RESPONSIVENESS SUMMARY
HUDSON RIVER PCBs SITE RECORD OF DECISION

JANUARY 2002

For

U.S. Environmental Protection Agency
Region 2
and
U.S. Army Corps of Engineers
Kansas City District

BOOK 2 OF 3

WHITE PAPERS

TAMS Consultants, Inc.

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LIST OF ACRONYMS AND ABBREVIATIONS

ACGIH	American Conference of Governmental Industrial Hygienists
ACHP	Advisory Council on Historic Preservation
AGC	Annual Guideline Concentration
AOC	Administrative Order on Consent
ANOVA	Analysis of Variance
APEG	Alkaline (Alkali Metal Hydroxide) Polyethylene Glycol
ARAR	Applicable or Relevant and Appropriate Requirement
ARCC	Adirondack Regional Chambers of Commerce
ARCS	Assessment and Remediation of Contaminated Sediments Program
ATSDR	Agency for Toxic Substance and Disease Registry
AWQC	Ambient Water Quality Criterion
BAT	Best Achievable Technology
BBL	Blasland, Bouck, and Lee
BCD	Base-Catalyzed Decomposition
BMR	Baseline Modeling Report
CADD	Computer-Aided Drafting and Design
CDF	Confined Disposal Facility
CDI	Chronic Daily Intake
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFR	Code of Federal Regulations
cfs	cubic feet per second
CIP	Community Interaction Program
CLU-IN	Hazardous Waste Clean-up Information (EPA web site)
COC	Chemical(s) of Concern
COPC	Chemical(s) of Potential Concern
CSF	Cancer Slope Factor
CSM	Conceptual Site Model
CT	Central Tendency
CWA	Clean Water Act
CZM	Coastal Zone Management
DEIR	Data Evaluation and Interpretation Report
DMR	Discharge Monitoring Report
DNAPL	Dense Non-Aqueous Phase Liquid
DOC	Dissolved Organic Carbon
DOSM	Depth of Scour Model
DOT	Department of Transportation
DRE	Destruction and Removal Efficiency
ECD	Electron Capture Detector
ECL	Environmental Conservation Law (New York)
EE/CA	Engineering Evaluation/Cost Analysis
EEC	Extreme Effect Concentration
EIS	Environmental Impact Statement
EO	Executive Order
EPA	Environmental Protection Agency
EPC	Exposure Point Concentration
ERA	Ecological Risk Assessment
ESA	Endangered Species Act
ETWG	Engineering/Technology Work Group
FAIR	Farmers Against Irresponsible Remediation

FDA	Food and Drug Administration
FEMA	Federal Emergency Management Agency
FR	Federal Register
FRTR	Federal Remediation Technologies Roundtable
FS	Feasibility Study
FSSOW	Feasibility Study Scope of Work
FWIA	Fish & Wildlife Impact Analysis
g/m^2	Grams per meter squared
GAC	Granular Activated Carbon
GC	Gas Chromatography
GCL	Geosynthetic Clay Liner
GE	General Electric Company
GIS	Geographic Information System
GLNPO	(EPA's) Great Lakes National Program Office
GRA	General Response Action
HDPE	High Density Polyethylene
HHRA	Human Health Risk Assessment
HHRASOW	Human Health Risk Assessment Scope of Work
HI	Hazard Index
HMTA	Hazardous Materials Transportation Act
hp	Horsepower
HQ	Hazard Quotient
HROC	Hudson River PCB Oversight Committee
HSI	Habitat Suitability Index
HTTD	High Temperature Thermal Desorption
HUDTOX	Upper Hudson River Toxic Chemical Model
IBI	Index of Biotic Integrity
IRIS	Integrated Risk Information System
ITT	Innovative Treatment Technologies (database)
kg	Kilogram
KPEG	Potassium polyethylene glycol
LOAEL	Lowest Observed Adverse Effect Level
LRC, LRCR	Low Resolution Sediment Coring Report
LTI	LimnoTech, Inc.
LTTD	Low Temperature Thermal Desorption
LWA	Length-Weighted Average
MANOVA	Multivariate Analysis of Variance
M&E	Metcalf and Eddy
MBI	Macroinvertebrate Biotic Index
MCA	Menzie-Cura and Associates
MCACES	Cost Estimating Software (USACE)
MCL	Maximum Contaminant Level
MCLG	Maximum Contaminant Level Goal
MDEQ	Michigan Department of Environmental Quality
MDPR	Molar Dechlorination Product Ratio
MEC	Mid-Range Effects Concentration
mg/kg	Milligrams per Kilogram (generally equivalent to parts per million, or ppm)
mg/L	Milligrams per Liter (generally equivalent to ppm)
MNA	Monitored Natural Attenuation
MPA	Mass per Unit Area

LIST OF ACRONYMS and ABBREVIATIONS (*cont'd*)

