consistent with the decrease in PCB concentrations recorded at the USGS Upper Hudson monitoring stations.

The results for the sediment core at River Mile -1.65 represent sediments accumulating in upper New York Bay, where the influence of both upriver and NYC metropolitan area inputs should be seen. As seen in Figure A.3-3, the PCB trend with time appears to have the same exponential decay rate as the upriver core, but is asymptotic to 0.5 mg/kg instead of zero. As of 1986, it appears that sediments influenced by the NYC metropolitan area inputs were accumulating with higher PCB levels than those found further upstream beyond the influence of the metropolitan region. Based on the absolute concentrations in the sediments at these two coring locations, Bopp and Simpson (1989) also concluded that the NYC metropolitan area related inputs in 1986 were of similar magnitude to those originating upriver. NYCDEP (1987) records of PCB levels indicate that Lower Hudson River stations from the New York-Bronx County Line to the Narrows had an average concentration of 0.488 mg/kg from 1983 to 1987, which is comparable to the Bopp and Simpson (1989) 0.5 mg/kg asymptote.

The PCB sediment studies of Lamont-Doherty also indicate the lack of biodegradation in the sediments of the Lower Hudson. Based on core dating, 30-year old sediment layers show little variation in the patterns of congeners constituting the PCB levels. Thus, sediments still retain patterns that can be readily described by the standard Aroclor mixtures and there is no apparent shift toward the less chlorinated PCBs (Bopp *et al.*, 1982). In addition, Bopp *et al.* (1982) observed increased ratios of some of the more highly chlorinated PCB peaks, a finding that may indicate the presence of Aroclor 1254.

On the basis of their sediment analysis, Bopp (1979) and Bopp and Simpson (1989) estimated the inventory of PCBs in the sediments of the Lower Hudson, including the dredge spoils excavated in recent years. These estimates are based on the combination of dated cores collected throughout the Lower Hudson and the matching PCB analysis. The inventory was estimated by determining the sediment accumulation rates based on the radionuclide records and by measuring the PCB concentrations in those sediments. These calculations yield a PCB burden of 187,000 lb (85,000 kg) in the sediments of the Lower Hudson and an additional 82,000 lb (37,000 kg) in the dredge spoils removed to the New York Bight. Table A.3-1 summarizes the PCB inventory estimates on an area-specific basis. The authors estimate the uncertainty of these estimates to be about a factor of two.

Several studies of the Lower Hudson also included suspended matter PCB measurements. Unlike the sediment studies which provide time-integrated samples, the water column samples are snapshots of instantaneous riverine conditions. Nonetheless, they provide a useful confirmation of the sediment records as well as a means to relate the sediment records to the water column inventory. Although the majority of samples collected for suspended matter PCB analysis were taken for the Upper Hudson (Bopp *et al.*, 1985), some results were obtained around River Mile 3 (Bopp and Simpson, 1984). The results are presented in Table A.3-2 and indicate that water column suspended matter correlates extremely well with the sediment core tops.

Lamont-Doherty investigators also addressed the partitioning of PCBs between suspended matter and water. The partitioning can be described by a constant (K_s) which relates the suspended matter PCB concentration to the dissolved PCB concentration. A K_s value greater than 1,000 generally indicates that the compound in question tends to strongly associate with the suspended matter. The first study consisted of *in situ* measurements. The second study involved a laboratory simulation, determining K_s under a broad range of conditions. On the basis of *in situ* measurements of dissolved and suspended matter PCB concentrations (Bopp, 1979) and laboratory simulations (Warren *et al.*, 1987), the K_s for PCBs was shown to vary with several factors. In general, K_s increased with increasing PCB chlorine content, decreasing dissolved organic carbon concentration and decreasing temperature. Based on these findings, Bopp (1979) and Bopp *et al.* (1985) conclude that total water column PCB concentration varies directly with suspended matter levels in the water column. The range of measured K_s's from Bopp (1979) was 1.5×10^4 to 4×10^5 for dichlorobiphenyls to hexachlorobiphenyls, respectively. Warren *et al.* (1987) found a similar relative range for these PCBs, although the absolute values shifted, depending on conditions.

A.3.2 Water

The USGS monitored total (dissolved and suspended matter) PCBs in the Lower Hudson water column from 1978-81, but they have discontinued their efforts since that period. Five locations on the Lower Hudson were monitored, with an average of five samples taken yearly at each location (Schroeder and Barnes, 1983a). The mean concentrations reported by Schroeder and Barnes are given below.

Water column measurements for the Lower Hudson show consistently lower PCB concentrations than the Waterford (Upper Hudson) values. In particular, the measurements at Castleton, the nearest downstream station to Waterford, can be taken to show the effects of dilution by the Mohawk River, which enters the Hudson below Waterford, and the increased watershed drainage area. Like the Upper Hudson, the PCB levels in the Lower Hudson water column showed a declining concentration trend in time over the monitoring period.

Mean PCB Concentrations From Five Lower Hudson Sites Compared to Upper Hudson Samples at Waterford[a]

Location	Mean PCB Concentration (µg/l)			
	1978	1979	1980	1981
Waterford	0.45	0.36	0.25	0.14
Castleton	0.23	0.20	0.11	0.08
Catskill	0.17	0.17	0.16	0.05
Staatsburg	0.14	0.16	0.19	0.07
Clinton Point	0.15	0.12	0.17	0.08
Highland Falls	--	0.17	0.07	--

[a]From Schroeder and Barnes (1983a) Table 6; Lower Hudson averages based on approximately 5 samples per year.

Schroeder and Barnes (1983a) caution that limited significance should be attached to the mean concentration values summarized above. These mean values are based on very few data points (typically no more than 5) and recent (1981) values are very close to analytical detection limits, such that they are subject to large uncertainty. Current levels of PCBs in the Lower Hudson water column are uncertain as is the relative contribution of upriver versus lower river PCB inputs.

Although few data exist on total water column PCB concentrations in the Lower Hudson after 1981, it is possible to make a rough estimation based on core top PCB levels as determined by Bopp and Simpson (1989). For the purposes of these calculations, it was assumed that the water column suspended matter PCB levels were equal to the PCB levels measured in the core tops on a mass basis, *i.e.*, suspended matter PCB concentration in mg/kg of suspended matter equals sediment PCB concentration in mg/kg of sediment. Sediment cores collected at River Miles 88.6 and -1.7 recorded sediment PCB levels of 0.8 and 1.5 mg/kg, respectively. In addition, Bopp and Simpson (1984) obtained measurements of PCBs in suspended matter River Mile 3 of 1.7 to 1.9 mg/kg in 1984.

Estimates were made of total water column PCB levels using the preceding PCB levels in sediment cores and suspended matter. The water column estimates assumed a partition coefficient of 10^5 (Bopp *et al.*, 1985) and suspended matter of 6 to 40 µg/l from 1984-1986 (Garvey, 1990). For 1984 the water column levels at River Mile 3 were estimated at 0.02 to 0.08 µg/l; at River Mile -1.7 they were estimated at 0.02 to 0.06 ug/l. In 1986, the water column levels at River Mile 88.6 were estimated at 0.01 to 0.04 µg/l. These values are generally consistent with the decreasing trend seen in the water column levels in earlier (1978-1981) USGS Survey data (Schroeder and Barnes, 1983). These estimates are subject to some degree of uncertainty, since the partition coefficient utilized for the more saline locations (River Miles 3 and -1.7) was originally based on data from freshwater systems. To date, no measurements of PCB partition coefficients are available for saline waters.

A.3.3 Fish

A.3.3.1 Overview of Previous Monitoring Programs

The Lower Hudson River historically supported a valuable commercial fishery of striped bass and other species. Although the Lower Hudson is considered to contain one of the most diverse fisheries found throughout the Atlantic Coastal systems (Beebe and Savidge, 1988), the striped bass fishery has been closed since 1976 because of elevated PCB levels. Since the early 1970s, NYSDEC has extensively monitored PCBs in fish in the Lower Hudson. Most of the NYSDEC studies of PCBs placed a major emphasis on the striped bass fishery as a consequence of the well documented commercial and recreational value of this resource.

The first report of PCB contamination in striped bass in the Lower Hudson was published in 1970 (Boyle, 1970). In 1971 NYSDEC added PCBs to their statewide analyses for pesticide residues in fish, including the analysis of the 1970 samples. Fish data collected and analyzed for PCBs in the 1970-74 period are summarized in Spagnoli and Skinner (1977).

Following the 1974 USEPA investigation of PCB contamination in the Fort Edward area (Nadeau and Davis, 1974), NYSDEC modified and intensified its PCB sampling program in Hudson River fish. Results of a total of 440 Hudson River fish samples analyzed in 1975-6 have been provided by NYSDEC and were included in the TAMS/Gradient computerized database.

Sampling and analytical procedures for the 1975-1976 collections are documented in Spagnoli and Skinner (1977). The 1975-76 fish collections were made by regional NYSDEC Fish and Wildlife personnel who were instructed on specific species and sizes of fish desired, location of stations and timetables for collection. Target species for the Lower Hudson included smallmouth bass (*Micropterus dolomieni*), largemouth bass (*Micropterus salmoides*), brown bullhead (*Ictalurus nebulosus*), goldfish (*Carassius auratus*), white sucker (*Catastomus commersoni*), striped bass (*Morone saxatilis*) and various other Lower Hudson River species. Other species were occasionally obtained as available. Attempts were made to sample small, medium (minimum legal) and large representatives of each species.

Analyses conducted by several different state laboratories were reported against standards for Aroclor 1242 or 1016 and Aroclor 1254. Aroclor 1221 was not analyzed. The nominal detection limit of the method was 0.01 ppm, although some of the labs reported results only as low as 0.1 ppm.

Following the 1975-1976 sampling effort, the commercial striped bass fishery in the Hudson was closed in February 1976 as a consequence of elevated PCB levels. As a condition of the 1975 Settlement Agreement signed by GE and NYSDEC, GE provided funds for continued PCB monitoring in fish collected annually at various locations below the discharge. A fish sampling program, redesigned in part to address the need for reassessment of closure of the commercial fishery, was implemented in Spring 1977.

In 1980, NYSDEC made a commitment to commercial fishermen to review the annual PCB data prior to mid-November of each year. This date was established to provide an adequate amount of time to prepare any necessary changes in the

regulations regarding the closure of the commercial fishery and to provide commercial fishermen with the opportunity to purchase new nets in the event that the fishery were reopened (Horn and Sloan, 1984). This mandate has resulted in periodic reports on the status of the striped bass fishery (Horn and Sloan, 1984; Sloan and Horn, 1986; Sloan *et al.*, 1986; Sloan *et al.*, 1987; Sloan *et al.*, 1988).

Methodology for the sampling effort since 1977 is described in Armstrong and Sloan (1981), Sloan and Horn (1986) and Sloan *et al.* (1988). Collection of resident species was as close as possible to the same time each year in order to minimize the effects of seasonal variation on PCB testing. Striped bass were typically collected during the spring migration (April, May and sometimes as late as June). Distinct summer and fall collections were made in several years to investigate seasonal variation. Because striped bass were caught during spring migration, the location at which they were caught probably bears little or no relationship to the PCBs in sediment and water at that location.

Aroclor concentrations were determined by comparison to standards for Aroclor 1221, Aroclor 1016 and Aroclor 1254 obtained from the Monsanto Corporation. The method utilized was not able to distinguish between Aroclors 1242 and 1016. The Aroclor detection limit for each Aroclor tested was 0.1 ppm wet weight through 1986; the detection limit for Aroclor 1221, which was never a large component of the total PCBs detected in striped bass, was changed to 0.05 ppm in 1987. For each sample, the percent lipid was determined as the percent by weight of tissue soluble in petroleum ether.

Since 1978, striped bass have been the species most commonly monitored, although a number of other species have been sampled as well. Table A.3-3 provides a summary of the major fish species sampled by year and the type of fish tissue analyzed as included in the NYSDEC database. Table A.3-4 presents the overall average lipid content of fish samples from the Lower Hudson. The most recent fish samples for which PCB results are available were collected in 1988. Fish samples collected and prepared for analyses in 1990 have been shipped recently to the analytical laboratory for PCB analyses (R. Sloan, pers. comm.).

A.3.3.2 Striped Bass

Time series trends of total PCBs, Aroclor 1016, and Aroclor 1254 on a ppm wet weight basis in the spring-collected striped bass from the Lower Hudson show a large decline from 1978 to 1979. Using all striped bass data below River Mile 80, no significant change (at the 95 percent confidence level) in arithmetic mean PCB levels was found for the period 1979 to 1987. A statistically significant decline was observed, however, in the geometric mean PCB concentrations (Sloan *et al.*, 1988). Geometric means since 1983 for striped bass showed a consistent decrease in both total PCBs and Aroclor 1254, but levels of Aroclor 1016 were found to remain relatively constant between 1983 and 1987. Sloan *et al.* conclude that "a general, significant decline of total PCB occurred from 1978 through 1981. In 1982 and 1983, concentration increased. Since 1983, levels have shown a slow but apparently steady reduction with the 1987 values ... similar to those observed in 1981." Lower PCB levels were also noted in samples which were taken from locations lower in the river.

Table A.3-5 summarizes data for striped bass from the Lower Hudson, obtained from the Lower Estuary (River Mile 12 to 76) and the Upper Estuary (River Mile 91 to 153). (The 1977 data for one sample only are omitted from this analysis.) The results for River Mile 27 (the Tappan Zee Bridge) and 153 (Federal Dam) are each shown separately in this table. The data for both locations are also included in the overall lower and upper estuary summaries in the same table. In the years 1986-1988, the median PCB concentrations in striped bass from the lower estuary have ranged from 2.4 - 3.0 ppm, whereas the median PCB levels in striped bass from the upper estuary have ranged from 4.8 - 11.2 ppm during this period.

Because of the tendency of PCBs to concentrate in lipids, some studies have analyzed data that have been based on lipid content of the samples. In the first four years of sampling (Sloan and Armstrong, 1980), data were generally found to be much more consistent when based on lipid content. Other studies have not been based on lipid content. Sloan *et al.* (1988) state that "expressing PCB data on a lipid basis (*i.e.* µg PCB/g-lipid) has not been useful in the past to explain

variability in PCB concentrations in migrant species." Therefore, in their study of striped bass, results were not based on lipid content.

In the present analysis, PCB data in fish have been evaluated on a lipid basis to provide consistency with results from the Upper Hudson. Time trends in lipid-based means for Aroclor 1016, Aroclor 1254 and their ratio for striped bass are shown in Figure A.3-4. Since 1978, the arithmetic means illustrate a drop in concentration, although the change is not as dramatic as the Sloan *et al.* (1988) geometric mean plots. The data summarized in Spagnoli and Skinner (1977) report average Aroclor 1254 concentrations (only) in striped bass in the whole Lower Hudson as approximately 15 ppm wet weight (or approximately 200 µg/g-lipid, assuming an average 6.8 percent lipid in striped bass).

While the averages show only slight declines in recent years, exponential decay functions can be fitted to the plots. Aroclor 1016 levels for striped bass below River Mile 80, excepting 1975-6 data, fit an exponential decay very closely, with an apparent half-life of 2.6 years (Figure A.3-5, 95 percent confidence bounds on regression shown by dotted lines). Aroclor 1254 levels do not provide quite as smooth a fit, but show an apparent half-life of 7.1 years. The apparent half-life for total PCBs (sum of Aroclor 1016, 1221 and 1254) is 5.3 years.

A comparison of lipid-based Aroclor 1016 and Aroclor 1254 levels in all Lower Hudson striped bass samples is shown in Table A.3-6. Although the concentration of both Aroclor mixtures has decreased dramatically since 1977/1978, the lower chlorinated congeners quantitated as Aroclor 1016 have decreased in concentration faster than the higher chlorinated congeners quantitated as Aroclor 1254, leading to an increase in the ratio of Aroclor 1254 to Aroclor 1016. Sloan *et al.* (1988) conclude that Aroclor 1016 levels in striped bass have declined to a level where "the 'Aroclor 1254' component is the determinant for the fate of PCB in Hudson River striped bass." This observation is significant, because sewage inputs of PCBs to the estuary from the NYC metropolitan area are thought to be characterized by a larger percentage of higher chlorinated congeners than the input to the estuary from the Upper Hudson.

A.3.3.3 Other Migrant/Marine Species

Other migrant/marine species monitored in significant numbers, but not sampled since 1985/1986, include American eel (*Anguilla rostrata*), American shad (*Alosa sapidissima*), Atlantic tomcod (*Microgadus tomcod*), alewife (*Alosa pseudoharengus*), and blueback herring (*Alosa aestivalis*).

In general, a correlation is expected between lipid content of fish and the concentration of PCBs, because of the lipophilic nature of the compound. An exception to this rule is the American shad where a significant correlation between total lipid and total PCBs was found in only one of 20 sample sets. There may be a lack of correlation for shad, because shad are transient in the estuary or do not feed there. For other migrant species including alewife, blueback herring and rainbow smelt, PCBs appear to accumulate at a rate related to body size, *i.e.* surface area to volume ratio (Sloan and Armstrong, 1980). PCB concentrations in marine species such as Atlantic tomcod, immature bluefish, Atlantic sturgeon and American eel are reported as showing significant correlations with lipid content, but not with length.

For the years from 1978 through 1981, a significant decrease in PCB concentrations in fish was observed for all species, but "most of the decline in PCB concentrations of migrant/marine species has been primarily due to the reduction of Aroclor 1016" (Sloan and Armstrong, 1980). The average percent decline in Aroclor 1016 calculated over those years was 42 percent, compared to five percent Aroclor 1254.

A relatively long time series of observations for a few migrant/marine species in the estuary are available at the Tappan Zee Bridge (River Mile 27). Trends in lipid-based PCB concentrations at this location are shown in Figure A.3-6 for striped bass and American shad, with 95 percent confidence limits on the arithmetic means. This data set from the lower estuary does not show the sharp drop off in PCB concentrations from 1978 to 1980 typical of the complete Lower Hudson data set. The shad show substantially lower bioaccumulation than the striped bass, reflecting their short residence in the estuary.

A.3.3.4 Resident Freshwater Species

A significant population of resident fish species, present in the freshwater portions of the river, form a valuable resource. Significant numbers (in space and time) of PCB analyses have been undertaken for the freshwater species largemouth bass (*Micropterus salmoides*), pumpkinseed (*Lepomus gibbosus*), redbreasted sunfish (*Lepomus auritus*), white perch (*Morone americana*), and yellow perch (*Perca flavescens*). The greatest number of samples in the upper estuary are available for largemouth bass and pumpkinseed. For these fish, no clear trend for PCB levels is apparent since 1980. Median PCB levels in largemouth bass at River Miles 112-114 go from a high of 31.2 ppm in 1978, to a low of 0.9 ppm in 1980 and rise again in 1986 to 10.2 ppm (Table A.3-7).

Lipid-based trends in total PCB concentrations in fish at Catskill (River Miles 112-114) are shown in Figure A.3-7. In contrast to the time series at Tappan Zee, the drop in PCB burdens from 1978 to 1980 is very clear in the largemouth bass population. Since 1980, total PCB concentrations in largemouth bass at Catskill have remained approximately steady. Although there are less data for yellow perch (the next most frequently sampled fish at this location), a similar pattern exists.

A large number of fish have been sampled in the upper estuary from Catskill (River Miles 112-114) and just below the Federal Dam at Troy (River Mile 153). Although technically in the Lower Hudson, fish species at River Mile 153 are very similar to those found just upstream. Furthermore, the PCB loading here is likely to come from upstream only and can be estimated from [illegible] measurements at Waterford. Therefore, the fish sampling just below the Federal Dam is most conveniently reviewed together with samples from the Upper Hudson and is discussed in conjunction with NYSDEC fish monitoring for the Upper Hudson River presented in Part B.

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A.4 Review of Lower Hudson PCB Mathematical Model

A variety of models developed within the past thirteen years have collectively attempted to link PCBs in the Upper Hudson to those found in the Lower Hudson. There have been efforts to develop specific transport models of PCBs (LMS, 1978, 1979; Apicella and Zimmie, 1978), models of ecosystem fate of PCBs (Hydroscience, 1979) and physio-chemical and striped bass food web models of PCB homologues (Thomann *et al.*, 1989).

Such a complex undertaking is not without difficulties and some models in the past have been criticized for not using "state-of-the-art" approaches (USEPA, 1981). Limburg (1986) has summarized and reviewed the successes and failures of both the LMS (1978; 1979) and Hydroscience (1979) models. For Phase 1, a preliminary understanding of some aspects and assumptions of the Thomann *et al.* (1989) model is appropriate in order to understand the intricate linkages of various physical, chemical and ecological parameters within the Lower Hudson.

A.4.1 Thomann Model

A.4.1.1 Overview

Thomann *et al.* (1989) considered many aspects of PCB transport, geochemistry and biogeochemistry in order to construct a fairly complete description of the PCB budget for the Lower Hudson. The model was designed to consider PCB dynamic patterns over long time and large spatial scales. The model includes water column transport, sediment interactions, degradation, dredging, gas exchange, biological interactions and tidal dispersion. In addition, the model considers individual PCB homologues and their potential environmental fate.

The model evaluates the time history of PCB inputs from 1946 to 1987 and calculates annual PCB budgets for the Lower Hudson. The work integrates and summarizes the effects of major PCB sources and sinks over time and provides insight concerning recent loads to the Lower Hudson. The mass of PCB input to the system on a source by source basis was calculated and the mass of PCBs lost

from the system via gas exchange, burial in the sediments, transport to the shelf waters and degradation was estimated. Thus, the model provides a means of understanding the current conditions in the river, how they developed, and how they may change in the future.

On the basis of the model, several conclusions were drawn on the environmental fate of PCBs in the Lower Hudson, New York Bight and Long Island Sound. The study concluded that the maximum PCB load to the Lower Hudson via the flow at the Federal Dam was 150 lb/day (68 kg/day), corresponding to the removal of the Fort Edward Dam in 1973. Since that time, the load decreased exponentially, to an estimate of 3 lb/day (1.4 kg/day) in 1987. The load of PCBs from the Upper to the Lower Hudson was dominated by the di- and trichlorobiphenyls (40 percent) and by the tetrachlorobiphenyls (40 percent). The remainder consisted of heavier PCB homologues.

The PCB loadings to the Lower Hudson from sewage discharges, atmospheric deposition and runoff also reached a maximum in the early 1970s at 30 lb/day (14 kg/day) and declined steadily since that time. However, the decline of these sources has not been as rapid as that of the upriver source. These additional sources dominate the PCB input to the entire system and represent about 46 percent of the total PCB loading to the Lower Hudson in 1980.

A mass balance for the Lower Hudson and estimates of the relative contribution of various sources and sinks were presented. A total of 595,000 lb (270,000 kg) were estimated to have been discharged into the Lower Hudson River through 1987. Of this total, 66 percent was estimated to have entered via the flow at the Federal Dam, 20 percent via tributaries and urban runoff, 12 percent from municipal sewage discharges and 2 percent from atmospheric deposition.

PCB losses from the Lower Hudson River through 1987 were estimated as follows: 66 percent was lost via gas exchange to the atmosphere, 19 percent was lost via transport to Long Island Sound and the New York Bight, 9 percent was lost via dredging of sediments, while 6 percent remained stored in the sediments of the Lower Hudson. In addition, the loss terms were homologue-dependent, with

gas exchange being more important for the lower chlorinated homologues and transport to the surrounding waters playing a more significant role for the highly chlorinated homologues.

The storage of PCBs in the Lower Hudson sediments did not always represent a net loss of PCB from the water column. The study concluded that in the 1970s the sediments acted as a net sink of PCBs. By 1987, however, the sediments were releasing PCBs to the water column with a resulting loss of about 10 percent of the maximum sediment PCB inventory. Although the sediments remain a net sink, their importance to the PCB mass balance as a sink term will continue to decrease with time.

As part of the investigation, a striped bass food web model of PCB homologues was constructed. In order to assemble the model, the authors analyzed the existing striped bass, PCB data base, formulated equations based on uptake and accumulation components and specified food web interactions and physiological parameters. Finally, they compared (calibrated) model output of total PCBs and PCB homologues to observed levels in certain age classes of striped bass and white perch collected in selected regions of the Lower Hudson.

The striped bass food web model encompassed a variety of trophic groups including the phytoplankton, zooplankton (*e.g. Gammarus*), "small fish" and white perch. The age-dependent (17 age classes) striped bass model was driven by outputs from the geochemical transport model and was "...successfully calibrated to white perch and striped bass in the mid and Lower Hudson using data from 1978-1987" (Thomann *et al.*, 1989).

Some of the major results of the striped bass food web model indicate that peak concentrations of PCBs (45 µg/g, wet weight) in striped bass occurred in the mid 1970s and have declined at an exponential rate of 0.057/yr since 1980-1982 (approximate half-life of 12 years). In addition, PCB concentrations in striped bass are linear to the phytoplankton uptake of PCBs from the water column (commonly termed bioconcentration). Furthermore, greater than 90 percent of the PCBs in striped bass are due to food web bioaccumulation and only a minor

percentage (approximately 10 percent) is accounted for by an uptake of PCBs from the water column.

The effort by Thomann *et al.* (1989) represents a vast undertaking, because so many parameters and fluxes were directly linked in order to produce an improved understanding of PCB dynamics in the Lower Hudson. Specific results of the modeling with respect to PCB mass transport estimates, geochemical processes and ecological parameters are presented below.

A.4.1.2 Mass Transport Estimates

PCB mass loads were estimated from PCB and river discharge measurements taken at the USGS monitoring station in Waterford. For the period 1977-1983, Thomann *et al.* (1989) estimated that approximately 19,000 kg (2,700 kg/yr on average) of PCBs were transported past Waterford from the Upper to Lower Hudson. The reported estimate is approximately 35 percent higher than the quantity computed by TAMS/Gradient, which is presented at B.4.

Mass loads of PCBs from the Upper to Lower Hudson prior to 1976 cannot be estimated from water column PCB concentrations, because none existed. (USGS monitoring began in 1976.) Prior to 1976, the model based mass load calculations on an approximate dating method applied to the PCBs measured in NYSDEC sediment core samples as reported by Hetling *et al.* (1978). Apparently, a constant sedimentation rate was assumed, which may be unrealistic in light of the historical channel destabilization and sediment scour following the removal of the Fort Edward Dam. Thomann *et al.* (1989) estimate that approximately 1,000 kg/yr of PCBs were transported from the Upper Hudson in the early to mid 1960s. A maximum PCB discharge of 24,600 kg/yr was estimated for 1973 (Thomann *et al.*, 1989), which is approximately five-fold higher than that suggested by radiologically dated cores from the Lower Hudson (Bopp and Simpson, 1989).

A.4.1.3 Geochemical Processes

In the construction of a PCB budget for the Lower Hudson River, the Thomann model was required to simulate several important geochemical processes. In particular, the model needed to simulate PCB partitioning in the water column between dissolved and suspended matter forms. It also simulated the gas exchange of PCBs between river water and the overlying ambient air. Although there is a need to simulate the above processes, estimates of parameters used in simulations may be subject to some uncertainty and, in some cases, may have been underestimated.

Of particular importance are the suspended matter (or sediment) to water partition coefficients (K_D's) and the gas exchange coefficients. Thomann used a model of water-sediment interactions for the K_D values, which consistently predicts K_D values below those measured by the Lamont-Doherty Geological Observatory (Bopp *et al.*, 1985; Warren *et al.*, 1987). This consistent bias in the K_D will result in the overestimation of the water-borne transport from the Lower Hudson, since estimates of the total water column concentration are directly dependent on the partition coefficient. It will also result in the overestimation of any flux, such as gas exchange, which is dependent upon the dissolved concentration of PCBs.

The model assumed a constant value for the gas exchange coefficient, derived from oxygen exchange across all congeners found in the river. This assumption ignores several important factors related to the large chemical differences among individual PCBs and between PCBs and oxygen and may overestimate the actual gas exchange rate by a factor of two or more. Depending upon the type of model used to simulate gas exchange, ignoring the differences among congeners can result in errors of 20 to 40 percent in the gas exchange coefficient. Furthermore, the lack of incorporation of additional gas exchange resistance related to the very low Henry's Law constant values for PCBs (about 10^{-2} to 10^{-3}, unitless) may underestimate the gas exchange coefficient (Bopp, 1983).

The gas exchange rate utilized as an input to the model is probably high, given literature estimates for PCBs (Bopp, 1979). In addition, the gas exchange problem is compounded by the underestimation of K_D as previously discussed. Since model calculations show gas exchange loss to represent the most important single flux of PCBs from the Lower Hudson, the potential error in all of the mass balance calculations is presumably high.

A.4.1.4 Ecological Parameters

The core of the striped bass food web model of PCB homologues links the change in concentration of PCBs in striped bass to three components: (1) uptake of PCBs from the water; (2) uptake of PCBs from other sources, mainly predation; and (3) elimination (depuration) of PCBs. Food chain uptake parameters which include bioconcentration factors (BCF) and bioaccumulation factors (BAF), are discussed below and defined in the Glossary.

Uptake of PCBs from Water (Bioconcentration)

Thomann *et al.* assigned the BCF for all levels above the phytoplankton as equivalent to the log octanol/water partition coefficient (K_{ow}). Since the primary route for accumulation from the water is by exchange across lipoprotein cell membranes, it appears valid to assign BCFs based on K_{ow}, although regression relationships between K_{ow} and BCF values would be preferred. It should be noted, however, that the efficiency of uptake increases with increased K_{ow} until a plateau is reached and then declines at log K_{ow} values greater than 6, which is in the range of the pentachlorobiphenyl and hexachlorobiphenyl PCB homologues.

The phytoplankton BCF was assigned a constant value of 30 *l*/g(w) [liters per gram (wet weight)], based largely on the results of Oliver and Niimi (1988). The constant value assigned is for PCB homologues dichlorobiphenyl through hexachlorobiphenyl, which actually ranged from 14 to 61 *l*/g(w). Other PCB homologs such as octachlorobiphenyl had a BCF of 100 *l*/g(w), according to the data presented by Oliver and Niimi (1988) for Lake Ontario plankton.

Thomann *et al.* use Oliver and Niimi's (1988) plankton data as indicative of phytoplankton BCF when, in fact, their plankton samples contain a mixture of phytoplankton and zooplankton. In addition, studies on the uptake kinetics of PCB by phytoplankton suggest a great deal of species-specific variation (Cosper, 1991 per. comm.). Thus, a reassessment of the phytoplankton BCF is probably warranted.

Uptake of PCBs from other Sources (Bioaccumulation)

The main avenue of PCB accumulation in fish is via consumption of food containing PCBs. As an animal consumes food, its growth will be less than that projected by the content of the food. One gram of food does not produce one gram of growth, since there are associated respiratory losses, which Thomann *et al.* (1989) correctly recognize. An important variable here is the assimilation rate, which diminishes with log K_{ow} above 6, but remains greater than zero at all times (Thomann *et al.*, 1989). Thus, the assimilation efficiencies used for the various homologues seem reasonable.

As Thomann points out, most studies indicate that striped bass shift from a primarily invertebrate diet to a primarily piscivorous mode of feeding at approximately two years of age or older. Depending on salinity, either *Gammarus* or *Neomysis* makes up the largest percentage of the diet for the young striped bass (age classes 0 to 2). Contrary to the Thomann assumption, however, white perch have not been shown to be "an important food source" of Hudson River striped bass populations and it is not appropriate to consider white perch as the "middle trophic level" for striped bass populations. Rather, it is documented that Hudson River striped bass (age classes > 2 years) are opportunistic piscivores and consume a variety of middle level trophic prey such as blueback herring, Atlantic tomcod, bay anchovy and mummichogs (Klauda and Setzler-Hamilton, per. comm.). The relative percent contribution of the various species to total striped bass dietary consumption is affected by seasonal changes and constraints in prey populations.

Another potential problem in the food web interactions presented by Thomann concerns the link between the phytoplankton and zooplankton compartments. Thomann *et al.* (1989) state that the phytoplankton are preyed upon by the zooplankton, which is represented by *Gammarus*. However, it is well established that Gammarid amphipods, including the genus *Gammarus*, are essentially bottom dwelling epibenthic organisms and commonly inhabit the surficial sediments. Gammarids are classified as detritus feeders or scavengers and do not utilize phytoplankton as a primary dietary source. In fact, Bek (1972) has shown that daily consumption of detritus by juvenile and adult gammarids may range from 60-100 percent of total body weight. These food links and consumption rates are important considerations in bioaccumulation assessments, since sedimentary concentrations of PCB are often orders of magnitude greater than those in the water column.

Elimination of PCBs

The term excretion is somewhat misleading, since what should be understood by the term are all those factors that result in a reduction of body concentration of PCBs. Such a change could come about from actual excretion, metabolism or dilution as the animal grows. In view of the extremely low water solubility of PCBs, the excretory route is probably low, as indicated by Thomann. On the other hand, animals do grow, leading to a possible dilution of the ingested PCBs.

A.4.2 Simulations Relevant to Upper Hudson Remediation

As part of their investigation, Thomann *et al.* (1989) developed a fairly intricate model of the food web, which was coupled to a geochemical transport model. Their effort is one of the first attempts to model the breadth and complexity of PCBs in the Lower Hudson River, New York Bight and Long Island Sound. Undoubtedly, the most important use of the model is for estimating future PCB levels in striped bass under various remedial scenarios.

The model was used to simulate the time it would take for 50 percent and 95 percent of the striped bass to be below the FDA threshold of 2 µg/g (wet weight) of PCBs under two scenarios: (1) complete elimination of Upper Hudson PCB sources and (2) status quo or a no action alternative. Conclusions based on these scenarios are a direct result of their estimates on the relative importance of the upriver source and how other inputs to the Lower Hudson will vary with time.

Under the no action scenario, the authors conclude that by 1992, the median concentration in three to six-year old striped bass would be below the FDA threshold of 2 µg/g (wet weight). If the upriver source were completely removed as of 1987, they conclude that this median concentration would be achieved a few years earlier (1990). When conditions are extrapolated to achieve a goal of 95 percent of striped bass below the 2 µg/g threshold level, both scenarios seem to yield the same result by the year 2004. Because Thomann *et al.* estimate that the Upper Hudson load contributes only 10 percent to PCB levels in striped bass, the Lower Hudson PCB inputs and sediment releases may be the most critical factor influencing the time required for PCBs in the striped bass population to drop below the 2 µg/g FDA threshold.

The assignment of various geochemical and physiological parameters to the model should be periodically reinvestigated in light of extant data. This preliminary assessment of the Thomann *et al.* (1989) model agrees with some of the final comments that "... simulation results should therefore be viewed as indicators of overall trends only." In any model, there is often a great deal of skepticism concerning the various simulations. As acknowledged by Thomann *et al.* (1989), spatial and temporal variability in PCB water concentrations, loading estimates, and variable striped bass migration patterns may contribute to overall model uncertainty.

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PART B

PHASE 1 REPORT

INTERIM CHARACTERIZATION AND EVALUATION

HUDSON RIVER PCB REASSESSMENT RI/FS

CONTENTS

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

TABLES

Tables Located in Volume 1 (2 of 2)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

FIGURES

Figures Located in Volume 1 (2 of 2)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PLATES

Plates Located in Volume 1 (2 of 2)

PART B

CONTENTS
(continued)

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SYNOPSIS
PHYSICAL SITE CHARACTERISTICS
(Section B.1)

The hydrology of the Upper Hudson River is described in detail (B.1.1), building upon the basic description of Hudson River hydrology given in Part A. Flow characteristics of the Upper Hudson generally show a strong seasonal dependence, with maximum flows during the annual spring thaw. Flows are partially regulated by wetlands as well as the Sacandaga Reservoir. Four major tributaries to the Upper Hudson below Fort Edward combine with flow upstream to produce an average annual flow of 7100 cfs at Waterford, above its confluence with the Mohawk. This basic flow regime governs the transport of PCBs in the Upper Hudson.

Water quality (B.1.2) is described according to New York State water quality classifications assigned on the basis of "best usages" and the results of water quality sampling data from a 1987-1988 survey. The water quality of the Upper Hudson at Fort Edward and Schuylerville was rated as poor, partly because of the fishing ban, as a result of historic PCB discharges.

Only the town of Waterford draws its drinking water from the Upper Hudson below Fort Edward. More commonly, the river water is used for industrial and commercial purposes, such as power-generation, and for domestic and agricultural use, such as watering lawns, gardens or crops. The Upper Hudson River is a navigational waterway; from Waterford to Fort Edward, it is co-incident with the Champlain Canal.

Population in the four counties bordering the Upper Hudson between Albany and Glens Falls is over half a million and land use is predominantly agricultural (B.1.3). Dairy farming is the principal form of agriculture. The region is also host to a number of industries, generally located in the vicinity of the population centers.

The Upper Hudson River represents a diverse fisheries resource (B.1.4); six fish surveys from 1933 to 1985 are reviewed. In general, these surveys show that the majority of fish species historically found in the Upper Hudson continue to reside there. The data also indicate a qualitative improvement in the fisheries resource.

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B. UPPER HUDSON CHARACTERIZATION

B.1 Physical Site Characteristics

B.1.1 Hydrology

The Upper Hudson River flows southerly from its source at Lake Tear-of-the-Clouds near Mt. Marcy in the Adirondacks to its confluence with the Mohawk River near the Federal Dam at Green Island, Troy, NY. The drainage area of this segment, shown in Plate A.1-1, is 4,630 square miles (Wagner, 1982). (Some discussion of overall Hudson River basin hydrology, presented in Part A, is not repeated here.)

The Upper Hudson River drains a major portion of the southern and central Adirondacks. Along its course, the main channel is intersected by several tributary branches, the most significant of which are the Sacandaga River, the Batten Kill, the Fish Creek and the Hoosic River (see Plate B.1-1). Water flow in this segment is regulated by several dams on the Hudson itself (see Plate B.1-2), as well as on tributary branches. Flow is further controlled by abundant wetlands located throughout the basin, which act as a buffer for high and low flow conditions.

The total mean annual fresh water flow from the Upper Hudson at its confluence with the Mohawk near Waterford is about 7,100 cfs. This flow represents more than a two-fold increase in flow from that at Hadley, NY. Before the river reaches the Bakers Falls - Fort Edward area, it is joined by the Sacandaga River, the largest single tributary in this area. The mean annual flow at Fort Edward is roughly 3,800 cfs, about 54 percent of the total flow at Waterford. Downstream of this location, the remaining tributaries are fairly evenly spaced, at roughly 10 to 15 miles between tributary junctions. The combined total of these tributaries doubles the flow of the Upper Hudson by the time it reaches Waterford. Of particular importance is the Hoosic River, which represents about 15 percent of the total drainage area south of Hadley.

Flow in the Hudson Basin is seasonally dependent, with flow patterns similar to those seen throughout the basin. The typical regime is one of fairly steady flow throughout nine months of the year. During the spring, flows in the Upper Hudson increase substantially in response to the melting of winter snow. The maximum flow at Waterford occurs one month before the maximum at Fort Edward; melting of winter snows in the southern portion of the Upper Hudson basin tends to occur earlier than in the portion of the Upper Hudson basin above Fort Edward. Figure B.1-1 shows mean monthly flows for water year 1986 at Fort Edward and Waterford, respectively. This seasonal pattern is duplicated in the flows at the Federal Dam at Green Island for the same year.

Flows for an atypical water year (1984) are shown in Figure B.1-2 for Fort Edward and Waterford, respectively. That water year was characterized as having an unusually warm winter with many major storm events throughout the year. The flow patterns at Fort Edward and Waterford in that year were also present at the Federal Dam and on the Wallkill River, implying that these unusual conditions were felt throughout the entire Hudson basin.

B.1.2 Water Quality and Use

B.1.2.1 Water Quality

New York State has classified its surface waters according to "best usages" and has established numerical water quality criteria (standards) to which those waters should conform. Waters that conform to the numerical criteria are considered suitable for their intended best use. Water quality classifications applicable to the Upper Hudson are listed in New York State's environmental regulations (6NYCRR700, et seq.) and are illustrated on Plate B.1-3. In summary, the Upper Hudson has assigned to its different reaches the following classes and uses:

Class A - drinking water (water supply)
Class B - primary contact recreation (swimming), fishing, fish propagation
Class C - fishing, fish propagation, swimming
Class D - fishing, fish passage, swimming

Numerical quality standards for each of the above classifications are found in the State's rules at 6NYCRR700-705. A recent NYSDEC Technical and Operational Guidance Series (TOGS) memo (September 25, 1990) has augmented the rules, particularly with regard to toxic constituents. The standards encompass conventional pollutant parameters, such as coliform levels, dissolved oxygen, turbidity and pH, as well as toxic constituents, such as heavy metals and organics. The toxic substance standards have been derived from health and environmental risk assessments performed either at the state level or derived from those performed by USEPA.

Historically, the state has used information from diverse sources to ascertain the condition of its surface waters and to evaluate their attainment of the designated best uses. Currently, a program of rotating intensive basin studies (RIBS) exists whereby NYSDEC monitors all surface waters on a six year cycle and uses data from other programs to provide continuity of information when RIBS sampling is not occurring. An intensive basin survey was conducted within the Upper Hudson during 1987 and 1988 and initial reports from that effort have now become available (NYSDEC, 1990).

As part of the Upper Hudson RIBS effort, NYSDEC evaluated water column conditions, toxics in bottom sediments and contaminant uptake by macroinvertebrates and fish. Table B.1-1 provides the six main parameters/media used in the RIBS program to rate water quality. Samples were collected at five locations along the river's main stem at North Creek, Corinth, Fort Edward, Schuylerville and Waterford. Although NYSDEC did not report direct conclusions concerning attainment of water quality standards in its RIBS document, a qualitative evaluation of overall conditions within particular river reaches is provided. Conclusions pertinent to those reaches incorporating the Fort Edward and Schuylerville sampling locations are summarized here.

Both the Fort Edward and Schuylerville water column samples exhibited elevated copper levels at a sufficient frequency to warrant considering copper a parameter of concern. Similarly, iron was found to be a parameter of concern at Schuylerville. No other trace constituents were detected at elevated

concentrations with sufficient frequency to be considered parameters of concern. Copper, a pervasive constituent of Upper Hudson River water, is also found in samples from the relatively pristine North Creek and Corinth river reaches. While sediment samples were obtained and analyzed from the Schuylerville location, such samples were not obtained at Fort Edward. According to RIBS, Schuylerville sediments were at the upper limit of background for cadmium and were slightly above background for lead and mercury. NYSDEC did not identify the background levels upon which their conclusions were based.

The 1987-1988 RIBS program found no water column toxicity to *Ceriodaphnia* at either Fort Edward or Schuylerville. Tissue from caddisflies collected at the two sites did not exhibit elevated levels of either heavy metals or of PCBs, in contrast to previous studies. Apparently macroinverterbrate tissue collected in 1987 at Hudson Falls (location not specified) did exhibit elevated lead and manganese levels. At both Fort Edward and Schuylerville, NYSDEC assessed overall water quality as poor, based on the RIBS results and the fishing ban, a result of historic PCB discharges to this river reach. At one point the RIBS document concludes that water quality is rated as poor primarily due to the fishing ban (NYSDEC,1990,p.77).

As a result of several amendments to the Clean Water Act, states are required to report in specific terms conditions of their surface waters. Clean Water Act Section 305(b) mandates that states submit water quality condition reports to USEPA every two years. The 305(b) report evaluates surface waters in relationship to their ability to sustain primary contact recreation and fish propagation uses. In addition, Clean Water Act Section 304(l) requires that states generate lists of surface waters that fail to meet water quality standards because of toxic pollutants, in general, and toxics from point sources, in particular.

New York's most recent 305(b) report was published in April 1990. That document provides, in part, a summary of waters wherein contamination in fish exceeds either FDA levels or other guidelines. For the Upper Hudson reach from Hudson Falls to Federal Dam, the only contaminant identified as exceeding either

the FDA levels or other guidelines is PCB (NYSDEC, 1990, Table 20). The 305(b) report also identifies PCB in sediment as the sole toxic that is responsible for use impairment in the Upper Hudson (NYSDEC, 1990, Table 17). Similarly, the state's 304(l) lists identify priority organics (PCBs) as being the toxic responsible for Upper Hudson use impairment.

B.1.2.2 Use

The Hudson River is used as a source for public water supplies (municipal and institutional drinking water) in sections of the river classified as Class AA or A. Along the Upper Hudson, three communities draw directly Hudson River water. Of these, Queensbury and Waterford have current average uses of more than 1 mgd (NYSDOH, 1991), as shown in Table B.1-2. The Waterford intake is located at the base of the Upper Hudson Basin near Lock 1. The Queensbury intake is located near Sherman Island Dam in Warren County. The Winebrook Hills Water District, the third Upper Hudson water supply drawing from Hudson, is located at the headwaters of the Hudson in Newcomb, Essex County.

A more common use of Hudson River water is for industrial and commercial purposes such as cooling, manufacturing processes and fire protection. Hudson River water is also extensively used for hydroelectric and thermal power generation. An inventory of facilities and plants that utilize Hudson River water can be found in reports for the Hudson River-Black River Regulating District (Malcolm Pirnie, 1984a) and for the NYSDOT (1984).

Hudson River water is also used for domestic (watering lawns and gardens) and agricultural purposes (irrigating crops). There are currently no records of water withdrawal for agricultural uses. Unlike the other intakes, permits are not needed to withdraw water from the Hudson for irrigation purposes (pers. comm., NYSDEC and NYSDOH, 1991).

The NYSDEC Division of Water, Source Surveillance Section provided a listing (March 7, 1991) of all significant active facilities with SPDES permits in the Upper Hudson River Basin. This search revealed that 27 facilities

discharge into the Upper Hudson Basin, with 15 discharging directly into the Hudson River. Five of the 15 facilities are municipal wastewater treatment plants, including Corinth Sewage Treatment Plant, Glens Falls Wastewater Treatment Plant, Saratoga County Sewer District #1 Wastewater Treatment Plant at Mechanicville, Stillwater Sewage Treatment Plant and Washington County Sewer District #2 Water Pollution Control Plant at Fort Edward.

The Champlain Canal is coincident with portions of the Hudson River; it extends from Waterford, New York on the Hudson River to Whitehall at the southern end of Lake Champlain. The Champlain Canal is part of the New York State Barge Canal System, also comprised of the Erie Canal, Oswego Canal and Cayuga-Seneca Canals. This network of waterways connects the Atlantic Ocean with the Great Lakes and the Saint Lawrence Seaway. The Champlain Canal is 60 miles in length, including 37 miles of canalized Hudson River from Waterford to Fort Edward and 23 miles of land-cut sections. The canal diverges from the river at Fort Edward just below Lock 7 and proceeds in a northeasterly direction to Lake Champlain. Additional land-cut areas exist at Stillwater, Northumberland and Fort Miller.

Natural flows provide a considerable portion of the water supply needs for the Hudson River portion of the Champlain Canal. The Hudson River at Fort Edward provides an average discharge of 5,244 cfs (USGS average for 1979-1990 water years) to the canal at the confluence below Lock 7. This flow is significantly influenced by regulation of flows from Great Sacandaga Lake. Below this point, the river is canalized to provide natural flows for the canal. Water is also supplied to the Champlain Canal via the Glens Falls Feeder Canal. The Feeder Canal diverts approximately 100 cfs of water from the Hudson River upstream of the Feeder Dam west of Glens Falls to the Champlain Canal Summit Level between Locks 8 and 9 on the canal near Smith's Basin. The Summit Level is the point of highest elevation on the canal and allows for the gravity flow of water in both directions as lockages are made (south to the Hudson River, north to Lake Champlain). Approximately 25 percent of the diverted water returns to the Hudson River (Malcolm Pirnie, 1984).

Commercial traffic has declined on the Champlain Canal and other canals in the Barge Canal system as a result of "the unreliability of the system for waterway transport" and competition from other modes of transportation for bulk products (Malcolm Pirnie, 1984b, US Army Corps of Engineers, 1977). Unlike the other three canals in the system, the Champlain Canal shows a steady decline in recreational use along the entire stretch of the canal on both the canalized Hudson River and the land-cut section north of Fort Edward (Malcolm Pirnie, 1984).

B.1.3 Population and Land Use

Four counties (Albany, Washington, Rensselaer and Saratoga) lie adjacent to the Upper Hudson between Albany and Glens Falls. All counties experienced growth between 1980 and 1990 with Saratoga having the greatest increase over the period and Rensselaer the lowest. Total population of these counties in 1990 was over 500,000.

Land use within a zone adjacent to the Upper Hudson River, depicted on Plate B.1-4, is mostly agricultural. Portions lie within New York State Agricultural Districts and include parcels considered to be prime farmland. Dairy farming is the major agricultural industry. The majority of the crops grown, such as corn and hay, are used for forage; small quantities of cash crops, such as oats and wheat, are produced. Industrial use is typically located near population centers. Major non-agricultural industries within the study area include: an industrial demolition company; several paper mills; hydroelectric plants; a grocery warehouse; and manufacturers of garden equipment, brake linings, brushes, paints, wallpaper, paper products, gun barrels, silicone products, abrasives, brass fittings and clothing. Forested and recreational land uses are scattered.

Existing recreational uses include Schaghticoke Canal Park at Lock 4 of the Champlain Canal and two town parks, which lie along the river in Fort Edward. Proposed for the Fort Edward area are a marina, to be located on the south end of Rogers Island, and a marina, trails and picnic areas to be located one mile

south of Fort Edward on the former Champlain Canal. Saratoga National Historic Park lies on the western bank of the River in the Town of Stillwater. Moreau State Park is located south of Glens Falls. At the confluence of the Mohawk and Hudson Rivers are Peebles Island State Park and the Van Schaik Island Country Club. Several parks and/or country clubs also front the River.

B.1.4 Fisheries

Fishery resources within the Hudson River from Federal Dam to Fort Edward are influenced by different physical features, as well as man-made structures such as locks, dams, guard gates, bridges, spillways and submerged power lines and cables. This array of physical features produces the following variety of different fish habitats:

- Outlets of streams and rivers;
- Shallow water areas (wetland and non-wetland);
- Designated ship channels of the canal (canalized river);
- Steep embankment areas with relatively swift current;
- Landcut (artificial) portions of the canal;
- Wet dumping grounds or spoil areas; and
- Various alternate channels separated from the main channel by an island.

This wide variation in habitats expands spatial heterogeneity and results in a complex fishery resource.

The New York State Conservation Department (Greeley, J.R. and Bishop, 1933) conducted an early and comprehensive fish inventory of the portion of the Hudson River between Hudson Falls and the mouth of the Hoosic River. Forty-one species of fish (Table B.1-3) were recorded. (The American shad, one might note, was listed as extinct.) The historical data developed as part of this 1933 fish inventory documented that there was an imbalance between the juvenile and adult

game fish. There appeared to be abundant juvenile game species from a few miles below Fort Edward to the mouth of the Hoosic River, but adult populations of game fish were uncommon.

An anonymous report, prepared by the Conservation Department in June 1960 (see Table B.1-3) at the request of the Stillwater Rod and Gun Club, are the only fish data that exist from 1933 to 1960. This 1960 report contains the observation that game fish resources declined between 1949 to 1959. Prior to 1949, angling success was supposedly satisfactory for "...bass, walleyes, northerns and pickerel." Several theories for this decline were advanced, such as industrial pollution from the Glens Falls and Fort Edward areas, effluent process changes at a Glens Falls paper and pulp mill, expanded boating activity and over-exploitation.

The next major fish inventory was conducted by Lane (1970) who found 13 species of fish. Lane stated the somewhat remarkable conclusion that: "The collection data indicate the absence of any significant fishery between Lock No. 1 and Fort Edward." With the exception of goldfish, few larger fish were found. Game fish collected in the river channel were represented largely by juvenile populations. Similar to 1933 observations, Lane noted a diminished amount of aquatic vegetation in the river channel. Although it is generally recognized that an adequate supply of aquatic vegetation is important for adult fish population maintenance and productivity, few historical and/or current data are available. One of the most unexpected results of Lane's survey was the absence of common carp. Although Lane concluded that "...conditions within the study area are not suitable for carp," later studies (Shupp, 1975; Makarewicz, 1983) do document the presence of the common carp within this section of the Upper Hudson.

Subsequently, NYSDEC began collecting fish for PCB analysis. Shupp (1975) collected fish samples in the 40-mile stretch of the Upper Hudson from Lock No. 1 to Hudson Falls and found an improvement, which he attributed to upgrading of treatment facilities and tougher regulations concerning industrial discharges. Although Shupp reported approximately 24 species of fish compared to the 13

species listed by Lane (1970), the fishing from Lock No. 1 to Hudson Falls was still considered poor, because of the overall low standing crop of fish and low numbers of adult fish compared to juveniles (Shupp, 1975). The reported preponderance of juvenile fish was similar to data from the 1933 and 1970 surveys. Sheppard (1976) indicated that "...some unknown factor is causing the exodus or demise of the mature segment of certain fish populations including the rock bass, pumpkinseed, yellow perch, walleye and chain pickerel." NYSDEC (R. Sloan, per. comm., 1991) has recently observed a greatly diminished number of both pumpkinseed and yellow perch populations during routine PCB assessments of resident fish in the Upper Hudson (Fort Edward to Federal Dam).

Since 1975, NYSDEC has continued to collect fish between Federal Dam and Fort Edward as part of their ongoing assessment and monitoring of PCB levels in fish flesh. The principal species collected and analyzed within this reach have been the brown bullhead, goldfish, largemouth bass, pumpkinseed and yellow perch. Because of the demise of the yellow perch and goldfish, current collection efforts have focused on the brown bullhead, common carp and largemouth bass (R. Sloan, per. comm.).

One of the most extensive fishery surveys since the 1933 survey was conducted approximately eight years ago by Makarewicz (1983). He surveyed 85 stations along the entire length of the Hudson between Federal Dam and Whitehall as part of the New York State Barge Canal Maintenance Dredging Program 1985-1995 for NYSDOT (Malcolm Pirnie, 1984b). The sampling stations included nine sampling reaches from Federal Dam to Fort Edward. A total of 46 species, including four migratory species (American eel, blueback herring, sea lamprey and striped bass), were found. Of the 42 resident freshwater species, the panfish ere the most prevalent (40 percent); demersal fish were second in abundance (22 percent); forage fish were the third most abundant group (14 percent); and game fish had the lowest relative abundance (9 percent). Dominant panfish members were bluegill, pumpkinseed, rock bass and yellow perch; demersal dominants were black bullhead and brown bullhead, common carp and white sucker; forage dominants were golden shiner, spotfin shiner and spottail shiner; and game fish dominants were

the largemouth bass and smallmouth bass. Collectively, these 13 species accounted for 85 percent of all resident freshwater species collected.

The most recent fish survey data available are from a study conducted by Green (1985) between Stillwater (Lock 4) and Schuylerville (Lock 5), covering approximately 13 river miles. Fewer overall species were taken (20) compared to the more intensive biological surveys in 1933 (41 species) and 1983 (46 species), both of which covered more of the river (Fort Edward to Federal Dam) than that covered in 1985 (Stillwater to Schuylerville).

Length at age comparisons for both smallmouth and largemouth bass indicate that growth rates were comparable to the New York Bass Study's average to fast growth rates (Green, 1985), an indication that some of the historical observations regarding the preponderance of juvenile fish and the paucity of adult game fish may no longer be valid for bass populations.

As shown in Table B.1-3, the list of species of fish in the 1983 study agrees quite well with the list for 1933. Although comparative percent contributions of the dominant game fish, panfish, forage fish and demersal fish with those species recorded in 1933 is not possible, because quantitative information are lacking in the historical study, all the above dominant species (with the exception of the spotfin shiner and black bullhead, see note 1, Table B.1-3) were also recorded in 1933. In addition, 31 species were similarly reported in both studies. An analysis of these two studies (Greeley and Bishop, 1933 and Makarewicz, 1983), which have spanned nearly 50 years, reveals considerable qualitative similarity of the fishery within the reach from Federal Dam to Fort Edward.

The construction of Federal Dam and various locks as part of the Champlain Canal Section of the New York State Barge Canal System blocked major upstream spawning migrations for a number of anadromous species, including the American shad, alewife, blueback herring, sturgeons and striped bass. Some migrants, as documented by Smith and Lake (1990), may be found periodically upstream of the

Federal Dam and Lock. Population pulses may enter and leave the lower region of the Upper Hudson through the interconnecting system of locks.

The majority of fish species listed in Table B.1-3 are freshwater residents of the Hudson River. Some migratory species, such as the striped bass, blueback herring, sea lamprey and American eel, still attempt to utilize sections of the Upper Hudson as migratory routes. With the exception of black bullhead, johnny darter, pearl dace and northern redbelly dace, all the fish species listed have also been found in various regions of the Lower Hudson River (Beebe and Savidge, 1988; Smith and Lake, 1990). Whereas many of the fish species are year-round freshwater residents, they are not unique to the Federal Dam/Fort Edward section of the Upper Hudson.

Some additional fish species, which were not found during the reported fish surveys summarized in Table B.1-3, have been reported in various sections of the entire Upper Hudson Region (Smith and Lake, 1990). These include:

American shad *(Alosa sapidissima)*
Stonecat *(Noturus flavus)*
Longnose sucker *(Catostomus catostomus)*
Lake chub *(Couesius plumbeus)*
Brassy minnow *(Hybognathus hankinsoni)*
Blacknose shiner *(Notropis heterolepis)*
Finescale dace *(Phoxinus neogaeus)*
Lake herring *(Coregonus artedi)*
Lake whitefish *(Coregonus clupeaformis)*
Round whitefish *(Prosopium cylindraceum)*
Lake trout *(Salvelinus namaycush)*
Rainbow smelt *(Osmerus mordax)*
Tessellated darter *(Etheostoma olmstedi)*

Although not found during the major fish surveys conducted within the Federal Dam/Fort Edward Region, the American shad has been known to occur within this particular region (per. comm., R. Sloan). Others such as the lake chub, lake herring, lake whitefish and lake trout are not commonly found in freshwater riverine systems and are not expected to occur to any great extent within this section of the Upper Hudson.

All studies reviewed to date indicate that the majority of species historically present in the lower section of the Upper Hudson continue to reside in this particular reach of the Hudson River. According to information submitted by Shupp (1987), the section of the Upper Hudson River between the Federal Dam and Fort Edward can support a diverse and high quality fishery resource. Shupp also cited evidence gathered from some NYSDEC studies between Mechanicville and Schuylerville, which suggested a "vast improvement" in smallmouth and largemouth bass stocks and other fish species from the early 1960s to the late 1980s. Shupp has stated, "Since 1984, the greatly improved warm water fish community in the Fort Edward to Troy (Upper Hudson) reach has stimulated interest in reopening the fishery."

Angler reports to The Warrensburg Fish Management Unit indicate a somewhat improved fishery for bass, yellow perch, black crappie and brown bullhead from 1969 to 1975 (Shupp, 1975). Analysis of the data presented by Lane (1970), Makarewicz (1983) and Green (1985) and prepared testimony statements by Shupp (1987) also suggest a qualitative improvement within the past twenty years.

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SYNOPSIS
SOURCES OF PCB CONTAMINATION
(Section B.2)

General Electric discharged PCBs from plants at Fort Edward and Hudson Falls between 1946 and 1977 (B.1.1). The total amount of PCBs released during 1957 to 1975, a period for which estimates can be made using historic data, ranges from 209,000 to 1,330,000 pounds.

Currently, six New York State facilities are permitted to discharge PCB-contaminated waste water to the basin of the Upper Hudson River (B.2.2). A facility in western Massachusetts discharges to a Hudson River tributary, the Hoosic River.

Other potential sources of PCBs to the Upper Hudson (B.3.3) are discussed.

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B.2 Sources of PCB Contamination

B.2.1 GE Discharges (To 1977)

The two GE plants at Fort Edward and Hudson Falls, New York began to use PCBs in 1946 and discontinued their utilization in 1977. In-plant sources of PCB discharges have been characterized as both minor spills and effluent from washing capacitor cans, with the latter being the major source. Capacitor cans were flood filled with dielectric fluid and then washed with detergent and water to remove excess material. Contaminated wash water was finally discharged, untreated, to the Hudson River (Brown, Jr. *et al.*, 1984).

Estimates of PCB releases at the two capacitor plants have been made on the basis of GE's overall usage of the chemical and considering discharges allowed pursuant to USEPA's discharge permit for the facilities. Figure B.2-1 illustrates the company's PCB usage, by Aroclor type, for the period 1946 to 1977. That figure shows the trend in Aroclor usage to be from relatively highly chlorinated forms in the mid-1950s (Aroclor 1254) to less chlorinated homologues in the 1960s (Aroclor 1242) and the 1970s (Aroclor 1016).

By using actual GE purchase records for the years 1966 thru 1975 and approximating GE purchases for the prior period on the basis of Monsanto's production records, GE's total PCB consumption for the period 1957 to 1975 has been estimated at 133,000,000 pounds (Limburg, 1985). Plant discharges during the 1960s have been approximated at 5 metric tons per year (Sofaer, 1976), a rate which is roughly compatible with that allowed by GE's 1975 discharge permit (30 pounds per day or about 11,000 pounds per year). Sanders (1989) provides anecdotal evidence of plant releases being less than one percent of plant consumption or less than 1,330,000 pounds from 1957 to 1975. Thus, one estimate of the range of releases to the river would be 209,000 to 1,330,000 pounds over the period 1957 to 1975, where the lower quantity is based on a continuous discharge of 30 pounds per day (or 5 metric tons/year) for a nineteen-year period.

B.2.2 Current Permitted Discharges

Six facilities in New York State, including GE, are permitted to discharge PCBs in the Upper Hudson Basin. NYSDEC (March 19, 1991) provided SPPDES permits and discharge monitoring reports (PCBs only) for each of these facilities. Table B.2-1 identifies these facilities, receiving waters and relevant information on PCB limits and measurements. Two facilities (GE, Fort Edward and James River Corporation, South Glens Falls Mill) are permitted to discharge PCBs directly into the Upper Hudson River, while one (GE, Old Fort Edward Site Remediation Project) discharges into the Old Champlain Canal in the vicinity of Fort Edward. In most cases, the concentration of PCBs in the final effluent is limited to the minimum reliable detection limit based on USEPA Method 608.

According to available Commonwealth of Massachusetts SPDES records, Sprague Electric Company is permitted to discharge PCBs (0.01 mg/l) directly into the Hoosic River, which flows into the Upper Hudson.

B.2.3 Other Sources

Table B.2-2 identifies inactive hazardous waste disposal sites located near the Upper Hudson River (above Federal Dam at Troy) in which PCBs have been dumped. This tabulation was obtained from NYSDEC, Division of Hazardous Waste Remediation, utilizing their annual inventories of disposal sites in New York State (April 1990). The NYSDEC priority classification codes stated in the table are: Code 2 - Significant threat to the public health or environment and action is required; Code 2a - Temporary classification assigned to sites with inadequate or insufficient data for inclusion in any of the other classes; Code 3 - Does not present a significant threat to the public health or environment, and action may be deferred; Code 4 -Site is properly closed and requires continued management.

The release of some contaminants from these inactive sites adds to the total PCB loadings in the Upper Hudson. In many instances, flow of surface water and groundwater from the sites are towards the Hudson River. "Unknown" material

was listed as being disposed of at many sites and is generally not included in the table. None of the sites identified by NYSDEC are classified as an imminent danger to the public or environment (Code 1). Many sites are classified as Code 2, suggesting that these sites may be potential sources of pollution to the Hudson River.

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SYNOPSIS

NATURE AND EXTENT OF CONTAMINATION

(Section B.3)

Available environmental data on the distribution of PCBs in the sediments, water, fish, air and plants of the Upper Hudson River as well as supporting data on flow and sediment transport are summarized and evaluated. As a foundation for continued analyses, available data have been compiled into a computerized, relational database management system (B.3.1).

Data on PCB concentrations in river-bottom sediments (B.3.2) are drawn primarily from the 1976-1978 NYSDEC sampling efforts and the 1984 Thompson Island Pool investigation, along with several other sources. Sediments are the major environmental repository for PCBs in the Upper Hudson, but there is a high degree of spatial variability in PCB concentrations. The 1984 study covered only the Thompson Island Pool and relatively little data have been collected since. It is difficult to determine the current mass and distribution of PCBs in sediments without further investigation.

The discussion of surface water monitoring (B.3.3) concentrates on data collected by the USGS. Transport of PCBs is affected by hydrologic processes, particularly flood events. A discussion of flow monitoring is followed by presentation of time series data, to the extent available, for suspended sediment and PCBs in the water column. Current full-year and summer average PCB concentrations are calculated, taking into account the problem of numerous measurements below analytical detection limits.

NYSDEC has monitored Upper Hudson fish on a regular basis since 1975; data are presently available for PCBs in fish through 1988. The extensive data collected in this program (nearly 3,000 Upper Hudson samples) are discussed (B.3.4). Total PCB burdens in fish declined sharply from 1978-1981. Levels of the higher chlorinated congeners in fish appear to have remained relatively constant since 1982. Results of NYSDOH macroinvertebrate monitoring are also described.

PCB monitoring data for air and plants near the Upper Hudson (B.3.5) are generally insufficient to assess the impact from PCBs in the river. Isolating the contribution of the river from other possible PCB sources is a particularly difficult problem.

For various other media there is a notable lack of monitoring data (B.3.6). Only limited groundwater sampling has been performed and surface soils near the river have not been monitored.

Data quality and analysis methods for the various monitoring programs are evaluated. PCBs have many different variations in chemical structure and differing physical properties. Uncertainties surrounding PCB measurement, particularly the specific variations in PCBs, results in considerable difficulties in interpreting the results. Furthermore, differing PCB measurement methods used for water, sediments or fish confound direct comparisons among them.

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B.3 Nature and Extent of Contamination

B.3.1 Overview of Sources and Database

Site data defining the current understanding of the nature and extent of PCB contamination, based on previous studies, are summarized in this section. Data synthesis efforts have focused on:

- obtaining the most complete, and current, data sources available;
- compiling these data into a computerized database;
- evaluating the PCB data for the media sampled;
- identifying current trends and relationships; and
- determining the adequacy of the existing data.

During the early 1970s, NYSDEC and several other agencies began the first comprehensive monitoring studies for PCBs in the Upper Hudson. Fish, which were some of the earliest environmental samples analyzed, showed high concentrations of PCBs. These early investigations began what is now over two decades of studies on PCBs in water, sediments, fish and other media affected by PCB discharges to the Upper Hudson. Table B.3-1 summarizes the major investigations.

Past USEPA documents, including the 1984 Feasibility Study and the EIS, have been reviewed for this work. Emphasis, however, was given to reviewing additional, more recent data and evaluating that along with the long-term monitoring record contained in the previous studies.

Previous investigations at the Hudson River PCB site have examined the nature and extent of contamination in several media, including fish, sediments, river water and, to a lesser extent, air. Each medium is discussed below. Additional studies, currently being performed by GE at the remnant deposit sites, are not available for this report. Before discussing each medium, a brief overview of the TAMS/Gradient database is provided below.

As the foundation for these Phase 1 data evaluation efforts as well as continued analyses during subsequent phases of the project and possibly future projects, the data gathered during Phase 1 have been compiled into a relational database management system using PC-based Paradox ™ software. This database currently contains approximately 30,000 records of information, primarily on sediments, water, fish and some other biota obtained from numerous sources (see Table B.3-1). Data input, verification and database management have been conducted by the TAMS/Gradient project personnel.

In the current sediment database, there are nearly 2,500 samples for the period 1976 through 1990. The fish database contains approximately 8,000 samples for the period 1973 through 1988. Additional data for other aquatic biota (macroinvertebrates and multiplate data) account for several hundred additional samples. Water column data, including daily and peak flow, suspended sediment and total PCBs, comprise the bulk of the remaining data in the database. The database contains a small number of samples summarizing PCB data for air and crop plants.

Data for separate media are linked primarily through sample date and location information. For each medium, data are organized such that a unique sample identification number links information among tables. Individual data tables are grouped by medium to contain similar kinds of information. As an illustration of the database format developed specifically for this Reassessment and its contents, excerpts from the sediment database tables are summarized below in four tables: Sample Information, Core Section, Chemical and Non-Chemical. Each of these tables is linked by a unique sample identification number, *e.g.*, sample ID numbers 30000, 30016 and 30032 shown here, such that sample IDs in each table correspond to data about a single sample.

The Sample table contains information about sample date, location (River Mile distance from bank and northing and easting coordinates, where available), sample type (grab versus core), the agency or investigator responsible for the analytical method(s), the reference report/location of the original data and other information as available.

Database Table Example: Sample Information Table

Sample ID	Type	M/D/YR	River Mile	Feet fr. West Bank	Northing (ft)	Easting (ft)	Sampler	Water Depth (ft)	Elev. (ft)	Ref	Agency
30000	Grab	5/21/77	168.8	330.0	1071755	685695	100			2	O'Brien & Gere
30016	Core	3/18/77	188.4	100.0	1163740	698970	100	5.8	119.6	1	O'Brien & Gere
30032	Core	3/18/77	183.4	60.0	1140410	669040	100	2.2	102.4	1	O'Brien & Gere

Core samples in the Sample table are linked with the Core Section table, which identifies the length of each core sample section and the depth beneath the river bottom, *i.e.*, the depth of sample penetration for the top and bottom of each section.

Sediment Database Example: Core Section Table

Sample ID	Core Section No.	Bottom of Section (in.)	Top of Section (in.)
30016	1	1	0
30016	2	2	1
• • •			
30016	12	12	11
30032	1	1	0
• • •			
30032	9	9	8

Selecting a sample ID from the Core Sample and Section tables and locating the same ID in the Chemical data table shows either the Aroclor results for an entire grab sample or section by section results for core samples. Additional information describing analytical measurement methods, *i.e.*, extraction method, are contained in the database as available. The Chemical data table also contains non-PCB chemical data, such as metals analyses (not shown here), where available.

Sediment Database Example: Chemical Data Table

Sample ID	Parameter	Core Section No.	Extraction Method	Concentration (ppm)
30000	Aroclor 1016		shake	1.0
30000	Aroclor 1221		shake	1.0
30000	Aroclor 1254		shake	1.0
30016	Aroclor 1016	4	soxhlet	6.0
• • •				
30016	Aroclor 1254	12	soxhlet	0.1
30032	Aroclor 1016	5	soxhlet	234.0
• • •				
30032	Aroclor 1254	5	soxhlet	163.0

Finally, non-chemical data, such as sediment texture class, percent volatile versus total solids, are contained in the Non-Chemical table.

Sediment Database Example: Non-Chemical Data Table

Sample ID	Core Section No.	Parameter	Value
30000		% total solids	78.93
30000		% volatile solids	0.85
30016	1	texture	GRAVEL
30016	4	% total solids	85.97
30016	4	% volatile solids	2.2
• • •			
30016	12	% total solids	89.23
30032	1	texture	CL-WC
• • •			
30032	5	% volatile solids	25.39

B.3.2 Sediment

Two primary sources of information provide the largest amount of data on sediment contamination in the Upper Hudson: (1) printed results (>1,000 samples) of the 1976-1978 NYSDEC sampling survey and (2) computer files (>2,000 samples) of the 1984 Thompson Island Pool survey. These data were entered into a computerized database (referenced here as the TAMS/Gradient database). Tables B.3-2 and B.3-3 provide a summary of these sediment samples in the TAMS/Gradient database. In addition to these two major NYSDEC sediment surveys, USEPA, the Lamont-Doherty Geological Observatory and GE have sampled sediments in the Upper Hudson.

B.3.2.1 1976-1978 NYSDEC Sampling

As reported by Tofflemire and Quinn (1979), NYSDEC conducted several sediment sampling surveys in the Hudson River between 1976 and 1978. Details about the sampling and analysis procedures for these studies are summarized in NYSDEC Technical Report No. 56 (Tofflemire and Quinn, 1979).

The data provided by NYSDEC contained a total of 1,167 sediment samples (396 cores and 771 grabs) taken during 1976, 1977 and 1978; 1,770 PCB analyses were reported for the 1,167 samples. The overwhelming majority of samples (1,091 of the 1,167 samples) from the 1976-78 data set were collected in the Upper Hudson River. Only five samples in this data set were obtained in the Lower Hudson River and all of these five were from River Mile 153, just south of the Troy Lock. Another sample in this set was identified as from the Lower Hudson, but other descriptions placed it in the Upper Hudson, while 70 samples had no information regarding location.

Aroclors 1016, 1221 and 1254 were identified as the PCB mixtures detected in the 1976-1978 sediment sampling effort. Total PCB concentrations were reported as the sum of these three aroclors. Analytical detection limits were not reported in this data nor was any indication given about a sample's detectable or non-detectable concentrations of PCBs. Because several concentra-

tions (1 ppm, 5 ppm, 10 ppm) occur an inordinate number of times in this data set, these concentrations are the probable detection limits for these samples.

Table B.3-4 summarizes the Aroclor concentrations for grab and core samples, respectively, within each of the nine river reaches sampled. In summarizing the mean and median PCB values for the core samples, each core was counted as a single sample, *i.e.*, the statistics are calculated over the entire core length. The reported N is the count of core sections, not of samples. One of the most striking aspects of the 1976-1978 results is the enormous variation of PCB concentration over very short distances. Samples taken only a few feet apart may have PCB levels varying by orders of magnitude. This extreme variability is highlighted in Figure B.3-1, which plots total PCB concentration in surface sediments (grab samples and the top section of all cores) by river mile. Some trend with river miles is shown by the median. The median PCB concentration by reach is highest in the Thompson Island Pool (River Mile ≈ 188 - 194), decreasing for several miles downstream, then showing an increase from River Mile ≈ 175 to River Mile ≈ 160. This pattern points to a relative scarcity of depositional areas between Stillwater and Northumberland.

Based on the results of the 1976-1978 survey, NYSDEC identified 40 hot-spots, areas containing more than 50 ppm total PCBs. The 1984 Thompson Island Pool survey re-evaluated the hot-spot locations and revised the designation to a series of over 100 polygons containing PCBs greater than 50 ppm. (The "hot-spot" term is used here only as a frame of reference to earlier studies and sediment areas previously defined as containing high levels of PCBs.) Subsequent to the 1976-1978 study, questions about the adequacy of the data were raised (NUS, 1984), as noted below:

- PCB concentrations exhibited wide spatial variability; samples along single transects not uncommonly ranged from non-detectable PCB concentrations to greater than 1,000 ppm.
- The sampling density was so low that, given the extreme spatial variability of the PCB concentrations, the accuracy of the hot-spot delineation is questionable.

- Changes in sediment deposits caused by the dynamics of the river greatly complicates comparisons between PCB concentrations in similar locations in different years.
- A major flood event that occurred in 1979 redistributed sediments significantly, again calling into the question the usefulness of the 1976-1978 sediment data, other than general purposes.
- The analytical techniques used to quantify PCBs have improved since this data was collected.

B.3.2.2 1984 NYSDEC Sampling

Methods and Procedures

In 1984, NYSDEC again undertook an extensive sediment sampling program. This effort focused on the Thompson Island Pool (M. P. Brown *et al.*, 1988b). The objective of this study was to identify areas of contaminated sediments that would be removed during the Hudson River PCB Reclamation Demonstration Project. Primarily these areas were the 20 hot-spots previously identified in the Thompson Island Pool and other areas with known or suspected high PCB concentrations.

The investigators identified 1,260 sampling locations in the approximately five-mile reach of the river. Many of these locations were determined by imposing a 125-foot triangular grid on previously defined hot-spots and areas that had PCB concentrations in excess of 50 ppm during the 1983 USEPA survey (NUS, 1984). In addition, sample locations were selected based on known or suspected sediment depositional areas, as indicated by location in the river and bathymetry measurements. Sample locations in the field were determined electronically by using a microwave locating system and generally agreed with predetermined locations.

Samples were collected by Normandeau Associates, Inc. between August 24, 1984 and November 30, 1984. In addition, 21 cores were collected during February 1-4, 1985. These later samples were taken through the ice on the river at locations that had been inaccessible by boat.

Whenever possible, the investigators collected core samples. At those locations where insufficient sediment was available for core samples, grab samples were attempted. At 80 locations, bedrock or coarse material precluded either sample method. In all, 1,016 locations throughout the Thompson Island Pool were sampled and yielded 674 grab and 408 core samples.

The depth of penetration for the core samples was fairly uniform, with an average depth of approximately 31 inches. The investigators divided core samples into sections based on a desire to define contaminated layers without introducing dilution from adjacent less contaminated layers. Sections were also chosen based on potential dredging considerations and a need to limit the number of chemical analyses.

NYSDOH and Versar, Inc. measured physical and chemical parameters of the sediments collected in this study. NYSDOH determined lengths of cores and sections, percent dry solids, dry specific weight (density) and textures, which were determined visually. Versar measured percent volatile solids and performed the gas chromatograph analyses for PCBs.

PCB Results

Versar reported PCBs as Aroclors 1242, 1254 and 1260 using the method of Webb and McCall (1973). Although the data contained a total PCB quantification, no mention is made in M. P. Brown *et al.* (1988b) of the method used to quantify or calculate this total. Examination of the data received indicates that the total was not simply the sum of the three Aroclor mixtures quantitated.

Wide variations in PCB concentrations in sediments were observed in the 1984 NYSDEC study throughout the Thompson Island Pool, even though sampling concentrated on areas of known contamination. The discrepancies between means and medians for both grabs and cores indicate that PCB concentrations have a highly skewed distribution over the area of the Thompson Island Pool. (Environmental monitoring results frequently exhibit this skewed pattern and a log-normal distribution is often a good approximation of the data.) Both grab

and core samples had significantly higher concentrations in the least chlorinated fraction that was quantified (Aroclor 1242) than in the more chlorinated fractions. Table B.3-5 provides Aroclor and total PCB summary statistics for both grab and core samples. Aroclor 1242 was the predominant Aroclor reported for these samples, with lower levels of Aroclors 1254 and 1260 also identified. On average, total PCBs in the samples were approximately 55 ppm, with maximum levels of >1,000 ppm detected in several samples.

Table B.3-6 presents the results of texture classifications determined by NYSDOH. Considering grab and core samples together, Thompson Island Pool sediments were classified most often as either gravel or fine sand, with a significant fraction of fine sand/wood chips and clay samples, particularly for core samples.

Because of their high adsorption (partition) coefficients, PCBs are generally expected to associate with the organic carbon fraction of the sediments. Although no measurements of organic carbon content were made as part of this study, organic carbon and organic matter content, which were measured frequently in the study as percent volatile solids, can be correlated. Thus, comparing PCB levels with organic matter content (volatile solids) provides a surrogate for comparing PCB concentration to sediment organic carbon content.

Figures B.3-2 through B.3-4 show the relationships between PCB concentrations and percent volatile solids within a texture classification for the three most commonly occurring textures (gravel, fine sand, fine sand/wood chips). Very little relationship appears to exist between total PCBs and volatile solids measured in the gravel texture class (Figure B.3-2). The fine sand (Figure B.3-3) and fine sand/wood chip (Figure B.3-4) categories exhibit a better correlation between PCB concentration and percent volatile solids, although percent volatile solids would still make a poor predictive measure of PCB concentration.

Comparison of 1976-78 and 1984 Studies

As mentioned previously, the 1984 study focused on the 20 hot-spot areas in the Thompson Island Pool as defined by areas exceeding 50 ppm PCBs in the 1976-1978 survey. M. P. Brown *et al.* (1988b) describe in detail the differences found between the two surveys. The PCB concentrations in sediments of the Thompson Island Pool exhibited lower concentrations in the 1984 survey than in the earlier study, as shown by the somewhat higher frequency of PCBs detected at <25 ppm (Figure B.3-5). A direct comparison of the relative frequency plot for the two studies is hindered by the fact that the 1984 survey specifically targeted potentially contaminated areas and areas with fine-grained sediments, which were thought to contain more PCBs. Thus, 1984 samples were potentially more heavily biased in a statistical sense to those areas of PCB contamination. If samples had been taken randomly from the Thompson Island Pool, the results would likely have yielded an even larger relative number of samples with lower concentrations.

Because of the different scope and sampling density of the two surveys and their different sampling and analytical methods, M. P. Brown *et al.* (1988b) indicate that direct quantitative comparisons between samples collected in similar areas are problematic. M. P. Brown *et al.* found that areas of high PCB concentration determined in the 1976-1978 survey appeared to be generally confirmed by the 1984 survey. Based on area-weighted average PCB concentrations in 138 polygon areas, these investigators calculated a total PCB mass of 23,200 kg (51,040 lb) in the top 1.5 m (59 inches) of Thompson Island Pool sediments (M.P. Brown *et al.*, 1988b). This mass estimate compares to Malcolm Pirnie's estimate in 1978 of approximately 61,000 kg (133,670 lb) for all of the Pool and approximately 48,000 kg (104,870 lb) for the 20 hot spots (Malcolm Pirnie, 1978). M. P. Brown *et al.* (1988b) offer as possible explanation for the varied estimates the differences in analytical PCB methods, depth integration/area averaging and sediment densities used.

The importance of random variability and sampling density for either the 1976-1978 or 1984 studies also complicates comparisons between the two surveys and affects mass calculation comparisons. For example, GE indicates that they have taken 30 samples from polygon 5 of the Thompson Island Pool (General Electric, John Claussen letter to USEPA, March 29, 1991). This polygon was estimated by M. P. Brown *et al.* (1988b) to have contained approximately seven percent of the PCB mass in the Thompson Island Pool based on two samples containing 39.7 and 6,587.8 ppm PCBs, yielding an average concentration of 2,437 ppm. GE has indicated that the average, based on their 30 samples in this polygon, is less than 20 ppm. (Results are not yet available to the project study team.) Although PCB mass differences in this one small area of the Thompson Island Pool do not necessarily mean that the overall conclusions of the 1984 survey are incorrect, they do suggest that:

- wide variations in PCB concentrations occur over relatively short distances;
- direct quantitative comparisons of PCB levels in samples from different years are problematic; and
- the mass and distribution of PCBs in the Upper Hudson are difficult to quantify.

B.3.2.3 Lamont-Doherty Geological Observatory Investigations

The Lamont-Doherty Geological Observatory, under contract to the NYSDEC, conducted a survey of PCB levels in the sediments, suspended matter and water column of the Upper Hudson River during 1983 and 1984 (Bopp *et al.*, 1985). The survey, a coring effort, collected 16 cores, covering the Upper Hudson from above Hudson Falls to the Albany area (Plate B.3-1), and analyzed them for radionuclides. Procedures were the same as those described in Section A.3 for the Lower Hudson. Many sections of these cores were analyzed for PCB levels, with an emphasis on homologue and congener-specific information. The investigation also involved PCB analyses of surface water samples (see Section B.3.3).

On the basis of these data, Bopp *et al.* (1985) were able to draw a number of important conclusions concerning the fate of PCBs in the Upper Hudson. In cores showing interpretable radionuclide chronologies, the occurrence of a maximum PCB concentration in sediments deposited circa 1973 (the removal of the Fort Edward Dam) could be seen in all areas of the Hudson, including the Thompson Island Pool (Figure B.3-6). This data demonstrated that the sediments of the Upper Hudson could be used to determine PCB transport history.