MS	Mass Spectroscopy
NAAQS	National Ambient Air Quality Standards
NAICS	North American Industry Coding System
NAS	National Academy of Sciences
NCP	National Oil Spill and Hazardous Substances Pollution Contingency Plan
NEPA	National Environmental Policy Act
ng/L	Nanograms per Liter, parts per trillion
NHPA	National Historic Preservation Act
NiMo	Niagara Mohawk Power Company
NOAA	National Oceanic and Atmospheric Administration
NOAEL	No Observed Adverse Effect Level
NPL	National Priorities List
NRC	National Research Council
NTCRA	Non-Time Critical Removal Action
NYCRR	New York Code of Rules and Regulations
NYSDEC	New York State Department of Environmental Conservation
NYSDOH	New York State Department of Health
NYSDOL	New York State Department of Labor
NYSDOT	New York State Department of Transportation
NYSPDES	New York State Pollutant Discharge Elimination System
O&M	Operation and Maintenance
OPRHP	Office of Parks, Recreation, and Historic Preservation
OSHA	Occupational Safety and Health Administration
OSWER	Office of Solid Waste and Emergency Response (EPA)
OU	Operable Unit
PCB	Polychlorinated Biphenyl
PCRDMP	Post-Construction Remnant Deposit Monitoring Plan
PEL	Probable Effects Level
PMCR	Preliminary Modeling Calibration Report
ppm	part(s) per million (mg/kg or mg/L)
PRG	Preliminary Remediation Goal
PSG	Project Sponsor Group
PVC	Polyvinyl Chloride
RAMP	Remedial Action Master Plan
RAO	Remedial Action Objective
RBC	Risk-Based Concentration
RBMR	Revised Baseline Modeling Report
REACH IT	Remediation and Characterization Innovative Technologies (EPA database)
RfD	Reference Dose
RI/FS	Remedial Investigation/Feasibility Study
RI	Remedial Investigation
RIMS	Remediation Information Management System
RM	River Mile
RME	Reasonable Maximum Exposure
ROD	Record of Decision
SARA	Superfund Amendments and Reauthorization Act of 1986
SAV	Submerged Aquatic Vegetation
SEC	Sediment Effect Concentration
SHPO	State Historic Preservation Office
SITE	Superfund Innovative Technology Evaluation Program
SPDES	State Pollution Discharge Elimination System

SQRT	Screening Quick Reference Tables
STC	Scientific and Technical Committee
T&E	Threatened and Endangered
TAG	Technical Assistance Grant
TAGM	Technical Assistance Guidance Memorandum (NYSDEC)
TBC	To-be-considered
TCDD	2,3,7,8-Tetrachlorodibenzo-p-dioxin
TCP	2,4,6-Trichlorophenol
TEC	Threshold Effect Concentration
TEF	Toxicity Equivalency Factor
TEQ	(Dioxin-like) Toxic Equivalent Quotient
TI	Thompson Island
TID	Thompson Island Dam
TIN	Triangulated Irregular Network
TIP	Thompson Island Pool
TLV	Threshold Limit Value
TOC	Total Organic Carbon
TOGS	Technical and Operational Guidance Series (NYSDEC)
TOPS	Trace Organics Platform Sampler
TQ	Toxicity Quotient
TR	Target Risk
TRV	Toxicity Reference Value
TSCA	Toxic Substances Control Act
TWA	Time-Weighted Average
UCL	Upper Confidence Limit
UET	Upper Effects Threshold
μg/kg	Micrograms per Kilogram, (generally equivalent to parts per billion, or ppb)
μg/L	Micrograms per Liter, (generally equivalent to parts per billion, or ppb)
USACE	United States Army Corps of Engineers
USBEA	United States Bureau of Economic Analysis
USBLS	United States Bureau of Labor Statistics
USC	United States Code
USDOC	United States Department of Commerce
USDOD	United States Department of Defense
USDOE	United States Department of Energy
USDOI	United States Department of Interior
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
VISITT	Vendor Information System for Innovative Treatment Technologies (EPA Program)
VLDPE	Very Low Density Polyethylene
WHO	World Health Organization

CONTAMINANT RISKS AND GEOCHEMISTRY

PCB Carcinogenicity
362702

No Figures or Tables

WHITE PAPER - PCB CARCINOGENICITY

(ID 362702)

ABSTRACT

EPA classifies PCBs as probable human carcinogens based on data showing that PCBs cause cancer in animals and inadequate but suggestive evidence that PCBs cause cancer in humans. EPA's guidelines for classifying the carcinogenicity of chemicals are consistent with the approaches used by other national and international agencies. Moreover, EPA's Weight of Evidence classification of PCBs as probable human carcinogens has been externally peer reviewed and is equivalent to the classifications of the National Toxicology Program, the National Institute of Occupational Safety and Health, and the International Agency for Research on Cancer, part of the World Health Organization.

In the Human Health Risk Assessment for the Hudson River PCBs Site, EPA used the current externally peer-reviewed toxicity values for PCB carcinogenicity (*i.e.*, cancer slope factors) contained in the Integrated Risk Information System, which is the Agency's consensus database of toxicity information. In the Human Health Risk Assessment, EPA summarized recent human epidemiological studies published since the 1996 PCB Cancer Reassessment. Based on a review of these newer studies, EPA determined that no change was necessary to EPA's classification of PCBs as probable human carcinogens. In the Human Health Risk Assessment, cancer risks from dioxin-like PCBs were calculated using current Toxicity Equivalency Factors developed by the World Health Organization. EPA submitted the Human Health Risk Assessment for external peer review. The peer reviewers agreed with the toxicity values EPA used in the Human Health Risk Assessment.

INTRODUCTION

The purpose of this paper is to provide an overview of EPA's process for evaluating the carcinogenicity of a chemical, development of cancer slope factors for PCBs, and the application of this toxicity information in the Human Health Risk Assessment for the Hudson River PCBs Site.

This paper is divided into four parts. The first part describes the history and development of the Agency's guidelines for carcinogenicity (USEPA, 1976, 1980, 1983a,b, 1984, 1986, 1994, 1996a, 1999a). Specific issues addressed in the guidelines include EPA's PCB Weight of Evidence classification, procedures for evaluating human epidemiological evidence and animal toxicity studies, and the use of this information in classifying the carcinogenicity of a chemical.
The second part of this paper describes the Agency's evaluation of the carcinogenicity of PCBs. It summarizes the important human epidemiological and animal studies evaluated during the 1996 Cancer Reassessment for PCB carcinogenicity (USEPA, 1996b), presents some of the new information on the cancer toxicity of PCBs evaluated by EPA since 1996, and presents the current cancer slope factors in the Integrated Risk Information System (IRIS), the Agency's consensus database of toxicity information (USEPA, 1999b).

The third part provides a list of published papers describing some of the PCB toxicity research conducted by EPA scientists in the past five years, including studies of the mechanisms by which PCBs cause cancer and other adverse health effects.

The fourth part of this paper addresses the use of PCB cancer toxicity information in the Human Health Risk Assessment (HHRA) for the Hudson River PCBs Site (USEPA, 2000a-d). Specifically, this section discusses the use of cancer toxicity information (*e.g.*, cancer slope factors) in IRIS and the toxic equivalency factors (TEFs) for dioxin-like PCBs. This section also describes the Agency's rationale for not using blood PCB levels in workers to evaluate cancer risks for people who eat PCB-contaminated fish from the Hudson River.

DEVELOPMENT OF EPA CARCINOGEN GUIDELINES

EPA's Carcinogen Guidelines (USEPA, 1976, 1983a,b, 1984, 1986, 1994, 1996a, 1999a) were used in determining the carcinogenicity of PCBs. These guidelines provide EPA's general framework for evaluating the cancer toxicity data (human and animal) for determining the Weight of Evidence classifications and cancer slope factors of chemicals. The Carcinogen Guidelines were developed after an evaluation of the procedures used by the International Agency for Research on Cancer (IARC), which is part of the World Health Organization (WHO) and the National Toxicology Program (NTP), which is part of the National Institutes of Health.

In 1976, EPA issued interim procedures and guidelines for health risks and economic impact assessments of suspected carcinogens (USEPA, 1976). In 1979, the Interagency Regulatory Liaison Group held a meeting regarding carcinogens and methods for evaluating the technical adequacy of animal toxicity studies (IRLG, 1979).

In 1982, IARC issued a monograph on the evaluation of the carcinogenic risk of chemicals to humans (IARC, 1982). In 1984, NTP's Ad Hoc Panel on Chemical Carcinogenesis Testing and Evaluation issued a report regarding selection of dose levels for long-term animal studies (NTP, 1984).

In 1984, EPA began its work on the Guidelines for Carcinogen Risk Assessment (USEPA, 1984). Draft guidelines were developed by a workgroup composed of expert scientists from throughout the Agency. The draft was externally peer reviewed by expert scientists in the field of carcinogenesis and related scientific disciplines, from universities, environmental groups, industry, labor and other governmental agencies. The guidelines were then proposed for public comment in the *Federal Register* (USEPA, 1984).

In 1986, EPA issued the Guidelines for Carcinogen Risk Assessment (September 24, 1986), which are the product of a two year Agency-wide effort, which has included many scientists from the larger scientific community (USEPA, 1986). These guidelines incorporated comments and responses to external peer review comments and comments from the Agency's Science Advisory Board and were finalized and published in the *Federal Register* (USEPA, 1986). The guidelines incorporate information from the previous documents and also information and procedures used by NTP and IARC (*e.g.*, the Weight of Evidence classification is based on the IARC approach). The 1986 Guidelines incorporated principles of the science for chemical carcinogens issued by the Office of Science and Technology Policy in 1985 (OSTP, 1985).

On April 23, 1996, the Proposed Guidelines for Carcinogen Risk Assessment were published in the *Federal Register* (USEPA, 1996a) for a 120-day public review and comment period. The Proposed Carcinogen Guidelines are a revision of EPA's 1986 Guidelines for Carcinogen Risk Assessment (USEPA, 1986) and, when final, will replace the 1986 cancer guidelines (USEPA, 1996a). The full text of the *Federal Register* notice is available on the web at www.epa.gov/ncea/.