Analysis of Upper Hudson sediments revealed that very recent sediment deposits (1980-1983) contained PCB congeners in ratios very similar to Aroclor 1242 and 1016, whereas older sediments, typically under anaerobic conditions, showed substantially different congener ratios. In general, these older sediments contained higher levels of mono through tetrachlorobiphenyls and lower levels of the higher chlorinated congeners, relative to a standard Aroclor 1242 mixture. Bopp *et al.* (1985) concluded on the basis of these results, that under anaerobic conditions, biologically driven dechlorination must be occurring. However, little or no dechlorination was occurring under the aerobic conditions of the uppermost sediment layers.

B.3.2.4 Other Studies

1983 USEPA Study

In August 1983, USEPA conducted a limited study to collect samples from locations that had been sampled in 1976-1978 (NUS, 1984). Sixty-six samples, of which 54 were core and 12 were grab samples, were collected within a nine-mile stretch of the river south of Rogers Island, including the Thompson Island Pool. Forty-two samples were collected from within or on the border of previously determined hot-spots. The results of this study tended to show that areas of high PCB concentrations in 1976-1978 exhibited high concentrations in 1983 as well. Nevertheless, direct comparisons between samples taken within 50 - 100 feet of each other during the two surveys indicated that variations by two orders of magnitude were not uncommon (NUS, 1984). In general, the concentrations observed in 1976-1978 were greater than those seen in 1983 at corresponding locations.

GE 1989 Baseline Studies (Remnant Remediation Project)

As part of the Remnant Remediation Project, General Electric conducted baseline pre-remediation sediment monitoring. Sediment samples were collected at five locations in the vicinity of the remnants: one location near Rogers Island; one location far upstream; one location between the remnants and Bakers Falls; and two downstream locations near Lock 6 and Waterford. Median PCB concentrations in river sediments in the remnant areas ranged from 0.47 ppm to 34 ppm, with a maximum of 99 ppm (Table B.3-7). Downstream samples contained median PCB concentrations of 0.64 to 2.1 ppm, whereas the control location had median PCB levels (seven samples) of 0.11 ppm. The sample location between Bakers Falls and the remnants had a median PCB concentration of 1.4 ppm. With the exception of the two downstream locations, PCBs were detected in all samples. The chromatograms were compared against Aroclor mixtures 1221, 1232, 1016, 1242, 1248, 1254 and 1260; Aroclor mixtures in the samples were reported to be Aroclors 1242 and/or 1254.

GE 1990 Sediment Sampling (Bioremediation Investigations)

General Electric has been conducting extensive research on biological dechlorination and/or degradation processes occurring within the river, which may have altered the composition of the PCB Aroclor patterns within the sediments. In conjunction with these studies, GE has recently collected samples from selected areas of the Upper Hudson for more detailed evaluation. General Electric provided preliminary results of their sediment sampling activities (Claussen, 1991b).

Harza Engineering, GE's contractor, collected 103 core samples from 12 hot-spots during 1990 and reported 275 PCB analyses. From three to eight cores were collected at most locations, with the exception of GE's H-7 location where 62 cores on a 12 x 12 foot grid were sampled. Samples were analyzed for PCB homologue groups and five Aroclors (1221, 1242, 1254, 1260 and 1268). The results of this sampling are summarized in Table B.3-8. With the exception of H-7 and Location 4, the median PCB concentration at the four other locations

within the Thompson Island Pool exceeded 100 ppm. Median PCB concentrations for the aforementioned two locations and the locations downstream of the Thompson Island Pool were less than 100 ppm.

Coincident samples and PCB measurements from the same hot-spots in the NYSDEC 1976-78, USEPA 1983, and GE 1990 samples are available for only six locations. The average PCB concentrations for each of these three surveys are summarized in the lower portion of Table B.3-8. In four out of six locations, the GE samples indicate average PCB levels above both the 1976-1978 and 1983 values; the remaining two locations show 1990 PCB levels lower than the 1976-1978 and/or 1983 results. Because of the very small sample sizes, few coincident locations and difficulty in determining whether these samples represent similar sediment zones over time, these results are inadequate to suggest a clear trend. Qualitatively, the results document the continued presence of PCBs in areas originally defined in 1976-1978 to be contaminated.

B.3.2.5 Other Chemicals in Sediments

In addition to PCBs in river sediments, other chemicals, particularly heavy metals, have been measured during 1976-1978 (Tofflemire and Quinn, 1979), 1984 Thompson Island Pool study (M. P. Brown *et al.*, 1988b) and by other investigators. Lead, cadmium, zinc, chromium, mercury and other metals have been measured. M.P. Brown *et al.* (1988b) indicate that anthropogenic sources, including a pigment manufacturer in Glens Falls, elevate lead, cadmium and chromium in river sediments above their naturally occurring levels. Based on their 1984 study, M.P. Brown *et al.* (1988b) reported mean metals concentrations in sediments for lead (217 µg/g), cadmium (21.6 µg/g), chromium (475 µg/g) and mercury (1.96 µg/g) along with several other chemicals. M. P. Brown *et al.* found that although lead and cadmium are the two metals most frequently found in sediments, standard leaching tests, *e.g.*, EP Toxicity, suggest they are not readily leachable.

Relatively few sediment samples have been tested for other organic priority pollutants. Four sections of two cores collected in 1983 by Dr. Richard Bopp from River Miles 188.5 and 191.1 (Thompson Island Pool) were submitted to NYSDOH and analyzed for dioxin and dibenzofurans. Six sediment samples collected in 1987 from three hot-spots were analyzed for dioxins, dibenzofurans, volatile and semi-volatile organics and pesticides (M. P. Brown *et al.*, 1988). With the exception of dibenzofurans, none of the other organic parameters were detected in the 1987 samples.

As reported by M. P. Brown *et al.* (1988b) tetrachlorodibenzo-p-dioxin (TCDD) and tetrachlorodibenzofuran (TCDF) as well as their 2,3,7,8- isomers were detected at less than part per billion levels in the 1983 samples. Total TCDD in these 1983 samples ranged from non-detected to 0.135 ppb; total TCDFs ranged from non-detected to 0.731 ppb. In two of the six 1987 samples, total TCDFs were detected at <0.2 ppb levels; TCDDs were not detected in the 1987 samples; detection limits ranged from 0.012 - 0.058 ppb. M. P. Brown *et al.* indicate that possible sources of the TCDFs in sediment include residual fall-out from coal and wood combustion, discharge from wood processing plants (by-product of chlorophenol pyrolysis) and discharge of chemical-wastes containing TCDDs and TCDFs as trace contaminants. Industrial PCB mixtures are known to contain trace levels of TCDFs.

B.3.2.6 Discussion

The study team encountered some difficulty in matching the contents of the data entries in the TAMS/Gradient database, especially the 1976-1978 sediment data, with data summaries provided in previous reports (Tables B.3-2 and B.3-3). The 1976-1978 raw data in printed form did not contain identification designations cross-referenced to date and sample location and no report containing such a cross-referenced summary was found. Sample identification numbers are shown on a marked-up copy of the 1977 Normandeau Associates, Inc. map in the NYSDEC offices, but do not cross-reference dates of the samples or laboratory identification numbers. The team could assess the completeness of the data set only by comparing sample dates and river location associated with the samples

with summaries in published reports. As shown in Table B.3-2, these comparisons identify some inconsistencies between the data in the TAMS/Gradient database and the number of samples reported by Tofflemire and Quinn (1979) in NYSDEC Technical Paper No. 56. For example, the TAMS/Gradient database contains 254 grab samples and 21 core samples reportedly collected in 1976, whereas Tofflemire and Quinn report 24 cores and 80 unspecified sediment samples for 1976. For 1977, the TAMS/Gradient database contains 446 grab and 246 core samples compared to 692 grab and 208 core samples reported by Tofflemire and Quinn (1979). For all samples collected between 1976 and 1978, the database contains 1,092 samples compared to 1,404 indicated by Tofflemire and Quinn.

Differences in the overall number of samples may be accounted for approximately by noting that:

- approximately 202 of the 672 summer 1977 grab samples taken by Normandeau Associates, Inc. (NAI) were not analyzed for PCBs and were not provided in the printed data summaries supplied for the TAMS/Gradient database;
- 200 spring 1978 remnant samples from Malcolm Pirnie, Inc. (MPI) were not provided for use in the TAMS/Gradient database.

If these 402 samples are subtracted from the Tofflemire and Quinn total of 1,404, there would be approximately 1,000 samples, more closely approximating the 1,092 samples in the TAMS/Gradient database. A comparison of the number of samples by river mile (Table B.3-3) also indicates differences in total numbers of samples reported by Tofflemire and Quinn and the data in the database. Because 15 years have passed since the 1976-1978 samples, their use for identifying precise locations and concentrations of contaminated sediments is limited.

The data summaries contained inconsistencies within samples and between samples. For example, the sums of individual core lengths did not always match the total core length; for approximately 3 percent of the samples, the data fields were incorrect; approximately 10 percent of the samples had no information

about northing-easting coordinates. The team could not evaluate independently the reason for the discrepancies nor the accuracy of the original data archival process/data summaries. These discrepancies in the sediment database are not considered to be significant at this time and an effort to resolve them will continue during the course of the project.

Differences between the TAMS/Gradient database and the results reported by M. P. Brown *et al.* (1988b) for the 1984 Thompson Island Pool survey appear to be slight. The database contains 1,141 samples, whereas M. P. Brown *et al.* report 1,205 for this survey, a difference of ~5 percent in total samples.

B.3.3 Surface Water Monitoring

Numerous surface water monitoring stations along the Upper Hudson are maintained by the USGS. These stations have monitored flow, suspended sediment, PCBs and other water quality parameters. The USGS data, obtained from WATSTORE and the Albany USGS office, provide the longest and most comprehensive record of surface water data for the Upper Hudson.

B.3.3.1 USGS Flow Records

The USGS has collected river discharge (flow) and water quality data at various points along the Upper Hudson River (Plate B.1-1). The USGS records of the monitoring stations located on the Hudson between Hadley, well above Fort Edward, and Green Island, below the confluence with the Mohawk River at Troy, were obtained for use in this investigation.

The majority of the USGS flow monitoring stations on the Upper Hudson have periods of record beginning in the 1970s, although continuous monitoring is available at Hadley since 1921. A lack of widespread flow measurements for earlier periods presents difficulties in analyzing the longer-term flow regime and flood probabilities. In particular, the flow record at Fort Edward, at the upper end of the Thompson Island Pool, commences only in 1976. No USGS monitoring is available at the Thompson Island Dam. Barge Canal stage data are

available at the guard gate at Crockers Reef (Gauge #118) in the Hudson River/Champlain Canal approximately parallel to the Thompson Island Dam, at the lower end of the pool. At the northern end of the Thompson Island Pool, Barge Canal stage data are available below Lock 7 (Gauge #119). These gauges report water elevations in reference to the Barge Canal datum. They have not been calibrated to river discharges and provide only qualitative data regarding flood discharges.

In order to extend the record of flow data for the Upper Hudson, it is necessary to move upstream to the confluence of the Hudson and Sacandaga Rivers, near Hadley (see Plate B.1-1). A monitoring station has been maintained on the Hudson at Hadley since July 1921. The Sacandaga River, a major tributary entering just below Hadley, has been monitored since 1907 at Stewarts Bridge near its confluence. By adding these two stations, USGS provides an estimate of the flow in the Hudson below the confluence with the Sacandaga. Between this point and Fort Edward there are several dams, but there are few additional tributaries. The drainage area above Fort Edward is 2,817 square miles, while that of the Hudson River at Hadley plus Sacandaga River at Stewarts Bridge is 2,719 square miles, representing only a 3.6 percent increase in contributing area. Estimates of flow in the Hudson below the Sacandaga, thus, provide an accurate estimate of the magnitude of flow at Fort Edward.

The daily average flow value records for USGS Upper Hudson stations are summarized in Table B.3-9, from upstream to downstream. USGS flow monitoring is available since 1976 only at the upstream end of the Thompson Island Pool (Fort Edward at Rogers Island). The closest functioning USGS monitoring station downstream is at Stillwater, which is 26 miles and three dams south. Thus, Fort Edward monitoring is most informative of hydraulic conditions in the Thompson Island Pool. Mean daily flow at Fort Edward is 5,244 cfs; daily flows range from 652 cfs to 34,100 cfs for the period of record. Additional inputs from tributaries and runoff increase the average daily flow to 7,933 cfs at Waterford and 13,642 cfs at Green Island, below the confluence of the Mohawk River.

Figures B.3-7a and B.3-7b display the daily average flows for 1973-1990. As Fort Edward monitoring commenced in water year 1977, the 1973-1976 flows are estimated from the calculated flows below Hadley, representing 97 percent of the contributing watershed area at Rogers Island. This record reveals the presence of several major flood events, which were associated with mass erosion of the remnant deposits. These events occurred on: April 2, 1976, when a daily average flow of 39,340 cfs was reported below Sacandaga River; April 29, 1979, when a flow of 31,700 cfs was reported at Rogers Island; and May 2, 1983, when 32,600 cfs was reported at Fort Edward. During these flood events flows were even higher for shorter time periods (peak flows). The maximum peak flows monitored at Fort Edward since December 1976 are 34,000 cfs on April 29, 1979 and 35,200 cfs on May 3, 1983.

The daily flows show evidence of a strong weekly periodicity. The seven-day cycle is the result of regulation of the Sacandaga Reservoir to supply power plants during the week, while maintaining the weekend recreational pool in Sacandaga Lake.

B.3.3.2 Suspended Sediments Monitoring

Information on time trends in suspended sediment data as well as the relationship between sediment and discharge is provided by USGS monitoring stations (Plate B.1-1). Several water quality stations were established on the Upper Hudson in 1969, but measurements of suspended sediment did not commence until 1975. Monitoring is not continuous or on a set schedule and there has been a tendency to focus on spring flood periods, with little data available for the winter months. Lack of a more extensive database and of regular time series creates difficulties in analyzing sediment data as well as other water quality parameters.

Summary statistics on the USGS suspended sediment monitoring for stations between Fort Edward and Waterford are given in Table B.3-10. The median suspended sediment concentration in the Upper Hudson above the confluence of the Mohawk ranges from 4 to 12 mg/l and increases downstream. Relationships between

suspended sediment levels and river flow are discussed in B.4.

B.3.3.3 USGS PCB Monitoring

Methods and Procedures

Regular monitoring of PCBs in the water column in the Upper Hudson was instituted by the USGS in late 1975 at Waterford and expanded to other upstream stations in 1977. Most other sampling programs, discussed later in this section, have been of short term duration. A recent search of the STORET database reveals that limited water-column PCB measurements are also available for some of the tributary rivers to the Upper Hudson. These data have not been reviewed. The USGS data are, thus, the primary source of time series information indicative of trends in water-column PCB concentrations.

USGS observations of PCB concentrations in the water column have been made at most of the same water quality stations as for sediment data (Plate B.1-1). Data sets of significant size are available at Fort Edward (River Mile 194.5), Schuylerville (River Mile 181), Stillwater (River Mile 168) and Waterford (River Mile 156.5), with a limited record at Fort Miller (River Mile 187). In addition background samples are taken upstream at Glens Falls (River Mile 200).

The original purpose of the USGS monitoring was to gather several years of data on PCB concentrations prior to removal of contaminated sediments (Schroeder and Barnes, 1983). Although this dredging plan has been delayed, monitoring has continued. Data are now available on WATSTORE, a USGS-maintained computerized database, through the end of water year 1989 (September 1989). Data for water year 1990 were collected by the USGS and have been turned over to NYSDEC for analysis, but the results are not yet available.

Methods of data collection and analysis are summarized in Turk and Troutman (1981) and Schroeder and Barnes (1983). According to the latter source, samples from the Upper Hudson were collected from bridges using depth-integrating samplers. The sampler held a wide-mouth glass bottle, which was lowered and

raised through the water column to obtain a depth-integrated sample (about 1 liter) for PCB analysis.

The USGS National Water Quality Laboratory in Doraville, GA performed the PCB analyses. Comparison was made to standard Aroclor mixtures. Results, however, were reported as total PCBs. Schroeder and Barnes (1983) reported that PCBs in the Hudson were "almost always in the composition range from [Aroclor] 1232 to 1248," but recognized that natural processes had likely altered the congener composition of the original Aroclor mixture.

Although the USGS laboratory reports a theoretical detection limit of 0.01 µg/*l* through water year 1983, the practical quantitation limit was considered to be 0.1 µg/*l*, because of the small size of the water sample (Bopp *et al.*, 1985). Data for this period recorded on WATSTORE contain both values entered as 0 and values coded as <0.10 µg/*l*. Apparently these are both intended to represent non-detects at the 0.1 detection level and the inconsistency is unintentional (Rogers, pers. comm., 1991). With water year 1984, the practical detection limit was lowered to 0.01 µg/*l*. Nevertheless, the 1984 and 1985 data are reported on WATSTORE *as if* they adhere to the previous detection limit of 0.1 µg/*l*. In 1986, the detection limit began to be reported as 0.01 µg/*l* in WATSTORE.

PCB Results

The USGS monitoring station at Glens Falls provided upstream background levels of PCBs in the Hudson for 1977-1983. Of 45 observations for total PCBs, only two had detectable levels of PCBs. These observations occurred on December 5, 1978 and September 28, 1980 and were both reported as 0.1 µg/*l*.

Summary statistics for the USGS monitoring of total PCBs in the water column between Fort Edward and Waterford are given in Table B.3-11. A remarkable fact evident from this table is that there is rather little variation in measured PCB concentration by river mile between Schuylerville and Waterford. At all stations, PCBs were detected in more than 60 percent of the water samples, with detection frequencies ranging from 63 percent to 89 percent. Average PCB

concentrations below Fort Miller range from a high of 0.29 µg/l at Stillwater to 0.23 µg/l at Waterford. Averages at Fort Miller are not directly comparable to other stations because of the short period of record. The average at Fort Edward, inflated by one very high measurement (77 µg/l), was calculated in Table B.3-11 by omitting this outlier. This procedure yields a long-term average approximately half of that observed downstream. Concentrations observed at Schuylerville, Stillwater and Waterford are approximately constant, although the average at Waterford is lower than the up-river values, primarily because of dilution from the Hoosic River. Indeed, the average concentration at Stillwater (0.29 µg/l) is slightly higher than that at the Schuylerville station upstream (0.26 µg/l). This finding suggests that there may be relatively little loss of water-column PCBs during transit in the Upper Hudson.

At the Green Island station, downstream of the confluence of the Mohawk River, the contributing watershed area is nearly double that at Waterford and PCB concentrations are correspondingly diluted. Twelve samples were analyzed for total PCBs between 1978 and 1985 and all were non-detects at the 0.1 µg/l level. No PCB measurements have been reported at Green Island since March 1985.

Figures B.3-8 through B.3-11 show the time series of PCB observations at Fort Edward, Schuylerville, Stillwater and Waterford, respectively. Observations reported as non-detect (or zero) are plotted at the detection limit (0.1 µg/l through September 1986 and 0.01 µg/l thereafter). While much of the variability observed near the detection limit may represent analytical noise, there is a clear similarity apparent between the PCB time-series plots at Schuylerville, Stillwater and Waterford (B.3-9 through B.3-11), particularly in the marked response to the 1979 spring flood. Despite the fact that samples were taken at somewhat erratic intervals, field personnel seem to have frequently visited each Upper Hudson station in succession, so that many samples for the whole reach, while not contemporaneous, are close together in time. All three stations are below the Thompson Island Pool. Although the response to floods appears somewhat different at Fort Edward, located below the remnant deposits but above the so-called hot-spots in the Thompson Island Pool, the 1979 event stands out.

Also notable in these time series is a general decline in water-column PCB levels from about 1979-1986. The question of whether there has been any genuine trend in PCB loading to water over time, or whether the apparent year to year trends are actually due to variability in the hydrologic regime, is discussed in Section B.4.

Current Full-Year Average PCB Concentrations in Water Column

Estimates of current average concentration of total PCBs in the water column are needed for the assessment of potential baseline health risks and other environmental impacts. As noted previously, seasonal variability and lack of continuous sampling confound the estimation of this average.

It is common practice when working with data in time series containing non-detects to develop an average based on treating the non-detects as if they were equal to one half of the detection limit. A more sophisticated way to approach this question is to use the Adjusted Log Normal Maximum Likelihood method of Cohn (1988), which overcomes the bias in the sample collection and variance in the data. Another method for including non-detects in the estimate of the mean, which however does not address the problem of sampling bias, is to use a log-probit analysis (Helsel and Cohn, 1988). Under the assumption that the logs of the data are normally distributed, they will fall on a straight line when plotted on a probit (probability) scale. The log-probit analysis essentially uses regression to extend this line past the detection limit to predict the values of the non-detect samples predicted by the observable part of the distribution.

In order to examine the sensitivity of the mean to the non-detects, the above three methods of calculation were used here: (1) simple mean with non-detects at 1/2 the detection limit; (2) the Adjusted Maximum Likelihood method; and (3) log-probit analysis. Results of all three methods and the upper 95 percent confidence limit on the estimates of the means, which is used in the risk analysis, are shown in Table B.3-12.

Little difference in estimates of the mean and the 95 percent upper confidence limit on the mean is produced by any of the three methods. Mean PCB levels in recent 1986-89 water samples are on the order of 0.05 µg/*l* at Fort Edward and drop to approximately 0.03 µg/*l* at Waterford. Upper 95 percent confidence limits on these means are not much higher, approximately 0.075 µg/*l* at Fort Edward and 0.035 µg/*l* at Waterford.

Summer Average PCB Concentrations

Average water-column PCB concentrations can be calculated from monitoring data based on either whole-year monitoring, including flood periods, or based on low-flow or seasonal monitoring only. Assessing average concentrations for the summer period, after the spring floods, is likely to be of greatest interest for assessing biological impact. This period has maximum biological production and is also the season during which most of the fish samples have been collected. There is also evidence to suggest that spring flood PCBs are largely sorbed on sediment particles, whereas concentrations associated with low flows are primarily dissolved or sorbed to very fine particles, *i.e.*, pass a 0.45-µm filter. Thus, they are more readily available to enter the food chain (Bopp *et al.*, 1985).

Average PCB concentrations during the summer (June-September) have been calculated. In doing so, the presence of many non-detects among the samples was addressed. The robust log-probit analysis method (Helsel and Cohn, 1988) was used to estimate averages in the presence of non-detects. No systematic bias toward higher concentration events is expected to apply to the summer observations, although this may be the case for spring observations. In 1986 there were no summer observations at Stillwater or Waterford, while summer observations at Schuylerville in this year as well as at Waterford in 1985 were all less than or equal to the detection limit of 0.1 µg/*l*. In the latter case the log-probit method cannot be used and the average has been arbitrarily set to one-half the detection limit. The calculated summer average PCB concentrations are shown in Figure B.3-12 and summarized in Table B.3-13. Recent 1988-1989 summer average PCB concentrations are fairly uniform for all locations and are on the order of

0.03 - 0.04 ug/l. Summer average concentrations at Fort Edward tend to be less than or equal to those downstream, whereas for the full year concentrations, including spring runoff, the average concentration at Fort Edward is higher than that found downstream.

B.3.3.4 Other Sources of Water Column Data

Waterford Treatment Plant Data

The City of Waterford operates a water works serving a population of approximately 12,000 persons in the Towns of Waterford and Halfmoon and the Village of Waterford. This is the first water treatment facility downstream of Fort Edward drawing water from the Hudson. In 1975, when the USGS began collecting PCB data in the river at Waterford, they also began collecting raw water input and finished water output data at the Waterford treatment plant, in cooperation with the Board of Water Commissioners of the Town of Waterford and the NYSDEC (Schroeder and Barnes, 1983b). The water for the treatment plant is drawn from a location 0.5 km upstream of the US Highway 4 bridge, where Hudson River water samples are also taken.

Data collected in cooperation with the USGS run through the end of water year 1983. In addition, approximately bimonthly data for November 1983 - February 1985 and March 1987 - October 1989 were available from the Waterford Water Works (Metcalf & Eddy, 1990). The data through 1983 were often collected concurrently with samples from the Hudson River at Waterford and can be used to provide a check on that data. Since September 1982, no PCBs have been reported above the detection limit (generally 0.1 µg/*l*), in either raw intake water or treated water.

Lamont-Doherty Study of 1983

A detailed study of PCB transport in the Upper Hudson, conducted by researchers of the Lamont-Doherty Geological Observatory in 1983 (Bopp *et al.*, 1985), involved an investigation of spring/summer 1983 PCB transport in the Upper

Hudson, which was a period of relatively high flows. The Lamont-Doherty study included the collection of data not available from USGS sampling. In addition to sediment cores, this study included 20 large-volume filter samples of suspended matter and fifteen 9-20 liter unfiltered samples containing water and suspended matter, collected from Troy to Glens Falls. Unlike USGS monitoring, detailed component analysis was undertaken for these samples. The data collected were used to form an empirical transport model, which used as input USGS measures of suspended sediment and flow, to predict total PCB load.

PCBs in samples of suspended matter were found to match standard Aroclor mixtures, *e.g.*, Aroclor 1242, reasonably well. Samples taken during a high-flow period were found to have significantly higher PCB levels (6.33 ppm) than those taken during a low-flow period (0.69 ppm). Some of the drop in PCB levels at locations downstream of Fort Edward was attributed to dilution from tributaries joining the Hudson.

Water samples were filtered and both filtrate and particles were tested for PCBs. Comparisons of water versus suspended matter PCB concentrations were made to derive a distribution coefficient. An inconsistency was found in that the concentration of PCBs on suspended matter filtered from the 9-20 liter samples was generally two to four times higher than the levels from the large-volume filter samples collected at the same locations on the river. This discrepancy could possibly be accounted for by an equilibration between water and suspended matter during storage of water samples prior to filtration and testing. Another possible explanation was the difference in filter size used to collect the particulate sample (1.2 microns) and to separate the water samples (0.7 micron).

Seasonal variations in concentrations of PCBs in water were found. Pre-spring runoff showed the lowest PCB levels. Summer samples showed higher PCB concentrations, accounted for by boat traffic and increased use of locks, which would resuspend sediments.

Bopp *et al.* (1985) were able to derive *in situ* partition coefficients for PCBs on a quasi-homologue basis, based on packed column analysis. These analyses also indicated the possibility that water-column PCB distributions may not reflect equilibrium conditions and that dissolved phase PCB concentrations may be higher than those predicted by equilibrium. The homologue distribution of PCBs on suspended matter was readily interpreted as Aroclor 1242-like, similar to PCBs in the aerobic sediment layers. The dissolved phase homologue distributions were not as readily explained, although preferential partitioning of the lower chlorinated homologues to the dissolved phase appeared to be a likely possibility, *i.e.*, lower chlorinated homologues have lower partition coefficients and, therefore, higher levels in the dissolved phase. The greater volatility of the lower chlorinated PCBs and their production in the anaerobic sediment layers could possibly confound this interpretation. The general homologue pattern agreement between water column and surface sediments led Bopp *et al.* (1985) to conclude that little or no release of PCBs from the anaerobic sediments was occurring on a substantive basis in comparison to the mixing and resuspension of the surficial sediments.

Bopp *et al.* concluded that PCB transport is tied to suspended matter transport, with the majority of PCB transport to the Lower Hudson occurring during the 10 to 20 days per year of major sediment transport.

NYSDOH Water Column PCBs

As part of their macroinvertebrate sampling program, NYSDOH also collected water samples and analyzed them for PCBs. This and other recent data received from Dr. Bush at NYSDOH, require additional evaluation. The macroinvertebrate studies are discussed in a later section.

Remnant Deposit Containment Monitoring Program

As part of remedial activities at the PCB remnant deposits at Fort Edward, General Electric is conducting a baseline environmental monitoring program, which will continue through and follow the in-place containment of Remnant Deposits 2,

3, 4 and 5 in accordance with the Administrative Order on Consent II CERCLA-90224. The baseline monitoring includes sampling of many environmental matrices, including water column PCB concentrations. Results of the first phase of this effort cover August through December 1989 (Harza Engineering, 1990).

During the baseline monitoring, samples were taken weekly or biweekly at ten water quality stations. Two of these stations are upstream of Fort Edward, one in the Sherman Island Pool and one just below Bakers Falls, while five are in the area of the remnant deposits at Fort Edward above Rogers Island. Station E-5 was located downstream of the Route 197 bridge on Rogers Island and is just below the USGS monitoring station. The remaining two stations were at Channel Marker 175, below Fort Miller Dam and Lock 6 (E-6), and at Channel Marker 13, two miles north of a NYSDEC boat ramp at the Erie Canal and Hudson River confluence, near Waterford (E-7). Raw water samples were analyzed for PCBs with an approximate analytical detection limit of 0.1 µg/*l*. In addition, dialysis bags, filled with 4 ml of hexane to concentrate PCBs, were suspended in the water column and analyzed biweekly. As the concentration factor for the dialysis bags is unknown, they can be used to indicate qualitatively the presence of PCBs, but not ambient concentrations.

The 1989 monitoring program unfortunately missed the spring runoff period. All raw water samples were reported to be below the detection limit of 0.1 µg/*l*. This result is consistent with USGS monitoring data for water year 1989 at Fort Edward, which had a lower detection limit and showed detectable concentrations in the 0.01-0.1 µg/*l* range. During the same period the dialysis bag concentrators occasionally detected PCBs at all stations except the uppermost (Sherman Island). All the detects were identified as Aroclor 1242.

B.3.4 Fish and Other Aquatic Biota

Substantial declines in average PCB burden in Upper Hudson fish were observed in the years after 1978 (Sloan *et al.*, 1983, 1984; M. P. Brown *et al.*, 1985). Analysis of the data reveals that these declines in concentration have proceeded at a slower rate in more recent years. It is unclear to what degree

the abnormally low spring floods of the 1980s have affected PCB levels and may be responsible for the observed declines. It does appear that total PCB concentrations in fish on a lipid (fat)-basis can be closely predicted from summer average water column PCB concentrations.

Because PCBs are typically stored (bioaccumulate) in fatty (lipid) tissues, it is sometimes useful to normalize the PCB levels in fish and express them on a lipid basis, *i.e.*, PCB content in fish expressed as ug-PCB/g-fish lipid. Whether or not normalized to lipid content, levels of the higher chlorinated congeners in largemouth bass were approximately stable from 1981-1988. Reported Aroclor 1016 levels, representing less chlorinated congeners, appear to have continued to decline for all species during this time period. Given the slow rate of reduction of Aroclor 1254, it is unclear when or if natural processes will reduce the PCB burden in fish to acceptable levels. Furthermore, potential changes in PCB levels in the water column and sediment caused by possible scour and resuspension of sediments would likely cause at least temporary increases in PCB levels in fish and aquatic biota.

In addition to fish data, some monitoring of invertebrates is also available for the Hudson. From 1976-1985 multiplate samples and chironomid larvae have been monitored by NYSDOH. These data are discussed at the end of this section, following the discussion of the fish monitoring program.

B.3.4.1 Fish Sampling

Data on concentrations of PCBs in Upper Hudson River fish collected by NYSDEC between 1975 and 1988 were used in this study. While over 30 species of fish are represented in the data, the majority (75 percent) of the samples are from half a dozen species including striped bass, largemouth bass, brown bullhead, pumpkinseed, American shad, and American eel. Approximately two-thirds of the samples tested were standard fillet samples.

Samples Prior to 1975

While PCBs are known to have been discharged into the Upper Hudson River since the 1940s, no testing for PCBs in fish is known to have been undertaken before 1970. In that year, a nationwide survey of chemical pollutants in game fish conducted by a popular magazine included a sample of spawning striped bass from the Hudson estuary in which 4.5 to 5 ppm PCBs in flesh and 11 to 12 ppm PCBs in eggs were reported (Boyle, 1970). NYSDEC had been analyzing fish for DDT and other pesticides statewide since the early 1960's. In 1971, NYSDEC added PCBs to their analyses, although no results were released publicly until 1975 (Sanders, 1989).

Fish data collected and analyzed for PCBs in the 1970-74 period are summarized by Spagnoli and Skinner (1977). These 1970-74 Hudson River samples include one smallmouth bass collected at Warrensburg (Upper Hudson) and 146 fish from 11 species collected below the Federal Dam (Lower Hudson). The highest observed concentration from below the Federal Dam was in a largemouth bass, reported as containing 53.81 ppm wet weight total PCBs in the 1970-72 period. This sample was taken prior to the removal of the Fort Edward Dam.

In August 1974, a USEPA team obtained water, sediment and fish samples from upstream and downstream of the GE discharge at Fort Edward. A sample of 42 shiner minnows from below the GE discharge showed an average concentration of 78 µg/g (ppm) PCB as Aroclor 1242, while one rock bass was reported with 342 µg/g (Nadeau and Davis, 1974). It should be noted that samples collected at control Station 0 above Bakers Falls were not non-detects. PCB levels at Station 0 were reported as 7.0 µg/gm in shiner minnows and as 17.0 µg/gm in yellow perch. The latter level is higher than the average for all NYSDEC yellow perch samples in the Upper Hudson. Few other samples have been reported from Bakers Falls.

Samples 1975-1976

Following the USEPA investigation, NYSDEC undertook more detailed monitoring of PCBs in fish from both the Upper and Lower Hudson. A total of 440 Hudson River (Upper and Lower) fish samples were analyzed in 1975-1976, the results of which NYSDEC provided for incorporation into the *TAMS/Gradient* database.

The 1975-1976 fish collections were made by regional USFWS personnel who were instructed on specific species and sizes of fish desired, location of stations and time tables for collection. Target species for the Hudson included smallmouth bass (*Micropterus dolomieni*), largemouth bass (*Micropterus salmoides*), brown bullhead (*Ictalurus nebulosus*), goldfish (*Carassius auratus*), white sucker (*Catastomus commersoni*), striped bass (*Morone saxatilis*) and various other estuarine species. Other species were occasionally obtained as available. Attempts were made to sample small, medium (minimum legal) and large representatives of each species.

Analyses were conducted by several different state laboratories, apparently using the methodology of Bush and Lo (1973) and reported against standards for Aroclor 1242 *or* 1016 and Aroclor 1254. Aroclor 1221 was not analyzed. The nominal detection limit of the method was 0.01 ppm, although some of the labs reported results only as low as 0.1 ppm.

Samples 1977-1988

In 1977-1979 NYSDEC monitoring methods were refined and standardized. Collection has continued to the present, although no results after 1988 are available. As of 1988, NYSDEC changed the frequency of sampling from yearly to every other year. Samples from 1990 were collected, but PCB analyses have not been completed. NYSDEC provided data covering the period of 1977 through 1988, which contains 7,373 Upper and Lower Hudson fish or fish composite analyses. These samples have been collected on a regular basis, with the intent of sampling given species at predetermined locations within two weeks of a specified date in

order to minimize potential seasonal effect. Records are very limited for 1981 and 1987. Table B.3-14 provides a summary of the total number of fish sampled for the Upper Hudson from 1975-1988.

Information in the fish database, in addition to chemical analyses, usually includes species, method of preparation, weight, length, percent lipid and, if determined, sex and age. In addition to a descriptive sampling location, river mile numbers and geographic designations are associated with each sample.

Sample collection, preparation and analytical methods are described in Armstrong and Sloan (1981). The desired sample size for each species collected was 30 individuals, although the availability of fish did limit sample size in some cases. A single fish sample generally consisted of a composite of one to three fish. For fish longer than 150 mm, standard fillets of whole sides of scaled fish were prepared for analysis. For brown bullhead samples, the skin was removed from the fillet. Samples of fish shorter than 150 mm were analyzed whole with head and viscera removed.

Aroclor concentrations in fish were determined by comparison to commercial Aroclor standards. This method was not able to distinguish between Aroclors 1242 and 1016. The detection limit for each Aroclor tested was 0.1 ppm. For each sample the percent lipid was determined as the percent by weight of tissue soluble in petroleum ether.

PCB Levels in Fish

This section summarizes the Upper Hudson PCB monitoring results for fish. Additional statistical analyses are presented in Section B.4. Samples collected at River Mile 153, just below the Federal Dam, are included with those of the Upper Hudson, since they represent a resident, freshwater, rather than estuarine, population, which is exposed to PCBs transported over the dam.

Although over 30 fish species have been sampled in the Hudson River, a few of these species account for the majority of the samples collected. These species are pumpkinseed, largemouth bass, brown bullhead, goldfish (carp), white perch and yellow perch. Overall average Aroclor levels for these species in River Miles 153 to 195 of the Upper Hudson for 1975-1988 are provided in Table B.3-15. Aroclors 1016 and 1254 are the dominant PCB mixtures reported; Aroclor 1221 represents the smallest PCB fraction. Overall, the highest average PCB concentrations have been found in goldfish (carp) with 1975-1988 average Aroclor levels of 32.0 ppm (Aroclor 1254) and 91.6 ppm (Aroclor 1016). That goldfish have the highest average PCB levels is not altogether surprising as they also have the highest lipid content (9.3 percent). The average 1975-1988 Aroclor levels in largemouth bass, perch and bullhead are <30 ppm, although total PCB levels in individual largemouth bass have been reported as high as 370 ppm.

The average of total PCB levels in all fish for recent years (1986-1988) also shown in Table B.3-15, is 10.9 ppm. The upper 95 percent confidence found on this mean, used in the preliminary human health risk assessment (B.6), is 12.0 ppm.

Tables B.3-16 through B.3-18 provide summary statistics by river mile sampled for largemouth bass, pumpkinseed, and brown bullhead, respectively. Recent data for 1986-88 show that median PCB levels in fish range from 1 to 30 ppm depending on species and river mile. Additionally, the variability of PCBs in fish from a given location and year, as measured by the standard error of the estimate of the mean, has dropped as the PCB levels have declined.

Figure B.3-13, which plots mean, total PCB levels in brown bullhead (skinned standard fillets) as a function of year and river mile, provides an overall perspective of the general decline of PCB levels over time and by river mile. PCB levels in the vicinity of River Mile 190 are the highest, but samples from only a few years are available here.

Surprisingly little fish data are available from the Thompson Island Pool area (River Mile 188-94). Regular monitoring there began in the latter half of the 1980s. The most complete data set available for the Upper Hudson above the Federal Dam is that at River Mile 175. Data here begin in 1975 and continue, with some interruptions, through 1988. Figure B.3-14 summarizes the trends in mean Aroclor levels for four fish species at this location as ppm wet weight Aroclor 1254 and Aroclor 1016. Aroclor 1016 levels are shown on a log scale, as they were extremely high for certain species in the 1977-78 time period. Aroclor 1221 was quantitated also, but generally at very low levels compared to the other two Aroclors. A notable trend is that during the 1977-78 period the ratio of Aroclor 1016/1254 levels in fish are elevated, *i.e.*, the lower chlorinated congeners (1016) dominated the higher chlorinated congeners (1254) during the late 1970s. This elevation may be attributable to scour of dechlorinated sediments from depth. Another possible explanation for the shift from Aroclor 1016 dominance to Aroclor 1254 in recent years is that the lower chlorinated congeners were more rapidly released and dissipated in the late 1970s and early 1980s, whereas the higher chlorinated congeners have tended to bioaccumulate more and are less rapidly released by fish.

Lipid-Based PCB Concentrations

Differences between PCB levels in different fish species are probably due to both differential feeding patterns and lipid content. Fish lipid (fat) typically accumulates PCBs more than other, less fatty tissues, because PCBs are highly lipophilic compounds, i.e., PCBs are more soluble in fat than water. Normalizing PCB levels on a fish lipid weight basis has sometimes proved useful in comparing measurements among species. The average percent lipid of goldfish was 9.2, pumpkinseed was 3.3, brown bullhead was 2.9 and largemouth bass was 1.4, with skewed distributions. There appears to be little correlation between weight and PCB body burdens in most species. Age of the fish sampled was not often determined.

Besides accounting for species differences, accounting for lipid content is important because fish lipid content appears to have changed from samples caught from one year to the next. Dividing the PCB concentration (wet weight) by the measured lipid content (g-lipid/g-fish) of the sample, one obtains the PCB concentration per gram (mass) of fish-lipid (ug-PCB/g-lipid). Looking at all samples for all species at all Upper Hudson sampling locations on a lipid basis, the median PCB levels have declined from 1829 ug-PCB/g-lipid in 1977 to 271 ug-PCB/g-lipid in 1988, with a 95 percent upper confidence bound estimate on the mean of 484 (Table B.3-19).

Lipid-based means of Aroclor 1254 and 1016 concentrations by year in the predominant fish species sampled at River Mile 175 are shown in Figure B.3-15. On a lipid basis, largemouth bass have usually shown the highest Aroclor 1254 levels. This may reflect their position as top carnivores in the aquatic food chain or their low fat content. Further, the lipid-based Aroclor 1254 levels in most species appear to have been relatively constant since 1982. Error bars are omitted from the multiple species plots for legibility. Trends in largemouth bass, with error bars, are shown in Figure B.3-16. (An error bar shown as a vertical line at each year representing the 95 percent upper and lower confidence bounds on the mean.)

In addition to the monitoring at River Mile 175, there is a good continuous record of sampling of brown bullhead at River Mile 153 just below the Federal Dam, for the period 1977-1988. PCB levels here were on average much lower than those observed at River Mile 175, presumably due to dilution of PCB concentrations by the flow of the Mohawk River. Average lipid-based PCB concentrations in brown bullhead show a regular exponential decline for Aroclor 1016 components and a less dramatic decline for Aroclor 1254 (Figure B.3-17).

B.3.4.2 Other Chemicals in Fish

The NYSDEC database contained analyses for chemicals in fish other than PCBs. Summary statistics for these other chemicals are shown in Table B.3-20.

Metals such as mercury, cadmium, chromium, copper and zinc were frequently detected. Median mercury and zinc concentrations were on the order of 0.4 ppm and 10 ppm, respectively. Organic compounds frequently detected include pesticides such as DDT, heptachlor, and dieldrin. Hexachlorobenzene and hexachlorocyclohexane were also detected frequently.

B.3.4.3 NYSDOH Macroinvertebrate Studies

As part of the Hudson River PCB Reclamation Demonstration Project, the New York State Department of Health (NYSDOH) conducted biomonitoring studies from 1976 to 1985 using caddisfly larvae, multiplate samples and chironomid larvae (Simpson *et al.*, 1986). These studies included long-term biomonitoring efforts from 1976 to 1985 as well as two short-term biological uptake studies in July and September of 1985.

Long-Term Biomonitoring Study

From 1976 through 1985, artificial substrate samplers (multiplates) were placed at 17 sites along the Hudson River from Hudson Falls to Nyack, New York (Novak *et al.*, 1988). These samplers were collected each year after a period of five weeks during the months of July, August and September. PCBs in the samples were reported as Aroclors 1016 and 1254. The resulting PCB concentrations in the multiplate samples represented a composite of sediment, algae, plankton and various macroinvertebrates. Invertebrates collected in the multiplate samplers included the following taxonomic groups: Chironomidae, Oligochataeta, Trichoptera, Ephemeroptera, Amphipoda and Elimidae. Chironomid larvae and pupae were the most abundant invertebrate component from Fort Edward to Saugerties, comprising up to 86 percent of the total macroinvertebrate population at Fort Miller and Waterford.

From 1978 to 1985, caddisfly larvae were collected by hand-picking individuals from rocks at five designated sites: Hudson Falls, Fort Edward, Fort Miller, Stillwater and Waterford. Caddisfly collections were made in June, July, August and September of each year.

Measured PCB levels in the 1985 multiplate samples for September ranged from 0.25 ppm at Hudson Falls (the control site) to over 6 ppm at Fort Edward. Multiplate monitoring from Fort Miller to Waterford resulted in PCB levels of 4 to 5 ppm. The multiplate samples at any one site appeared to show a consistent decline in PCB concentrations from early summer to later summer in any particular year. Larger scale trends or relationships in either time or with sample location are difficult to detect, because of the extremely wide variation in the sample results. Average PCB concentrations in multiplate samples generally showed a decline from 1976 - 1980. Nevertheless, average PCB concentrations increased in 1981 and remained high through 1985. Multiplate samples from the Thompson Island Pool and downstream showed significantly higher PCB concentrations than samples taken upstream of Fort Edward. Yet, no significant trends are apparent when comparing Fort Edward results with those at Waterford.

The results of the caddisfly biomonitoring efforts show a decline in total PCB concentrations from 1978 to 1980. As is true of the multiplate data, spatial trends are not readily apparent from the caddisfly samples. Measured PCB levels in macroinvertebrate tissues generally ranged from 20 to 60 ppm (dry weight) in 1979 and from 20 to approximately 40 ppm (dry weight) in 1985. During the same collection periods, macroinvertebrates collected at Hudson Falls, the control site, exhibited PCB tissue residues less than 10 ppm.

The data from this study exhibit a great deal of random variability. The bulk of the data are from multiplates, which collect sediment as well as living matter. In theory, it should be possible to distinguish sediment versus biologically-based PCBs by adjusting the observations to a lipid basis; lipid content of samples was reported. An additional factor contributing to the variability is that in almost every year a downward trend by month was observed at most stations, based generally on three samples. The cause of this phenomenon is not known, but the limited number of observations in this yearly cycle may have obscured the influence of other factors.

Examination of the complete run of samples at the Fort Miller multiplate and caddisfly station (PCB-5) shows what appears to be a largely random pattern of total PCBs on a dry weight basis (Figure B.3-18). The larger mean PCB concentration for the period 1981-82 is largely attributed to the increased variability of the data for this period. Adjusting to a lipid basis actually increases the total variability represented by outlying data points. Only a slight trend is suggested in this figure, where PCB levels may have declined from 1976 to 1980, then remained approximately stable. At Stillwater, the lipid-based values appear to be almost entirely random (Figure B.3-19).

A comparison between the confidence intervals on the overall means for all sample years by sample locations (Figure B.3-20) shows differences between the low values observed above Fort Edward and values at downstream stations. Lipid-based means at all stations by year (Figure B.3-21) appear to show a decline from 1976 to 1979, then relatively constant levels except for a jump upward in 1982. Again, this apparent trend is difficult to confirm as a consequence of the limited number of samples and possible differences in sampling or analytical protocols throughout the duration of the monitoring.

Short-Term Biomonitoring Study

Short-term biomonitoring investigations using the chironomid larvae, *Chironomus tentans*, were also performed by the NYSDOH during July and September 1985 (Novak *et al.*, 1990). The monitoring method consisted of placing 25 laboratory-reared chironomid larvae in nylon mesh envelopes or packets that were exposed to the water column. Envelopes were placed, in groups of ten, in steel mesh baskets at the primary collection site and monitored at 0, 1, 2, 4, 8, 12, 24, 48, 72 and 96 hours. Chironomids were placed at four sites, including two at Thompson Island Pool, one at Bakers Falls and one at Fish Creek, and monitored at 96 hours. Packets of chironomids exposed to the sediment at a collection site located on the eastern shore of the Thompson Island Pool were also collected at 96 hours. Water column samples were obtained during the same collection intervals for each site.

Results of this investigation indicated that chironomids accumulated PCBs ranging from 0.1 to 7 ppm after 1 to 96 hours of continuous exposure, whereas larvae exposed to sediments near the Thompson Island Pool for 96 hours contained over 100 ppm. Water column PCB levels were in the range of 0.03 - 0.1 µg/l during the experiment. The ratio of the PCB levels in the chironomids (in ppb) to the ambient PCB concentrations in the water column (defined as the bioaccumulation factor or BAF) were on the order of 10^4 to 10^5.

A significant conclusion from this study was that the PCB congener pattern found in tissues of chironomid larvae differed substantially from the congener pattern observed in water. Using capillary column gas chromatography, the investigators were able to isolate PCB congeners in both the water column and chironomids. The most abundant congeners in chironomid tissues were 2,4,2',5'-tetrachlorobiphenyl and 2,3,6,4'-tetrachlorobiphenyl. In contrast, the predominant congeners in water were 2,2'-dichlorobiphenyl, 2,6-dichlorobiphenyl, and 2,6,2'-trichlorobiphenyl. These findings suggest a number of possible explanations. One explanation could be that chironomid larvae may selectively bioconcentrate the more highly chlorinated congeners which are present at relatively low concentrations in water. Another factor which could explain the observed results is that the lower chlorinated congeners were present in the water but below detection limits.

Although this study presents some very interesting congener-specific results, they are too limited in scope to provide clear indications of either congener-specific bioaccumulation or congener-specific comparisons of PCBs in sediments. (No congener-specific sediment data were obtained.)

B.3.5 PCB Concentrations in Air and Plants

B.3.5.1 Air

Monitoring Near Fort Edward

Air monitoring efforts for PCBs and other air toxics have been conducted in the Upper Hudson River study area from late 1976 to 1982 by NYSDEC/NYSDOH and various researchers and as recently as 1989/90 by contractors (Harza/Yates-Auberle) for General Electric.

From January through August of 1977, NYSDOH collected air samples at five locations in the Upper Hudson Valley to determine ambient PCB concentrations. While the Glens Falls and Warrensburg samples showed no detections above the 0.020 $\mu g/m^3$ detection limit, results from the Hudson Falls and the Fort Edward samples demonstrated high levels of total PCBs, ranging from 0.060 $\mu g/m^3$ to 3.26 $\mu g/m^3$ (Malcolm-Pirnie, 1978). Atmospheric levels of PCBs in the Fort Edward area were reported to decrease from 1 $\mu g/m^3$ down to 0.3 $\mu g/m^3$ after the cessation of PCB use by General Electric in their Hudson Falls and Fort Edward capacitor plants in 1977 (Limburg, 1984).

In 1979, NYSDEC conducted an air monitoring survey for PCBs around various dumps and landfills (Caputo and Fort Miller dumps, Remnant Area, Moreau and Site 3a and Buoy 212) in the Hudson Falls/Fort Edward area bordering the Upper Hudson River (see Table B.3-21). Values ranged from 5 to 15 $\mu g/m^3$ total PCBs at the Moreau and remnant areas and 24 to 300 $\mu g/m^3$ total PCBs at the Fort Miller and Caputo dumps, respectively. At the Caputo dump, where the soil was reported to contain 5,000 mg/kg PCBs, air monitoring for PCBs before and after capping of the site showed that average ambient PCB concentrations decreased from 118 $\mu g/m^3$ before capping to 0.26 $\mu g/m^3$, once the site was capped (Shen, 1982).

Two air samples taken over Lock 6 in the summer of 1980 yielded Aroclor 1242 concentrations ranging from 0.11 $\mu g/m^3$ to 0.52 $\mu g/m^3$ (Table B.3-21). Aroclors 1221 and 1254 were not detected (NYSDEC, 1981). Ambient PCB levels

monitored at farm fields in 1981 near the Hudson River, however, showed low PCB concentrations of approximately 0.005 $\mu g/m^3$ (NUS, 1984).

In the early 1980s, NYSDEC and the Boyce Thompson Institute for Plant Research of Cornell University conducted a joint air/plant monitoring effort near the tailwater of Lock 6 to determine if volatilization of PCBs from the water column was occurring (Buckley and Tofflemire, 1983 and 1984). Between August 1981 and September 1981, seven air samples were taken. Aroclor 1242 was detected in all seven samples, ranging from 0.031 to 0.06 $\mu g/m^3$. Aroclor 1254 was detected in three of the samples at levels up to 0.0013 $\mu g/m^3$. During this study, a vertical PCB gradient was also noticed, when airborne PCBs were measured simultaneously at heights of 1 and 4.5 meters above the water.

In August 1986, NYSDEC collected three sets of ambient grab samples in duplicate at the proposed containment site (Site G), the Fort Edward Landfill, the Bourgoyne Avenue School and Lock 7 of the Champlain Canal (USEPA/NYSDEC, 1987). The highest ambient PCB concentration measured was 0.083 $\mu g/m^3$ at Lock 7. Site G and the Fort Edward landfill samples contained PCB levels below the detection limit of 0.007 $\mu g/m^3$. The Bourgoyne Avenue School sample information was not available at the time of the report.

Again in 1987, NYSDEC conducted air monitoring from April 2, 1987 to July 16, 1987 at the Kingsbury Landfill, located north of Fort Edward. In 76 of 105 samples taken over April and May 1987, Aroclor 1016/1242 was detected with a maximum concentration of 0.49 $\mu g/m^3$. Aroclor 1248 was detected in 5 of 105 samples with a maximum concentration of 0.52 $\mu g/m^3$. Both the Kingsbury and Fort Edward municipal landfills were the burial site of several thousand tons of PCBs. Neither these landfills, nor others in the area, are part of the current investigation of the Hudson River Superfund Site. The results are presented here as evidence of PCBs having been monitored in the air within the vicinity of the site.

Most recently, in connection with its Remnant Remediation Project, GE conducted baseline pre-remediation air monitoring from August to November 1989 (Harza, 1990). Fixed air monitoring stations were planned at five locations along the river: two residential areas, one upwind location, one downwind location and a farming receptor location two miles south of the remnant area. Because of site access problems, only three of the five sites were monitored. Once every three days, two 1-liter/minute, 24-hour air samples were taken using a two-channel (two separate samples taken simultaneously) air monitoring station three feet above the ground. The samples were analyzed by NIOSH Method 5503 (desorption of fluorisil tubes with solvent followed by GC-ECD analysis) with a detection limit of 0.05 µg/sample. In total, 84 samples were collected. Seven samples showed levels of PCBs above the detection limit (0.05 µg/sample), with a maximum value of 0.23 $\mu g/m^3$. Of these seven detects, three were from a residential area (location A2), two from a downwind receptor (location A4) and one from the farm area (location A5). Although PCBs were detected in this investigation, the two sample channels often gave inconsistent results; one channel contained PCBs while the other did not. This occurrence may have been due to sampling or analytical problems or both.

Maximum ambient background PCB concentrations in air measured by New York State during a statewide monitoring effort, listed at the bottom of Table B.3-21, provide a perspective on PCB levels in air in the vicinity of the site. Maximum ambient air PCB concentrations measured in this effort ranged from 0.002 $\mu g/m^3$ in Syracuse and Rensselaer (urban areas) to 0.007 $\mu g/m^3$ in Staten Island (NYSDEC, 1982-4). These maxima are one to two orders of magnitude lower than the maxima detected during GE's baseline monitoring study and values measured by NYSDEC near Lock 7 in August 1986.

Lamont-Doherty Investigations

Three studies of PCB volatilization and properties governing volatilization were conducted at the Lamont-Doherty Geological Observatory. The first two of these studies (Bopp, 1979 and Bopp, 1983) dealt with the estimation of PCB properties that govern volatilization and the application of these properties to

etals such as mercury, cadmium, chromium, copper and zinc were frequently ed. Median mercury and zinc concentrations were on the order of 0.4 ppm 10 ppm, respectively. Organic compounds frequently detected include icides such as DDT, heptachlor, and dieldrin. Hexachlorobenzene and xachlorocyclohexane were also detected frequently.

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Besides accounting for species differences, accounting for lipid content is important because fish lipid content appears to have changed from samples caught from one year to the next. Dividing the PCB concentration (wet weight) by the measured lipid content (g-lipid/g-fish) of the sample, one obtains the PCB concentration per gram (mass) of fish-lipid (ug-PCB/g-lipid). Looking at all samples for all species at all Upper Hudson sampling locations on a lipid basis, the median PCB levels have declined from 1829 ug-PCB/g-lipid in 1977 to 271 ug-PCB/g-lipid in 1988, with a 95 percent upper confidence bound estimate on the mean of 484 (Table B.3-19).

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Results of this investigation indicated that chironomids accumulated PCBs ranging from 0.1 to 7 ppm after 1 to 96 hours of continuous exposure, whereas larvae exposed to sediments near the Thompson Island Pool for 96 hours contained over 100 ppm. Water column PCB levels were in the range of 0.03 - 0.1 µg/l during the experiment. The ratio of the PCB levels in the chironomids (in ppb) to the ambient PCB concentrations in the water column (defined as the bioaccumulation factor or BAF) were on the order of 10^4 to 10^5.

A significant conclusion from this study was that the PCB congener pattern found in tissues of chironomid larvae differed substantially from the congener pattern observed in water. Using capillary column gas chromatography, the investigators were able to isolate PCB congeners in both the water column and chironomids. The most abundant congeners in chironomid tissues were 2,4,2',5'-tetrachlorobiphenyl and 2,3,6,4'-tetrachlorobiphenyl. In contrast, the predominant congeners in water were 2,2'-dichlorobiphenyl, 2,6-dichlorobiphenyl, and 2,6,2'-trichlorobiphenyl. These findings suggest a number of possible explanations. One explanation could be that chironomid larvae may selectively bioconcentrate the more highly chlorinated congeners which are present at relatively low concentrations in water. Another factor which could explain the observed results is that the lower chlorinated congeners were present in the water but below detection limits.

Although this study presents some very interesting congener-specific results, they are too limited in scope to provide clear indications of either congener-specific bioaccumulation or congener-specific comparisons of PCBs in sediments. (No congener-specific sediment data were obtained.)

Most recently, in connection with its Remnant Remediation Project, GE conducted baseline pre-remediation air monitoring from August to November 1989 (Harza, 1990). Fixed air monitoring stations were planned at five locations along the river: two residential areas, one upwind location, one downwind location and a farming receptor location two miles south of the remnant area. Because of site access problems, only three of the five sites were monitored. Once every three days, two 1-liter/minute, 24-hour air samples were taken using a two-channel (two separate samples taken simultaneously) air monitoring station three feet above the ground. The samples were analyzed by NIOSH Method 5503 (desorption of fluorisil tubes with solvent followed by GC-ECD analysis) with a detection limit of 0.05 µg/sample. In total, 84 samples were collected. Seven samples showed levels of PCBs above the detection limit (0.05 µg/sample), with a maximum value of 0.23 µg/m^3. Of these seven detects, three were from a residential area (location A2), two from a downwind receptor (location A4) and one from the farm area (location A5). Although PCBs were detected in this investigation, the two sample channels often gave inconsistent results; one channel contained PCBs while the other did not. This occurrence may have been due to sampling or analytical problems or both.

Maximum ambient background PCB concentrations in air measured by New York State during a statewide monitoring effort, listed at the bottom of Table B.3-21, provide a perspective on PCB levels in air in the vicinity of the site. Maximum ambient air PCB concentrations measured in this effort ranged from 0.002 µg/m^3 in Syracuse and Rensselaer (urban areas) to 0.007 µg/m^3 in Staten Island (NYSDEC, 1982-4). These maxima are one to two orders of magnitude lower than the maxima detected during GE's baseline monitoring study and values measured by NYSDEC near Lock 7 in August 1986.

Lamont-Doherty Investigations

Three studies of PCB volatilization and properties governing volatilization were conducted at the Lamont-Doherty Geological Observatory. The first two of these studies (Bopp, 1979 and Bopp, 1983) dealt with the estimation of PCB properties that govern volatilization and the application of these properties to

monitored at farm fields in 1981 near the Hudson River, however, showed low PCB concentrations of approximately 0.005 µg/m³ (NUS, 1984).

In the early 1980s, NYSDEC and the Boyce Thompson Institute for Plant Research of Cornell University conducted a joint air/plant monitoring effort near the tailwater of Lock 6 to determine if volatilization of PCBs from the water column was occurring (Buckley and Tofflemire, 1983 and 1984). Between August 1981 and September 1981, seven air samples were taken. Aroclor 1242 was detected in all seven samples, ranging from 0.031 to 0.06 µg/m³. Aroclor 1254 was detected in three of the samples at levels up to 0.0013 µg/m³. During this study, a vertical PCB gradient was also noticed, when airborne PCBs were measured simultaneously at heights of 1 and 4.5 meters above the water.

In August 1986, NYSDEC collected three sets of ambient grab samples in duplicate at the proposed containment site (Site G), the Fort Edward Landfill, the Bourgoyne Avenue School and Lock 7 of the Champlain Canal (USEPA/NYSDEC, 1987). The highest ambient PCB concentration measured was 0.083 µg/m³ at Lock 7. Site G and the Fort Edward landfill samples contained PCB levels below the detection limit of 0.007 µg/m³. The Bourgoyne Avenue School sample information was not available at the time of the report.

Again in 1987, NYSDEC conducted air monitoring from April 2, 1987 to July 16, 1987 at the Kingsbury Landfill, located north of Fort Edward. In 76 of 105 samples taken over April and May 1987, Aroclor 1016/1242 was detected with a maximum concentration of 0.49 µg/m³. Aroclor 1248 was detected in 5 of 105 samples with a maximum concentration of 0.52 µg/m³. Both the Kingsbury and Fort Edward municipal landfills were the burial site of several thousand tons of PCBs. Neither these landfills, nor others in the area, are part of the current investigation of the Hudson River Superfund Site. The results are presented here as evidence of PCBs having been monitored in the air within the vicinity of the site.

PHASE 1 REPORT - REVIEW COPY
INTERIM CHARACTERIZATION AND EVALUATION

HUDSON RIVER PCB REASSESSMENT RI/FS

EPA WORK ASSIGNMENT NO. 013-2N84

AUGUST 1991

Region II

ALTERNATIVE REMEDIAL CONTRACTING STRATEGY (ARCS)
FOR
HAZARDOUS WASTE REMEDIAL SERVICES

EPA Contract No. 68-S9-2001

VOLUME 1
(BOOK 1 OF 2)

TAMS CONSULTANTS, Inc.
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PART B

PHASE 1 REPORT

INTERIM CHARACTERIZATION AND EVALUATION

HUDSON RIVER PCB REASSESSMENT RI/FS

CONTENTS

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

TABLES

Tables Located in Volume 1 (2 of 2)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

FIGURES

Figures Located in Volume 1 (2 of 2)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PART B

CONTENTS
(continued)

PLATES

Plates Located in Volume 1 (2 of 2)

PART B

CONTENTS
(continued)

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SYNOPSIS
PHYSICAL SITE CHARACTERISTICS
(Section B.1)

The hydrology of the Upper Hudson River is described in detail (B.1.1), building upon the basic description of Hudson River hydrology given in Part A. Flow characteristics of the Upper Hudson generally show a strong seasonal dependence, with maximum flows during the annual spring thaw. Flows are partially regulated by wetlands as well as the Sacandaga Reservoir. Four major tributaries to the Upper Hudson below Fort Edward combine with flow upstream to produce an average annual flow of 7100 cfs at Waterford, above its confluence with the Mohawk. This basic flow regime governs the transport of PCBs in the Upper Hudson.

Water quality (B.1.2) is described according to New York State water quality classifications assigned on the basis of "best usages" and the results of water quality sampling data from a 1987-1988 survey. The water quality of the Upper Hudson at Fort Edward and Schuylerville was rated as poor, partly because of the fishing ban, as a result of historic PCB discharges.

Only the town of Waterford draws its drinking water from the Upper Hudson below Fort Edward. More commonly, the river water is used for industrial and commercial purposes, such as power-generation, and for domestic and agricultural use, such as watering lawns, gardens or crops. The Upper Hudson River is a navigational waterway; from Waterford to Fort Edward, it is co-incident with the Champlain Canal.

Population in the four counties bordering the Upper Hudson between Albany and Glens Falls is over half a million and land use is predominantly agricultural (B.1.3). Dairy farming is the principal form of agriculture. The region is also host to a number of industries, generally located in the vicinity of the population centers.

The Upper Hudson River represents a diverse fisheries resource (B.1.4); six fish surveys from 1933 to 1985 are reviewed. In general, these surveys show that the majority of fish species historically found in the Upper Hudson continue to reside there. The data also indicate a qualitative improvement in the fisheries resource.

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B. UPPER HUDSON CHARACTERIZATION

B.1 Physical Site Characteristics

B.1.1 Hydrology

The Upper Hudson River flows southerly from its source at Lake Tear-of-the-Clouds near Mt. Marcy in the Adirondacks to its confluence with the Mohawk River near the Federal Dam at Green Island, Troy, NY. The drainage area of this segment, shown in Plate A.1-1, is 4,630 square miles (Wagner, 1982). (Some discussion of overall Hudson River basin hydrology, presented in Part A, is not repeated here.)

The Upper Hudson River drains a major portion of the southern and central Adirondacks. Along its course, the main channel is intersected by several tributary branches, the most significant of which are the Sacandaga River, the Batten Kill, the Fish Creek and the Hoosic River (see Plate B.1-1). Water flow in this segment is regulated by several dams on the Hudson itself (see Plate B.1-2), as well as on tributary branches. Flow is further controlled by abundant wetlands located throughout the basin, which act as a buffer for high and low flow conditions.

The total mean annual fresh water flow from the Upper Hudson at its confluence with the Mohawk near Waterford is about 7,100 cfs. This flow represents more than a two-fold increase in flow from that at Hadley, NY. Before the river reaches the Bakers Falls - Fort Edward area, it is joined by the Sacandaga River, the largest single tributary in this area. The mean annual flow at Fort Edward is roughly 3,800 cfs, about 54 percent of the total flow at Waterford. Downstream of this location, the remaining tributaries are fairly evenly spaced, at roughly 10 to 15 miles between tributary junctions. The combined total of these tributaries doubles the flow of the Upper Hudson by the time it reaches Waterford. Of particular importance is the Hoosic River, which represents about 15 percent of the total drainage area south of Hadley.

Flow in the Hudson Basin is seasonally dependent, with flow patterns similar to those seen throughout the basin. The typical regime is one of fairly steady flow throughout nine months of the year. During the spring, flows in the Upper Hudson increase substantially in response to the melting of winter snow. The maximum flow at Waterford occurs one month before the maximum at Fort Edward; melting of winter snows in the southern portion of the Upper Hudson basin tends to occur earlier than in the portion of the Upper Hudson basin above Fort Edward. Figure B.1-1 shows mean monthly flows for water year 1986 at Fort Edward and Waterford, respectively. This seasonal pattern is duplicated in the flows at the Federal Dam at Green Island for the same year.

Flows for an atypical water year (1984) are shown in Figure B.1-2 for Fort Edward and Waterford, respectively. That water year was characterized as having an unusually warm winter with many major storm events throughout the year. The flow patterns at Fort Edward and Waterford in that year were also present at the Federal Dam and on the Wallkill River, implying that these unusual conditions were felt throughout the entire Hudson basin.

B.1.2 Water Quality and Use

B.1.2.1 Water Quality

New York State has classified its surface waters according to "best usages" and has established numerical water quality criteria (standards) to which those waters should conform. Waters that conform to the numerical criteria are considered suitable for their intended best use. Water quality classifications applicable to the Upper Hudson are listed in New York State's environmental regulations (6NYCRR700, et seq.) and are illustrated on Plate B.1-3. In summary, the Upper Hudson has assigned to its different reaches the following classes and uses:

Class A - drinking water (water supply)
Class B - primary contact recreation (swimming), fishing, fish propagation
Class C - fishing, fish propagation, swimming
Class D - fishing, fish passage, swimming

Numerical quality standards for each of the above classifications are found in the State's rules at 6NYCRR700-705. A recent NYSDEC Technical and Operational Guidance Series (TOGS) memo (September 25, 1990) has augmented the rules, particularly with regard to toxic constituents. The standards encompass conventional pollutant parameters, such as coliform levels, dissolved oxygen, turbidity and pH, as well as toxic constituents, such as heavy metals and organics. The toxic substance standards have been derived from health and environmental risk assessments performed either at the state level or derived from those performed by USEPA.

Historically, the state has used information from diverse sources to ascertain the condition of its surface waters and to evaluate their attainment of the designated best uses. Currently, a program of rotating intensive basin studies (RIBS) exists whereby NYSDEC monitors all surface waters on a six year cycle and uses data from other programs to provide continuity of information when RIBS sampling is not occurring. An intensive basin survey was conducted within the Upper Hudson during 1987 and 1988 and initial reports from that effort have now become available (NYSDEC, 1990).

As part of the Upper Hudson RIBS effort, NYSDEC evaluated water column conditions, toxics in bottom sediments and contaminant uptake by macroinvertebrates and fish. Table B.1-1 provides the six main parameters/media used in the RIBS program to rate water quality. Samples were collected at five locations along the river's main stem at North Creek, Corinth, Fort Edward, Schuylerville and Waterford. Although NYSDEC did not report direct conclusions concerning attainment of water quality standards in its RIBS document, a qualitative evaluation of overall conditions within particular river reaches is provided. Conclusions pertinent to those reaches incorporating the Fort Edward and Schuylerville sampling locations are summarized here.

Both the Fort Edward and Schuylerville water column samples exhibited elevated copper levels at a sufficient frequency to warrant considering copper a parameter of concern. Similarly, iron was found to be a parameter of concern at Schuylerville. No other trace constituents were detected at elevated

concentrations with sufficient frequency to be considered parameters of concern. Copper, a pervasive constituent of Upper Hudson River water, is also found in samples from the relatively pristine North Creek and Corinth river reaches. While sediment samples were obtained and analyzed from the Schuylerville location, such samples were not obtained at Fort Edward. According to RIBS, Schuylerville sediments were at the upper limit of background for cadmium and were slightly above background for lead and mercury. NYSDEC did not identify the background levels upon which their conclusions were based.

The 1987-1988 RIBS program found no water column toxicity to *Ceriodaphnia* at either Fort Edward or Schuylerville. Tissue from caddisflies collected at the two sites did not exhibit elevated levels of either heavy metals or of PCBs, in contrast to previous studies. Apparently macroinverterbrate tissue collected in 1987 at Hudson Falls (location not specified) did exhibit elevated lead and manganese levels. At both Fort Edward and Schuylerville, NYSDEC assessed overall water quality as poor, based on the RIBS results and the fishing ban, a result of historic PCB discharges to this river reach. At one point the RIBS document concludes that water quality is rated as poor primarily due to the fishing ban (NYSDEC,1990,p.77).

As a result of several amendments to the Clean Water Act, states are required to report in specific terms conditions of their surface waters. Clean Water Act Section 305(b) mandates that states submit water quality condition reports to USEPA every two years. The 305(b) report evaluates surface waters in relationship to their ability to sustain primary contact recreation and fish propagation uses. In addition, Clean Water Act Section 304(l) requires that states generate lists of surface waters that fail to meet water quality standards because of toxic pollutants, in general, and toxics from point sources, in particular.

New York's most recent 305(b) report was published in April 1990. That document provides, in part, a summary of waters wherein contamination in fish exceeds either FDA levels or other guidelines. For the Upper Hudson reach from Hudson Falls to Federal Dam, the only contaminant identified as exceeding either

the FDA levels or other guidelines is PCB (NYSDEC, 1990, Table 20). The 305(b) report also identifies PCB in sediment as the sole toxic that is responsible for use impairment in the Upper Hudson (NYSDEC, 1990, Table 17). Similarly, the state's 304(l) lists identify priority organics (PCBs) as being the toxic responsible for Upper Hudson use impairment.

B.1.2.2 Use

The Hudson River is used as a source for public water supplies (municipal and institutional drinking water) in sections of the river classified as Class AA or A. Along the Upper Hudson, three communities draw directly Hudson River water. Of these, Queensbury and Waterford have current average uses of more than 1 mgd (NYSDOH, 1991), as shown in Table B.1-2. The Waterford intake is located at the base of the Upper Hudson Basin near Lock 1. The Queensbury intake is located near Sherman Island Dam in Warren County. The Winebrook Hills Water District, the third Upper Hudson water supply drawing from Hudson, is located at the headwaters of the Hudson in Newcomb, Essex County.

A more common use of Hudson River water is for industrial and commercial purposes such as cooling, manufacturing processes and fire protection. Hudson River water is also extensively used for hydroelectric and thermal power generation. An inventory of facilities and plants that utilize Hudson River water can be found in reports for the Hudson River-Black River Regulating District (Malcolm Pirnie, 1984a) and for the NYSDOT (1984).

Hudson River water is also used for domestic (watering lawns and gardens) and agricultural purposes (irrigating crops). There are currently no records of water withdrawal for agricultural uses. Unlike the other intakes, permits are not needed to withdraw water from the Hudson for irrigation purposes (pers. comm., NYSDEC and NYSDOH, 1991).

The NYSDEC Division of Water, Source Surveillance Section provided a listing (March 7, 1991) of all significant active facilities with SPDES permits in the Upper Hudson River Basin. This search revealed that 27 facilities

discharge into the Upper Hudson Basin, with 15 discharging directly into the Hudson River. Five of the 15 facilities are municipal wastewater treatment plants, including Corinth Sewage Treatment Plant, Glens Falls Wastewater Treatment Plant, Saratoga County Sewer District #1 Wastewater Treatment Plant at Mechanicville, Stillwater Sewage Treatment Plant and Washington County Sewer District #2 Water Pollution Control Plant at Fort Edward.

The Champlain Canal is coincident with portions of the Hudson River; it extends from Waterford, New York on the Hudson River to Whitehall at the southern end of Lake Champlain. The Champlain Canal is part of the New York State Barge Canal System, also comprised of the Erie Canal, Oswego Canal and Cayuga-Seneca Canals. This network of waterways connects the Atlantic Ocean with the Great Lakes and the Saint Lawrence Seaway. The Champlain Canal is 60 miles in length, including 37 miles of canalized Hudson River from Waterford to Fort Edward and 23 miles of land-cut sections. The canal diverges from the river at Fort Edward just below Lock 7 and proceeds in a northeasterly direction to Lake Champlain. Additional land-cut areas exist at Stillwater, Northumberland and Fort Miller.

Natural flows provide a considerable portion of the water supply needs for the Hudson River portion of the Champlain Canal. The Hudson River at Fort Edward provides an average discharge of 5,244 cfs (USGS average for 1979-1990 water years) to the canal at the confluence below Lock 7. This flow is significantly influenced by regulation of flows from Great Sacandaga Lake. Below this point, the river is canalized to provide natural flows for the canal. Water is also supplied to the Champlain Canal via the Glens Falls Feeder Canal. The Feeder Canal diverts approximately 100 cfs of water from the Hudson River upstream of the Feeder Dam west of Glens Falls to the Champlain Canal Summit Level between Locks 8 and 9 on the canal near Smith's Basin. The Summit Level is the point of highest elevation on the canal and allows for the gravity flow of water in both directions as lockages are made (south to the Hudson River, north to Lake Champlain). Approximately 25 percent of the diverted water returns to the Hudson River (Malcolm Pirnie, 1984).

Commercial traffic has declined on the Champlain Canal and other canals in the Barge Canal system as a result of "the unreliability of the system for waterway transport" and competition from other modes of transportation for bulk products (Malcolm Pirnie, 1984b, US Army Corps of Engineers, 1977). Unlike the other three canals in the system, the Champlain Canal shows a steady decline in recreational use along the entire stretch of the canal on both the canalized Hudson River and the land-cut section north of Fort Edward (Malcolm Pirnie, 1984).

B.1.3 Population and Land Use

Four counties (Albany, Washington, Rensselaer and Saratoga) lie adjacent to the Upper Hudson between Albany and Glens Falls. All counties experienced growth between 1980 and 1990 with Saratoga having the greatest increase over the period and Rensselaer the lowest. Total population of these counties in 1990 was over 500,000.

Land use within a zone adjacent to the Upper Hudson River, depicted on Plate B.1-4, is mostly agricultural. Portions lie within New York State Agricultural Districts and include parcels considered to be prime farmland. Dairy farming is the major agricultural industry. The majority of the crops grown, such as corn and hay, are used for forage; small quantities of cash crops, such as oats and wheat, are produced. Industrial use is typically located near population centers. Major non-agricultural industries within the study area include: an industrial demolition company; several paper mills; hydroelectric plants; a grocery warehouse; and manufacturers of garden equipment, brake linings, brushes, paints, wallpaper, paper products, gun barrels, silicone products, abrasives, brass fittings and clothing. Forested and recreational land uses are scattered.

Existing recreational uses include Schaghticoke Canal Park at Lock 4 of the Champlain Canal and two town parks, which lie along the river in Fort Edward. Proposed for the Fort Edward area are a marina, to be located on the south end of Rogers Island, and a marina, trails and picnic areas to be located one mile

south of Fort Edward on the former Champlain Canal. Saratoga National Historic Park lies on the western bank of the River in the Town of Stillwater. Moreau State Park is located south of Glens Falls. At the confluence of the Mohawk and Hudson Rivers are Peebles Island State Park and the Van Schaik Island Country Club. Several parks and/or country clubs also front the River.

B.1.4 Fisheries

Fishery resources within the Hudson River from Federal Dam to Fort Edward are influenced by different physical features, as well as man-made structures such as locks, dams, guard gates, bridges, spillways and submerged power lines and cables. This array of physical features produces the following variety of different fish habitats:

- Outlets of streams and rivers;
- Shallow water areas (wetland and non-wetland);
- Designated ship channels of the canal (canalized river);
- Steep embankment areas with relatively swift current;
- Landcut (artificial) portions of the canal;
- Wet dumping grounds or spoil areas; and
- Various alternate channels separated from the main channel by an island.

This wide variation in habitats expands spatial heterogeneity and results in a complex fishery resource.

The New York State Conservation Department (Greeley, J.R. and Bishop, 1933) conducted an early and comprehensive fish inventory of the portion of the Hudson River between Hudson Falls and the mouth of the Hoosic River. Forty-one species of fish (Table B.1-3) were recorded. (The American shad, one might note, was listed as extinct.) The historical data developed as part of this 1933 fish inventory documented that there was an imbalance between the juvenile and adult

game fish. There appeared to be abundant juvenile game species from a few miles below Fort Edward to the mouth of the Hoosic River, but adult populations of game fish were uncommon.

An anonymous report, prepared by the Conservation Department in June 1960 (see Table B.1-3) at the request of the Stillwater Rod and Gun Club, are the only fish data that exist from 1933 to 1960. This 1960 report contains the observation that game fish resources declined between 1949 to 1959. Prior to 1949, angling success was supposedly satisfactory for "...bass, walleyes, northerns and pickerel." Several theories for this decline were advanced, such as industrial pollution from the Glens Falls and Fort Edward areas, effluent process changes at a Glens Falls paper and pulp mill, expanded boating activity and over-exploitation.

The next major fish inventory was conducted by Lane (1970) who found 13 species of fish. Lane stated the somewhat remarkable conclusion that: "The collection data indicate the absence of any significant fishery between Lock No. 1 and Fort Edward." With the exception of goldfish, few larger fish were found. Game fish collected in the river channel were represented largely by juvenile populations. Similar to 1933 observations, Lane noted a diminished amount of aquatic vegetation in the river channel. Although it is generally recognized that an adequate supply of aquatic vegetation is important for adult fish population maintenance and productivity, few historical and/or current data are available. One of the most unexpected results of Lane's survey was the absence of common carp. Although Lane concluded that "...conditions within the study area are not suitable for carp," later studies (Shupp, 1975; Makarewicz, 1983) do document the presence of the common carp within this section of the Upper Hudson.

Subsequently, NYSDEC began collecting fish for PCB analysis. Shupp (1975) collected fish samples in the 40-mile stretch of the Upper Hudson from Lock No. 1 to Hudson Falls and found an improvement, which he attributed to upgrading of treatment facilities and tougher regulations concerning industrial discharges. Although Shupp reported approximately 24 species of fish compared to the 13

species listed by Lane (1970), the fishing from Lock No. 1 to Hudson Falls was still considered poor, because of the overall low standing crop of fish and low numbers of adult fish compared to juveniles (Shupp, 1975). The reported preponderance of juvenile fish was similar to data from the 1933 and 1970 surveys. Sheppard (1976) indicated that "...some unknown factor is causing the exodus or demise of the mature segment of certain fish populations including the rock bass, pumpkinseed, yellow perch, walleye and chain pickerel." NYSDEC (R. Sloan, per. comm., 1991) has recently observed a greatly diminished number of both pumpkinseed and yellow perch populations during routine PCB assessments of resident fish in the Upper Hudson (Fort Edward to Federal Dam).

Since 1975, NYSDEC has continued to collect fish between Federal Dam and Fort Edward as part of their ongoing assessment and monitoring of PCB levels in fish flesh. The principal species collected and analyzed within this reach have been the brown bullhead, goldfish, largemouth bass, pumpkinseed and yellow perch. Because of the demise of the yellow perch and goldfish, current collection efforts have focused on the brown bullhead, common carp and largemouth bass (R. Sloan, per. comm.).

One of the most extensive fishery surveys since the 1933 survey was conducted approximately eight years ago by Makarewicz (1983). He surveyed 85 stations along the entire length of the Hudson between Federal Dam and Whitehall as part of the New York State Barge Canal Maintenance Dredging Program 1985-1995 for NYSDOT (Malcolm Pirnie, 1984b). The sampling stations included nine sampling reaches from Federal Dam to Fort Edward. A total of 46 species, including four migratory species (American eel, blueback herring, sea lamprey and striped bass), were found. Of the 42 resident freshwater species, the panfish ere the most prevalent (40 percent); demersal fish were second in abundance (22 percent); forage fish were the third most abundant group (14 percent); and game fish had the lowest relative abundance (9 percent). Dominant panfish members were bluegill, pumpkinseed, rock bass and yellow perch; demersal dominants were black bullhead and brown bullhead, common carp and white sucker; forage dominants were golden shiner, spotfin shiner and spottail shiner; and game fish dominants were

the largemouth bass and smallmouth bass. Collectively, these 13 species accounted for 85 percent of all resident freshwater species collected.

The most recent fish survey data available are from a study conducted by Green (1985) between Stillwater (Lock 4) and Schuylerville (Lock 5), covering approximately 13 river miles. Fewer overall species were taken (20) compared to the more intensive biological surveys in 1933 (41 species) and 1983 (46 species), both of which covered more of the river (Fort Edward to Federal Dam) than that covered in 1985 (Stillwater to Schuylerville).

Length at age comparisons for both smallmouth and largemouth bass indicate that growth rates were comparable to the New York Bass Study's average to fast growth rates (Green, 1985), an indication that some of the historical observations regarding the preponderance of juvenile fish and the paucity of adult game fish may no longer be valid for bass populations.

As shown in Table B.1-3, the list of species of fish in the 1983 study agrees quite well with the list for 1933. Although comparative percent contributions of the dominant game fish, panfish, forage fish and demersal fish with those species recorded in 1933 is not possible, because quantitative information are lacking in the historical study, all the above dominant species (with the exception of the spotfin shiner and black bullhead, see note 1, Table B.1-3) were also recorded in 1933. In addition, 31 species were similarly reported in both studies. An analysis of these two studies (Greeley and Bishop, 1933 and Makarewicz, 1983), which have spanned nearly 50 years, reveals considerable qualitative similarity of the fishery within the reach from Federal Dam to Fort Edward.

The construction of Federal Dam and various locks as part of the Champlain Canal Section of the New York State Barge Canal System blocked major upstream spawning migrations for a number of anadromous species, including the American shad, alewife, blueback herring, sturgeons and striped bass. Some migrants, as documented by Smith and Lake (1990), may be found periodically upstream of the

Federal Dam and Lock. Population pulses may enter and leave the lower region of the Upper Hudson through the interconnecting system of locks.