Changes since the 1986 Carcinogen Guidelines (USEPA, 1986) are summarized in the 1996 Proposed Carcinogen Guidelines (USEPA, 1996a), as follows:

> "Since the publication of the 1986 cancer guidelines, there is a better understanding of the variety of ways in which carcinogens can operate. Today, many laboratories are moving toward adding new test protocols in their programs directed at mode of action questions. Therefore, the Proposed Guidelines provide an analytical framework that allows for the incorporation of all relevant biological information, recognize a variety of situations regarding cancer hazard, and are flexible enough to allow for consideration of future scientific advances."

In 1999, EPA proposed revised Carcinogen Guidelines (USEPA, 1999a) in response to comments by the EPA Science Advisory Board. The approaches outlined in the proposed revised guidelines are consistent with the 1996 Cancer Reassessment for PCBs (USEPA, 1996a). The 1999 proposed guidelines were developed to address issues regarding children's risk from exposure to carcinogens. On November 21, 2001, EPA published an announcement in the *Federal Register* soliciting additional scientific information and comments on the draft revised Carcinogen Guidelines that could assist EPA in completing the final Guidelines (USEPA, 2001). *This Federal Register* notice also stated that, until final Guidelines are issued, the July 1999 draft revised Guidelines will serve as EPA's interim guidance to EPA risk assessors preparing cancer risk assessments.

As outlined above, the carcinogenicity guidelines were developed within the Agency, published in the *Federal Register* for comment, and externally peer-reviewed. EPA responded to comments on the proposed guidelines and made changes based on a review of the comments submitted by these groups and individuals. The guidelines were also submitted for review to EPA's Science Advisory Board, an external scientific review panel.

EPA'S EVALUATION OF PCB CARCINOGENICITY

EPA classified PCBs as probable human carcinogens in 1988 (USEPA, 1988) and reaffirmed this classification in 1996 (USEPA, 1996b). EPA's classification is based on a weight of the evidence. The available classifications for chemicals are a) carcinogenic to humans, b) probably carcinogenic to humans, c) possibly carcinogenic to humans, d) not classifiable as to human carcinogenicity, and e) evidence of non carcinogenicity to humans. The EPA classification of PCBs as probable human carcinogens is equivalent to the NTP, NIOSH, and IARC classifications for PCBs (NTP, 1981, 2000; NIOSH, 1977; IARC, 1978, 1987).

Following the 1988 evaluation of the carcinogenicity of PCBs, EPA conducted a reassessment of the carcinogenicity of PCBs in 1996 (USEPA, 1996b, see also www.epa.gov/ncea). In developing EPA's cancer reassessment for PCBs, EPA circulated the document within the Agency to more than 40 expert Agency scientists who reviewed and commented on the document. In addition, the document was submitted for external peer review to a panel of 16 experts in various areas of PCB toxicity, exposure and carcinogenicity including a scientist from the General Electric Company (USEPA, 1996b,c). The panel agreed with EPA's conclusions (USEPA, 1996b,c) regarding the carcinogenicity of PCBs and recommended that the Agency use the Brunner *et al.* (1996) study to develop the cancer slope factor for PCBs. Following review by the Agency and a panel of external reviewers (Koller, 1996), EPA used data from the Brunner *et al.* (1996) study in the 1996 PCB Cancer Reassessment (USEPA, 1996b). This information was also incorporated into the IRIS file for PCBs (USEPA, 1999b), submitted to Congress in October 1996 and published in an article by the Agency's lead author of the 1996 PCB Cancer Reassessment (Cogliano, 1998).

The 1996 PCB Cancer Reassessment was conducted consistent with the 1996 Proposed Cancer Guidelines (USEPA, 1996a, pp. 6, 55-56), as follows:

> "This new assessment adopts a related approach that distinguishes among PCB mixtures by using information on environmental processes. Environmental processes have profound effects that can decrease or increase toxicity, so toxicity of an environmental mixture is only partly determined by the original commercial mixture. This new assessment, therefore, considers all cancer studies (which used commercial mixtures only) to develop a range of dose-response slopes, then uses information on environmental processes to provide guidance on choosing an appropriate slope for representative classes of environmental mixtures and different exposure pathways."

The 1996 PCB Cancer Reassessment is also consistent with the 1999 Revised Carcinogen Guidelines, which address children's health (USEPA, 1999a).

EPA considered data from human epidemiological studies and animal studies in determining that PCBs are probable human carcinogens. In 1988, EPA concluded there was inadequate but suggestive evidence that PCBs cause cancer in humans and sufficient evidence that PCBs cause cancer in animals (USEPA, 1988). In 1996, EPA reaffirmed this classification, concluding (USEPA, 1996b), "Overall, the human studies have been considered to provide limited…to inadequate…evidence of carcinogenicity. The animal studies, however, have been considered to provide sufficient evidence of carcinogenicity" (USEPA, 1996b).