The majority of fish species listed in Table B.1-3 are freshwater residents of the Hudson River. Some migratory species, such as the striped bass, blueback herring, sea lamprey and American eel, still attempt to utilize sections of the Upper Hudson as migratory routes. With the exception of black bullhead, johnny darter, pearl dace and northern redbelly dace, all the fish species listed have also been found in various regions of the Lower Hudson River (Beebe and Savidge, 1988; Smith and Lake, 1990). Whereas many of the fish species are year-round freshwater residents, they are not unique to the Federal Dam/Fort Edward section of the Upper Hudson.

Some additional fish species, which were not found during the reported fish surveys summarized in Table B.1-3, have been reported in various sections of the entire Upper Hudson Region (Smith and Lake, 1990). These include:

American shad *(Alosa sapidissima)*
Stonecat *(Noturus flavus)*
Longnose sucker *(Catostomus catostomus)*
Lake chub *(Couesius plumbeus)*
Brassy minnow *(Hybognathus hankinsoni)*
Blacknose shiner *(Notropis heterolepis)*
Finescale dace *(Phoxinus neogaeus)*
Lake herring *(Coregonus artedi)*
Lake whitefish *(Coregonus clupeaformis)*
Round whitefish *(Prosopium cylindraceum)*
Lake trout *(Salvelinus namaycush)*
Rainbow smelt *(Osmerus mordax)*
Tessellated darter *(Etheostoma olmstedi)*

Although not found during the major fish surveys conducted within the Federal Dam/Fort Edward Region, the American shad has been known to occur within this particular region (per. comm., R. Sloan). Others such as the lake chub, lake herring, lake whitefish and lake trout are not commonly found in freshwater riverine systems and are not expected to occur to any great extent within this section of the Upper Hudson.

All studies reviewed to date indicate that the majority of species historically present in the lower section of the Upper Hudson continue to reside in this particular reach of the Hudson River. According to information submitted by Shupp (1987), the section of the Upper Hudson River between the Federal Dam and Fort Edward can support a diverse and high quality fishery resource. Shupp also cited evidence gathered from some NYSDEC studies between Mechanicville and Schuylerville, which suggested a "vast improvement" in smallmouth and largemouth bass stocks and other fish species from the early 1960s to the late 1980s. Shupp has stated, "Since 1984, the greatly improved warm water fish community in the Fort Edward to Troy (Upper Hudson) reach has stimulated interest in reopening the fishery."

Angler reports to The Warrensburg Fish Management Unit indicate a somewhat improved fishery for bass, yellow perch, black crappie and brown bullhead from 1969 to 1975 (Shupp, 1975). Analysis of the data presented by Lane (1970), Makarewicz (1983) and Green (1985) and prepared testimony statements by Shupp (1987) also suggest a qualitative improvement within the past twenty years.

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SYNOPSIS
SOURCES OF PCB CONTAMINATION
(Section B.2)

General Electric discharged PCBs from plants at Fort Edward and Hudson Falls between 1946 and 1977 (B.1.1). The total amount of PCBs released during 1957 to 1975, a period for which estimates can be made using historic data, ranges from 209,000 to 1,330,000 pounds.

Currently, six New York State facilities are permitted to discharge PCB-contaminated waste water to the basin of the Upper Hudson River (B.2.2). A facility in western Massachusetts discharges to a Hudson River tributary, the Hoosic River.

Other potential sources of PCBs to the Upper Hudson (B.3.3) are discussed.

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B.2 Sources of PCB Contamination

B.2.1 GE Discharges (To 1977)

The two GE plants at Fort Edward and Hudson Falls, New York began to use PCBs in 1946 and discontinued their utilization in 1977. In-plant sources of PCB discharges have been characterized as both minor spills and effluent from washing capacitor cans, with the latter being the major source. Capacitor cans were flood filled with dielectric fluid and then washed with detergent and water to remove excess material. Contaminated wash water was finally discharged, untreated, to the Hudson River (Brown, Jr. *et al.*, 1984).

Estimates of PCB releases at the two capacitor plants have been made on the basis of GE's overall usage of the chemical and considering discharges allowed pursuant to USEPA's discharge permit for the facilities. Figure B.2-1 illustrates the company's PCB usage, by Aroclor type, for the period 1946 to 1977. That figure shows the trend in Aroclor usage to be from relatively highly chlorinated forms in the mid-1950s (Aroclor 1254) to less chlorinated homologues in the 1960s (Aroclor 1242) and the 1970s (Aroclor 1016).

By using actual GE purchase records for the years 1966 thru 1975 and approximating GE purchases for the prior period on the basis of Monsanto's production records, GE's total PCB consumption for the period 1957 to 1975 has been estimated at 133,000,000 pounds (Limburg, 1985). Plant discharges during the 1960s have been approximated at 5 metric tons per year (Sofaer, 1976), a rate which is roughly compatible with that allowed by GE's 1975 discharge permit (30 pounds per day or about 11,000 pounds per year). Sanders (1989) provides anecdotal evidence of plant releases being less than one percent of plant consumption or less than 1,330,000 pounds from 1957 to 1975. Thus, one estimate of the range of releases to the river would be 209,000 to 1,330,000 pounds over the period 1957 to 1975, where the lower quantity is based on a continuous discharge of 30 pounds per day (or 5 metric tons/year) for a nineteen-year period.

B.2.2 Current Permitted Discharges

Six facilities in New York State, including GE, are permitted to discharge PCBs in the Upper Hudson Basin. NYSDEC (March 19, 1991) provided SPPDES permits and discharge monitoring reports (PCBs only) for each of these facilities. Table B.2-1 identifies these facilities, receiving waters and relevant information on PCB limits and measurements. Two facilities (GE, Fort Edward and James River Corporation, South Glens Falls Mill) are permitted to discharge PCBs directly into the Upper Hudson River, while one (GE, Old Fort Edward Site Remediation Project) discharges into the Old Champlain Canal in the vicinity of Fort Edward. In most cases, the concentration of PCBs in the final effluent is limited to the minimum reliable detection limit based on USEPA Method 608.

According to available Commonwealth of Massachusetts SPDES records, Sprague Electric Company is permitted to discharge PCBs (0.01 mg/l) directly into the Hoosic River, which flows into the Upper Hudson.

B.2.3 Other Sources

Table B.2-2 identifies inactive hazardous waste disposal sites located near the Upper Hudson River (above Federal Dam at Troy) in which PCBs have been dumped. This tabulation was obtained from NYSDEC, Division of Hazardous Waste Remediation, utilizing their annual inventories of disposal sites in New York State (April 1990). The NYSDEC priority classification codes stated in the table are: Code 2 - Significant threat to the public health or environment and action is required; Code 2a - Temporary classification assigned to sites with inadequate or insufficient data for inclusion in any of the other classes; Code 3 - Does not present a significant threat to the public health or environment, and action may be deferred; Code 4 -Site is properly closed and requires continued management.

The release of some contaminants from these inactive sites adds to the total PCB loadings in the Upper Hudson. In many instances, flow of surface water and groundwater from the sites are towards the Hudson River. "Unknown" material

was listed as being disposed of at many sites and is generally not included in the table. None of the sites identified by NYSDEC are classified as an imminent danger to the public or environment (Code 1). Many sites are classified as Code 2, suggesting that these sites may be potential sources of pollution to the Hudson River.

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SYNOPSIS

NATURE AND EXTENT OF CONTAMINATION

(Section B.3)

Available environmental data on the distribution of PCBs in the sediments, water, fish, air and plants of the Upper Hudson River as well as supporting data on flow and sediment transport are summarized and evaluated. As a foundation for continued analyses, available data have been compiled into a computerized, relational database management system (B.3.1).

Data on PCB concentrations in river-bottom sediments (B.3.2) are drawn primarily from the 1976-1978 NYSDEC sampling efforts and the 1984 Thompson Island Pool investigation, along with several other sources. Sediments are the major environmental repository for PCBs in the Upper Hudson, but there is a high degree of spatial variability in PCB concentrations. The 1984 study covered only the Thompson Island Pool and relatively little data have been collected since. It is difficult to determine the current mass and distribution of PCBs in sediments without further investigation.

The discussion of surface water monitoring (B.3.3) concentrates on data collected by the USGS. Transport of PCBs is affected by hydrologic processes, particularly flood events. A discussion of flow monitoring is followed by presentation of time series data, to the extent available, for suspended sediment and PCBs in the water column. Current full-year and summer average PCB concentrations are calculated, taking into account the problem of numerous measurements below analytical detection limits.

NYSDEC has monitored Upper Hudson fish on a regular basis since 1975; data are presently available for PCBs in fish through 1988. The extensive data collected in this program (nearly 3,000 Upper Hudson samples) are discussed (B.3.4). Total PCB burdens in fish declined sharply from 1978-1981. Levels of the higher chlorinated congeners in fish appear to have remained relatively constant since 1982. Results of NYSDOH macroinvertebrate monitoring are also described.

PCB monitoring data for air and plants near the Upper Hudson (B.3.5) are generally insufficient to assess the impact from PCBs in the river. Isolating the contribution of the river from other possible PCB sources is a particularly difficult problem.

For various other media there is a notable lack of monitoring data (B.3.6). Only limited groundwater sampling has been performed and surface soils near the river have not been monitored.

Data quality and analysis methods for the various monitoring programs are evaluated. PCBs have many different variations in chemical structure and differing physical properties. Uncertainties surrounding PCB measurement, particularly the specific variations in PCBs, results in considerable difficulties in interpreting the results. Furthermore, differing PCB measurement methods used for water, sediments or fish confound direct comparisons among them.

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B.3 Nature and Extent of Contamination

B.3.1 Overview of Sources and Database

Site data defining the current understanding of the nature and extent of PCB contamination, based on previous studies, are summarized in this section. Data synthesis efforts have focused on:

- obtaining the most complete, and current, data sources available;
- compiling these data into a computerized database;
- evaluating the PCB data for the media sampled;
- identifying current trends and relationships; and
- determining the adequacy of the existing data.

During the early 1970s, NYSDEC and several other agencies began the first comprehensive monitoring studies for PCBs in the Upper Hudson. Fish, which were some of the earliest environmental samples analyzed, showed high concentrations of PCBs. These early investigations began what is now over two decades of studies on PCBs in water, sediments, fish and other media affected by PCB discharges to the Upper Hudson. Table B.3-1 summarizes the major investigations.

Past USEPA documents, including the 1984 Feasibility Study and the EIS, have been reviewed for this work. Emphasis, however, was given to reviewing additional, more recent data and evaluating that along with the long-term monitoring record contained in the previous studies.

Previous investigations at the Hudson River PCB site have examined the nature and extent of contamination in several media, including fish, sediments, river water and, to a lesser extent, air. Each medium is discussed below. Additional studies, currently being performed by GE at the remnant deposit sites, are not available for this report. Before discussing each medium, a brief overview of the TAMS/Gradient database is provided below.

As the foundation for these Phase 1 data evaluation efforts as well as continued analyses during subsequent phases of the project and possibly future projects, the data gathered during Phase 1 have been compiled into a relational database management system using PC-based Paradox ™ software. This database currently contains approximately 30,000 records of information, primarily on sediments, water, fish and some other biota obtained from numerous sources (see Table B.3-1). Data input, verification and database management have been conducted by the TAMS/Gradient project personnel.

In the current sediment database, there are nearly 2,500 samples for the period 1976 through 1990. The fish database contains approximately 8,000 samples for the period 1973 through 1988. Additional data for other aquatic biota (macroinvertebrates and multiplate data) account for several hundred additional samples. Water column data, including daily and peak flow, suspended sediment and total PCBs, comprise the bulk of the remaining data in the database. The database contains a small number of samples summarizing PCB data for air and crop plants.

Data for separate media are linked primarily through sample date and location information. For each medium, data are organized such that a unique sample identification number links information among tables. Individual data tables are grouped by medium to contain similar kinds of information. As an illustration of the database format developed specifically for this Reassessment and its contents, excerpts from the sediment database tables are summarized below in four tables: Sample Information, Core Section, Chemical and Non-Chemical. Each of these tables is linked by a unique sample identification number, *e.g.*, sample ID numbers 30000, 30016 and 30032 shown here, such that sample IDs in each table correspond to data about a single sample.

The Sample table contains information about sample date, location (River Mile distance from bank and northing and easting coordinates, where available), sample type (grab versus core), the agency or investigator responsible for the analytical method(s), the reference report/location of the original data and other information as available.

Database Table Example: Sample Information Table

Sample ID	Type	M/D/YR	River Mile	Feet fr. West Bank	Northing (ft)	Easting (ft)	Sampler	Water Depth (ft)	Elev. (ft)	Ref	Agency
30000	Grab	5/21/77	168.8	330.0	1071755	685695	100			2	O'Brien & Gere
30016	Core	3/18/77	188.4	100.0	1163740	698970	100	5.8	119.6	1	O'Brien & Gere
30032	Core	3/18/77	183.4	60.0	1140410	669040	100	2.2	102.4	1	O'Brien & Gere

Core samples in the Sample table are linked with the Core Section table, which identifies the length of each core sample section and the depth beneath the river bottom, *i.e.*, the depth of sample penetration for the top and bottom of each section.

Sediment Database Example: Core Section Table

Sample ID	Core Section No.	Bottom of Section (in.)	Top of Section (in.)
30016	1	1	0
30016	2	2	1
• • •			
30016	12	12	11
30032	1	1	0
• • •			
30032	9	9	8

Selecting a sample ID from the Core Sample and Section tables and locating the same ID in the Chemical data table shows either the Aroclor results for an entire grab sample or section by section results for core samples. Additional information describing analytical measurement methods, *i.e.*, extraction method, are contained in the database as available. The Chemical data table also contains non-PCB chemical data, such as metals analyses (not shown here), where available.

Sediment Database Example: Chemical Data Table

Sample ID	Parameter	Core Section No.	Extraction Method	Concentration (ppm)
30000	Aroclor 1016		shake	1.0
30000	Aroclor 1221		shake	1.0
30000	Aroclor 1254		shake	1.0
30016	Aroclor 1016	4	soxhlet	6.0
• • •				
30016	Aroclor 1254	12	soxhlet	0.1
30032	Aroclor 1016	5	soxhlet	234.0
• • •				
30032	Aroclor 1254	5	soxhlet	163.0

Finally, non-chemical data, such as sediment texture class, percent volatile versus total solids, are contained in the Non-Chemical table.

Sediment Database Example: Non-Chemical Data Table

Sample ID	Core Section No.	Parameter	Value
30000		% total solids	78.93
30000		% volatile solids	0.85
30016	1	texture	GRAVEL
30016	4	% total solids	85.97
30016	4	% volatile solids	2.2
• • •			
30016	12	% total solids	89.23
30032	1	texture	CL-WC
• • •			
30032	5	% volatile solids	25.39

B.3.2 Sediment

Two primary sources of information provide the largest amount of data on sediment contamination in the Upper Hudson: (1) printed results (>1,000 samples) of the 1976-1978 NYSDEC sampling survey and (2) computer files (>2,000 samples) of the 1984 Thompson Island Pool survey. These data were entered into a computerized database (referenced here as the TAMS/Gradient database). Tables B.3-2 and B.3-3 provide a summary of these sediment samples in the TAMS/Gradient database. In addition to these two major NYSDEC sediment surveys, USEPA, the Lamont-Doherty Geological Observatory and GE have sampled sediments in the Upper Hudson.

B.3.2.1 1976-1978 NYSDEC Sampling

As reported by Tofflemire and Quinn (1979), NYSDEC conducted several sediment sampling surveys in the Hudson River between 1976 and 1978. Details about the sampling and analysis procedures for these studies are summarized in NYSDEC Technical Report No. 56 (Tofflemire and Quinn, 1979).

The data provided by NYSDEC contained a total of 1,167 sediment samples (396 cores and 771 grabs) taken during 1976, 1977 and 1978; 1,770 PCB analyses were reported for the 1,167 samples. The overwhelming majority of samples (1,091 of the 1,167 samples) from the 1976-78 data set were collected in the Upper Hudson River. Only five samples in this data set were obtained in the Lower Hudson River and all of these five were from River Mile 153, just south of the Troy Lock. Another sample in this set was identified as from the Lower Hudson, but other descriptions placed it in the Upper Hudson, while 70 samples had no information regarding location.

Aroclors 1016, 1221 and 1254 were identified as the PCB mixtures detected in the 1976-1978 sediment sampling effort. Total PCB concentrations were reported as the sum of these three aroclors. Analytical detection limits were not reported in this data nor was any indication given about a sample's detectable or non-detectable concentrations of PCBs. Because several concentra-

tions (1 ppm, 5 ppm, 10 ppm) occur an inordinate
set, these concentrations are the probable detecti

Table B.3-4 summarizes the Aroclor concen
samples, respectively, within each of the nine
summarizing the mean and median PCB values for the
counted as a single sample, *i.e.*, the statistics a
core length. The reported N is the count of core s
of the most striking aspects of the 1976-1978 resu
of PCB concentration over very short distances. S
apart may have PCB levels varying by orders o
variability is highlighted in Figure B.3-1, which
in surface sediments (grab samples and the top se
mile. Some trend with river miles is shown by
concentration by reach is highest in the Thompson I
194), decreasing for several miles downstream, t
River Mile = 175 to River Mile = 160. This pattern
of depositional areas between Stillwater and North

Based on the results of the 1976-1978 surve
spots, areas containing more than 50 ppm total PCI
Pool survey re-evaluated the hot-spot locations a
a series of over 100 polygons containing PCBs gre
spot" term is used here only as a frame of refere
sediment areas previously defined as containing high
to the 1976-1978 study, questions about the adequa
(NUS, 1984), as noted below:

- PCB concentrations exhibited wide spat
  single transects not uncommonly ran
  concentrations to greater than 1,000
- The sampling density was so low tha
  variability of the PCB concentrations,
  delineation is questionable.

- Changes in sediment deposits caused by the dynamics of the river greatly complicates comparisons between PCB concentrations in similar locations in different years.
- A major flood event that occurred in 1979 redistributed sediments significantly, again calling into the question the usefulness of the 1976-1978 sediment data, other than general purposes.
- The analytical techniques used to quantify PCBs have improved since this data was collected.

B.3.2.2 1984 NYSDEC Sampling

Methods and Procedures

In 1984, NYSDEC again undertook an extensive sediment sampling program. This effort focused on the Thompson Island Pool (M. P. Brown *et al.*, 1988b). The objective of this study was to identify areas of contaminated sediments that would be removed during the Hudson River PCB Reclamation Demonstration Project. Primarily these areas were the 20 hot-spots previously identified in the Thompson Island Pool and other areas with known or suspected high PCB concentrations.

The investigators identified 1,260 sampling locations in the approximately five-mile reach of the river. Many of these locations were determined by imposing a 125-foot triangular grid on previously defined hot-spots and areas that had PCB concentrations in excess of 50 ppm during the 1983 USEPA survey (NUS, 1984). In addition, sample locations were selected based on known or suspected sediment depositional areas, as indicated by location in the river and bathymetry measurements. Sample locations in the field were determined electronically by using a microwave locating system and generally agreed with predetermined locations.

Samples were collected by Normandeau Associates, Inc. between August 24, 1984 and November 30, 1984. In addition, 21 cores were collected during February 1-4, 1985. These later samples were taken through the ice on the river at locations that had been inaccessible by boat.

Whenever possible, the investigators collected core samples. At those locations where insufficient sediment was available for core samples, grab samples were attempted. At 80 locations, bedrock or coarse material precluded either sample method. In all, 1,016 locations throughout the Thompson Island Pool were sampled and yielded 674 grab and 408 core samples.

The depth of penetration for the core samples was fairly uniform, with an average depth of approximately 31 inches. The investigators divided core samples into sections based on a desire to define contaminated layers without introducing dilution from adjacent less contaminated layers. Sections were also chosen based on potential dredging considerations and a need to limit the number of chemical analyses.

NYSDOH and Versar, Inc. measured physical and chemical parameters of the sediments collected in this study. NYSDOH determined lengths of cores and sections, percent dry solids, dry specific weight (density) and textures, which were determined visually. Versar measured percent volatile solids and performed the gas chromatograph analyses for PCBs.

PCB Results

Versar reported PCBs as Aroclors 1242, 1254 and 1260 using the method of Webb and McCall (1973). Although the data contained a total PCB quantification, no mention is made in M. P. Brown *et al.* (1988b) of the method used to quantify or calculate this total. Examination of the data received indicates that the total was not simply the sum of the three Aroclor mixtures quantitated.

Wide variations in PCB concentrations in sediments were observed in the 1984 NYSDEC study throughout the Thompson Island Pool, even though sampling concentrated on areas of known contamination. The discrepancies between means and medians for both grabs and cores indicate that PCB concentrations have a highly skewed distribution over the area of the Thompson Island Pool. (Environmental monitoring results frequently exhibit this skewed pattern and a log-normal distribution is often a good approximation of the data.) Both grab

and core samples had significantly higher concentrations in the least chlorinated fraction that was quantified (Aroclor 1242) than in the more chlorinated fractions. Table B.3-5 provides Aroclor and total PCB summary statistics for both grab and core samples. Aroclor 1242 was the predominant Aroclor reported for these samples, with lower levels of Aroclors 1254 and 1260 also identified. On average, total PCBs in the samples were approximately 55 ppm, with maximum levels of >1,000 ppm detected in several samples.

Table B.3-6 presents the results of texture classifications determined by NYSDOH. Considering grab and core samples together, Thompson Island Pool sediments were classified most often as either gravel or fine sand, with a significant fraction of fine sand/wood chips and clay samples, particularly for core samples.

Because of their high adsorption (partition) coefficients, PCBs are generally expected to associate with the organic carbon fraction of the sediments. Although no measurements of organic carbon content were made as part of this study, organic carbon and organic matter content, which were measured frequently in the study as percent volatile solids, can be correlated. Thus, comparing PCB levels with organic matter content (volatile solids) provides a surrogate for comparing PCB concentration to sediment organic carbon content.

Figures B.3-2 through B.3-4 show the relationships between PCB concentrations and percent volatile solids within a texture classification for the three most commonly occurring textures (gravel, fine sand, fine sand/wood chips). Very little relationship appears to exist between total PCBs and volatile solids measured in the gravel texture class (Figure B.3-2). The fine sand (Figure B.3-3) and fine sand/wood chip (Figure B.3-4) categories exhibit a better correlation between PCB concentration and percent volatile solids, although percent volatile solids would still make a poor predictive measure of PCB concentration.

Comparison of 1976-78 and 1984 Studies

As mentioned previously, the 1984 study focused on the 20 hot-spot areas in the Thompson Island Pool as defined by areas exceeding 50 ppm PCBs in the 1976-1978 survey. M. P. Brown *et al.* (1988b) describe in detail the differences found between the two surveys. The PCB concentrations in sediments of the Thompson Island Pool exhibited lower concentrations in the 1984 survey than in the earlier study, as shown by the somewhat higher frequency of PCBs detected at <25 ppm (Figure B.3-5). A direct comparison of the relative frequency plot for the two studies is hindered by the fact that the 1984 survey specifically targeted potentially contaminated areas and areas with fine-grained sediments, which were thought to contain more PCBs. Thus, 1984 samples were potentially more heavily biased in a statistical sense to those areas of PCB contamination. If samples had been taken randomly from the Thompson Island Pool, the results would likely have yielded an even larger relative number of samples with lower concentrations.

Because of the different scope and sampling density of the two surveys and their different sampling and analytical methods, M. P. Brown *et al.* (1988b) indicate that direct quantitative comparisons between samples collected in similar areas are problematic. M. P. Brown *et al.* found that areas of high PCB concentration determined in the 1976-1978 survey appeared to be generally confirmed by the 1984 survey. Based on area-weighted average PCB concentrations in 138 polygon areas, these investigators calculated a total PCB mass of 23,200 kg (51,040 lb) in the top 1.5 m (59 inches) of Thompson Island Pool sediments (M.P. Brown *et al.*, 1988b). This mass estimate compares to Malcolm Pirnie's estimate in 1978 of approximately 61,000 kg (133,670 lb) for all of the Pool and approximately 48,000 kg (104,870 lb) for the 20 hot spots (Malcolm Pirnie, 1978). M. P. Brown *et al.* (1988b) offer as possible explanation for the varied estimates the differences in analytical PCB methods, depth integration/area averaging and sediment densities used.

The importance of random variability and sampling density for either the 1976-1978 or 1984 studies also complicates comparisons between the two surveys and affects mass calculation comparisons. For example, GE indicates that they have taken 30 samples from polygon 5 of the Thompson Island Pool (General Electric, John Claussen letter to USEPA, March 29, 1991). This polygon was estimated by M. P. Brown *et al.* (1988b) to have contained approximately seven percent of the PCB mass in the Thompson Island Pool based on two samples containing 39.7 and 6,587.8 ppm PCBs, yielding an average concentration of 2,437 ppm. GE has indicated that the average, based on their 30 samples in this polygon, is less than 20 ppm. (Results are not yet available to the project study team.) Although PCB mass differences in this one small area of the Thompson Island Pool do not necessarily mean that the overall conclusions of the 1984 survey are incorrect, they do suggest that:

- wide variations in PCB concentrations occur over relatively short distances;
- direct quantitative comparisons of PCB levels in samples from different years are problematic; and
- the mass and distribution of PCBs in the Upper Hudson are difficult to quantify.

B.3.2.3 Lamont-Doherty Geological Observatory Investigations

The Lamont-Doherty Geological Observatory, under contract to the NYSDEC, conducted a survey of PCB levels in the sediments, suspended matter and water column of the Upper Hudson River during 1983 and 1984 (Bopp *et al.*, 1985). The survey, a coring effort, collected 16 cores, covering the Upper Hudson from above Hudson Falls to the Albany area (Plate B.3-1), and analyzed them for radionuclides. Procedures were the same as those described in Section A.3 for the Lower Hudson. Many sections of these cores were analyzed for PCB levels, with an emphasis on homologue and congener-specific information. The investigation also involved PCB analyses of surface water samples (see Section B.3.3).

On the basis of these data, Bopp *et al.* (1985) were able to draw a number of important conclusions concerning the fate of PCBs in the Upper Hudson. In cores showing interpretable radionuclide chronologies, the occurrence of a maximum PCB concentration in sediments deposited circa 1973 (the removal of the Fort Edward Dam) could be seen in all areas of the Hudson, including the Thompson Island Pool (Figure B.3-6). This data demonstrated that the sediments of the Upper Hudson could be used to determine PCB transport history.

Analysis of Upper Hudson sediments revealed that very recent sediment deposits (1980-1983) contained PCB congeners in ratios very similar to Aroclor 1242 and 1016, whereas older sediments, typically under anaerobic conditions, showed substantially different congener ratios. In general, these older sediments contained higher levels of mono through tetrachlorobiphenyls and lower levels of the higher chlorinated congeners, relative to a standard Aroclor 1242 mixture. Bopp *et al.* (1985) concluded on the basis of these results, that under anaerobic conditions, biologically driven dechlorination must be occurring. However, little or no dechlorination was occurring under the aerobic conditions of the uppermost sediment layers.

B.3.2.4 Other Studies

1983 USEPA Study

In August 1983, USEPA conducted a limited study to collect samples from locations that had been sampled in 1976-1978 (NUS, 1984). Sixty-six samples, of which 54 were core and 12 were grab samples, were collected within a nine-mile stretch of the river south of Rogers Island, including the Thompson Island Pool. Forty-two samples were collected from within or on the border of previously determined hot-spots. The results of this study tended to show that areas of high PCB concentrations in 1976-1978 exhibited high concentrations in 1983 as well. Nevertheless, direct comparisons between samples taken within 50 - 100 feet of each other during the two surveys indicated that variations by two orders of magnitude were not uncommon (NUS, 1984). In general, the concentrations observed in 1976-1978 were greater than those seen in 1983 at corresponding locations.

GE 1989 Baseline Studies (Remnant Remediation Project)

As part of the Remnant Remediation Project, General Electric conducted baseline pre-remediation sediment monitoring. Sediment samples were collected at five locations in the vicinity of the remnants: one location near Rogers Island; one location far upstream; one location between the remnants and Bakers Falls; and two downstream locations near Lock 6 and Waterford. Median PCB concentrations in river sediments in the remnant areas ranged from 0.47 ppm to 34 ppm, with a maximum of 99 ppm (Table B.3-7). Downstream samples contained median PCB concentrations of 0.64 to 2.1 ppm, whereas the control location had median PCB levels (seven samples) of 0.11 ppm. The sample location between Bakers Falls and the remnants had a median PCB concentration of 1.4 ppm. With the exception of the two downstream locations, PCBs were detected in all samples. The chromatograms were compared against Aroclor mixtures 1221, 1232, 1016, 1242, 1248, 1254 and 1260; Aroclor mixtures in the samples were reported to be Aroclors 1242 and/or 1254.

GE 1990 Sediment Sampling (Bioremediation Investigations)

General Electric has been conducting extensive research on biological dechlorination and/or degradation processes occurring within the river, which may have altered the composition of the PCB Aroclor patterns within the sediments. In conjunction with these studies, GE has recently collected samples from selected areas of the Upper Hudson for more detailed evaluation. General Electric provided preliminary results of their sediment sampling activities (Claussen, 1991b).

Harza Engineering, GE's contractor, collected 103 core samples from 12 hot-spots during 1990 and reported 275 PCB analyses. From three to eight cores were collected at most locations, with the exception of GE's H-7 location where 62 cores on a 12 x 12 foot grid were sampled. Samples were analyzed for PCB homologue groups and five Aroclors (1221, 1242, 1254, 1260 and 1268). The results of this sampling are summarized in Table B.3-8. With the exception of H-7 and Location 4, the median PCB concentration at the four other locations

within the Thompson Island Pool exceeded 100 ppm. Median PCB concentrations for the aforementioned two locations and the locations downstream of the Thompson Island Pool were less than 100 ppm.

Coincident samples and PCB measurements from the same hot-spots in the NYSDEC 1976-78, USEPA 1983, and GE 1990 samples are available for only six locations. The average PCB concentrations for each of these three surveys are summarized in the lower portion of Table B.3-8. In four out of six locations, the GE samples indicate average PCB levels above both the 1976-1978 and 1983 values; the remaining two locations show 1990 PCB levels lower than the 1976-1978 and/or 1983 results. Because of the very small sample sizes, few coincident locations and difficulty in determining whether these samples represent similar sediment zones over time, these results are inadequate to suggest a clear trend. Qualitatively, the results document the continued presence of PCBs in areas originally defined in 1976-1978 to be contaminated.

B.3.2.5 Other Chemicals in Sediments

In addition to PCBs in river sediments, other chemicals, particularly heavy metals, have been measured during 1976-1978 (Tofflemire and Quinn, 1979), 1984 Thompson Island Pool study (M. P. Brown *et al.*, 1988b) and by other investigators. Lead, cadmium, zinc, chromium, mercury and other metals have been measured. M.P. Brown *et al.* (1988b) indicate that anthropogenic sources, including a pigment manufacturer in Glens Falls, elevate lead, cadmium and chromium in river sediments above their naturally occurring levels. Based on their 1984 study, M.P. Brown *et al.* (1988b) reported mean metals concentrations in sediments for lead (217 µg/g), cadmium (21.6 µg/g), chromium (475 µg/g) and mercury (1.96 µg/g) along with several other chemicals. M. P. Brown *et al.* found that although lead and cadmium are the two metals most frequently found in sediments, standard leaching tests, *e.g.*, EP Toxicity, suggest they are not readily leachable.

Relatively few sediment samples have been tested for other organic priority pollutants. Four sections of two cores collected in 1983 by Dr. Richard Bopp from River Miles 188.5 and 191.1 (Thompson Island Pool) were submitted to NYSDOH and analyzed for dioxin and dibenzofurans. Six sediment samples collected in 1987 from three hot-spots were analyzed for dioxins, dibenzofurans, volatile and semi-volatile organics and pesticides (M. P. Brown *et al.*, 1988). With the exception of dibenzofurans, none of the other organic parameters were detected in the 1987 samples.

As reported by M. P. Brown *et al.* (1988b) tetrachlorodibenzo-p-dioxin (TCDD) and tetrachlorodibenzofuran (TCDF) as well as their 2,3,7,8- isomers were detected at less than part per billion levels in the 1983 samples. Total TCDD in these 1983 samples ranged from non-detected to 0.135 ppb; total TCDFs ranged from non-detected to 0.731 ppb. In two of the six 1987 samples, total TCDFs were detected at <0.2 ppb levels; TCDDs were not detected in the 1987 samples; detection limits ranged from 0.012 - 0.058 ppb. M. P. Brown *et al.* indicate that possible sources of the TCDFs in sediment include residual fall-out from coal and wood combustion, discharge from wood processing plants (by-product of chlorophenol pyrolysis) and discharge of chemical-wastes containing TCDDs and TCDFs as trace contaminants. Industrial PCB mixtures are known to contain trace levels of TCDFs.

B.3.2.6 Discussion

The study team encountered some difficulty in matching the contents of the data entries in the TAMS/Gradient database, especially the 1976-1978 sediment data, with data summaries provided in previous reports (Tables B.3-2 and B.3-3). The 1976-1978 raw data in printed form did not contain identification designations cross-referenced to date and sample location and no report containing such a cross-referenced summary was found. Sample identification numbers are shown on a marked-up copy of the 1977 Normandeau Associates, Inc. map in the NYSDEC offices, but do not cross-reference dates of the samples or laboratory identification numbers. The team could assess the completeness of the data set only by comparing sample dates and river location associated with the samples

with summaries in published reports. As shown in Table B.3-2, these comparisons identify some inconsistencies between the data in the TAMS/Gradient database and the number of samples reported by Tofflemire and Quinn (1979) in NYSDEC Technical Paper No. 56. For example, the TAMS/Gradient database contains 254 grab samples and 21 core samples reportedly collected in 1976, whereas Tofflemire and Quinn report 24 cores and 80 unspecified sediment samples for 1976. For 1977, the TAMS/Gradient database contains 446 grab and 246 core samples compared to 692 grab and 208 core samples reported by Tofflemire and Quinn (1979). For all samples collected between 1976 and 1978, the database contains 1,092 samples compared to 1,404 indicated by Tofflemire and Quinn.

Differences in the overall number of samples may be accounted for approximately by noting that:

- approximately 202 of the 672 summer 1977 grab samples taken by Normandeau Associates, Inc. (NAI) were not analyzed for PCBs and were not provided in the printed data summaries supplied for the TAMS/Gradient database;
- 200 spring 1978 remnant samples from Malcolm Pirnie, Inc. (MPI) were not provided for use in the TAMS/Gradient database.

If these 402 samples are subtracted from the Tofflemire and Quinn total of 1,404, there would be approximately 1,000 samples, more closely approximating the 1,092 samples in the TAMS/Gradient database. A comparison of the number of samples by river mile (Table B.3-3) also indicates differences in total numbers of samples reported by Tofflemire and Quinn and the data in the database. Because 15 years have passed since the 1976-1978 samples, their use for identifying precise locations and concentrations of contaminated sediments is limited.

The data summaries contained inconsistencies within samples and between samples. For example, the sums of individual core lengths did not always match the total core length; for approximately 3 percent of the samples, the data fields were incorrect; approximately 10 percent of the samples had no information

about northing-easting coordinates. The team could not evaluate independently the reason for the discrepancies nor the accuracy of the original data archival process/data summaries. These discrepancies in the sediment database are not considered to be significant at this time and an effort to resolve them will continue during the course of the project.

Differences between the TAMS/Gradient database and the results reported by M. P. Brown *et al.* (1988b) for the 1984 Thompson Island Pool survey appear to be slight. The database contains 1,141 samples, whereas M. P. Brown *et al.* report 1,205 for this survey, a difference of ~5 percent in total samples.

B.3.3 Surface Water Monitoring

Numerous surface water monitoring stations along the Upper Hudson are maintained by the USGS. These stations have monitored flow, suspended sediment, PCBs and other water quality parameters. The USGS data, obtained from WATSTORE and the Albany USGS office, provide the longest and most comprehensive record of surface water data for the Upper Hudson.

B.3.3.1 USGS Flow Records

The USGS has collected river discharge (flow) and water quality data at various points along the Upper Hudson River (Plate B.1-1). The USGS records of the monitoring stations located on the Hudson between Hadley, well above Fort Edward, and Green Island, below the confluence with the Mohawk River at Troy, were obtained for use in this investigation.

The majority of the USGS flow monitoring stations on the Upper Hudson have periods of record beginning in the 1970s, although continuous monitoring is available at Hadley since 1921. A lack of widespread flow measurements for earlier periods presents difficulties in analyzing the longer-term flow regime and flood probabilities. In particular, the flow record at Fort Edward, at the upper end of the Thompson Island Pool, commences only in 1976. No USGS monitoring is available at the Thompson Island Dam. Barge Canal stage data are

available at the guard gate at Crockers Reef (Gauge #118) in the Hudson River/Champlain Canal approximately parallel to the Thompson Island Dam, at the lower end of the pool. At the northern end of the Thompson Island Pool, Barge Canal stage data are available below Lock 7 (Gauge #119). These gauges report water elevations in reference to the Barge Canal datum. They have not been calibrated to river discharges and provide only qualitative data regarding flood discharges.

In order to extend the record of flow data for the Upper Hudson, it is necessary to move upstream to the confluence of the Hudson and Sacandaga Rivers, near Hadley (see Plate B.1-1). A monitoring station has been maintained on the Hudson at Hadley since July 1921. The Sacandaga River, a major tributary entering just below Hadley, has been monitored since 1907 at Stewarts Bridge near its confluence. By adding these two stations, USGS provides an estimate of the flow in the Hudson below the confluence with the Sacandaga. Between this point and Fort Edward there are several dams, but there are few additional tributaries. The drainage area above Fort Edward is 2,817 square miles, while that of the Hudson River at Hadley plus Sacandaga River at Stewarts Bridge is 2,719 square miles, representing only a 3.6 percent increase in contributing area. Estimates of flow in the Hudson below the Sacandaga, thus, provide an accurate estimate of the magnitude of flow at Fort Edward.

The daily average flow value records for USGS Upper Hudson stations are summarized in Table B.3-9, from upstream to downstream. USGS flow monitoring is available since 1976 only at the upstream end of the Thompson Island Pool (Fort Edward at Rogers Island). The closest functioning USGS monitoring station downstream is at Stillwater, which is 26 miles and three dams south. Thus, Fort Edward monitoring is most informative of hydraulic conditions in the Thompson Island Pool. Mean daily flow at Fort Edward is 5,244 cfs; daily flows range from 652 cfs to 34,100 cfs for the period of record. Additional inputs from tributaries and runoff increase the average daily flow to 7,933 cfs at Waterford and 13,642 cfs at Green Island, below the confluence of the Mohawk River.

Figures B.3-7a and B.3-7b display the daily average flows for 1973-1990. As Fort Edward monitoring commenced in water year 1977, the 1973-1976 flows are estimated from the calculated flows below Hadley, representing 97 percent of the contributing watershed area at Rogers Island. This record reveals the presence of several major flood events, which were associated with mass erosion of the remnant deposits. These events occurred on: April 2, 1976, when a daily average flow of 39,340 cfs was reported below Sacandaga River; April 29, 1979, when a flow of 31,700 cfs was reported at Rogers Island; and May 2, 1983, when 32,600 cfs was reported at Fort Edward. During these flood events flows were even higher for shorter time periods (peak flows). The maximum peak flows monitored at Fort Edward since December 1976 are 34,000 cfs on April 29, 1979 and 35,200 cfs on May 3, 1983.

The daily flows show evidence of a strong weekly periodicity. The seven-day cycle is the result of regulation of the Sacandaga Reservoir to supply power plants during the week, while maintaining the weekend recreational pool in Sacandaga Lake.

B.3.3.2 Suspended Sediments Monitoring

Information on time trends in suspended sediment data as well as the relationship between sediment and discharge is provided by USGS monitoring stations (Plate B.1-1). Several water quality stations were established on the Upper Hudson in 1969, but measurements of suspended sediment did not commence until 1975. Monitoring is not continuous or on a set schedule and there has been a tendency to focus on spring flood periods, with little data available for the winter months. Lack of a more extensive database and of regular time series creates difficulties in analyzing sediment data as well as other water quality parameters.

Summary statistics on the USGS suspended sediment monitoring for stations between Fort Edward and Waterford are given in Table B.3-10. The median suspended sediment concentration in the Upper Hudson above the confluence of the Mohawk ranges from 4 to 12 mg/l and increases downstream. Relationships between

suspended sediment levels and river flow are discussed in B.4.

B.3.3.3 USGS PCB Monitoring

Methods and Procedures

Regular monitoring of PCBs in the water column in the Upper Hudson was instituted by the USGS in late 1975 at Waterford and expanded to other upstream stations in 1977. Most other sampling programs, discussed later in this section, have been of short term duration. A recent search of the STORET database reveals that limited water-column PCB measurements are also available for some of the tributary rivers to the Upper Hudson. These data have not been reviewed. The USGS data are, thus, the primary source of time series information indicative of trends in water-column PCB concentrations.

USGS observations of PCB concentrations in the water column have been made at most of the same water quality stations as for sediment data (Plate B.1-1). Data sets of significant size are available at Fort Edward (River Mile 194.5), Schuylerville (River Mile 181), Stillwater (River Mile 168) and Waterford (River Mile 156.5), with a limited record at Fort Miller (River Mile 187). In addition background samples are taken upstream at Glens Falls (River Mile 200).

The original purpose of the USGS monitoring was to gather several years of data on PCB concentrations prior to removal of contaminated sediments (Schroeder and Barnes, 1983). Although this dredging plan has been delayed, monitoring has continued. Data are now available on WATSTORE, a USGS-maintained computerized database, through the end of water year 1989 (September 1989). Data for water year 1990 were collected by the USGS and have been turned over to NYSDEC for analysis, but the results are not yet available.

Methods of data collection and analysis are summarized in Turk and Troutman (1981) and Schroeder and Barnes (1983). According to the latter source, samples from the Upper Hudson were collected from bridges using depth-integrating samplers. The sampler held a wide-mouth glass bottle, which was lowered and

raised through the water column to obtain a depth-integrated sample (about 1 liter) for PCB analysis.

The USGS National Water Quality Laboratory in Doraville, GA performed the PCB analyses. Comparison was made to standard Aroclor mixtures. Results, however, were reported as total PCBs. Schroeder and Barnes (1983) reported that PCBs in the Hudson were "almost always in the composition range from [Aroclor] 1232 to 1248," but recognized that natural processes had likely altered the congener composition of the original Aroclor mixture.

Although the USGS laboratory reports a theoretical detection limit of 0.01 µg/*l* through water year 1983, the practical quantitation limit was considered to be 0.1 µg/*l*, because of the small size of the water sample (Bopp *et al.*, 1985). Data for this period recorded on WATSTORE contain both values entered as 0 and values coded as <0.10 µg/*l*. Apparently these are both intended to represent non-detects at the 0.1 detection level and the inconsistency is unintentional (Rogers, pers. comm., 1991). With water year 1984, the practical detection limit was lowered to 0.01 µg/*l*. Nevertheless, the 1984 and 1985 data are reported on WATSTORE *as if* they adhere to the previous detection limit of 0.1 µg/*l*. In 1986, the detection limit began to be reported as 0.01 µg/*l* in WATSTORE.

PCB Results

The USGS monitoring station at Glens Falls provided upstream background levels of PCBs in the Hudson for 1977-1983. Of 45 observations for total PCBs, only two had detectable levels of PCBs. These observations occurred on December 5, 1978 and September 28, 1980 and were both reported as 0.1 µg/*l*.

Summary statistics for the USGS monitoring of total PCBs in the water column between Fort Edward and Waterford are given in Table B.3-11. A remarkable fact evident from this table is that there is rather little variation in measured PCB concentration by river mile between Schuylerville and Waterford. At all stations, PCBs were detected in more than 60 percent of the water samples, with detection frequencies ranging from 63 percent to 89 percent. Average PCB

concentrations below Fort Miller range from a high of 0.29 µg/l at Stillwater to 0.23 µg/l at Waterford. Averages at Fort Miller are not directly comparable to other stations because of the short period of record. The average at Fort Edward, inflated by one very high measurement (77 µg/l), was calculated in Table B.3-11 by omitting this outlier. This procedure yields a long-term average approximately half of that observed downstream. Concentrations observed at Schuylerville, Stillwater and Waterford are approximately constant, although the average at Waterford is lower than the up-river values, primarily because of dilution from the Hoosic River. Indeed, the average concentration at Stillwater (0.29 µg/l) is slightly higher than that at the Schuylerville station upstream (0.26 µg/l). This finding suggests that there may be relatively little loss of water-column PCBs during transit in the Upper Hudson.

At the Green Island station, downstream of the confluence of the Mohawk River, the contributing watershed area is nearly double that at Waterford and PCB concentrations are correspondingly diluted. Twelve samples were analyzed for total PCBs between 1978 and 1985 and all were non-detects at the 0.1 µg/l level. No PCB measurements have been reported at Green Island since March 1985.

Figures B.3-8 through B.3-11 show the time series of PCB observations at Fort Edward, Schuylerville, Stillwater and Waterford, respectively. Observations reported as non-detect (or zero) are plotted at the detection limit (0.1 µg/l through September 1986 and 0.01 µg/l thereafter). While much of the variability observed near the detection limit may represent analytical noise, there is a clear similarity apparent between the PCB time-series plots at Schuylerville, Stillwater and Waterford (B.3-9 through B.3-11), particularly in the marked response to the 1979 spring flood. Despite the fact that samples were taken at somewhat erratic intervals, field personnel seem to have frequently visited each Upper Hudson station in succession, so that many samples for the whole reach, while not contemporaneous, are close together in time. All three stations are below the Thompson Island Pool. Although the response to floods appears somewhat different at Fort Edward, located below the remnant deposits but above the so-called hot-spots in the Thompson Island Pool, the 1979 event stands out.

Also notable in these time series is a general decline in water-column PCB levels from about 1979-1986. The question of whether there has been any genuine trend in PCB loading to water over time, or whether the apparent year to year trends are actually due to variability in the hydrologic regime, is discussed in Section B.4.

Current Full-Year Average PCB Concentrations in Water Column

Estimates of current average concentration of total PCBs in the water column are needed for the assessment of potential baseline health risks and other environmental impacts. As noted previously, seasonal variability and lack of continuous sampling confound the estimation of this average.

It is common practice when working with data in time series containing non-detects to develop an average based on treating the non-detects as if they were equal to one half of the detection limit. A more sophisticated way to approach this question is to use the Adjusted Log Normal Maximum Likelihood method of Cohn (1988), which overcomes the bias in the sample collection and variance in the data. Another method for including non-detects in the estimate of the mean, which however does not address the problem of sampling bias, is to use a log-probit analysis (Helsel and Cohn, 1988). Under the assumption that the logs of the data are normally distributed, they will fall on a straight line when plotted on a probit (probability) scale. The log-probit analysis essentially uses regression to extend this line past the detection limit to predict the values of the non-detect samples predicted by the observable part of the distribution.

In order to examine the sensitivity of the mean to the non-detects, the above three methods of calculation were used here: (1) simple mean with non-detects at 1/2 the detection limit; (2) the Adjusted Maximum Likelihood method; and (3) log-probit analysis. Results of all three methods and the upper 95 percent confidence limit on the estimates of the means, which is used in the risk analysis, are shown in Table B.3-12.

Little difference in estimates of the mean and the 95 percent upper confidence limit on the mean is produced by any of the three methods. Mean PCB levels in recent 1986-89 water samples are on the order of 0.05 µg/*l* at Fort Edward and drop to approximately 0.03 µg/*l* at Waterford. Upper 95 percent confidence limits on these means are not much higher, approximately 0.075 µg/*l* at Fort Edward and 0.035 µg/*l* at Waterford.

Summer Average PCB Concentrations

Average water-column PCB concentrations can be calculated from monitoring data based on either whole-year monitoring, including flood periods, or based on low-flow or seasonal monitoring only. Assessing average concentrations for the summer period, after the spring floods, is likely to be of greatest interest for assessing biological impact. This period has maximum biological production and is also the season during which most of the fish samples have been collected. There is also evidence to suggest that spring flood PCBs are largely sorbed on sediment particles, whereas concentrations associated with low flows are primarily dissolved or sorbed to very fine particles, *i.e.*, pass a 0.45-µm filter. Thus, they are more readily available to enter the food chain (Bopp *et al.*, 1985).

Average PCB concentrations during the summer (June-September) have been calculated. In doing so, the presence of many non-detects among the samples was addressed. The robust log-probit analysis method (Helsel and Cohn, 1988) was used to estimate averages in the presence of non-detects. No systematic bias toward higher concentration events is expected to apply to the summer observations, although this may be the case for spring observations. In 1986 there were no summer observations at Stillwater or Waterford, while summer observations at Schuylerville in this year as well as at Waterford in 1985 were all less than or equal to the detection limit of 0.1 µg/*l*. In the latter case the log-probit method cannot be used and the average has been arbitrarily set to one-half the detection limit. The calculated summer average PCB concentrations are shown in Figure B.3-12 and summarized in Table B.3-13. Recent 1988-1989 summer average PCB concentrations are fairly uniform for all locations and are on the order of

0.03 - 0.04 ug/l. Summer average concentrations at Fort Edward tend to be less than or equal to those downstream, whereas for the full year concentrations, including spring runoff, the average concentration at Fort Edward is higher than that found downstream.

B.3.3.4 Other Sources of Water Column Data

Waterford Treatment Plant Data

The City of Waterford operates a water works serving a population of approximately 12,000 persons in the Towns of Waterford and Halfmoon and the Village of Waterford. This is the first water treatment facility downstream of Fort Edward drawing water from the Hudson. In 1975, when the USGS began collecting PCB data in the river at Waterford, they also began collecting raw water input and finished water output data at the Waterford treatment plant, in cooperation with the Board of Water Commissioners of the Town of Waterford and the NYSDEC (Schroeder and Barnes, 1983b). The water for the treatment plant is drawn from a location 0.5 km upstream of the US Highway 4 bridge, where Hudson River water samples are also taken.

Data collected in cooperation with the USGS run through the end of water year 1983. In addition, approximately bimonthly data for November 1983 - February 1985 and March 1987 - October 1989 were available from the Waterford Water Works (Metcalf & Eddy, 1990). The data through 1983 were often collected concurrently with samples from the Hudson River at Waterford and can be used to provide a check on that data. Since September 1982, no PCBs have been reported above the detection limit (generally 0.1 µg/*l*), in either raw intake water or treated water.

Lamont-Doherty Study of 1983

A detailed study of PCB transport in the Upper Hudson, conducted by researchers of the Lamont-Doherty Geological Observatory in 1983 (Bopp *et al.*, 1985), involved an investigation of spring/summer 1983 PCB transport in the Upper

Hudson, which was a period of relatively high flows. The Lamont-Doherty study included the collection of data not available from USGS sampling. In addition to sediment cores, this study included 20 large-volume filter samples of suspended matter and fifteen 9-20 liter unfiltered samples containing water and suspended matter, collected from Troy to Glens Falls. Unlike USGS monitoring, detailed component analysis was undertaken for these samples. The data collected were used to form an empirical transport model, which used as input USGS measures of suspended sediment and flow, to predict total PCB load.

PCBs in samples of suspended matter were found to match standard Aroclor mixtures, *e.g.*, Aroclor 1242, reasonably well. Samples taken during a high-flow period were found to have significantly higher PCB levels (6.33 ppm) than those taken during a low-flow period (0.69 ppm). Some of the drop in PCB levels at locations downstream of Fort Edward was attributed to dilution from tributaries joining the Hudson.

Water samples were filtered and both filtrate and particles were tested for PCBs. Comparisons of water versus suspended matter PCB concentrations were made to derive a distribution coefficient. An inconsistency was found in that the concentration of PCBs on suspended matter filtered from the 9-20 liter samples was generally two to four times higher than the levels from the large-volume filter samples collected at the same locations on the river. This discrepancy could possibly be accounted for by an equilibration between water and suspended matter during storage of water samples prior to filtration and testing. Another possible explanation was the difference in filter size used to collect the particulate sample (1.2 microns) and to separate the water samples (0.7 micron).

Seasonal variations in concentrations of PCBs in water were found. Pre-spring runoff showed the lowest PCB levels. Summer samples showed higher PCB concentrations, accounted for by boat traffic and increased use of locks, which would resuspend sediments.

Bopp *et al.* (1985) were able to derive *in situ* partition coefficients for PCBs on a quasi-homologue basis, based on packed column analysis. These analyses also indicated the possibility that water-column PCB distributions may not reflect equilibrium conditions and that dissolved phase PCB concentrations may be higher than those predicted by equilibrium. The homologue distribution of PCBs on suspended matter was readily interpreted as Aroclor 1242-like, similar to PCBs in the aerobic sediment layers. The dissolved phase homologue distributions were not as readily explained, although preferential partitioning of the lower chlorinated homologues to the dissolved phase appeared to be a likely possibility, *i.e.*, lower chlorinated homologues have lower partition coefficients and, therefore, higher levels in the dissolved phase. The greater volatility of the lower chlorinated PCBs and their production in the anaerobic sediment layers could possibly confound this interpretation. The general homologue pattern agreement between water column and surface sediments led Bopp *et al.* (1985) to conclude that little or no release of PCBs from the anaerobic sediments was occurring on a substantive basis in comparison to the mixing and resuspension of the surficial sediments.

Bopp *et al.* concluded that PCB transport is tied to suspended matter transport, with the majority of PCB transport to the Lower Hudson occurring during the 10 to 20 days per year of major sediment transport.

NYSDOH Water Column PCBs

As part of their macroinvertebrate sampling program, NYSDOH also collected water samples and analyzed them for PCBs. This and other recent data received from Dr. Bush at NYSDOH, require additional evaluation. The macroinvertebrate studies are discussed in a later section.

Remnant Deposit Containment Monitoring Program

As part of remedial activities at the PCB remnant deposits at Fort Edward, General Electric is conducting a baseline environmental monitoring program, which will continue through and follow the in-place containment of Remnant Deposits 2,

3, 4 and 5 in accordance with the Administrative Order on Consent II CERCLA-90224. The baseline monitoring includes sampling of many environmental matrices, including water column PCB concentrations. Results of the first phase of this effort cover August through December 1989 (Harza Engineering, 1990).

During the baseline monitoring, samples were taken weekly or biweekly at ten water quality stations. Two of these stations are upstream of Fort Edward, one in the Sherman Island Pool and one just below Bakers Falls, while five are in the area of the remnant deposits at Fort Edward above Rogers Island. Station E-5 was located downstream of the Route 197 bridge on Rogers Island and is just below the USGS monitoring station. The remaining two stations were at Channel Marker 175, below Fort Miller Dam and Lock 6 (E-6), and at Channel Marker 13, two miles north of a NYSDEC boat ramp at the Erie Canal and Hudson River confluence, near Waterford (E-7). Raw water samples were analyzed for PCBs with an approximate analytical detection limit of 0.1 µg/*l*. In addition, dialysis bags, filled with 4 ml of hexane to concentrate PCBs, were suspended in the water column and analyzed biweekly. As the concentration factor for the dialysis bags is unknown, they can be used to indicate qualitatively the presence of PCBs, but not ambient concentrations.

The 1989 monitoring program unfortunately missed the spring runoff period. All raw water samples were reported to be below the detection limit of 0.1 µg/*l*. This result is consistent with USGS monitoring data for water year 1989 at Fort Edward, which had a lower detection limit and showed detectable concentrations in the 0.01-0.1 µg/*l* range. During the same period the dialysis bag concentrators occasionally detected PCBs at all stations except the uppermost (Sherman Island). All the detects were identified as Aroclor 1242.

B.3.4 Fish and Other Aquatic Biota

Substantial declines in average PCB burden in Upper Hudson fish were observed in the years after 1978 (Sloan *et al.*, 1983, 1984; M. P. Brown *et al.*, 1985). Analysis of the data reveals that these declines in concentration have proceeded at a slower rate in more recent years. It is unclear to what degree

the abnormally low spring floods of the 1980s have affected PCB levels and may be responsible for the observed declines. It does appear that total PCB concentrations in fish on a lipid (fat)-basis can be closely predicted from summer average water column PCB concentrations.

Because PCBs are typically stored (bioaccumulate) in fatty (lipid) tissues, it is sometimes useful to normalize the PCB levels in fish and express them on a lipid basis, *i.e.*, PCB content in fish expressed as ug-PCB/g-fish lipid. Whether or not normalized to lipid content, levels of the higher chlorinated congeners in largemouth bass were approximately stable from 1981-1988. Reported Aroclor 1016 levels, representing less chlorinated congeners, appear to have continued to decline for all species during this time period. Given the slow rate of reduction of Aroclor 1254, it is unclear when or if natural processes will reduce the PCB burden in fish to acceptable levels. Furthermore, potential changes in PCB levels in the water column and sediment caused by possible scour and resuspension of sediments would likely cause at least temporary increases in PCB levels in fish and aquatic biota.

In addition to fish data, some monitoring of invertebrates is also available for the Hudson. From 1976-1985 multiplate samples and chironomid larvae have been monitored by NYSDOH. These data are discussed at the end of this section, following the discussion of the fish monitoring program.

B.3.4.1 Fish Sampling

Data on concentrations of PCBs in Upper Hudson River fish collected by NYSDEC between 1975 and 1988 were used in this study. While over 30 species of fish are represented in the data, the majority (75 percent) of the samples are from half a dozen species including striped bass, largemouth bass, brown bullhead, pumpkinseed, American shad, and American eel. Approximately two-thirds of the samples tested were standard fillet samples.

Samples Prior to 1975

While PCBs are known to have been discharged into the Upper Hudson River since the 1940s, no testing for PCBs in fish is known to have been undertaken before 1970. In that year, a nationwide survey of chemical pollutants in game fish conducted by a popular magazine included a sample of spawning striped bass from the Hudson estuary in which 4.5 to 5 ppm PCBs in flesh and 11 to 12 ppm PCBs in eggs were reported (Boyle, 1970). NYSDEC had been analyzing fish for DDT and other pesticides statewide since the early 1960's. In 1971, NYSDEC added PCBs to their analyses, although no results were released publicly until 1975 (Sanders, 1989).

Fish data collected and analyzed for PCBs in the 1970-74 period are summarized by Spagnoli and Skinner (1977). These 1970-74 Hudson River samples include one smallmouth bass collected at Warrensburg (Upper Hudson) and 146 fish from 11 species collected below the Federal Dam (Lower Hudson). The highest observed concentration from below the Federal Dam was in a largemouth bass, reported as containing 53.81 ppm wet weight total PCBs in the 1970-72 period. This sample was taken prior to the removal of the Fort Edward Dam.

In August 1974, a USEPA team obtained water, sediment and fish samples from upstream and downstream of the GE discharge at Fort Edward. A sample of 42 shiner minnows from below the GE discharge showed an average concentration of 78 µg/g (ppm) PCB as Aroclor 1242, while one rock bass was reported with 342 µg/g (Nadeau and Davis, 1974). It should be noted that samples collected at control Station 0 above Bakers Falls were not non-detects. PCB levels at Station 0 were reported as 7.0 µg/gm in shiner minnows and as 17.0 µg/gm in yellow perch. The latter level is higher than the average for all NYSDEC yellow perch samples in the Upper Hudson. Few other samples have been reported from Bakers Falls.

Samples 1975-1976

Following the USEPA investigation, NYSDEC undertook more detailed monitoring of PCBs in fish from both the Upper and Lower Hudson. A total of 440 Hudson River (Upper and Lower) fish samples were analyzed in 1975-1976, the results of which NYSDEC provided for incorporation into the TAMS/*Gradient* database.

The 1975-1976 fish collections were made by regional USFWS personnel who were instructed on specific species and sizes of fish desired, location of stations and time tables for collection. Target species for the Hudson included smallmouth bass (*Micropterus dolomieni*), largemouth bass (*Micropterus salmoides*), brown bullhead (*Ictalurus nebulosus*), goldfish (*Carassius auratus*), white sucker (*Catastomus commersoni*), striped bass (*Morone saxatilis*) and various other estuarine species. Other species were occasionally obtained as available. Attempts were made to sample small, medium (minimum legal) and large representatives of each species.

Analyses were conducted by several different state laboratories, apparently using the methodology of Bush and Lo (1973) and reported against standards for Aroclor 1242 *or* 1016 and Aroclor 1254. Aroclor 1221 was not analyzed. The nominal detection limit of the method was 0.01 ppm, although some of the labs reported results only as low as 0.1 ppm.

Samples 1977-1988

In 1977-1979 NYSDEC monitoring methods were refined and standardized. Collection has continued to the present, although no results after 1988 are available. As of 1988, NYSDEC changed the frequency of sampling from yearly to every other year. Samples from 1990 were collected, but PCB analyses have not been completed. NYSDEC provided data covering the period of 1977 through 1988, which contains 7,373 Upper and Lower Hudson fish or fish composite analyses. These samples have been collected on a regular basis, with the intent of sampling given species at predetermined locations within two weeks of a specified date in

order to minimize potential seasonal effect. Records are very limited for 1981 and 1987. Table B.3-14 provides a summary of the total number of fish sampled for the Upper Hudson from 1975-1988.

Information in the fish database, in addition to chemical analyses, usually includes species, method of preparation, weight, length, percent lipid and, if determined, sex and age. In addition to a descriptive sampling location, river mile numbers and geographic designations are associated with each sample.

Sample collection, preparation and analytical methods are described in Armstrong and Sloan (1981). The desired sample size for each species collected was 30 individuals, although the availability of fish did limit sample size in some cases. A single fish sample generally consisted of a composite of one to three fish. For fish longer than 150 mm, standard fillets of whole sides of scaled fish were prepared for analysis. For brown bullhead samples, the skin was removed from the fillet. Samples of fish shorter than 150 mm were analyzed whole with head and viscera removed.

Aroclor concentrations in fish were determined by comparison to commercial Aroclor standards. This method was not able to distinguish between Aroclors 1242 and 1016. The detection limit for each Aroclor tested was 0.1 ppm. For each sample the percent lipid was determined as the percent by weight of tissue soluble in petroleum ether.

PCB Levels in Fish

This section summarizes the Upper Hudson PCB monitoring results for fish. Additional statistical analyses are presented in Section B.4. Samples collected at River Mile 153, just below the Federal Dam, are included with those of the Upper Hudson, since they represent a resident, freshwater, rather than estuarine, population, which is exposed to PCBs transported over the dam.

Although over 30 fish species have been sampled in the Hudson River, a few of these species account for the majority of the samples collected. These species are pumpkinseed, largemouth bass, brown bullhead, goldfish (carp), white perch and yellow perch. Overall average Aroclor levels for these species in River Miles 153 to 195 of the Upper Hudson for 1975-1988 are provided in Table B.3-15. Aroclors 1016 and 1254 are the dominant PCB mixtures reported; Aroclor 1221 represents the smallest PCB fraction. Overall, the highest average PCB concentrations have been found in goldfish (carp) with 1975-1988 average Aroclor levels of 32.0 ppm (Aroclor 1254) and 91.6 ppm (Aroclor 1016). That goldfish have the highest average PCB levels is not altogether surprising as they also have the highest lipid content (9.3 percent). The average 1975-1988 Aroclor levels in largemouth bass, perch and bullhead are <30 ppm, although total PCB levels in individual largemouth bass have been reported as high as 370 ppm.

The average of total PCB levels in all fish for recent years (1986-1988) also shown in Table B.3-15, is 10.9 ppm. The upper 95 percent confidence found on this mean, used in the preliminary human health risk assessment (B.6), is 12.0 ppm.

Tables B.3-16 through B.3-18 provide summary statistics by river mile sampled for largemouth bass, pumpkinseed, and brown bullhead, respectively. Recent data for 1986-88 show that median PCB levels in fish range from 1 to 30 ppm depending on species and river mile. Additionally, the variability of PCBs in fish from a given location and year, as measured by the standard error of the estimate of the mean, has dropped as the PCB levels have declined.

Figure B.3-13, which plots mean, total PCB levels in brown bullhead (skinned standard fillets) as a function of year and river mile, provides an overall perspective of the general decline of PCB levels over time and by river mile. PCB levels in the vicinity of River Mile 190 are the highest, but samples from only a few years are available here.

Surprisingly little fish data are available from the Thompson Island Pool area (River Mile 188-94). Regular monitoring there began in the latter half of the 1980s. The most complete data set available for the Upper Hudson above the Federal Dam is that at River Mile 175. Data here begin in 1975 and continue, with some interruptions, through 1988. Figure B.3-14 summarizes the trends in mean Aroclor levels for four fish species at this location as ppm wet weight Aroclor 1254 and Aroclor 1016. Aroclor 1016 levels are shown on a log scale, as they were extremely high for certain species in the 1977-78 time period. Aroclor 1221 was quantitated also, but generally at very low levels compared to the other two Aroclors. A notable trend is that during the 1977-78 period the ratio of Aroclor 1016/1254 levels in fish are elevated, *i.e.*, the lower chlorinated congeners (1016) dominated the higher chlorinated congeners (1254) during the late 1970s. This elevation may be attributable to scour of dechlorinated sediments from depth. Another possible explanation for the shift from Aroclor 1016 dominance to Aroclor 1254 in recent years is that the lower chlorinated congeners were more rapidly released and dissipated in the late 1970s and early 1980s, whereas the higher chlorinated congeners have tended to bioaccumulate more and are less rapidly released by fish.

Lipid-Based PCB Concentrations

Differences between PCB levels in different fish species are probably due to both differential feeding patterns and lipid content. Fish lipid (fat) typically accumulates PCBs more than other, less fatty tissues, because PCBs are highly lipophilic compounds, i.e., PCBs are more soluble in fat than water. Normalizing PCB levels on a fish lipid weight basis has sometimes proved useful in comparing measurements among species. The average percent lipid of goldfish was 9.2, pumpkinseed was 3.3, brown bullhead was 2.9 and largemouth bass was 1.4, with skewed distributions. There appears to be little correlation between weight and PCB body burdens in most species. Age of the fish sampled was not often determined.

Besides accounting for species differences, accounting for lipid content is important because fish lipid content appears to have changed from samples caught from one year to the next. Dividing the PCB concentration (wet weight) by the measured lipid content (g-lipid/g-fish) of the sample, one obtains the PCB concentration per gram (mass) of fish-lipid (ug-PCB/g-lipid). Looking at all samples for all species at all Upper Hudson sampling locations on a lipid basis, the median PCB levels have declined from 1829 ug-PCB/g-lipid in 1977 to 271 ug-PCB/g-lipid in 1988, with a 95 percent upper confidence bound estimate on the mean of 484 (Table B.3-19).

Lipid-based means of Aroclor 1254 and 1016 concentrations by year in the predominant fish species sampled at River Mile 175 are shown in Figure B.3-15. On a lipid basis, largemouth bass have usually shown the highest Aroclor 1254 levels. This may reflect their position as top carnivores in the aquatic food chain or their low fat content. Further, the lipid-based Aroclor 1254 levels in most species appear to have been relatively constant since 1982. Error bars are omitted from the multiple species plots for legibility. Trends in largemouth bass, with error bars, are shown in Figure B.3-16. (An error bar shown as a vertical line at each year representing the 95 percent upper and lower confidence bounds on the mean.)

In addition to the monitoring at River Mile 175, there is a good continuous record of sampling of brown bullhead at River Mile 153 just below the Federal Dam, for the period 1977-1988. PCB levels here were on average much lower than those observed at River Mile 175, presumably due to dilution of PCB concentrations by the flow of the Mohawk River. Average lipid-based PCB concentrations in brown bullhead show a regular exponential decline for Aroclor 1016 components and a less dramatic decline for Aroclor 1254 (Figure B.3-17).

B.3.4.2 Other Chemicals in Fish

The NYSDEC database contained analyses for chemicals in fish other than PCBs. Summary statistics for these other chemicals are shown in Table B.3-20.

Metals such as mercury, cadmium, chromium, copper and zinc were frequently detected. Median mercury and zinc concentrations were on the order of 0.4 ppm and 10 ppm, respectively. Organic compounds frequently detected include pesticides such as DDT, heptachlor, and dieldrin. Hexachlorobenzene and hexachlorocyclohexane were also detected frequently.

B.3.4.3 NYSDOH Macroinvertebrate Studies

As part of the Hudson River PCB Reclamation Demonstration Project, the New York State Department of Health (NYSDOH) conducted biomonitoring studies from 1976 to 1985 using caddisfly larvae, multiplate samples and chironomid larvae (Simpson *et al.*, 1986). These studies included long-term biomonitoring efforts from 1976 to 1985 as well as two short-term biological uptake studies in July and September of 1985.

Long-Term Biomonitoring Study

From 1976 through 1985, artificial substrate samplers (multiplates) were placed at 17 sites along the Hudson River from Hudson Falls to Nyack, New York (Novak *et al.*, 1988). These samplers were collected each year after a period of five weeks during the months of July, August and September. PCBs in the samples were reported as Aroclors 1016 and 1254. The resulting PCB concentrations in the multiplate samples represented a composite of sediment, algae, plankton and various macroinvertebrates. Invertebrates collected in the multiplate samplers included the following taxonomic groups: Chironomidae, Oligochataeta, Trichoptera, Ephemeroptera, Amphipoda and Elimidae. Chironomid larvae and pupae were the most abundant invertebrate component from Fort Edward to Saugerties, comprising up to 86 percent of the total macroinvertebrate population at Fort Miller and Waterford.

From 1978 to 1985, caddisfly larvae were collected by hand-picking individuals from rocks at five designated sites: Hudson Falls, Fort Edward, Fort Miller, Stillwater and Waterford. Caddisfly collections were made in June, July, August and September of each year.

Measured PCB levels in the 1985 multiplate samples for September ranged from 0.25 ppm at Hudson Falls (the control site) to over 6 ppm at Fort Edward. Multiplate monitoring from Fort Miller to Waterford resulted in PCB levels of 4 to 5 ppm. The multiplate samples at any one site appeared to show a consistent decline in PCB concentrations from early summer to later summer in any particular year. Larger scale trends or relationships in either time or with sample location are difficult to detect, because of the extremely wide variation in the sample results. Average PCB concentrations in multiplate samples generally showed a decline from 1976 - 1980. Nevertheless, average PCB concentrations increased in 1981 and remained high through 1985. Multiplate samples from the Thompson Island Pool and downstream showed significantly higher PCB concentrations than samples taken upstream of Fort Edward. Yet, no significant trends are apparent when comparing Fort Edward results with those at Waterford.

The results of the caddisfly biomonitoring efforts show a decline in total PCB concentrations from 1978 to 1980. As is true of the multiplate data, spatial trends are not readily apparent from the caddisfly samples. Measured PCB levels in macroinvertebrate tissues generally ranged from 20 to 60 ppm (dry weight) in 1979 and from 20 to approximately 40 ppm (dry weight) in 1985. During the same collection periods, macroinvertebrates collected at Hudson Falls, the control site, exhibited PCB tissue residues less than 10 ppm.

The data from this study exhibit a great deal of random variability. The bulk of the data are from multiplates, which collect sediment as well as living matter. In theory, it should be possible to distinguish sediment versus biologically-based PCBs by adjusting the observations to a lipid basis; lipid content of samples was reported. An additional factor contributing to the variability is that in almost every year a downward trend by month was observed at most stations, based generally on three samples. The cause of this phenomenon is not known, but the limited number of observations in this yearly cycle may have obscured the influence of other factors.

Examination of the complete run of samples at the Fort Miller multiplate and caddisfly station (PCB-5) shows what appears to be a largely random pattern of total PCBs on a dry weight basis (Figure B.3-18). The larger mean PCB concentration for the period 1981-82 is largely attributed to the increased variability of the data for this period. Adjusting to a lipid basis actually increases the total variability represented by outlying data points. Only a slight trend is suggested in this figure, where PCB levels may have declined from 1976 to 1980, then remained approximately stable. At Stillwater, the lipid-based values appear to be almost entirely random (Figure B.3-19).

A comparison between the confidence intervals on the overall means for all sample years by sample locations (Figure B.3-20) shows differences between the low values observed above Fort Edward and values at downstream stations. Lipid-based means at all stations by year (Figure B.3-21) appear to show a decline from 1976 to 1979, then relatively constant levels except for a jump upward in 1982. Again, this apparent trend is difficult to confirm as a consequence of the limited number of samples and possible differences in sampling or analytical protocols throughout the duration of the monitoring.

Short-Term Biomonitoring Study

Short-term biomonitoring investigations using the chironomid larvae, *Chironomus tentans*, were also performed by the NYSDOH during July and September 1985 (Novak *et al.*, 1990). The monitoring method consisted of placing 25 laboratory-reared chironomid larvae in nylon mesh envelopes or packets that were exposed to the water column. Envelopes were placed, in groups of ten, in steel mesh baskets at the primary collection site and monitored at 0, 1, 2, 4, 8, 12, 24, 48, 72 and 96 hours. Chironomids were placed at four sites, including two at Thompson Island Pool, one at Bakers Falls and one at Fish Creek, and monitored at 96 hours. Packets of chironomids exposed to the sediment at a collection site located on the eastern shore of the Thompson Island Pool were also collected at 96 hours. Water column samples were obtained during the same collection intervals for each site.

Results of this investigation indicated that chironomids accumulated PCBs ranging from 0.1 to 7 ppm after 1 to 96 hours of continuous exposure, whereas larvae exposed to sediments near the Thompson Island Pool for 96 hours contained over 100 ppm. Water column PCB levels were in the range of 0.03 - 0.1 µg/l during the experiment. The ratio of the PCB levels in the chironomids (in ppb) to the ambient PCB concentrations in the water column (defined as the bioaccumulation factor or BAF) were on the order of 10^4 to 10^5.

A significant conclusion from this study was that the PCB congener pattern found in tissues of chironomid larvae differed substantially from the congener pattern observed in water. Using capillary column gas chromatography, the investigators were able to isolate PCB congeners in both the water column and chironomids. The most abundant congeners in chironomid tissues were 2,4,2',5'-tetrachlorobiphenyl and 2,3,6,4'-tetrachlorobiphenyl. In contrast, the predominant congeners in water were 2,2'-dichlorobiphenyl, 2,6-dichlorobiphenyl, and 2,6,2'-trichlorobiphenyl. These findings suggest a number of possible explanations. One explanation could be that chironomid larvae may selectively bioconcentrate the more highly chlorinated congeners which are present at relatively low concentrations in water. Another factor which could explain the observed results is that the lower chlorinated congeners were present in the water but below detection limits.

Although this study presents some very interesting congener-specific results, they are too limited in scope to provide clear indications of either congener-specific bioaccumulation or congener-specific comparisons of PCBs in sediments. (No congener-specific sediment data were obtained.)

B.3.5 PCB Concentrations in Air and Plants

B.3.5.1 Air

Monitoring Near Fort Edward

Air monitoring efforts for PCBs and other air toxics have been conducted in the Upper Hudson River study area from late 1976 to 1982 by NYSDEC/NYSDOH and various researchers and as recently as 1989/90 by contractors (Harza/Yates-Auberle) for General Electric.

From January through August of 1977, NYSDOH collected air samples at five locations in the Upper Hudson Valley to determine ambient PCB concentrations. While the Glens Falls and Warrensburg samples showed no detections above the 0.020 $\mu g/m^3$ detection limit, results from the Hudson Falls and the Fort Edward samples demonstrated high levels of total PCBs, ranging from 0.060 $\mu g/m^3$ to 3.26 $\mu g/m^3$ (Malcolm-Pirnie, 1978). Atmospheric levels of PCBs in the Fort Edward area were reported to decrease from 1 $\mu g/m^3$ down to 0.3 $\mu g/m^3$ after the cessation of PCB use by General Electric in their Hudson Falls and Fort Edward capacitor plants in 1977 (Limburg, 1984).

In 1979, NYSDEC conducted an air monitoring survey for PCBs around various dumps and landfills (Caputo and Fort Miller dumps, Remnant Area, Moreau and Site 3a and Buoy 212) in the Hudson Falls/Fort Edward area bordering the Upper Hudson River (see Table B.3-21). Values ranged from 5 to 15 $\mu g/m^3$ total PCBs at the Moreau and remnant areas and 24 to 300 $\mu g/m^3$ total PCBs at the Fort Miller and Caputo dumps, respectively. At the Caputo dump, where the soil was reported to contain 5,000 mg/kg PCBs, air monitoring for PCBs before and after capping of the site showed that average ambient PCB concentrations decreased from 118 $\mu g/m^3$ before capping to 0.26 $\mu g/m^3$, once the site was capped (Shen, 1982).

Two air samples taken over Lock 6 in the summer of 1980 yielded Aroclor 1242 concentrations ranging from 0.11 $\mu g/m^3$ to 0.52 $\mu g/m^3$ (Table B.3-21). Aroclors 1221 and 1254 were not detected (NYSDEC, 1981). Ambient PCB levels

monitored at farm fields in 1981 near the Hudson River, however, showed low PCB concentrations of approximately 0.005 $\mu g/m^3$ (NUS, 1984).

In the early 1980s, NYSDEC and the Boyce Thompson Institute for Plant Research of Cornell University conducted a joint air/plant monitoring effort near the tailwater of Lock 6 to determine if volatilization of PCBs from the water column was occurring (Buckley and Tofflemire, 1983 and 1984). Between August 1981 and September 1981, seven air samples were taken. Aroclor 1242 was detected in all seven samples, ranging from 0.031 to 0.06 $\mu g/m^3$. Aroclor 1254 was detected in three of the samples at levels up to 0.0013 $\mu g/m^3$. During this study, a vertical PCB gradient was also noticed, when airborne PCBs were measured simultaneously at heights of 1 and 4.5 meters above the water.

In August 1986, NYSDEC collected three sets of ambient grab samples in duplicate at the proposed containment site (Site G), the Fort Edward Landfill, the Bourgoyne Avenue School and Lock 7 of the Champlain Canal (USEPA/NYSDEC, 1987). The highest ambient PCB concentration measured was 0.083 $\mu g/m^3$ at Lock 7. Site G and the Fort Edward landfill samples contained PCB levels below the detection limit of 0.007 $\mu g/m^3$. The Bourgoyne Avenue School sample information was not available at the time of the report.

Again in 1987, NYSDEC conducted air monitoring from April 2, 1987 to July 16, 1987 at the Kingsbury Landfill, located north of Fort Edward. In 76 of 105 samples taken over April and May 1987, Aroclor 1016/1242 was detected with a maximum concentration of 0.49 $\mu g/m^3$. Aroclor 1248 was detected in 5 of 105 samples with a maximum concentration of 0.52 $\mu g/m^3$. Both the Kingsbury and Fort Edward municipal landfills were the burial site of several thousand tons of PCBs. Neither these landfills, nor others in the area, are part of the current investigation of the Hudson River Superfund Site. The results are presented here as evidence of PCBs having been monitored in the air within the vicinity of the site.

Most recently, in connection with its Remnant Remediation Project, GE conducted baseline pre-remediation air monitoring from August to November 1989 (Harza, 1990). Fixed air monitoring stations were planned at five locations along the river: two residential areas, one upwind location, one downwind location and a farming receptor location two miles south of the remnant area. Because of site access problems, only three of the five sites were monitored. Once every three days, two 1-liter/minute, 24-hour air samples were taken using a two-channel (two separate samples taken simultaneously) air monitoring station three feet above the ground. The samples were analyzed by NIOSH Method 5503 (desorption of fluorisil tubes with solvent followed by GC-ECD analysis) with a detection limit of 0.05 µg/sample. In total, 84 samples were collected. Seven samples showed levels of PCBs above the detection limit (0.05 µg/sample), with a maximum value of 0.23 $\mu g/m^3$. Of these seven detects, three were from a residential area (location A2), two from a downwind receptor (location A4) and one from the farm area (location A5). Although PCBs were detected in this investigation, the two sample channels often gave inconsistent results; one channel contained PCBs while the other did not. This occurrence may have been due to sampling or analytical problems or both.

Maximum ambient background PCB concentrations in air measured by New York State during a statewide monitoring effort, listed at the bottom of Table B.3-21, provide a perspective on PCB levels in air in the vicinity of the site. Maximum ambient air PCB concentrations measured in this effort ranged from 0.002 $\mu g/m^3$ in Syracuse and Rensselaer (urban areas) to 0.007 $\mu g/m^3$ in Staten Island (NYSDEC, 1982-4). These maxima are one to two orders of magnitude lower than the maxima detected during GE's baseline monitoring study and values measured by NYSDEC near Lock 7 in August 1986.

Lamont-Doherty Investigations

Three studies of PCB volatilization and properties governing volatilization were conducted at the Lamont-Doherty Geological Observatory. The first two of these studies (Bopp, 1979 and Bopp, 1983) dealt with the estimation of PCB properties that govern volatilization and the application of these properties to

actual conditions in the river. In particular, Bopp developed estimates for the molecular diffusivity, Henry's Law constant and gas exchange rate of PCB homologues at Troy and Poughkeepsie. Assuming a 20-day water transit time between the two points, a simple model of gas exchange based on the derived parameters worked quite well. The data and model both showed gas exchange rates for PCBs in the Hudson that varied inversely with the degree of chlorination. For low chlorinated homologues (di- and trichlorobiphenyls), Bopp (1983) estimates that about 40 percent is lost, because of gas exchange, as a given parcel of water travels from Troy to Poughkeepsie. For the tetra- and pentachlorobiphenyls, 10 to 20 percent is lost as a result of gas exchange. For the higher chlorinated homologues, Bopp (1983) calculated even lower loss rates and, in fact, was unable to measure any water column loss attributable to gas exchange. This finding has important ramifications for the fate of the more highly chlorinated homologues in the Lower Hudson. Since gas exchange and biodegradation did not appear to remove these PCBs, they would remain in the sediments of the Lower Hudson or be transported out to the New York Bight.

The remaining study from the Lamont Doherty Geological Observatory was conducted by Warren *et al.* (1985) and consisted of a series of laboratory studies to refine earlier estimates of Henry's Law constants for PCBs on a quasi-congener basis. Warren *et al.* (1985) determined Henry's Law constants for individual PCB congeners or pairs of similar congeners under a range of temperature conditions, which are directly applicable to conditions formed in the Hudson.

B.3.5.2 PCB Uptake by Plants

Early Studies

In 1977, PCBs were found in vegetation growing on dump sites in the Fort Edward area in a monitoring study performed by Weston Environmental Consultants (Weston, 1977). Following the discovery of high PCB concentrations in vegetation at the Fort Edward dump, NYSDEC and the Boyce Thompson Institute for Plant Research at Cornell University conducted a number of studies from 1978 to 1981 to measure airborne-PCB uptake in several vegetative species in the Fort

Edward/Hudson Falls vicinity. Their research evaluated PCB levels in trembling aspen along easterly transects from the Fort Edward dump, the Buoy 212 dredge spoil site, and a riffle area near Lock 6 (Tofflemire *et al.*, 1981). PCB measurements in aspen leaves ranged from 180 mg/kg at the dump, decreasing to 0.15 mg/kg at a distance of 820 meters from the dump. A similar declining trend was reported for the Buoy 212 dredge spoil site and the riffle area.