Human Epidemiological Studies

The peer reviewers of EPA's 1996 PCB Cancer Reassessment found inadequacies in the epidemiological data with regard to limited cohort size, problems in exposure assessment, lack of data on confounding factors, and the fact that occupational exposures may be to different congener mixtures than found in environmental exposures. The peer reviewers stated (USEPA, 1996c):

> "Most researchers think that PCBs act mainly as tumor promoters. Thus, at nontoxic doses, PCBs might be expected to increase cancer risk mainly in humans that have sustained cancer initiation due to exposure to genotoxicants or to the presence of a mutant gene. For common cancers that have complex and multiple etiologies, promotive effects will be seen by epidemiology only if specifically looked for. Epidemiological studies have not thus far tested this hypothesis. "

EPA has summarized the human epidemiological studies used to classify PCBs as probable human carcinogens (USEPA, 1996b, 1999b). The human epidemiological evidence is described in USEPA (1999b) as follows (SMR=standard mortality ratio, CI=confidence interval, p=level of statistical significance):

> "Inadequate. A cohort study by Bertazzi *et al.* (1987) analyzed cancer mortality among workers at a capacitor manufacturing plant in Italy. PCB mixtures with 54%, then 42% chlorine were used through 1980. The cohort included 2100 workers (544 males and 1556 females) employed at least 1 week. At the end of follow-up in 1982, there were 64 deaths reported, 26 from cancer. In males, a statistically significant increase in death from gastrointestinal tract cancer was reported, compared with national and local rates (6 observed, 1.7 expected using national rates, SMR=346, CI=141-721; 2.2 expected using local rates, SMR=274, CI=112-572). In females, a statistically significant excess risk of death from hematologic cancer was reported, compared with local, but not national, rates (4 observed, 1.1 expected, SMR=377, CI=115-877). Analyses by exposure duration, latency, and year of first exposure revealed no trend; however, the numbers are small.
>
> A cohort study by Brown (1987) analyzed cancer mortality among workers at two capacitor manufacturing plants in New York and Massachusetts. At both plants the Aroclor mixture being used changed twice, from 1254 to 1242 to 1016. The cohort included 2588 workers (1270 males and 1318 females) employed at least 3 months in areas of the plants considered to have potential for heavy exposure to PCBs. At the end of follow-up in 1982, there were 295 deaths reported, 62 from cancer. Compared with national rates, a statistically significant increase in death from cancer of the liver, gall bladder, and biliary tract was reported (5 observed, 1.9 expected, SMR=263, $p<0.05$). Four of these five occurred among females employed at the Massachusetts plant. Analyses by time since first employment or length of employment revealed no trend; however, the numbers are small.
>
> A cohort study by Sinks *et al.* (1992) analyzed cancer mortality among workers at a capacitor manufacturing plant in Indiana. Aroclor 1242, then 1016, had been used. The cohort included 3588 workers (2742 white males and 846 white females) employed at least 1 day. At the end of follow-up in 1986, there were 192 deaths reported, 54 from cancer. Workers were classified into five exposure zones based on distance from the impregnation ovens. Compared with national rates, a statistically significant excess risk of death from skin cancer was reported (8 observed, 2.0 expected, SMR=410, CI=180-800); all were malignant melanomas. A proportional hazards analysis revealed no pattern of association with exposure zone; however, the numbers are small.

> Other occupational studies by NIOSH (1977), Gustavsson *et al.* (1986) and Shalat *et al.* (1989) looked for an association between occupational PCB exposure and cancer mortality. Because of small sample sizes, brief follow-up periods, and confounding exposures to other potential carcinogens, these studies are inconclusive.
>
> Accidental ingestion: Serious adverse health effects, including liver cancer and skin disorders, have been observed in humans who consumed rice oil contaminated with PCBs in the "Yusho" incident in Japan or the "Yu-Cheng" incident in Taiwan. These effects have been attributed, at least in part, to heating of the PCBs and rice oil, causing formation of chlorinated dibenzofurans, which have the same mode of action as some PCB congeners (ATSDR, 1993; Safe, 1994)."

Animal Data

EPA determined that PCBs cause cancer in animals based on animal bioassay data. The NTP and IARC also conclude that PCBs are animal carcinogens (NTP, 1981; IARC, 1987). ATSDR's Toxicological Profile (ATSDR, 2000) states, "there is conclusive evidence that commercial PCB mixtures are carcinogenic in animals based on induction of tumors in the liver and thyroid". EPA’s evaluation (USEPA, 1996b, 1999b) of the animal bioassay data for PCBs is summarized below:

> “A 1996 study found liver tumors in female rats exposed to Aroclors 1260, 1254, 1242, and 1016, and in male rats exposed to 1260. These mixtures contain overlapping groups of congeners that, together, span the range of congeners most often found in environmental mixtures. Earlier studies found high, statistically significant incidences of liver tumors in rats ingesting Aroclor 1260 or Clophen A 60 (Kimbrough *et al.*, 1975; Norback and Weltman, 1985; Schaeffer *et al.*, 1984). Mechanistic studies are beginning to identify several congeners that have dioxin-like activity and may promote tumors by different modes of action. PCBs are absorbed through ingestion, inhalation, and dermal exposure, after which they are transported similarly through the circulation. This provides a reasonable basis for expecting similar internal effects from different routes of environmental exposure. Information on relative absorption rates suggests that differences in toxicity across exposure routes are small.”

Varying Dose Levels Tested

EPA evaluated a number of animal bioassays regarding the carcinogenicity of PCBs that were conducted at varying dose levels, not only at the Maximum Tolerated Dose (MTD). Consistent with NTP and IARC protocols (NTP, 1984; IARC, 1982, 1987), animal studies are conducted at varying levels below the MTD to aid in establishing a dose-response curve. Data at or near the MTD level were evaluated consistent with EPA's 1986 Carcinogen Guidelines (USEPA, 1986), which state: "Long-term animal studies at or near the MTD are used to ensure an adequate power for the detection of carcinogenic activity."

EPA's 1996 PCB Cancer Reassessment (Table 2-1, USEPA, 1996b), which showed the liver tumor incidences in rats from lifetime exposure studies from 1975 to 1985, generally included a control group of rats not exposed to PCBs and other groups exposed to varying concentrations of PCBs (*i.e.*, 25 ppm, 50 ppm, and 100 ppm). The cited studies include Kimbrough *et al.* (1975), NCI (1978), Schaeffer *et al.* (1984), and Norback and Weltman (1985). The Brunner *et al.* (1996) rat study (later published as Mayes *et al.*, 1998) included doses of PCBs ranging from the control (0 ppm), to 25 ppm, 50 ppm, 100 ppm and 200 ppm. The Brunner et al. (1996) lifetime study data, in which rats were exposed to PCBs at levels less than the MTD for 104 weeks, demonstrated that the rats fed diets of PCBs had statistically significant, dose-related, increased incidences of liver tumors from each Aroclor mixture (USEPA, 1996b).

In addition, the partial lifetime studies that were evaluated by EPA also included exposures to various concentrations of PCBs. Kimbrough *et al.* (1972) included dose levels of 0 ppm, 20 ppm, 100 ppm, 500 ppm, or 1,000 ppm for Aroclor 1254 or 1260. Other studies include Kimbrough and Linder (1974), in which BALB/cJ mice were exposed to 300 ppm of Aroclor 1254 for 11 months or for six months followed by five months without exposure to PCBs. Kimura and Baba (1973) exposed Donryu rats to diets ranging from 38 to 462 ppm of Kanechlor (a trade name for PCBs) 400. Ito *et al.* (1973) exposed dd mice to 0 ppm, 100 ppm, 250 ppm or 500 ppm of Kanechlor 300, 400 or 500. Ito *et al.* (1974) exposed Wistar rats to diets of 0, 100, 500, or 1,000 ppm of Kanechlor 300, 400, or 500 ppm. Rao and Banerji (1988) exposed male Wistar rats to diets of 0 ppm, 50 ppm or 100 ppm of Aroclor 1260.

Gender Differences in Tumors

EPA followed appropriate guidelines and policies in extrapolating the data from the Brunner *et al.* (1996) rat study to humans. As stated in the PCB Cancer Reassessment (USEPA, 1996b, see p. 44), "the different responses for male and female rats (Brunner *et al.*, 1996) suggest the possibility of developing different potency values for males and females. In view of the 91% response in male Wistar rats (Schaeffer *et al.*, 1984), as well as the sensitivity of male mice (Kimbrough and Linder, 1974; Ito *et al.*, 1973), it is premature to conclude that females are always more sensitive. The PCB Cancer Reassessment (USEPA, 1996b) provides summary tables of the ranges of potency values based on data from both males and females. The potencies are based primarily on the range of Aroclors 1260, 1254, 1242 and 1016 tested in female Sprague-Dawley rats, but other studies were considered also.

Benign and Malignant Tumors

Consistent with the framework set forth in the Agency's Carcinogen Guidelines (USEPA, 1986, 1996a, 1999a), EPA considered benign as well as malignant tumors in evaluating the carcinogenicity of PCBs because both benign and malignant tumors are considered to be representative of related responses to the PCBs. Benign tumors progressed to malignant tumors in multiple studies.

EPA is not alone in using this approach to evaluate tumor data in assessing the carcinogenicity of chemicals. The Agency's 1996 proposed Carcinogen Guidelines (USEPA, 1996a) noted,

> "As in the approach of the National Toxicology Program and the International Agency for Research on Cancer, the default is to include benign tumors observed in animal studies in the assessment of animal tumor incidence if they have the capacity to progress to the malignancies with which they are associated. This treats the benign and malignant tumors as representative of related responses to the test agents, which is scientifically

> appropriate. This is a science policy decision that is somewhat more conservative of public health than not including benign tumors in the assessment. Nonetheless, in assessing findings from animal studies, a greater proportion of malignancy is weighed more heavily than a response with a greater proportion of benign tumors. Greater frequency of malignancy of a particular tumor type in comparison with other tumor responses observed in an animal study is also a factor to be considered in selecting the response to be used in dose response assessment".