From 1978-1980, total background PCB concentrations were measured in goldenrod and trembling aspen within Washington and Saratoga Counties and were found to be decreasing with time (Buckley, 1983). Average PCB concentrations in goldenrod decreased from 0.32 mg/kg (ppm dry weight) in 1978 to 0.18 mg/kg in 1980, whereas average PCB levels in trembling aspen decreased from 0.12 mg/kg in 1978 to 0.07 mg/kg in 1980. Also in Washington and Saratoga counties, background levels of total PCBs were measured in crops such as hay, corn, timothy grass, perennial rye, brome grass, and orchard grass. Average total PCB background concentrations ranged from 0.02 mg/kg (corn/silage) to 0.12 mg/kg (brome grass/hay), as shown in Table B.3-22.

In September 1979, total PCB concentrations were measured in aspen, sumac and goldenrod at five sites located at various distances (<1,200 m) and directions from the Patterson Road PCB dump in Fort Miller, New York (Buckley, 1982). PCB levels in aspen ranged from 0.1 mg/kg to 58.2 mg/kg. PCB levels in sumac suggested similar PCB uptake, with concentrations ranging from 0.11 mg/kg to 68.6 mg/kg. Measurements for goldenrod ((0.26 mg/kg to 182 mg/kg) showed approximately twice the rate of PCB uptake at the same sites as the aspen and sumac measurements. This result suggests that PCB uptake by vegetation may be species-dependent.

During September 1980, vapor-phase PCB accumulation in vegetation was measured in the leaves of two varieties of sumac near an abandoned PCB dump in the Fort Edward/Hudson Falls area (Buckley and Tofflemire, 1983). The data, shown in Table B.3-22, demonstrate PCB concentrations higher than background levels with PCB concentrations ranging from 0.97 mg/kg to 5.2 mg/kg. The lowest PCB concentrations were found in samples furthest (230m) from the source, whereas

the highest PCB levels were in samples nearest (60m) the source.

In 1978 and 1981, air and vegetation measurements for total PCBs were made near the tailwater of Lock 6. This study was conducted to investigate whether PCB concentrations in air and vegetation were elevated above background concentrations measured in Saratoga and Washington counties, because of PCB volatilization from turbulent sediment resuspension at the tailwater of the dam. The data indicate a general decrease in foliar PCB concentrations from 1978 to 1981, although the small sample size precludes a conclusive result. The PCB concentrations in vegetation near the tailwater of the Lock 6 dam, which ranged from 0.23 to 1.07 mg/kg total PCBs, are considerably higher than background levels in the same species measured in 1979.

More Recent Studies

In July and August 1984, researchers from NYSDOH performed translocation and transplantation studies on purple loosestrife at two locations to determine PCB uptake from soil and air (Bush *et al.*, 1986). The study site was located alongside the Upper Hudson River, near Albany. The control site was two miles from the river. Their results indicated that the main route of PCB uptake in purple loosestrife is via the root system. Plants transplanted from the control site to the Hudson River site showed an increase in PCB levels from 0.010 mg/kg to 0.210 mg/kg total PCBs. Additional evidence that soil, as opposed to air, served as the major pathway of PCB uptake was provided by congener analyses. The congeners in the plants that showed an appreciable increase over time were those present in the soil samples, but not in air samples.

Although the primary uptake of PCBs in purple loosestrife appeared to be systemic through the roots, PCB uptake at the air-leaf interface also occurred. This pathway is suggested by the increased PCB concentrations measured in purple loosestrife plants that were translocated in plastic bags from the control site with control site soil to the experimental site at the river. Further field and laboratory investigations revealed that in areas with high PCB levels in air (0.140 $\mu g/m^3$), plants scavenged PCB congeners from the air. At lower ambient PCB

levels (0.008 μg/m^3), monochloro- and dichlorobiphenyl were emitted from the plant. This finding suggests that at least two processes are possible; PCB accumulation from the air into the lipophilic, waxy sections of the leaf and volatilization of the less chlorinated PCBs from the leaf to the air.

A similar research project involving PCB uptake in crops and their fruits was undertaken during July 1985 by the NYSDOH (Shane and Bush, 1989). The study site was Patroon Island, situated in the Upper Hudson River, near Albany. Soil analysis for PCBs at the study site revealed an average PCB concentration in soil of 0.145 mg/kg. The control site was located in a rural area near Guiderland, with an average total PCB soil concentration of 0.009 mg/kg.

Corn and soybeans were planted at both sites, but corn did not take root at the study site. Stringbeans and pinto beans, already growing at the study site, also were analyzed during the study. Corn plants were both transplanted and translocated from the control site to the experimental site. Composite leaf samples were taken from the end of July through mid - September for PCB analysis. At the end of the experiment, plant fruits, *e.g.* beans and corn ears, were also collected.

The average PCB concentrations in the 17 sets of samples taken from the study site ranged from 0.001-0.050 mg/kg total PCBs, but were generally less than 0.025 mg/kg. PCB levels in fruits were 0.00013 mg/kg for corn kernels and 0.00055 mg/kg for soybeans, approximately 100 times lower than the leaf concentrations. The results of the study indicated that all four plant species preferentially accumulated eight mono-, di-, tri- and tetrachlorobiphenyl congeners, but not in the same order. Mono- and dichlorobiphenyls (2-mono-, 2,4'-di- and 2,2'-dichlorobiphenyl) were present as the highest percentages in all species; the remaining five congener percentages varied by species.

None of the bean species yielded a significant increase in accumulation over time. In addition, for both the translocated and transplanted corn plants, a significant decrease over time in the content of the major congeners (mono- and dichlorobiphenyl) was recorded. Shane and Bush (1989) note that although leaves

of corn contain lipophilic waxy cutin, which would be likely to accumulate PCBs, a greater elimination route for the less chlorinated, more water-soluble, congeners exists by way of transpiration through the stomates of the corn leaf. Thus, Shane and Bush concluded that although PCB accumulates in corn seedlings, emissions or metabolism reduce the PCB load to the same levels as uncontaminated plants in approximately three weeks of growth after emergence.

Very low PCB concentrations in vegetation were found in the studies by Shane and Bush (1986) and Bush *et al.* (1989). Their results and the data from the earlier studies by NYSDEC and Boyce Thompson Institute demonstrate that crop plants potentially accumulate PCBs from either soil, air or both. The older studies show more dramatic uptake in response to the relatively high historical PCB levels in air near concentrated dump sources.

B.3.6 Other Media

Few monitoring data for PCBs are available for other media of possible concern. Limited groundwater sampling has been performed in the late 1970s, but no more recent data were located. Surface soils near the river and adjacent croplands have not been monitored. Virtually no data exists for upland biota.

As discussed in the 1984 FS (NUS, 1984), Weston Environmental Consultants (1978) collected groundwater samples in 1977 from several dredge spoil and PCB dump areas. Groundwater samples from the spoil and dump areas reportedly contained PCBs at levels ranging from 16.7 µg/*l* (ppb) near Site 212 to as high as 693 µg/*l* in groundwater at the Old Fort Edward landfill. Weston calculated PCB migration potential in groundwater from dredge spoil sites to be on the order of 10^{-4} to 10^{-1} lb/year. These estimates were several orders of magnitude lower than potential surface runoff/erosion losses of PCBs from dredge spoil areas, which were estimated to be as much as 24.2 lb/yr (Weston Environmental Consultants, 1978, as reported in NUS, 1984).

B.3.7 Adequacy of PCB and Aroclor Measurement

B.3.7.1 Overview

In general, the sediment, fish and water column PCB monitoring data sets reviewed here are acceptable for assessing trends in PCB concentrations. Nevertheless, it is difficult to draw comparisons between data sets, because different studies selected different approaches for quantitating (measuring) PCBs and reporting them as various Aroclor mixtures. Most of the reported Aroclor concentrations are based on the concentrations of only a portion of the congeners present in the sample mixture. Furthermore, the actual distribution of congeners in an environmental sample, collected years or decades after the original release, bear little resemblance to a commercial Aroclor mixture. For most, if not all, the reported Aroclor results, *e.g.* Aroclor 1016, 1242, 1254, *etc.*, the data should be interpreted as representative of lower versus higher chlorine containing PCB congeners, rather than as indicative of a true commercial Aroclor-like mixture.

PCBs - Aroclors, Congeners, and Analysis Methods

PCBs are a class of chemicals theoretically consisting of 209 different congeners, of which approximately half are found in various Aroclor mixtures. A single Aroclor mixture may consist of dozens of congeners. The number of different congeners reportedly found in an Aroclor mixture may vary considerably, depending upon the type of analysis performed and the quantity analyzed (Alford-Stevens, 1986). For example, in a review article by Alford-Stevens, the number of congeners reportedly found in Aroclor 1242 ranged from 27 to 74. In a more recent study, Schulz *et al.* (1989) report a total of 77 different congeners detected in Aroclor 1242 along with weight percent information.

The weight percent of each congener or group of congeners reported in the literature for a given Aroclor mixture has also varied. The weight percent information for homologue groups found in different Aroclors cited by USEPA (1983) differs from the weight percent information reported by Webb and McCall

(1973). The extent to which these differences can be attributed to analytical technique or methodology and batch to batch variation in Aroclor mixtures is unknown. O'Brien and Gere (1977) report that the results of an inter-laboratory analysis of reference samples suggest that "ostensibly pure Aroclor mixtures from different sources contained different percentages of PCB homologs." The Aroclor standards were obtained commercially and directly from Monsanto.

Early determinations of Aroclor mixture compositions involved separation of the mixture on a packed column gas chromatograph (GC) followed by quantitation using an electron capture detector (ECD). A single Aroclor mixture can be resolved on a packed column GC into two to eighteen separate peaks depending upon the Aroclor mixture analyzed. Due to the poor resolution of PCBs on a packed column, each packed column GC peak may represent as many as eight or more individual congeners. The majority of the PCB analyses conducted for the Hudson River investigations have been packed column gas chromatograph (GC) methods. Only recently have more precise, capillary column GC techniques capable of resolving individual congeners been employed. These techniques still generally use the ECD for quantitation.

The sensitivity of the ECD to an individual congener varies directly with the degree of chlorination. In fact, the ECD response may vary by as much as a factor of 100 over the entire range of PCB congeners. Thus, the peak area generated by a mono- or dichlorobyphenyl may be 100 times less than that generated by a nona- or decachlorobiphyenyl, if all are present at the same concentration in the sample. This finding has important ramifications when analysis considers total peak area and not individual peaks to quantify PCB levels or Aroclor mixtures. Minor changes in the highly chlorinated PCB concentrations will be more readily reflected in the total peak area than more substantial changes in the lower chlorinated PCB concentrations.

In 1973, Webb and McCall formalized an approach for quantifying Aroclor mixture concentrations using GC-ECD and the weight percent of PCBs represented by each packed column GC peak. Using elemental analysis, GC-ECD and GC-mass spectrometry (GC-MS), Webb and McCall were able to calculate the weight percent

of PCBs contained in commercial solutions of Aroclors 1221 through 1260. The weight percent of packed column peaks corresponding to Aroclors 1242, 1254 and 1260 are shown in Figure B.3-22. Individual peaks are identified by their retention time relative to the internal standard pp'-DDE which is assigned a value of 100. With this information, individual response factors can be calculated for each packed column GC peak in a standard Aroclor mixture. The total amount of PCBs present in any sample can then be calculated from the sum of the amounts of all the individual peaks.

Webb and McCall peaks corresponding to Aroclor mixtures quantitated for Hudson River samples are summarized in the tabulation that follows.

Webb and McCall Peaks Used to Quantitate Aroclor Mixtures

Source	Aroclor 1221	Aroclor 1016	Aroclor 1242	Aroclor 1254	Aroclor 1260
1977 Sediment Survey	11 or 21	28, 47, 58		8 peaks ID unknown	
1984 Sediment Survey			28, 47, 58	See text	See text
Fish Monitoring	11	37, 40		125, 146, 174	
1983 USEPA (in NUS FS, 1984)			See text	See text	See text
1985 Bopp *et al.*	Analyzed for Total PCBs as Sum of Peaks 28 - 174				
3985 Bopp *et al.*			47		

Interpreting Reported Aroclor Results

The Webb and McCall approach works well for estimating concentrations of total PCBs in environmental samples including weathered samples where gas chromatograms bear little resemblance to standard Aroclor mixtures. A problem arises, however, when total PCBs are calculated in a weathered sample using the Webb and McCall approach, but then reported as the concentration of a single Aroclor mixture or the combination of two or more Aroclor mixtures.

Weathered samples may be defined as samples containing PCB mixtures no longer resembling pure Aroclor standards. Weathering of Aroclors occurs naturally in the environment, since each PCB congener possesses different physical and chemical properties. For example, Henry's Law constants may vary by a factor of ten (Brunner, 1990). Association constants may vary by over two orders of magnitude (Lara, 1989). Solubility may vary over several orders of magnitude (Dickhut, 1986) depending upon the PCB congener. Overall, the lesser chlorinated PCBs are generally more soluble and subject to losses through volatilization and photolysis. The more chlorinated PCBs are generally less soluble and more likely to accumulate in sediments and biota. Even though natural transformation processes may alter PCB compositions so they no longer resemble commercial Aroclor mixtures, PCB concentrations in environmental samples continue to be reported as Aroclor mixtures. For weathered samples, Aroclor designations are no longer descriptive of the congeners present and their relative amounts.

The Webb and McCall approach for analyzing environmental samples containing more than one Aroclor fosters the tendency to report PCB concentrations in all samples, even weathered samples, as Aroclor concentrations. Peaks with relative retention times 11 to 70 are compared to peaks in Aroclor 1242; peaks with relative retention times 84 to 174 are associated with Aroclor 1254; and peaks with relative retention times longer than 174 are associated with Aroclor 1260. Some exceptions to this division of peak assignments are presented by Webb and McCall along with a flow chart for assigning Aroclor standards to be used for quantitation. The scheme presented for assigning Aroclor standards to quantitate

peaks found in specified regions of the chromatogram may lead to an incorrect assumption of congeners present, if the concentrations are reported as Aroclor mixture concentrations or if GC operating conditions are altered, such as using a different stationary phase in the column. Webb and McCall used a SE-30 stationary phase and noted that DC-200, OV-17, and OV-101 phases are also appropriate. QF-1 and OV-225 stationary phases are not appropriate for this technique, since the PCBs will elute in a different order and the peak assignments made by Webb and McCall will no longer be valid.

B.3.7.2 Discussion of Data Quality Assurance

1976-1978 Sediment Survey

O'Brien and Gere conducted the PCB analyses for the sediment samples from the 1976-1978 survey. The O'Brien and Gere (1977) report contains a summary of the quality control (QC) data produced during analyses of sediment samples collected during the 1977 sediment survey. Method blanks (sodium sulfate and sand) were incorporated with every batch of samples analyzed. Two different extraction procedures were employed in the study -- soxhlet extraction and a two hour shaker method. Extraction efficiencies were evaluated by performing a second extraction on the same sample and assuming that all recoverable PCBs were extracted after two extractions. This approach does not account for PCBs which may be irreversibly bound to the matrix. Duplicate analyses where samples are split into two portions and analyzed as two samples in the same batch were employed to evaluate the reproducibility of results. In addition, an inter-laboratory study of reference samples was conducted between five different laboratories to assess the variability between laboratories.

PCB contamination (> 1 ppm) was observed in 5 out of 37 soxhlet extraction blanks and 0 out of 120 shaker extraction blanks. Sediment samples analyzed in the same batch as unacceptable blank samples were re-analyzed. Extraction efficiencies averaged over 80 percent for both methods. Sediment analyses were not corrected for extraction efficiency, since extraction efficiencies varied as much as duplicate analyses on the same sample.

The relative percent difference between duplicate analyses of sediment samples extracted using the soxhlet extraction technique averaged 125 percent for Aroclor 1016, 160 percent for Aroclor 1254 and 174 percent for Aroclor 1221. These differences are quite large compared to the relative percent differences observed for the shaker method -- 105 percent for Aroclor 1016, 85 percent for Aroclor 1221 and 71 percent for Aroclor 1254. The target, relative percent difference for the study was 100 percent. After rejecting outliers, the relative percent difference between duplicates improved considerably so that only Aroclor 1221 measured with the soxhlet technique remained outside the target level with a value of 111 percent (Tofflemire *et al.*, 1979). The large relative percent differences averaged over all sediment samples may have been due to the extreme heterogeneities of the samples. Bopp *et al.* (1985) noted that one sediment sample collected from the Upper Hudson River contained centimeter-sized pieces of paper containing about 6,000 ppm PCBs.

The overall relative standard deviation (RSD) of PCB concentrations measured in the inter-laboratory study was 49 percent. During the inter-laboratory study, it was determined that pure Aroclor mixtures from different sources contained different amounts of PCB congeners. Since these pure Aroclor mixtures were used as standards for quantitation, some of the variation between laboratories may be attributed to the use of Aroclors from different sources. Unfortunately, there is no indication of the magnitude of this effect.

In the 1977 sediment survey, the O'Brien and Gere laboratory reported PCBs as Aroclor concentrations. Aroclors 1221, 1016 and 1254 were used as standards. GC peaks were identified by their actual retention times and not retention times relative to pp'-DDE. As a result, peak identities according to the Webb and McCall scheme are not certain. O'Brien and Gere note that Aroclor chromatogram patterns may be altered drastically in environmental samples, making the identification of specific Aroclors difficult, if not impossible. Nevertheless, the laboratory reported PCB levels as Aroclor concentrations.

Aroclor 1221 was not always identified in the samples analyzed by O'Brien and Gere. When Aroclor 1221 was identified, quantitation was based on a single peak (most likely Webb and McCall peak 11 and possibly peak 21). Peak 11 accounts for 32 percent of the PCBs found in Aroclor 1221 and only one percent of PCBs found in Aroclor 1242. As Aroclor 1221 was rarely identified and peak 11 is difficult to quantitate, little importance should be attached to Aroclor 1221 levels reported in the 1977 sediment survey.

Aroclor 1016 levels were quantitated based on three different peaks (Webb and McCall peaks 28, 47, and 58). Aroclor 1254 levels were quantitated based on eight peaks (possibly Webb and McCall peaks 70, 84, 98, 104, 125, 146, 178, and 203). For each peak, a separate estimate was derived of the respective Aroclor mixture and the average for the Aroclor was reported for the sample. Because some of the peaks used to quantitate Aroclor 1016 and 1254 occur in other PCB mixtures, *e.g.*, peaks 47 and 58 occur in both Aroclor 1016 and 1254, an overestimation error results from the use of these peaks to quantitate any given Aroclor mixture. If equal amounts of Aroclors 1016 and 1254 were contained in a sediment sample, then the Aroclor 1016 concentration would be overestimated by almost 30 percent. The higher the relative amount of Aroclor 1254 in the sample, the greater the overestimation.

The same type of situation exists for quantitation of Aroclor 1254. If equal amounts of Aroclor 1242 and 1254 were contained in a sample, estimated Aroclor 1254 levels would be higher by almost 15 percent and total PCB levels (Aroclor 1242 plus Aroclor 1254) would be overestimated by approximately 22 percent. Once again, the degree of overestimation would depend upon the relative levels of Aroclors 1242 and 1254 in the sample.

The degree of overestimation would also likely depend upon the extent of weathering in the sample. Aroclor levels and total PCB levels, based on the sum of the individual Aroclors in lightly weathered samples, are most likely to be overestimated. The weathering process is not well enough known to predict the probable direction of error in estimating PCBs in heavily weathered samples.

In spite of the large errors introduced by the above approach, the error in estimating Aroclor and total PCB levels using the above approach is small compared to the large relative percent difference measured between duplicate analyses obtained by O'Brien and Gere. The large variability in duplicate measurements may be due to the heterogeneity of the sediments.

1984 Sediment Survey

Sediment samples collected in the 1984 sediment survey were analyzed by Versar, Inc. Versar's internal quality assurance program included evaluation of duplicate analyses and Aroclor 1242 spike recoveries. In addition, blind samples were submitted to Versar in duplicate but in different batches. M. P. Brown *et al.* (1988b) provide a summary of Versar duplicate and blind duplicate analyses as well as Aroclor 1242 spike recoveries.

The average relative percent difference between Versar-submitted duplicate analyses was 27 percent. This value is much lower than the values reported by O'Brien and Gere for the 1977 sediment survey. The average relative percent difference between blind duplicate samples was almost twice as large at 52 percent, but was still lower than the percent difference values reported by O'Brien and Gere.

The mean percent recovery of Aroclor 1242 was 86 percent. M. P. Brown *et al.* suggest that the reason for this low recovery might be the method used for quantifying Aroclor 1242. Versar originally used the method of Webb and McCall (1973) to quantify Aroclor 1242 levels, but only used peaks with relative retention times of 21 to 84. The omission of peaks with relative retention times of 11, 16, 98, 104, 125, and 146 would result in an underestimation of total PCB levels by approximately 11 percent for a true Aroclor 1242 mixture. An actual recovery of 100 percent would result in a calculated recovery of only 89 percent. This value of 89 percent is very close to the mean percent recovery of 86 percent. Thus, a recovery reported to be approximately 100 percent would actually overpredict PCB concentrations of the sample by approximately 11 percent to 14 percent. The data were not corrected for the percent recovery, so it is

anticipated that the reported levels are fairly close to the true levels.

In the 1984 sediment survey, Versar estimated PCB concentrations as Aroclors 1254 and 1260 using the method of Webb and McCall. It should be noted that there was not a strict correspondence between relative retention times reported by Versar and those of Webb and McCall, because of differences in the instrument packed columns and GC operating conditions.

Aroclor 1254 quantitation is uncertain for these samples because peaks 98 and 104 were not accurately quantitated (detection frequency of 2.6 and 2.9 percent, because of interference from the internal standard pp'-DDE which, by definition, elutes at a relative retention of 100. Peaks 98 and 104 contain approximately 21 percent of the PCBs found in Aroclor 1254. Thus, Aroclor 1254 levels are likely underestimated.

No Aroclor 1221 levels were reported in the 1984 sediment survey and, indeed, Aroclor 1221 is difficult to quantitate using GC-ECD. This situation results from the relatively poor sensitivity of the method to mono- and dichlorobiphenyls, the major constituents of Aroclor 1221. Early eluting peaks (Webb and McCall peaks 11 and 16) were identified in only two to three percent of the samples analyzed. Early eluting peaks were also rarely identified in the 1977 sediment survey.

Versar also reported total PCB levels for the 1984 sediment survey. As the methods used to calculate total PCB levels were not described, it is impossible to evaluate the data quality. Based on the reported Aroclor and total PCB levels, it was clear that total PCB levels were not simply the sum of the quantified Aroclor mixtures.

Packed column GC peak area data for the 1984 sediment survey are contained in the TAMS/Gradient database. (Access to the NYSDEC data was provided by GE.) This peak area data allows for an evaluation of the PCB quantitation procedure with respect to weathered samples. The average normalized peak areas for peaks 47 and 58 in sediment extracts are approximately 35 percent greater than the

average normalized peak areas for the same peaks in an Aroclor 1242 standard. The average normalized peak areas for peak 28 are approximately equal in extracts and standard. This finding suggests that using these peaks will tend to overestimate Aroclor 1242 levels to a greater extent in a weathered sample compared to a standard Aroclor 1242 mixture. This factor may account for the 40 percent higher Aroclor 1242 estimates obtained by M. P. Brown *et al.* (1988b) using a revised Webb and McCall procedure compared to the initial estimates obtained by Versar using the standard Webb and McCall procedure. It is not known to what extent this overestimation may offset the underestimation in Aroclor 1254 levels due to interference from the internal standard.

Other Sediment Data

In addition to the two major sediment sampling events just discussed, a number of other sediment surveys have been conducted. In 1983, the USEPA collected 66 sediment samples from the Upper Hudson River. These data were summarized in Volume 1 of the Feasibility Study for the Hudson River PCBs Site published by NUS (1984). Two different methodologies were employed to quantitate Aroclor concentrations. If the sample appeared to be characterized by a single Aroclor, then the sum of the areas of all PCB peaks was used to calculate the Aroclor concentration. If more than one Aroclor appeared to be present in the sample, then the Webb and McCall method was used.

Using the sum of the areas of all PCB peaks to quantitate a single Aroclor mixture will likely overestimate total PCBs present, if the sample is enriched in more highly chlorinated PCBs. This procedure will underestimate total PCBs present, if the sample is enriched in lesser chlorinated PCBs, because of the relative ECD response factors for lower and higher chlorinated PCBs. As mentioned earlier, the Webb and McCall procedure provides the best estimate of total PCBs in weathered samples.

In another study, sixty-five core sections were collected from the Upper Hudson River, extracted and analyzed for PCBs by Bopp *et al.* (1985), at the Lamont Doherty Geological Observatory. Total PCBs were quantitated using the

method of Webb and McCall (1973). Individual peak concentrations were calculated for peaks with relative retention times of 28 to 174. Webb and McCall-analyzed Aroclor 1242 and 1254 standards were used in the analyses to eliminate the uncertainty in using weight percent data supplied by Webb and McCall with Aroclor standards from other sources. In addition, Bopp *et al.* estimated original Aroclor 1242 concentrations in some samples by comparing a single peak (peak 47) in the sample to the same peak found in the standard, since Peak 47 was observed to be relatively stable in all core section extracts.

Total PCB estimates using the Webb and McCall procedure should be reasonably accurate. However, only peaks 28 to 174 were quantitated. Assuming that Aroclors 1242 and 1254 are found in the sediments, 15 percent of Aroclor 1242 may be omitted by neglecting peaks 11 to 21 and 3 percent of Aroclor 1254 may be omitted by neglecting to quantitate peaks 203 and 232. Observation of a major shift to higher relative amounts of lesser chlorinated PCBs in subsurface samples may increase the error in estimating total PCBs, since lower chlorinated PCBs eluting in peaks 11, 16 and 21 were not quantitated.

Hudson River Fish Samples

PCB concentrations have been measured in numerous species of fish collected throughout the Hudson River. Hazleton Laboratories America Inc. performed all the fish analyses since 1977 in an attempt to maintain some consistency in the analyses. PCB sample results of the project have been periodically published in numerous reports (Sloan *et al.*, 1983, 1984, 1986; Armstrong and Sloan, 1980a, 1980b) and are also in the TAMS/Gradient database.

Quality control data generated as part of the Hudson River Fish Monitoring Project consisted of: 1) analysis of periodic performance samples to assess accuracy; 2) analysis of one blank sample for every 20 fish samples; 3) analysis of one duplicate sample for every set of 20 fish samples; and 4) evaluation of the spike recovery of three Aroclors (1221, 1016, and 1254) in one sample for every 20 fish samples reported.

Summary quality control data generated between the years 1978 - 1981 of the Hudson River Fish Monitoring Project are presented in an appendix to a report prepared by Sloan and Armstrong (1980). All blank analyses were found to contain no PCBs. The average relative percent difference in Aroclor concentrations reported for duplicate analyses was small and varied from approximately 10 to 20 percent. The average recovery of Aroclors 1221, 1016 and 1254 spiked in fish flesh varied from 65 to 98 percent. In general, spike recoveries were highest for Aroclor 1254 and lowest for Aroclor 1221.

Aroclor mixtures in fish samples have been reported as concentrations of Aroclors 1221, 1016 and 1254. Quantitation was done by comparing the areas of one or several peaks to those produced by the respective Aroclor. Aroclor 1221 levels were calculated based on a single peak (Webb and McCall peak 11). Aroclor 1016 levels were quantitated based on two peaks (Webb and McCall peaks 37 and 40). Interestingly, Aroclor 1242 standards were used to quantitate Aroclor "1016" levels. Early in the project, it was determined that Aroclor 1242 patterns obtained from the fish extract were indistinguishable from that produced by Aroclor 1016 (Armstrong and Sloan, 1980). As a result, all Aroclor 1016/1242 levels have been reported as Aroclor 1016. Aroclor 1254 levels were quantitated based on three peaks (Webb and McCall peaks 125, 146 and 174).

Estimation errors of PCBs in the fish monitoring program can be evaluated in a fashion analogous to the evaluation of errors in the 1977 sediment survey. Peaks 37 and 40 were used to quantify Aroclor 1242 and are not found in Aroclor 1254. Peaks 125, 146 and 174, used to quantify Aroclor 1254, contain approximately 34 percent of the PCBs contained in Aroclor 1254 and only 2.6 percent of the PCBs found in Aroclor 1242. Peak 11 was used to quantify Aroclor 1221 in the fish samples. It represents 32 percent of the PCBs in Aroclor 1221 and only one percent of those found in Aroclor 1242. This selection of peaks for Aroclor quantitation appears to be a much better selection of peaks than those used in the 1977 and 1984 sediment surveys, because there is little overlap in Aroclors 1221, 1242 and 1254. Thus, on this basis, the reported levels in fish are anticipated to be fairly accurate to the extent that the chromatographic elution profile found in the fish actually resembled the Aroclor mixture quantified. If,

however, the chromatographic elution profiles resemble a heavily altered sample, there is little means of assessing the adequacy of the data.

USGS Water Column Data

The USGS began regular monitoring of PCB concentrations water samples collected from the Hudson River at Waterford in 1975. In 1977, water samples collected at Rogers Island near Fort Edward, Schuylerville and Stillwater were also included in the monitoring program. All samples were shipped chilled on ice to the USGS National Water Quality Laboratory in Doraville, Ga., where all PCB analyses were done.

Analysis for total recoverable PCB concentration was performed on unfiltered samples. Results, therefore, include the dissolved as well as the suspended fraction. Dissolved PCB concentrations were determined on samples filtered through a 0.45 μm silver oxide filter. Dissolved concentrations were determined on only a small percentage (<5 percent) of the samples.

Elapsed time from sample collection to laboratory analysis was reported as two to six weeks in 1984. Analysis of split samples retained in Albany for up to seven months indicated that storage time had no discernible effect on concentration. It is further contended that allowance of several days' contact time between solvent and water directly in the sample bottle is desirable because of PCB's affinity for particulates (Schroeder and Barnes, 1983).

Total concentrations were calculated by dividing the area of a sample's identified PCB peaks by the area of all peaks for an Aroclor standard, then multiplying this ratio by the concentration of the Aroclor standard. Requirements for a calculation were that at least 60 percent of the peaks in a standard be present in a sample and both relative peak ratios and column detention time must match. In a few samples, chromatographic peaks resembled a mixture of two Aroclors. In such cases, calculations were based on a standard containing two Aroclors. The data are generally reported as concentration of total PCBs, supplemented by notes as to presence or absence of specific Aroclors, but with

quantitation to specific Aroclors not given.

The uncertainty introduced by the use of the entire sample peak area to calculate the total PCB concentration in the USGS samples will vary to the degree that the elution profile does not match the standard profiles. It is not clear how great an error a 60 percent match will introduce, but it is clear that the error will increase the more the sample pattern is shifted to the lower chlorinated congeners peaks relative to the standard.

Concentrations were reported uncorrected for incomplete extraction. Schroeder and Barnes, however, contend that extraction efficiency is high (>80 percent) for Hudson River water, because the river is relatively low in suspended sediment and dissolved organic carbon concentrations. Extraction efficiency may, however, be an issue for periods of high suspended sediment.

B.3.7.3 Summary

PCB concentrations in Hudson River sediment, water and fish samples have been reported as Aroclors, despite the fact that heavily weathered samples may bear little resemblance to original Aroclor mixtures. A number of methods have been devised to quantitate PCBs as Aroclors and considerable variation exists between the methods. Most methods tend to overpredict total PCBs present in the sample.

Sediment Survey (1977-1978)

- PCB extraction efficiencies were on the order of 80 percent which leads to an underprediction of total PCBs.
- Aroclor 1242 levels were likely overestimated by basing quantitation on peaks containing congeners also found in Aroclor 1254; Aroclor 1254 levels were likely overestimated by basing quantitation on peaks containing congeners also found in Aroclor 1242.
- Lower chlorinated PCBs (Aroclor 1221) were rarely detected.
- Higher chlorinated PCBs (Aroclor 1260) were not quantitated.
- Summing aroclors may have led to an overprediction of total PCB con-

centrations.

Sediment Survey (1984)

- It is difficult to predict whether Aroclor concentrations presented in the 1984 sediment survey overpredicted or underpredicted total PCB levels.
- Mean Aroclor 1242 extraction efficiency was 86 percent, possibly leading to an underprediction of Aroclor 1242.
- Lower chlorinated PCBs (Aroclor 1221) were not quantitated.
- Aroclor 1254 levels were likely underestimated, because of interference from the internal standard pp'-DDE.
- Total PCB levels were quantified, but the method of quantification could not be evaluated.

Fish Survey (1977-1988)

- The Aroclor measurements and quantitation were performed by one laboratory, giving what should be a consistent set of results. Aroclor results appear to be reliable.

USGS Water Column Data (1975-1989)

- Total PCBs were reported (not Aroclor mixtures), thus avoiding the inherent ambiguities and uncertainties of modelling an environmentally sampled PCB with an Aroclor mixture. (USGS reported that most samples were in the Aroclor 1232 to 1248 composition range.)
- Few (<5 percent) of the water samples were analyzed for dissolved PCBs.
- Data may be subject to uncertainties via quantification based on total peak area.

Other Data Sources

- Other data sources that provide information on PCB levels in sediment at various points in time exist.
- These records do not constitute a continuous monitoring program, but can provide a cross-reference on other data.

The data examined here are adequate for assessing gross trends. In most cases direct comparisons of Aroclors reported for different media are problematic, because of the different procedures used to measure Aroclors in each medium. Aroclor concentrations reported for the same medium are generally adequate for assessing temporal or spatial trends in the data, but different quantitation methods used by different laboratories complicate direct comparisons. Congener-specific PCB analyses would alleviate problems in estimating total PCBs and allow comparisons of data among groups.

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SYNOPSIS

DATA SYNTHESIS AND EVALUATION OF TRENDS

(Section B.4)

Detailed interpretation and analyses of the Upper Hudson monitoring data have focused on the potential for migration and redeposition of PCBs in sediments and evaluation of statistical relationships among PCB concentrations in sediments, water and fish.

Three questions posed as the objectives of these analyses concern the cycling of PCBs in sediments and water and their impact on the fish population (B.4.1).

Flood flow and sediment transport (B.4.2) are addressed first. USGS data are used to analyze flood recurrence intervals and the relationships between flow and sediment load. The flood frequency analysis suggests that other investigations may have overestimated the magnitude of the 100-year flood in the Thompson Island Pool by 25 percent. This finding implies that the potential for scour of contaminated sediments may be less than previously estimated. Analyses also suggest that a decline in suspended sediment load has occurred over time.

An investigation of the relationship between PCB concentrations and flow and estimation of mass loading from the Upper to Lower Hudson is presented. PCBs in the water column and mass discharge (B.4.3) are difficult to evaluate, because relatively few samples are taken at a station in a typical year, whereas PCB concentrations may change rapidly with changes in river flow. PCB concentrations and trends must, therefore, be inferred from an incomplete time series of measurements. PCB concentrations in the Upper Hudson have shown a bimodal relationship to flow, increasing at both high and low river flows. Separate multiple regression models, fit for high and low flow regimes at each station, do not yield great predictive strength. A negative correlation between PCB concentration and year is found at all stations, indicating a gradual decline of PCB concentrations in the water column over time. Estimates of PCB mass loading from the Upper to the Lower Hudson are evaluated. PCB measurements are biased toward high flow events. Mass load, not measured directly, must be estimated statistically. To correct the sampling bias, this analysis adopted a new method of estimating the load. In the period of 1984-1989, little increase in total load between Fort Edward and Waterford appears to have occurred. This finding led to the unexpected conclusion that much of the load in recent years appears to come from locations upstream of the Thompson Island Pool. Use of the new analysis also indicates that of PCB loading from the Upper to the Lower Hudson may have been overestimated previously.

PCB levels in fish have declined over the last ten years, exhibiting approximate exponential decline patterns, with a leveling out or stabilization in recent years (B.4.4). For the 11-year sampling record at River Mile 175, levels of a less chlorinated mixture of PCBs (Aroclor 1016) in fish exhibit an apparent half-life of 3 to 4 years. The rate of decline of the higher chlorinated congeners (Aroclor 1254) appears to be much slower, with an apparent half-life of 7 to 40 years. Time-trend regression equations are used to obtain an approximate estimate of

total PCB levels in fish over the next 30 years, assuming the current declining trend continues.

PCB mass transport and PCB levels in fish in the Upper Hudson both exhibit generally declining trends over time. Despite the large number of sediment samples that have been analyzed, shifting sediments, widely disparate sampling densities and uncertainties in PCB measurement methods all confound the interpretation of the sediment sample results. Available data are insufficient to relate PCB concentrations in fish to PCBs in sediments. In order to understand better the exchange of PCBs between sediment, water and fish, detailed PCB analyses related to specific forms of PCBs (congeners) will be necessary. Among the questions still to be answered are whether the PCB levels will continue their observed decline and what specific conditions would alter their decline.

B.4 Data Synthesis and Evaluation of Trends

B.4.1 Phase 1 Objectives

In general terms, a benefit/risk analysis of existing conditions and possible remedial actions in the Upper Hudson depends on an understanding of how PCBs transfer, accumulate and dissipate in sediments, water, fish and other media. Perhaps the most visible net effect of the exchange of PCBs in this system has been PCB uptake in fish. Use of the PCB levels in fish as a barometer of the overall status of PCBs in the Upper Hudson is, therefore, considered appropriate and may ultimately provide a means to evaluate benefits and risks of remedial measures *versus* current trends. If the fish data are to be utilized as a future barometer to evaluate the effects of remedial actions, then the following questions need to be answered:

(1) What is the potential for migration and redeposition of PCBs in sediments?

(2) How are PCBs in sediments transferred to the water-column?

(3) What is the effect of (1) and (2) on bioaccumulation of PCBs by fish?

These three areas of investigation, shown schematically in Figure B.4-1, provide an initial framework within which to reassess the PCBs in the Hudson River and focus attention on those questions most important to the reassessment. To meet this end, initial results of scour and flow relationships and correlations of PCBs in water and fish are examined.

B.4.2 Flood Flow and Sediment Transport

B.4.2.1 Flood Frequency Analysis

This section evaluates USGS flow data to derive flood recurrence probabilities for the Fort Edward and the Thompson Island Pool areas. This

evaluation suggests that earlier investigations of flood flows significantly overestimated the magnitude of the 100-year flood for the upper river.

Analysis of both average flow and flood recurrence intervals is important for assessing the probability of future high flow events, which could scour PCB-contaminated river sediments. Figure B.4-2 shows the flow-duration curves generated from 13 years of daily average flow observations at Fort Edward. The curves show the exceedance probability for any specified flow. For instance, at Fort Edward, only 1 percent of observed daily average flows exceeded 20,800 cfs, while 50 percent of the flows exceeded 4,060 cfs. The period of record at Fort Edward is relatively short (13 years), which could lead to inaccurate estimates for extreme events. A 69-year record upstream at the Hudson River below the confluence with the Sacandaga River, reported by USGS from flows at Hadley plus flows at Stewarts Bridge in the Sacandaga, is also shown in Figure B.4.2. This record provides better estimates for extreme events.

To estimate a flood recurrence interval it is necessary to form a partial duration series of the flood events. Series were formed for both daily average and peak value flows. The partial duration series of interest here is the annual maxima series (largest flood event in each year of record). This evaluation provides an estimate of the annual flood recurrence interval or return period. Using observed annual peak flows, this recurrence interval can be calculated as (N+1)/M, where M is the rank order of the observed flood (*e.g.*, for the largest flood M=1) and N is the total number of years of observation. The time series available at Fort Edward is not sufficiently long to provide a reliable estimate of the extremes of the distribution. The occurrence of floods can be estimated from the longer record (the upstream station of the Hudson River at Hadley to which discharges from the Sacandaga River have been added) and the floods can be translated to the Thompson Island Pool and other areas downstream (Figure B.4-3a).

Before proceeding with the flood frequency analysis for the Thompson Island Pool, two caveats must be stated:

(1) The longer (69-year) monitoring period for Hadley is inadequate to estimate extreme floods with recurrence intervals exceeding the length of the record (*e.g.*, >70 years). A Log Pearson Type III extreme value distribution provides a means to estimate floods of longer recurrence intervals (USGS, 1982). Figure B.4-3a plots the empirical flood recurrence values for the 69-year record, together with estimates of larger (100-year) floods at Hadley.

(2) A plot of the *annual* maxima by year (Figure B.4-3b) reveals that the three highest flows on record all occurred in the 1920s. This finding suggests that the construction of the Sacandaga dam in 1930 significantly altered the flow regime and reduced the magnitude of floods.

Flood frequencies calculated using a Log Pearson Type III distribution (USGS, 1982) on the post-1930 data only yield estimates of daily floods as shown below for the Hudson River below Sacandaga River.

Estimated (Log Pearson Type III) Daily Flood Events, Hudson below Sacandaga (Post 1930)

Recurrence Period T (yrs)	1931-1989 Flood Flow Q_T (cfs)
5	27,964
10	32,027
25	36,876
50	40,314
100	43,621

During the 14 years following removal of the Fort Edward Dam, the maximum daily flood observed in the Upper Hudson occurred in April 1976, with a daily average flow of 39,340 cfs reported below Sacandaga. The above analysis implies that this flow is less than a 50-year flood. Frequency analysis with a log normal distribution (Bras, 1990) yields a similar result, with the 50-year flood estimated at 41,144 cfs for a daily average flow.

Peak flows (as opposed to maximum daily flows) are of most interest for the assessment of flood damage and the determination of maximum erosional shear stresses leading to sediment scour. Peak flows were not regularly monitored at Fort Edward until 1976. USGS does not measure peak flows for the Hudson immediately below the confluence with Sacandaga River (see Plate B.1-1). Because the Sacandaga is controlled, peaks on the two rivers are unlikely to coincide. Therefore, a method had to be developed to estimate peak flow at Fort Edward from peak flow data for the Hudson River at Hadley for the period prior to 1976. The 100-year peak flow flood at Fort Edward has been estimated at 41,[illegible]0 cfs (Malcolm Pirnie, 1975). This estimate was obtained using a *direct* translation of the 100-year flood at Hadley downstream, without proration for drainage area and further assuming that the flow in the Sacandaga would be zero during flooding in the Hudson. Examination of flow records in the Sacandaga shows that the zero flow assumption is not always valid. The method of the Federal Emergency Management Administration (FEMA, 1982 and 1984) for estimating flood recurrence at Fort Edward based on peaks at Hadley was to assume that the Sacandaga reservoir would contribute 8,000 cfs of flow to the Hudson River during extreme flood events. This discharge results from the opening of one control valve, a procedure often followed during major storms to prevent topping of the dam. With this assumption, FEMA used a Log Pearson III distribution, fit to the period of record for the Hudson at Hadley, and accounted for the increased drainage area between Hadley and Fort Edward to predict flood flows at Fort Edward. Assuming a constant 8,000 cfs contribution from the Sacandaga is likely to overestimate the magnitude of floods, is a conservative assumption for a flood insurance study.

Examination of the record shows that daily average flow in the Sacandaga at Stewarts Bridge was often near zero on the days of peak flow at Hadley; it exceeded 8,000 cfs only twice between 1930 and 1976. For the present study, a modified approach was undertaken. Peak flows at Fort Edward prior to the period of record were estimated from peak flows at Hadley plus measured daily flows in the Sacandaga at Stewarts Bridge, prorated downstream according to the following equation:

$$Q_{PK}(FE) = [Q_{PK}(H) + Q_{DA}(S)] \times \left(\frac{2817}{2719}\right)^{.75}$$

where

$Q_{PK}(FE)$	=	estimated flood peak at Fort Edward (cfs);
$Q_{PK}(H)$	=	measured flood peak at Hadley (cfs);
$Q_{DA}(S)$	=	daily average flow of the same date at Stewarts Bridge on Sacandaga River (cfs); and
(2817/2719)	=	ratio of contributing areas.

This equation is similar to the FEMA (1982 and 1984) method, except that it does not assume a constant input from the Sacandaga Reservoir.

Based on the measured daily flows at Stewarts Bridge, the greatest flows in the Hudson below Sacandaga did not always occur during peak flows at Hadley; instead, they sometimes occurred later, after the passage of the flood peak at Hadley, when the dam was reopened on the Sacandaga. This occurrence apparently applies only to flood flows in the lower part of the regime (<28,000 cfs). The calculated annual peak partial duration series at Fort Edward was checked against the annual partial duration series of daily maxima and the latter value was substituted into the peak file when greater. This procedure necessarily underestimates some unmonitored flood peaks in the Hudson below Sacandaga, but the procedure is not thought to have a major impact on estimated recurrence frequency of extreme events.

Peak flow data are available directly at Fort Edward for December 1976 through September 1990. Therefore, the above method was used to estimate floods at Fort Edward prior to the period of record, for water years 1930 through 1976. These were then combined with actual measured peak flows at Fort Edward for 1976-1990 and a Log Pearson III distribution fit to the full data. The annual maximum peak and daily average flood recurrence intervals calculated by this method at Fort Edward are shown in Table B.4-1.

Estimation of the magnitude and recurrence interval of future expected floods is of significance in assessing the future potential erodibility of the remnant and hot spot deposits. Examining the flood recurrence interval estimates in Table B.4-1 suggests that the magnitude of expected floods may have been overestimated by past researchers. For instance, this TAMS/Gradient estimate of the 100-year recurrence flood expected at Fort Edward is approximately 8,000 cfs less than that estimated by FEMA (1984).

The largest peak flow experienced at Fort Edward since the removal of the dam was that of April 1-2, 1976. Although a gauge was not in operation, it was estimated by FEMA (1984) that the peak flows in 1976 were greater than 50,000 cfs and approximated a 100-year flood. This analysis may not be accurate. Translating the combined flows in the Hudson River at Hadley and in the Sacandaga River at Stewarts Bridge downstream suggests that the flood peak at Fort Edward in 1976 should have been in the neighborhood of 45,000 cfs. This would be approximately a 100-year flood by TAMS/Gradient estimates, but less than a 50-year flood by FEMA estimates.

There is considerable doubt concerning the estimated peak flow magnitude of floods prior to the institution of actual gauging. The estimated daily average flow at Fort Edward during the flood event of April 2, 1976 (about 40,750 cfs) was also less than a 100-year daily flood. The next downstream station operating at this time was at Green Island (below the Mohawk River), where an annual peak flow of 106,000 was reported on April 2, 1976. On this date, flow in the Mohawk at Cohoes was 52,100 cfs, leaving only 53,900 cfs attributable to the upstream Hudson. This flow would correspond to a flow at Fort Edward in the neighborhood of 39,000 cfs. The data suggest that the magnitude of the 1976 peak and daily flood events may have been overestimated. Possibly, an extreme high stage observed at Fort Edward in 1976 did not correspond to the estimated flow, which would be derived from the stage-discharge curve developed in 1977 by the USGS.

The magnitude of major flood events has considerable significance for the analysis of erodibility of contaminated sediments in the Thompson Island Pool. An attempt was made to assess erodibility in this area in 1985 (Zimmie, 1985), using the HEC-6 computer model. In Zimmie's study, the 10-year and 100-year peak flood events for the Thompson Island Pool were assumed to be 46,600 and 63,700 cfs, respectively, citing the Fort Edward flood insurance studies (FEMA, 1980, 1984). Actually, these figures are the peak discharge estimates proposed by FEMA (1980, 1982) for the *downstream corporate limits* of the *Town* of Fort Edward; these limits are located *below* Fort Miller, two dams downstream of the Thompson Island Pool. FEMA (1982) estimates for the 10 and 100-year peak discharges within the Thompson Island Pool, upstream of the confluence with Moses Kill, are 41,900 and 56,800 cfs, about 10 percent less in magnitude. The FEMA calculations were based on estimates of 10 and 100-year peak discharges at the Village of Fort Edward of 38,800 and 52,400 cfs. The current analysis, with the benefit of a longer record, suggests that these values are overestimates.

Using the TAMS/Gradient peak flood calculations at Fort Edward, peak discharges for the 10 and 100-year events in the Thompson Island Pool, upstream of the confluence with Moses Kill, would be about 37,000 and 49,000 cfs respectively. Discharges modeled by Zimmie may have overestimated the 100-year peak discharge in the Thompson Island Pool by about 14,000 cfs and, indeed, were in excess of the expected 500-year peak discharge, using values computed for this study and presented in Table B.4-1.

B.4.2.2 Suspended Sediment Discharge

Understanding the movement of sediments in the Upper Hudson is essential to evaluating the movement of PCBs adsorbed to the sediments. The natural rate of sediment transport in the Upper Hudson River is relatively low compared to many other eastern North American rivers of similar size, because of comparatively low rates of erosional sediment input, combined with extensive sediment trapping behind the many dams on the river. Nevertheless, a period of extensive, PCB-contaminated sediment scour and transport occurred after removal of the Fort Edward Dam in 1973.

Important early data on sediment transport in the Upper Hudson was documented in the engineering studies investigating conditions following the removal of the Fort Edward Dam (Malcolm Pirnie, 1975, 1976). This dam, completed in September 1822, was reconstructed in 1898 as a rock-filled timber crib, approximately 586 feet long, with an average height of approximately 19 feet. The dam created a pool about two and one-half miles long and about 400-800 feet wide. Extensive sediment deposits, largely derived from lumber industry debris, accumulated in the pool behind the dam.

Because of severe deterioration of the structure, the Federal Power Commission granted permission to remove the dam and it was removed in July-October 1973. With removal of the dam, the impounded pool disappeared and the river eroded a channel into the entrapped sediments, leaving large sediment deposits or remnants exposed along the river banks. Now called remnant deposits, these are shown in Plate I.2-3. Spring floods of 1974 mobilized large volumes of sediment and debris from the former dam pool; by the beginning of the navigation season in 1974, the channel was blocked by debris to about one quarter mile below Lock 7. It is estimated that 850,000 cubic yards of debris were scoured from the former dam pool between July 1973 and July 1974, but that 775,000 cubic yards of this total were deposited between the dam site and Lock 7, from which they were dredged in 1974-75 (Malcolm Pirnie, 1976). It was later determined that these sediments contained large amounts of PCBs.

The sediments behind the former Fort Edward Dam were not typical natural sediments, but contained a significant proportion of sawdust particles and other wood debris, derived from historical Adirondack timber operations. This organic material was characterized by a low specific gravity and a low settling rate, which had impact on subsequent transport. Borings behind the dam in 1970 (Dames and Moore, quoted in Malcolm Pirnie, 1975) indicated that the bottom of the pool was covered by a weak, compressible organic silt and paper waste from 2 to 14 feet deep. Another 1970 study, designed to assess metal contamination behind the dam, concluded that the sediments behind the dam consisted of a "transient deposit" of brown fibrous sludge, covering a black sandy silt and stated: "No

appreciable amount of organic silt was encountered" (Clarkeson & Clough, quoted in Malcolm Pirnie, 1975).

Major spring floods (peak of approximately 100-year recurrence interval) occurred between March 31 and April 8, 1976. Malcolm Pirnie estimated that an additional 36,000 tons of sediment were scoured from the remnant deposits and transported into the Thompson Island Pool by this flood, while about 45,000 tons of sediment were transported out of the lower end of the Thompson Island Pool (Malcolm Pirnie, 1976). The net increase in sediment load leaving the pool likely included substantial scour of sediments deposited in 1974.

The 10-year recurrence daily average flood at Fort Edward is estimated to be about 31,200 cfs (see Table B.4-1). Daily floods in excess of this magnitude have been experienced only twice since 1976, on April 29, 1979 (31,700 cfs) and on May 2, 1983 (32,600 cfs).

Information on time trends in suspended sediment data, as well as discharge, are provided by USGS monitoring stations (see B.3). Suspended sediment measurements did not commence until 1975 nor is monitoring continuous or on a set schedule. Measurements have focused on spring flood periods, with little data available for the winter months. Lack of a more extensive data base and of a regular time series creates difficulties in analyzing sediment data and other water quality parameters.

Use of the monitoring records to establish relationships between suspended sediment concentration and discharge assumes that conditions have not changed over time. Figures B.4-4 through B.4-7 show plots of suspended sediment concentration (mg/l) versus discharge. The rating curves show a steady increase in sediment concentration moving downstream from Fort Edward to Waterford. The relationship appears to exhibit breakpoints or sills, previously noted by Zimmie (1985), which occur when suspended sediment concentrations remain at a rather steady, low level until a critical (threshold), flow velocity is reached. Thereafter, concentrations increase as a function of discharge. Such behavior is thought to represent an approximate critical shear stress for sediments in the

river. At Fort Edward, the sill appears to extend to about 10,000-12,000 cfs (283-340 m^3/sec). At Schuylerville, the relationship is not as clear, perhaps, because the station is just below the confluence with Batten Kill. Breakpoints in the sediment response also appear further downstream at Stillwater and Waterford, with an apparent increase to a range around 19,000 cfs by Waterford.

The destabilization of the channel following the removal of the Fort Edward Dam in 1973 suggests that a decline in average suspended sediment levels was to have been expected as the river gradually recovered to a more equilibrium condition and the remnant remediation was completed. Time trends of observed sediment load at Fort Edward and Schuylerville, shown in Figures B.4-8 and B.4-9, suggest a decline over time, particularly at Schuylerville. Although high sediment loads typically occur during spring flood periods, greater sediment load is shown for the moderate floods of 1981-1982 than for the major floods of 1979 and 1983. It could be that a limited sampling schedule missed the sediment transport peak during the major flood years. Between 1984 and 1989, daily average flows greater than 28,000 cfs occurred only in spring 1987 and a clear sediment load peak is evident in response to this event.

Empirical Trend Analysis

Time trends in suspended sediment concentration, corrected for discharge, can be examined through multiple regression relating total suspended sediment concentration (TSS) to discharge and year to better understand sediment transport relationships. A log transformation of concentration is necessary to stabilize the residual variance. Models can then be fit in the following form:

$$LN\ (TSS + 1) = \alpha + (\beta_1 \times Q) + (\beta_2 \times Yr)$$

where TSS is the sediment concentration (mg/l), Q is the measured or estimated instantaneous discharge (cfs) and Yr is the years since 1900. (Because a log transformation is used for TSS, a "1" is added to TSS, to handle zeros in the

record.) Very similar models can be fit for Fort Edward, Schuylerville and Stillwater.

Fort Edward

$$LN\ (TSS + 1) = 2.87 + 9.5x10^{-5} \times Q - 0.0223 \times Yr$$

$$R^2 = 70 \text{ percent}$$

Schuylerville

$$LN\ (TSS + 1) = 4.61 + 8.6x10^{-5} \times Q - 0.0386 \times Yr$$

$$R^2 = 63 \text{ percent}$$

Stillwater

$$LN\ (TSS + 1) = 3.80 + 9.9x10^{-5} \times Q - 0.0291 \times Yr$$

$$R^2 = 68 \text{ percent}$$

For a constant discharge, the relationships imply that sediment concentrations show an exponential decline against year. All the regression coefficients in these equations are significant at the 95 percent level. The average slope against year is -0.03, which indicates sediment load corrected for variability in flow has declined since 1977 with an apparent half-life of 23 years. Such a decline may to some degree represent a depletion of readily erodible sediments from behind the former Fort Edward Dam. The decline may also be a consequence of somewhat lower than average flows in the mid 1980s or a combination of these and other factors.

B.4.3 PCBs in the Water Column and Mass Discharge

This section presents extended statistical analyses of water column monitoring data, with particular emphasis on exploring relationships to aid in understanding and predicting the concentrations and mass transport of PCBs in the water column.

B.4.3.1 PCB-Discharge Relationships

Regression Analyses

As was the case with sediment data, PCB measurements at USGS monitoring stations are gathered at irregular intervals in time, so that standard time series statistical tests cannot be applied.

The time series of PCB concentrations in the water column show a high degree of similarity among Schuylerville, Stillwater and Waterford. The coincidence of peaks among these records would naturally seem to be controlled by hydrologic events, particularly flood/scour episodes. This relationship is shown in Figures B.4-10 and B.4-11, which plot daily average flows at Fort Edward together with the total PCB measurements at Fort Edward and at Schuylerville, the nearest long-established station downstream of the Thompson Island Pool. The time series show a clear response in PCB levels to the major flood event of 1979 as well as to the smaller flood events in 1981. The extreme flood event of 1983 seems, however, to have produced only a modest response at Schuylerville in PCB concentrations; subsequent floods also seem to have produced less of a response than is found in the 1979-1980 period. One explanation is that the amount of PCB-contaminated sediment available for scour and suspension into the water column has been steadily reduced. In particular, the 1979 flood may have removed much of the easily erodible, contaminated sediment in the remnant deposits and Thompson Island Pool, while burying other contaminated sediments.

There are some other functional differences between the periods before and after 1979. For instance, between 1972 and 1979 there was channel maintenance dredging yearly in the reaches between Schuylerville and Fort Edward, whereas no dredging has occurred there since 1980. Dredging around Lock 7 and Fort Edward occurred in 1974, 1976, 1977, 1978 and 1979 (Ryan, 1991). Maximum volumes were removed from this area in 1974 (351,000 cubic yards) and 1979 (66,930 cubic yards), following the floods that redistributed the remnant deposits. Destabilization of the channel margins following dredging may have excaberated the erosion of contaminated sediments. Furthermore, 1974-75 dredge spoils placed

on Rogers Island at Fort Edward and may have provided a source of contaminated washoff into the river.

Variations in PCB concentrations in the water column are related to variations in river discharge, but the relationship is complex. Various authors (Turk and Troutman, 1981; Schroeder and Barnes, 1983; NUS, 1984; Bopp *et al.*, 1985) have proposed that total PCB concentrations in the water column may have a bimodal relationship to flow, with measured concentrations increasing at both high and low flow extremes. The high flow peak has been attributed to increased scour of contaminated sediments, whereas the low flow peak has been attributed to decreased dilution. Increasing PCB concentrations at low flows presume a relatively constant rate of loading of PCBs, either by desorption from the sediments or in base flow, accompanied by little or no erosional input, yielding concentrations inversely related to flow.

Measurements of both dissolved and adsorbed PCBs in the water column, involving filtration of raw river water samples and subsequent analysis of the two fractions (Bopp, 1979); Tofflemire, 1980; Turk and Troutman, 1981; Bopp *et al.*, 1985) indicate that PCBs are often detectable in the water column in both adsorbed and dissolved form. The dissolved component actually refers to concentrations that will pass a fine filter and, thus, includes both truly dissolved constituents and PCBs sorbed to tiny organic particles (Bopp *et al.* 1985). The dissolved component is thought to predominate at low flows, exhibiting "surprisingly high" concentrations (NUS, 1984) above what would be predicted for solution equilibrium and up to 0.5 ppb (Tofflemire, 1980).

At high river flows, the PCBs in the water column are more likely to be predominantly sorbed to sediment. This occurrence is theorized to represent the scour of contaminated sediments during high flow events, resulting in transient mass fluxes of PCBs in the water column. At such times the dissolved and adsorbed PCB fractions may not be in equilibrium with the sediments and water.

Previous analysis of the PCB concentration-discharge relationship has been based primarily on observations prior to 1980. Figures B.4-12 through B.4-15 plot PCB concentrations versus daily average flows at Fort Edward, Schuylerville, Stillwater and Waterford for the full period of record (1976-1989). A visual inspection of these plots reveals surprisingly little obvious relationship between flow and concentration, with non-detects occurring at almost any flow level. This occurrence is particularly true at Fort Edward. At the stations below Thompson Island, the bimodal relationship of concentration to flow is more evident, with peaks found in both the low flow and high flow parts of the graph. While high concentrations are not found at middle level flows, low concentrations continue to be found at all flow levels, indicating that flow alone is not a particularly good predictor of PCB concentration.

As shown on these plots, the apparent bimodal relationship to flow is most apparent in the pre-1985 observations. The more recent PCB concentration data seem to bear little clear relationship (direct or inverse) to flow alone, without correcting for other influences. Similarly, the relationship between levels of suspended solids levels and PCB concentrations in the water is surprisingly weak (see Figure B.4-16 for the relationship at Stillwater). Nevertheless, it does appear that the highest PCB concentrations in the water column generally occur at high flows and accompany high sediment loads.

The relationship between PCB concentrations and flow may be obscured by joint correlation with other related variables. This possibility was explored through the use of stepwise multiple regression, in which flow, the inverse of flow, sediment concentration, year and month in year were considered as possible regression variables. A transformation to the log of PCB concentration was used to stabilize the residual variance for high flows. The bimodal hypothesis suggests that PCB concentrations should begin to rise with flow when the shear stress at the sediment water interface reaches a certain critical value, resulting in the resuspension of PCB-contaminated sediments. This hypothesis implies that separate models should be fit for observations above and below some critical flow value, distinguishing scouring versus non-scouring flows. From the sediment rating curves, the non-scour/scour breakpoints appear to be at about

11,000 cfs at Fort Edward, 12,000 cfs at Schuylerville, 16,000 cfs at Stillwater and 19,000 cfs at Waterford. The relative values correspond to increasing drainage area moving downstream and are in agreement with the conclusion of NUS (1984) that "the transition from one form of PCB (transport) to the other varies at flows ranging from 10,000 cfs to 20,000 cfs."

Regression models were tested for both high and low flow regimes. Non-detected PCB values were set at one half the detection limit for the regressions. The F-test statistic at a 95 percent significance level was set as the criterion to retain a variable as "significant" in these regressions. Table B.4-2 summarizes the variables that are significant in these regression equations. As shown by the relatively low correlation (R^2) coefficients in this table (ranging from 0.25 to 0.70), these regression equations do not have great *predictive* strength. The regression coefficients on the variables, however, are significant at a 95 percent confidence level and, thus, there are strong correlations between these variables and total PCB concentrations. For all four stations, high-flow PCB concentrations are positively correlated with either discharge or sediment concentration and these two variables are strongly correlated with one another. A negative correlation between PCB concentration and year is also noted. This relationship may result, in part, from the decrease in the detection limit from 0.1 to 0.01 µg/l at the start of water year 1987. While the peak PCB concentrations in the water column of 1979-80 have not been observed since, the models still show a significant response of concentrations to high flow/erosional events. For low flows at all stations, there is also a decline with time as well as an inverse relationship to discharge (the dilution effect). The decline with time may represent a gradual depletion of the more readily soluble congeners in the upper layers of sediment.

The regression coefficients with time (year) for the Fort Edward station are somewhat lower than the respective coefficients at the other downstream stations. This outcome suggests a somewhat less significant decline of PCB concentrations in water over time at Fort Edward.

Non-parametric Tests of Trend for Water Column PCBs

As noted above, the negative correlation detected with time in the regression equations may be partly accounted for by the change in detection limit. Nevertheless, there has been a statistically significant downward trend in concentrations during the period of monitoring. For instance, if all values less than 0.1, detect or non-detect, are set to 0.1, the regression still shows a significant negative correlation between concentration and year at the downstream stations. The interpretation of "significant" depends on assumptions regarding the distribution of the parameter, *i.e.*, normal, lognormal, *etc.* Further, the extreme variability of concentrations during flood events may obscure the trend. For this reason, NUS (1984) advocated assessing trends with time on low flow concentrations only. They indicated that low flow concentrations at all stations had decreased with time, with the decrease being statistically significant between 1979 and 1980.

A non-parametric test, which makes no distributional assumptions and is not overly sensitive to the variability of high concentration events, was undertaken in order to confirm the trend. The non-parametric test used here is that advocated for water quality data by Lettenmaier *et al.* (1991), which was applied to the time series of PCB observations at Schuylerville. The presence of multiple detection limits is treated by: 1) setting all data flagged as non-detect to the highest detection limit; and 2) setting all measured concentrations below the highest detection limit to the highest detection limit. The non-parametric Spearman Rank Correlation test was then applied to examine the correlation between PCB concentration and discharge. As this test did not show significance at the five percent level, flow-adjusted concentrations were not needed for the test.

The time series of yearly averages was then examined using the Mann-Kendall test (Gilbert, 1987), a non-parametric test of trend in which missing values are allowed to occur. For the 1977-89 series of annual mean PCB concentrations in the water column at Schuylerville, the Mann-Kendall test results indicate a statistically significant decline (>95 percent confidence) in PCB concentration

with time. Similar results hold for the other stations downstream of the Thompson Island Pool. The only exception to this result is mean annual PCB concentrations at Rogers Island (1978-1989) for which the trend is not significant with time at the 95 percent confidence level. In other words, the robust, non-parametric test confirms that the trend at Fort Edward is, indeed, toward lower mean levels, but the evidence is not sufficiently strong to conclude that this is not due to random variation.

B.4.3.2 Mass Transport Estimates

Estimates of PCB mass transport rates are critical to assessing the impact of the Upper Hudson contamination on the PCB problem in the estuary and to evaluating remedial actions in the Upper Hudson. Where such rates can be determined, they reveal the magnitude of the problem and the relative contribution of various source areas.

Accurate estimation of PCB mass transport is a difficult problem in light of the available data. While river flow has been monitored on a continuous basis, PCB measurements in the water column consist of 10-50 small-volume samples per year, taken at irregular time intervals. Major portions of the yearly load may be transported during a few brief flood events. The problem lies in filling the gaps in the PCB monitoring record in order to estimate the integrated total mass over an entire year. For this reason, estimates of annual PCB load have varied widely (summarized in NUS, 1984). Two approaches to estimating annual PCB load were considered here, each with particular problems. One approach is to develop a regression analysis on the available data, relating observed PCB concentrations to flow, thus creating a continuous PCB record. A second approach is to develop a direct estimate of the yearly load. As will be seen below, the first approach is inherently biased. A bias is also present in the monitoring data, which would tend to an overestimation of load. Robust methods of estimating the average can correct for this problem and were developed for this study. The results are expected to be unbiased, but, for the periods of high transport, they have wide uncertainty ranges, *i.e.*, large error bars.

Regression Approach

In the first approach, regression equations (previously discussed) would relate PCB concentration to time, flow and sediment concentration. Sediment load, which is not monitored continuously, cannot be used in the regression to be useful for continuous prediction. Furthermore, PCB load is not measured directly; instead, it must be estimated from the PCB concentration multiplied by the discharge (Q). Thus, it is *not* appropriate to develop a regression equation relating PCB load (L) to discharge, because discharge would appear on both sides of such an equation. Because PCB concentration must also be predicted from flow, the residuals from such an equation will not meet independent normality assumptions, but will be dependent in magnitude on discharge. A substantial proportion of the yearly total mass transport is dependent on a few high flow-high PCB concentration events, which will fall into the statistical noise of the regression/extrapolation procedure. An estimate of yearly load based on the regression of concentration on flow would tend to underestimate the total load contributed by high flow events and, thus, underestimate the yearly totals. Indeed, use of the regression equations derived for concentration to predict loads from flows results in estimates of total load that are consistently smaller in magnitude than estimates obtained through a robust estimate of the mean.

Schroeder and Barnes (1983a), building on earlier work of Turk and Troutman (1981), estimated low flow PCB loads using regression analysis. They divided the flow regime at each monitoring station into scouring and non-scouring segments, defined by a 600 m^3/sec flow (21,186 cfs) at Waterford. The high flow (scouring regime) PCB load was identified as highly variable and dependent on the specific source areas contributing flows to the event. Therefore, no model was proposed for this portion of the load, even though it may be the dominant contributor to total mass, and total mass calculations were not made. Schroeder and Barnes did propose yearly (water year) linear regression models for non-scouring-flow loads, for which they reported statistically significant relationships of PCB transport rates to inverse of discharge and relatively high R^2 values (50-98 percent). They adopted a no-intercept model, in which PCB concentration (C) during non-scouring flows is modeled as inversely related to discharge (Q):

$$C = \beta * \frac{1}{Q}$$

where β is the regression coefficient. Imposition of a no-intercept model essentially assumes that concentration declines to zero at infinite flow, via dilution. This assumption is dubious, as the model is explicitly restricted to no-scour situations and the maximum non-scouring flow does not exhibit infinite dilution. By fitting the following non-zero intercept model:

$$C = \alpha + \frac{\beta^*}{Q}$$

the intercept (α) is found to be more significant than the slope parameter (β^*), and the R^2 value for the regression declines to very low levels (typically <10 percent). The implication is that the significance of the relationship is focused in the constant intercept value. Indeed, what Schroeder and Barnes have actually done is to fit a *constant* load model to the non-scouring regime, wherein the load (L) is not dependent upon discharge at all:

$$L = C \times Q = \frac{\beta}{Q} \times Q = \beta.$$

This model implies that under no-scour conditions a constant load of PCBs enters the river, perhaps through baseflow and constant desorption/diffusion from the sediments, so that concentration varies only by dilution. This is a simple and appealing paradigm, but the approach reduces the regression problem to one equivalent to determining a mean mass loading rate solely as a function of mean concentration, although treating only a restricted flow range. This method is also likely to result in biased estimates if used to calculate cumulative loading, because the regression approach makes the assumption that concentrations are normally distributed. Loads, however, will not be normally distributed, but will be a function of Q, which will be approximately log-normally distributed. For instance, the distribution of loads under non-scouring flows observed at Stillwater in 1983 is shown in Figure B.4-17. The arithmetic mean of these observations is 1.74 kg/day, whereas the regression approach yields a β value of

1.0 kg/day. In essence, β is estimated as the central point or median of the distribution, but this estimate is not equivalent to the mean in a skewed distribution. Therefore, use of the Schroeder and Barnes model to estimate non-scouring loads will result in an underestimation of total loading over the course of time.

Direct Estimation Approach

The second approach to estimating PCB loading over time is to use the USGS measurements of PCB concentration (C_i) and instantaneous discharge (Q_i) directly. Their product yields a series of estimates of load rate, i.e., $L_i = C_i \times Q_i$. Because PCB levels were not measured continuously, an estimate of yearly load provided by summing the instantaneous loads does not yield an unbiased load estimate. PCB measurements have not been spread randomly throughout the year, but instead are focused in the months of April and August, with the apparent intent of providing good coverage of the high and low flow regimes. For instance, at Stillwater, there are 85 PCB observations in April, but only 3 in January (see Figure B.4-18). Measurements at other stations follow a similar pattern. The problem, as partially recognized by NUS (1984), is that the available monitoring data provide a biased estimate of the load and the bias is toward high flow events. At Schuylerville, the PCB observations are associated with discharges having a mean of 11,136 cfs and a median of 6,422 cfs, whereas the expected mean discharge, prorated from the Fort Edward gauge, is about 4,900 cfs. Because the PCB concentration is highly variable, the mean is likely to be influenced by outlying extreme high values, so that it is unlikely that an accurate estimate can be obtained from a small number of samples. The uneven distribution of measurements during the year will tend to overestimate the true average load rate.

Additional problems arise because the PCB concentration samples are (practically) instantaneous rather than continuous. Thus, the observed concentration response to a flood event is likely to be dependent on whether the observation caught the rising or falling limb of a flood wave. Determination of the mean is also influenced by the presence of non-detects in the concentration

measurements. Various methods can be used to address estimation in the presence of non-detects in a sample, *e.g.* Helsel and Cohn (1988). No straightforward method is available to address the problem of estimating the effect of such non-detects on the estimation of load, which is the product of the weakly correlated variables, concentration and discharge.

Most authors who have attempted to estimate the total yearly PCB load (*e.g.*, Tofflemire, 1980; Brown and Werner, 1985; Barnes, 1987) seem to have used similar methods, based on the observation that PCB concentrations exhibit a bimodal distribution against discharge and begin to increase, on average, beyond a certain critical flow value. The bias inherent in over-sampling high flow episodes can then be corrected by taking yearly averages for *both* scouring and non-scouring flow regimes, then weighting these by the actual rate of occurrence of daily average flows above and below the assumed scour limit during the course of the water year or calendar year.

USGS estimates of PCB mass transport are summarized in Barnes (1987), who analyzed data through water year 1983. As with earlier authors, the transport regime was divided into scouring and non-scouring flows. For non-scouring flows, annual loads were calculated by direct averaging of instantaneous loads calculated from PCB concentration and water discharge data. Barnes found that PCB transport rates at Schuylerville, Stillwater and Waterford through 1983, were approximately equal to one another and greater than those at Fort Edward (Rogers Island). No confidence limits were attached to the estimates nor was it stated how non-detects were treated. For PCB transport during high flow, Barnes notes that PCB yield at Waterford depends on the percent of discharge originating from the sub-basin above Fort Edward. He then presents a graph of annual transport by both scouring and non-scouring flows across Federal Dam. The method of calculating the total annual scouring loads is not documented nor are the exact figures given.

Other, earlier efforts to calculate total PCB loading to the estuary are summarized in NUS (1984), including calendar year estimates by Brown and Werner (1983). None of these estimates appears to include confidence limits or any

detailed statistical analysis of the computation of the scouring flow loads. Results of Tofflemire (1980), Barnes (1987) and Brown and Werner (1983) are summarized in Table B.4-3.

The foregoing methods of PCB load calculation remain highly sensitive to mis-estimation of high flow loads, as a few such events may contribute a major part to the total calculation. One way around this problem is to adopt a robust estimation method, based on the fact that the observed loads, for both scouring and non-scouring regimes, appear to follow lognormal distributions.

The current analysis uses a corrected mean method, which is derived from the median PCB concentration. The median is much more resistant to bias introduced by outliers in a distribution than is the mean. In addition, the data appear to be well described by lognormal distributions. The arithmetic mean for a lognormal distribution is related to the median through:

$$\bar{x} = m_x \exp\left[\frac{\sigma^2_{lnx}}{2}\right]$$

where $\bar{x}$ is the mean, m_x is the median of the untransformed data and σ^2_{lnx} is the variance of the natural logs of the data.

Calculation by this formula yields an alternative estimate of the arithmetic mean, which is relatively insensitive to outliers in the data. The arithmetic variance can also be estimated from the log-space parameters as:

$$\sigma^2_x = \bar{x}^2 \left(\exp[\sigma^2_{lnx}] - 1\right).$$

To implement this method, the data for each year are first divided into scouring and non-scouring flow regimes. These regimes are defined by flow values determined for each monitoring station, rather than by simply determining high flow days by flows at Waterford. The same critical flow values as were selected

for the regression equations earlier in this section are used, except in the case of Waterford where the figure proposed by Schroeder and Barnes (1983) is used, *i.e.*, 21,000 cfs at Waterford. Log-space parameters for each flow regime were calculated and used to determine adjusted arithmetic parameters by the equations given above. These were then weighted by the actual number of scouring and non-scouring flow days observed in a given year to obtain the corrected mean. The variances were also weighted and pooled. Non-detects among the concentration observations are included in the load calculations as one-half the concentration detection limit times the flow.

Annual PCB load estimates using the corrected and uncorrected mean methods are summarized in Table B.4-4. In general, the corrected mean method yields lower estimates of PCB load than use of average annual PCB concentration multiplied by average annual flow, particularly in the earlier years. For instance, in 1983 the uncorrected mean estimates of loads past Fort Edward and Waterford are 4200 and 3900 kg, respectively, whereas the corrected mean estimates are 1700 and 980 kg.

Total mass of PCBs transported per year at all monitoring stations, except for the short run at Fort Miller, is plotted in Figure B.4-19. Figures B.4-20 and B.4-21 provide the 95 percent confidence intervals on the load calculations for Waterford and Fort Edward, respectively. In general, the error bounds are quite large for years in which there were a significant number of scouring flows, due to the high variability of PCB loads in these flows.

A number of interesting inferences can be drawn from the plots of annual PCB loads. For the early years, through 1979, it is clear that there was a substantial gain in PCB load over the length of the Thompson Island Pool, reflected in the differences between loads at Fort Edward and downstream stations. This regime seems to have been altered by the significant flood of April 1979 (34,000 cfs at Rogers Island), which may have removed much of the readily erodible PCB-contaminated sediment in the Thompson Island Pool. For 1980 through 1982 the load gain from Fort Edward downstream is less dramatic, with annual loads at Stillwater (see Figure B.4-21) about twice those at Fort

Edward (an increase in the range of 300 to 800 kg/yr). The spring flood in 1983 (35,200 cfs) was even greater than that of 1979 and PCB loads increased sharply during this year. Loads since 1983 have continued a downward trend, with only a moderate increase shown for the high flow in 1987.

After 1983, there appears to have been little or no gain in annual PCB load between Fort Edward and downstream stations. This finding suggests that, at least for the flows experienced in this period, the Thompson Island Pool has not contributed any significant increase to the PCB load above the load already present upstream at Rogers Island, presumably because most of the easily erodible contaminated sediments were removed by earlier floods. That period, however, was one of lower than average spring floods. The only significant spring flood event from 1984-1989 was that of 1987, which did produce an apparent gain from Fort Edward to Schuylerville and further downstream. Regardless of whether sediment scour has been less during this period, it appears that a significant PCB load is in the river upstream of the hot spot areas (see Figure B.4-19).

For the observations at Waterford, average PCB concentrations are lower, due to dilution, and flows higher than those at stations upstream. Nevertheless, the annual PCB load estimated at Waterford very closely tracks that estimated for Stillwater and Schuylerville (Figure B.4-19). Evidently there is no significant difference in load between Schuylerville (the first station with a long-term record downstream of Thompson Island Pool) and Waterford (the last station before Federal Dam), implying that most PCBs mobilized in the Upper Hudson are transported through to the Lower Hudson. These observations fit with the relative annual water-column PCB concentrations (see B.3). The contributing watershed area at Waterford (4,611 square miles) is 1.6 times that at Fort Edward (2,817 square miles) and 1.3 times that at Schuylerville (3,440 square miles). If the load is simply throughput from Fort Edward past Waterford, then concentrations should decline, by dilution, as the inverse ratio of the contributing area. That is, concentrations at Waterford should be 77 percent of those at Schuylerville and 63 percent of those at Fort Edward. The estimated current (1986-1989) annual (full year) average concentrations (Table B.3-12) imply that concentrations at Waterford are 81 percent of those at Schuylerville and 62

percent of those at Fort Edward, fitting the prediction very closely. It should, however, be noted that the same relationship is not apparent for summer low flows (Table B.3-13).

The calculations of total PCB mass passing Waterford presented here are somewhat higher than those of either Brown and Werner (1983) or Barnes (1987). Their estimates are still well within the confidence limits of the present estimates. It is also possible that the earlier estimates included some consistent underestimation of total mass.

An independent analysis of mass transport rates is provided by the detailed study of PCB transport in the Upper Hudson for Spring-Summer 1983 conducted by Bopp *et al.* (1985), supplemented by collection of data not available from USGS sampling. Bopp *et al.* developed equations to estimate PCB transport, utilizing USGS monitoring of mean daily discharge and mean suspended matter concentrations, together with experimentally determined PCB dissolved/adsorbed distribution coefficients and a measure of PCB component concentration on suspended matter that was determined from ten large-volume water samples, which varied significantly from season to season. The PCB loads estimated are, thus, determined *independently* of the USGS monitoring of total PCB concentrations. Using this method, Bopp *et al.* estimated the total PCB mass transport past Stillwater for March 1 to September 30, 1983 to be 940 kg, of which 840 kg was transported in spring runoff, between April 11 and June 10. The TAMS/Gradient mean estimate for the March to September time period is 830 kg total PCBs transported past the Stillwater station on a total of 21 scouring flow days (>18,000 cfs). This result is in close agreement with the figure given by Bopp. The TAMS/Gradient mean estimate of total transport during this period, is 1,400 kg. The TAMS/Gradient estimate of non-scouring transport is, thus, much higher compared to that of Bopp and yields 570 kg rather than 100 kg. However, the 95 percent confidence limits on the estimate for this period are also very large (±1,000 kg). For the same time period, the mean estimated transport at Schuylerville was only about 800 kg.

One interpretation of PCB mass load estimates presented here is that the Thompson Island Pool has not been a major source of PCB mass in recent years (see Figure B.4-19). The bulk of the PCB mass presently transported in the river is already present in the water column at Rogers Island (above the Thompson Island Pool) and is simply throughput downstream without significant loss. This finding is reflected in the fact that recent PCB concentrations observed in the water column at Fort Edward are higher than in any station downstream of the Thompson Island Pool. Presumably the source of the load passing Rogers Island has been the area beginning adjacent to the Hudson Falls Plant and continuing to Rogers Island, unless there is some as yet undetected contaminated sediment source in the Hudson Falls area. This hypothesis should be investigated through continued monitoring, subsequent to the remediation of the remnant deposits, of PCB concentrations at Fort Edward, Fort Miller and Schuylerville. Additionally, the probability of a future significant flood event remobilizing large quantities of buried contaminated sediments must be evaluated.

B.4.3.3 Discussion of Mass Transport from Upper to Lower River

PCBs in the Upper Hudson enter the Lower Hudson system by passing over the Federal Dam at Troy. The rate of mass transport over the dam is thus a factor in the impact of PCBs in the estuarine system. There are also other sources of PCB contamination in the estuary (see Part A). Assessing the relative importance of these various sources for current PCB contamination in the estuary presents a difficult problem.

It has been estimated that GE capacitor plants discharged between 209,000 and 1.33 million pounds of PCBs to the Hudson River between 1957 and 1975 and an unknown quantity prior to 1957. Much of that total was sorbed to sediments in the pool behind the former Fort Edward Dam and redistributed following the removal of that dam in 1973. Large portions of the contaminated sediment were removed by dredging in 1973-1974, but M. P. Brown *et al.* (1988b) estimated that approximately 23,000 kilograms (51,000 pounds) of PCBs remain in the Thompson Island Pool sediments, whereas the Draft Supplement to the Final EIS (USEPA, 1986) estimated that PCB mass in the lower reaches of the Upper Hudson were on

the order of 50,000 kg (110,000 pounds) and that the remnant deposits contained on the order of 21,000 kg (46,000 pounds).

The PCBs presently stored in the sediments of the Upper Hudson, dredge spoil and containment sites constitute only a fraction of the total releases. Unknown portions of the total mass have been lost by volatilization to the atmosphere. Tofflemire and Quinn (1979) estimated volatilization losses of about 600 kg/yr total PCBs from the Upper Hudson. Most of the balance of the mass released into the Upper Hudson has been transported across the Federal Dam and into the Lower Hudson estuary and, ultimately, the Atlantic Ocean.

Based on the PCB loads past Waterford, as developed previously, one can estimate that approximately 15,000 kilograms of PCBs were transported from the Upper to the Lower Hudson between 1977 and 1989. This estimate is obtained by summing the annual load obtained by the corrected mean method; it is somewhat higher than would be calculated using the annual loads determined by Barnes (1987). The 95 percent confidence limits on the sum are approximately ±3,000 kg (*i.e*, 12,000 to 18,000 kg total). Of this total, about 13,000 kg are estimated to have been transported between calendar years 1977 and 1983. There has been a steady decline in the rate of loading, reflecting both the decline in water column PCB concentrations and the generally lower flows of the mid 1980s. Figure B.4-22 shows the estimated annual loads past Waterford (calendar year basis). Although there is year to year variability, the loads appear to have declined exponentially, with a half-life of about three years.

For the period before 1977 there is no water column monitoring of PCBs available. A record of the sediment transport history is, however, preserved in undisturbed sediment cores. Extensive work on the sediment record of contamination in the Hudson, particularly the Lower Hudson (see Part A), have been carried out by the Lamont Doherty Geological Observatory (summarized in Bopp and Simpson, 1989).

Study of dated sediment cores from the Lower Hudson above the salt wedge shows a maximum total PCB concentration in suspended sediment in the early to mid 1970s. This occurrence reflects the large volume of contaminated sediment transported downstream following the removal of the Fort Edward Dam in 1973; in several cores the peak concentration appears to be at or very near 1973. A core collected at River Mile 88.6 (Bopp and Simpson, 1989) exhibits an exponential decrease in sediment concentration by year since 1973 with a half-life of about 3.5 years, which is very similar to the three-year half-life decline in mass loads past Waterford. The core data do not directly yield historic water column PCB concentrations, but do provide information on relative concentration from year to year. The cores suggest that peak concentrations *circa* 1973 may have been about five times those experienced around 1980, *i.e.* on the order of 5,000 kg/yr. Prior to the peak, the cores show a gradual, consistent increase in PCB deposition rates from about 1950 on. Core concentrations dated around 1960 appear to be similar to those found *circa* 1980, suggesting that in this period the mass flux of total PCBs from the Upper Hudson to the estuary was on the order of 500 to 1,000 kg/year.

Another analysis of the yearly loading past Waterford is contained in a modeling study of PCBs in the Hudson River estuary (Thomann *et al.*, 1989). This study used sediment data to estimate loads through 1975 and USGS monitoring at Waterford to estimate loads for 1976-87. The results they obtained from analysis of the USGS data differ from those presented in the previous section and show significantly higher transport rates than estimated here for the period before 1984. For the year 1980, Thomann *et al.* estimate average daily loading across the Federal Dam, based on Waterford monitoring, as 3.8 kg/day, which is 90 percent higher than this report's estimate of 2.0 kg/day. Thomann *et al.* estimate about 20.3 kg/day PCB transport in 1977, while this report's estimate is 11.2 kg/day. The primary reason for the difference is that Thomann *et al.* did not attempt to correct for the fact that the USGS monitoring is intentionally biased toward higher flow (and possibly higher PCB concentration) events, while the winter months are under-represented. Thomann *et al.* (1989) do not state how non-detects in the data were handled.

The loads estimated by Thomann *et al.* are very close to the *uncorrected* mean loads presented for Waterford, *e.g.*, 4.1 kg/day for 1980, 21.7 kg/day for 1977. By neglecting the bias in the data, Thomann *et al.* have likely overestimated the PCB loading from the Upper to the Lower Hudson. Compared to the TAMS/Gradient estimate that ~13,000 kg of PCBs were transported past Waterford from 1977-83, Thomann *et al.* (1989) estimated total loading across the Federal Dam for the same period to be ~19,000 kg, or about 35 percent more.

Thomann *et al.* (1989) also used sediment data to attempt to estimate PCB mass transport prior to 1976. Their estimates are on the order of 1,000 kg/year around 1960-1965, but then indicate a very sharp increase to a peak of 24,600 kg/yr in 1973. This peak is nearly five times that suggested by the Lower Hudson cores. The calculation was based on analysis of several Upper Hudson sediment cores reported by Hetling *et al.* (1978). Approximate, dated annual PCB concentrations were established in these cores, which were taken to represent the PCB concentrations in suspended sediment during the same year. The dating seems to have been based on the assumption of a constant sedimentation rate, which is probably unrealistic in terms of the historic channel destabilization and sediment mobilization following the removal of the Fort Edward Dam.

For the analysis of Thomann *et al.*, annual sediment PCB concentrations from the cores in river reaches 1-5 (see Plate B.1-2) were prorated to the expected concentrations at the exit of Reach 1. The daily PCB load was then calculated as:

$$W_T = 5.391 \times 10^{-6} \frac{r_2 Q m_1}{f_{p1}}$$

in which W_T is the total PCB discharge (lb/day), Q is the annual average flow over the Federal Dam (cfs), m_1 is the estimated solids concentration in the water column at Waterford, r_2 is the sediment PCB concentration from the core data, and f_{p1} is the fraction of total PCB associated with the solids (Thomann *et al.*, 1989). Estimation via this equation is obviously highly sensitive to the value of f_{p1}.

Thomann *et al.* use a value of f_{p1}=0.637, based on a partition coefficient of 20,500 *l*/kg. The dissolved portion of the mass was therefore taken to represent a constant, relatively large fraction of 36 percent. During some periods of high flood scour, however, the bulk of PCB mass transported is sorbed to particulate matter, without being in equilibrium with the water column (Bopp *et al.*, 1985). In addition, the major portion of the total PCB mass transported in a given year may be accounted for by scouring transport on a few high flow days. Thus, use of the equilibrium partition coefficient means that the dissolved portion and, thereby, the total mass of PCBs transported may be substantially overestimated. This overestimate would result particularly for the period immediately after 1973, when large volumes of sediment from the former Fort Edward Dam pool were moved by spring floods. The calculations of Thomann *et al.* further assume a single suspended sediment concentration value of 85.6 mg/*l* (undocumented), whereas USGS observations for 1975-1989 show a median sediment concentration of 12 mg/l and an average of 63.3 mg/l at Waterford. Taken together these two assumptions could result in overestimation of PCB load by a factor of around two. Finally, the adjustment factor normalizing Hetling's (1978) data to the exit of Reach 1 is also highly uncertain.

A more reliable estimate of pre-1976 loading over the Federal Dam is provided by examining the radionuclide-dated core samples from below the dam and comparing these to later monitoring.

B.4.4 Analysis of PCBs in Fish

Plots of concentration versus time for fish in the Upper Hudson (see B.3) indicate that PCB levels in all fish species appear to have declined in recent years. This trend was discussed in Sloan *et al.* (1983), Sloan *et al.* (1984) and M. P. Brown *et al.* (1985), who noted that the levels of all Aroclors in fish in the Upper Hudson have tended to decline since about 1980. Data from recent years suggest that this trend alone, one of apparent exponential decay, will not be sufficient to reduce PCB burdens in many fish species to acceptable levels in a reasonable period of time.

There have not been any major flood erosion events since 1976; the effect of such an event in the future on fish PCB levels remains to be assessed.

In monitoring since 1975, peak PCB levels in most species of fish in the Upper Hudson seem to have occurred in 1977-78. The plots of Aroclor concentrations versus time show that the ratio of Aroclor 1016 to Aroclor 1254 in fish was also elevated in this period, typically to around five versus an earlier ratio near one. Presumably this result represents a significant input from buried, dechlorinated sediments to the water column during this period or the extensive use of Aroclor 1016 by General Electric from 1972 to 1976. Since 1980, however, 1016/1254 ratios have been around one and slowly declining in most fish species. This result appears to represent a gradual depletion of lower congeners from the aquatic system, resulting from the absence of significant new input from the deep sediments and/or the mixing of older and newer sediments, resulting in the return to the dominant, pre-1972 PCB mixture in the riverine system. No similar pattern was found for Aroclor 1221 relative to 1016 or 1254; levels of this Aroclor (representing mono and dichloro congeners) were consistently low.

B.4.4.1 Evaluation of Time Trends

Non-Parametric Trend Test

Many tests used to examine the significance of trends exhibited in time-series data, such as the fish data, rely on assumptions regarding the underlying probability distribution (*e.g.*, normal, lognormal, *etc.*) of the data. Thus, the results of such tests depend on the validity of the basic assumption that the data adequately follow the probability distribution selected. Non-parametric tests offer an alternate means of examining the significance of time-series trends without requiring that the data follow any particular probability distribution.

To test whether the apparent decline in Aroclor concentrations in fish is significant, the non-parametric Mann-Kendall test was used on the medians of the yearly data (Gilbert, 1987). This is the same test used previously in evaluating the time-series trends for water-column PCB levels.

The entire time series of Aroclor 1254 and Aroclor 1016 measurements at River Mile 175 for largemouth bass, brown bullhead, pumpkinseed and goldfish (Table B.4-5) was examined. Although the negative values obtained from the Mann Kendall statistics confirm the visual impression of a general downward trend for all concentrations, the trends are not significant at the 95 percent confidence level for either Aroclor 1254 or 1016 in brown bullhead, nor for Aroclor 1254 in largemouth bass nor Aroclor 1016 in pumpkinseed. This result does not mean that real declines are not occurring in these cases, but indicates that the trend cannot be distinguished at the 95 percent confidence level from random variability in the data without making further assumptions regarding the probability distribution of the data.

Apparent Aroclor Half-Lives in the Fish Population

The fact that the Aroclor 1016/1254 ratio has declined fairly continuously since about 1980 in most species suggests the absence of any major resuspension of buried, lightly chlorinated or dechlorinated sediments since that time. The 1980-1988 data are, thus, appropriate for estimating the rate of removal of the steady state, Aroclor 1254-like PCB components from the system. These components are taken to represent the more persistent fraction of the PCBs. Estimates of removal rate or half-life depend on multiple factors, many or most of which may be unknown or unquantified. This empirical determination is limited to the range of conditions found in the river over the course of the monitoring. In particular, there is a strong connection between average summer PCB concentrations in water and concentrations in fish. PCB concentration in water, in turn, may depend on discharge and the 1980s were generally notable for low flows. Thus, the rate of decay of Aroclor concentrations observed in recent fish data may, in part, reflect only the temporary variations in climate experienced during this period.

Just as an exponential decline and half-life were estimated for PCB mass transport in the previous section, an exponential decay pattern tends to hold for Aroclor concentrations in Hudson River fish. In chemical data characterized by exponential decay patterns, a plot of the log of Aroclor concentration versus time is linear:

$$LN(Aroclor) = \alpha - (\beta \cdot t)$$

where α is a regression constant, β is the linear rate of change in the LN of Aroclor concentration relative to t (1/time) and t is time (years). The half-life ($t_{1/2}$) of the compound can be calculated as LN(2)/β. For 1980-1988 fish data, the Aroclor 1254 decay rate factors and half-life values at River Mile 175 are shown below.

Aroclor 1254 Half-Lives in Upper Hudson Fish (River Mile 175)

Species	Time Constant, β (1/years)	Half-Life, $t_{1/2}$ (years)
Largemouth Bass	0.046	15.1
Pumpkinseed	0.095	7.3
Brown Bullhead	0.017	40.7

When PCB levels are based on lipid content, the half-lives for Aroclor 1254 do not change much for the 1980-88 data, yielding 11.7 years for largemouth bass, 9.4 years for pumpkinseed and 37.3 years for brown bullhead. The slope of the regression for largemouth bass, however, is entirely the result of the relatively high 1980 measurements. For 1981-1988 there has actually been a slightly increasing trend in lipid-based Aroclor 1254 levels in largemouth bass.

The half-life values calculated above can, in theory, be used to estimate the length of time it will take to reduce levels of the more resistent higher chlorinated congeners in fish populations, assuming the continuation of recent hydrological patterns and lack of additional major input from deep sediments. For example, the most recent (1988) observed Aroclor concentrations in fish (wet weight, not lipid-based) are shown below.

Mean Aroclor 1254 Concentrations (ppm), 1988

Species	RM 190 (Fort Edward)	RM 175 (Stillwater)	RM 153 (Federal Dam)
Brown Bullhead	6.9 [6.3]	4.1 [3.8]	1.2 [1.2]
Goldfish	33.1 [30.9]		
Pumpkinseed	4.2 [3.9]	2.2 [2.1]	
Largemouth Bass	5.3 [3.3]	1.9 [1.3]	2.8 [1.6]

Values in [ ] are geometric mean values.

Based on the half-life estimates, it is expected to take 70 years to reduce the Aroclor 1254 component in brown bullhead to below 2 ppm at River Mile 190. On the other hand, largemouth bass might reach such levels of Aroclor 1254 in about 25 years.

Aroclor 1016 levels in fish have shown a steady decline since 1980. Figure B.4-23 shows the exponential regression line for Aroclor 1016 decay in largemouth bass (lipid-based). Calculated Aroclor 1016 half-lives at River Mile 175 were rather similar for different species, yielding 3.5 years for largemouth bass, 4.5 years for pumpkinseed and 3.8 years for brown bullhead.

This discussion of half-life calculations is for River Mile 175 only and depends on whether or not lipid-based values are used. A fuller utilization of the data, but still not considering variations in hydrology, may be accomplished by considering the following multiple regression relating measured Aroclor content to these variables: year of measurement (Y), river mile (M), percent lipid (LP) and sample weight in grams (W):

$$LN(\text{Aroclor}) = \alpha + \beta_1 Y + \beta_2 M + \beta_3 (LP) + \beta_4 (W).$$

Fish sample data from 1980-1988 taken at Troy and Fort Edward for largemouth bass, pumpkinseed and brown bullhead were used in these multiple regressions. As a weighted average, the apparent half-life for Aroclor 1254

components in all fish is 9.3 years, while the apparent half-life for Aroclor 1016 components is 4.1 years. The R^2 values of these regressions are fairly low, ranging from 28 percent to 60 percent. Better performing models (R^2=80 to 84 percent) can be constructed by considering total PCBs, rather than individual Aroclors.

B.4.4.2 Projected PCB Concentrations in Fish

A 30-year projected average PCB concentration in fish is necessary for the evaluation of baseline health risks. This average is calculated through a regression approach similar to that just described above. There are extensive data sets of PCB concentrations in largemouth bass, brown bullhead and pumpkinseed in the Upper Hudson. The pumpkinseed data are not very useful for assessing average concentrations in terms of potential human consumption, since an effort was made to gather in September primarily yearling pumpkinseed, which are smaller than commonly edible size. PCB concentrations in largemouth bass and the bullhead data were, therefore, averaged to obtain a representative value for human exposure.

The multiple regression here is based on the natural log of the *sum* of Aroclors 1016, 1221 and 1254 taken to represent total PCBs. The log of percent lipid, rather than percent lipid, provided a better fit in these regressions. Regression equations used Aroclor measurements for brown bullhead and largemouth bass sampled for River Miles 153 to 190. The best fit equations are:

$$LN(PCB_{L.Bass}) = 12.45 + 0.031\,M - 0.188\,Y + 0.997\,P + 0.0003\,W$$

R^2 = 86 percent
SE = 0.553

$$LN(PCB_{B.Bullhead}) = 6.4 + 0.069\,M - 0.200\,Y + 0.894\,P + 0.00045\,W$$

R^2 = 81 percent

SE = 0.520

where

M = River Mile;
Y = Years since arbitrary datum of 1900 (Year - 1900);
P = Natural log of percent lipid content of sample;
W = Weight of fish (grams); and
SE = Standard error of regression estimate.

Using these equations, the 30-year projected median PCB concentrations in these species at River Mile 175 are 0.19 ppm for largemouth bass and 0.16 ppm for brown bullhead.[1] The average PCB concentration over the entire 30-year period will be greater. Because the regressions use lognormal variables, a straightforward expression for this average cannot be developed. Thus, a simple Monte Carlo sampling procedure was employed to provide an estimate of this 30-year average.

Steps in the Monte Carlo procedure are outlined below.

- Incrementally select years (*e.g.* 30 years) ranging from 1991 to 2020.
- Uniformly sample fish between Federal Dam and Fort Edward.
- Select a random sample for percent lipid (P) and weight (W) assuming they are lognormally distributed, *i.e.* LN(*P*) and LN(*W*) are normally distributed with the following parameters:

LN(*P*) mean	s_x	LN(*W*) mean	s_x
-0.2925	1.1597	6.2666	0.6186 (bass)
0.7918	0.9575	5.8370	0.4269 (bullhead).

- Calculate the annual average PCB levels in the fish using the samples of *P* and *W* in the above regression equations, which also involves adding a random error term, which is modeled using the standard error (SE) of the regression equations.

[1]Values in the equations are P=-0.29, W=608.867 (bass); P=0.681, W=373.754 (bullhead); years since 1900 Y=105 and M=175.

- Repeat the calculation a large number of times (500 here) and calculate the overall 30-year mean PCB estimate for brown bullhead and largemouth bass.

The frequency distribution of the annual average PCB concentration after 500 simulations (300,000 fish realizations) is shown in Figure B.4-24. Note that over a 30-year time horizon the distribution remains skewed and is best described as lognormal. The simulated average of largemouth bass and brown bullhead has a grand mean (mean of the means) of 1.23 ppm and a standard deviation of 0.148. The upper 95 percent confidence limit on the grand mean is 1.55 ppm. Statistics on the 30-year average ($\bar{x}$) for individual species follow:

Largemouth Bass: $\bar{x} = 0.99$; $s_{\bar{x}} = 0.18$; $\bar{x}_{.95}$ (upper 95 percent confidence limit) = 1.38 ppm.
Brown Bullhead: $\bar{x} = 1.47$; $s_{\bar{x}} = 0.24$; $\bar{x}_{.95}$ (upper 95 percent confidence limit) = 2.00 ppm.

B.4.4.3 Relation Between PCB Concentrations in Fish and Water

An important determinant of PCB concentrations in fish, although not necessarily the only one, is likely to be ambient water concentrations. In particular, ambient PCB concentrations in water during the summer low flow season (June-September), which is also the period of maximum biological production, are likely to provide a good indicator.

M. P. Brown *et al.* (1985) provided what appeared to be a nearly exactly linear relationship between 1977-1983 mean summer water column PCB concentration and median lipid-based total PCB concentrations in yearling pumpkinseed collected annually in September at River Mile 175. Total PCBs were calculated by adding concentrations calculated for individual Aroclor mixtures and the median was employed as a robust indicator of central tendency. (Mean and median are very close to one another in this data set.) Only yearling pumpkinseed were included, which necessitated the exclusion of a few older individuals collected in 1981-1983.

Figure B.4-25 provides a plot similar to that of M. P. Brown *et al.* (1985) extended to include the 1984-1988 data. Data for fish in 1987 and data for summer PCB concentrations in water for 1986 could not be included as they do not exist. Lipid-based total PCB concentrations in yearling pumpkinseed at River Mile 175 are plotted against summer PCB concentrations in water at River Mile 168. The general linearity of the relationship is maintained, although the 1984 and 1985 data points do not lie directly on the earlier line. The apparent strength of the relationship is largely due to the 1979 and 1980 observations, when summer PCB concentrations in the water column were high. Water column PCB concentrations have been much lower in subsequent years and it is not necessarily clear that the relationship in the lower part of the scale is linear.

Consideration of individual Aroclor measurements in pumpkinseed (Figure B.4-26) suggests that both Aroclor 1016 and Aroclor 1254 in the fish are linearly related to total PCB concentrations measured in the summer water column at River Mile 168, although with differing slopes. Measurements of Aroclor 1254 in particular look as though there is an asymptotic leveling off in the lower part of the water concentration range. This observation may indicate a non-linear bioaccumulation effect. As water column concentrations of PCBs become very low, the body burden of PCBs in fish may begin to decline at a slower rate, perhaps, as a consequence of some direct food pathway from sediment to fish.

The plots in Figures B.4-25 and B.4-26 have compared the PCB concentrations in pumpkinseed against concentrations in the water column at River Mile 168, downstream from River Mile 175 where the pumpkinseed were collected. The data can also be compared to PCB concentrations in the water column at Schuylerville, upstream from River Mile 175, with rather similar results (Figure B.4-27). Use of the Schuylerville data allows a point for 1986 to be plotted.

Yearling pumpkinseed may be expected to provide the best relationship among the fish monitored to ambient PCB concentrations in the water column as they are sedentary, relatively low on the trophic chain and all the same year class. Total lipid-based PCBs in all largemouth bass sampled at River Mile 175 bear a

rather similar relationship to water column concentrations as shown in Figure B.4-28. A single sample from 1979 is omitted here as possibly anomalous. There appears to be even more indication of an asymptotic leveling off of body burden in the lower part of the water column concentration range.

Total lipid-based PCBs in brown bullhead collected at River Mile 175 also exhibit a strong linear relationship to summer water concentration as shown in Figure B.4-29, except for two points that lie outside the 95 percent confidence limits.

Given the apparent linear relationship, simple regressions can be developed to indicate dependence of fish PCB burden on summer water concentration, in the form:

$$PCB = \alpha + \beta S$$

where *PCB* is the median lipid-based total PCB concentration (ppm) at River Mile 175 for a given species in a given year, and S is the average summer water concentration (μg/*l*) at River Mile 168 for the same year. The results for pumpkinseed, largemouth bass and brown bullhead are shown below.

Relationship of Lipid-Based PCBs in Fish and Summer PCB Concentrations in Water

Species	α	β	R^2 (%)	SE	p value
Pumpkinseed	131.0 (p=.004)	1602.8 (p=.00001)	97.1	53.1	.00001
Largemouth Bass	-76.50 (p=.722)	6804.3 (p=.00002)	96.0	381.2	.00002
Brown Bullhead	134.45 (p=.431)	2407.6 (p=.0015)	83.6	296.6	.0015

The α coefficients (a constant) for the largemouth bass and brown bullhead are not statistically significant at the 95 percent confidence interval, which would be consistent with fish PCB concentrations going to zero as summer water concentrations go to zero. The β coefficients, all highly significant, differ from one another by up to more than a factor of 3. These coefficients represent the species lipid-based bioaccumulation factor (BAF) for total PCBs relative to

water column PCB concentration. (Note that BAF = $\beta \times 1{,}000$, since PCBs in fish are in ppm and water concentrations are in ppb.)

An interesting question is whether these BAFs, calculated at River Mile 175, also apply at other locations. If PCB concentrations in fish respond both to PCB concentrations in water and sediment, perhaps via a benthic food chain pathway, the answer might well be yes. The data are not sufficient to resolve this question. Despite recent data for several species of fish in the Thompson Island Pool (River Mile 190), there is no water column PCB monitoring at the same location. The fish data from River Mile 190 are all from 1984 or later, by which time PCB concentrations in fish, in general, had declined. If largemouth bass data at River Mile 190 (1984, 1985, 1986 and 1988) are plotted against downstream PCB concentrations in the water column at Stillwater, all the data points lie slightly above the regression line. Similar results are found for limited samples of other species. As noted earlier, water column PCB concentrations increase upstream. Thus, it is not clear if the relationship at River Mile 190 is any different from that observed at River Mile 175, particularly as no observations are available from the high PCB concentration periods before 1980.

Except for the data at River Mile 175, there are no other significantly long runs of observation (>5 years) on a single species at a single location in the Upper Hudson. There is a long record (1977-1988) for samples of brown bullhead at River Mile 153 just below the Federal Dam. Here the limited PCB samples from the water column were all non-detects at the 0.1 µg/l level, so the relationship between fish PCB burden and water concentration can not be directly determined.

B.4.5 Summary

Summarized here are the questions posed at the beginning of Section B.4 and the information currently developed to help answer them.

(1) What is the potential for migration and redeposition of PCBs in sediments?

- Flood frequency analysis results have yielded improved estimates of the high flow recurrence intervals, which may govern sediment scour and mass redistribution of PCBs in sediments.

- Sediment scour appears to be influenced by minimum scouring flows necessary to suspend appreciable sediment loads. Minimum scouring flows are approximately >10,000 - 20,000 cfs.

- Few large spring floods have occurred in the 1980s, but the high flows in spring 1983 and somewhat lower high flow in spring 1987 both resulted in somewhat increased PCB loads. The potential for resuspension of significant amounts of PCBs during a future major flood remains an important unanswered question.

(2) What is the relationship between PCBs in sediments and PCBs in the water column?

- Multiple regression analyses suggest PCBs are directly related to flow at high flows and inversely related to flow at low flows, although the relationships do not have very great *predictive* strength.

- It is unclear from available data under what flow conditions PCBs in the water column are predominantly dissolved or adsorbed to suspended sediment. It appears, however, that the major portion of the annual PCB transport occurs during the highest flows, a finding that suggests that PCBs attached to sediments are a major component of the PCB load.

- The TAMS/Gradient estimate of PCB mass transport from the Upper to Lower Hudson from 1977-1989 is approximately 15,000 kg. Estimates prepared by previous investigators of PCB stored in the Upper Hudson sediments total approximately 94,000 kg: 23,000 kg in the Thompson Island Pool; 50,000 kg in the other reaches of the Upper Hudson; and 21,000 kg in the remnant areas.

- Empirical time trends suggest that the PCB loads, transported in the water column, have diminished since the late 1970s, with a half-life of approximately three years.

(3) What is the effect of (1) and (2) on the levels of PCBs in fish?

- PCB levels in fish have declined from the high levels in the late 1970s; however, the rate of decline has been very low in recent years.

- A strong linear relationship between PCBs in the water column and PCBs in fish holds for the range of PCB concentrations observed in both water and fish. Field BAF values relating lipid-based PCB levels in fish to levels in water range from 1.6 million to 6.8 million.

- Assuming current trends, the projected 30-year average (1991 - 2020) PCB concentrations in fish is on the order of 1.5 ppm. This projected average is a "best-case" estimate in that it assumes no major resuspension of PCBs in sediments.

- Lack of sufficient paired fish and sediment samples (in time and space) precludes an analysis of the correlation between PCBs in sediments and PCBs in fish.

The long record of PCB measurements in fish and surface water provide an extensive and reliable database, with measurements as recent as 1988 and 1989, respectively. Although sediments have been sampled by GE as recently as 1990, the monitoring record for sediments lacks the extended coverage in both time and space of the water and fish samples. Thus, the sediment data provide a somewhat less reliable database from which to extrapolate trends and relationships with PCBs in water and fish. Other media, including other aquatic biota, air and upland plants, have not been studied as extensively as fish, surface water and sediments.

SYNOPSIS

SEDIMENT TRANSPORT MODELING

(Section B.5)

Development and calibration of hydraulic and sediment transport models have been initiated. Because PCBs in the river are bound primarily to sediments, scour of sediments is a crucial mechanism to the movement of PCBs. A mathematical model provides one tool to predict potential scour and redeposition of sediments containing PCBs. A basic modeling framework has been developed contemporaneously with the data analysis in order to determine what type of modeling may later be appropriate and feasible. The current modeling effort is limited to implementation and calibration of a hydraulic model of the Thompson Island Pool and the preliminary development of a sediment transport model for this reach of the Upper Hudson.

The WASP4 family of models (B.5.1) is selected for use, because of its flexible format and data handling capabilities. Although it can accept sediment transport information, it does not have a sediment scour or transport routine. Thus, one objective is to develop sediment transport routines that can work in conjunction with WASP4.

Previous attempts at sediment transport modeling in the Thompson Island Pool undertaken with the HEC-6 model (B.5.2) are reviewed. There are difficulties, such as the presence of cohesive organic sediment, that make modeling sediment transport in the Upper Hudson difficult. This difficulty and problems in calibrating hydrodynamic models have resulted in a low degree of success for previous efforts by other researchers.

DYNHYD5, the hydrodynamic module of WASP4, is described (B.5.3). A detailed two-dimensional link-node mathematical representation of the Thompson Island Pool is developed using a Geographic Information System. The model is calibrated to river stage data recorded in the barge canal at both the upper and lower ends of Thompson Island Pool. Comparison of results predicted by the model with measured stage observations yielded a fit to within approximately one-tenth of a meter for several flood episodes.

A sediment transport model, STREAM, is described (B.5.4). This model is used to simulate stream bed and streambank erosion, sediment deposition and resuspension and sediment transport. The potential application of this model to the Upper Hudson is still in development. Although preliminary results indicate an improvement in prediction over previous HEC-6 modeling, calibration of this model has not been completed.

Continued modeling efforts will depend on obtaining additional data to define current sediment bed deposits and grain-size distribution and to measure suspended sediment at both upstream and downstream sections of the Thompson Island Pool.

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B.5 Sediment Transport Modeling

B.5.1 Overview

PCB loads in surface water are highest during high river flows, although the magnitude of response appears to have diminished in recent years. It is currently uncertain if this apparent declining response of PCBs in the water column is a result of the generally low flows of the 1980s and/or if higher than normal and flood flows could cause significant erosion, downstream transport, and increased PCB concentrations in the water column and fish. Potential sediment scour, resuspension, and redeposition, in response to high flows or floods must be evaluated in order to extrapolate from current conditions (*e.g.*, PCB levels in sediment, water fish) and assess likely future trends. Furthermore, PCB dissolution and dispersion as a consequence of possible maintenance or remedial dredging actions must be evaluated. Both of these evaluations are facilitated by hydraulic and sediment transport models, which can be operated to test the response of sediments to flood flows and evaluate changes in dissolved and adsorbed PCB concentrations in the water column.

In contrast to mathematical transport models, evaluation of historical monitoring data provides information on the interrelationships of PCBs in sediment, water and biota (fish) based on the *observed data*. Thus, the relationship between flow and sediment discharge suggests a possible threshold scouring flow phenomena. Other examples of empirical relationships supported by an evaluation of the data are: regression and time-trend relationships among flow, suspended sediment concentration and PCBs; or correlations between PCB levels in fish and those in the water column. The data record may be too short or data from all media insufficient, however, to prove accurate as a predictive indication of future trends for conditions other than those covered by the historical record. In Phase 1, available flow, suspended sediment and water column PCB data have been evaluated to assess completeness and analyze statistical trends. In addition, some initial hydraulic and sediment transport model testing have been performed to assess whether detailed modeling is feasible and appropriate.

The ability of transport models to provide useful results depends on a number of factors. Sufficient data must be available to calibrate the models and determine whether the models adequately mimic observed flows, sediment transport, and PCB concentrations in the water column. The model must be sufficiently detailed to capture the important physical processes influencing flow and sediment/PCB transport. The models should be tested and supported, yet flexible enough to allow modifications for site-specific conditions.

The WASP4 family of models was chosen for Upper Hudson hydraulic and PCB transport models. This modeling package is widely used and supported by the USEPA Center for Exposure Assessment Modeling (CEAM) in Athens, GA. It provides a flexible framework for incorporating modified submodel components as necessary. As currently distributed by CEAM, the WASP4 package consists of independent, but fully compatible components DYNHYD and WASP. DYNHYD is a link-node hydrodynamic model that provides input to WASP. WASP is the water quality module and can be compiled by substitution of submodels as EUTRO, a eutrophication and conventional pollutant model, or TOXI, a toxic pollutant model. In addition, the package also contains a Beta test version of FCHAIN, a food chain model.

The WASP4 package is based on a hydrodynamic model of sufficient complexity to provide useful sediment transport parameters, yet is straightforward enough to be implemented and calibrated with available data. Use of DYNHYD to model two dimensional hydrodynamic flows provides the capability of emulating hydraulics of at least trapezoidal (non-rectangular) reaches.

Although WASP4 is designed to accept sediment transport information, the CEAM package notably lacks any sophisticated sediment scour/transport routines. Thus, an important facet of modeling efforts is to develop sediment transport routines, appropriate to conditions in the Upper Hudson, which can take the hydrodynamic model DYNHYD as input and, in turn, provide input to the water quality model WASP. Summarized here are hydraulic and sediment transport model development and calibration results, as developed in Phase 1.

B.5.2 Previous Modeling Studies

Attempts to model sediment and PCB transport from the Thompson Island Pool using the HEC-6 computer model have been made by Lawler, Matusky & Skeller (LMS) in 1978 and 1979 and Zimmie in 1985. LMS (1979) modeled sediment transport from Fort Edward into the Lower Hudson and modeled reaches from Lock 7, just below Rogers Island, to the Federal Dam. Zimmie (1985) applied the same model to the Thompson Island Pool area, following the extensive resurvey of sediments there in 1984.

Both efforts were based on the HEC-6 model (US Army Corps of Engineers, 1977) and both are subject to limitations.

1. The HEC-6 model is designed to address transport of cohesionless sediments, and cannot explicitly model cohesive organic sediments. Such sediments may play an important role in Hudson River PCB transport.

2. HEC-6 is a one-dimensional sediment model and, thus, cannot replicate the lateral variability in sediment composition in the Thompson Island Pool. This imposes limitations on the ability to model flood effects on specific hot spots.

3. It is unclear whether the calibration data are sufficient to accurately fit a model of sediment transport.

4. As discussed at B.4, Zimmie (1985) appears to have overestimated the magnitude of flood flows and resulting probability of sediment scour.

The purpose of the LMS study was to predict PCB transport downstream under no action and mitigated (dredged) conditions. The focus was on expected impacts of average rather than extreme conditions. Thus, hydraulic simulation was undertaken over observed flows of the 20-year period 1958-1977, rather than generating specific flood recurrence events. The conclusion of LMS (1979) was that the hot spot dredging project would reduce the time needed to deplete the input of significant amounts of PCBs to the estuary from 69 to 44 years.

A detailed critique of the LMS application of the HEC-6 model is provided in the Feasibility Study (NUS, 1984). The hydraulic submodel (HEC-2) was considered adequate for the application; similarly, the one dimensional nature of the model would not have introduced significant difficulties, given the relative length of reaches and heights of dams between Fort Edward and Troy. Significant problems were, however, noted in the calibration of the hydraulic submodel. For the Thompson Island Pool, the only calibration data used were apparently a limited set of 1976-77 observations at Lock 7, which is the *up*stream end of the reach. Because the grade over the reach is quite low, elevations at Lock 7 should be closely controlled by the downstream boundary at the Thompson Island Dam, although calibration data at both the upstream and downstream ends of the reach would of course be preferable. The NUS study notes that the predicted model water surface elevations at Lock 7 consistently exceeded observations and, indeed, "the surface elevation within the drawdown curve at the dam already exceeds the observed elevation at the upstream end of the reach." An error in specification of the rating curve at the dam was thought to be the cause. The result is that flow rates corresponding to a given water surface elevation are overestimated by about 30 percent, which would imply that the average velocities driving the sediment model are overestimated.

More severe problems appear to apply to the sediment transport portion of the LMS modeling effort. Upstream sediment loading was simply calculated by a regression relation at Glens Falls. The first sediment data available for calibration were at Lock 4, Stillwater. Model predictions here consistently underestimated measured sediment concentrations by about a factor of two for high flows and up to an order of magnitude for low flows. Further, the HEC-6 application predicted sediment deposition throughout the reaches between Lock 7 and Lock 4 at all flows below the one percent exceedance value. NUS (1984) demonstrated that significantly better performance was achieved by simply taking the Glens Falls sediment loads and routing it through to Lock 4 with no scour or deposition. Thus, the sediment transport predictions of the model appear to be unreliable and, in turn, cast doubt on PCB transport predictions.

Among the reasons for poor performance of the sediment transport model may be a lack of adequate calibration data. The problem may also have been compounded by assigning the entire silt component to the coarsest model category (0.032 to 0.062 mm). More generally, the inability of the model to reproduce observed sediment behavior may be a result of neglecting the role of cohesive organic sediments. In a recent, detailed study of a river in Wisconsin, which possesses extensive PCB contamination together with input from pulp and paper industries, Gailani *et al.* (1991) demonstrate that both scour and deposition of PCB-contaminated organic sediments must be modeled as time-dependent processes. The rate of settling is dependent on the state of flocculation of the cohesive sediments, a time-dependent process, while resuspension is controlled by the state of compaction, which also increases over time. The critical shear stress for resuspension of these sediments was found to vary by an order of magnitude with time, since deposition. Together these factors create a situation in which sediments, once suspended, have a tendency to keep on moving, with rates of deposition less than would be predicted from a model that does not treat cohesive sediments.

Zimmie (1985) also used the HEC-6 model to analyze the Thompson Island Pool. Using approximately 50,000 depth soundings of the Thompson Island Pool obtained by Raytheon in 1982 and a computer program to convert these soundings to average cross-sectional profiles, Zimmie defined 31 reaches, from 634 to 1,267 feet in length, between Lock 7 and the Thompson Island Dam. The cross-sectional average geometry of the Thompson Island Pool was thus very fine, although still one-dimensional. In addition, Zimmie used detailed sediment size data for each model segment. Zimmie's results indicated that water discharges of less than 32,000 cfs would not cause bed changes of more than 0.14 feet in any modeled segment. A 63,700 cfs discharge event (taken to represent a 100-year flood) also predicted only small amounts of local bed changes, with a maximum scour of 1.5 feet in one reach.

Zimmie's application was subject to the same theoretical limitations of the HEC-6 model as was the LMS (1979) study. Most notably, the transport of cohesive organic sediments was not addressed. A number of practical difficulties in

implementation, similar to those of LMS, may be of even more significance. The first concern is the adequacy of calibration. Zimmie's HEC-6 calibration was to observed stage data at the staff gauge at Lock 7, at the mouth of the Champlain Canal, just above the uppermost reach of the model. Observed errors in HEC-6 predictions of stage at this gauge are ≤3 percent, which is a substantial improvement over the LMS (1978) calibration. Nevertheless, it is unclear why Zimmie did not employ similar barge canal stage data from the Crockers Reef Guard Gate, at the downstream end of the pool. Use of such data would have enabled further refinement of the calibration, *i.e.*, a better fit could have been obtained by using variable, Manning's roughness coefficients for model segments.

Calibration of the sediment model was problematic; 1977 and 1984 data were available for grain size distribution of *bottom* sediments, but no *suspended* sediment monitoring was available at the downstream end of the model. Thus, predictions of sediment output from the Thompson Island Pool cannot be considered to be calibrated.

Other serious problems apply to predictions of scour expected to occur under given recurrence interval floods. Zimmie assumed that the 10-year recurrence peak discharge event was 46,600 cfs and the 100-year recurrence event was 63,700 cfs. These flows were applied at the upstream end of the Thompson Island Pool and routed through the system with no additional inflow. As noted at B.4, these values are actually those proposed by FEMA for the *downstream corporate limits* of the *Town* of Fort Edward, located *below* Fort Miller. The FEMA estimates for 10 and 100-year discharges at the Village of Fort Edward (above Lock 7) are 38,800 and 52,400 cfs. Additional analysis in Section B.4 suggests that even these values are too high and that the 10 and 100-year discharges at Lock 7 are actually on the order of 33,300 and 44,300 cfs. This finding implies that the 10-year recurrence event modeled by Zimmie was actually greater than the 100-year recurrence event. The likelihood of significant scour in the Thompson Island Pool may, thereby, be significantly overestimated.

B.5.3 Hydrodynamic Model Description

B.5.3.1 Use of WASP4 Family of Models

TAMS/Gradient modeling efforts are founded on the WASP4 family of models. The current hydrodynamic model in WASP4 is DYNHYD, Version 5.02 (DYNHYD5). This is an updated version of DYNHYD4, documented in Ambrose *et al.* (1988). Important modifications for Version 5 include the ability to accommodate (mildly) trapezoidal channels and evaporation and precipitation (Wool, 1990). The model is an enhancement of the Potomac Estuary hydrodynamic model, DYNHYD2 (Roesch *et al.*, 1979).

DYNHYD5 is constructed as a link-node model, which is essentially a finite volume approach, allowing a two-dimensional problem to be reduced to a series of one-dimensional transport problems. Essentially, the nodes may be conceived of as junctions which *store* water, while the links are idealized channels which *convey* water between a pair of junctions. Taken together, the junctions account for all the water volume in the river, while the channels account for all the water movement in the river. This approach yields a computationally efficient system; at each time step, the equation of motion is solved on all the links, giving flow velocities, while the equation of continuity is solved at the nodes, giving water elevations (hydraulic head).

B.5.3.2 Governing Equations

DYNHYD5 is based on the equations of continuity and momentum, which describe the propagation of a long wave through a shallow water system. The full equations of continuity and momentum for flow in an open channel are usually referred to as the St. Venant equations, which can be derived on a unit control volume (e.g., Bras, 1990). The momentum, or motion, equation is:

$$\frac{\partial V}{\partial t} + V\frac{\partial V}{\partial x} + \frac{g}{A}\frac{\partial(\bar{y}A)}{\partial x} + \frac{V\bar{Q}}{A} = a_f + a_g + a_w \qquad (1)$$

where

V = local velocity of flow in direction x;
A = channel cross-sectional area;
g = acceleration of gravity;
$\bar{y}$ = depth to the centroid of the flow cross section;
$\bar{q}$ = spatially averaged lateral inflow rate;
a_f = frictional acceleration;
a_g = gravitational acceleration; and
a_w = wind acceleration.

The terms on the left hand side of the equation represent, respectively, the local acceleration in velocity; the Bernoulli acceleration, or rate of momentum change by mass transfer; the momentum change induced by pressure differentials related to large water-surface changes; and the momentum change caused by the incoming mass of lateral inflow.

The second equation, the continuity or mass balance equation, is given by:

$$\frac{\partial Q}{\partial x} + \frac{\partial A}{\partial t} = \bar{q}. \qquad (2)$$

For the DYNHYD5 solution of the momentum equation it is first assumed that the flood wave length is significantly greater than the depth and, therefore, the momentum change induced by pressure differentials in (1) can be ignored. For this reason, the model is not appropriate to dam break situations. It is also assumed that lateral inflow occurs only at nodes and not at the links, so that the fourth term on the left side of (1) is also zero.

The right hand side of (1) contains terms for the gravitational and frictional acceleration. In DYNHYD5 a separate term is developed for wind acceleration. Although it is an important factor for flow in estuaries or large

lakes, it is insignificant in the Upper Hudson. The wind acceleration term will, therefore, be ignored in this application.

Gravitational acceleration a_g is driven by the slope of the water surface h_L:

$$a_g = -g \sin (h_L). \tag{3}$$

When h_L is small, $\sin(h_L)$ may be replaced by h_L, which can then be written in terms of the rate of change in head, H, yielding

$$a_g = -g \frac{\partial H}{\partial x}. \tag{4}$$

Frictional acceleration a_f is equal in magnitude and opposite in sign to the gravitational acceleration a_g during steady flow. During unsteady flow, it must also balance other changes in momentum and, thus, can be written in a form similar to a_g, but in terms of the energy slope, S, as:

$$a_f = g \frac{\partial S}{\partial x}. \tag{5}$$

The assumption is then made that the flow is *nearly* steady, so that the energy slope can be determined in the same manner as in steady, uniform flow. In particular, the empirical Manning equation for steady, uniform flow states, for V in units of meters/second, is:

$$V = \frac{R^{2/3}}{n} \left(\frac{\partial S}{\partial x}\right)^{1/2} \tag{6}$$

in which *n* is Manning's roughness coefficient and R is the hydraulic radius.

Solving this equation for the energy gradient and substituting into (5) yields the following expression for the friction acceleration:

$$a_f = -g \frac{n^2}{R^{4/3}} V |V| \qquad (7)$$

in which V^2 has been replaced with $V \cdot |V|$ to insure that the friction acceleration will always oppose the direction of flow.

The complete momentum equation can then be rewritten, neglecting the wind acceleration, as:

$$\frac{\partial V}{\partial t} = - V\frac{\partial V}{\partial x} - g\frac{\partial H}{\partial x} - g\frac{n^2}{R^{4/3}} V |V| . \qquad (8)$$

The equation of continuity (2) is likewise applied to channels by assuming that lateral inflow occurs only at nodes. For flow in a rectangular channel of constant width B, (2) may be rewritten as:

$$\frac{\partial H}{\partial t} = - \frac{1}{B} \frac{\partial Q}{\partial x} \qquad (9)$$

in which the term on the left is the rate of water surface elevation change with respect to time (m/sec) and the terms on the right represent the rate of water volume change with respect to distance per unit width (m/sec).

For mildly trapezoidal channels, such as that of the Upper Hudson, in which the width B is much greater than the depth, the depth can be taken as nearly equal to the hydraulic radius R and the solution for the trapezoidal channel can then be undertaken on an approximately equivalent rectangular channel. For B>>R and moderate side slopes, this method introduces negligible error into the solution.

In addition to the equations of momentum and continuity, solution requires the imposition of boundary conditions. For a river these typically include an upstream inflow condition and a downstream boundary or outflow condition. Variable inflows can be specified as piecewise linear functions, which are interpolated in time.

DYNHYD5 is set up to specify downstream boundary conditions as: 1) an outflow specification; 2) a fixed head boundary; or 3) a tidal boundary. None of these conditions is appropriate for study of the Thompson Island Pool, where the downstream boundary consists of a broad-crested dam. Flow over this dam is not constant nor fixed head, but depends on the elevation behind the dam. Therefore, the model was modified to accept a dynamic discharge condition. That is, outflow is specified as head-dependent and calculated via the Hulsing equation for flow over dams used as weirs:

$$Q = mbH^{3/2} \qquad (10)$$

in which Q is outflow (cfs); b is the effective length of the dam crest (760 feet); and H is the head on the weir (feet), defined as the height of the level liquid surface above the crest. In this equation m is an experimental factor and taken as a calibration parameter. The theoretical value for a trapezoidal weir, with 4:1 end slopes, is m=3.367.

B.5.3.3 Model Implementation

The link-node computational network is solved by writing the governing equations in a finite difference form, yielding one equation for each channel and one equation for each junction. A solution is then obtained explicitly via a modified Runge-Kutta method.

The momentum equation (8) can be written in finite difference form as:

$$\frac{V_i^{t+1}-V_i^t}{\Delta t} = - V_i^t \frac{\Delta V_i^t}{\Delta x_i} - g\frac{\Delta H_i^t}{\Delta x_i} - g\frac{n_i^2}{(R_i^t)^{4/3}} V_i^t |V_i^t| \qquad (11)$$

in which the subscript i refers to channel; the superscript t refers to the present time step of length Δt (sec); and Δx_i is the length of channel i (meters).

The water surface gradient $\Delta H_i/\Delta x$, is computed from the junction heads at either end of the channel. However, the velocity gradient cannot be computed directly from upstream and downstream channel velocities, because of the possible branching in the network. Therefore, an expression for the velocity gradient with the channel must be derived from the continuity equation (2). On substituting $V \cdot A = Q$, the velocity gradient is:

$$\frac{\partial V}{\partial x} = - \frac{1}{A} \frac{\partial A}{\partial t} - \frac{V}{A} \frac{\partial A}{\partial x}. \tag{12}$$

Writing $B \cdot R = A$ and $B \cdot \Delta H = \delta A$, the velocity gradient can be expressed in finite difference form as:

$$\frac{\Delta V_i}{\Delta x_i} = - \frac{1}{R_i} \frac{\Delta H_i}{\Delta t} - \frac{V_i}{R_i} \frac{\Delta H_i}{\Delta x_i} \tag{13}$$

wherein $\Delta H_i/\Delta t$ can be computed as the average water surface elevation change between the junctions at each end of channel i during a given time step. Substituting (13) into (11) yields the explicit solution for the equation of motion in each channel for each time step, again neglecting terms for wind acceleration:

$$V_i^{t+1} = V_i^t + \Delta t \left[\frac{V_i^t}{R_i^t} \frac{\Delta H_i^t}{\Delta t} + \left(\frac{(V_i^t)^2}{R_i^t} - g \right) \frac{\Delta H_i^t}{\Delta x_i} - g \frac{n_i^2}{(R_i^t)^{4/3}} V_i^t \, |V_I^T| \right]. \tag{14}$$

The equation of continuity (9) can be written in finite difference form as:

$$\frac{H_j^{t+1} - H_j^t}{\Delta t} = - \frac{\Delta Q_j^t}{B_j^t \, \Delta x_j} \tag{15}$$

in which j is the junction number. ΔQ_j is the total sum of all i=1 to n flows entering and leaving junction j, while $B_j \cdot \Delta x_j$ is equal to the surface area of the junction As_j.

This yields an explicit solution for the head at each junction at each time step:

$$H_j^{t+1} = H_j^t - \Delta t \frac{\sum_{i=1}^{n} Q_{ij}^t}{As_j^t}. \qquad (16)$$

The solution implementation uses a modified Runge-Kutta procedure. To proceed from time t to time $t + \Delta t$, the following steps are taken.

1. For time $t + \Delta t/2$, predict the mean velocity in each channel using the channel velocities, cross sectional areas and junction heads from time t.

2. Predict the flow in each channel for time $t + \Delta t/2$, using the velocity obtained in (1) and the cross-sectional area from time t.

3. Compute the head at each junction at time $t + \Delta t/2$, using the flows derived in (2).

4. Compute the cross-sectional area of each channel at time $t + \Delta t/2$, using the heads computed in (3).

5. Predict the mean velocity for each channel at time $t + \Delta t$, using the velocities, cross-sectional areas and junction heads computed for time $t + \Delta t/2$.

6. Compute the flow in each channel at time $t + \Delta t$, using the velocity for time $t + \Delta t$ computed in (5) and the cross sectional area computed for $t + \Delta t/2$ in (4).

7. Compute the head at each junction at time $t + \Delta t$, using the flows computed in 6).

8. Compute the cross-sectional area of each channel at time $t + \Delta t$ computed in (7).

B.5.3.4 Model Setup for Thompson Island Pool

Initial modeling efforts have focused on the Thompson Island Pool. If results from these efforts prove promising, they will be expanded to cover more of the river during Phase 2. In order to implement the hydrodynamic flow model, the geometry of this reach of the river was defined or discretized and the flow model was calibrated to observed USGS flow records.

The links in the DYNHYD5 model will later form the model segments for implementation of the transport model (WASP4). During Phase 1, both one-dimensional and two-dimensional versions of the DYNHYD5 were implemented. Although a two-dimensional flow and sediment model will likely be required to investigate the potential erodibility of specific areas, the sediment model was developed and calibrated in one dimension, which required a one-dimensional hydraulic model. The link-node discretization was defined in such a way that the one-dimensional implementation is a direct approximation of the two-dimensional model. This approximation is accomplished by collapsing lateral nodes in the two-dimensional discretization into single nodes for the one-dimensional model. The two-dimensional link-node network is, therefore, described first.

The river cross-sections, bathymetry and initial geometric discretization of the Thompson Island Pool were based on the work in this reach by Zimmie (1985). Although some specific limitations of that effort, notably the limitations of its calibration were addressed earlier, the basic geometry employed by Zimmie was adopted for the DYNHYD5 model. Adoption of this geometry was necessary to avoid the extensive effort of recalculating pool bathymetry from raw data. The cross-sections developed by Zimmie were used in recognition of the fact that current river bathymetry may differ from that determined in 1982 and that additional data may be needed during Phase 2.

In Zimmie's effort, the Thompson Island Pool was described by 32 averaged cross sections, from River Mile 188.5 (Thompson Island Dam) to River Mile 193.64 (just below the mouth of the Champlain Canal near Lock 7 and the tip of Rogers Island). The geometry of the river bed was defined by approximately 50,000 depth

soundings taken in late spring of 1982 by Raytheon for NYSDEC. (Data are apparently now available only in a hardcopy printout.) According to Zimmie (1985), these data were processed by computer to generate cross sections every 115 to 35 feet along the river axis. The cross sections were then grouped into sub-reaches of like physical dimensions and hydraulic characteristics, then interpolated by hand, with the intention of preserving major bedforms, to represent sections varying from 634 to 1267 feet in length. (Sections were drawn to Barge Canal datum, which is 1.177 feet higher than the National Geodetic Vertical Datum.) Overbank elevations were then approximated from FEMA studies. The upstream end of each cross section was assigned a river mile station ID; thus, the true position of each averaged cross section is between the transect lines shown by Zimmie. Most of the river mile transect lines appear to correspond to sediment sampling transects shown by Gahagan and Bryant (1982).

The Zimmie cross sections do not represent exact, discrete cross-sections, but instead represent average conditions throughout a short reach. As such, they should be appropriate for hydrodynamic modeling. Because they are average cross-section descriptions, when they are transferred onto an actual map of the river, it is often necessary to do minor shrinking or stretching to fit the river width at the modeled cross-section.

Adopting Zimmie's average cross-section description represents a compromise. On the one hand, these cross-sections were based on detailed bathymetry of the Thompson Island Pool. On the other hand, the original sounding data are not readily available and the accuracy of the interpolations cannot readily be checked. It is possible that re-discretization of the reach will be needed for future work.

To implement a two-dimensional link-node network, the cross-sections were divided into up to three subsections to reflect bedforms. For example, a given reach might contain a relatively deep central channel and shallower, near-shore areas. Some cross-sections did not indicate any clear change in bedform across the width (*e.g.*, a channel of uniform depth, without shallow near-shore areas), in which case only a single horizontal node was used. A model node was

In order to calibrate the model it is desirable to have at least two reference points. These are provided by the staff gauges maintained during the navigation season on the Hudson River/Champlain Canal by NYSDOT. Gauge #119 is located at the mouth of the Champlain Canal below Lock 7c and, thus, is approximately coincident with the uppermost node of the link-node model. Gauge #118, Crockers Reef, is located at the guard gate in the land cut bypassing the Thompson Island Dam and is approximately parallel with the eastern section of the dam. The canal entrance is, however, approximately 0.5 miles upstream of the dam. Given the low energy gradients normally experienced in the Thompson Island Pool, levels at this gauge are expected to be approximately equal to those at Node 13 of the two-dimensional model (Node 5 of the one-dimensional model). Both gauges are normally read twice a day from mid-April to mid-December and the two values averaged. (Observations are sometimes omitted during floods.)

Initial calibration, using both the two-dimensional and one dimensional networks, was undertaken on flows of Spring 1978. As this year was a relatively low flow year, the model was relatively insensitive to specification of calibration parameters. In other words, moderate flows pass through the pool with little dynamic character; specification of typical Manning roughness coefficient values of 0.027 for the uppermost, dredged channel reaches and 0.030 for the remaining reaches suffices to give an excellent fit to the staff gauge data. Additional calibration of the one dimensional model was performed on the major spring flood of April-May 1983. In this year, a daily average flow of 32,600 cfs was reported at Fort Edward on May 2, 1983 and a peak flow of 35,200 cfs was reported on May 3, 1983. These flows are on the order of 20-year recurrence interval events, but are the largest flood events on record since monitoring began at Fort Edward in December 1976. Inputs to the model were taken as the daily average flows recorded at Fort Edward, because the full record of hourly peaks was not readily available. The hydrodynamic model was run on a six second time step, with the inflow rates interpolated between daily midpoints.

At these high flow rates, model performance is more sensitive to the calibration parameters. Reasonable calibration was obtained by setting the Manning roughness coefficient to 0.026 in the uppermost two nodes, 0.027 in the

next three lower nodes, 0.028 in the node below that, and 0.029 in the remaining nodes down to the Thompson Island Dam. In addition, the weir discharge parameter was set to 3.36 to fine tune stage elevations in the lower end of the pool.

Comparison of predicted and measured stage observations for the flood period and for the subsequent declining limb of the flood is shown in Figure B.5-3. A reasonable fit (usually within a tenth of a meter) is provided, considering that: 1) only daily average flow and not full hydrographs were used as input; and 2) there may be noticeable differences in stage between the main channel and barge canal gauges during extreme floods.

Additional calibration of the hydrodynamic model is expected in Phase 2, depending on the data needs of the sediment model.

B.5.4 Sediment Transport Model

The sediment transport model STREAM (Borah *et al.*, 1982a; Borah and Bordoloi, 1989a, 1989b, 1991) is used here to simulate bed and bank scour, sediment deposition and resuspension, and sediment transport for the Thompson Island Pool. Spatial and temporal variations of the flow conditions and hydraulic parameters are obtained from the output of DYNHYD5, described in the previous section, and provide the hydraulic parameters, *e.g.*, flow, velocity, *etc.*, needed for the sediment model.

B.5.4.1 Streambed Erosion and Deposition

The amount of sediment transported in, deposited in or eroded from an alluvial stream bed is the result of imbalances between sediment transport capacity of the flow and the incoming sediment. Such an imbalance is determined by considering local conservation of mass. During an erosion condition, particle entrainment occurs, if the particles on the bed surface are transportable with the existing flow conditions. Otherwise the particles remain on the bed surface as part of an armor layer. These processes are simulated using the algorithms discussed in the subsections below. The sediment is divided into small size

groups, based on the particle size distributions with each group represented by an average diameter. Each sediment group is considered individually and the total response is determined by adding responses of all the groups.

Sediment Transport Capacity

The sediment transport capacity of a flow may be expressed as a function of the flow parameters, such as depth and velocity and the particle size. There are many sediment transport formulas available today; a review of these is given by Vanoni (1975) and evaluations of some are made by Alonso *et al.* (1981). Such formulas are directly applicable when modeling the transport of uniform sediment; they are not directly applicable for simulating transport of nonuniform sediment.

A given flow has a characteristic capacity for transporting different sediment size groups; sediment transport capacity is calculated separately for each particle size group. Thus, as the transport capacity for each size group is calculated, transport capacity for the remaining size groups is reduced. A variable called the residual transport capacity accounts for this incremental transport capacity calculation:

$$T_{ri} = T_i \Omega \tag{17}$$

$$\Omega = 1 - \sum_{j=1}^{N} \frac{C_j}{T_j} \tag{18}$$

where

T_{ri} = the residual transport capacity for size group i;

T_i = the sediment transport capacity for group i;

C_j = the volumetric concentration of sediment group j; and

N = the total number of sediment size groups considered in the simulation.

The term Ω in Equation (17) is used as an indicator for entrainment/erosion when Ω>0, deposition when Ω<0 or equilibrium when Ω=0.

Active Bed Layer

The materials available for entrainment are essentially those exposed at the bed surface. The model assumes entrainment of particles starting from the smallest size group exposed at the surface. Entrainment then continues with larger sizes, deeper into the bed until either Ω=0 or the bed is covered with non-transportable materials forming an armor layer. This process is assumed to be taking place from an upper bed layer called the active layer. The thickness, porosity and particle size distribution of this layer can vary throughout the simulation, but the layer is assumed homogeneous at all times. For bed materials having enough non-transportable sizes to form an armor layer, the thickness of the active layer is expressed as:

$$\tau_a = \frac{d_L}{(1-\lambda)\sum_{i=L}^{N} P_i} \qquad (19)$$

where

τ_a = the thickness of the active layer under armoring conditions;

d_L = the size of sediment group L;

L = the smallest non-transportable sediment group present in the bed material;

λ = the porosity of the bed material; and

P_i = the fraction of sediment group i.

Equation (19) is valid if the bed material consists of enough non-transportable particles to form an armor layer. In case of bed materials not having enough non-transportable particles, several layers will be scoured away

until the flow achieves equilibrium. Under such conditions, the following expression is used to compute the active layer thickness:

$$\tau_n = \frac{d_{85}}{0.15\ (1-\lambda)} \tag{20}$$

in which τ_n = the thickness of active layer under non-armoring conditions; and d_{85} = the sediment size under which 85 percent particles are finer.

In general, Equation (19) is used only if $d_L < d_{85}$; otherwise Equation (20) is used. This equation was based on model testing by Borah and Bordoloi (1989a) on Little and Mayer's (1972) experimental data.

Bed Erosion

Particle entrainment from an active layer is simulated using an ordering procedure. In this procedure, it is assumed that entrainment begins with materials from the first (smallest size) sediment group exposed at the surface. Next, materials from the second (next larger size) group, which were already exposed or newly exposed at the surface due to entrainment of the materials from the first group, may then entrain. This entrainment may be followed immediately by the entrainment of additional particles from the first group, which become exposed after removal of the particles from the second group. Next, materials from the third group may entrain followed by the materials from first and second groups, which were directly underneath those third group materials, and then the first group materials, which were underneath these second group materials. This sequence of material entrainment for the number of size groups N are summarized in a matrix called the entrainment frequency matrix. The elements of this matrix are obtained from the following expressions:

$$F_{ij} = 2^{i-j-1}, \quad for\ j<i \tag{21}$$

$$F_{ij} = 1, \quad for\ j=i \tag{22}$$

$$F_{ij} = 0, \quad for\ j>i \tag{23}$$

where

- i = the matrix row representing a sediment size group exposed at the bed surface;
- j = the matrix column representing a sediment group hidden under group i; and
- F_{ij} = the element of the entrainment frequency matrix representing probable frequency of entrainment of group j after group i (i>j) is entrained.

Volumes of potential entrainment from different size groups, based on the entrainment frequency matrix, are computed and arranged in the corresponding elements of another matrix called the volume entrainment matrix. Erosion or entrainment volumes for different size groups during a time interval are computed from this matrix and are expressed as:

$$E_j = e \sum_{i=j}^{M} V_{ij} \tag{24}$$

$$V_{ij} = \frac{F_{ij}V_iV_j}{\sum_{k=j}^{N} F_{kj}V_k} \tag{25}$$

$$V_j = \sum_{i=1}^{N} V_{ij} = W\tau(1-\lambda)P_j \tag{26}$$

where

- E_j = the erosion volume of sediment group j per unit length during a time interval;

e = sediment erodibility parameter, $0 \le e \le 1$, ranging from 0 for non-erodible material to 1 for detached and easily erodible sediments;

v_{ij} = the element of the volume entrainment matrix representing volume of size group j in the active layer per unit length which is exposed at the surface due to the removal of volume v_{ii} of group i;

M = the row number in the volume entrainment matrix v_{ij} for $\Omega=0$;

V_j = the volume of group j present in the active layer per unit length; and

W = the active bed width; and τ = the active layer thickness in Equation (19) or (20).

The computation of entrainment volume in Equation (24) is accomplished through a search procedure starting from the first row of the volume entrainment matrix. Every time entrainment from an element is computed, the term Ω is updated using Equation (18) after increasing C_j by the eroded volume. Entrainment computation continues towards higher rows until $\Omega=0$ or the bed is armored ($T_i = 0$).

The sediment erodibility parameter (e) represents the resistance to erosion due to cohesion or other bonding properties (Borah *et al.*, 1982a). While testing STREAM with Little and Mayer's (1972) experimental data, Borah and Bordoloi (1989) found this parameter to be 1.0 for noncohesive bed materials. For Hudson River sediments, it will serve as an initial calibration parameter for modeling the effects of cohesive sediments. More detailed cohesive models may be necessary after further model testing.

Sediment Deposition

Whenever $\Omega<0$, deposition is assumed and particles beginning with the largest size group are dropped out. The simulation continues until the flow is no longer overloaded ($\Omega=0$) or all the material from different size groups present in the flow are settled. Thus, the deposition volume during a time interval is computed as:

$$D_j = AC_j, \quad if\ |T_{xj}| \geq C_j\ or\ T_j = 0 \tag{27}$$

$$D_j = A\,|T_{xj}|, \quad if\ |T_{xj}| < C_j \tag{28}$$

in which D_j = the deposition volume of size group j per unit length during a time interval. The amount of sediment reaching the bed depends on the particle settling velocity. Therefore, D_j is adjusted by a correction factor of $2w_j\Delta t^*/h$ (<1) to account for slow deposition of particles with low settling velocities, where w_j = the average fall velocity of size j; Δt^* = the time interval computed in the characteristic solution of sediment continuity equation (discussed later); and h = the average flow depth.

Bed Elevation Change

The change in bed elevation during a computational time interval is computed as:

$$\Delta Z = \frac{\Delta t}{W\,\Delta t^*} \sum_{j=1}^{N} \frac{D_j - E_j}{1-\lambda} \tag{29}$$

in which ΔZ = the bed elevation change and Δt = the computational time interval selected for the simulation.

B.5.4.2 Streambank Erosion

The lateral erosion of a cohesive riverbank is given by Arulanandan *et al.* (1980) and is expressed as (Osman and Thorne, 1988):

$$\Delta W = \frac{R(\tau-\tau_c)\ \Delta t}{\gamma \tau_c} \tag{30}$$

$$R = 0.0022\ \tau_c\ e^{-0.13\tau_c} \tag{31}$$

where

ΔW = the bank erosion distance (in m) in one of the banks;

τ_c = the critical shear stress (in dynes/cm^2) for cohesive soils with given sodium adsorption ratio, pore fluid salt concentration and dielectric dispersion;

γ = the soil unit weight (in KN/m^3);

τ = the average shear stress (in dynes/cm^2); and

Δt = the computational time interval (in minutes).

The rate at which the eroded bank material is added to the sediment load is computed by introducing a new linear relation.

The bank height (H) is updated by subtracting ΔZ from its initial value. The bank height above the zone of lateral erosion (H') is computed by subtracting $\Delta W \tan\theta$ from the initial bank height, where θ is the bank angle. Using the ratio of these heights $(H/H')_m$, the critical height ratio for unstable bank is computed as (Osman and Thorne, 1988):

$$\left(\frac{H}{H'}\right)_c = \frac{1}{2}\left[\frac{\lambda_2}{\lambda_1} + \sqrt{\left(\frac{\lambda_2}{\lambda_1}\right)^2 - 4\left(\frac{\lambda_3}{\lambda_1}\right)}\right] \tag{32}$$

$$\lambda_1 = (1-K^2)\,(\sin\beta\cos\beta - \cos^2\beta\tan\theta) \tag{33}$$

$$\lambda_2 = 2(1-K)\ \frac{\sigma}{\gamma H'} \tag{34}$$

$$\lambda_3 = \frac{\sin\beta\ \cos\beta\ \tan\theta - \sin^2\beta}{\tan\theta} \tag{35}$$

$$\beta = \frac{1}{2}\left(\tan^{-1}\left[\left(\frac{H}{H'}\right)_m^2 (1-K^2)\tan\theta\right] + \phi\right) \tag{36}$$

where

H	=	the bank height;
H'	=	the bank height above zone of lateral erosion;
$(H/H')_c$	=	the critical height ratio for an unstable bank;
θ	=	the initial bank angle;
β	=	the failure plane angle;
ϕ	=	the angle of internal friction;
σ	=	the cohesion;
K	=	the ratio of crack depth to bank height (y/H); and
y	=	the depth of tension cracking.

If the bank height ratio $(H/H')_m$ is approximately equal to the critical bank height ratio $(H/H')_c$, bank failure would occur. The failure block width and volume are computed as (Osman and Thorne, 1988):

$$BW = \frac{H - y}{\tan\beta} - \frac{H'}{\tan\theta} \tag{37}$$

$$VB = \frac{1}{2}\left(\frac{H^2-y^2}{\tan\beta} - \frac{H^2}{\tan\theta}\right) \quad (38)$$

in which BW = the width of failed bank on one side; and VB = the volume of failed block per unit length on one side.

After the bank has initially failed, the new bank slope becomes β and remains the same for parallel bank retreat. The subsequent bank failure width and volume are computed using Equations (32) through (38) after substituting $\theta = \beta$, $K = 0$, $y = 0$ and $\lambda_3 = \lambda_1$.

The volumes of materials generated from lateral bank erosion $(\Delta W)^2\tan\theta$ and bank failure VB are not readily available to the entire cross sectional flow. Therefore, these materials are stored at the bank toes and the volume of lateral sediment contribution (Equation 44) during a computational time interval is computed by multiplying this storage by the fraction $2\Delta W/(W+2\Delta W)$. This procedure is a modification of the original procedure of Osman and Thorne (1988), where the total sediment transport capacity was used to transport these materials.

Sediment Routing

The routing algorithm of nonuniform sediment is primarily based on the conservation of mass for the sediment load and materials on the bed per size group. The total sediment load (concentration or discharge) is computed by adding the corresponding values from all the size groups. The continuity equation for a size group may be written as:

$$\frac{\partial Ac}{\partial t} + \frac{\partial Q_s}{\partial x} = q_s - \delta + \eta \quad (39)$$

where

A = the flow area;

c = the volumetric concentration of the sediment load;

Q_s = the volumetric sediment discharge;

q_s = the volumetric rate of lateral sediment inflow per unit length;

δ = the volumetric rate of sediment deposition per unit length;

η = the volumetric rate of sediment entrainment per unit length;

t = time; and

x = the longitudinal distance.

Assuming constant flow conditions and constant q_s, δ and η within small intervals of time and space and assuming sediment moving with the same velocity as water, Equation (39) is expressed in a quasi-linear hyperbolic form, governing the propagation of sediment load waves, and solved by the method of characteristics which yields (Borah *et al.*, 1982a):

$$C_{j,m+1} = C_j + \frac{E_j}{A} - \frac{D_j}{A} \tag{40}$$

$$C_j = C^n_{j,m} + \frac{q_{s,j}\,\Delta x}{Q} \tag{41}$$

$$E_j = \eta_j\,\Delta t^* \tag{42}$$

$$D_j = \delta_j\,\Delta t^* \tag{43}$$

$$q_{s,j} = \frac{2\,\Delta W\,[(\Delta W)^2 \tan\theta + 2\,VB]\,P_j}{(W + 2\,\Delta W)\,\Delta t} \tag{44}$$

$$\Delta t^* = \frac{\Delta x}{V} \qquad (45)$$

in which m = the subscript representing space increment node; n = the superscript representing time interval node, Δx = the space increment ($x_{m+1} - x_m$); Δt^* = the time increment ($t - t^n$) given by Equation 45; Q = the water discharge flow rate; and V = the average water velocity.

As indicated earlier, Equation (44) is a new addition to the original procedure. After the first bank failure, the angle θ is replaced by β (Equation 11e) and kept the same afterwards.

In the above solution, Q, V and A are constant within each space element (Δx) and time increment (Δt^*). The concentration $c_{j,m+1}$ (Equation 40) occurs at x_{m+1} and $t^n+\Delta t^*$. Concentration $c^{n+1}_{j,m+1}$ at x_{m+1} and $t^{n+1} = t^n+\Delta t$ is computed by interpolating or extrapolating $c^n_{j,m+1}$ (initial value) and $c_{j,m+1}$ between t^n and $t^n+\Delta t^*$. This concentration is multiplied by Q to compute sediment discharge for the size group j. Adding sediment discharges for all the groups, the total sediment discharge is computed. From these values, size distribution of the sediment load is computed.

The bed material volume V_j per unit length of size group j in the active layer is updated by adding D_j and subtracting E_j. From all the updated bed material volumes, the new size distribution of active layer is computed based on which new active layer thickness is computed. During bed erosion, the new thickness may be deeper than the remaining thickness from the previous time interval. In such cases, additional parent bed materials from the excess thickness are added to the current active layer materials and size distribution of the active layer is revised.

The bed elevation within Δx is updated by adding ΔZ in Equation (29) to the initial value. The bed porosity is computed in the model using the following empirical relation given by Komura and Simmons (1967):

$$\lambda = 0.245 + \frac{0.0864}{(d_{50})^{0.21}} \quad (46)$$

in which d_{50} = the median particle size in cm.

B.5.4.3 Initial Calibration Efforts

The initial calibration efforts for the STREAM sediment transport model are ongoing. These initial efforts for the Thompson Island Pool section of the Upper Hudson are being applied to a one-dimensional segmentation of the river reach, containing 31 segments in the five-mile river reach. Two scenarios are being investigated: 1) a moderate flow event which occurred in mid May 1983; and 2) a flood event in late April 1983.

In order to complete the sediment model calibration, additional field data defining suspended sediment loads over the Thompson Island Dam and further bed sediment data are needed. Continuing calibration efforts and on-going modeling may be performed in subsequent phases, depending on the need.

B.5.5 Summary

During Phase 1, transport modeling efforts have focused on proposing and selecting an appropriate transport model and defining preliminary hydraulic and sediment transport scenarios upon which to base the calibration of the models. At present, the hydraulic model has been calibrated with results showing an improvement over previous HEC-6 model efforts by others. The sediment transport model has been implemented, but final calibration requires additional data.

SYNOPSIS

PRELIMINARY HUMAN HEALTH RISK ASSESSMENT
(Section B.6)

This section sets forth the objectives, method and results of a preliminary baseline human health risk assessment. The main objective of the preliminary assessment (B.6.1) is to examine the quality of the available site data for risk assessment purposes and identify where additional data are needed to perform a more complete assessment of potential health risks. For the purposes of this assessment, only risks associated with exposure to PCBs are evaluated. Future land uses in the area are assumed to be similar to current land uses.

There are several potential pathways by which people might be exposed to PCBs originating from the Hudson River (B.6.2). Potential exposure to PCBs originating from the Hudson River via dietary intake includes exposure from ingestion of fish, home-garden crops, beef or dairy products, human breast milk and drinking water. Of these dietary intake pathways, only potential exposures from ingestion of fish and drinking water are quantified. Remaining dietary intake pathways could not be quantified, because insufficient data exist to determine whether or from what source PCB exposure may occur. While potential exposure as a consequence of inhalation of PCBs in ambient air is discussed, exposures occurring via this pathway could not be evaluated quantitatively. Sampling data are too sparse and/or inadequate to determine: 1) representative PCB concentrations in ambient air; or 2) the contribution of volatilization of PCBs from the Hudson River as opposed to contributions from other sources. Recreational exposures include dermal contact with sediments and river water as well as incidental sediment ingestion during recreational activities, all of which were quantified. The analysis has revealed that estimated PCB intake through consumption of fish from the Hudson River appears to be the most significant, potential pathway of human exposures to PCBs from the site.

A discussion of the current understanding of potential carcinogenic and non-carcinogenic toxic effects associated with exposure to PCBs (B.6.3) summarizes methods used by USEPA to derive toxicity values and estimate potential health risks associated with exposures to PCBs.

Quantitative exposure estimates are evaluated in conjunction with the toxicity information in order to predict the potential for human health effects (B.6.4) associated with exposure to Hudson River PCBs. Two types of health risk evaluations are presented: non-carcinogenic health effects and carcinogenic risks. The potential health risks associated with all quantified pathways other than the ingestion of fish are estimated to be within the acceptable range.

Based on available data, there appear to be unacceptable potential cancer and non-cancer risks associated with regular ingestion of fish from the Upper Hudson River. Assuming consumption of PCB-contaminated fish for 30 years, cancer risks are estimated to be as high as 2 in an exposed population of 100. With respect to non-cancer risks, the average daily exposure to PCBs resulting from consumption of fish from the Upper river may be as high as 51 times the reference dose. This evaluation shows that the population that regularly consumes

fish from the Upper Hudson River is at risk from PCB exposure. The working assumption is that people still consume fish, despite the fishing ban. This assumption may require further quantification, because, as the operative risk, it will be useful to ascertain the effectiveness of the fishing ban.

B.6 Preliminary Human Health Risk Assessment

B.6.1 Phase 1 Objectives

USEPA has prepared this preliminary, baseline human health risk assessment for several reasons. First, the 1984 ROD did not include any quantitative information concerning the risks to human health from PCBs in the Hudson River and USEPA would like to initiate early public review of this complex topic. Second, USEPA needed an evaluation of those pathways, such as airborne PCBs or ingestion of food crops or livestock, where sufficient empirical data are not available, so as to determine an appropriate data acquisition program. Third, USEPA needed to know whether the historical assumption that fish were the significant pathway of potential human exposure was empirically verifiable in order to consider whether institutional control, i.e., a fishing ban, might be required, regardless of other concurrent remedial alternatives, or might be used because it is most protective. Further, USEPA wanted to disclose all current working assumptions used in this risk assessment, because various reviews are underway that may require subsequent modifications to this baseline assessment. By identifying these areas of uncertainty now, the final result will be more easily understood, regardless of whether any adjustments are necessary or not.

This preliminary evaluation is based on appropriate conservative exposure assumptions (as defined by USEPA guidance), the most recent toxic potencies estimated by appropriate USEPA offices and the most current regulatory criteria and standards of NYSDEC and USEPA. The potential baseline risks addressed are the current public health risks, which might result from taking no remedial action, *i.e.*, the no action alternative for the river. Risks associated with any proposed remedial action for the river or natural biodegradation of contaminants will be assessed in Phase 3 - Feasibility Study.

As PCBs have been identified as the major sources of health risks associated with exposure to site related media, this preliminary assessment is limited to evaluating potential health risks associated with PCBs.

Unlike many NPL sites where current use differs significantly from plausible use in the future, it is assumed here that future use would be the same as present use along the river and would not alter the potential for exposure of area residents. Therefore, no alternate analysis for a hypothetical future exposure scenario is provided. Also, this baseline assessment does not consider the potential occurrence of a major scour event.

The scope of the preliminary evaluation is limited to addressing those risks stemming from PCB contamination that currently exists in the Hudson River or from PCB contamination that can be directly attributed to the river. For example, potential risks due to exposures to PCB-contaminated fish, sediments and surface water are evaluated in this assessment, since there is little question that this contamination can be quantitatively and directly attributed to riverine levels. Potential risks stemming from exposures to PCB-contaminated soils or products produced on those soils can not be evaluated, since sufficient data are not available, the source of the PCBs in soils is unclear and, if present, such PCBs may not be directly attributed to the river.

In interpreting the findings of this report, it is important to note that new data regarding PCB concentrations, e.g., in fish or other media, in the river system will be utilized as they become available. In addition, ongoing studies regarding the toxicology of PCBs and human risks associated with PCB exposures will be incorporated, if accepted by USEPA through a scientific review process before the completion of the Reassessment RI/FS. Subsequent phases of the Hudson River reassessment process will incorporate such new information and analyses.

B.6.2 Exposure Assessment

B.6.2.1 Introduction

Following identification of PCBs as the chemical of concern, an examination of potential exposure routes, potentially exposed populations and exposure pathways was performed. To the extent data were available, the magnitude of

exposures via each pathway was estimated in a two step process, considering both contaminant concentration and human exposures.

Contaminant concentrations in each of the environmental media of concern (*e.g.*, water, sediments, air, *etc.*) are determined at relevant receptor points. Determination of media of concern is based on analyses of mechanisms of contaminant release from the site and environmental fate and transport as well as consideration of locations and mechanisms of human contact with site contaminants.

As suggested in USEPA's recent Risk Assessment Guidance for Superfund (1989b), "reasonable maximum" individual exposure concentrations are calculated to the extent appropriate. Geographic variations in environmental concentrations are considered in determining appropriate exposure point concentrations. Duration of exposure and the likelihood of exposure pathways occurring are also evaluated.

Human exposures to PCBs are quantified using the environmental concentrations together with estimates of media intake. These scenarios, under which exposures are evaluated, include assumptions regarding physiological parameters, such as body weights, media intake rates, such as soil ingestion rates, and activity patterns, such as frequency of contact at the site. In some instances, standard exposure assumptions are included in the assessment. For example, throughout this assessment, a 30-year duration of residence in the Hudson River area, and a 70-year lifespan are incorporated into exposure calculations, based on recent USEPA guidance (USEPA, 1989a). Similarly, a lifetime average body weight of 70 kg is assumed. In other cases, assumptions are tailored to site-specific conditions as appropriate.

The population of concern in the evaluation of the Upper Hudson River consists of the inhabitants of the towns, cities, and rural areas surrounding the River. Exposure by these populations to PCBs present along the river and to PCBs that have migrated from the River could occur via a number of potential pathways, as illustrated in Figure B.6-1.

Several gaps in the available information emerged during the process of quantifying an exposure dose and precluded a thorough, quantitative exposure assessment for some exposure pathways. Concentration data for PCBs in some of the media of concern were either non-existent, out of date, or of questionable applicability. In addition to data limitations, several pathways cannot be quantitatively assessed, because they are not considered complete at this time. Rather than calculating exposures (and associated risks) from data of questionable relevance, it is considered more appropriate to point out the limitations and suggest possible means of acquiring better, more relevant data. The potential exposure pathways considered in this analysis and the type of evaluation performed for each pathway are summarized in the tabulation on the following page.

B.6.2.2 Dietary Intake

Fish Consumption

Because fish effectively bioaccumulate PCBs, fish provide a pathway, frequently the predominant pathway, for human exposures to PCBs. Studies conducted on Michigan residents established that those who regularly ate Lake Michigan fish had serum PCB levels up to 30 times greater than those who did not eat these fish (Humphrey, 1987). Data on PCBs in Hudson River fish, discussed in Section B.3, clearly indicate that fish consumption can result in human exposures to PCBs.

Recent studies (NYSDEC, 1990) indicate that the Hudson River continues to draw a significant number of anglers. Estimates are that 26,870 (±3,440) individuals fish the Hudson River for a total of 232,110 (±51,310) angler days. Over 38 percent of these individuals claim to fish the Upper Hudson (section north of Federal Dam at Troy) for an estimated 87,060 (±22,090) angler days along

Summary of Potential Upper Hudson Exposure Pathways

Exposure Pathway/Medium	Potentially Exposed Population	Type of Evaluation
Fish Ingestion	Adult local fishing population – Upper Hudson from Albany to Fort Edward	Reasonable maximum exposure (RME) – Two Scenarios: (1) Recent fish data 1986-1988 (2) 30-year projections based on current trends • 50th percentile fish ingestion rate for local anglers (≈50 half-pound servings per year).
Drinking Water	Adult residents using the River as a potential regular source of drinking water	Screening type assessment: Possible future drinking water use near Fort Edward with no PCB removal in water treatment.
Dermal PCB Absorption from Sediments	Adults and children during summertime recreational activities on the River – Thompson Island Pool	Wading: Hands and feet exposed.
Dermal PCB Absorption from Water	Adults and children during summertime recreational activities on the River – Thompson Island Pool	Swimming: Full body exposed.
Ingestion of PCBs from Sediments	Adults and children during summertime recreational activities on the River – Thompson Island Pool	Incidental Uptake: Hand to mouth activity
Pathways Not Evaluated Due to Inadequate Data		
Inhalation of PCBs in Air	Limited recent air measurements are available. It is difficult to determined whether PCBs in air are from the river, dump sites, or other (unknown) sources.	
Ingestion of PCBs in Local Snapping Turtles (Cooked)	A few snapping turtles in the Upper Hudson have been found to contain PCBs. It is uncertain whether the small sample is representative and further unknown whether turtles are caught and consumed by local residents.	
Ingestion of Garden or Agricultural Crops	Historical studies have shown some PCB contamination in plants (e.g., trees and pasture) near the river; recent data are lacking. It is difficult to determine whether PCBs from the river are the major or only source of possible crop contamination.	
Ingestion of Milk or Meat	No data are available.	
Ingestion of Breast Milk	No data are available.	

this portion of the river. (The source of this information does not specifically indicate the proportion of days in the Fort Edward to Federal Dam reach.) From the NYSDEC study, it appears that local populations do most of the fishing along the river and the average distance traveled per fishing trip is approximately 34 miles.

NYSDOH has issued a Health Advisory that recommends against consumption of any fish taken from the Upper Hudson River (Hudson Falls to Federal Dam) and limited intake of many species from the Lower Hudson. It is, however, USEPA policy not to assume that fishing bans or other similar types of institutional controls have any significant long-term effectiveness in reducing the intake of contaminated fish. Therefore, this baseline risk assessment evaluates the possible risks that would be associated with consumption of fish and the potential risk that would arise from consumption of fish at a rate that might be expected to occur in the absence of or despite the Health Advisory. This approach is prudent in light of information from the NYSDEC angler survey indicating that individuals may be unaware that they are consuming potentially contaminated fish, even though they are aware of fishing advisories (NYSDEC, 1990). Or individuals may feel that their fish are safe to eat, even though the water from which the fish come are known to be polluted (Belton *et al.*, 1986).

Specific examples of disregard for fishing advisories have been addressed in a study conducted by the New Jersey Department of Environmental Protection (NJDEP) and Rutgers University (Belton, 1986). In this survey of anglers along the Lower Hudson River, New York Bay, and Newark Bay, observations by NJDEP personnel indicated that fishing advisories were being ignored. While half of the anglers surveyed claimed knowledge of warnings, two-thirds of those who admitted to eating their catch considered them to be totally safe to eat, while others considered them to be polluted but not harmful. The fishermen were found to hold several beliefs that made them resistant to advisories. For example, some stated that fish were able to avoid pollution through a self-cleansing mechanism; other fishermen felt that they could determine whether a fish was safe to eat based on the physical appearance, smell, taste, or behavior of the fish.