With respect to PCB carcinogenicity, in 1996, EPA described a study by Norback and Weltman (1985) that demonstrated tumor progression as follows (USEPA, 1996b):

> "Norback and Weltman (1985). Groups of male or female Sprague-Dawley rats were fed diets with 0 or 100 ppm Aroclor 1260 for 16 months; the latter dose was reduced to 50 ppm for 8 more months. After 5 additional months on the control diet, the rats were killed and their livers were examined. Partial hepatectomy was performed on some rats at 1, 3, 6, 9, 12, 15, 18, and 24 months to evaluate sequential morphologic changes. In males and females fed Aroclor 1260, liver foci appeared at 3 months, area lesions at 6 months, neoplastic nodules at 12 months, trabecular carcinomas at 15 months, and adenocarcinomas at 24 months, demonstrating progression of liver lesions to carcinomas. By 29 months, 91 percent of females had liver carcinomas and 95 percent had carcinomas or neoplastic nodules; incidences in males were lower, 4 and 15 percent, respectively (see table 2–1)."

EPA also evaluated PCB carcinogenicity based on lifetime and stop studies of rats fed diets containing Aroclors 1260, 1254, 1242 or 1016, using data from Brunner *et al.* (1996). From the lifetime study data, in which rats were exposed to PCBs for 104 weeks, EPA concluded that the rats fed diets of PCBs had statistically significant, dose-related, increased incidences of liver tumors from each Aroclor mixture (USEPA, 1996b; Cogliano, 1998). From the stop study data, in which the rats were exposed to PCBs for 52 weeks and then PCB exposure was stopped, EPA determined that, for Aroclors 1254 and 1242, tumor incidences were approximately half those of the lifetime study; that is, nearly proportional to exposure duration. In contrast, for Aroclor 1016, stop-study tumor incidences were zero, while for Aroclor 1260 they were generally greater than half as many as in the lifetime study.

Earlier studies found high, statistically significant incidences of liver tumors in various strains of rats ingesting Aroclor 1260 or Clophen A60 (Kimbrough *et al.*, 1975, Norback and Weltman, 1985; Schaeffer *et al.*, 1984). Kimbrough *et al.* (1975) found significantly increased hepatocellular carcinomas in rats fed Aroclor 1260. Schaeffer *et al.* (1984) found male Wistar rats in the shortest exposed group (16.4 months) had preneoplastic liver lesions, and after 23 months had hepatocellular carcinomas. Norback and Weltman (1985) studied Sprague-Dawley rats exposed to Aroclor 1260 and found that by 29 months 91% of females had liver carcinomas. In addition, the Brunner *et al.* (1996) study found several of the tumors were hepatocholangiomas, a rare bile duct tumor seldom seen in control rats.

The data from the studies described above are the basis for EPA’s determination that PCBs cause cancer in animals. Benign tumors progressed to malignant tumors in multiple studies, in different strains of rats, and at different dose levels of PCBs.

Cancer Slope Factor (CSF)

The quantification of carcinogenicity is a value called a cancer slope factor (CSF). As outlined in the EPA Carcinogen Guidelines (USEPA, 1986; 1996a), EPA favors basing CSFs on human epidemiological studies, which requires quantitative information on both exposure and response. However, for PCBs, EPA concluded that the human epidemiological data are insufficient to develop CSFs (USEPA, 1996b). During the peer review of EPA's 1996 PCB Cancer Reassessment (USEPA, 1996c), EPA included charge questions to the peer-reviewers requesting specific evaluation of human epidemiological evidence as a basis for developing the CSFs for PCBs. The peer reviewers supported EPA's conclusion that it is not feasible to use the human epidemiological data to develop CSFs for PCBs (USEPA, 1996c).

EPA used the proposed 1996 Carcinogen Guidelines (USEPA, 1996a) to develop the CSFs for PCBs. Following review of the carcinogenicity data and based primarily on the Brunner *et al.* (1996), EPA developed separate PCB CSFs for inhalation and ingestion, and provided a recommendation for exposure by dermal contact. The oral CSF for PCBs developed in 1988 (USEPA, 1988) was revised downward in 1996 from 7.7 mg/kg-day^{-1} to 2.0 mg/kg-day^{-1}. In the 1996 PCB Cancer Reassessment (USEPA, 1996b, p. 35), EPA explained,

> "This difference in cancer slope factor is attributable to three factors, each responsible for reducing the slope by approximately one-third: the rat liver tumor reevaluation (Moore *et al.*, 1994), use of the new cross-species scaling factor (USEPA, 1992) and not using a time weighted average dose."

Similarly, when these factors are applied to the CSF derived from the Norback and Weltman (1985) study, the CSF is reduced from 7.7 mg/kg-day^{-1} to 2.2 mg/kg-day^{-1}.

As part of EPA's 1996 PCB Cancer Reassessment, EPA evaluated an approach regarding PCB congener persistence in the body (Brown (1994). EPA identified some limitations of using this approach in the development of CSFs for PCBs, as follows (USEPA, 1996b):

> "Reconstruction of past exposure is problematic because different mixtures had been in use over the years, the distribution of exposure and absorption by route and congener is unknown, and congener persistence in the body varies greatly from congener to congener (Brown, 1994) and person to person (Steele *et al.*, 1986)."

Human Epidemiological Studies Since the 1996 PCB Cancer Reassessment

Since the 1996 PCB Cancer Reassessment (USEPA, 1996b), additional studies regarding the carcinogenicity of PCBs in humans have been published (*e.g.*, Gustavsson and Hogstedt, 1997; Hardell *et al.*, 1996; Rothman *et al.*, 1997; Tironi *et al.*, 1996; Yassi *et al.*, 1994; Loomis *et al.*, 1997; Kimbrough *et al.,* 1999 [discussed separately]).