Almost half of the respondents indicated that the fish must be safe to eat, "if you ate them and had no reaction within a day or two."

Given the presumption that people do fish in the Upper Hudson below Hudson Falls, a key ingredient in estimating exposure via this route is the amount of fish caught and eaten by local anglers. USEPA (1989a) suggests use of the intake rate of 30 g/day or 140 g/day for the 50th and 90th percentile values, respectively, of annual average daily intake. These numbers represent values from studies by Puffer (1981) and Pierce (1981), which investigated the consumption of fish by individuals who were known to fish regularly from piers in southern California and Puget Sound. While these study locations are quite distinct from the Upper Hudson River, the values generated in these studies are considered by USEPA to be more representative of actual annual consumption rates for recreational anglers (USEPA, 1989a) than other values available in the literature. Other values are usually based on the amount of fish that enter the commercial market divided by the entire civilian population of the US, thus, are averaged out over non-consumers and do not adequately represent consumption by recreational (or subsistence) anglers.

According to the NYSDEC (1990) study of angler activity, the average consumption values suggested by USEPA also appear to be consistent with consumption rates among anglers resident in New York State. The NYSDEC reports an overall fish consumption rate for statewide anglers of 45.2 meals per year, or an estimated intake of 27.7 grams per day (g/d). Consumption rates higher than the national average were found for all sociodemographic groups, with a consumption increase associated with increasing age, increasing income, and increasing education. In light of these state-specific data, use of the USEPA recommended value for an average ingestion rate (30 grams/day) appears justified. This value of 30 g/d is prorated over 365 days/year and is equivalent to a one-half pound serving approximately 50 times per year.

Another issue important to assessing possible PCB exposure from fish consumption includes the species of fish that are actually consumed by human populations. According to the most recent survey of New York anglers, 38 percent

of the days spent fishing on the Hudson River (Upper and Lower) are spent fishing for bass. Another 6.5 percent of days is spent fishing for brown trout. Over 22 percent of the days are spent in pursuit of "no specific type" of fish, and 33 percent of days are spent for "other" species. No information specific to the Upper Hudson is available. According to data regarding species preferences on a statewide basis, bass are the fish most frequently sought after, followed sequentially by brown trout, rainbow/steelhead trout, yellow perch, and walleye (yellow pike). The statewide values appear to echo the data specific to the Hudson.

The only available PCB data for the species preferred by fishermen are for PCBs in bass. Since a large percentage of the individuals polled in the NYSDEC survey indicated that they fished for other species or no specific type of species, all species sampled were used to provide the best characterization of the exposure concentration. Additionally, in evaluating the data it became clear that any differences in concentrations among species were not of sufficient magnitude to necessitate treating each species separately.

PCB concentrations in fish fillet or eviscerated fish were used according to available data. The fillet represents the part of the fish most commonly consumed. Although concentrations of some organochlorine compounds in fish may decrease while cooking (Stachiw, 1988), the data for PCBs are not consistent. One study reported a small decrease in PCB concentrations with cooking (Zabik, 1982), while another study reported a wide range of PCB decrease after cooking (Cordle, 1982). Because, no specific value can be derived based on the available data and no information on PCB concentrations after cooking the species of concern from the Hudson are available, no adjustments were made.

Because of the uncertainty in predicting future trends from current levels of PCBs in fish, potential exposures from PCBs in fish are evaluated for two scenarios: 1) using the most recent concentration data, *i.e.*, 1986-1988, to represent appropriate exposure concentrations, and 2) using a 30-year average concentration, estimated by assuming that the observed exponential rate of decline in tissue concentrations of PCBs continues from 1991 to 2020.

For the first scenario, the exposure concentration for PCBs in fish is the 95th percent confidence limit of the arithmetic average PCB concentration in fish sampled from 1986 to 1988, across all species sampled from River Miles 153 to 195. This 95th percent upper confidence limit of 12.0 ppm (12.0 mg/kg) was determined to represent a reasonable upper bound estimate of exposure concentration for several reasons. First, guidelines for assessing baseline human health risks for Superfund sites state that for "any estimate of exposure concentrations, the upper confidence limit (*i.e.*, the 95 percent upper confidence limit) on the arithmetic average will be used for this variable" (USEPA, 1989b). Second, fish tend to migrate at least over a limited distance and thus, exposure concentrations would be integrated over distance. Third, anglers fishing the Hudson River over the 30-year exposure duration would typically fish more than one location over time. Taken together, the second and third points indicate that it would <u>not</u> be reasonable to assume that either the most contaminated fish, nor fish from the most contaminated location are consumed continuously over a long period of time.

For the second scenario, the exposure concentration is based on future average concentrations of PCBs in fish over the period of the next 30 years, as calculated from the available data on largemouth bass and brown bullhead. A value of 1.5 ppm is considered predictive of the 95th percent confidence interval of the 30-year mean, barring any unforeseen redistribution of PCBs within the river. The full derivation of this value is described at B.4.4. The validity of the 30-year concentration value should be assessed as new information on potential resuspension and redistribution of PCBs in sediments is developed.

In summary, exposure to PCBs from ingestion of contaminated fish is calculated assuming an annual average daily intake rate of fish of 30 g/d and exposure concentrations of PCBs in fish of either 12.0 ppm or 1.5 ppm, considered to reasonably approximate the range of concentrations. Fish consumption is assumed to occur each year, over a 30-year residence time in the vicinity of the river. Also assumed are a lifetime average bodyweight of 70 kg and 100 percent gastrointestinal absorption of PCBs from the fish.

These exposure assumptions are summarized in Table B.6-1. The calculation used to estimate exposure to PCBs from ingestion of fish is:

$$\text{Human Intake of PCBs from fish ingestion (mg/kg-day)} = \frac{FIR \times FC \times GI \times EF \times ED \times CF}{BW \times AT}$$

where

- FIR = Fish ingestion rate, 30 g/day (USEPA, 1989a)
- FC = Fish concentration of PCBs (1.5 mg/kg and 12.0 mg/kg)
- GI = Gastrointestinal absorption, 100%
- EF = Exposure frequency, 365 days/year (Ingestion rate is annual daily average)
- ED = Exposure duration (30 years; national upper-bound time at one residence; USEPA, 1989b)
- CF = Conversion factor (0.001 kg fish/g fish)
- BW = Body weight (70 kg)
- AT = Averaging time(365 days/year x 70 years for carcinogenic effects; 365 days/year x 30 years for non-cancer effects)

Under these assumptions, annual average daily exposure over the assumed 30-year exposure duration is calculated to be 5.1×10^{-3} mg/kg-d, assuming current average concentrations of PCBs in fish of 12.0 ppm, and 6.4×10^{-4} mg/kg-d, using the estimated future average concentration of 1.5 ppm. Averaged over a 70-year lifetime, the chronic daily intake (CDI) corresponding to the two exposure concentrations is 2.2×10^{-3} mg/kg-d or 2.8×10^{-4} mg/kg-d, respectively. The average daily exposures and chronic daily intakes for this pathway are listed in Tables B.6-5 and B.6-6.

Snapping Turtles

Available literature indicates that snapping turtles exhibit an exceptional ability to bioaccumulate PCBs from their environment (Stone *et al.*, 1980). Since these animals may be consumed by some human populations, they can constitute a possible exposure pathway.

Neither recent data showing specific concentrations of PCBs in Hudson River turtles nor intake rates, as in turtle soup, are available. Therefore, no exposure calculation is made.

Ingestion of Agricultural or Home Garden Crops

Information available in the literature indicates that plants may become contaminated with PCBs. It has been established that some plants may become contaminated with PCBs by root uptake from contaminated soil or water or by uptake of volatilized PCBs from air (see B.3). It is possible that vegetables from vegetable gardens or farms in the vicinity of the Hudson River may contain PCBs and pose a route for human exposure. There are, however, no data on soil concentrations of PCBs in this study area for this computation. Water used for crops or backyard gardens is not believed to be a significant concern, given the levels of PCBs in the water. Thus, the data are insufficient to predict the amount of plant uptake of PCBs from soil or water. Other data indicate that air in the Hudson River area may contain PCBs and that uptake of PCBs from air could also occur, but a Hudson River source of the air-borne PCBs cannot be isolated from other sources.

There are many uncertainties in the estimation of human health risks related to the ingestion of PCB-contaminated vegetables. Uncertainties associated specifically with this pathway of ingestion include, first, that the background data from the area on the concentration of PCBs in crops, collected in the 1970s and early 1980s, are not necessarily representative of current conditions. Some reports indicate that the fruits of plants (corn and beans) contain lower concentrations of PCBs than the leaves by an order of 100-fold (Shane and Bush, 1989). In addition, different species of plants may be more efficient in eliminating PCBs congeners (Shane and Bush, 1989). PCB concentrations in plants are declining over time (Buckley, 1983). For example, the levels of PCBs measured in aspen decreased about seven-fold over the period of 1978 to 1981, while sumac PCB concentrations declined about two-fold (Buckley and Tofflemire, 1983).

In the absence of data demonstrating that the Hudson River site contributes to the concentration of PCBs in local crop species, any quantitative estimate of exposures from this pathway would be speculative and, therefore, have not been made.

Ingestion of Beef or Dairy Products

Should locally raised livestock consume PCB-contaminated feedstock or water, they would tend to accumulate the PCBs in their adipose tissue and could present a pathway for human exposure, if the meat or milk from these animals were consumed. In the Hudson River area, water fed to animals is, for the most part, from streams or wells on individual properties, which are not known to be contaminated with PCBs. In addition, no data regarding the concentration of PCBs in soils in the vicinity are available.

Many uncertainties exist in the analysis of risks related to these exposure pathways. As described above, PCB concentrations in plants are apparently declining over time; thus, the use of available data (concentrations measured in 1979) is likely to overestimate potential exposures. Also lacking are data on the proportionate consumption of supplemental feed versus feed grown locally, especially in the case of dairy cows, where the amount of supplemental feed is probably significant (Fries, 1982).

Breast Milk

Because human breast milk is high in lipid content, lipophilic PCBs may also be present in breast milk and constitute a source of exposure for a nursing infant. Some researchers (Wolff, 1983; Yakushiji, 1978; Brilliant, 1978) have suggested that the concentrations of organochlorine residues (such as PCBs) in breast milk are close to those theoretically predicted by diffusion directly from adipose tissue. That is, the level in milk fat is practically the same as that in adipose tissue, with a ratio of adipose concentration to milk fat concentration of approximately 1.1:1. This transfer of contaminants from adipose tissue to milk fat, which occurs indirectly via the blood, is very efficient

because blood flow to both mammary tissue and adipose tissue greatly exceeds milk production. Therefore, adipose and milk fat attain equilibrium concentrations of dissolved organochlorine residues (Wolff, 1983).

Studies conducted on fish-consuming populations have specifically correlated fish consumption with PCB levels in human breast milk (Schwartz, 1983) or in human adipose tissue (Rogan, 1986), and indicate that breast milk from women who regularly consume fish from contaminated waters may contain PCBs at an average concentration as high as 1.8 µg PCB/g milk fat (54 ng/g whole milk) (Rogan, 1986). There is limited evidence that the specific PCB congeners that bioaccumulate in fish may be detected at elevated levels in the breast milk of women who eat those fish (Bush, 1991).

One study, conducted on breast milk from residents of two communities of Upstate New York, found that there was no difference in the congener profile of breast milk collected from a population of Oswego county (near the Great Lakes region) than in breast milk collected from Albany (along the Hudson River), a finding that makes determination of the source of exposure difficult to distinguish. This study also indicates that the mean concentration of PCBs in breast milk among women resident in Upstate New York is in the range of 26.5 ng/g (whole milk), a concentration that could result in a young child approaching an unsafe level of intake (Cordle, 1982).

No studies have been conducted specifically to assess the PCB concentrations present in breast milk among the residents along the Hudson River. Similarly, there are no known human biomonitoring data for other tissue types for this population that might allow extrapolation to or modeling of breast milk concentrations. Consequently, it is impossible to provide an adequate evaluation of the potential risks that might be presented to nursing infants via this route of exposure.

A NYSDOH study is characterizing the PCB level in breast milk and other human tissues among residents of the St. Regis Reservation in Massena, NY (Bush, 1991), because of concerns relating to exposure from the St. Lawrence River.

Human biological samples are being collected from this population as well as populations along the Great Lakes and control populations from Warren, Schoharie, Rensselaer, and Oswego counties. Information from this study could provide useful information regarding the background concentrations of PCBs in human tissues as well as PCB concentrations in other populations of concern, such as residents of the Indian reservation and residents of northern New York state. Results of this NYSDOH study are expected to be released in the summer or fall of 1991.

Drinking Water

Data regarding the concentrations of PCBs in water from the Upper Hudson River are available for five locations: Fort Edward, Fort Miller, Schuylerville, Stillwater, and Waterford. The concentrations of PCBs in these water samples have been consistently low. Specifically, based on data collected from 1986 to 1989, the adjusted mean concentrations from these different locations does not exceed 0.05 ppb. The highest concentrations were detected in Fort Edward, where the upper 95th percent confidence interval on the adjusted mean is 0.06 ppb; almost one full order of magnitude below the current Maximum Contaminant Level (MCL) of 0.5 ppb (see B.6.3.5). Consistently, the lowest measured levels were found at the Waterford sampling station, where the upper 95th percent confidence interval on the adjusted mean is 0.034 ppb.

The only populations between Fort Edward and Poughkeepsie known to use the Hudson River as a source of municipal water are the populations of Waterford and Halfmoon, which are served by the Waterford Water Works (Berger, personal communication). Recently, an extensive study of the quality of the water at this intake found that the PCB concentration in Hudson River water and treated water has been below the 0.1 ppb detection limit since September 1983 (Metcalf and Eddy, 1990). Thus, these measurements are below the federal MCL of 0.5 ppb and the NYSDOH action level of 0.1 ppb.

The USGS monitoring station at Waterford has detected PCBs above their 0.01 ppb detection limit. In recent years, the concentration of PCBs measured by USGS at this station has averaged on the order of 0.03 ppb. Despite the consistently low levels of PCBs in water samples collected in the recent past and the fact that PCB concentrations fall below regulatory limits for PCBs in water, an initial screening level exposure and risk estimate was calculated for drinking water. In this screening step, the exposure scenario utilized the upper 95 percent confidence limit on the mean concentration from recent (1986-1989) river water samples at Roger's Island as the exposure concentration (0.06 mg/l). The exposure evaluation assumes a daily intake of 2 liters of water for 30 years. Exposure by a 70 kg adult was assumed. The exposed population was assumed to drink untreated water from the Fort Edward area, where the highest concentrations of PCBs were consistently detected. PCBs taken in via this pathway are assumed to be 100 percent absorbed. All the exposure assumptions employed in evaluating exposure through drinking water ingestion are consistent with EPA guidance on evaluating exposures using reasonable maximum assumptions.

The annual average daily exposure to PCBs from this exposure scenario is estimated to be 1.7×10^{-6} (mg/kg-day) for adults. The chronic daily intake over a 70-year lifetime is calculated to be 7.3×10^{-7} (mg/kg-day). Any drinking water exposures that may occur are likely to be lower than the screening level exposures presented here due to water treatment.

B.6.2.3 Inhalation Exposures

Exposure from Air

Air monitoring (see B.3) for PCBs has been conducted in the Upper Hudson River study area from the late 1970s to as recently as 1989. None of the sampling efforts to date, however, has adequately characterized the contribution of the Hudson River to PCBs in ambient air of resident populations. The current database raises concerns about the possibility of inhalation exposure to PCBs, but the inadequacy of the data proscribes any sound quantitative evaluation of inhalation exposure.

B.6.2.4 Recreational Exposures

Direct contact with contaminated sediments is a possible route of contaminant exposure, particularly where such sediments occur close to residential and recreational areas. Routes of exposure during a visit to a contaminated site include absorption following dermal contact with sediments, the incidental ingestion of sediments during subsequent hand to mouth contact and dermal contact with water. Under a recreational use scenario, considered the predominant use along the Hudson, such exposures are likely to occur occasionally during the course of a resident's life.

Dermal Absorption From Contact With Sediments

Individuals wading along the Hudson River may be exposed to PCBs by dermal contact with contaminated sediments. The dose of PCBs absorbed through the skin depends on many factors, including the skin surface area exposed to sediments, adherence of sediments to the skin, the frequency and duration of exposure, the concentration of PCBs in the sediments, and the amount of PCBs transferred from the sediment through the skin. The derivation of the values used for each of these exposure factors is provided below and summarized in Table B.6-2.

Contact with river sediments is most likely to occur during recreational activities. Because activity patterns change with the age of the population, exposure by young children (ages 1-6), older children and teenagers (ages 7-18), and adults (age 18 and above) were considered separately. It is assumed that infants under one year of age would not come into direct contact with river sediments. For children aged 1-6 and for adults, it is assumed that use of the river occurs approximately seven days per year (USEPA, 1988). For older children and teenagers, use of the river is estimated to occur 24 days per year, based on the assumption of use twice per week for the three summer months. Age-specific body weights are also used. The skin surface area available for contact with sediments depends on clothing. For children between the ages of 1-18, it is assumed that legs, feet, arms, and hands are available for sediment contact. These areas correspond to roughly 55 percent of the total body surface area for

children. For adult exposures, it is assumed that lower legs, feet, forearms, and hands are available for sediment contact. Following USEPA guidance (USEPA, 1989b), 50th percentile body surface areas were used to estimate dermal contact rates. The specific surface area values, found in the USEPA's Exposure Factors Handbook (USEPA, 1989a), are presented in Table B.6-2.

The rate of adherence of sediments to skin and the absorption of PCBs from the sediment across the skin are based on a review of the literature. The values of 1 mg/cm^2 and 3 percent are used for these parameters, respectively.

Data regarding soil-to-skin adherence factors (AF) are limited. USEPA (1989b) currently recommends default values of 1.45 mg/cm^2 (potting soil) and 2.77 mg/cm^2 (kaolin clay) based on experimental values for soil-related dust adherence reported by the Toxic Substance Control Commission of the State of Michigan (Harger, 1979). However, data presented by other investigators suggest that the average amount of soil adhering to human skin may be as much as 10-fold lower. AF values ranging between 0.2 and 0.7 mg/cm^2 have been reported by Lepow *et al.* (1975), Roels *et al.* (1980), Que Hee *et al.* (1985), and USEPA (1989c), wherein USEPA calculated an upper-bound (95th percentile) soil-to-skin adherence factor of 1.0 mg/cm^2. Based on data from these four studies, 1.0 mg/cm^2 was chosen to be the most appropriate AF for Hudson River sediments. It was selected because it is an upper-bound adherence value derived from studies of diverse soil types, a condition likely to be representative of areas surrounding the Hudson River, and is based on the most recent USEPA evaluation of this issue (USEPA, 1989c).

Data regarding absorption (AB) of PCBs through the skin are limited. PCB absorption (bioavailability) can be estimated by comparison to absorption values for 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD), since both chemicals have similar chemical properties (Oak Ridge National Laboratory, 1989). Based on a study of dermal absorption of TCDD by Poiger and Schlatter (1980) in rats, USEPA (Schaum, 1984) recommended using three percent as an upper-bound AB for PCBs in humans. This value is consistent with recent guidance from USEPA indicating dermal absorption rates of between one and ten percent and is supported by data

presented in other investigations (Shu, 1988; USEPA 1989c) together with the similarity of the chemical properties of TCDD and PCBs.

Because of the spatially and temporally disjointed nature of the available data on sediment concentrations of PCBs in the Hudson River, choosing a best exposure concentration for use in the risk assessment is difficult. The most recent data for surface sediments would appear to be reasonable, but few recent data are available. GE conducted a limited sampling effort in 1989 where three to eight cores from previously identified hotspots were obtained (see B.3). Their results indicate depth-averaged PCB levels of <1 ppm to 918 ppm. A review of the 1984 Thompson Island Pool survey, the most recent large-scale monitoring effort, reveals that approximately 80-85 percent of the samples in the Thompson Island Pool had PCB levels less than 100 ppm. The highest concentrations measured exceed 1,000 mg/kg in the Thompson Island Pool. Taking all surface (including all grab and top core section samples = 569 samples) sediment samples from the 1984 Thompson Island Pool survey, the 95 percent upper confidence bound on the mean is 66.2 mg/kg. This estimate provides a reasonable maximum exposure level for sediment-bound PCBs. Subsequent analyses in Phase 2 and 3 may focus attention on evaluating sediment exposures as a function of river reach and areas of particularly high PCB concentrations.

The dose of PCBs absorbed through the skin was calculated as:

$$\text{Absorbed Dose (mg/kg/day)} = \frac{CS \times CF \times SA \times AF \times AB \times EF \times ED}{BW \times AT}$$

where

- CS = PCB Concentration in Sediments (mg/kg)
- CF = Conversion Factor (10^{-6} kg/mg)
- SA = Skin Surface Area Available for Contact (3,931, 7,420 and 5,170 cm^2 for ages 1-6, 7-18 and adults, respectively)
- AF = Sediment-to-Skin Adherence Factor (1.0 mg/cm^2)
- AB = Absorption Factor (0.03)
- EF = Exposure Frequency (7 days/year for ages 1-6 and adults; 24 days/yr ages 7-18)
- ED = Exposure Duration (30 years)
- BW = Body Weight (15, 42 and 70 kg for ages 1-6, 7-18 and adults, respectively)
- AT = Averaging Time (70 years x 365 days/year).

The annual average daily exposure to PCBs from this pathway is estimated to be 1.0×10^{-5}, 2.3×10^{-5}, and 2.8×10^{-6} mg/kg-d for young children, older children, and adults, respectively. The chronic daily intake over a 70-year lifetime is calculated to be 5.3×10^{-6} mg/kg-d. These values are listed in Tables B.6-5 and B.6-6.

Incidental Ingestion of River Sediments

Contaminated sediments can adhere to the skin of individuals swimming or wading in the Hudson River. Subsequent hand-to-mouth activities, particularly prevalent in young children, will lead to the ingestion of adhered sediments and contaminating PCBs. The extent of human exposure that will result from sediment ingestion depends on the concentration of PCBs in sediment, the bioavailability of PCBs from the sediment, sediment ingestion rates, exposure frequency and duration, and body weight. The specific values used to calculate exposures via ingestion of contaminated sediment are presented in Table B.6-3.

All of the values used in calculating sediment ingestion exposures are discussed above for dermal exposure to sediment, with the exception of the sediment intake rate. Based on guidance from USEPA (Porter, 1989), daily sediment ingestion rates are 200 mg/day for children 1 to 6 years of age and 100 mg/day for individuals 7 or more years of age. In both cases, 100 percent of the PCBs in the sediments are assumed to be absorbed. Although the sediment ingestion rates are expressed as a daily rate, most soil and sediment ingestion is expected to occur during outdoor play. This exposure assessment assumes that the entire daily sediment intake occurs at the Hudson River.

The intake of PCBs due to the incidental ingestion of sediments was calculated as follows (USEPA, 1989b):

$$\text{Intake (mg/kg/day)} = \frac{CS \times CF \times IR \times EF \times ED}{BW \times AT}$$

where

CS = PCB Concentration in Sediments (mg/kg)
CF = Conversion Factor (10^{-6} kg/mg)
IR = Sediment Ingestion Rate (200 and 100 mg/day for ages 1-6 and >age 6, respectively)
EF = Exposure Frequency (7 days/yr for ages 1-6 and adults; 24 days/yr for ages 7-18)
ED = Exposure Duration (30 years)
BW = Body Weight (15, 42 and 70 kg for ages 1-6, 7-18 and adults, respectively)
AT = Averaging Time (70 years x 365 days/year).

Annual average daily exposure to PCBs resulting from the ingestion of sediments is calculated to be 1.7 x 10^{-6} mg/kg-d, 1.0 x 10^{-6} mg/kg-d, and 1.8 x 10^{-6} mg/kg-d for young children, older children, and adults, respectively. Over the assumed 30-year duration of residence near the Hudson River and a 70-year lifetime, the chronic daily intake (CDI) of PCBs from sediments is calculated to be 3.5 x 10^{-6} mg/kg-d. These values are listed in Tables B.6-5 and B.6-6.

Dermal Absorption from Water

Another possible exposure pathway is dermal absorption of PCBs directly from water while swimming or wading in the Hudson River. The amount of exposure (dose) from swimming activities is dependent on the concentration of PCBs in the water, the frequency and duration of human contact with the water, the skin surface area available for contact, the amount of PCBs that cross the skin (permeability) and the body weight of the exposed individual. The values used for these exposure conditions are listed in Table B.6-4 and addressed below.

Exposures from swimming are assumed to occur during recreational use of the river as described for exposure from dermal contact and ingestion of sediment, except that the entire surface of the body is assumed to contact the water. The values used for bodyweight, exposure frequency and exposure duration are age-specific and are the same as those described above. The skin permeability constant of 3.2 x 10^{-2} cm/hr is derived from the literature. The duration of a swimming event is assumed to be 2.6 hours per day (USEPA, 1989a). The dermal

permeability constant for PCBs is unknown, but is estimated to be 3.2×10^{-2} cm/hr based on the considerations described below.

- Dermal permeability has been found to be well described by Fick's first law, which states that the steady state flux across the skin is proportional to the concentration gradient, the permeability coefficient, and the reciprocal of skin thickness (Scheuplein, 1977). The concentration of permeating chemicals within the body is conservatively assumed to be zero so that the full external concentration of the chemical contaminant appears as the concentration gradient. Fick's second law predicts that a lag time ranging from minutes to hours occurs before the flux through the skin builds up to its steady state value. To be health protective in calculations, it is assumed that the full steady state flux exists instantly at the initiation of expsoure.

- The permeability coefficient is a crucial factor controlling uptake of chemicals via dermal exposure. For pure liquids of a given family (*e.g.*, the alcohols, methanol, ethanol, propanol, and decanol), the permeability coefficient goes down with increasing molecular weight. For chemicals presented in solution, the characteristics of solute, solvent and skin are all important in determining the magnitude of the permeability coefficient. The stratum corneum is lipophilic (*i.e.*, hydrophobic), so that, for chemicals presented in aqueous solution, it is experimentally found that over a range of octanol-water partition coefficients (K_{ow}'s), the chemical-specific dermal permeability coefficient (K_p), is directly related to that chemical's K_{ow} (Flynn, 1990).

- Hawker and Connell (1988) have reported that PCBs have log K_{ow} values ranging between 4.5 (for monochlorobiphenyls; MW=189) and 8.2 (for decachlorobiphenyl; MW = 499). Flynn (1990), who has tabulated and modeled the permeability coefficients of a wide variety of substances, predicts a value of $K_p = 1.6 \times 10^{-2}$ cm/hr for all PCBs. Flynn also suggests that the fitted curve be given a bias moving it upward in permeability so that 85 percent of all the values fall below it in order to derive an upper bound estimate of dermal permeability. This process assigns a value of $K_p = 3.2 \times 10^{-2}$ to the skin permeability coefficient for PCBs. This value is unlikely to underestimate the health risks posed by PCBs in water and, hence, $K_p = 3.2 \times 10^{-2}$ cm/hr is used as the PCB-specific dermal permeability constant for the calculations below. (This value is within the range of those values presented in USEPA's recently released Interim Guidance for Dermal Exposure Assessment.)

Concentrations of PCBs in Hudson River water have been evaluated and discussed in Section B.3. For reasons described in that section and to be consistent with current USEPA guidance on determination of exposure concentration (USEPA 1989b), this exposure assessment uses the 95th percent confidence limit value of the adjusted log normal maximum likelihood estimate of the mean value. Since the concentration of PCBs in water at Fort Edward is consistently higher than for other sampled locations, data from that location were selected for use in the exposure assessment. Specifically, the exposure concentration of 0.06 µg/l is incorporated into the exposure calculations.

The dose of PCBs absorbed through the skin from direct contact with Hudson River water is calculated as follows (USEPA, 1989b):

$$\text{Absorbed Dose (mg/kg/day)} = \frac{CW \times CF \times SA \times K_p \times DE \times EF \times ED}{BW \times AT}$$

where

CW = PCB Concentration in Water (0.06 ug/l)
CF = Conversion Factor (10^{-3} l/cm^3)
SA = Skin Surface Area Available for Contact (100%, or 6,880, 13,100 and 18,150 cm^2 for ages 1-6, 7-18 and adults, respectively)
K_p = Chemical-Specific Dermal Permeability Constant (3.2×10^{-2} cm/hr)
DE = Duration of Event (2.6 hr/day)
EF = Exposure Frequency (7 days/year for ages 1-6 and adults; 24 days/year for ages 7-18)
ED = Exposure Duration (30 years)
BW = Body Weight (15, 42, and 70 kg for ages 1-6, 7-18 and adults, respectively)
AT = Averaging Time (70 years x 365 days/year).

Under these assumptions, annual average daily exposure to PCBs resulting from dermal absorption is calculated to be 4.4×10^{-6} mg/kg-d, 1.0×10^{-7} mg/kg-d, and 2.5×10^{-6} mg/kg-d for young children, older children, and adults, respectively. Over the assumed 30-year duration of residence near the Hudson River and a 70 year lifetime, the chronic daily intake (CDI) of PCBs from sediments is calculated to be 2.6×10^{-6} mg/kg-d. These values are listed in Tables B.6-5 and B.6-6.

B.6.3 Toxicity Assessment

B.6.3.1 Introduction

PCBs generally have low acute toxicity but are of public health concern due to their persistence in the environment, the potential to bioaccumulate in animal and human tissues, and their potential for chronic or delayed toxicity. The major target organs of PCB toxicity are the liver and the skin. Occupational exposures to relatively high concentrations of PCBs have resulted in changes in serum levels of liver enzymes and skin effects such as chloracne (ATSDR, 1987). In individuals who accidentally consumed PCBs in contaminated rice oil in Japan (Yusho patients), routine liver function tests were abnormal. PCBs have also been shown to cause some developmental effects and neurological effects in Yusho patients, occupationally exposed individuals, and in individuals exposed via the consumption of contaminated fish.

USEPA has developed several sets of toxicity values to provide quantitative estimates of the potency of chemicals and resultant toxic effects. The reference dose (RfD) and the cancer slope factor (CSF) are the toxicological values of relevance for this assessment. The RfD and the CSF are fundamentally different in their assumptions of the relationship between dose and response. For carcinogenic effects, it is assumed that there is no threshold below which no effect will occur. Some risk, however small, is associated with every level of exposure. In contrast, RFDs for non-carcinogenic effects assume that there is a threshold dose below which there will be no deleterious effect.

Verified RfDs and CSFs are available on USEPA's Integrated Risk Information System (IRIS). These toxicity values and other health risk assessment information are included in IRIS after a comprehensive review of chronic toxicity data by work groups of USEPA scientists. Verified RfDs and CSFs are considered to be the most reliable basis for estimating noncarcinogenic and carcinogenic risks resulting from chronic chemical exposures. If toxicity values are not available for the chemicals of concern in IRIS, EPA's secondary source known as the Health Effects Assessment Summary Tables (HEAST) may be reviewed. This is

a quarterly review of toxicity values for carcinogenic or noncarcinogenic effects. If no toxicity values are available in HEAST, guidance documents published by USEPA's Environmental Criteria and Assessment Office (ECAO) may provide useful toxicity information.

Because PCBs comprise a class of chemicals, but are largely considered to be one compound as far as regulatory approaches to their evaluation, additional discussion regarding the characteristics of the diverse class of compounds, and the possible impact on regulation is also provided.

B.6.3.2 Noncarcinogenic Effects

Reference Doses (RfDs) provide a benchmark for the daily dose (with a confidence level of an order of magnitude) to which humans, including sensitive populations such as children or pregnant women, may be subjected without an appreciable risk of deleterious effects during a lifetime of exposures. The same unit system commonly used for dose (mg chemical/kg body weight - day) is also used for RfDs.

The basis of an RfD calculation is usually the highest dose level that did not cause observable adverse effects (*i.e.*, the No Observed Adverse Effect Level, or NOAEL) after chronic exposure in animal experiments. The NOAEL is then divided by uncertainty (safety) factors and, occasionally, an additional modifying factor to obtain the RfD. In general, the uncertainty factor is determined by multiplying by a factor of 10 to account for interspecies variation and a factor of 10 to account for sensitive human populations. Additional factors of 10 are included in the uncertainty factor, when the RfD is based on the Lowest Observed Adverse Effect level (LOAEL) instead of the NOAEL, if an experiment was conducted over less than a lifetime of exposure, or if there are inadequacies in the database.

For PCBs, an oral RfD of 1×10^{-4} mg/kg/day for Aroclor 1016 was developed in 1987. Other RfDs for other Aroclor mixtures are not available, and thus it is assumed that the same RfD applies to all PCB mixtures. The basis of the RfD

was a study in rhesus monkeys where offspring of a group exposed to 1 ppm PCBs in food were significantly smaller than the controls. The NOAEL in this study was 0.25 ppm in the diet, equivalent to a dose of 0.0105 mg/kg-d. An uncertainty factor of 100 was applied to this NOAEL to generate an RfD of 1×10^{-4} mg/kg-d.

The most recent consultation of the IRIS database, in which PCBs were updated as of January 1990, revealed that no oral RfD for PCBs currently exists. USEPA is in the process of reviewing the RfD for PCBs. Recent conversations with and publications by USEPA staff (Dourson, personal communication; Clark, personal communication; Dourson and Clark, 1990) indicate that this RfD of $1x10^{-4}$ mg/kg-d is being considered for use by some USEPA regional offices and is under consideration for adoption by the Environmental Criteria Assessment Office of USEPA.

Other, more restrictive, RfDs for PCBs are currently being proposed by state agencies. For example, based on research conducted by Fein (1984), which correlates consumption of PCB-contaminated fish with lower birth weight, smaller head circumference, shorter gestational age, and poorer neuromuscular maturity in infants, the Minnesota Department of Health has proposed an RfD for human reproductive effects of 5×10^{-5} mg/kg-d (Shubat, 1991). Researchers in human development have suggested a threshold for the developmental effects of PCBs in the range of 1 to 3.4 ppm PCBs in breast milk (fat basis). Application of pharmacokinetics modeling to these threshold values indicates a threshold dose for a 60 kg woman on the order of 3×10^{-4} to 1×10^{-3} mg/kg-d. Incorporating a 10-fold uncertainty factor for intra-species variability suggests an RfD based on human studies in the range of 3×10^{-5} to 1×10^{-4} (Shubat, 1991).

There is some concern that an RfD based on studies of Aroclor 1016 in monkeys may underestimate the risks associated with human exposure to PCBs. Not only do human studies indicate a greater possible sensitivity in humans than animals, but Aroclor 1016 is also one of the less highly chlorinated PCB mixtures generated commercially. Until the RfDs based on human studies can undergo verification by USEPA, it is anticipated that the RfD of 1×10^{-4} would provide an adequate indication of potential non-cancer toxicity, at least within an order

of magnitude range (Clark, personal communication). Consequently, this RfD is used here to evaluate non-carcinogenic risks from PCB exposure.

B.6.3.3 Carcinogenic Effects

Definition

PCBs have been classified by USEPA as B2 carcinogens, or probable human carcinogens. The carcinogen classification system is based on the strength of the evidence that a compound is a human carcinogen. Compounds for which there is sufficient human evidence of carcinogenicity are in Class A. Chemicals for which there is limited human evidence and sufficient evidence of carcinogenicity in animals are in Class B1; those for which there is inadequate evidence in humans, but sufficient evidence in animals are in Class B2. Chemicals in classes B1 and B2 are considered to be probable human carcinogens.

USEPA's Carcinogen Assessment Group (CAG), which reviews human, animal, and *in vitro* data on suspected chemical carcinogens, has calculated cancer slope factors (CSFs) for those determined to be carcinogenic. CSFs are used to estimate the excess cancer risk due to continuous exposure to a chemical throughout the course of a 70-year lifetime. CSFs are based on data from lifetime animal bioassays, although human data are used when available. Since the evidence of carcinogenicity from human studies is inconclusive for PCBs, the CSF is based on studies conducted in animals. For animal data, USEPA uses a mathematical, linearized multistage model to extrapolate from the high doses used in the bioassay to the low doses expected to result from environmental exposure. The CSF represents the upper 95 percent confidence limit of the slope of the linear portion of the dose-response curve. Excess carcinogenic risk for the experimental animal is extrapolated from this slope to the excess carcinogenic risk expected for humans.

USEPA Cancer Slope Factor

USEPA has determined that the cancer slope factor (CSF) for all PCB mixtures in humans is 7.7 $(mg/kg\text{-}day)^{-1}$. This CSF is based on a study by Norback and Weltman (1985) in which rats (70 male and 70 female) were fed a diet containing Aroclor 1260 mixed in corn oil for two years (100 ppm for 16 months and 50 ppm for 8 months) followed by a standard diet alone (0 ppm) for another five months. Controls were fed a diet containing corn oil for 18 months followed by a standard diet alone for another five months. Among animals surviving more than 18 months, the incidence of hepatocellular carcinomas in females and males was 91 percent (43/47) and 4 percent (2/46) respectively; the incidence in corresponding controls was 0 percent. Among these survivors, an additional 9 percent of the females (4/47) and 11 percent of the males (5/46) had neoplastic nodules; the incidence of total neoplasms in the corresponding controls was 2 percent (1/49) and 0 percent (0/32) for females and males, respectively.

During the course of this study, concurrent liver morphology studies were performed on tissue samples obtained by partial hepatectomies of exposed rats (3 male and 3 female) at eight time points. These studies demonstrated the sequential progression of liver lesions to hepatocellular carcinomas. This progression from nodules to benign tumors to carcinomas has been used by USEPA to justify its calculation of CSFs on the basis of both malignant tumors <u>plus</u> benign tumors and nodules. Formerly, CSFs were determined on the basis of malignant tumors alone and thus were lower.

USEPA views the Norback and Weltman study as a positive study since: (a) it used an adequate number of animals (≥ 50/group); (b) it spanned the natural life span of the rat; and (c) Sprague-Dawley rats have a low incidence of spontaneous hepatocellular neoplasms. Confidence in this study is apparently enhanced by the fact that the current CSF for Aroclor 1260 is similar to the old value of 3.9 $(mg/kg\text{-}day)^{-1}$, based on data from an earlier study (Kimbrough, 1975).

USEPA views the CSF for Aroclor 1260 as representative of all other PCB mixtures. First, there is no information about which Aroclor congeners are carcinogenic and, second, the Norback Weltman study using Aroclor 1260 is superior to all other studies of PCB mixtures (USEPA, 1988). Although this may be a conservative assumption (USEPA, 1990), USEPA believes that given the data available, the public health is best protected by assuming that Aroclor 1260 is representative of all other mixtures (Mukerjee, personal communication).

In this assessment, USEPA protocol is used and the evaluation of carcinogenic risk uses the CSF of 7.7 $(mg/kg\text{-}d)^{-1}$.

Other Cancer Slope Factor Studies

USEPA headquarters has calculated a CSF for Aroclor 1254 of 2.6 $(mg/kg\text{-}day)^{-1}$ (USEPA, 1988) in contrast to the value of 7.7 $(mg/kg\text{-}d)^{-1}$ calculated for Aroclor 1260. This value was based on a 1978 National Cancer Institute (NCI) study in which statistically significant, dose-related increases in hepatic neoplastic nodules and carcinomas were seen in female (Fischer 344) rats fed a diet containing Aroclor 1254. The separate CSF for Aroclor 1254 has not been promulgated by headquarters due to uncertainties in the underlying study and so is not used in the current assessment.

In establishing the CSF for Aroclor 1260, USEPA (1988a) chose to exclude a rat study by Schaffer *et al.* (1984). That study evaluated the carcinogenicity of Clophens 60 and 30, commercial PCB mixtures with a congener profiles similar to Aroclors 1260 and 1016 respectively (Schaeffer, 1988; ATSDR, 1987).

USEPA excluded this study because there was a discrepancy in the published tables, which prevented a determination of the number of animals at risk, and the study used male (Wistar) rats. Norback and Weltman showed that female (Sprague-Dawley) rats are more sensitive than males. Although these objections have merit, the study by Schaeffer, in many ways, resembles that of Norback and Weltman. Both used an adequate number of animals (≥ 50/group), were of similar duration (lifespan of the rat), used rats with a low incidence of spontaneous

hepatocellular neoplasms (less than 10 percent), and used doses which did not increase mortality, as shown below.

	NORBACK & WELTMAN	SCHAEFFER *ET AL*
Strain of male rat	Sprague-Dawley	Wistar
Exposure route (vehicle)	Oral (diet)	Oral (diet)
Tumor site (type)	Liver (carcinoma and neoplastic nodules)	Liver (carcinoma and neoplastic nodules)
Aroclor/Clophen	none 1260	none 30 60
Nominal dose in feed (ppm)	0 100	0 100 100
TWA-D_R (mg/kg/day)	0 3.45	0 5.0 5.0
D_M (mg/kg/day)	0 0.59	0 0.85 0.85
Tumor incidence	0/32 7/46	6/122 42/130 122/125
Tumor incidence (percent)	0 15	5 32 98
Exposure duration	720 days (1/2 dose days 480 to 720)	832 days
Observation period	days 540 to 870	days 501 to 832
Observation duration	330 days	331 days
Study duration	870 days	832 days

If females are indeed the most sensitive sex, the data of Schaeffer *et al.* (1984) on male rats could not be used to determine CSFs for PCBs. However, a comparison of tumor incidence between rats exposed to Clophens 30 and 60 does indicate that less chlorinated Aroclors may also be less carcinogenic. The CSF for Aroclor 1016 could be estimated given the relative change in CSFs between Clophens 60 and 30 as well as the CSF for Aroclor 1260.

B.6.3.4 Toxicity of Specific PCB Congeners

Few congener-specific analysis of PCBs in environmental samples from the Hudson River have been performed. Because of the different chemical characteristics of various PCB congeners, different congeners will partition differently throughout environmental media. Therefore, the congeners to which humans may be exposed may differ from the congeners present in the original source of commercial PCB formulations and may also differ from one exposure route to another.

In December 1990, USEPA hosted a workshop to evaluate the possibility of shifting the toxicity assessment of PCBs from an Aroclor-specific approach to a congener-specific approach, which would permit a more specific toxicity assessment for Aroclor mixtures as well as for environmental mixtures of PCBs. The consensus was that information is currently insufficient to develop a Toxic Equivalency Factor (TEF) approach for PCBs similar to that used for polychlorinated dibenzo-dioxins and -furans. Some of the issues related to toxicity are listed below.

- PCBs do not induce one set of toxic endpoints via the same mechanism of action. To the contrary, it appears that various structural classes of PCBs may exert a number of different toxic effects via different mechanisms. As a result, a TEF scheme would have to be developed for each mechanism which produces an effect of concern.
- Some PCBs (substituted in both the para and one meta position of both rings) produce effects similar to dioxin and act via the same mechanism. A proposed TEF scheme for these dioxin-like effects ranks the toxicity of these PCBs relative to dioxin, based on the ability of each congener to induce aryl hydrocarbon hydroxylase (AHH) activity or exert other dioxin-like effects.
- Coplanar PCBs (meta and para substituted but not ortho substituted) are the most similar to dioxin and are given the highest TEF values for dioxin-like effects.
- Increasing ortho-substitution of these PCBs decreases their ability to induce the AHH enzyme system and decreases their similarity to dioxin. Such PCBs are given lower TEF values.

- Some PCBs seem to promote cancer via a mechanism different than the dioxin-like AHH enzyme induction. The current understanding is that PCBs substituted in both the para and at least one meta position on each ring and having some degree of ortho substitution are potent carcinogen promoters by inhibiting intercellular communication or stimulating cell proliferation. The carcinogenic potential of most commercial PCB mixtures is probably attributed to this mechanism. No TEF approach has yet been proposed for these effects.

- PCB metabolites, such as arene oxides, may exert still different toxic effects via different mechanisms.

- Recent studies have shown that lightly chlorinated ortho-substituted PCB congeners may be associated with developmental neurotoxicity, e.g. statistically significant decrease in scores on psychomotor development tests associated with exposure of human infants to PCBs *in utero* or via breast milk. Lightly chlorinated congeners were found to accumulate in the central nervous system of monkeys, and to reduce cellular dopamine concentrations. For humans, this neurological effect endpoint may be more sensitive than the cancer endpoint for PCBs, which do not seem to be complete carcinogens.

- Recent findings indicate that there may be reproductive effects, e.g., sperm motility or estrogenic effects, to which humans may be very sensitive.

B.6.3.5 Epidemiological Studies

Epidemiological studies of accidental or occupational exposures to PCBs are few and for the most part inconclusive, because of small study size or low incidence rates for the endpoints being evaluated. Several epidemiological studies do point to an association between exposure to PCBs and some forms of cancer and other adverse effects. Most of the studies are inconclusive, either because they lack statistical power or because exposures are often not to PCBs alone. Nevertheless, findings are usually consistent with those from animal research.

Some of these studies contain enough information to estimate exposure or dose. In subsequent phases of the study process, further examination of the epidemiological studies may provide insights into the human dose-response relationships for PCBs.

Cancer Effects

Table B.6-7 outlines the findings of some epidemiological studies that have examined cancer mortality in PCB-exposed populations. Malignant melanoma, brain cancer, pancreatic cancer, rectal cancer, liver cancer, gall bladder and biliary tract cancer, hematologic neoplasms, gastrointestinal tract cancer, and stomach cancer have each been associated with PCB exposure in at least one study. Reports of liver cancer are most common, most statistically significant, and most consistent with the results of animal studies. As these studies vary in quality, a more intensive critique of their study designs is necessary before considering the significance of the results.

Non-Cancer Effects

Table B.6-8 summarizes the results of some recent studies that have examined the non-cancer effects of PCB exposures in humans. It is well documented from studies of workplace or environmental exposures that high levels of PCB exposure causes chloracne and dermal effects. Workers exposed to high levels of PCBs have also complained of lassitude, loss of appetite, loss of libido, burning of eyes, nose and throat, nausea, and dizziness. Associations have been made between PCB exposure and increased blood pressure, liver injury and abnormal liver function, and reduced gestational periods. People exposed to PCBs, along with chlorinated furans and quaterphenyls in the Yusho and Yucheng poisonings reported dermal effects, itching, swelling, eye discharge, jaundice, weakness, numbness, fever, and hyperemic conjunctivae. In many of the patients, respiratory symptoms persisted and many have chronically infected airways. Delayed or impaired immune response has been found in some of the poisoning victims (Kimbrough, 1987).

Even for studies that demonstrate a statistically significant association between PCB exposure and a toxic endpoint, the results must be interpreted with caution since the studies are generally unable to control for exposures to other chemicals or to the dioxins and furans which are common contaminants of commercial PCBs.

B.6.3.6 Other Health-Based Regulatory Limits or Guidelines

Reviewed below are several of the established regulations and guidelines for PCBs in various media, which were developed with considerations of the health effects of PCBs.

FDA: Tolerance for PCBs in Fish

The Food and Drug Administration (FDA) promulgated a regulation lowering the tolerance level for PCBs in fish destined for interstate commerce from 5 ppm to 2 ppm (44 FR 38330, 1979). This regulation, proposed in 1977 and promulgated in 1979, was stayed in the courts and did not become effective until 1984. The tolerance level was based on weighing the results of a risk assessment against the magnitude of potential food loss resulting from a lowered tolerance level. It is important to point out that the methodology of the FDA risk assessment precludes application of its results to the Hudson River risk assessment for fish ingestion. The FDA tolerance was not developed using methodology consistent with current USEPA guidance for risk assessment. Additionally, the FDA specifically states that this tolerance is intended to apply to fish entering interstate commerce and that this level may not be protective for locally caught fish from contaminated areas.

To arrive at a tolerance of 2 ppm, the FDA considered national per capita fish consumption, looking at the general distribution of PCB levels in fish for sale across the U.S. The FDA risk assessment was performed by assuming that with a tolerance level of 2 ppm this level would be the maximum concentration in fish encountered by the heavy fish consumer and that PCB concentrations in fish consumed would be distributed below 2 ppm in a manner reflecting a mix of fish from diverse sources (Cordle, 1982). The tolerance is not based on the assumption that all fish consumed contains 2 ppm PCBs. Because the distribution of PCB concentrations in fish caught in the Hudson River by local anglers is likely to be different from the distribution of PCB concentrations in fish for sale across the United States, the risk associated with regularly eating Hudson

River fish will differ from the risks associated with the FDA assessment for a 2 ppm tolerance, even if Hudson River fish do not exceed 2 ppm PCBs.

The FDA has derived a cancer slope factor of 0.34 $(mg/kg\text{-}day)^{-1}$ from the NCI bioassay using Aroclor 1254. A USEPA derivation of a cancer slope factor based on the NCI bioassay calculated a slope factor of 2.6 $(mg/kg\text{-}day)^{-1}$ (USEPA, 1988). The reasons for the difference cannot be evaluated without additional details on the FDA methodology.

USEPA: Drinking Water

USEPA used a cancer slope factor of 7.7 $(mg/kg\text{-}day)^{-1}$ to estimate that ingestion of drinking water containing 0.5, 0.05, and 0.005 µg/l PCBs corresponds to increased cancer risks of 10^{-4}, 10^{-5}, and 10^{-6}, respectively. The CSF used in this determination is consistent with the most recent value recommended by USEPA and is based on the study by Norback and Weltman.

The Maximum Contaminant Level (MCL) promulgated by USEPA for drinking water is 0.5 µg/l PCBs, which corresponds to a lifetime risk of 10^{-4} assuming lifetime ingestion of 2 liters of water per day, and a CSF of 7.7 $(mg/kg\text{-}day)^{-1}$. The MCL is set equal to the practical quantitation level for PCBs, which USEPA has determined reflects the level that can be measured by good laboratories under normal operating conditions within specified limits of precision and accuracy (54 FR 22062, 1989).

USEPA: Ambient Water

USEPA has issued ambient water quality criteria for PCBs of 7.9×10^{-5} µg/l, 7.9×10^{-6} µg/l, and 7.9×10^{-7} µg/l corresponding to lifetime risks of 10^{-5}, 10^{-6}, and 10^{-7}, based on the ingestion of fish and shellfish and ingestion of drinking water. The risks are primarily attributable to ingestion of fish and remain constant whether ingestion of drinking water is considered or not. These values were derived using a previous CSF of 4.34 $(mg/kg\text{-}day)^{-1}$. Ambient water quality criteria for consumption of fish and shellfish assumes a fish consumption

rate of 6.5 g/day, a 3 percent average lipid content for the edible portion of fish and shellfish consumed in the United States, and a bioconcentration factor for fish of 10,385 per percent lipid in the fish.

New York State: Ambient Water

New York State has issued ambient PCB water criteria for surface waters. These criteria are 0.001 µg/l for waters where ingestion of fish and shellfish may occur and 0.01 µg/l for waters where ingestion of drinking water is the only source of human exposure. The NYS 0.001 µg/l ambient water quality criteria is slightly higher than the USEPA-derived ambient water quality criteria (0.00079 µg/l as described above), which corresponds to a lifetime cancer risk of 10^{-5} based on consumption of fish.

USEPA Advisory Levels for PCB Superfund Cleanup

In 1990, USEPA's Office of Solid Waste and Emergency Response issued a guidance document containing preliminary cleanup goals, or action levels, for various media at Superfund sites with PCB contamination. These concentrations represent "the level above which unrestricted exposure may result in risks exceeding protective levels." The guidance establishes the MCL (0.5 ug/L) as an action level for groundwater and an action level of 19 ug PCBs per gram of organic carbon for freshwater sediments (USEPA, 1990). It should be noted that action levels are not cleanup levels.

Guidelines for PCB cleanup in sediments are currently being reviewed by USEPA. A proposed action level for freshwater sediments (19 ug PCBs per gram organic carbon) is based on a USEPA ambient water quality criteria of 0.014 ug/L for protection of aquatic life (OSWER, 1990). This is derived through the use of an equilibrium partitioning approach which relates the PCB concentration in sediment to that in the interstitial pore water of the sediment. The 1990 OSWER guidance states that where sediment values exceed the listed action levels, a monitoring program of indigenous biota should be instituted. This guidance does

not derive an action level for PCBs in freshwater sediments based on human consumption of contaminated fish.

National Academy of Sciences: Suggested No-Adverse-Response Level

The Safe Drinking Water Committee calculated Suggested No-Adverse-Response Levels (SNARLs) for 24-hour and 7-day exposures to PCBs in drinking water in 1977 and updated them in 1980. The SNARLs were derived by looking at induction of mixed-function oxidase enzymes in the liver of mammals as shown in studies by Grant (1974) and Bruckner (1977) in which rats were fed Aroclor 1254. Based on these studies, a no-adverse-response level was determined to be 1 mg/kg in animals; an uncertainty factor of 100 was applied. The resulting SNARL for PCBs is 0.35 mg/l water (0.01 mg/kg-d) for 24-hour exposures, and 0.05 mg/l for 7-day exposures (0.0014 mg/kg-d). The NAS determined that a reliable chronic SNARL could not be calculated for PCBs, because they are suspected carcinogens and represent a complex mixture of isomers and impurities having various biological activities and environmental fates.

Standards for Occupational Exposures

Both the Occupational Safety and Health Administration (OSHA) and the American Council of Government and Industrial Hygienists (ACGIH) have recommended 8-hour time-weighted average exposure limits for PCBs in air of 1 mg/m^3 for Aroclor 1242 and 0.5 mg/m^3 for Aroclor 1254 (ACGIH, 1986; CFR, 1990). These limits appear to be based on preventing skin irritation and chloracne, although the ACGIH documentation paper notes that the 1 mg/m^3 standard for Aroclor 1242 "will offer reasonably good protection against systemic intoxication, but may not guarantee complete freedom from chloracne." The OSHA levels are legally enforceable in the workplace. These exposure limits are not based on protecting workers from possible carcinogenicity. Assuming an inhalation rate of 10 m^3 per 8-hour work day and exposure to PCBs at the enforceable limit (1 mg/m^3) for 40 hours per week over 30 years, the resulting estimate of cancer risk is 3.4×10^{-1} from PCB exposure, based on USEPA's CSF of 7.7 $(mg/kg\text{-}day)^{-1}$.

The National Institute for Occupational Safety and Health (NIOSH) has recommended a 10-hour time-weighted average occupational exposure limit of 0.001 mg/m^3 or less (NIOSH, 1977). The NIOSH recommended exposure levels are guidelines and are not legally enforceable in the workplace.

New York State: Ambient Air Guidelines

The State of New York has issued guidelines for both short-term and long-term ambient air concentrations for PCBs. The short-term guideline concentration for ambient air is 0.5 mg/m^3 PCBs, which is based on and equivalent to the ACGIH Time Weighted Average-Threshold Limit Value (TWA-TLV) for exposure to Aroclor 1254. The annual ambient air guideline concentration for long-term exposures to PCBs in air is 1.19 ug/m^3 (NYSDEC, 1989). The TLV does not consider carcinogenic effects. The preamble to the TLV specifically states that these values are not to be applied to the general population and several investigators (Calabrese, 1988; USEPA Inhalation RfD documentation; Jarabek, 1990) have specifically cautioned against the use of the TLV for the general population.

B.6.4 Risk Characterization

B.6.4.1 Definition

The risk characterization step of the risk assessment process defines the potential threats to human health posed by the contaminants in the Hudson River. In characterizing and presenting such risks, carcinogenic and noncarcinogenic health risks are described for a reasonable maximum exposure scenario where possible.

In contrast to the quantitative assessment of carcinogenic health risk, assessment for non-cancer endpoints involves comparison of average daily exposure levels with established references doses (RfDs) to determine whether estimated exposures exceed recommended limits. Typically this comparison is expressed as a Hazard Quotient (HQ), which is the ratio of the estimated exposure to the RfD:

$$HQ = \frac{\text{Daily Exposure}}{\text{RfD}}$$

When HQ exceeds unity, unacceptable exposures may be occurring. Both exposure levels and RfDs are typically expressed in units of mg of PCB intake per kg of body weight per day (mg/kg-d). Actual daily exposures over the assumed exposure period of 30 years, rather than the lifetime (70 year) average exposures, are of concern when evaluating noncarcinogenic effects. Thus, unlike evaluation of carcinogenic effects, exposures of less than lifetime duration are not averaged over an entire lifetime.

Quantitative assessment of carcinogenic risks involves evaluation of lifetime average daily exposure levels and applications of toxicity factors which reflect carcinogenic potency. Excess lifetime cancer risk is calculated as:

$$Risk = CSF \times CDI$$

where CSF is the carcinogenic slope factor and CDI is the chronic daily intake (averaged over a lifetime) of PCBs from the river. Exposure levels are expressed in units of mg of PCB intake per kg of human body weight per day (mg/kg-d). When exposure occurs only over some portion of a lifetime, the lifetime average exposure, which is believed to reflect the contribution of the exposure to lifetime risk, is approximated by dividing the total PCB intake over the period of exposure by the total lifetime (assumed to be 70 years). Since carcinogenic slope factors (CSFs) are expressed in units that are the reciprocal of those for exposure (*i.e.*, the CSF is expressed as $(mg/kg\text{-}d)^{-1}$, multiplication of the exposure level by the CSF yields a unitless estimate of cancer risk.

The risk estimate for cancer reflects the incremental increase in the probability of getting cancer following site-specific exposure compared to the background probability or the probability associated without exposure to site contaminants. For example, a risk of 2.0×10^{-6} means that an additional 2 people in an exposed population of 100,000 people who are actually receiving the

dose from which the risk was calculated are estimated to manifest cancer during a lifetime of exposure.

Table B.6-5 summarizes the carcinogenic risks from exposure to PCBs from the Hudson River, for all exposure pathways. Non-carcinogenic risks, in the form of the Hazard Quotient, are summarized in Table B.6-6. These risks are discussed below, by exposure pathway.

B.6.4.2 Dietary Intake

Fish

The risk calculations for recreational anglers assume that a person consumes contaminated fish from the Hudson River for 30 years. The assumptions presented in Table B.6-1 are the basis of the risk calculations. The risks are calculated using the 95th percent confidence limits of mean PCB concentrations in fish for an estimated 30-year projection and for current conditions (1986-1988 data). For the projected 30-year mean concentration, the excess cancer risk is estimated to be 2×10^{-3} and the HQ is 6. The estimated excess cancer risk for exposure under current conditions is 2×10^{-2} and the Hazard Quotient is 51.

Drinking Water

Recent monitoring data on Hudson River water indicate that surface water PCB concentrations are below regulatory guidelines for drinking water. Thus, only a screening level risk calculation for the drinking water pathway was performed. This scenario used the 95 percent upper confidence limit of the mean PCB concentration (0.06 µg/l) based on samples taken at Rogers Island from 1986 - 1989. Using this exposure concentration and assuming direct consumption of water without treatment yields an upper bound risk estimate of 6×10^{-6}. The non-carcinogenic hazard quotient for this scenario is less than unity (HQ<1).

Other

Risks that may be associated with consumption of vegetables, meat, dairy milk or breast milk were not calculated, because insufficient data are available (see B.6.2).

B.6.4.3 Inhalation Exposures

Air concentration data for PCBs are inadequate to permit a quantitative assessment of human exposure and risk from the Hudson River site. To do so, would require monitoring concentrations of contaminants in areas along the Hudson River and at distances from it and relating measured concentrations to the Hudson River as the source.

B.6.4.4 Recreational Exposures

Dermal Exposure to River Sediment

The risk calculations assume that an individual is dermally exposed to river sediments over a 30-year period. The magnitude of this exposure changes over time because of different behavioral and physical characteristics at different ages. The risks are calculated assuming that the PCB level in sediment is 66.2 ppm.

The estimated lifetime excess cancer risk from exposure is 4×10^{-5}. The Hazard Quotient for dermal exposures never exceeds unity for any exposed age group. There are many uncertainties regarding the current PCB levels in sediments. Because most recent sediment sampling efforts have focused on previously defined hot spots, the 66.2 ppm RME value is probably biased toward sediments with relatively higher PCB concentrations as neither a random sampling program nor sampling of beach sediments was employed. Thus, this excess cancer risk value must be considered only a rough estimate.

Incidental Ingestion of River Sediment

As with the evaluation of dermal exposures, risk calculations for incidental ingestion of river sediment assume that an individual is exposed to river sediments over a 30-year period. The magnitude of this exposure changes over time due to different behavioral and physical characteristics at different ages.

The estimated lifetime excess cancer risk for ingestion exposure is 2×10^{-6}. The Hazard Quotient for ingestion exposures never exceeds unity for any exposed age group. As noted above, the limitations in the current database for PCB levels in sediment result in a cancer risk that should be considered only a rough estimate.

Contact with River Water

The risk calculations assume that an individual is dermally exposed to river water during recreational outings over a 30-year period. The magnitude of this exposure changes over time due to different behavioral and physical characteristics at different ages. The risks are calculated using the 95th percent confidence limit of the mean water concentration from Fort Edward.

The estimated lifetime excess cancer risk from water exposure is 2.0×10^{-7}. The Hazard Quotient for exposures to water never exceeds unity for any exposed age group. Since the exposure concentration incorporated into the risk calculations is from the sampling location that demonstrates the highest level of PCBs of any sampling location, it can be anticipated that risks associated with exposure to water from other locations would be lower.

B.6.4.5 Risk Characterization Compared to Human Studies

The risk estimates are based on studies of the effects of PCBs on various animal species given very high doses in controlled laboratory tests. Specifically, the evaluation of cancer risk is based on studies performed on rats; the

evaluation of the risk of non-cancer endpoints is based on studies in monkeys. There is considerable uncertainty in extrapolating from animals with high doses to humans with low doses. Ideally, an assessment of risks to humans from low-level environmental exposures would be based on toxicological studies from human populations exposed at low doses. Although low-exposure human studies are virtually non-existent, epidemiological studies conducted on humans exposed in occupational settings can provide information useful to evaluating risks of low level exposure.

Due to the lack of adequate exposure data and the inability to control adequately for other chemical exposures in epidemiological studies, it is impossible to establish whether the quantitative estimate of risk that is derived from animal studies is consistent with findings from exposed human populations. A review of available studies does, however, demonstrate that PCBs have been associated with adverse health effects among human populations and that the endpoints of concern from animal studies are consistent with observed human toxicities (see B.6.3.5 and Tables B.6-7 and B.6-8).

B.6.4.6 Analysis of Uncertainties

The process of estimating human health risks posed by PCBs in the Hudson River contains multiple steps. Inherent in each step are uncertainties that ultimately affect the final risk estimate. Generally, these uncertainties belong to one of the major categories of risk assessment: hazard identification, exposure assessment, and toxicity assessment.

In this assessment, no formal hazard identification process was undertaken, since previous investigations of the site had identified PCBs as the predominant contaminant. The absence of sufficient other chemical data precludes an evaluation of the interaction among PCBs and other chemicals. These interactions may ameliorate or exacerbate the toxicity of PCBs. An interaction in the latter category would be the co-presence of PCBs and a primary carcinogen. Unlike primary carcinogens, evidence is accumulating that PCBs do not initiate neoplastic growths; rather they appear to promote the development of latent

neoplasms. Thus, if PCBs and a primary carcinogen were both present in the Hudson River, an evaluation of PCBs alone would underestimate the cancer risks associated with river contact.

The intake quantity of PCBs has uncertainties associated with the variables used to make the intake calculation. Of these variables, the concentration of PCBs in individual media and the rate at which human populations contact these media are major sources of uncertainty. For example, contact rates are based on standardized national statistics that may not apply to the local population.

Because PCBs are a unique class of chemicals, yet are regulated as one compound, significant uncertainties also exist in the toxicological values used in this analysis. Cancer and non-cancer toxicity values for PCBs have been derived from studies using mixtures of PCBs. These values are based on animal studies where doses administered are high relative to typical environmental exposures experienced by humans. In the extrapolation for high laboratory doses to low environmental doses, uncertainties arise. In addition, these toxicity values are assumed to be representative of the toxicity values of all mixtures of PCBs and of individual PCB congeners, even though the toxic potency of different PCB congeners can vary significantly. In the future, specific cancer slope factors may be assigned to each Aroclor mixture and possibly to environmental mixtures or specific congeners.

Recent changes in the liver tumor classification system may also influence conclusions regarding carcinogenicity, both qualitative and quantitative, of PCBs. In 1986 the National Toxicology Program developed a new classification system for proliferative liver lesions in rats. This change occurred after the pathological criteria used to classify some of the lesions had evolved based upon new knowledge. For example, the older liver tumor classification systems included the classification of "neoplastic nodules." A more recent understanding of the pathogenesis of cancer has resulted in a division of this single category into two categories (Maronpot, 1986); 1) foci of cellular alteration, which are non-neoplastic changes in liver cells, and 2) hepatocellular adenomas, a benign form of liver tumor which may progress towards cancer.

A reevaluation of the carcinogenicity of dioxins (TCDD) was recently completed using the new classification (Maine Department of Human Services, 1990). In this reread of pathology slides, the total number of liver tumors declined from 33 to 18 in the high dose group. In addition, it was observed that carcinogenicity appeared to be associated with liver toxicity. Thus, cancer slope factor estimates, which are based in part upon neoplastic nodules of the liver, are likely to change with review of the slides. General Electric is currently in the process of funding a similar reevaluation of the pathology slides from PCB cancer bioassays (Neal, personal communication). Because the cancer slope factor for PCBs is also based on evidence of liver cancer, it is possible that information from this reevaluation could support a downward adjustment of the CSF for PCBs.

Using the example of the reevaluation of the carcinogenicity of TCDD discussed above, the reduction in the upper 95 percent confidence limit of the linearized multistage (LMS) model, as applied by the Maine Bureau of Health, showed less than an order of magnitude decrease in the cancer slope factor of TCDD. In fact, the upper 95 percent confidence limit declined only by a factor of about four. In contrast, the maximum likelihood estimate showed a much greater decline with the reread of the slides - well over an order of magnitude demonstrating the insensitivity of the upper 95 percent confidence limit to changes in the dose-response curve.

Such a finding reflects the degree to which cancer slope factors, based on the upper 95 percent confidence limit of the LMS, are strongly influenced by the dose levels (typically based on the Maximum Tolerated Dose or MTD) of the particular experiment. Since selection of the MTD (the basis for dose selection in many chronic bioassays) reflects, in part, acutely toxic effects, it is not surprising that a correlation exists between the dose levels of a study and the carcinogenic slope factor estimate.

Two additional issues are important to recognize. First, based on the currently available data, the primary risks estimated to exist result from consumption of contaminated fish by anglers who either disregard the fishing

advisory issued by the NYSDOH or are unaware of it. Second, the risks associated with some potential pathways of exposure were not quantified in this assessment due to inadequate data. Preliminary analysis indicates that risks might be associated with these other pathways and that further data are needed to derive a total risk across all exposure pathways. Table B.6-1 summarizes potential exposure pathways, including those for which quantitative risk assessments were not performed.

B.6.5 Lower Hudson Discussion

The extent of area encompassed by the Lower Hudson River and estuary and the volume of data that would be required to conduct a quantitative exposure and risk assessment for this area combine to make such an effort difficult to accomplish with the available data. Issues such as potential PCB exposure differences in the fresh versus salt water portions of the Lower Hudson, including assessing the impacts of metropolitan and industrial sources of PCBs, will have an important bearing on a risk assessment for the Lower Hudson. At this time, data for the Lower Hudson are sufficient to provide only some general comparisons with the preliminary risk assessment for the Upper Hudson.

- Fish have been sampled extensively at various locations in the fresh and salt water portions of the Lower Hudson. PCBs transported from the Upper to Lower Hudson contribute to the PCBs found in the freshwater fish population of the Lower Hudson. A comparison of the concentrations of PCBs in the fish from this freshwater portion of the Lower Hudson indicate that overall concentrations in the Lower Hudson are slightly below those in fish from the Upper Hudson, but they are on the same order of magnitude. Therefore, risks associated with human consumption of these fish -- assuming exposure patterns are similar for the Upper and Lower Hudson -- would be similar to the risks associated with consumption of fish from the Upper Hudson.

- The available data on PCBs in river water from the Lower Hudson are both fragmentary and out of date. The data available are limited to a few samples collected between 1978 and 1981 (Schroeder and Barnes, 1983). These data are not adequate to characterize possible exposures that might occur via river contact.

- In the Lower Hudson, samples were taken from river bottom sediments, which in many cases were in areas unlikely to receive human exposure. This situation makes it difficult to apply the data to human exposure scenarios. In order to provide an adequate characterization of the human health risks associated with exposure to river sediments in the Lower Hudson, sampling of sediments from areas of human contact, such as river banks, would be required.

- Data on PCB levels in air of the Lower Hudson are even more limited than for the Upper Hudson. Furthermore, it is likely that there are sources of PCBs in the airshed of the Lower Hudson that are not adequately characterized and it may be difficult to correlate airborne PCBs with a specific source.

- A substantial effort would be required to evaluate how exposure pathways along the Lower Hudson differ from those of the Upper Hudson. This effort is particularly difficult for the Lower Hudson, because exposures may be unique not only in comparison to the Upper Hudson, but may change within different parts of the Lower Hudson, because of the differing influences of salt water and freshwater.

The preliminary assessment of the human health risks associated with the Upper Hudson indicates that the risks posed by exposure to PCBs from consumption of fish outweigh risks associated with other routes of exposure by several orders of magnitude. It is likely that exposures from consumption of fish taken from the Lower Hudson will similarly present the greatest risk, because extensive bioaccumulation of PCBs occurs in fish. Since exposures via fish consumption overshadow other exposures to such a great extent, the total risks from PCBs in the Lower Hudson would not be expected to change significantly even with collection of adequate data for additional exposure pathways. If, however, a comparison or evaluation of the possible risks from other uses of the river (recreational contact, drinking water, etc.) are desired, additional data must be collected.

SYNOPSIS

INTERIM ECOLOGICAL RISK ASSESSMENT
(Section B.7)

The objective of this interim ecological risk assessment (B.7.1) is to present available information considered pertinent to a subsequent more definitive assessment of risks to non-human populations exposed to PCBs derived from the Upper Hudson River. This assessment is considered interim, because sufficient data are not available to provide a comprehensive and fully quantitative risk assessment. The ecosystem is defined as the river (Fort Edward to Troy) and adjacent uplands.

As background, a description of the ecosystem is given (B.7.2), drawing from previous environmental investigations and other published information. The PCB exposure assessment (B.7.3) presents a conceptual food chain of PCB exposure, uptake and transfer pathways. A simplified aquatic food chain (benthic insect, benthic fish, carnivorous fish, bird and mammal) is developed and five indicator species (chironomid larvae, brown bullhead, largemouth bass, herring gull and mink) are selected, based upon data available. PCB exposure concentrations and possible PCB uptake and transfer via sediment, water and food intake are discussed. The analysis utilizes measurements of PCBs in sediments, water, fish and some aquatic insects.

The toxicity assessment (B.7.4) relies upon a review of the literature on PCB toxicity in aquatic species, fish-eating birds and mammals as well as proposed criteria and guidelines. The risk characterization (B.7.5) compares monitored and estimated PCB exposure levels of selected indicator species to published toxicity information and proposed PCB guidelines.

Based upon the limited available data, it is premature to conclude whether ecological risks specifically attributable to PCB contamination from the Upper Hudson River exist. In spite of the many years of PCB monitoring data for water and fish and sporadic monitoring of sediments, only one known study of potential adverse ecological effects of PCBs in the Upper Hudson has been published, but could not pinpoint PCBs as the causal agent.

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B.7 Interim Ecological Risk Assessment

B.7.1 Phase 1 Objectives

The objective of this Interim Ecological Risk Assessment is to present information pertinent to a subsequent and more definitive ecological risk assessment of PCBs in the Upper Hudson River. This preliminary discussion adheres to current USEPA Superfund guidance for ecological risk assessments (USEPA, 1989d) and is the first step in the ecological risk assessment procedure. Depending on subsequent analyses, interpretations and conclusions will be presented in following phases. In the ensuing discussion, the term ecosystem refers to the river and adjacent upland areas from the vicinity of Hudson Falls/Fort Edward to the Federal Dam, but does not refer to the entire Upper Hudson drainage basin.

Although considerable data have been gathered on PCBs in the Upper Hudson River, the monitoring programs have been conducted by a variety of agencies and researchers. Therefore, the data gathered do not follow a singular, orchestrated plan to assess ecological risk per se. Although it may be tempting to assume that the many years of monitoring should provide definitive conclusions on the ecological health of the Upper Hudson site, the fact remains that very little data have been gathered to relate the measured PCB concentrations in sediment, water and biota to observed ecological effects. Thus, this preliminary assessment relies on published information concerning PCB toxicity in relation to measured PCB concentrations in Upper Hudson River sediments, water and biota. A comprehensive ecological risk assessment, including population, community and ecosystem interactions in response to PCB exposure, is not possible with the available monitoring data. Continued efforts in subsequent phases of the reassessment will evaluate the need for additional ecological monitoring.

PCB contamination in the Upper Hudson River ecosystem is most evident in the river sediments, water column and fish. Although fish have been the most extensively monitored component, some PCB data for aquatic invertebrates exist and a small amount of data on PCBs in crop/plant species have been collected (see

discussion at B.3). Because very limited data exist concerning PCB levels in piscivorous (fish-eating) birds, mammals, and other wildlife, a semi-quantitative evaluation of possible PCB exposure for these groups is presented. Indicator species have been chosen on the basis of species occurrence, PCB sensitivity and/or availability of toxicity and PCB monitoring data. This approach provides an initial evaluation of potential ecological risks for selected species.

B.7.2 Ecosystem Description

B.7.2.1 Terrestrial Habitats

Habitats

The terrestrial ecosystem is characterized by mixed deciduous and coniferous forests, croplands, pasture, marshes, and seasonally or semi-permanently flooded evergreen and deciduous forests (see Plate B.1-4). The forested area is representative of northeastern transition woodlands (Andrle and Carroll, 1988). Forested areas are particularly apparent along the eastern shoreline and nearby uplands. Forests west of the river are sparser and discontinuous, but are apparent along much of the shoreline and surrounding area north of the Saratoga National Historical Park (north of Stillwater). Woodland species common to these areas include the white-tailed deer, raccoon, opossum, various squirrels and other small mammals.

Agricultural land use along much of the Upper Hudson is dominated by pasture and croplands that support the local dairy industry. Deciduous forests as well as meadows and grasslands containing a variety of herbaceous shrubs, grasses and wildflowers often border the farm fields. Such diversity in cover types provides an abundance of food and suitable habitat for white-tailed deer, foxes, birds and small mammals. Several species of raptors, including the red-tailed hawk and the great horned owl, frequent these open areas.

The New York State Department of Environmental Conservation has defined and designated "significant habitat," considered to be areas of special concern for aquatic and terrestrial species. Within the Upper Hudson River area, deer wintering areas are listed as significant habitats (Malcolm Pirnie, 1984b). Other, possibly sensitive or important habitats are fish spawning areas, waterfowl wintering areas, pine barrens, and bog-wetlands. Wetlands surrounding the Upper Hudson are largely riverine as classified by Cowardin *et al.* (1979).

Vegetation

The Upper Hudson River lies in the Hudson Lowlands. Vegetative classes of the pockets of wetlands along the river include: emergent, moss-lichen, floating leaved bed and submergent bed plant varieties. At higher elevations, there is a transition to a northern hardwood forest with species such as sugar maple, beech, yellow birch, hemlock and white pine (Malcolm Pirnie, 1984b).

Birds

The Hudson River Valley is part of the Atlantic flyway migratory route and Champlain-Hudson Valley crossing (Schierbaum *et al.*, 1959). Rivers and wetlands along this route provide ample bird feeding and breeding habitat for such species as black duck, mallard, wood duck, both blue and green winged teal and Canada goose (Malcolm Pirnie, 1984b).

Resident or breeding species include numerous songbirds, wintering waterfowl and raptors. Based on Andrle and Carroll's (1988) atlas of all breeding birds within New York State, the following is a representative list of suspected or confirmed breeding species within the Upper Hudson River ecosystem.

Mallard
American Black Duck
Northern Pintail
Blue-winged Teal
Belted Kingfisher
American Bittern
Spotted Sandpiper
Red-winged Blackbird
European Starling
Northern Flicker
Eastern Meadowlark
Common Nighthawk
Wild Turkey
Red-necked Pheasant

Herring Gull	Ruffed Grouse
American Crow	Red-tailed Hawk
Mourning Dove	American Kestrel
Northern Cardinal	Great Horned Owl

Mammals

Mammals are especially sensitive to human encroachment as range size decreases and competition for food increases. Nevertheless, habitat, such as deciduous and evergreen forests bordered by openlands and wetlands, has sufficient diversity to support a moderate mammalian population. Game species, such as cottontail rabbit and white tail deer, abundant muskrat populations and moderate populations of mink, beaver and weasels are found (Malcolm Pirnie, 1984b). Wildlife field guides for the Upper Hudson provide the following list of other species likely to inhabit the vicinity.

Red and Gray Fox	Hoary Bat
Raccoon	Striped Skunk
Red and Gray Squirrel	Woodchuck
Eastern Chipmunk	Meadow Vole
Southern Bog Lemming	River Otter
White-footed Mouse	Muskrat

Amphibians and Reptiles

Several species of amphibians (frogs, toads and salamanders) inhabit the river's shores, shallow tributaries and areas of dense forest litter. Salamanders common to the area are the spotted, dusky and two-lined salamanders. The American toad as well as bullfrogs, green frogs and wood frogs are also common throughout the surrounding wetlands.

Many reptilian species are found along the river and adjacent uplands. Characteristic aquatic species include the snapping turtle, painted turtle, brown snake and the northern water snake. Terrestrial or upland species include the

wood turtle, smooth green snake, eastern ribbon snake and the rare timber rattlesnake.

Threatened and Endangered Species

A number of species found in the Upper Hudson River Valley are listed by New York State as endangered, threatened or species of special concern (Buffington, 1991). These protected species are named below by category.

Endangered
- Bald Eagle*
- Peregrine Falcon*
- Shortnose Sturgeon*
- Bog Turtle

Threatened
- Mud Sunfish
- Osprey
- Timber Rattlesnake
- Red-shouldered Hawk
- Northern Harrier

Special Concern
- Least Bittern
- Cooper's Hawk
- Black Rail
- Upland Sandpiper
- Common Barn Owl
- Common Nighthawk
- Henslow's Sparrow
- Grasshopper Sparrow
- Vesper Sparrow
- New England Cottontail
- Small-footed Bat
- Southern Leopard Frog
- Spotted Salamander
- Banded Sunfish
- Blackchin Shiner

*Also considered endangered under federal regulations.

In addition to being listed as endangered by New York State, the bald eagle, peregrine falcon, and shortnose sturgeon are currently listed as endangered by the federal government (USFWS, 1991). No existing field data were found to document the presence of either the bald eagle or peregrine falcon within the site area. Although the shortnose sturgeon's range is generally limited to the Lower Hudson south of Troy, it may access the upper reaches of the river through the series of locks.

B.7.2.2 Aquatic Ecosystem

Studies conducted in the Upper Hudson by the New York State Conservation Department (now NYSDEC) as early as 1933 document a number of major pollution problems and associated biotic perturbations (Farrell, 1933). Pulp and sewage contaminants were often cited as major factors contributing to a shift in the benthic (bottom) community structure from pollution intolerant taxa, *e.g.*, mayflies and caddisflies, to pollution tolerant taxa, *e.g.*, midges and tubificids. This structural shift was also reflected in the fish populations with many more tolerant species, *e.g.*, bullheads, and suckers, caught in extremely polluted areas. Although Farrell (1933) gave a very general account of pollution sources, it was one of the earliest attempts to document the biotic impacts of pollution in the Upper Hudson River.

Boyle (1969), in his book *The Hudson River*, described the general characteristics of the Upper Hudson between Fort Edward and Troy in a rather blunt narrative fashion:

> The character and appearance of the Hudson change entirely when the river leaves the foothills of the Adirondacks at Fort Edward, the head of navigation. Locks and dams now choke the flow and turn the river into a forty-mile chain of sluggish lakes. From Fort Edward down to Troy, the river serves mainly as a highway for pleasure boats and self-propelled barges...
>
> Ecologically, the canalized Hudson is sort of an entity unto itself, a world apart from the rushing Adirondack parent and sealed off by locks and dams from the estuary to the south. Before the construction of the Troy Dam in 1826, fishes from the Atlantic used to make their way upstream as far north as Glen Falls, a journey of 209 miles. Shad were found in the

tributaries of the Upper Hudson River such as Battenkill...In colonial times, sturgeon were abundant in this stretch of the Hudson. ...Striped bass, too, came in numbers...Nowadays, a stray striper or shad may work its way through the lock system, but they are markedly rare above Troy. The most conspicuous fishes are strangers--black bass... and carp...

In the canalized Hudson, especially where the river slows, the aquatic insects differ from those in the rushing river of the Adirondacks. There are, in certain clean coves and backwaters, a profusion of dragonflies and damselflies...There presence indicates, by rough rule of thumb, whether or not the water is badly polluted. Alas, the canalized Hudson probably does not have as many dragonflies as it did in times of the past. This stretch of the river has been greatly despoiled and disfigured by pollution, much of which is from pulp and paper mills...thick, gray mats of pulp wastes...drift downstream, where they sink and pile up against dams. ...Instead of dragonflies and fishes...one may find..."index organisms"...sludge worms,...leeches,..and rattail maggots.

From the time Boyle's book was published in 1969 to the present, positive changes in the Upper Hudson have taken place. A number of sewage treatment facilities have been upgraded and industrial discharges are more stringently regulated (Shupp, 1975).

Earlier in this document (see A.1), the conceptual framework of an aquatic food chain for the Hudson was presented. Because PCBs bioaccumulate through the food chain, that approach is also used here to provide a foundation for evaluating the ecological exposure and risks posed by PCBs in the Upper Hudson ecosystem. Where necessary, information from the previous discussion is summarized briefly.

Conceptual Ecosystem Framework

Recent evaluations of the Hudson River ecosystem by Limburg *et al.* (1986) and Gladden *et al.* (1988) discuss the four major categories of organic resources in the aquatic ecosystem:

- primary producers -- phytoplankton, periphyton and macrophytes;
- detritus -- particulate organic matter and associated microbial biomass;

- dissolved organic matter; and
- consumers -- microzooplankton, macrozooplankton, benthic invertebrates and fish.

Phytoplankton

There are very limited data on resident phytoplankton populations in the Upper Hudson River. Data collected by NYSDEC (data provided by R. Bode) indicate that species of green algae, blue-green algae, diatoms and dinoflagellates were found in the Upper Hudson from 1974-1978. Not surprisingly, the dominant genera recorded in the Upper Hudson were also found throughout the freshwater portion of the Lower Hudson River (Frederick *et al.*, 1976; Storm and Heffner, 1976; Weinstein, 1977).

During a seasonal survey of phytoplankton throughout the Hudson River, Storm and Heffner (1976) found that representatives of the diatom genus *Cyclotella* and unidentified green nannoplankton were the most common taxa. Of the 16 river stations surveyed by Storm and Heffner (1976), only one was located in the Upper Hudson, approximately two miles above the Federal Dam in the vicinity of Waterford. Although an overall species list by station was not included, Storm and Heffner (1976) found that during the warmer months the phytoplankton communities in the vicinity of Waterford shifted from diatoms to those dominated by green and blue-green algae. Temporal patterns of diatom dominance from late fall through spring and green/blue-green algae dominance during the late summer months are typical for most temperate aquatic systems, including the Lower Hudson (McFadden *et al.*, 1978).

Periphyton

The growth of periphyton is linked to the velocity of the river. Some species need much slower current than others to maintain adequate population density (Whitton, 1975). For example, many of the benthic diatoms, *e.g.*, *Navicula*, are attached to silt particles and may form large mats, while others,

such as *Cladophora*, usually occur in faster flowing water and form long resistant filaments. No periphyton data are known to be available within the Fort Edward to Federal Dam river reaches.

Macrophytes

There are very limited data on resident macrophyte (macroscopic forms of aquatic vegetation) populations. Muenscher (1933) surveyed the "abundant" and "common" aquatic vegetation from the mouth of Snook Kill to Mechanicville. This study found pickerelweed (*Pontedaria*), arrowhead (*Sagittaria*), and dense beds of floating heart (*Nymphoides*) in relatively shallow, slow-flowing coves and in areas behind islands adjacent to the various landcut portions of the Champlain Canal. Since this survey, there have been no additional inventories of aquatic macrophytes within this part of the Upper Hudson. Thus, the present composition and extent of macrophytes cannot be determined. In a recent assessment (Feldman, 1991, SUNY Binghampton, pers. comm.) of macroinvertebrate taxa associated with aquatic macrophyte species, *Trapa* (Water Chestnut) and *Vallisneria* (Water Celery) were collected in the vicinity of the Thompson Island Pool. Given the paucity of data, the contribution of macrophytes to primary production in the Upper Hudson is not discernible.

Invertebrate Community

Invertebrates can be categorized in three major groups, based on habitat preference and size. These are: microzooplankton, macrozooplankton and benthic invertebrates.

The zooplankton components of most freshwater systems are dominated by rotifers, pelagic cladocerans and copepods. Although considered an important food resource for a variety of plankton-feeding fish, including emerald shiners, spottail shiners and tesselated darters (Gladden *et al.*, 1988), no surveys have been conducted on resident zooplankton populations within the Upper Hudson.

The benthic invertebrates comprise a large and heterogeneous assemblage of organisms. Some of the major groups of benthic invertebrates found in many freshwater systems, likely to occur in the Upper Hudson, are listed below.

Aquatic Insects
- Dragonflies (Odonata)
- Damselflies (Odonata)
- Mayflies (Ephemeroptera)
- Stoneflies (Plecoptera)
- Caddisflies (Trichoptera)
- Midges (Diptera)

Mollusks
- Bivalve clams and mussels (Pelecypoda)
- Univalve snails (Gastropoda)

Annelids
- Aquatic earthworms (Oligochaetes)

Roundworms (Nematodes)

Crustaceans
- Scuds (Amphipods)
- Sowbugs (Isopods)
- Seed shrimp (Ostracods)

Surveys of benthic invertebrates during 1972 (Simpson *et al.*, 1974), 1977 (Bode, 1979) and 1987-1988 (RIBS Sampling, 1987-1988), conducted within selected parts of the Fort Edward to Federal Dam section of the Upper Hudson, indicate that the region is dominated by midge larvae (Chironomidae). All the above-cited studies utilized multiplate samplers (Hester and Dendy, 1962) suspended in the water column. These samplers consisted of tempered hardboard plates and spacers, which were held together with an aluminum turnbuckle (Simpson *et al.*, 1974). Artificial multiplate samplers are biased for freely colonizing benthic species, such as midge larvae, and sampling results may not adequately represent the overall benthic community, including resident infauna components. In the following discussion, major results of all benthic studies conducted to date within the Fort Edward/Federal Dam reaches of the Upper Hudson are reviewed and analyzed.

During the summer of 1972, NYSDEC conducted a benthic survey at 12 stations from Troy to an area upstream of Corinth (Simpson *et al.*, 1974). Benthic collections from Fort Edward to Troy (stations 5-12) indicate that this stretch of the Upper Hudson is dominated by midge larvae (Diptera: Chironomidae) and oligochaete worms (Naididae). An average of 14 taxa were recorded, including the dominant genera listed below.

Midge larvae (Diptera: Chironomidae)
Cricotopus
Pentaneura
Polypedilum
Rheotanytarsus

Dance Flies (Diptera: Empididae)
Hemerodromia

Aquatic earthworms (Oligochaetes)
Nais
Pristinella

The midge larvae and oligochaetes collectively account for greater than 85 percent of the total benthic invertebrates sampled. In most collections, midge larvae were approximately two to three times more abundant than the oligochaetes.

During the summer of 1977 (Bode, 1979), three stations (UHUD- 4A, 6 and 11) within the Federal Dam/Fort Edward reaches were sampled. These stations were located in the vicinities of Fort Edward, Fort Miller and Waterford. Once again, the midge larvae (73.5 percent of total) and oligochaetes (22.4 percent of total) dominated the benthic invertebrate community. An average of 21 taxa were recorded. In addition to midge larvae and oligochaetes, four species of Ephemeroptera (mayflies), fourteen species of Trichoptera (caddisflies) and one species of Plecoptera (stoneflies) were collected within the Fort Edward to Waterford region during 1977. The more intensive 1972 study at seven stations within the same region recorded only two species of mayflies, three species of caddisflies and no stoneflies. The increased representation of these groups during the 1977 study is of interest, since mayflies, stoneflies and caddisflies are widely recognized as being far less tolerant to poor water quality (low

dissolved oxygen, high BOD, etc.) than either midge larvae or oligochaetes (Wilson, 1984). In addition, there was an increase in species richness from an average of 14 taxa in 1972 to 21 taxa in 1977.

The only study specifically designed to sample benthic infauna was conducted in 1988 by NYSDEC (Preddice and Karcher, 1990). The field sampling included quantitative collections of benthic macroinvertebrates by ponar grab in order to ascertain if significant community changes were associated with bridge maintenance at Fort Miller. Targeted samples were taken from soft sediment areas; ponar sampling difficulties precluded collecting samples from gravelly areas. Both these events bias the samples toward a preponderance of soft bottom (sediment) dwelling infauna. Preddice and Karcher (1990) found an average of 13 taxa in the abundant groups listed below.

Group and Taxa (Percent of Total)

Midge larva (29.9 percent)
- *Tribelos*
- *Procladius*
- *Chironomus*
- *Polypedilum*

Olicochaetes (22.9 percent)
- *Tubificidae*
- *Lumbriculus*
- *Limnodrilus*

Round worms -- Nematoda (17.6 percent)

Fingernail Clams (13.4 percent)
- *Sphaerium*
- *Pisidium*

Side Swimmers (6.7 percent)
- *Gammarus*
- *Hyalella*

Caddisfly Larvae (2.9 percent)
- *Phylocentropus*

Although midge larvae and oligochaetes were still the most abundant groups, there was an increased representation of nematodes, clams and amphipods compared to the 1972 and 1977 studies (Simpson *et al.*, 1974; Bode, 1979). These differences are somewhat expected, given the way the samples were secured. The 1972 and 1977 studies used multiplate samplers suspended in the water column and the 1988 study (Preddice and Karcher, 1990) utilized a ponar grab sampler which, unlike the multiplate samplers, is extremely effective for sampling soft bottom benthic infauna. Similar to the 1977 study, four genera of mayfly larvae were recorded. In addition, the four genera of caddisflies found are known to inhabit silt or mud substrates (Bode, 1991, per. comm.).

A recent multiplate survey of benthic invertebrates (data provided by R. Bode, NYSDEC) within the Federal Dam/Fort Edward reaches of the river was conducted as part of a RIBS (Rotating Intensive Biological Survey) study of drainage basin 11 during 1987-1988.

The 1987 and 1988 surveys sampled three stations in the vicinity of Fort Edward, Schuylerville and Waterford, approximately the same locations as the 1972 and 1977 surveys. Midge larvae dominated the benthic community and accounted for 72 percent of all organisms sampled. The most striking dissimilarity, however, was in the relative percent contribution of the oligochaetes. The oligochaetes comprised one to two percent of the total benthic community compared to an average contribution of 25 percent recorded during the 1972 and 1977 surveys. An average of 21 taxa were recorded within the Fort Edward to Waterford region during the 1987-1988 RIBS program. The following is a list of dominant genera.

Midge larvae (Diptera: Chironomidae)
- *Corynoneura*
- *Cricotopus*
- *Dicrotendipes*
- *Hudsonimyia*
- *Orthocladius*
- *Polypedilum*
- *Rheotanytarsus*
- *Synorthocladius*
- *Tanytarsus*
- *Thienemanniella*

Caddisfly larvae
Cheumatopsyche
Chimarra
Hydropsyche
Neureclipsis

Mayfly larvae
Baetis
Stenonema

Flatworms (Platyhelminthes)
Undetermined

Analogous to the 1977 study, the RIBS 1987-1988 biological samples revealed that caddisflies (10 species), mayflies (8 species) and stoneflies (1 species) are present within the Fort Edward to Waterford region of the Upper Hudson. The continued presence of these pollution intolerant (sensitive) groups suggests that water quality improvements have occurred in the Upper Hudson since 1972.

Fish

A number of studies have attempted to categorize the fisheries within the 40-mile stretch of the Upper Hudson between Federal Dam and Fort Edward. (Refer to B.1.4 for a more detailed review of relevant fisheries surveys.) Historical surveys by Greeley and Bishop (1933) and recent surveys (Makarewicz, 1983; Malcolm Pirnie, 1984b; Green, 1985) indicate rather diverse fish fauna. The vast majority of species are year-round residents. Although some anadromous species such as American shad, alewife, blueback herring and striped bass may be present in the Upper Hudson, the construction of the Federal Dam and Champlain Canal has essentially blocked major upstream spawning migrations.

The diversity of freshwater residents in the Upper Hudson is indicative of the varied habitats that occur in this section of the Hudson. The wide variation in habitats expands spatial heterogeneity and results in a quite complex fishery resource. For example, Makarewicz (1983) surveyed nine different habitats within the Fort Edward to Federal Dam section of the Upper Hudson.

Malcolm Pirnie (1984b) found that relative fisheries abundance varied among areas sampled. For instance, far fewer fish (13 individuals) were recorded in the natural river channel compared to areas adjacent to outlets of streams (637 individuals). Approximately 69 percent of all fish collected (Makarewicz, 1983), subsequently summarized by (Malcolm Pirnie, 1984b), were indicative of various slow-flowing, shallow water zones and stream mouths.

All fish species collected by Makarewicz (1983) were categorized into four functional groups (Malcolm Pirnie, 1984b), including game fish, panfish, forage fish and demersal fish. Listed below are the numerically dominant members of these four functional groups that were collected within the Federal Dam/Fort Edward reaches.

Group	Dominant Members
Game Fish	Largemouth bass Smallmouth bass
Panfish	Bluegill Pumpkinseed Rock bass Yellow perch
Forage	Golden shiner Spotfin shiner Spottail shiner
Demersal	Black bullhead Brown bullhead Common carp White sucker

Collectively, the above 13 species account for 85 percent of the total number of resident freshwater fish collected during the Makarewicz (1983) survey and represent a variety of habitat preferences. For example, bullhead, pumpkinseed, bluegill and yellow perch prefer relatively protected slow-flowing, shallow habitats, whereas smallmouth bass and spottail shiner seem to prefer more moderate flowing waters and/or the outlets of small to large streams (Scott and Grossman, 1973; Malcolm Pirnie, 1984b).

All the major qualitative studies reviewed indicate that the fish species historically present in the Upper Hudson continue to reside in the Fort Edward to Troy reaches.

Summary of Aquatic Ecosystem

A number of studies (see reviews by Wetzel, 1975 and Mann, 1975) have concluded that various autochthonous primary production inputs (phytoplankton, periphyton and macrophytes) make important contributions to aquatic food webs. Unfortunately, the paucity of data in the Upper Hudson makes it impossible to determine the relative contribution of the various primary producers. Furthermore, contributions of allochthonous carbon from upstream areas and the surrounding watershed are not known, but may be more refractory in nature and of reduced or limited nutritional value compared to the autochthonous sources. Whatever the relative sources of organic carbon, data reviewed for the Upper Hudson indicate that the system is capable of supporting diverse fish fauna.

Many species of fish living in river systems can exploit a variety of food resources, including invertebrates, detritus and other fish (Weinstein, 1977, Moran and Limburg, 1986 and Gladden *et al.*, 1988). Although no fish studies in the Federal Dam/Fort Edward region of the Upper Hudson have included routine analyses of stomach contents, many resident species seem to have diverse and opportunistic feeding habits (Malcolm Pirnie, 1984b). The exploitation of various resources by fish populations may lead to a pattern of trophic partitioning, as exemplified by the dietary preferences (Moran and Limburg, 1986 and Gladden *et al.*, 1988) listed below.

Zooplankton
- Emerald shiner
- Tessellated darter
- Spottail shiner

Benthic/Detrital
- Common carp
- Goldfish
- Eastern silvery minnow
- Golden shiner

White catfish
Brown bullhead
Bluegill
White sucker

Epibenthic invertebrates
Yellow perch
Pumpkinseed
Redbreast sunfish

Fish/Macroinvertebrates
Largemouth bass

In addition to differences in trophic strategy, the year class strength of some species may be linked to the timing or availability of food and the foraging success of larval and juvenile fish. Thus, the temporal variability in fish numerical abundance and spawning success may be linked to spatial and temporal trends of trophic resources, although studies of seasonal populations would be needed to confirm this possibility.

The apparent improvement in the Upper Hudson fisheries (Shupp 1975, 1987) appears to correspond with upgrades of sewage treatment plants and more stringent regulations concerning industrial discharges. The improvement in the fish community seems also to be reflected in the invertebrate populations.

Comparisons of recent RIBS (1987-1988) data on benthic invertebrates to studies in 1977 and 1972 (Bode, 1979; Simpson *et al.*, 1974) suggest a general water quality improvement in this section of the Hudson. The water quality improvement is indicated by the trends in the average number of intolerant (sensitive) species such as mayflies, stoneflies and caddisflies and by the average species richness (total number of species or taxa) in multiplate samples from 1972-1988.

The number of mayflies, stoneflies and caddisflies species (EPT, as defined below) and species richness are two of a variety of parameters currently in use by the NYSDEC in order to determine overall stream water quality. Bode *et al.* (1991) state:

"...EPT denotes the total number of species of mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera) found in an average 100-organism subsample. These are considered to be mostly clean-water organisms, and their presence generally is correlated with good water quality (Lent, 1987). Expected ranges from most streams in New York State are: greater than 10, non-impacted; 6-10, slightly impacted; 2-5, moderately impacted; and 0-1, severely impacted."

It must be emphasized that the reference to "good water quality" above refers generally to standard parameters such as dissolved oxygen, biological oxygen demand, *etc.*, and does not necessarily reflect the possible influences of PCBs.

A preliminary analysis of the average EPT and average species richness within the Fort Edward to Federal Dam reaches of the Upper Hudson is presented below.

EPT and Species Richness Summary

	1972	1977	1987	1988
EPT	0.2	6.0	7.4	8.8
Species Richness	14.0	21.0	20.0	22.0
Overall Assessment*	1	2	2	2

* 1 = severely to moderately impacted
2 = slightly impacted

Since 1972, there has been an increased representation of the more pollution intolerant (sensitive) groups and an almost two-fold increase in species richness from 1972 to 1977-1988. In addition, the organic waste tolerant Naid oligochaetes (Bode *et al.*, 1991) have declined from a relative abundance of approximately 25 percent (1972 and 1977) to 1-2 percent (1987-1988).

B.7.3 PCB Exposure Assessment

This ecological exposure assessment identifies probable routes of PCB exposures, quantifies such exposures where possible and presents a conceptual trophic pathway or food chain approach to indicate possible PCB transfer pathways within the ecosystem. No attempt is made to define the aquatic and terrestrial food web in detail. The *"New York State Barge Canal Environmental Report"* (Malcolm Pirnie, 1984b) contains a summary of available environmental/habitat and species studies relevant to the site, but the information therein is not adequate to describe the food web in detail. Furthermore, data available specific to PCBs are inadequate to evaluate species, population and community health and dynamics, which are necessary components of an ecosystem approach. The available data are used here in a simplified ecological framework to develop an initial assessment of PCB exposure levels (measured and extrapolated) and possible risks for selected indicator species that are representative of the aquatic food web.

B.7.3.1 Exposure Pathways

Both exposure and uptake of PCBs by organisms in the ecosystem depend on: ambient (time and space-dependent) PCB levels in water, sediments and air; species growth and feeding habits; PCB storage; kinetic transfer rates; and metabolism. Because the PCB contamination in the Upper Hudson is largely within the river, aquatic species, such as benthic invertebrates, fish and piscivorous (fish-eating) wildlife, are more susceptible to PCB exposure and uptake than terrestrial herbivores and omnivores. Volatilization of PCBs from the river may also contribute to elevated PCB levels in plants, but this pathway is likely to be less significant than aquatic exposures. (Because PCB levels in plants near the river are also not well-documented, such an analysis is precluded at this time.) Similarly, because of the currently very low PCB concentrations in water, direct ingestion of contaminated river water by upland species is not likely to lead to PCB uptake to the degree expected as a consequence of consumption of fish from the river. Thus, aquatic organisms and predators of fish from the river are the focus of this ecological exposure evaluation.

Two primary modes of PCB exposure are examined: 1) direct contact and absorption with PCBs in water and sediments; and 2) dietary ingestion of PCBs and potential food chain transfers. Because PCBs are lipophilic, they tend to accumulate in body tissues, especially body lipid (fat), with the frequent result that PCB levels in body tissues increase with higher trophic level. Therefore, species at the top of the food chain may accumulate high PCB levels, even though they may not be exposed directly to PCBs in the water and sediments. (A possible exception may be species associated with the benthic environment, which are exposed to very high PCB concentrations in sediments and interstitial water.)

A conceptual food chain of PCB exposure, uptake and transfer pathways in the Upper Hudson can be described as a producer/consumer system, consisting of:

- *primary producers*, such as aquatic macrophytes, filamentous algae, phytoplankton and periphyton;
- *primary consumers*, such as micro- and macrozooplankton, benthic invertebrates and various aquatic insects;
- *secondary consumers*, including forage or planktivorous fish, such as shiners and several demersal species;
- *carnivorous (game) fish*, such as largemouth bass and northern pike; and
- *fish eating (piscivorous) birds and mammals* at the top or near-top trophic levels considered here.

Organisms associated with the benthos throughout their life history include certain invertebrate primary consumers (some insect larvae, oligochaete worms, snails and mussels) and a variety of demersal fish (catfish, bullhead and carp), collectively categorized as secondary consumers. Other secondary consumers, including various forage fish (emerald shiner, spottail shiner, tesselated darter) and panfish (pumpkinseed, rock bass and yellow perch), feed primarily on zooplankton and epibenthic invertebrates. Carnivorous (game) fish, such as largemouth bass and northern pike, typically feed on aquatic insects and invertebrates as juveniles and on other fish and macroinvertebrates as adults. While the different feeding behavior of fish may result in different exposure risks to

PCBs in the Upper Hudson, PCB uptake in food has not been reported and cannot be assessed at this time.

Birds and mammals are the top trophic level considered here. Wildlife in this trophic level include the herring gull, heron, osprey, mink and river otter, all of which consume various species of omnivorous and carnivorous fish. Omnivorous birds, such as ducks, geese, and various shorebirds, may be exposed to PCBs in the river when they feed on several species of submerged and emergent plants, plankton and aquatic insects.

Other terrestrial exposure pathways are not part of this analysis, as additional field studies would be required to permit quantification of terrestrial exposure/uptake pathways.

B.7.3.2 Identification of Indicator Species

The table below identifies indicator species, covering various trophic levels in the food chain. They were selected, based on their probability of exposure, significance in the food web and the availability of applicable exposure concentrations, *i.e.* PCB monitoring data.

Indicator Species

Component	Species	Exposure Route
Benthic Insect	Chironomid Larvae	Sediment, Water
Benthic Fish	Brown Bullhead	Sediment, Water, Benthos (diet)
Carnivorous (Game) Fish	Largemouth Bass	Sediment, Water, Fish (diet)
Piscivorous Bird	Herring Gull	Fish (diet)
Piscivorous Mammal	Mink	Fish (diet)

The aquatic indicator species (chironomid, brown bullhead and largemouth bass) were chosen, because site-specific monitoring data are available for these species.

No site-specific data adequate to assess PCBs in non-aquatic species were found. Therefore, non-aquatic species were selected based on the availability of published information regarding PCB uptake and response. Non-aquatic indicator species chosen in this preliminary assessment may not necessarily represent the preferred species from an ecological perspective, but their selection was constrained by available data related to PCBs. For example, the heron, osprey or even bald eagle would probably be preferred to the herring gull as a fish-eating indicator bird species, because their diets more predominantly come from fish than the diet of the herring gull. The herring gull is an omnivore; it eats fish, but fish is, perhaps, a less significant fraction of its overall diet. PCB toxicity data are, however, available for the herring gull, whereas no data were found in the literature for heron, osprey or eagle. Furthermore, there is no site-specific information documenting that either ospreys or bald eagles are significant inhabitants of the site, while herring gulls are common.

Mink are often considered quite sensitive mammalian species that obtain a major portion of their diet from fish (Newell *et al*. 1987). As discussed earlier, mink are known to inhabit the Upper Hudson site (Malcolm Pirnie, 1984b), although their population has not been reported. Furthermore, PCB toxicity in mink has been widely reported.

Pending continued evaluation of the Upper Hudson ecosystem related to PCB contamination, other indicator species may be selected and monitored in future phases.

B.7.3.3 Exposure Quantification

Table B.7-1 summarizes measured and estimated PCB exposure levels for the indicator species in the Upper Hudson, including an indication of the level of confidence or reliability in these values. Reliability is considered high for exposures based on measured values with extensive spatial and temporal coverage (PCBs in water and fish). Reliability is considered moderate for exposures based

on monitoring data that may lack adequate temporal or spatial coverage but for which a large data set is available (sediments, macroinvertebrates). Reliability is considered low for estimated exposures based on assumed dietary intake levels.

As indicated in Table B.7-1, PCB concentrations in body tissues have been measured for the chironomid and fish indicator species. There is no direct monitoring data for dietary intake by any of the indicator species and dietary intake (where given in the table) is estimated, based on possible feeding rates, as discussed below. The lack of dietary PCB uptake adds emphasis to the previous discussion that despite all of the available PCB monitoring data, gaps in the data exist, which hinder a definitive evaluation of PCB exposures and transfers for species which have not been monitored, *e.g.*, birds and mammals.

Ambient Water and Sediment Exposures

Ambient water is a significant route of exposure for all aquatic dwelling organisms. A strong correlation exists between PCB levels measured in fish and those in water (see B.4). Whether this relationship is due to direct partitioning through gill and body surfaces, contact with sediments, ingestion of contaminated food or a combination of these factors is not determined.

The measurements of PCBs in the water column in 1986-89 provide a reasonable baseline for this investigation. Upper 95 percent confidence bounds on mean PCB concentrations in the Upper Hudson for this period range from 0.06 µg/*l* (Fort Edward) to 0.034 µg/*l* (Waterford), as explained in Section B.3. Summer low-flow PCB concentrations could provide the best indicators of possible ecological harm, because there is less flushing during low flows and water temperatures are generally higher, leading to increased biological activity. These concentrations, however, do not differ from full-year values to such an extent as would affect the risk analysis.

Ambient PCB levels in river sediments are less well-defined in time and space and are far more locally variable than those in water. Recent sediment data, preferred for evaluating baseline conditions, are relatively sparse. PCB

levels measured by GE in a recent study indicate median PCB levels on the order of <10 mg/kg to >100 mg/kg at the 12 locations sampled, targeted at previously defined hot spot locations. Earlier data collected in 1977 by NYSDEC yielded median PCB levels within river reaches on the order of 5 to <30 mg/kg, with hot spots commonly having >100 mg/kg average PCB levels. The 1984 Thompson Island Pool survey results indicate that approximately 80 percent of the samples had PCB levels less than 100 mg/kg. The highest concentrations measured exceed 1,000 mg/kg in the Thompson Island Pool. Taking all surface sediment samples (including all grab and top core section samples = 569 samples) from the 1984 Thompson Island Pool survey, the 95-percent upper confidence bound on the mean is 66.2 mg/kg. This estimate provides a reasonable maximum exposure level for sediment-bound PCBs. Subsequent analyses in Phase 2 and 3 may focus attention on evaluating sediment exposures as a function of river reach and areas of particularly high PCB concentrations.

Chironomid Larvae

The results of the NYSDOH multiplate monitoring (see B.3.4 for a more detailed discussion of the monitoring results) indicate that chironomids (midges) were the most abundant macroinvertebrate component of the 1985 samples, comprising up to 86 percent of the total macroinvertebrate population at Fort Miller and Waterford (Simpson *et al.*, 1986). As larvae and pupae, these organisms often live in burrows or flocculent tubes created from bits of sediment held together by a secretion of their silk glands. Uptake of PCBs for these life stages is likely to reflect this direct association with contaminated sediments and interstitial water.

The 96-hour biomonitoring studies conducted by NYSDOH in 1985 indicated that PCB concentrations in the tissue of laboratory-reared chironomid larvae that were exposed to ambient water in the Thompson Island Pool ranged from 5 to >7 mg/kg (dry weight) (NYSDOH, 1986). Corresponding PCB concentrations in ambient water during these periods ranged from 0.06 to 0.13 µg/*l* in July and from 0.03 to 0.07 µg/*l* for September. Although the NYSDOH study was short-term, *i.e.* 96

hour exposures, PCB accumulation appeared to reach a quasi-steady state level during the course of the study.

The NYSDOH study also investigated PCB uptake in laboratory-reared chironomid larvae exposed to sediments in the Thompson Island Pool. These chironomid larvae had greater than 100 mg/kg (dry weight) of PCBs after 96 hours of exposure, an apparent reflection of the higher PCB concentrations in the sediments and interstitial water.

In light of the very sparse database of information from which these short-term exposure estimates were drawn, the baseline PCB levels in chironomid are uncertain. Nevertheless, the limited data suggest that considerable bioaccumulation takes place.

Fish

Exposure levels of PCBs in fish are needed to evaluate:

- potential adverse effects on the health of fish caused by the PCB levels in fish tissue, and;
- potential adverse effects on fish-eating wildlife as a result of consumption of PCB-containing fish.

As discussed in B.4, PCB levels in fish have generally displayed an exponential decline from 1975 to 1988, although the PCB levels fluctuate from year to year and average concentrations in 1988 were generally greater than those observed in 1987. PCB levels in fish for the three most recent years for which samples are available (1986-1988) are summarized below.

Upper-Bound Mean PCB Concentrations* in Fish at Three Upper Hudson Locations
(mg/kg -- wet weight)

Year	RM 153-155			RM 175			RM 190-195 (Thompson Island Pool)		
	LMB	BB	PKSD	LMB	BB	PKSD	LMB	BB	PKSD
1986		3.3		7.9	13.9	7.0	12.6	48.7	
1987	3.1	2.3	4.3					16.9	5.9
1988	5.8	2.7		3.6	13.7	5.5	11.7	20.3	8.1

*Values are 95-percent upper confidence bounds on the mean taken from Tables B.3-16, B.3-17, and B.3-18.

Abbreviations:

RM River Mile
LMB Largemouth Bass (fillets)
BB Brown Bullhead (fillets)
PKSD Pumpkinseed (yearling samples -- head and viscera removed)

Although PCB concentrations in fish have been monitored for over 15 species, pumpkinseed, largemouth bass and brown bullhead samples represent the largest and most recent data available. These three species also comprise a large proportion of the overall Upper Hudson fish fauna. Therefore, PCB concentrations listed above for these three species are considered representative of recent baseline conditions in the Upper Hudson.

PCB concentrations in these species vary with River Mile, with the highest PCB levels in the Thompson Island Pool samples, lowest concentrations in those samples caught near Federal Dam and intermediate PCB levels for the samples near Stillwater. Recent (1986-1988) 95-percent upper confidence bound total PCB concentrations in fish for Thompson Island Pool samples range from >5 ppm (pumpkinseed) to <50 ppm (brown bullhead). Representative upper-bound PCB levels near Federal Dam range from >2 ppm (brown bullhead) to <6 ppm (largemouth bass); PCB levels for these three species near Stillwater are on the order of 5 to 10 ppm.

Because many toxicity studies report adverse affects of PCBs to birds and mammals in terms of dietary exposure levels, assessing potential risks to fish predators would be based ideally on whole-fish PCB concentrations, because fish-eating wildlife consume whole fish. PCB levels were measured, however, in fish

fillet (muscle) tissue. Filleting and skinning the fish samples remove a large proportion of lipid-containing tissue where PCBs tend to accumulate. Thus, PCB levels in whole-fish samples from these species are likely to be somewhat greater than the levels reported above. (Too few whole-fish samples were reported to provide an adjustment.) Niimi and Oliver (1989, 1983) found that whole-fish PCB concentrations exhibited up to five-fold higher PCB concentrations than PCB levels in muscle in four salmonid species studied. Using their results, and ignoring possible species differences, whole-fish PCB body burdens may be approximately five times greater than the levels reported above.

Although it is possible to adjust the fillet PCB concentrations to whole-fish values, based on literature studies such as those cited above, this adjustment was not considered necessary for several reasons. No site and species-specific data to make the adjustment was found. A factor of five adjustment as suggested by the Niimi and Oliver studies would not affect this preliminary assessment. No site-specific data on the type or amount of fish actually consumed by fish-eating wildlife in the Upper Hudson Site vicinity is available; approximate dietary intake is estimated from published sources where needed. Thus, uncertain foraging and food preferences and unknown fish intake were considered much larger uncertainties than possible differences between fillet versus whole-fish PCB levels. Finally, the data currently available are inadequate to provide quantitative risk estimates for fish or fish-eating wildlife, precluding the need for such an adjustment.

Estimated Fish Dietary Intake

Brown bullhead are exposed to PCBs by ingestion of sediment matter, dietary intake, including various species of cladocerans, copepods, chironomids, algae and fish, ingestion of ambient water and direct absorption. For some dietary components of brown bullhead, PCB concentrations may be surmised from the results of the NYSDOH long-term multiplate monitoring. These samples reflect PCB concentrations in algae, zooplankton and macroinvertebrates as well as organic detrital matter and inorganic sediment components. Multiplate samples taken from the Upper Hudson at Fort Miller have shown little change over the period of

measurement (1976-86), with typical values on the order of 10 - 60 µg/g (mg/kg) on a dry-weight basis. As mentioned above, chironomid larvae tissues contained approximately five to greater than seven mg/kg PCBs when exposed to ambient water. Larvae exposed to sediments showed approximately a ten-fold increase over these levels. Although their major dietary component consists of several species of zooplankton and aquatic insects, including chironomids, bullhead also feed heavily on both dead and live fish. Studies on adult black bullhead indicate that fish may represent from 1 to over 50 percent (by volume) of the total dietary intake (Repsys *et al.* 1976; Applegate and Mullan, 1966). Using the 1986-1988 fish monitoring data described above, average dietary PCB concentrations in the three prominent fish species in the Upper Hudson that are possibly consumed by brown bullhead range from approximately >2 to <50 mg/kg.

Largemouth bass are carnivorous game fish. As juveniles, largemouth bass feed predominantly on various species of cladocerans or water fleas and other zooplankton. In contrast, adults are opportunistic; top carnivores feed primarily on fish, but also consume large insects, crayfish and other benthic invertebrates (Pflieger, 1978).

Stomach contents of bullhead and bass have not been analyzed to determine food preferences and PCB levels in their food. Such studies may prove useful in the future to determine mechanisms and rates of PCB uptake in fish. Results of multiplate, short-term chironomid sampling and fish sampling all indicate PCBs in these species to be on the order of several to >10 mg/kg. Thus, average PCBs in fish diets in the Upper Hudson may contain on the order of several to >10 mg/kg of PCBs.

Herring Gull

Fish-eating birds are exposed to PCBs in the fish they consume and, to a lesser extent, through ingestion of water. Herring gulls are essentially opportunistic and may feed on a variety of fish, shellfish, insects and carrion. Whittow and Rahn (1984) report that their diet consists of approximately 50 percent fish. Given that the recent PCB levels in the Upper Hudson fish fillet

samples (summarized above) are approximately >2 to <50 mg/kg, a weighted concentration of PCBs in gull diets in this area, assuming their other dietary sources are free of PCBs, may be approximately 1 - 25 mg/kg.

The PCB dose, expressed as the average daily PCB intake per unit body weight, depends both on PCB levels in the food, the quantity of food ingested and body weight. Herring gulls weigh from 0.5 to 1.3 kg and consume approximately 20 percent of their body weight(BW)/day. Assuming a range of dietary concentrations of 5 to 50 mg/kg, a body weight of 1 kg and a daily intake rate of approximately 0.1 kg fish/day (50 percent fish), the daily dietary PCB exposures for herring gulls is estimated to range from 0.1 to 2.5 mg/kg$_{BW}$-d. Actual exposures would vary depending on PCB concentrations in fish typically eaten, the percentage of fish in their diet and other dietary intakes containing PCBs.

Birds in the site vicinity have not been monitored for PCBs. Therefore, the relationship between the amount of PCBs ingested and the tissue levels in birds (say gulls) remains unknown. One approach to estimate PCB body burdens, which ignores the mechanism and rates of PCB transfers from food to body tissues, relies on the use of bioaccumulation factors (BAFs) or the ratio of PCBs in bird tissues to the PCB concentrations in their diet. Braun and Norstrum (1989) of the Canadian Wildlife Service have determined from field studies on herring gulls in the Great Lakes that these birds accumulate approximately 93 times the PCB concentration in their diet ($BAF_{tissue} = 93$). This BAF is based on PCB levels of approximately 0.5 mg/kg measured in alewife (fish), a major part of the diet of the herring gull in their study. Braun and Norstrum determined that the eggs of the herring gulls studied accumulated approximately 32 times the PCB concentration in whole fish ($BAF_{egg} = 32$). Adopting these BAF values as approximations and assuming as discussed above that the weighted-average gull diet contains approximately 1 - 25 mg/kg PCBs, the resulting PCB levels in herring gulls would be approximately 93 - 2,325 mg/kg (whole body) and 32 - 800 mg/kg (egg). Without field sampling to confirm PCB levels in fish-eating birds in the vicinity, these estimates are very uncertain.

Mink

According to Linscombe *et al*. (1982) and Aulerich (1973), the mink's diet is comprised of approximately 30 to 50 percent fish. Other dietary components include crayfish, amphibians and other small mammals. Assuming mink consume 50 percent of their diet as fish from the Upper Hudson, they could ingest a weighted average of 1 - 25 mg/kg in their diet. Newell *et al*. (1987) indicate the average body weight and daily food consumption for mink are 1 kg and 0.15 kg/d, respectively. Based on these assumptions, the mink daily intake or dietary dose of PCBs could be on the order of 0.15 - 3.8 mg/kg-day. Because PCB toxicity to mink has been studied based on the PCBs in the mink diet and not on the PCB levels in mink body tissues, possible levels of PCBs in mink are not estimated.

B.7.4 Toxicity Assessment

B.7.4.1 Types of Toxicity

The toxicity of PCBs to aquatic and terrestrial organisms can vary considerably depending on congener and Aroclor composition. Large differences in factors such as percent chlorine, solubility, congener structure, organism sensitivity and species-specific sensitivity contribute to the overall complexity in evaluating PCB toxicity. In spite of the congener- and organism-specific toxicity response, it is generally true that PCBs are largely chronic toxicants in the environment, rather than acutely toxic. Ambient concentrations of PCBs in the environment are frequently not high enough to pose acute toxic effects, such as death, but exhibit a cumulative effect whereby toxicity increases as the length of PCB exposure increases. Furthermore, PCB toxicity generally increases as the degree of PCB chlorination increases. Exceptions to this pattern include highly chlorinated congeners, which may be physically hindered from accumulating in tissues, and congeners with a low chlorine content, which are more bio-available and, thus, are more readily metabolized. (The toxicity of these less highly chlorinated congeners may be due to the phenolic metabolites produced during metabolism.)

A further complexity in the determination of PCB toxicity arises as a result of the differing toxicological characteristics among congeners with very similar chlorine content. For example, 20 of the 209 possible PCB congeners exhibit coplanar properties due to non-ortho substitution in the biphenyl ring (Tanabe, 1988). Coplanar molecules have an approximately flat structure whereby the two benzene rings of the PCB molecules lie in the same plane. Examples include 3,3',4,4',5-pentachlorobiphenyl and 3,3',4,4',5,5'-hexachlorobiphenyl, compounds that are structurally similar to 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and 2,3,4,7,8-pentachlorodibenzofuran. Although coplanar PCBs are constituents of Aroclor mixtures in minute quantities, consideration of their dioxin-like responses is warranted. Studies have indicated that coplanar congeners are 10 to 1,000 times more toxic than similar non-coplanar PCB congeners (Eisler, 1986). Such unique responses have initiated the development of Toxic Equivalency Factors (TEFs), a process that consists of normalizing enzyme induction potencies against known TCDD-induced responses (Safe, 1990). This TEF approach is gaining acceptance for assessing biological risks. Its use, however, depends upon reliable congener-specific PCB analyses, which are largely unavailable for the Hudson.

In addition to coplanar toxicity research, other studies implicate the contaminant dibenzofuran as the principal toxic agent in PCB mixtures. Dibenzofurans are co-products of most Aroclor mixtures and are also produced as PCB metabolites or as photochemical derivatives (Kalmaz and Kalmaz, 1979).

Toxic effects resulting from chronic exposures of PCBs include enzyme inhibition and induction, immunotoxicity, liver disorders, tissue lesions, reproductive impairment, reduced growth and, in some cases, death. The varying species tolerances to Aroclor mixtures and specific congeners can result from the presence or absence of detoxifying mechanisms. Such mechanisms include the cytochrome P-450 mixed function oxidase (MFO) system or the 3-methylcholanthrene-inducible form (3-MC) of cytochrome P-450 referred to as aryl hydrocarbon hydroxylase (AHH) or cytochrome P-448 (Sipes and Gandolfi, 1986). These microsomal systems are responsible for the initial biotransformation of a wide variety of xenobiotics, including several PCB congeners. Induction of 3-MC

activity, which includes AHH and ethoxyresorum O-deethylase (EROD) induction, is usually associated with the transformations of coplanar congeners, whereas non-3-MC and MFO activity is considerably less specific.

The detoxifying process catalyzed by enzymatic oxidative reactions is essential to the organism's ability to metabolize congeners to more soluble products, which are excreted in urine and bile. Studies indicate that congeners having ortho-positioned chlorines may be easily metabolized and rapidly released, whereas congeners with chlorines in the para position are typically less soluble and are subsequently retained by various tissues (USEPA, 1980).

Microsomal enzyme induction has been used extensively as toxicity endpoints for several aquatic and terrestrial species, including fish (Huckins *et al.*, 1988), birds (Miranda *et al.*, 1987) and various mammals (Tanabe, 1988). Although induction of MFO activity is often correlated with PCB exposure, quantification of adverse effects related to induction is limited. Consequently, enzyme induction does not implicitly constitute irreparable harm.

PCBs are most often considered to be chronic toxicants, a characteristic that is associated, at least in part, with their tendency to bioconcentrate or bioaccumulate. Biological uptake of PCBs and accumulation in body tissues are influenced by several factors, such as direct exposure in water and sediment, PCB levels in food, organism lipid composition, and depuration (loss/release) rates. Both assimilation and depuration of PCBs tend to be related to chlorine content of PCBs.

The following sections present toxicity information, bioconcentration and bioaccumulation factors and proposed guidelines and criteria for PCBs in water, sediments and wildlife. To the extent possible, bioconcentration factors (BCFs) are used for laboratory-derived ratios of PCBs in organisms to PCBS in water. Bioaccumulation factors (BAFs) refer to values based on field measurements. (See glossary for a definition of BCF versus BAF.) Organisms for which toxicity and uptake data are available include plants, plankton, aquatic invertebrates, fish, birds and a limited number of terrestrial and semi-aquatic mammals.

B.7.4.2 Toxicity Literature Review

Plants

Reduced growth, reproductive alterations and inhibited photosynthesis are some of the documented responses exhibited by plants when exposed to PCBs. Exposure may be due to foliar contact from aerial deposition, vapor sorption of volatilized PCBs and root contact or uptake with contaminated soils. The latter is far less detrimental to plant survival (Pal *et al.*, 1980). Direct contact with PCBs in water is likely to be an important exposure pathway for aquatic plants.

Pal *et al.* (1980) investigated the effects of PCBs on carrots, sugarbeets, corn and tomatoes. Weber *et al.* (1979) found that soybeans exhibited decreased plant height and fresh weight when exposed to Aroclor 1254 concentrations of 1 to 1,000 ppm. Duckweed, an aquatic plant, suffered decreased colony formation when exposed to 5 mg/*l* Aroclor 1242 in water and complete growth inhibition at 100 mg/*l* (Mahanti, 1975).

Buckley and Tofflemire (1983) reported that plants in the vicinity of the Upper Hudson River have accumulated PCBs in the leaves and, to a lesser extent, in the stems, with little translocation to plant fruits (see Section B.3.5). Several studies have examined PCB uptake from contaminated soils (Moza *et al.*, 1976; Iwata *et al.*, 1974). Iwata *et al.* (1974) reported that translocation of PCBs from carrot roots to other plant tissues (*e.g.*, leaves) was minimal; 97 percent of the PCBs were retained in the outer peel and only 3 percent translocated into other plant tissues. Studies by Sawhney and Hankin (1984) have indicated that uptake from PCB-amended soils by beets, turnips and beans was ultimately influenced by PCB solubility and volatility. Sawhney and Hankin found that these plants accumulated Aroclors with the following preferences: Aroclor 1248 > Aroclor 1254 > Aroclor 1260.

Pal *et al.* (1980) reported BAFs, defined as the ratio of PCB concentration in plant tissue to PCB concentrations in the soil, ranging from 0.002 to 0.96 for several crop plants. In contrast, reported accumulation factors for aquatic plants ranged from 289 to 814 (Moza *et al.*, 1976).

Planktonic Species

The range at which adverse effects are exhibited by zooplankton and algae is quite wide, varying from exposures of 1.3 to over 2,000 µg/*l* (USEPA, 1980). When exposed to Aroclor 1254, various species of green algae exhibit adverse reactions, such as reduced carbon fixation and reduced growth rates at concentrations as low as 0.1 to 10 µg/*l* (USEPA, 1980). Exposure to Aroclor 1242 at 30 µg/*l* resulted in no significant effects to the amphipod *Hyalella azteca*, whereas exposures of 100 µg/*l* resulted in complete mortality (Borgmann *et al.*, 1990). *Daphnia magna*, a common aquatic organism used in establishing toxicity benchmarks, is very sensitive to PCB exposure. Lethal exposures for this species range from 1.2 µg/*l* for Aroclor 1248 to 2.5 µg/*l* for Aroclor 1254 (Eisler, 1986). Table B.7-2 summarizes these effects.

As with most aquatic species, zooplankton and algae accumulate PCBs in concentrations that greatly exceed ambient water column levels. BCFs for various species of phytoplankton have been reported in the range of 1,000 to 100,000 (USEPA, 1980). Eisler (1986) reports a BCF value of 47,000 for *Daphnia magna*.

Aquatic Macroinvertebrates and Insects

Although very few reports of the effects of PCBs on aquatic insects and macroinvertebrates exist, many of these species appear to exhibit sensitivities similar to those of *Daphnia magna*. The evaluation of exposure and toxic effects for aquatic insects is complicated by lifecycle stages, whereby insects are submerged as larvae, but winged upon emergence and, thereby, removed from a more contaminated medium, *e.g.*, water. This difference in morphology and physiology contributes to the difficulty in predicting exposures and subsequent effects. In contrast, aquatic invertebrates such as oligochaete worms, mussels, snails and

crayfish are less mobile benthic organisms, lacking multiple life stages with limited morphological change. These aquatic invertebrates may be continuously exposed to contaminated water and sediments, and, as a result, exhibit more predictable body burdens than organisms exposed to PCBs for only a portion of their lifecycle.

Eisler (1986) reports that chironomids (midges) are sensitive to Aroclor 1254 at concentrations of 0.5 to 1.2 µg/*l*. Exposure to Aroclor 1254 at 1.5 µg/*l* resulted in complete inhibition of mosquito emergence (Sanders and Chandler, 1972). The 7-day LC_{50} values, defined as the PCB concentration in water resulting in 50 percent mortality, for crayfish exposed to Aroclor 1242 and 1254 are 30 µg/*l* and 100 µg/*l*, respectively (Mayer, *et al.*, 1977). These authors report that grass shrimp exposed to Aroclor 1254 exhibited a 7-day LC_{50} value of 3 µg/*l*.

Chironomid larvae monitored during NYSDOH's short-term biomonitoring study in the Upper Hudson exhibited BAFs on the order of 10,000 to 200,000 (Simpson *et al.*, 1986).

Fish

Much of the available literature on PCB toxicity in aquatic environments addresses the effects of PCBs on various fish species. Common responses in fish exposed to PCBs include enzyme induction and inhibition, reproductive impairment, lesions, tumors, fin rot and, at high enough concentrations, mortality (USEPA, 1980). Mayer *et al.* (1977) report long term LC_{50} values for trout, bluegill and catfish ranging from 3 to 177 µg/*l* (Table B.7-2).

Studies of several freshwater species in Lake Michigan report that PCB concentrations of 0.01 µg/*l* in water and 1 µg/g in food caused significant reduction in lake trout survival (Willford *et al.*, 1981). Other studies have shown that exposures of 0.7 and 1.5 µg/*l* cause mortality in brook trout and largemouth bass, respectively (USEPA, 1980). Although very few studies address the issues of PCB toxicity associated with tissue residues, a PCB concentration

of 0.4 mg/kg (ppm) in whole-body fish tissues has been observed to cause reproductive impairment in rainbow trout (Eisler, 1986). Trout are particularly sensitive to PCBs and other species may be more tolerant to PCB exposures.

Laboratory-derived BCFs for several species of freshwater fish are on the order of 10,000 to 100,000 (USEPA, 1980; Mayer *et al.*, 1977). BAFs obtained from field surveys have ranged from 120,000 for rainbow trout to over 1,600,000 for lake trout (USEPA, 1980). As explained in Section B.4, the slope of the regression between PCBs in fish fillets (expressed on a lipid basis) and PCBs in the water column indicates that BAFs for Hudson River samples are on the order of 10^6. Recent studies by Jones *et al.* (1989) indicate that caged fathead minnows studied in the Upper Hudson selectively accumulated tetra- and penta-chlorinated biphenyls, suggesting a congener-specific bioaccumulation preference.

Birds

Aquatic and terrestrial birds have exhibited behavioral changes, reproductive impairment, enzyme induction, reduced growth and mortality when exposed to PCBs. Reported acute LD_{50} values, defined as the dose resulting in 50 percent mortality, for various birds range from 604 to over 6,000 mg/kg (concentration in the diet) for northern bobwhite and from 1,975 to 3,182 mg/kg for mallards (Eisler, 1986). It is evident from these values that birds may exhibit a high degree of initial resistance to PCB toxicity. Based on a six-week study of white leghorn hens, Britton and Huston (1973) reported that dietary levels of Aroclor 1242 at 20, 40, and 80 mg/kg yielded PCB concentrations in egg yolks of 6.2, 5.4, and 5.6 mg/kg, respectively, and resulted in a significant decrease in hatchability and growth. Newell *et al.* (1987), citing the Britton and Huston (1973) study, indicate a No Observed Effect Level (NOEL) of 0.224 mg/kg-day as a daily intake dose. [It is unclear from the Newell *et al.* (1987) report whether they are referring to the same Britton and Huston study, because Newell *et al.* report that the Britton and Huston study was for nine weeks, using Aroclor 1248. The cited Britton and Huston study was based on six-week feedings of Aroclor 1242.] Eisler (1986) reports that 40 mg/kg in the diet of doves caused changes in mating behavior.

Although several studies have investigated the effects of PCB levels on eggshell thickness in species such as cormorants (Koeman *et al.*, 1973), screech owls (Anne *et al.*, 1980) and western grebes (Lindvall and Low, 1980), conclusive evidence correlating PCB levels with eggshell thinning does not exist. It is possible, however, that eggshell thinning may be ascribed to the synergistic effects of PCBs, organochlorine insecticides and their metabolites (Lindvall and Low, 1980).

Birds may store relatively high concentrations of PCBs in fatty tissues, muscle, liver and brain. It appears that depuration is generally slow, but significant reductions in PCB body burdens have been observed as a result of egg production. Screech owls exposed to a diet containing 3 ppm of Aroclor 1248 laid eggs containing 3.9 to 17.8 ppm (Anne *et al.*, 1980). Other dietary exposures of PCBs from 10 to 20 ppm have resulted in PCB concentrations in eggs of 14 to 16 ppm (Eisler, 1986).

Mammals

Mink have been found to be very sensitive to PCBs. Aroclor 1254 fed to mink at concentrations of 0.64 to 2 mg/kg has led to reproductive failure (Platonow and Karstad, 1973; Aulerich and Ringer, 1977). Long term LD_{50} values of 0.1 mg/kg for hexachlorobiphenyl, 8.6 mg/kg for Aroclor 1242 and 6.7 mg/kg for Aroclor 1254 have also been reported (Eisler, 1986; Ringer, 1983).

Other sublethal effects exhibited by mammals include lethargy, weight loss, liver disorders, enzyme induction and inhibition and hormonal effects. Studies on rabbits, monkeys, mice and rats indicate that PCBs are also teratogenic and carcinogenic (USEPA, 1980). Other studies have shown that PCBs may even alter sleep and hibernation patterns in white-footed mice (Sanders and Kirkpatrick, 1977) and raccoons (Montz *et al.*, 1982).

B.7.4.3 Proposed Criteria and Guidelines

There exist no promulgated *standards* for PCBs in surface water, sediments, and wildlife. A number of federal and state agencies, including USEPA, NOAA, USFWS, and NYSDEC, have developed *criteria* and *guidelines* for assessing environmental thresholds for PCBs as summarized in Table B.7-3.

Ambient Water Quality Criteria

The USEPA has established a freshwater Ambient Water Quality Criterion (AWQC) for the protection of aquatic life including algae, zooplankton, aquatic insects, invertebrates, fish and piscivorous wildlife. The 24-hour average AWQC for PCBs in ambient (surface) water is 0.014 µg/*l* (USEPA, 1980). As set forth in USEPA guidance, this criterion is deemed to provide a conservative threshold level of PCBs in freshwater aquatic environments to protect against long-term exposure and bioaccumulation.

Mink, identified as a species very sensitive to PCBs, are at the apex of the aquatic food chain. The AWQC for PCBs is based, at least in part, on the assumption that mink represent a highly sensitive wildlife species in aquatic environments. Surface water PCB concentrations less than the AWQC are thought to protect other, less sensitive species in the aquatic environment. The AWQC was established using a dietary threshold of 0.64 mg/kg linked to reproductive failure in mink (Platanow and Karstad, 1973). Additionally, as fish are a major component of the mink diet, PCB levels in fish were taken as the major aquatic exposure pathway for PCB uptake in mink. Adopting BCFs, which the guidelines use to relate PCB levels in water to those in fish, the AWQC guideline is given by (USEPA, 1980) as:

$$\text{AWQC} = \frac{0.64 \text{ mg/kg}}{45{,}000 \text{ mg/kg} \,/\, \text{mg/l}} = 0.014 \text{ µg/l}$$

where the BCF (45,000 mg/kg / mg/*l*) is a geometric mean of 3 BCFs for rainbow and brook trout (USEPA, 1980).

The USFWS has reviewed the applicability of the AWQC guideline and concluded that the criterion is environmentally protective, but also concludes that it should be changed to reflect a *maximum* of 0.014 µg/*l* rather than a 24-hour average (Eisler, 1986). The New York State Ambient Water Quality Guidance Criterion is established at 0.001 µg/*l*, derived from an acute (96-hour) LC_{50} (NYSDEC, 1985).

Sediment Quality Guidelines

The development of criteria or guidelines for PCBs in sediments is under study by the USEPA Science Advisory Board. Several approaches are being considered including the Apparent Effects Threshold (AET), co-occurrence analysis (COA) and Equilibrium Partitioning (EP) methods.

The AET approach, developed from data derived in Puget Sound, Washington, combines chemical concentration data in sediments and a biological indicator of injury, such as sediment bioassays and altered benthic infauna abundance, to determine the contaminant concentrations above which significant biological response can be expected (PTI, 1988). Long and Morgan (1990) summarize the AET values for several Puget Sound studies and report AET values for San Francisco Bay, California. A range of AET values for PCBs, from 0.05 to 3.1 mg/kg (ppm -- dry weight), were reported for saltwater environments. Although the AET approach has been used primarily in saltwater systems, a similar approach may be useful in the Upper Hudson.

The Co-occurrence (COA) method, like the AET, relies on field-collected data matching chemical concentrations with observed biological effects. COA bioassay results for Hudson-Raritan Bay indicated biological responses to PCBs as low as 0.64 mg/kg. COA results for freshwater environments indicated effects ranging from 0.13 mg/kg to 1,141 mg/kg (Long and Morgan, 1990).

The EP method is based on theoretical partitioning of PCBs on sediments into the interstitial pore space of the sediment. A pore water PCB concentration equal to the AWQC is used to set a target PCB threshold in sediment. Because sediment-water partitioning depends on chemical and sediment properties, no single sediment value can be calculated. For Aroclor 1254, the EP value is 0.42 ppm (dry weight) for sediment with a one percent total organic carbon fraction (Long and Morgan, 1990).

As part of the National Status and Trends program, NOAA has recently reviewed studies that examine the AET, EP and other methods, such as spiked sediment bioassays (Long and Morgan, 1990). NOAA reviewed over 150 reports or studies on contaminated sediments around the United States. From these, Long and Morgan identified 34 that contained adequate and reliable data from which to synthesize sediment guideline values. Using these, NOAA has determined an Effects Range (ER). This range is determined by the 10th percentile (the low ER, or ER-L) and median (ER-M) chemical concentration thresholds for the 34 studies. The findings of NOAA's investigation indicate that PCB concentrations as low as 0.050 ppm (ER-L) with a median of 0.4 ppm (ER-M) have been shown to be associated with biological response. It must be emphasized that NOAA states clearly that the sediment thresholds are intended only as guidance values possibly to be used to identify the need for site-specific monitoring studies, and do not represent NOAA standards (Long and Morgan, 1990).

Guidelines for PCBs in Fish

Eisler (1986) presents a synoptic review of PCB toxicity in wildlife and presents proposed dietary and body tissue guidelines of the USFWS for PCBs in fish. The USFWS reported whole-body and egg-residue guidelines for rainbow trout of 0.4 mg/kg and 0.33 mg/kg, respectively. These tissue residues have been associated with reproductive impairment, decreased hatch success and fry deformities (Eisler, 1986). As mentioned previously, trout are particularly sensitive to PCBs and other hazardous organic compounds, such that the levels proposed by the USFWS are deemed to be conservative and protective of other, less sensitive fish.

As part of the Niagara River Biota Contamination Project, NYSDEC Division of Fish and Wildlife developed fish flesh criteria for piscivorous wildlife (Newell *et al.* 1987). These criteria, which are based on a review of mammalian and avian laboratory studies, establish threshold PCB levels in fish set to protect fish-eating wildlife. Thus, they are thresholds for fish consumers, rather than values set to protect the health of fish. Mink, because of their largely fish diet and demonstrated sensitivity to PCBs, are the piscivorous wildlife species used to establish the criterion. The noncarcinogenic fish flesh criterion for the protection of piscivorous wildlife established by NYSDEC is 0.13 mg/kg. This criterion was determined by using an adjustment factor of 0.2 to the dietary intake threshold (0.64 mg/kg) causing reproductive impairment in the study by Platanow and Karstad (1973). The adjustment factor is used to convert this Lowest Observed Effect Level (LOEL) to a NOEL.

Guidelines for PCBs in Birds

The most sensitive body tissue indicators of potential risks to aquatic and terrestrial birds have been found to be PCB accumulation in brain and egg tissues. Stickel *et al.* (1984) reported that Aroclor 1254 concentrations in brain tissue ranging from 349 to 763 ppm (wet weight) resulted in mortality in cowbirds, grackles, starlings and red-winged blackbirds. Birds that survived the same dietary dosage of 1,500 ppm had brain PCB residues of 54 to 301 ppm. This study concluded that brain PCB levels greater than 301 ppm are highly correlated with death. Based on these findings as well as other studies correlating brain PCB residues of 76 to 180 ppm with mortality in cormorants, the USFWS has recommended a threshold brain PCB concentration of 54 mg/kg (Eisler, 1986).

Although brain PCB residues are good indicators of possible lethality, residues of PCBs in eggs provide a more sensitive endpoint for determining lethal and sublethal effects as well as population effects as a consequence of decreased hatch success. Based on the findings of Britton and Huston (1973) discussed earlier, the USFWS has recommended a whole egg PCB concentration of 0.4 ppm (wet weight) as a threshold value (Kubiak, 1991, pers. comm.).

Guidelines for PCBs in Mammals

Guidelines for the protection of mammals are most conservatively represented by dietary studies on mink. Such proposed guidelines include the findings of Platonow and Karstad (1973) whereby a mink diet of 0.64 mg/kg Aroclor 1254 led to reproductive impairment. Ringer (1983) determined a NOEL in mink, on the basis of body weight (BW) intake, of 0.1 mg/kg_{BW}-d (approximately 0.67 ppm in the diet) from a study where mink were given a dietary dose of 0.225 mg/kg_{BW}-d, which resulted in significant reproductive impairment. The USFWS recommended dietary tolerance level for mink is 1.54 $\mu g/kg_{BW}$-d (Eisler, 1986).

B.7.5 Risk Characterization

This interim ecological risk characterization is limited to comparisons of PCB exposure levels in selected indicator species with: 1) proposed criteria and guidelines for ecological protection; and 2) toxicity endpoints or thresholds.

B.7.5.1 Ambient Water

As discussed earlier, upper 95 percent confidence bounds on recent (1986-89) mean concentrations of PCBs in the water column range from 0.034 (Waterford) to 0.06 µg/*l* (Fort Edward). These ambient levels are approximately two to five times greater than the USEPA AWQC of 0.014 µg/*l*; they are from 30 to 60 times greater than the more conservative New York State water quality criterion of 0.001 µg/*l*.

Site-specific reports or monitoring studies of adverse effects to aquatic life in the Upper Hudson resulting from PCB exposures do not exist. A review of the toxicity studies summarized in Table B.7-2 reveals that the baseline PCB concentrations in water are below levels that have been measured to cause aquatic toxicity. Thus, although the water column PCB levels may be elevated somewhat above AWQC values, toxicological data in the literature do not corroborate an imminent harm to algal, macroinvertebrate, insect and fish species from direct contact with PCBs in water.

B.7.5.2 Sediment

Concentrations of PCBs in sediments of the Upper Hudson, ranging from <1 ppm to >1,000 ppm are generally above various proposed sediment guideline levels discussed previously. As reported in NOAA's National Status and Trends report, with "very few exceptions" biological effects, such as reduced benthic macroinvertebrate diversity or reduced chironomid and mayfly survival, were observed in sediments with PCBs above 0.37 ppm (Long and Morgan, 1990).

Long and Morgan attribute a "moderate" degree of confidence in the threshold effects values for PCBs in sediment. However, species richness/diversity and species-specific biological responses to PCBs listed in the Long and Morgan study are difficult to correlate with ecological risks in the Upper Hudson. For example, Long and Morgan (1990) report that 0.7 ppb PCBs in Mississippi River sediments caused reduced chironomid survival. Concentrations of PCBs in the sediments of the Upper Hudson exceed these levels by orders of magnitude, but, as discussed previously, chironomid species appear to be increasing in abundance in the river.

Likewise, the PCB threshold concentration in sediment, according to the equilibrium partitioning (EP) approach, is on the order of 2 ppm, using the Aroclor 1254 partition coefficient and approximately five percent total organic carbon, based on volatile solids content of Hudson River sediment. Sediments in the Upper Hudson contain PCBs well above this threshold. Again, the AWQC is a water column concentration based on bioaccumulation and protection of fish-eating wildlife. Yet the connection between pore-water PCB concentrations and food-chain uptake and transfers is largely uncertain. Thus, it is not clear what ecological impacts are occurring because of PCBs in the sediments.

B.7.5.3 Fish

The USFWS has recommended a guideline of 0.4 ppm for PCBs in fish tissues for the protection of fish. Recent PCB levels for fish (fillets) in the Upper Hudson range from 2.3 to 48.7 ppm for brown bullhead and 3.1 to 12.6 for

largemouth bass. The levels of PCBs in both indicator fish species are over 10 times the recommended USFWS guidelines.

The USFWS guideline is based on studies of rainbow trout in which whole body residues of 0.4 ppm resulted in significant fry mortality and deformities (USEPA, 1980; Eisler, 1986). Although trout are very sensitive to PCBs, they are not a major species in the Upper Hudson site vicinity. It is unclear if present PCB levels in the Upper Hudson fish species represent a significant detrimental impact on the indigenous population.

Kim *et al.*(1989) conducted the only known study of the possible pathological response of Upper Hudson fish to contamination the river. In their study, brown bullhead were sampled near Griffin Island (River Mile 190) and Stillwater (River Mile 175); these fish were designated the experimental fish from contaminated stretches of the river. Fish sampled near Corinth (River Mile 200) were designated as control samples. PCB levels in the contaminated fish ranged from 16.3 to 102 mg/kg, with a mean of 38.3 mg/kg. PCB levels in the control fish ranged from 0.38 mg/kg to 1.4 mg/kg, with a mean of 0.61 mg/kg. No significant differences in gross abnormalities, such as outward physical characteristics, were found in the experimental versus control fish. In contrast, histopathological (cell pathology) results indicated a statistically significant increase in bile-duct hyperplasia (abnormal cell growth) in the contaminated fish (78 incidences) compared to the control fish (13 incidences). Kim *et al.* (1989) note:

> "The most significant finding was the high frequency of bile-duct hyperplasia, accompanied by hepatic and renal hemosiderosis among brown bullhead from the contaminated section of the Hudson River. Since the hepatobiliary system plays a major role in removing toxic materials from the blood after absorption from the gastrointestinal tract and is the site where biotransformation and excretion of xenobiotics is taking place, the observed bile-duct abnormalities could serve as an indicator of chemical contamination of an aquatic environment."

No single chemical could be identified as the causal agent in their study, although the authors note that studies by Norback and Weltman (1985) have shown bile-duct hyperplasia to be a notable response of rats exposed to PCBs in their diets during laboratory experiments. Further indication that the observed pathological differences were possibly induced by toxic organics was the measured increase in cytochrome P-450 in the livers of two contaminated fish compared with the levels in the control fish (Kim *et al.*, 1989).

While the Kim *et al.* (1989) results indicate that toxic organic contaminants, including PCBs, hexachlorobenzne, and octachlorostyrene found in the fish, are possible agents responsible for the observed pathological abnormalities, they conclude that "the specific chemicals or metabolites responsible for the observed bile-duct hyperplasia remain to be investigated" (Kim *et al.*, 1989).

B.7.5.4 Fish-Eating Birds

Birds in the Upper Hudson area have not been monitored for PCBs. Weighted concentrations of PCBs in the fish diet of herring gulls could be on the order of 1 to 25 ppm, based on TAMS/Gradient estimates. The USFWS proposed dietary guideline is 3.0 mg/kg for the protection of birds. This threshold concentration is based on feeding studies in screech owls where a diet of 3 ppm resulted in high PCB residues in owl eggs. Several other studies cited previously have reported that diets of 20 to 80 ppm in chickens caused significant reproductive impairment. Eisler (1986) reports LD_{50} values for mallard ducks of several *thousand* ppm.

TAMS/Gradient estimates of PCBs in the fish diets of herring gulls are in the same order of magnitude as the USFWS guidelines and lower than those levels causing reproductive impairment, as suggested by other studies. Additional monitoring data will be required if PCB uptake in birds is to be assessed.

Transfers of PCBs to developing eggs have been implicated as sensitive toxicological endpoints in several species of birds. Kubiak (pers. comm., 1991) reports that PCB residues of 0.4 ppm in eggs of chickens decreased hatch success.

Eisler (1986) indicates that 16 ppm in turtle doves caused delayed development. No site-specific information is available on the PCBs in eggs of local birds. BAF values relating PCBs in the diet to PCBs in herring gull eggs were taken from the literature and used to estimate PCB levels as high as 32 - 800 ppm. This estimate, however, is highly uncertain. Furthermore, no information is available on the effects of PCBs in the eggs of herring gulls.

B.7.5.5 Mammals

Like the bird pathway, possible dietary exposure levels of PCBs for fish-eating mammals have not been monitored. Assuming 50 percent of their diet consists of fish from the Upper Hudson, mink diets may contain on the order of 1 - 25 ppm PCBs. This dietary PCB concentration exceeds the 0.64 ppm level shown to cause reproductive failure (Planatow and Karstad, 1973). These estimated intake levels also exceed the NYSDEC fish flesh criteria of 0.13 ppm (Newell, *et al*. 1987).

B.7.5.6 Summary

Measured baseline (1986-89) PCB concentrations in surface water of the Upper Hudson exceed the USEPA ambient water quality criterion by as much as a factor of five. Sediments, for which there are no clearly defined PCB criteria, are contaminated with PCBs far above the <1 ppm guidelines outlined by NOAA as levels indicating biological effects. Levels of PCBs in Upper Hudson fish exceed the USFWS recommended guidelines for trout by approximately an order of magnitude. Finally, estimated PCB levels in the diets of fish eating birds and mammals at the site appear to be on the same order or somewhat higher than dietary levels recommended by USFWS and NYSDEC.

Aside from the comparison with published toxicity values and measured PCB concentrations in Hudson River sediments, water and biota, no quantitative ecological risks attributable directly to PCBs can be adequately presented based on available site data. Future phases of the reassessment will address data limitations and better define ecological risks due to PCBs in the River.

SYNOPSIS

PHASE 1 FEASIBILITY STUDY

(Sections C.1 through C.7)

The purpose of the Feasibility Study (FS) for the Hudson River PCB site is to identify and evaluate alternatives for mitigating PCB contamination and controlling its effects on public health or the environment. By selecting from the range of technologies available for site clean-up, a response can be formulated that is technically feasible, protects public health and the environment, is cost-effective and is consistent with applicable or relevant and appropriate requirements (ARARs). This Part provides the results of initial steps taken to date as part of the comprehensive Feasibility Study for the project. Three aspects of the FS are discussed: (1) remedial objectives and response actions; (2) potential clean-up technologies; and (3) an initial screening of technologies.

National legislation has established the purposes of remedial actions and the process by which remediation alternatives are developed and evaluated (C.1). Remedial objectives and potential, general response actions are explained (C.2).

Section 121(d) of SARA and the NCP require that CERCLA remedial actions comply with all federal ARARs. State requirements must also be attained, if they are legally enforceable and consistently enforced statewide. A listing of the specific federal and state requirements (C.3) cover three kinds of ARARs: chemical-specific (govern the extent of site cleanup); location-specific (pertain to existing site features); and action-specific (pertain to proposed site remedies and govern implementation of the selected site remedy).

The technologies and processes identified (C.4) are: containment, natural PCB biodegradation in sediments, removal, disposal and treatment technologies. The treatment technologies include physical, chemical, thermal and biological treatment. Innovative technologies (C.5) are also reviewed.

An initial screening of technologies is performed (C.6). Although no particular technology has been eliminated from further consideration in Phases 2 and 3, preliminary judgments concerning remedial options are presented. Since the decision to remediate the Upper Hudson's PCB-contaminated sediments will be significantly influenced by the feasibility and costs of the technologies, recommendations for treatability studies are also presented (C.7).

Part C initiates the feasibility study process, which will continue through Phase 2 as additional site characterization work is performed and will be finalized in Phase 3.

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PART C

PHASE 1 REPORT

INTERIM CHARACTERIZATION AND EVALUATION

HUDSON RIVER PCB REASSESSMENT RI/FS

CONTENTS

PART C

CONTENTS
(continued)

PART C

CONTENTS
(continued)

PART C

CONTENTS
(continued)

TABLES

Tables Located in Volume 1 (2 of 2)

PART C

CONTENTS
(continued)

FIGURES

Figures Located in Volume 1 (2 of 2)

PART C

CONTENTS
(continued)

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C. PHASE 1 FEASIBILITY STUDY

C.1 Introduction

Remedial actions, as defined by 300.5 of the National Contingency Plan (NCP), are those responses to releases that are consistent with a permanent remedy to protect against or minimize release of hazardous substances, pollutants or contaminants so they do not migrate to cause substantial danger to current or future human health and welfare or the environment.

In formulating a remedy, CERCLA requires USEPA to emphasize risk reduction through destruction or treatment of hazardous waste. Section 121 of SARA establishes a statutory preference for remedies that permanently and significantly reduce the mobility, toxicity or volume of hazardous waste over remedies that do not use such treatment. Section 121 also requires USEPA to select a remedy that is protective of human health and the environment, is cost-effective and utilizes permanent solutions and alternative treatment technologies to the maximum extent practicable. Furthermore, Section 121 requires that, upon completion, remedies must attain applicable or relevant and appropriate requirements (ARARs), unless specified waivers are granted.

Section 300.430 of the NCP, in conjunction with USEPA guidance on conducting a Feasibility Study (FS), sets forth the development and evaluation process for remedial alternatives (USEPA, 1988). This process consists of the following steps:

- Identify the nature and extent of contamination and threat presented by the release (300.430[d][2]);
- Identify general response objectives for site remediation (300.430[e][2][i]);
- Identify and evaluate remedial technologies potentially applicable to wastes and site conditions (300.430[e][2][ii]);

- Develop alternatives to achieve site-specific response objectives (300.430[e][2][iii]);
- Conduct initial screening of alternatives (300.430[e][7]); and
- Conduct detailed analysis of alternatives (300.430[e][9]).

As an initial step, both CERCLA and the NCP require identification of the nature and extent of site contamination. The nature and distribution of contamination and the threat posed by the release of contaminants from the Upper River has been presented in Part B. Phase 2 will continue to define the nature and extent of contamination. This part identifies potential clean-up technologies and presents an initial screening of these technologies.

Figure C.1-1, an overview of the FS process as described above, highlights the Phase 1 FS activities reported here.

C.2 Remedial Objectives and Response Actions

Remedial action objectives are developed in order to set goals for protecting human health and the environment early in the alternative development process. The goals should be as specific as possible, but should not unduly limit the range of alternatives that can be developed.

In the original 1984 FS, the contaminant of interest was PCBs, which remain the specific contaminant of interest for this reassessment. The media of interest are the Upper Hudson River sediments.

It is apparent from the preliminary assessments that the impact of PCBs on aquatic life and consumers of aquatic life will drive the clean-up of the site. Target levels for clean-up will need to be determined once these assessments are complete.

General response actions for remediation of the Hudson River PCB site include the following:

1) No Action;

2) Containment;

3) *In situ* treatment;

4) Complete or partial removal with on-site or off-site disposal; and

5) Removal with on-site or off-site treatment and disposal.

Table C.2-1 lists general response actions for the Hudson River site and identifies the technologies screened for each response action. A no action alternative will be considered throughout each phase of the FS. Under the no action alternative, contaminated river sediments would be left in place without treatment or containment. Institutional controls such as fishing bans, site access restrictions and monitoring may be continued.

Remedial technologies and processes associated with the general response actions are discussed in the following sections. A preliminary assessment of each technology's applicability and state of development is also provided in order to focus succeeding FS phases and the Phase 2 field investigations.

C.3 Potentially Applicable or Relevant and Appropriate Requirements (ARARs)

C.3.1 Definition of ARARs

Section 121(d) of SARA and the NCP require that CERCLA remedial actions comply with all federal ARARs. State requirements must also be attained under Section 121(d)(2)(c) of SARA, if they are legally enforceable and consistently enforced statewide. ARARs are used to determine the appropriate extent of site clean-up, identify and formulate remedial action alternatives and govern the implementation and operation of the selected action. According to SARA, requirements may be waived by USEPA, provided protection of human health and the environment is still assured, under the following six specific conditions:

- The selected remedial action is an interim remedy;
- Compliance with such requirements will result in greater risk to human health and the environment than alternative options;
- Compliance with such requirements is technically impracticable from an engineering perspective;
- The selected remedial action will provide a standard of performance equivalent to other approaches required under applicable regulations;
- The requirement is a state requirement that has been inconsistently applied; or
- Attainment of the ARAR would entail extremely high costs relative to the added degree of reduction of risk afforded by the standard (i.e., fund balancing).

To consider ARARs and, more importantly, to incorporate consideration of ARARs in the FS and remedial response processes, the NCP and SARA have defined both applicable requirements and relevant and appropriate requirements, as described below.

C.3.1.1 Applicable Requirements

Applicable requirements are those federal and state requirements that would be legally applicable, either directly or as incorporated by a federally authorized state program, if response actions were not taken pursuant to Sections 104 or 106 of CERCLA.

Requirements that are applicable to and have jurisdiction over given situations are considered "applicable requirements." An example of an applicable requirement would be MCLs for a site that exhibits groundwater contamination entering a public water supply.

C.3.1.2 Relevant and Appropriate Requirements

Relevant and appropriate requirements are those federal and state requirements that, while not legally "applicable," can be applied if, in the decision-maker's best professional judgment, site circumstances are sufficiently similar to those situations that are jurisdictionally covered and use of the requirement makes good sense. During the FS process, relevant and appropriate requirements are intended to have the same weight and consideration as applicable requirements.

The term "relevant" was included so that a requirement initially screened as nonapplicable because of jurisdictional restrictions would be reconsidered and, if appropriate, included as an ARAR for the site. For example, MCLs would be nonapplicable, but relevant and appropriate for a site that exhibited groundwater contamination in a potential (as opposed to an actual) drinking water source.

C.3.1.3 Other Requirements To Be Considered

Other requirements to be considered are federal and state nonregulatory requirements *e.g.*, guidance documents or criteria.

Nonpromulgated advisories or guidance documents do not have the status of ARARs. However, where there are no specific ARARs for a chemical or situation or where such ARARs are not sufficient to be protective, guidance or advisories should be identified and used to ensure that a remedy is protective.

C.3.2 Development of ARARs

Under the description of ARARs set forth in the NCP and SARA, many federal and state environmental requirements must be considered. These requirements include ARARs that are:

- chemical-specific (i.e., govern the extent of site clean-up)
- location-specific (i.e. pertain to existing site features)
- action-specific (i.e., pertain to proposed site remedies and govern implementation of the selected site remedy)

C.3.2.1 Chemical-Specific ARARs

Chemical-specific ARARs govern the extent of site clean-up and provide either actual clean-up levels or a basis for calculating such levels. For example, surface water criteria and standards, as well as air standards, provide necessary clean-up goals for the Hudson River PCB contaminant.

Chemical-specific ARARs are also used to indicate acceptable levels of discharge to determine treatment and disposal requirements and to assess the effectiveness of remedial alternatives. Table C.3-1 lists and summarizes potential chemical-specific ARARs. Chemical-specific ARARs will apply to every alternative developed in later phases.

C.3.2.2 Location-Specific ARARs

Location-specific ARARs pertain to natural site features such as wetlands and floodplains, as well as manmade features, including existing landfills, disposal areas and historic buildings. Location-specific ARARs generally place

restrictions on the concentration of hazardous substances or the conduct of activities, because of the site's particular characteristics or location. Consideration of such ARARs provides a basis for assessing existing site conditions and subsequently aids in defining potential location-specific ARARs. Potential location-specific ARARs are presented in Table C.3-2.

C.3.2.3 Action-Specific ARARS

Action-specific ARARS are usually technology- or activity-based limitations that control actions at CERCLA sites. After remedial alternatives are developed, action-specific ARARs pertaining to proposed site remedies provide a basis for assessing the feasibility and effectiveness of the remedies. For example, action-specific ARARs may include hazardous waste transportation and handling requirements, air and water emissions standards and landfilling and treatment requirements of TSCA and RCRA. Potential action-specific ARARs are presented in Table C.3-3.

C.3.3 Statutes and Regulations

More detailed descriptions of some of the federal and state statutes referenced in Tables C.3-1, C.3-2 and C.3-3 are presented below.

C.3.3.1 Federal Statutes and Regulations

Toxic Substances Control Act (TSCA) 15 USC 2601, 40 CFR 761

TSCA provides USEPA with authority to require testing of both new and existing chemical substances entering the environment and to regulate them where necessary. TSCA requirements do not apply to PCBs at concentrations less than 50 ppm; PCBs can not be diluted, however, to escape TSCA requirements.

TSCA establishes prohibitions and requirements for the manufacturing, processing, distribution in commerce, use, disposal, storage and marking of PCBs. Section 2605 includes provisions for incineration, disposal, storage for

disposal, chemical waste landfills, decontamination, clean-up policy, recordkeeping and reporting for PCBs.

Subpart D of 40 CFR 761 contains the following applicable provisions regarding PCBs:

- 40 CFR 761.60(a)(5) states that "all dredged material and municipal sewage treatment sludge that contain equal or greater than 50 ppm of PCBs shall be disposed of:

  (i) in an incinerator pursuant to section 40 CFR 761.70,
  (ii) in a chemical waste landfill described in 40 CFR 761.75, or
  (iii) by a disposal method approved by the Agency's Regional Administrator in the region in which the PCBs are located."

  Applications for disposal by methods other than those specified in subparagraphs (i) and (ii) above may be made in writing and sent directly to the Regional Administrator. Application procedures and contents of the application are addressed in 40 CFR 761.60 (5)(iii).

- 40 CFR 761.70 covers the incineration of PCBs. Incinerators for the burning of PCBs must be approved by the appropriate USEPA Regional Administrator or the Director, Exposure Evaluation Division, pursuant to section 40 CFR 761.70 (d), which lists application requirements.

- 40 CFR 761.75 applies to facilities used to dispose of PCBs. In general, a chemical waste landfill for PCBs must be approved by the Agency Regional Administrator. The landfill must meet technical requirements, which include, but are not limited to, the following: soil consistency surrounding the landfill, flood protection, topography and appropriate record maintenance. 40 CFR 761.75 (b).

- 40 CFR 761.60(a)(6) provides that PCB articles with concentration levels equal to or greater than 50 ppm must be stored prior to disposal in compliance with section 40 CFR 761.65.

- 40 CFR 761.65 states that PCB articles must be removed from storage within one year from the time they were placed in storage. In addition, the regulation lists storage facility requirements and container requirements. An exemption from this regulation exists where PCBs are stored for 10 days or less at a transfer facility. This section may be applicable should dredged materials be stored before incineration or landfill disposal.

USEPA has proposed amendments to the storage and disposal rules for PCBs in 40 CFR 761, subpart D. The tentative changes were published at 55 Fed. Reg.

46,470 (1990). The purpose of this proposed rule is to set out criteria and procedures for revoking and suspending PCB storage and disposal approvals. Current USEPA rules require persons operating PCB disposal facilities to obtain an approval for such activities. These rules do not, however, prescribe when or how such approvals may be suspended or revoked.

Resource Conservation and Recovery Act (RCRA) - 42 USC 6901, 40 CFR 260

RCRA is an applicable statute because it establishes a cradle to grave regulatory program for present hazardous waste activities. RCRA mandates the national policy on how hazardous waste will be stored, treated and disposed of to alleviate any potential threat to human health and the environment. PCBs alone are not a RCRA hazardous waste; they are, however, subject to land disposal restrictions, as discussed below, to the extent that the waste would otherwise be considered hazardous under RCRA.

40 CFR 264 lists standards applicable for an owner or operator of hazardous waste treatment, storage and disposal facilities. This part includes, but is not limited to, general facility standards, releases from the facility contingency plan and emergency procedures to the generator of hazardous waste, landfills and incinerators.

40 CFR 268 imposes land disposal restrictions. The purpose is to restrict hazardous wastes from land disposal and to define limited circumstances in which a hazardous waste may be disposed. Subparts include a schedule for land disposal prohibitions and establishment of treatment standards, prohibitions on land disposal, treatment standards and prohibitions on storage.

40 CFR 270 establishes the USEPA-administered permit programs, *e.g.*, hazardous waste permit programs. This regulation subparts set forth permit application conditions and procedures. Specifically, Subpart F part 270.62 addresses hazardous waste incinerator permits.

Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) 42 USC 9601, 40 CFR 372

Under CERCLA 101 (14) a "hazardous substance" is any substance USEPA has designated under specified sections of the Clean Water Act, Clean Air Act or TSCA, and any "hazardous waste" under RCRA. A list of such hazardous substances, which includes PCBs, may be found at 40 CFR part 302.

At 40 CFR 372 are set forth requirements for the submission of information relating to the release of toxic chemicals under section 313 of Title III of SARA. The information collected under this part is intended to inform the general public and communities surrounding regulated facilities about releases of toxic chemicals. "Release" is defined as any spilling, leaking, pumping, pouring, emitting, emptying, discharging, injecting, escaping, leaching, dumping or disposing into the environment (including the abandonment or discarding of barrels, containers and other closed receptacles) of any toxic chemical. Notification requirements are set forth within this section, as well a toxic chemical listing, which includes PCBs. The section is applicable, since there is the possibility that PCBs could be released into the surrounding area in which the remedial action is occurring.

Clean Air Act - 42 USC 7401

This Act promotes the protection and enhancement of the quality of the nation's air resources through research and development and air pollution control and prevention. For example, if the remedial action includes incineration of sediments, then the Act may be relevant and appropriate.

42 USC 7412 establishes national emission standards for hazardous air pollutants. The term "hazardous air pollutant" means an air pollutant that may reasonably be anticipated to result in an increase in mortality or an increase in serious irreversible, or incapacitating reversible illness. In addition, the regulation provides operational standards, designs, equipment and work standards. PCBs have not previously been listed as a hazardous air pollutant. However,

under Title III of the Clean Air Act Amendments of 1990, PCBs (Aroclors) are included on the list of toxic air pollutants for which USEPA will have to establish standards.

C.3.3.2 New York State Statutes and Regulations

New York Environmental Conservation Law Article 11, Title 5

This statute is applicable, because it regulates the disposal of pollutants into the Hudson River, but it does not specifically address PCB levels.

The statute prohibits the disposal of "deleterious or poisonous substances" into any public or private waters which affect the welfare of wildlife, waterfowl and fish inhabiting those waters. The statute does not define the term "deleterious or poisonous substance" nor does it define, "disposal." This statute also provides that "oil, acid, sludge, cinders or ashes from a vessel of any type shall not be thrown, dumped or allowed by any person to run into the waters of the Hudson or Mohawk rivers." Finally, the statute prohibits the disposal of earth, soil, refuse or other solid substances into streams or tributaries inhabited by trout. The statute does not establish the maximum acceptable levels for PCBs.

New York Water Classification and Quality Standards

This statute is applicable as it specifically lists the permissible ambient water quality standard for PCBs and provides water classifications for the Upper and Lower Hudson.

The statute provides water classifications for the protection and propagation of fish, shellfish and wildlife, and for recreation in and on water, and takes into account the use and value of water supplies for: public, agricultural, industrial and other purposes, including navigation. The classification of water depends on permissible usage. Differences in water classifications are determined according to what discharge is made to the water and where the water discharge flows.

For each classification, a best usage standard of water is established. Within each classification, the standards for determining water quality, which include toxic waste and deleterious substances are established. The standards for determining water quality must be consistent with the best usage of the water. In addition to these general standards, parts 702.3-702.4 create special classifications and standards for the Upper and Lower Hudson.

Ambient water quality standards are listed in Appendix 31; the established standards for PCBs applicable to the classification of the Upper and Lower Hudson River are as follows:

PCBs - Human Life and Other Usage	0.01 ppb
PCBs - Aquatic Life and Other Usage	0.001 ppb

Groundwater Classifications

This statute is applicable as it is specific to PCB concentrations in groundwater. It provides groundwater quality standards and water classes. The maximum level for PCBs is 0.1 ppb.

New York State Constitution Article XV, Section 1-4

Article XV provides that the state has sole ownership and management of the canal system. The state has the power of granting revocable permits for the occupancy or use of such barge canal lands or structures. The state may lease or may transfer the barge canal and terminals and facilities to the federal government. The lease or transfer to the federal government may be for the purposes of operation, improvement and inclusion in the national system of inland waterways, including flood control, conservation and utilization of water resources. This arguably would include dredging for remediation purposes.

The above article from the New York Constitution refers to the Canal Law Article XV (1938), which provides that the acquisition of land and waters necessary for the improvement, maintenance or repair of the canal system shall be through appropriations by the New York Superintendent of Public Works.

New York Environmental Conservation Law Article 21, Title 5

The Tri-State compact attempts to control pollution around the Lower Hudson through the Interstate Sanitation Commission. The compact is relevant and appropriate.

This statute was created to control future pollution and to abate existing pollution in the waters of the densely populated Tri-state district bordering New York, Connecticut and New Jersey. In general, the states agree that no sewage or other polluting matters will be discharged or will be permitted to flow into the waters of the district. Each state signatory pledges to act in full cooperation through the Interstate Sanitation Commission, which has jurisdiction to enforce the provisions of the compact. The compact specifically includes the Lower Hudson River in the district. The compact does not further define "sewage or other polluting matters," nor does it establish maximum concentration levels for PCBs.

New York Environmental Conservation Law Article 24, Freshwater Wetlands

The purpose of this statute is to declare the public policy of New York State regarding the preservation, protection and conservation of freshwater wetlands.

Regulated activities, among others, are: dredging, draining excavation and removal of sand, soil, mud, shells, gravel and other aggregate from any freshwater wetland. Dredging or filling of navigable waters of the state must be done pursuant to this article and any other applicable law. The statute does not more explicitly define navigable waters.

New York Environmental Conservation Law Article 25, Tidal Wetlands

The regulation protects the state's interest in tidal wetlands. The regulations cover activities such as dredging, draining and excavation of tidal wetlands.

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C.4 Technology and Process Identification

Technologies are presented here in the following order: *in situ* technologies including containment and treatment; dredging and excavation methods; sediment and treatment techniques, including physical, chemical, thermal and biological; and disposal alternatives. Several emerging sediment treatment systems are discussed at C.5.

C.4.1 Containment

In situ control and containment measures are intended to reduce dispersion and leaching of contaminated sediments to other areas of a water body. Methods include retaining dikes and berms and capping. Retaining dikes and berms include earthen embankments, bulkheads and sheet pile walls. The structures can be constructed perpendicular to the direction of stream flow to prevent suspended particulate matter from flowing downstream or parallel to a river bank.

A wide variety of materials can be used to cap contaminated sediments in order to minimize leaching and prevent their erosive transport. Cover materials include inert materials such as silt, clay, sand, cement, or a geotextile; alternatively, active materials can be applied to the surface or mixed with the sediment in an attempt to limit mobility. Issues such as the capping material's susceptibility to scour and resuspension, ability to withstand leaching, and effect on the ecosystem must be studied in order to determine suitability for a given site. Capping may increase anaerobic activity and has been used as a component of bioremediation programs.

Containment using capping was the remedial action chosen for the Hudson River remnant deposit sites. A manufactured soil cover material (claymax) is being used along with two feet of soil cover at these deposit sites. Work has now largely been completed on that project and post construction monitoring will begin during this year (1991).

C.4.2 Natural PCB Biodegradation in Sediments

C.4.2.1 Aroclor Patterns

Much study by GE and other researchers has focused on the issue of naturally occurring, biologically mediated PCB transformations in Hudson River sediments. Laboratory controlled experiments have been conducted in an attempt to confirm dechlorination or biodegradation processes. Monitoring data have also been evaluated specifically with respect to addressing the question of whether biological transformations are responsible for the altered appearance of the PCBs found in the river versus the original PCBs (Aroclor mixtures) discharged to the river.

In reviewing the data from Hudson River sediment sample analyses, Brown, Jr *et al.* (1984) have observed that the concentration distribution of PCB congeners in sediment varies from that in the Aroclor mixtures (predominantly 1242) believed to have been originally deposited there. Uncertainty in the amount of different Aroclors that actually entered the river, differential partitioning and transport of different PCB congeners and biodegradation are all contributing factors to this observation.

In 1984, Brown, Jr. *et al.* analyzed sediment samples taken from eight surface locations and two sediment cores in River Reach 8; an additional surface sample was collected from Reach 6. They reported that a number of different PCB congener distributions could be distinguished in the gas chromatograms generated from the analysis of these samples. The three primary patterns observed were:

- Pattern A, which looked very much like Aroclor 1242 but showed somewhat increased portions of penta- and hexachlorinated PCBs, somewhat diminished portions of tetra- and trichlorobiphenyls and markedly diminished proportions of mono- and dichlorobiphenyls;
- Pattern B, which showed a reduction in levels of penta-, hexa- and some tetrachlorobiphenyls and more mono- and dechlorinated congeners as compared to Pattern A; and

- Pattern C, which was distinguished from pattern B by less reduction in the levels of penta-, hexa- and tetrachlorobiphenyls, but greater reduction in the levels of tri- and some dichlorobiphenyls, and much greater concentrations of mono- and some dechlorinated congeners.

C.4.2.2 Aerobic Biodegradation of PCBs

Brown, Jr. *et al.* (1984) attributed Pattern A to aerobic biodegradation of Aroclor 1242 for several reasons. Pattern A was typically associated with surface deposit. Aerobic microbial degradation of mono- and dechlorinated PCB congeners had been reported in laboratory experiments (Furukawa, 1982; Shiaris and Sayler, 1982; Kong and Sayler, 1983; Safe, 1984). Bacteria capable of degrading some tri- and tetra-chlorinated PCB congeners were detected in every Reach 8 sediment sample collected. The PCB congener distributions in Hudson River surface sediment samples looked more similar to that produced from Aroclor biodegradation by the most commonly encountered microbial populations than by mammalian mixed function oxidases (MFOs), by anaerobes or by water extraction. The microbial activity observed by Brown, Jr. *et al.* (1984) and others (Furukawa, 1982; Safe, 1984; and Bedard, 1990) generally decreased with the number of chlorine atoms on the molecule. Thus, an environmental sample containing a biodegraded Aroclor mixture would be expected to show a pattern of PCBs like that of Pattern A, in which the lesser chlorinated congeners are reduced in concentration relative to the more chlorinated congeners. Non-biologically mediated process, such as selective dissolution or partitioning of congeners, are other mechanisms that might also explain the Pattern A congener distribution.

More recent work supports the conclusion that biodegradation may account for at least some of the observed patterns of PCB concentration distributions in surface sediments. Bedard *et al.* (1986, 1987a and 1987b), Sing *et al.* (1988), and Unterman *et al.* (1988) have shown in laboratory studies that there are at least 25 different naturally occurring microbial strains capable of degrading one or more PCB congeners. The congener specificities exhibited by these strains suggests that the initial aerobic microbial attack on PCBs is mediated by one of

two types of enzymes, a 2,3- or a 3,4-dioxygenase (Abramowicz, 1990; Bedard, 1990).

While Pattern A may be the result of aerobic PCB biodegradation in surface sediments, the *rate* at which biodegradation is occurring in Hudson River sediments is extremely difficult to determine. Attempting to extrapolate a rate from existing concentrations is not possible, since the initial concentrations of PCB congeners in surface sediments are unknown and the present concentrations have undoubtedly been affected by sediment scouring, sediment deposition and sediment-water partitioning, as well as biodegradation. Laboratory measured biodegradation rates can not be extrapolated to the river, because of differences in temperature and nutrient concentrations. No time series data, which might allow estimation of rates *in situ*, have been systematically collected from the river. Furthermore, it is likely that different biodegradation rates are occurring in different areas of the river, as a consequence of the presence of different microbial populations and nutrient concentrations at different locations.

Brown, Jr. *et al.* (1984) have suggested, based on the levels of Aroclor 1221 and 1016 reported in fish between 1977 and 1981 (Sloan *et al.* 1983), that the mono- and dichlorinated congeners are being biologically degraded from surface sediments with a half-life of one to two years. They note, however, that the rate of Aroclor 1221 and 1016 disappearance reflects the disappearance of congeners that can be degraded by a 2,3-dioxygenase; the degradation of other mono- and dichloro congeners is likely slower. This estimate is subject to several sources of error: the small number of time points (five) from which it was made; the variability inherent in biological data; and insufficiently detailed data to rule out factors other than biodegradation, *e.g.* evaporative loss of lesser chlorinated congeners from the water column.

Even if an aerobic biodegradation rate could be determined with some accuracy, such a rate, by definition, would be characteristic of only the upper most oxygenated layers of sediment. The rate in the remainder of the sediment

column would be essentially zero. Such depth limitations to aerobic biodegradation may partially explain the fact that most of the surface sediment samples discussed by Brown, Jr. *et al.* (1984, 1987b) actually showed patterns that resembled a mix of surface and subsurface sediment rather than Pattern A exclusively; these samples typically had higher levels of mono- and dichlorinated congeners than found in Aroclor 1242.

C.4.2.3 Anaerobic Dechlorination

Brown, Jr. *et al.* (1984, 1987a, 1987b, 1988) have suggested that anaerobic dechlorination produces Patterns B and C in the subsurface. The evidence they present to support their hypothesis is four-fold. First, Patterns B and C were reportedly associated with subsurface hot spots. Second, sediments exhibiting these patterns showed changes in homologue distribution from that of Aroclor 1242 corresponding to one-third of the chlorine lost. Third, subsurface sediments tended to have PCB congener distributions, *i.e.* a greater percentages of mono- and dichlorobiphenyls, which were difficult to explain on the basis of selective extraction of the lesser chlorinated congeners upstream with redeposition downstream. Fourth, there was a selective loss of *meta* and *para*, as opposed to *ortho* chlorines in Patterns B and C, suggestive of a biological specificity. The hypothesis of Brown, Jr. *et al.* is supported by the work of Quensen *et al.* (1988, 1990), who have shown that a bacterial inoculum cultured from Hudson River sediments can dechlorinate PCBs when incubated under an N_2-CO_2 atmosphere, at 25°C.

The ability of microorganisms from Hudson River sediments to dechlorinate PCB mixtures reductively has been confirmed by researchers at GE's Biological Sciences Laboratory, the University of Michigan, New York University Medical Center and Wadsworth Center for Laboratories and Research (Abramowicz *et al.*, 1989; Vogel *et al.*, 1989; Alder *et al.*, 1990; Rhee and Bush, 1990). Abramowicz *et al.* (1989) have shown the sequential, microbially mediated removal of chlorines from 2,3,4,3'4,-pentachlorobiphenyl to form tetra-, tri-, di-, and

monochlorobiphenyls. To date, however, no anaerobic species demonstrating PCB dechlorination have been isolated (see review by Abramowicz, 1990).

While it seems plausible that reductive dechlorination is occurring in the subsurface sediments of the Upper Hudson, the average rate at which this degradation takes place is difficult to determine and may be fairly slow. The rate may be expected to slow further as the more reactive congeners are dechlorinated and the more recalcitrant congeners accumulate (Brown, Jr. *et al.*, 1984). Chen *et al.* (1988) found no evidence of anaerobic PCB congener biodegradation in sediments, unless they were incubated with the addition of cultured bacteria. Rhee *et al.* (1990) detected no decrease in individual PCB congener concentrations after anaerobic incubation of Moreau sediments *in situ* for seven months. Brown, Jr. *et al.* (1984) have calculated very rough half-lives of ten years for the penta- and tetra- chlorinated congeners and two to three years for the reducible tetra-, tri- and dechlorinated congeners, based on a single core available (core 18-6) for analysis.

Figure C.4-1 shows the weight percentages of congeners with one, two, three, four or five chlorines reported for core 18-6 as a function of depth. Since the core was collected in 1977 and sediments at 19 inches were tentatively dated at 1952, this core represents sediments deposited over the course of approximately 25 years, with progressively older sediments at greater depths. Between the surface and eight inches, Brown, Jr. *et al.* characterize the PCB pattern in the core as a mixture of Patterns A and B. Thus, the changes in congener percentages over this section of the core may reflect a mixture of aerobic and anaerobic processes. Between a depth of 8 and 19 inches, however, the core is characterized by Pattern B alone. If the ratio of various Aroclor mixtures input to the river and the sedimentation rate were relatively constant over this time period and if reductive dechlorination is producing Pattern B, this section of the core should show progressively smaller percentages of the more chlorinated congeners with contaminant increases in the percentages of terminal dechlorination products. The weight percentages of all congeners, however, actually remain relatively constant through this section. This finding

suggests either that the input of Aroclor mixtures was *not* constant, that the source of Pattern B is *not* reductive dechlorination or that limited dechlorination occurred over the time period represented by this section of the core.

C.4.3 Removal Technologies

Removal technologies include dredging and excavation. Excavation techniques are not discussed further in this report.

Dredging system alternatives have been evaluated extensively by the US Army Corps of Engineers, in general, and by USEPA and numerous consultants for specific Superfund Projects. Dredging systems identified in the literature fall into the hydraulic, mechanical and specialty-type categories, with each category serving particular applications. Fugitive sediment releases from dredging equipment will be evaluated in subsequent phases, utilizing published reports.

The cutterhead hydraulic pipeline dredge is among the most commonly used dredging systems. By combining mechanical cutting action with hydraulic suction, this dredge has the capability of efficient excavation and removal of material to disposal sites without rehandling. Material would be drawn-up by the dredge and pumped at an appropriate solids concentration through a floating or submerged pipeline to shore and to ultimate disposal. The system would include booster pumps and tugs and would necessitate a water treatment plant capable of handling large quantities of water.

The clamshell mechanical dredging system consists of barge-mounted cranes outfitted with suitable clamshell buckets. Excavated material is placed in scows or hopper barges for transport to the disposal site. In case of the Hudson River project, the barge contents would probably be slurried for removal by a hydraulic pump-out system located on shore. Where circumstances permit, bottom dump scows can be used in concert with a mechanical system and would discharge dredged material at sub-aqueous disposal sites.

Results of recent field studies conducted by the US Army Corps of Engineers Waterways Experiment Station indicated that the cutterhead dredge was the most successful in limiting sediment resuspension into the water column, followed by the hopper and clamshell dredges. Modifications such as overflow prevention or use of an enclosed bucket may improve resuspension characteristics of the hopper and clamshell dredges. Specialty dredges were also tested (the modified dustpan and matchbox dredges) and compared with the cutterhead. No reduction in sediment resuspension was found with use of the specialty dredges.

Historically, contaminated sediments were removed from the river during NYSDOT's routine channel maintenance dredging. As the river's PCB problem became better understood, remedial alternatives, including bank-to-bank dredging of the river, full-scale dredging of the 40 PCB hot spots in the river and reduced-scale dredging of the most contaminated hot spots, were considered. Due to limited funding under the Clean Water Act, a reduced-scale dredging program had been recommended by the USEPA and the NYSDEC in earlier studies. The NYSDEC currently has an Action Plan for site remediation that incudes dredging and encapsulation of river sediments at an upland site in proximity to the river (Site 10).

C.4.4 Treatment Technologies

C.4.4.1 Physical and Chemical Treatment Technologies

Evaluation

To date, incineration and disposal in landfills are the most widely practiced and permitted methods for management of PCB-contaminated soils and sediment. However, other technologies have now emerged and are considered technically and economically feasible alternatives to incineration and landfilling in certain circumstances. In this section, a range of physical and chemical treatment technologies and their general applicability to the Hudson River site and level of development are presented.

Technologies screened in previous studies of the Hudson River, including the NUS Feasibility Study Report (1984), the Engineering Report by Malcolm Pirnie, Inc. (MPI, 1985) as part of the Hudson River PCB Reclamation/Demonstration Project and a Research Triangle Institute Report (1987) are reexamined herein. The Ebasco Feasibility Study for the New Bedford Harbor site (1990) has also been reviewed, as have emerging technologies demonstrated through the USEPA Superfund Innovative Technology Evaluation (SITE) Program.

The technologies discussed include: 1) those that were retained in the previous studies for further test and evaluation and have been developed to at least a pilot-scale operation; 2) those not retained in previous studies, but which have undergone significant development since the previous studies; and 3) those new technologies that have recently emerged and which have been demonstrated on a pilot-scale in the USEPA SITE program.

Table C.4-1 lists the treatment technologies reviewed and screened in the NUS and MPI studies. The majority of technologies reviewed at the time of publication of the two reports were in the early stage of development; little information was known about their environmental effects and costs. The preferred physical/chemical technologies in the NUS study were KOHPEG and wet air oxidation. Results of the MPI study indicated that none of the technologies evaluated had been demonstrated to treat PCB-contaminated waste effectively and that further research was required before a preferred method could be selected for a project of the magnitude of the Hudson River project.

In 1987, the Research Triangle Institute (RTI), under USEPA sponsorship, conducted a study to identify emerging technologies and their level of development. From the initial screening of 64 treatment technologies, eleven processes, of which ten were physical/chemical processes and one was biological, were selected for further technological assessment. Table C.4-2 lists the ten physical/chemical processes.

According to RTI, a significant number of solvent washing/extraction processes had emerged since the NUS Feasibility Study was published in 1984. The solvent washing/extraction processes are physical treatment technologies which do not destroy PCBs, but transform them from a solid matrix to a solvent matrix. PCBs extracted to the solvent matrix can be further separated from the solvent, forming a concentrated PCB stream, which then requires final disposal. In comparison to thermal systems, solvent extraction may be easier to permit, since it is not subject to the same regulatory restraints as thermal treatment. It may also be economically feasible for treating a wider range of PCB wastes and can be less costly with respect to energy usage. RTI recommended that three technologies be retained for thorough testing and evaluation, of which two were physical/chemical. These were the Basic Extraction Sludge Treatment (B.E.S.T.) process and the UV/Ultrasonics Technology process. None of these technologies has been applied on a large scale.

In 1990, Ebasco conducted bench/pilot-scale treatment tests on seven selected PCB treatment technologies (Table C.4-3). Five were physical/chemical processes, one was a dewatering process and the other was bioremediation. The two physical/chemical technologies retained for the development of remedial alternatives were the B.E.S.T. process and solidification/stabilization with the use of Portland cement. According to Ebasco, solvent extraction technology (B.E.S.T. process) was not yet fully developed for PCB waste treatment applications. Incineration was still considered the only Best Demonstrated Available Technology (BDAT) for PCB wastes.

Physical and chemical technologies chosen in previous studies for further analysis are summarized below.

PROCESSES	NUS (1984)	RTI (1987)	EBASCO (1990)
KOHPEG	X		
Wet Air Oxidation	X		
B.E.S.T. Process		X	X
UV/Ultrasonics Technology		X	
Solidification/Stabilization			X

To be retained here, a technology must have been preliminarily tested for its environmental compatibility, its cost effectiveness and technical capability. In some cases, however, where technologies have undergone preliminary testing, they have not been developed further. For example, the wet air oxidation process, a preferred technology in the NUS Feasibility Study Report, has been found to be a non-viable application for effective destruction of PCBs in a slurry matrix (J. R. Nicholson, personal communication, March 1991). The UV/Ultrasonics technology, rated as a preferred process in the RTI Report, has not been commercially developed (E. Pedzy, personal communication, February 1991). Many process developers identified in previous reports have left the market.

The physical and chemical treatment technologies that were retained for further analysis in previous studies and that have undergone further development since those studies include:

- KOHPEG Process (Galson Remediation Corporation)
- B.E.S.T. Process (Resources Conservation Company)

Technologies that were not retained for analysis in previous studies, but which have gained significant advancement in technical development include:

- Low Energy Extraction Process (ART International, Inc.)
- Propane Extraction Process (CF Systems, Inc.)

These four technologies are discussed below.

KOHPEG

The KOHPEG process, developed by Galson Remediation Corporation (GRC) of Syracuse, NY, is a chemical treatment technology using potassium hydroxide (KOH) in a solution of mixed polyethyleneglycol (PEG) and a phase-transfer catalyst,

dimethyl sulfoxide (DMSO), to dechlorinate PCBs. The dechlorination reaction takes place in the liquid phase. Therefore, it is necessary to extract the contaminants from the soil surface into the reagent phase, where they can react. The end products are glycol ether and potassium chloride, which are water soluble, low toxicity materials. GRC commercialized the process and has marketed it under the name of APEG-PLUS since 1986; its initial unit is a mobile decontamination facility designed for treating contaminated soil and sludges at a capacity of 20-200 tons per day. The material flow sequences of the system are shown in Figure C.4-2.

GRC conducted a bench-scale test on New Bedford Harbor sediments. The results indicate that PCB removal efficiencies of above 99 percent were achieved for samples containing PCB concentrations of 440 to 7,300 ppm. However, the recovery of reagent and sediment-solids was low, 75 percent for DMSO and 43 percent for solids. This ultimately proved to be a materials handling problem. Later, a separate bench-scale simulation test conducted by GRC on PCB-contaminated soil reported that the recovery of both reagent and solids had improved to 81 percent for DMSO and 102 percent for solids (Galson, 1991). Costs for treating New Bedford Harbor sediments using the APEG-PLUS system were estimated to be $98 and $120 per ton based on 500,000 and 50,000 cubic yards of sediment treated, respectively.

The APEG-PLUS system is primarily used for treating soils. Modifications in the material handling equipment would be required for treating the Hudson River sediments. The moisture content of the sediment may affect the heating requirements and reactor cycle time of the treatment process. The affected operation parameters should be determined through pilot tests using actual site samples. Toxicity of the technique's reaction products and their long-term effect on environmental conditions remain to be confirmed.

B.E.S.T. Solvent Extraction Process

The B.E.S.T. process utilizes triethylamine (TEA) as a solvent. TEA is completely soluble with water at temperatures below 65°F. Above this temperature, TEA and water are only partially miscible. This inverse miscibility property of the solvent is employed to separate PCB-contaminated sediment into PCB/oil, water, and solids fractions. A block diagram for the B.E.S.T. process is presented in Figure C.4-3. The extraction of the contaminated material is conducted at a temperature of approximately 40°F. At this temperature, the TEA freely mixes with the water and the PCB/oil fraction of the sediment matrix. Therefore, the extract solution contains most of the water in the feed and PCB/oil fractions. The extract solution is then heated to temperatures above 130°F. At this elevated temperature, the water separates from the TEA/PCB/oil fraction. The TEA solvent is recovered for reuse from the separated phases via steam stripping. The PCB/oil fraction is disposed of by incineration or chemical dechlorination processes at a permitted facility.

The Resources Conservation Company (RCC) conducted a bench-scale study of its B.E.S.T. process on New Bedford Harbor sediment (Ebasco, 1990). PCB removal efficiencies of above 99 percent were achieved with three extraction stages from initial PCB concentrations of 5,800 and 420 ppm. The PCB concentrations in treated sediment residues were 130 and 11 ppm, respectively, for the high and low initial concentrations. A test with an initial PCB concentration of 11,000 ppm resulted in a residue containing 16 ppm PCB after six extraction stages.

Recently, RCC has completed bench-scale treatability testing with PCB-contaminated soil samples from natural gas pipeline compressor stations. The PCBs in the feed were in the range of 500-2,000 ppm. Removal efficiencies of above 99.8 percent were obtained, leaving residual PCB in the treated soil at less than 2 ppm (Weimer, 1991). Similar PCB extraction efficiencies using the B.E.S.T. process were obtained in other tests. Various PCB levels in the treated residue were reported. The factors that affect the extraction efficiencies and the residual PCB in treated matrices are believed to be the initial concentration of

PCB, stages of extraction and matrix characteristics. Those factors are site-specific determinants, which should be verified by bench or pilot-scale treatability tests.

RCC has recently completed a B.E.S.T. pilot plant configured for processing liquid sludge (sediment) or contaminated soils. The pilot plant is configured with a washer/dryer vessel for extracting and drying soils. The washer/dryer is a horizontal, cylindrical vessel that has a rotating shaft with mixing paddles attached. Performance of the washer/dryer is key to successfully treating sediments with the B.E.S.T. process.

Low Energy Extraction Process (LEEP)

The Low Energy Extraction Process (LEEP) was developed by a New York University research team. Study of its application in removing PCBs from soil, sediment and sludge was funded by USEPA. LEEP technology was being developed and commercialized by Remediation Technology, Inc., which is now Applied Remediation Technology (ART) International, Inc.

The process is a solvent extraction technology based on the combined use of hydrophilic (water miscible) and hydrophobic (water immiscible) solvents. Contaminants (PCBs) are leached from solid material with acetone (a hydrophilic leaching solvent) and then concentrated in kerosene (a hydrophobic extractant) by liquid-liquid extraction. While the acetone solvent is recycled internally, the kerosene containing PCBs is removed from the process for final destruction. Decontaminated solids and water are returned to the environment. A schematic diagram of the primary steps of LEEP is shown in Figure C.4-4.

LEEP was accepted into the SITE Emerging Technologies Program in 1989. ART International, Inc. has conducted bench-scale process simulation (a treatability study) on Waukegan, Illinois harbor sediment. The experiment resulted in selection of acetone as a leaching solvent by virtue of its high leaching efficiency and good settling characteristics for solid fines. Kerosene was chosen as

the liquid-liquid extractant for its good affinity to PCBs, availability, cost, relatively low toxicity and immiscibility in the mixture of water and acetone. The treated sediments contained 42 percent by weight of water and initial PCB concentration in the sample was 33,600 ppm on a dry weight basis. The study showed 99.9 percent PCB removal. A pilot plant with a nominal throughput of 200 lb/hr of dry solids is under construction. Design and engineering of a trailer-mounted commercial unit is scheduled to be completed in 1992.

The LEEP technology uses basic unit operations and is constructed with commercially available equipment. Selection of equipment for application to the Hudson River sediment requires specific engineering analyses. Concurrent Hudson River studies may lead to a better definition of equipment in use. A major disadvantage of LEEP is that the system is not yet available for full-scale applications. Also, feed stream particle size limitation may limit process applicability. Further development and demonstration at a large scale are needed.

Propane Extraction Process

C.F. Systems, a Morrison Knudsen Company in Woburn, Massachusetts, is the developer of the liquefied propane extraction technology. The process uses propane at ambient temperatures and at pressures over 200 psi to extract PCBs along with other oily organics from a sediment-water slurry. C.F. Systems has been operating a commercial-scale unit to treat petroleum refinery sludge at a capacity of 100 barrels per day. This unit was selected to demonstrate their pilot-scale system using New Bedford Harbor sediments. The unit is trailer mounted and is designed to handle pumpable soils, sludge or sediments.

The basic operating steps, shown in Figure C.4-5, include extraction, phase separation and solvent recovery. A mixture of liquefied propane and butane is used as the extracting medium. Pumpable (slurried) solid waste is fed into the top of an extractor. Then the solvent, a propane/butane mix, is condensed by compression and allowed to flow upward through the same extractor. In the

extractor the solvent makes non-reactive contact with waste materials, dissolving the waste's organics. Following this extraction procedure, the residual mixture of clean water or water/solids can be removed from the base of the extractor. The mixture of solvent and organics leaves the top of the extractor and passes to a separator through a valve, which partially reduces pressure. The reduction of pressure causes the solvent to vaporize out of the top of the separator. It is then collected and recycled through the compressor as fresh solvent. The organics left behind are drawn from the separator.

PCB removal efficiencies of 90 percent were achieved for New Bedford Harbor sediments containing PCBs ranging from 350 to 2,500 ppm with up to ten passes or recycles through the treatment unit. The propane extraction process was not retained as a viable technology for that particular site because of problems with materials handling, system operating parameters, extraction efficiencies and low throughput rates observed during the pilot demonstration. Material treated with the propane extraction process must be pumpable. The Hudson River sediments would have to be prepared in a slurry form that can move through the system without clogging. Formation of settled particles or the presence of abrasive particles in the sediment would require special handling equipment in order to keep the material flowing through the system.

C.4.4.2 Thermal Treatment Technologies

Introduction

Thermal treatment is the application of heat to a substance to reduce or eliminate its toxicity. The thermal level can be extreme, *e.g.*, using a plasma-arc for destruction, which generates temperatures in the thousands of degrees, or it can be relatively low, *e.g.*, passing the contaminated substance through a dryer operating at a temperature of 400°F.

The regulatory framework for the thermal destruction of PCBs is the Toxic Substances Control Act (TSCA). Incineration under TSCA requires that PCBs be maintained at a temperature of 2,200°F for at least two seconds. Additional incineration requirements include a minimum combustion efficiency of the burner(s), PCB destruction efficiency, acid gas scrubbing, and control and recording requirements. Residual materials from thermal treatment processes must contain no more than 2 ppm PCBs.

Remedial thermal systems can be permanent, mobile or portable. Mobile systems are brought to a site and then removed at the conclusion of the clean-up. They normally include all of the equipment and subsystems necessary for operation of the facility, such as electric power generation equipment, a fuel supply and equipment to collect and dispose of wastewater. Transportable equipment differs from mobile equipment in that it requires a significant installation effort. This equipment is provided in modular components and must be assembled before use. A process water supply is needed on-site. Wastewater discharge would also be disposed of on-site, although water or wastewater discharge treatment facilities may be required. Transportable systems are designed so that they can be dismantled, removed and re-installed at another site.

The anticipated load for this project, as well as its geographical distribution, would probably necessitate installation of one or more on-site thermal systems.

Low Temperature Desorption

Low temperature desorption is the application of low level heat (400°F to 800°F) to a material in a primary chamber. Organic materials are released in the primary chamber and are directed to a secondary chamber. In the secondary chamber, the organics are heated to 2,200°F for destruction and are then passed through an air emissions control system to remove acid gases and particulate matter.

The low temperatures at which the system operates allow use of conventional materials in the primary chamber and reduce the amount of supplemental fuel required. By varying the amount of air admitted to the primary chamber, the amount of volatiles released can be controlled. With less air (approaching a pyrolysis condition), the release of organics from soils will increase.

PCBs bind tightly to soils. A temperature of 400°F to 800°F is not sufficient to release the contaminant from soil, unless air flow is severely restricted. In a restricted oxygen atmosphere (pyrolysis), however, the PCBs will be encouraged to form other compounds, such as dioxins, which are considered more toxic than PCBs. Consequently, thermal desorption is not a recommended thermal treatment technology for PCB-contaminated sediments.

Rotary Kiln Incineration

The rotary kiln incinerator is a horizontal cylinder, lined with refractory material, which turns about its horizontal axis. Waste, no greater than two inches mean particle size, is deposited in the kiln at one end and is reduced to an ash by the time it reaches the opposite end of the kiln. Kiln rotational speed is in the range of three-quarters to two revolutions per minute.

A source of heat is required to bring the kiln to operating temperature and to maintain its temperature during incineration of the waste feed. Supplemental fuel is normally injected into the kiln through a conventional burner. The kiln will dry and burn solids and will volatilize organic material, including PCBs. All organics will generally not be incinerated in the kiln and a high temperature must be maintained in an afterburner at a specific residence time for destruction. To meet regulatory requirements, the afterburner is designed for a residence time of two seconds at a temperature of 2,200°F.

A rotary kiln system used for the incineration of toxic waste would include the kiln, provisions for feeding, supplemental fuel injection, an afterburner and an ash collection system. The gas discharge from the afterburner is directed to

an air emissions control system. An induced draft fan is provided within the emissions control system to draw gases from the kiln through the equipment line and then to discharge them via a stack to the atmosphere. The fan (or another prime mover) is sized to maintain a negative pressure throughout the system so that gas leakage is always into, not out of, the kiln system.

Multiple Hearth Incineration

The multiple hearth furnace is a vertical cylindrical structure, lined with refractory material. It is designed for sludges or other wastes that require drying. Waste is dropped on one of a series of six to ten horizontal hearths. A center shaft rotates within the incinerator, wiping waste across one hearth. At the edge of the hearth waste drops to a lower hearth.

Waste loses most of its moisture on the top hearths, burns or loses volatiles at the center of the furnace, and burns to a sterile ash at the lower hearths of the incinerator. The multiple hearth furnace does not sustain temperatures in excess of approximately 1,500°F at its outlet. When it is used for PCB destruction, an afterburner is required to obtain the 2,200°F temperature specified in USEPA's rules.

Off-gas from the incinerator is directed toward an air emissions control system, which normally includes a quench section, a Venturi scrubber and a tray tower for control of particulate and acid gases. The multiple hearth system is a very large system, with many pieces of equipment and many movable components. Because of its size and its complexity, it is not adaptable for transportable use and is not recommended for the Hudson River site.

Fluid Bed Incineration

The fluid bed furnace is a cylindrical, refractory-lined shell with a supporting structure above its bottom surface to hold a sand bed (fluidized bed). The structure has a series of tuyeres, which allow the passage of air upward into

the bed while tending to prevent the passage of sand. Air is introduced into a windbox and then through the tuyeres into the sand bed. A high degree of turbulence is created in the sand bed by the passage of the air stream, which creates a motion on the top of the bed that has the appearance of a fluid.

Fluidized bed systems are relatively large and operate at temperatures less than 1,600°F. They are not adaptable to transportable systems and they do not develop sufficiently high temperatures to destroy PCBs. Their use for the Hudson River site is not recommended.

Circulating Fluid Bed Incineration

In the circulating bed concept, a high air/gas velocity (from 15 to 20 feet/second) is introduced into a fluid bed. This high velocity, which is ten times the velocity in a conventional fluid bed furnace, elutriates both the bed and the combustible waste. Circulating material rises through a reaction zone to the top of a combustion chamber and passes through a hot cyclone. Hot gas passes through the cyclone, while the majority of solids drop to the bottom of the cyclone and are re-injected into the bed of the furnace. The hot flue gases pass to a gas cooler and then to a baghouse for removal of particulate.

Feed is sized to less than a one-half inch average particle size and is introduced into the leg between the cyclone and the bed of the reactor. Waste is fed to the system through a feeding bin. A metering screw conveys waste from the bin to the feed leg. The waste feed rate is automatically adjusted to maintain a pre-set oxygen concentration in the flue gas. Lime can be added to the waste feed through a lime metering system to neutralize acid-generating constituents of the waste, such as PCBs.

The design operating temperature for circulating systems is normally 1,600°F, although the system can withstand temperatures up to 2,000°F on a continuous basis. A combustion air fan provides air to the bed for fluidization and oxidation. Furnace draft is maintained by an induced draft fan downstream

of the cyclone. Flue gas exiting the cyclone passes through a conventional exhaust gas treatment system, which removes particulate and other undesirable constituents from the gas stream.

Retention time of material within the system is controlled by monitoring the discharge from the ash cooler. The cyclone bottom ash discharges to the reactor, but this ash flow can also be removed from the system through a water-cooled ash conveyor. By increasing the speed of this conveyor, additional material is removed from the furnace system and the residence time within the system is likewise controlled. For instance, by lowering the conveyor discharge rate, less material would be discharged from the system and the solids retention time would be increased.

A major feature of the circulating fluid bed system is its ability to control the residence time of wastes to over 50 seconds. Generally, destruction of organics, including PCBs, has been found to occur at temperatures below, 1,600°F in this system. This lower temperature translates to lower supplemental fuel requirements, less refractory maintenance, less severe bed eutectics, *etc.*

Conveyor Furnace

The conveyor furnace is essentially a conveyor belt system passing through a long chamber lined with refractory material. An induced draft fan maintains a negative pressure throughout the system. Combustion air is introduced at the discharge end of the belt. Air will pick up heat from the hot burning waste as waste and air travel countercurrent to each other. Supplemental heat is provided by electric infrared heating elements or by conventional fossil fuel burners within the furnace above the belt. Cooling air is injected into the incinerator chamber to prevent local hot spots in the immediate vicinity of the heaters/burners and is used as secondary combustion air within the furnace. The furnace is designed to provide and maintain a temperature of 1,600°F above the travelling conveyor.

The conveyor belt is woven wire mesh made of high temperature alloy steel that will withstand the 1,300°F to 1,600°F temperatures encountered within the furnace. The refractory material used is ceramic felt rather than brick. The furnace does not have a high capacity for holding heat and can be started from a cold condition relatively quickly, in one to two hours. Soil or other wastes are fed by gravity on to the belt and are immediately leveled to a depth of two to three inches. The waste must be sized to no more than a two-inch effective diameter. The belt speed and travel time are chosen to provide burnout of the waste with minimal agitation. This feature results in a relatively low level of particulate emissions.

This furnace system has been adapted for the treatment of soils contaminated with trace organics. The soils are heated in the basic unit to release their organic contamination. The organics are directed, through the exiting gas stream, to an external afterburner where they are fired at a sufficient temperature and a specific residence time for destruction. Depending on the furnace manufacturer, electric power, natural gas, propane or fuel oil is used as supplemental fuel for the system.

Electric Pyrolyzer

Waste-bearing materials must be processed to a maximum two-inch average particle size prior to introduction into this system. Soil can have a moisture content of up to 25 percent before drying is necessary. Solid waste is dropped by gravity through the reactor, or pyrolyzer, while liquid feed is injected into the system.

The electric pyrolyzer process promotes the release of organics from the surface of soil or other material. As waste is dropped into the pyrolyzer, it passes through a high temperature zone where the majority of the organics volatilize. The soil, or other solid waste, drops to the bottom of the pyrolyzer, which is maintained at a high enough temperature to keep the soil and other inorganics in a molten state.

Supplemental electrodes within the melt assure that its temperature will be maintained at a relatively high level and will be uniform throughout the melt. Any metals present will be found in their elemental form or as a salt, they will be removed from the melt on a continuous basis from an appropriately placed tap. Other taps are located at other levels of the reactor wall to provide a means for discharge of slag and other materials generated by the process. The tapped materials fall into a water bath where they cool immediately. The cooled residual has the appearance of dark glass. Any organics that did not volatilize as the soil dropped through the reactor will be destroyed within the melt.

Even though the supplemental electrodes generate a temperature of approximately 4,000°F, the overall melt is maintained at a much lower temperature. Chemistry of the melt can be controlled by additives such as lime, salts or other compounds. By adjusting melt composition, neutralization of acid gas components of the waste occurs and properties of the by-product slag can be controlled.

Off-gas from the reactor passes through a cyclone where the majority of particulate that may be elutriated into the gas stream is removed. A baghouse removes the balance of particulate matter. A wet scrubber placed downstream of the baghouse will remove any halogenated (acid) gases that were not neutralized within the reactor.

Contract Thermal Destruction

There are several commercial incineration facilities that dispose of PCB-contaminated waste on a contract basis. All of these facilities utilize rotary kiln systems for the disposal of wastes. One facility also has a rotary reactor, which has been developed for the treatment of contaminated soils.

The rotary reactor is a hollow, three compartment horizontal cylinder that rotates at 10 to 30 revolutions per minute. It acts as a horizontal fluid bed reactor utilizing various inert medium for the bed, *e.g.*, sand. Solid and

semi-solid wastes, in addition to the inert medium, are mechanically lifted on internal radial fins and cascade through the combustion gases in the combustion zone of the reactor. This cascading action provides mass and heat transfer. The recycling feature of this system provides relatively high residence time for the solid material and allows operation at lower temperatures than with more conventional incineration equipment. The normal operating temperature for this reactor is approximately 1,600°F. Lime can be added to the unit to neutralize acid gases generated from the burning of halogenated organics, such as PCBs.

Status of Thermal Destruction Technologies

Several of the above described systems have been identified as not recommended for further analysis. Of those remaining, several have not been developed further than pilot-scale. The screening of thermal destruction technologies follows.

- Low temperature desorption. Additional analyses not recommended.
- Incineration
  - Rotary kiln. Recommended for consideration. Has been applied to the destruction of PCBs in sediments, most recently the clean-up of Waukegan Harbor outside of Chicago.
  - Multiple hearth. Additional analyses not recommended.
  - Fluid bed incinerator. Additional analyses not recommended.
  - Circulating fluid bed incinerator. Recommended for consideration. Full-scale systems are operating in California and Alaska on contaminated soils.
  - Conveyor furnace. Recommended for consideration. Full-scale systems have been used for the clean-up of contaminated soils. Five conveyor systems utilizing electric power are on the market, and a number of systems utilizing fossil fuel are also operating.
- Electric Pyrolyzer. Recommended for consideration. This equipment has been developed at pilot-scale and mounted on two trailers. It is available for demonstrations.

- Contract disposal. Not recommended for further consideration.

C.4.4.3 Biological Treatment Technologies

Bioremediation

Bioremediation is a technique in which the physical, chemical and biological conditions of a contaminated medium are manipulated to accelerate the natural biodegradation and mineralization processes. (Biodegradation is a process whereby microorganisms alter the structure of a chemical. Mineralization is the complete biodegradation of a chemical to carbon dioxide, water and simple inorganic compounds.) In nature, both partial biodegradation and complete mineralization take place; the processes, however, are frequently slow.

Bioremediation has been used in the treatment of sewage for a number of years. It has been used fairly successfully under some conditions to treat petroleum products, creosote and pesticide contamination. PCBs, however, pose greater challenges to bioremediation than many other types of contamination. Research is necessary before effective full-scale, biological treatment is available for these compounds.

Paramount to successful PCB bioremediation is the identification of a microbial population capable of degrading a large number of different PCB congeners. Abramowicz (1990) has recently reviewed the PCB degrading capability of aerobic and anaerobic strains. As summarized in that review, more than 25 aerobic strains demonstrating varying degrees of PCB-degrading competence and specificity have been isolated. Some strains, such as *Psuedomonas* sp. LB400 and *Alcaligenes eutrophus* H850, have shown the ability to degrade a large number of congeners, including several penta-, hexa- and heptachlorinated congeners. Other species have shown PCB-degrading capabilities complementary to LB400 and H850, suggesting that treatment by two or more strains could yield even more degradation than treatment with one of these strains alone. Aerobic biodegrada-

tion, however, is generally limited to the less chlorinated PCB congeners. To date, no aerobic strain has shown the ability to degrade Aroclor 1260.

Those organisms that have exhibited PCB-degrading capabilities have not shown the ability to mineralize PCBs. Rather, the microbes degrade PCBs to chlorobenzoates (in some cases chloroacetophenones) and five carbon aliphatic compounds. Bioremediation of highly chlorinated congeners and the products of aerobic PCB biodegradation will, therefore, require a consortium of microbes that can degrade these compounds as well. DETOX Industries indicate that they have identified a consortium of microbes that will mineralize PCBs, including the more highly chlorinated congeners of Aroclor 1260 (Philip Bails, DETOX Industries, personal communication). As the organisms are proposed for commercial bioremediation, the details regarding this consortium are proprietary and the information can not be verified.

Anaerobic organisms have shown the ability to dechlorinate reductively heavily chlorinated PCB congeners. Dechlorination does not change the total molar concentration of PCBs, since the products are less chlorinated biphenyls. Dechlorination, however, does yield products that can be degraded by aerobes. Thus, sequential anaerobic/aerobic treatment may enable treatment of more chlorinated PCB mixtures.

Theoretically, an alternative to the sequential treatment of PCBs with several strains is the genetic engineering of a wide range of biodegradative capabilities into a single or reduced number of coexisting organisms. Toward this end, Mondello (1989) has cloned genes encoding PCB degradative enzymes from several PCB-degrading strains. Enzyme activity in recombinant *E. coli* containing these genes from *Psuedomonas sp.* LB400 was nearly as great as in the donor strain. More research is still required to engineer recombinants with the ability to degrade a wide variety of congeners.

In addition to the identification of PCB-degrading microbes, successful bioremediation will require identification of the environmental factors

controlling biodegradation. Research sponsored by General Electric is ongoing to define the environmental conditions most conducive to PCB biodegradation. The results of their research to date indicate that optimum aerobic microbial activity requires: 1) microbial growth on biphenyl or chlorobiphenyl (Bedard, 1990; Mondello, 1989; McDermott, 1989); 2) temperatures elevated above those that would be characteristic of Hudson River sediments (McDermott *et al.*, 1989); 3) aeration (McDermott *et al.*, 1989); and 4) sufficient PCB bioavailability. The inherent insolubility of PCBs and high concentrations of natural organic matter concentrations can reduce bioavailability (Brooks, 1989; Harkness and Bergeron, 1990). Optimum anaerobic activity for Hudson River strains or consortia appears to require: 1) the absence of inhibitors, such as sulfate (Tiedje *et al.*, 1989); 2) elevated PCB concentrations, *i.e.* greater than 50 ppm (Tiedje *et al.*, 1987); 3) the presence of certain inorganic nutrients (Abramowicz *et al.*, 1989); 4) a supplemental carbon source (Tiedje *et al.*, 1989; Nies *et al.*, 1990; Alder *et al.*, 1990); and 5) temperatures elevated above those that would be characteristic of river sediments (Tiedje *et al.*, 1989). GE will be conducting site-specific tests this summer to identify the important environmental variables affecting bioremediation of Hudson River sediments.

Once an acceptable microbial consortium and proper environmental variables have been identified, one of three different engineering approaches to bioremediation can be taken: an *in situ* approach, a land-based approach or a bioreactor approach.

In Situ Approach

For *in situ* treatment of Hudson River sediments, the contaminated sediments would be left in place. This approach obviously limits the amount of control that can be exercised over environmental variables during bioremediation and can pose significant engineering difficulties in the uniform introduction and mixing of microbes or nutrients that may be required. Additionally, mixing would require a containment system to prevent suspension and transport of contaminated

sediments. Regular monitoring of sediment conditions and PCB concentration would be necessary to assess remedial progress.

McDermott *et al.* (1989) have conducted some small-scale field tests of *in situ* soil bioremediation at a former racing drag strip in Glens Falls, New York. The top 15 cm of soil at this site was contaminated with 50 to 500 ppm of Aroclor 1242. A test area was prepared by rototilling the top 20 cm of soil to homogenize vertically the PCB contamination. Two test plots were marked off and covered with a transparent tent to protect the site from the elements. One plot was dosed three times a week with LB400 (2×10^9 cells/ml); the control plot was dosed three times a week with buffer. One-half of each test plot was mixed by rototilling prior to each dosing. After 20 weeks, approximately 25 percent of the PCBs had been biodegraded in the top three cm of the LB400 dosed, unmixed soil. The amount of biodegradation in the lower 17 cm of this soil was not reported. Approximately ten percent of the PCBs had been biodegraded in the LB400 dosed, mixed soil. No biodegradation was observed in either the mixed or unmixed portions of the control plot. McDermott *et al.* (1989) suggest that these results might be improved, if soil and moisture conditions were less extreme. (Temperatures inside the tent sometimes exceeded 50°C, rapidly drying the soil and desiccating the bacteria.) Genetic engineering of a bacteria that could withstand environmental extremes and/or dosing with a bacterial culture showing PCB degradative capacities complementary to LB400 might also improve results (McDermott *et al*, 1989).

No full-scale, *in situ* bioremediation of PCB-contaminated sediments has been conducted to date. At Sheboygan River and Harbor Superfund Site, however, a type of limited *in situ* bioremediation not involving the introduction or mixing of microbes and nutrients is planned for some of the PCB-contaminated sediments. These sediments will be capped, with the expectation that this capping will prevent transport of the sediments and increase anaerobic conditions, thereby enhancing anaerobic dechlorination already believed to be in progress. Capping will take place only in areas that will not be disturbed by dredging. PCB-contaminated sediments that are not capped will be removed. While relatively

easily accomplished from an engineering standpoint, this bioremedial solution may provide little reduction in total molar concentration of PCBs. Anaerobic processes are likely limited to reductive dechlorination without cleavage of the biphenyl ring and certain congeners may prove resistant to dechlorination by the indigenous microbes.

Land-Based Approach

Two types of land-based biological treatment approaches can be used for bioremediation of sediments: composting and land farming. Implementation of either treatment for the Hudson River site would require dredging and may require dewatering of the sediments. Since anaerobic conditions would be difficult to maintain in both composting and landfarming, these systems would be appropriate only if bioremediation can be carried out aerobically. Additionally, some other type of treatment system would be necessary for the water resulting from dewatering.

In composting, sediments would be placed in large piles. A typical compost pile may be six to eight feet in height and contain from 4,000 to 10,000 cubic yards of sediment. Sediments would be placed on top of a prepared clay or plastic liner with a leachate collection system in compliance with RCRA minimum technology requirements. Oxygen would be supplied to the material through a piping system. Installation of a watering system to maintain appropriate moisture levels and deliver nutrients and possibly microbes to the piles would likely also be necessary. In cool weather, steam or heated water may be delivered to the piles to maintain elevated temperatures as well as to supply moisture and nutrients. Collected leachate can be recirculated. The final component of each pile would be a cover to minimize particulate emissions and surface run-off. Depending upon the permeability of the sediments, they may require mixing with a bulking agent prior to piling to develop a permeability that will allow penetration of nutrients, water, air and/or microbes. Regular monitoring of both pile conditions and changes in PCB concentration would be necessary to assess remedial progress.

In land farming, sediments would be spread over liners and leachate collection systems in 9 to 18-inch layers or lifts, for treatment. Although land farms require more space for treatment than compost piles, a number of lifts can be land-farmed sequentially over the same surface area. A sprinkler system would be used to deliver microbes, nutrients and moisture to the land farm. Oxygen can be supplied through periodic tilling. As land farms are generally not covered, there is the potential for loss to the air of more volatile congeners, especially during initial spreading and tilling.

Compost piles and land farms have been used at a number of sites, with varying degrees of success to treat soil contaminated with solvents, petroleum products or creosote. These contaminants, however, are generally more readily biodegraded than PCBs. Furthermore, some of the treatment success attributed to these approaches is probably a result of compound loss through volatilization or air stripping rather than actual compound destruction. No full-scale PCB bioremediation projects using these techniques have been completed to date; few vendors have experience treating any compounds in soil or sediment volumes greater than 10,000 cubic yards.

Bioreactor Approach

In a bioreactor approach to bioremediation of the Hudson River sediments, sediments would be dredged and then biologically treated in a container or reactor. Because sediments are completely contained, a bioreactor approach offers greater control over environmental variables than either an *in situ* or land-based system. Furthermore, this approach is the only one of the three that is conducive to the sequential anaerobic/aerobic treatment that may be required to treat more heavily chlorinated PCBs. Remediation in a bioreactor, however, may provide little advantage over alternative remedial techniques in terms of sediment handling requirements and destruction efficiency.

Several commercial vendors claim experience with bioreactor technology, but few vendors with demonstrated pilot-scale (or larger scale) experience in

treating PCBs in bioreactors were identified. DETOX Industries has completed a pilot demonstration project in which 500 pounds of PCB-contaminated mixed-wastes, including sludge, soil, electrical capacitor oil and water, were placed in an open-air reactor. Initial concentrations of PCBs (some Aroclor 1260 as well as lesser chlorinated mixtures) were as high as 2,000 ppm in sludge and 44,000 ppm in the liquid phase. Concentrations were aerobically reduced to 4 ppm overall within 16 months (Joe Daily, DETOX Industries, personal communication). ENSR has completed pilot-scale tests at the French Limited Superfund Site. Heavy sludge contaminated with 1,600 ppm levels of lesser chlorinated PCBs (Aroclors 1232, 1242 and 1248) were aerobically treated in 2,000 gallon tanks. At the time the tests were terminated, total PCB concentration had been reduced to 66 ppm. Additional reduction in concentration may have been achievable, if the tests had been carried out for longer periods of time (Richard Woodward, ENSR, personal communication). The effect of volatilization on the performance of open air bioreactors is also a consideration for its use.

Using the results of treatability tests carried out at the University of Michigan, Blasland and Bouck plan to conduct pilot-scale tests on PCB-contaminated sediments from Sheboygan River and Harbor. It is expected that treatment will include anaerobic/aerobic cycles. These tests are scheduled to take place within one year.

Additional evaluation of the applicability of bioremediation for treatment of Hudson River sediments should include:

1) Laboratory demonstration that a consortium of microbes can degrade all of the PCB congeners present in Hudson River sediments as well as potentially undesirable biodegradation products;

2) Identification of optimum environmental conditions for consortium activity;

3) Estimation, based on laboratory experiments, of maximum PCB biodegradation rates achievable and the time required to biologically treat Hudson River sediments;

4) Demonstration that biodegradation results comparable to those obtained in the laboratory, can be obtained on a larger scale in the field, *i.e.*, need for pilot-scale tests;

5) Comparative analysis of in-river bioremediation results and land-based system results; and

6) The effect of PCB concentrations on bioremedial techniques.

C.4.5 Disposal Technologies

Disposal technologies include upland disposal in a lined landfill, off-site disposal in a permitted disposal facility and confined aquatic disposal. There are no off-site, permitted facilities in the project area.

The use of subaqueous depressions or borrow pits can provide confinement of contaminated material in an open-water disposal scenario. The US Army Corps of Engineers has conducted research on subaqueous confinement and has utilized the technology in the New York Harbor area. For the Upper Hudson, this technology will not be considered further, since this part of the river has a relatively steep gradient.

Disposal in a secure, lined landfill provides for long-term storage of contaminated material and necessitates siting, design, construction, operation, closure, post-closure monitoring and maintenance. The facility must meet regulatory requirements, including groundwater standards, for landfills in which PCB-contaminated materials will be stored. The landfill may be capped with an impermeable cover after each dredging season. A roughing and storage pond, surge pond, water treatment plant, pump station, leachate collection system, stormwater drainage system and chemical feed system are components of the design. Construction site electrical services, fencing and seeding are required. A secure landfill provides a high degree of isolation of the contaminated material with a low probability of subsequent discharge. The technology for landfill design is routine and is readily available.

In 1982, a location in the Town of Fort Edward, known as Site 10, was chosen for the continued disposal of contaminated river sediments. In 1983, approvals for Site 10 were revoked by the New York State Supreme Court on grounds, among others, of violating local zoning regulations. The decision was upheld by New York's Court of Appeals in 1985. Several alternate disposal sites were subsequently analyzed. A site known as Site G, also located in the Town of Fort Edward, was selected as a preferred alternative for a reduced-scale disposal effort. Subsequently, a 1987 amendment to the New York State Siting Law allowed the continued study of disposal Site 10. Site 10 has now been chosen by the NYSDEC Project Sponsor Group as the preferred disposal site for approximately three million cubic yards of river sediments, remnant deposits and PCBs previously removed from the Hudson (NYSDEC, 1989).

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C.5 Innovative Treatment Technologies (USEPA SITE Program)

Innovative technologies that have at least preliminarily demonstrated a capacity to separate PCBs from solids, sludge or sediments include:

• AOSTRA TACIUK Process System	SoilTech Inc. Englewood, CO 80112
• Solidification/Stabilization Processes	USEPA Risk Reduction Engineering Lab (RREL) Cincinnati, OH

The SoilTech system has previously been named the Alberta Oil Sands Technology and Research Authority (AOSTRA) - Taciuck process. SoilTech, Inc. holds the exclusive US license to apply this technology and is owned by Canonie Environmental Services Corp. and LIMATAC Industrial Processes. The SoilTech concept is a continuous pyrolysis system using a rotary drum heater with an inner core and two subatmospheric processing chambers. The first chamber operates at variable temperatures to about 600 °F and volatilizes water and light hydrocarbons from injected wastes. The water and organics are collected and condensed. The second chamber operates at temperatures of 1,000 to 1,100°F. In this chamber, heavier hydrocarbons are evaporated and partially pyrolyzed to yield lighter gas fractions and some coke. The vapor from this chamber is also collected and condensed. For feed soils or sludges containing PCBs, the PCBs are recovered as condensate in an oil phase.

Solids residues pass out of the second or light temperature chamber into the outer shell of the drum. Gas or oil-fired burners at the end of the outer shell provide primary process heat with the injection of combustion air. The solids are heated in this zone to about 1,300 to 1,400°F and some hot solids are recycled back into the second chamber to create effective heat transfer. Clean solids are output from the unit. Other components of the major process equipment are a flue gas train for removing particulate matter and acid gases, a preheat vapor train and a retort vapor train, which condenses high-boiling vapors and separates vapors from liquids.

The process has been developed at production levels and has been applied to treating PCB-contaminated soil at Wide Beach, New York and PCB-contaminated sediment at Waukegan Harbor, Illinois. SoilTech's pilot demonstration unit has a nominal capacity of five tons per hour; a commercial transportable unit has a capacity of ten tons per hour. The latter is currently being used in treating 21,000 tons of PCB-contaminated soils at the Wide Beach, NY Superfund site. The SoilTech system is retained here for further analyses and bench-scale testing.

A group of chemical fixation technologies that immobilize contaminants within the waste have emerged through the USEPA SITE Program. These technologies involve mixing waste material with settling agents to enhance the physical properties of the waste. Numerous commercial settling agents have been tested. These agents either eliminate free water from the waste or alter the chemical form of the contaminants to make them resistant to leaching.

A bench-scale study of solidification/stabilization as a treatment technology for New Bedford Harbor sediments was conducted by the US Army Corp of Engineers (1989). Composite sediment samples were processed with various dosages of settling agent formulations, including Portland cement, Portland cement with Firmax proprietary additive and a Silicate Technology Corporation (STC) proprietary additive. Batch leaching tests showed that the leachability of PCBs was reduced by factors of 10 to 100. Costs for treating New Bedford Harbor sediments using the tested agents have been estimated at $100 per ton. While the solidification/stabilization approaches offer potential low cost treatment options, data on the long-term aging effects of the stabilized/solidified matrix should be developed further.

Recently, USEPA's Risk Reduction Engineering Laboratory (RREL) in Cincinnati initiated a project with RMC Environmental of West Plains, Missouri to conduct controlled experiments on PCB-contaminated soils. The experiments were conducted to investigate declining concentrations of PCB over time, which were observed at contaminated sites that were stabilized through the addition of lime and other alkaline materials. The study has been recently published and will be reviewed.

C.6 Initial Screening of Technologies

While no particular technology has been removed from further consideration for subsequent phases, it is possible at this initial screening stage to render some judgments concerning remedial options, based upon their applicability and current level of development. Data upon which the initial screening was based were obtained from numerous sources, including reports for other Superfund sites, USEPA's technology assessment documents and direct communications with equipment manufacturers. Results of this initial screening effort are illustrated as Figure C.6-1.

Several technologies associated with particular response actions were not screened in this preliminary reassessment. These include methods to excavate remnant deposits, if necessary. In addition, technologies applicable to treating water resulting from sediment dewatering operations have not yet been evaluated. A wide range of well-proven, commercially viable technologies are available to treat effluent from dewatering operations. These will be evaluated in subsequent phases.

Mechanical, hydraulic and specialty dredging systems or conventional excavation methods are available to remove contaminated sediments. While hydraulic systems have been preferred at other Superfund sites and have been shown to minimize sediment resuspension during removal operations, these systems result in the need to handle significant quantities of by-product water and tend to be most cost-effective for dredging relatively large quantities of sediment. Thus, the three generic dredging systems have been retained for further assessment, when additional information on materials characteristics and quantities will become available.

Should a decision be made to remediate the Hudson River site by removing some or all its contaminated bottom materials, it would be necessary either to landfill the removed materials or to treat those materials and landfill the treated residuals. Physical, chemical, biological and thermal processes or technologies are available to treat PCB-contaminated solids. Of the large range of treatment alternatives available, those considered to be either commercially

available or sufficiently well developed for further consideration are illustrated on Figure C.6-1. The decision to remove treatment technologies from further consideration this time was based on data contained in reports for other Superfund sites and on personal communications with system vendors.

As an alternative to removal, technologies applicable to remediating the Hudson River site without removing its contaminated sediments have also been considered. *In situ* technologies include those that contain the bottom sediments as well as those that treat them. Containment systems, such as capping and retaining structures, are methods for controlling sediment resuspension under the range of expected river hydraulic conditions. *In situ* treatment methods, primarily chemical and biological technologies, are also undergoing testing and have been retained for further assessment. As illustrated on Figure C.6-1 generic *in situ* remedial technologies will continue to be evaluated and will be considered applicable to either partial or complete remediation of the river's contaminated sediments.

A final category of remedial technologies considered here is that which involves disposal of either untreated contaminated soils or the residuals from treatment of contaminated soils. Off-site disposal at commercial landfills has been discounted at this time because of the lack of nearby permitted landfills and the relatively large quantities of materials that would have to be hauled a considerable distance to any commercial facilities. On-site or in-river disposal and upland disposal options are being retained for evaluation in the next phase. Upland disposal, which would involve obtaining approval to construct a proximate landfill specifically for the river's contaminated sediments, is the disposal technology being pursued by the NYSDEC Project Sponsor Group at this time.

C.7 Treatability Studies

The decision to remediate the Hudson's contaminated sediments will be significantly influenced by the feasibility and costs of remedial technologies. Treatability studies can be an effective technique to evaluate remedial technologies, because such studies reduce equipment performance uncertainties and lead to improved estimates of treatment system cost. Since treatability studies have not yet been performed on Hudson sediments, even a limited program would be expected to provide considerable, useful information.

Bench-scale treatability studies are particularly appropriate where emerging chemical/physical technologies are being evaluated. Bench-scale tests are usually performed in a laboratory and require relatively small quantities of the contaminated material to achieve their objective. These tests are conducted to determine the effectiveness of a particular processes chemistry and to test a wide range of process operating variables. Bench-scale work can also be used to set parameters for full-scale tests, should a particular technology warrant further consideration.

Four physical/chemical technologies, KOHPEG, B.E.S.T., LEEP and Propane Extraction, are being brought forward for further consideration on the basis of preliminary screening. Bench-scale work to establish basic operating parameters and costs for these would be appropriate in subsequent project phases. The developers of selected technologies will be contacted to establish time frames, costs, material quantities, and general goals for a treatability program. In general, between two and five months would be required for the bench-scale work (depending on the technology) and it would be necessary to obtain about ten pounds of bottom material for each test.

Bench-scale analyses related to thermal treatment systems are not recommended at this time. It will be necessary, at some point, to establish parameters such as the temperature at which PCBs are released from the river sediments, the sediment ash fusion temperature as well as the composition of the bottom material in terms of total chlorides, heavy metals, and total organics. Since thermal systems have been extensively applied to remediating contaminated

soil, there is less need to establish their general feasibility at this time. In addition, cost estimates for thermal systems are available from numerous other projects and can, on a preliminary basis, be adapted to the Hudson River.

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GLOSSARY

PHASE 1 REPORT

INTERIM CHARACTERIZATION AND EVALUATION

HUDSON RIVER PCB REASSESSMENT RI/FS

Adjusted Mean: Used here to refer to an estimate of the mean adjusted to account for bias in the sampling process.

Absorb: A chemical/physical bonding which is normally not reversible.

Adsorb: A chemical/physical surface adhesion which is normally reversible.

Adsorption Coefficient: The ratio of the amount of a chemical adsorbed per unit weight of organic carbon soil or sediment to the concentration of the chemical in solution at equilibrium. (see K_d K_{oc}.)

Allocthonous: Inputs of organic carbon from the surrounding terrestrial watershed and/or those carbon inputs generated outside the riverine system.

Anadromous: Fish that ascend rivers from the sea for spawning.

ARAR: Applicable or Relevant and Appropriate Requirements.

Arithmetic Mean (Average): For a set of data, the total sum of all sample values divided by the number of samples.

Aroclor: A trade name applied to mixtures of PCBs. When used to refer to a particular PCB mixture, it is usually followed by a numerical code which indicates the percentage of chlorine in that mixture.

Aryl Hydrocarbon Hydroxylase (AHH): A type of enzyme produced by the liver. Concentration or activity of AHH in the liver may increase in response to exposure to specific PCB congeners or other toxic compounds.

Autocthonous: Inputs of organic carbon that are generated within the riverine system.

Average Lifetime Daily Dose (ALDD): Exposure expressed as mass of a substance contacted per unit body weight per unit time (mg/kg-d), averaged out over a lifetime.

BAF: Bioaccumulation factor (BAF) is the ratio of the concentration of a chemical in an organism living in contaminated water and consuming contaminated food, to the concentration of the same chemical in the surrounding water. See

also bioconcentration factor (BCF).

BCF: Bioconcentration factor (BCF) is the uptake and tissue accumulation of a chemical from water. No other sources such as sediments and food are included.

Benthos: Animals associated with the aquatic substrata.

Bioaccumulation: The uptake and tissue accumulation of a chemical by biota from water and food sources.

Bioconcentration: The partitioning of a chemical between water and biota (often fish). Due to the chemical composition of water and biota, many organic chemicals have a greater tendency to partition into (*i.e.* concentrate in) biota rather than to remain in water.

Biodegradation: The microbially mediated transformation of a chemical.

BOD: Biological oxygen demand.

Cancer Potency Factor (CPF): An index of the cancer-causing ability of a compound, based on the slope of the dose-response curve. The potency factor is used to estimate an upper-bound probability of an individual developing cancer as a result of a lifetime of exposure to a particular level of a potential carcinogen.

Capillary Column Gas Chromatograph: An instrument used to separate, detect, and quantify chemicals. Separation of the chemicals employs a relatively small diameter column. This instrument can separate many chemicals a packed column chromatograph can not separate. See packed column chromatograph.

Carcinogen: A chemical or physical agent (such as radiation) capable of inducing cancer.

Carcinogenic: Capable of producing or inciting cancer.

Catadromous: Fish that live in fresh water and go to the sea to spawn, *e.g.*, eels.

CERCLA: Comprehensive Environmental, Response and Compensation Liability Act.

cfs: cubic feet per second; english unit measurement of river discharge.

Chlophen: PCB mixture manufactured in Germany.

cm/hr: centimeters per hour.

Confidence Interval: An interval defining the probable range of a random variable with the given confidence. For example, a 95 percent confidence interval about the sample mean is the range within which the true mean can be

expected to fall 95 percent of the time in repeated trials.

Confidence Limit: The value defining the end of a confidence interval.

Congener: A generic term used to refer to any possible PCB molecule. (See also isomer and homologue)

Coplanar PCB: A subset of PCB congeners whose physical configuration is such that the biphenyl molecules lie in approximately the same plane in space, resulting in a relatively flat molecular structure.

Correlation Coefficient: Measure of the degree to which two variables are linearly related; the range is 0 (no correlation) to 1 (perfect correlation).

Demersal: Bottom dwelling.

Detection Limit: Technically, the lowest concentration of a chemical that can be observed using a particular analytical method. Frequently used to refer to the lowest concentration of a chemical that can be routinely quantified using a particular analytical method.

Dissolved Oxygen (percent saturation): For comparative purposes, dissolved oxygen concentration in mg/l is often expressed in terms of the percentage saturation of the sample (% saturation = 100 G/G') where G is the observed concentration of the gas and G' is its solubility in water of the appropriate *in situ* temperature and salinity. Values may be greater than 100% (supersaturated) or less than 100% (undersaturated). Supersaturated conditions are generally caused by algae blooms, whereas undersaturated conditions are due to respiration by plants and animals, sediment oxygen demand and oxidation of organic particulates within the water column.

Dose-Response Curve: The line or curve that depicts the relationship between the exposure dose of a compound and the toxic response of the exposed population; usually taken from a graph showing laboratory experimental results where the dose is presented on the X-axis, and the response presented on the y-axis.

Dry weight: The weight of a substance (*e.g.* fish tissue or sediment) after all water has been removed from it.

Eh: Measure of the state of oxidation or reduction of a system in millivolts (mv). It is basically a measure of the ability of the system to supply or use up electrons. The Eh scale ranges from +500 mv (extremely oxidizing) to -500 mv (extremely reducing).

Error Bars: Graphical designation of a confidence interval.

Euryhaline: Organisms that can tolerate a wide range of salinities.

Exposure Pathway: The course a chemical or physical agent takes from a source to an exposed individual. An exposure pathway describes a unique mechanism by which an individual or population is exposed to chemicals or physical agents at or from a site. Each exposure pathway includes a source or release from a source, an exposure point, and an exposure route. If the exposure occurs at a point different from the exact source, then a transport/exposure medium (*e.g.*, air) also is involved.

F-Test: A statistical test of significance of the ratio of variances of random variables.

FDA: Food and Drug Administration.

Flow-Duration Curve: A plot showing river discharges and their corresponding exceedance probabilities.

Gas Chromatography: An analytical technique used to separate, detect and quantify chemicals.

Geometric Mean (Average): For a set of data, the anti-log arithmetic mean of the natural logarithms of the data.

Hazard Quotient: The ratio of a single substance exposure dose over a specified time period (*e.g.*, chronic) to a reference dose for that substance derived from a similar exposure period.

Hemosiderosis: A pathological condition marked by the deposition of hemosiderin in the tissues as a result of the breakdown of red blood cells.

Henry's Law Constant: Provides a measure of the extent of chemical partitioning between air and water at equilibrium. The higher the Henry's Law constant, the more likely a chemical is to volatilize than to remain in the water.

Hepatic Carcinoma: A malignant new growth in the liver made up of epithelial cells.

Hepatic Neoplastic Nodules: A small rounded mass of tissue in the liver that is in the form of a swelling, knot, or protuberance, which is a new and abnormal growth, and in which the growth is progressive but is not malignant (*i.e.*, does not spread to other parts of the body).

Hepatocellular Carcinoma: A malignant new growth in the liver made up of epithelial cells of hepatic origin, and tending to infiltrate the surrounding tissues and to be malignant (*i.e.*, spreads from one area to another in the body).

Hepatocellular Neoplasms: Any new and abnormal growth in the liver; specifically a new growth of tissue in which the growth is uncontrolled and progressive, and may or may not be malignant (*i.e.*, may or may not spread to other parts of the body).

Histopathology: Study of cellular pathology.

Homologues: Molecules making up a series in which each successive group of the series is characterized by one more atom, or group of atoms, than the preceding group. For example, mono- through decachlorinated biphenyls make up a homologous series in which each successive homologue group contains compounds with one more chlorine atom than the preceding group. Members of this series are referred to as homologues.

Hyperplasia: An abnormal increase in the cells of a tissue.

Interstitial: Referring to the sediment pore space.

In Vitro: In an artificial environment outside of the living organism, as in a test tube.

Instantaneous Discharge: A measurement of river flow at a specific point (*e.g.*, hourly) in time.

IRIS: Integrated Risk Information System. IRIS is an on-line database created by the EPA and maintained on the National Library of Medicine's Toxicology Data Network system. The database contains EPA health risk and regulatory information on some 400 chemicals, with both carcinogenic and non-carcinogenic risk assessment data for oral and inhalation routes of exposure. These data include Reference Doses (RfD), indicators of non-carcinogenic risks, and Unit Risks, indicators of carcinogenic risks. The regulatory information relates to environmental statutes such as the Clean Air Act, Clean Water Act, and SUPERFUND legislation. IRIS is further supplemented with EPA Drinking Water Health Advisories, substance identification, chemical and physical properties, acute toxicity, and aquatic toxicity.

Isomers: Molecules containing the same number and types of atoms but differing in arrangement. For example, 2,3-dichlorobiphenyl and 2,4-dichlorobiphenyl are isomers of each other.

K_d: Provides a soil or sediment-specific measure of the extent of chemical partitioning between soil or sediment and water, unadjusted for dependence upon organic carbon. The higher the K_d, the more likely a chemical is to bind to soil or sediment than to remain in water. See adsorption coefficient and partition coefficient.

K_{oc}: Provides a measure of the extent of chemical partitioning between organic carbon and water at equilibrium. The higher the K_{oc}, the more likely a chemical is to bind to soil or sediment than to remain in water.

K_{ow}: Provides a measure of the extent of chemical partitioning between water and octanol at equilibrium. The greater the K_{ow}, the more likely a chemical is to partition to octanol than to remain in water. Octanol is used as a surrogate for lipids (fats) and K_{ow} can be used to predict bioconcentration in aquatic

organisms.

Lipid: One of a class of compounds that contain long chain aliphatic hydrocarbons and their deviatives such as acids (fatty acids), amino alcohols and aldehydes. The presence of the long aliphatic chain confers distinct solubility properties and has led to the traditional definition of lipids as substances which are insoluble in water but soluble in ether, chloroform and benzene.

Linear Regression: A method of fitting a linear relationship between variables, in which one variable (the dependent variable) is estimated from one or more other variables (independent variables). The best fit relationship is usually accomplished by the method of least squares.

Linearized Multistage (LMS) Model: A mathematical model used to extrapolate the risk of cancer from high dosed observed in animal studies to the lower doses usually experienced by humans from environmental exposures. A multistage extrapolation model is based on the theory that several distinct changes are necessary to transform a normal cell into a malignant cell, and that human cancer can arise from a single transformed cell. The presence of a linear term in the model insures near linearity for the confidence limit of the extrapolation.

Log Transformation: A data transformation made by taking the logarithms of the sample data, often used to stabilize, or minimize, variance in environmental data.

Log Pearson Type III Distribution: An extreme-value probability distribution proposed for flood frequency analysis by the U.S. Water Resources Council.

Lotic: Refers to any system of flowing water such as streams and rivers.

Lowest-Observed-Adverse-Effect-Level (LOAEL): In dose-response experiments, this is the lowest exposure level at which there are statistically or biologically significant increases in frequency or severity of adverse effects in an exposed population.

m^3/s: cubic meters per second; metric unit measurement of river discharge.

Macrophyte: Macroscopic forms of vegetation.

Mann-Kendall Test: A non-parametric trend test used here to determine statistically significant declines in PCB concentrations in various media; a useful test in cases of missing data and non-detects.

Mass Spectrometry: An analytical technique used to identify chemical structures and to determine chemical concentrations. It is often used in combination with gas chromatography.

Maximum Tolerated Dose (MTD): The highest dose that causes no more than a 10 percent weight decrement in the exposed animal population, and does not produce mortality, clinical signs of toxicity, or pathologic lesions (other than those related to cancer) that would be predicted to shorten the animals' natural lifespan.

Maximum Contaminant Level (MCL): MCLs are the maximum allowable concentrations of contaminants in drinking water as established by federal and state agencies responsible for regulating public water systems. These are legally enforceable standards. MCLs are set to protect the public from acute or chronic health effects or an "unacceptable" cancer risk. They must also take into account technological feasibility and economic impact.

Median: For a set of data, defined as the midpoint in the data set, *e.g.*, half of the sample values are lower and half higher than the median.

mg/cm^2: milligrams per square centimeter.

mg/kg-d: a dose rate of exposure to contaminants expressed in milligrams of contaminant per kilogram of body weight per day.

Multiple Regression: Linear regression with multiple dependent variables.

ng/g: nanograms per gram (parts per billion), representing one-trillionth (10^{-9}) of a gram of chemical per gram of media (*e.g.* sediment).

NIOSH: National Institute of Safety and Health.

No-Observed-Adverse-Effect-Level (NOAEL): In dose-response experiments, a chemical exposure level at which there are no statistically or biologically significant adverse effects between a population exposed to the chemical versus a control population; some effects may be produced at this level, but they are not considered to be adverse, nor precursors to specific adverse effects.

NOAA: National Oceanic and Atmospheric Association.

Non-detects: Samples in which a particular chemical can not be measured at a concentration exceeding the detection limit.

Non-Parametric Test: A statistical test which does not depend on assumptions regarding the nature of the probability distribution defining a given data set (*e.g.*, no assumption of normality, lognormality, *etc.* is made about the data).

NPL: National Priority List.

NYSDEC: New York State Department of Environmental Conservation.

NYSDOH: New York State Department of Health.

NYSDOT: New York State Department of Transportation.

OSWER: Office of Solid Waste and Emergency Response within the USEPA.

Packed Column Analysis: The detection and quantitation of chemicals using a packed column gas chromatograph.

Packed Column Gas Chromatograph: An instrument used to separate, detect, and quantify chemicals. Separation of the chemicals employs a relatively large diameter column which may be incapable of separating some very similar chemicals. Chemicals which can not be separated are difficult to accurately quantitate. See capillary column chromatograph.

Partition Coefficient: A constant describing the equilibrium distribution of a chemical between two materials. This constant is the ratio of the chemical concentration in one material (in environmental applications, usually a hydrophobic material) to the concentration of the same chemical in a second material (in environmental applications, usually water). K_{ow} and K_{oc} are two examples.

PCB Metabolite: Following absorption of PCBs, the body metabolizes them. Metabolism of PCBs is dependent upon on the number and position of chlorine atoms, with lesser chlorinated isomers metabolized more readily than more chlorinated isomers. PCB metabolites are the products of this metabolism, and tend to be more water soluble (and thus more easily excreted) than the parent compound.

PCB: A polychlorinated biphenyl (PCB) is a member of the chemical class of chlorinated, aromatic hydrocarbons. A PCB consists of two connected rings of six carbon atoms each (this ring structure is called a biphenyl), to which one or more chlorine atoms are attached at any of 10 available sites.

PCDF: Polychlorinated dibenzofuran, a chemical related to TCDD.

Percentile: Value at which a given percent of the probability mass of a variable is exceeded. Commonly reported percentiles are the 50^{th} percentile, or median, and the 95^{th} percentile, which is a value exceeded by 5 percent of the sample data.

Periphyton: Microfloral growth upon the aquatic sediments.

pH: Measure of the acidity or alkalinity of a substance based on a 0-14 scale and is a function of the free hydrogen ions or protons. A pH of 7.0 is considered neutral and values below pH 7.0 are more acidic and values above pH 7.0 are more basic.

Phytoplankton: Free-floating microscopic algae.

Picocurie: One-trillionth of a Curie.

Piscivorous: Fish-eating.

Planktivorous: Plankton-eating.

ppb: Parts per billion, equivalent to µg/l in water at low solute concentrations.

ppm: Parts per million, equivalent to mg/l in water at low solute concentrations.

Primary Contact Recreation: Recreational activities where the human body may come in direct contact with raw water to the point of complete body submergence, such as swimming, diving, *etc.*

RCRA: Resource Conservation and Recovery Act. RCRA (1976) was passed by Congress in response to the potential problems posed by disposal of wastes generated by chemical and other industrial processes. RCRA promotes continuous management from point of generation to final disposal of hazardous wastes. The program works through requirements for hazardous waste generators, transporters, and treatment, storage, and disposal facilities. In 1984, RCRA was amended by the Hazardous and Solid Waste Amendments that required EPA to focus on permitting land disposal facilities and eventually phasing out land disposal of some wastes.

Reference Dose (RfD): A benchmark for the daily dose to which humans, including sensitive populations (such as children or pregnant women), may be subjected without an appreciable risk of adverse non-carcinogenic health effects during a lifetime of exposure.

Residual Variance: Portion of total variation in a data set due to chance or error.

RI/FS: Remedial Investigation/Feasibility Study. The RI/FS is the framework for determining appropriate remedial actions at Superfund sites. Remedial investigations are conducted to characterize the contamination at the site and to obtain information needed to identify, evaluate, and select cleanup alternatives. The feasibility study includes an analysis of remedial action alternatives based on National Contingency Plan evaluation criteria.

RIBS: Rotating Intensive Basin Studies, NYSDEC monitoring program for the assessment of ambient water quality.

ROD: Record of Decision, typically for a Superfund site.

Salt Front: The furthest upstream point at which the influence of seawater can be measured in the freshwater, typically about 500 ppm of salinity.

SARA: Superfund Amendments and Reauthorization Act of 1986.

Secondary Contact Recreation: Recreational activities where contact with water is minimal and where ingestion of water is not probable, such as fishing, boating, *etc.*

Shear Stress (τ): Force per unit area exerted by a moving fluid on its substrate in the plane of contact, used in this report in reference to the stress imparted by water flowing over river bottom sediments.

SITE: Superfund Innovative Technology Evaluation.

Skewed Distribution: A probability distribution which is not symmetric about the mean, *e.g.* the lognormal distribution.

Sorb: The process of either adsorbing and/or absorbing a chemical species onto a solid medium. (See definitions of absorb and adsorb.)

SPDES: State Pollutant Discharge Elimination System, a permit program adopted pursuant to Article 17, Title 8 of the Environmental Conservation Law and Section 402 of the Clean Water Act that imposes discharge limitations on point sources.

Spearman Rank Correlation: A non-parametric measure of correlation between two variables, in which rank orders rather than actual values are compared.

Stage-Discharge Curve: Plot showing the relationship between river height (stage) and flow (discharge).

Suggested No Adverse Response Level: Values suggested by the National Academy of Sciences that represent exposure concentrations of contaminants in drinking water or air that are expected to be safe for human exposure. Generally, an acceptable time period of exposure (*e.g.*, 24-hour, 7-day or long-term exposure) is also indicated, and different exposure concentrations may be suggested for each period of exposure.

Taxa: A major taxonomic category or subdivision such as class, order or family.

TCDD: Tetrachlorodibenzodioxin, of which the 2,3,7,8-TCDD isomer is considered to be the most toxic.

Time Weighted Average - Threshold Limit Values (TWA-TLV): The average concentration for a normal 8-hour work-day and a 40-hour workweek, to which nearly all workers may be repeatedly exposed, day after day, without adverse effect.

Toxic Equivalency Factor (TEF): An approach used to characterize the toxicity of a complex mixture of toxic compounds when adequate toxicity information is available only for some of the components of the mixture. In general, the toxicity of one component of the mixture is known. This compound's potency is set to 1.0, and the toxicity of other components is estimated by comparison against this base compound. Ideally, use of the TEF approach accounts for the

differing toxicological properties of the various components of the mixture, and yields a more realistic and scientifically supported estimate of the potential risks from exposure to the mixture.

Trophic: Refers to food level, *e.g.* primary producer, consumer, etc.

µg/l: micrograms per liter, equivalent to ppb for water at low solute concentrations.

µg/g: micrograms per gram (parts per million), representing one-millionth (10^{-6}) of a gram of chemical per gram of media (*e.g.* sediment).

µg/m³: micrograms per cubic meter, equivalent to ppt (parts per trillion) for water at low solute concentrations.

USEPA: United States Environmental Protection Agency.

USFWS: United States Fish and Wildlife Service.

USGS: United States Geological Survey.

Water Year: The USGS Water Year is defined as running from October 1 of the previous calendar year to September 30 of the current calendar year.