EPA has noted issues with many of the studies of occupationally exposed individuals working in industrial plants in the U.S. and internationally (USEPA, 1996b). Issues include the small number of tumors found, making it difficult to associate the exposures with specific manufacturing processes in the plant studied by the investigators (*i.e.*, high exposure, medium exposures, or low exposure areas); mortality rather than morbidity as a study objective; the lack of historical data on exposures; and confounding from exposures to chemicals other than PCBs within the plant. A brief summary of the studies and their conclusions regarding the carcinogenicity of PCBs is provided below by type of cancer and population studied.

Breast Cancer

Recent studies have investigated PCB exposures and breast cancer. EPA has evaluated these studies and concluded that it is not possible to attribute a cause and effect association between PCB exposure and breast cancer given the sparse data available (USEPA, 1997).

Study results suggested that PCBs increase the risk of breast cancer after menopause (Moysich *et al.*, 1998) and research has suggested a mechanism by which PCBs can contribute to cancer, including breast cancer (Oakley *et al.*, 1996). Other studies have failed to show an association between PCB exposure and breast cancer (*e.g.*, Hoyer *et al.*, 1998, see studies reviewed in USEPA, 1997 and Table D-1 of USEPA, 2000a).

Researchers have suggested the need to consider PCB levels in women prior to the time of breast cancer diagnosis (*e.g.*, Adami *et al.*, 1995). The critical or sensitive period of exposure for the developing breast tissue may be as an infant or during puberty, in which case the current procedure of measuring blood PCB levels at the time of diagnosis may not be an appropriate biomarker of exposure.

Organ Sites Excluding Breast Cancer

EPA has also evaluated studies on PCB exposures and cancers other than breast cancer. Based on the available epidemiological evidence, EPA believes that the data are inconclusive with respect to the association of PCBs and cancer in humans, including hepatobiliary, hematological, malignant melanoma, rectal, gastrointestinal tract, pancreatic, and endometrial cancers based on the limitations of the epidemiological studies (USEPA, 1999b).

Kimbrough et al. (1999a) Occupational Study

In 1999, Dr. Kimbrough and colleagues published a study of cancer mortality in workers exposed to PCBs (Kimbrough *et al.*, 1999a). The paper describes a study of workers from two GE capacitor manufacturing plants in New York State. In this study, mortality (deaths) from all cancers was determined for 7,075 females and males who worked at the GE facilities for at least 90 days between 1946 and 1977. The total number of deaths from all causes was 1,195 people, and the total number of deaths caused by cancer was 353 people. No significant elevations in mortality for any site-specific cause were found in the hourly worker cohort (*i.e.*, group). No significant elevations were seen in the most highly exposed workers. Mortality from all cancers was significantly below expected in hourly male workers and comparable to expected for hourly female workers. Several researchers submitted Letters to the Editor identifying limitations of the Kimbrough *et al.* (1999a) study, which were published in the Journal of Occupational and Environmental Medicine (Bove *et al.*, 1999; Frumkin and Orris, 1999). The response to these letters was also published (Kimbrough *et al.* (1999b).

EPA performed a preliminary review of the Kimbrough *et al.* (1999a) study and identified aspects of the study that suggest that the study will not change the Agency's conclusions regarding the carcinogenicity of PCBs (USEPA, 2000a-c). The primary limitation, which is shared by other similar epidemiological studies, is that the degree of exposure is not well characterized.

As part of its review, EPA sent copies of the Kimbrough *et al.* (1999a) paper to several researchers requesting an evaluation regarding whether this new paper would change the Weight of Evidence classification of PCBs as probable human carcinogens. The findings from these letters are summarized below:

Dr. D. Ozonoff of the Boston University School of Public Health concluded (Ozonoff, 1999):

> "In short, we have here another "data point". It should be judiciously interpreted and used with the caution appropriate to studies of this type. In particular, this means not giving undue weight to its failure to show associations previously revealed, since there are too many factors that would mitigate against being able to show them in this study."

Dr. M. Harnois of the Massachusetts Department of Environmental Protection concluded (Harnois, 1999):

> "A subgroup that is masked in this study is the one containing hourly male workers exposed to Aroclor 1254 by dermal contact, incidental ingestion, and inhalation for at least 5 years and followed for at least 20 years. This group could have different cancer frequencies from those presented in the report, being definitely exposed to a known carcinogenic mixture for a prolonged interval and observed for an interval that could allow development of tumors.
>
> This report deals mostly with deaths due to cancer effects, but we know that reproductive, nervous and immunological effects can also occur. These are beyond the scope of the research report, but may be ignored by readers who assume that cancer is the only effect of PCBs."

Dr. T. Mack of the University of Southern California, Norris Comprehensive Cancer Center concluded (Mack, 1999):

> "I guess my bottom line is that the summary statements ("lack of any significant elevations adds important information" and "lack of consistent findings --- would suggest a lack of an association") in the paper are appropriate. I think that it is appropriate to downgrade the priority given to PCB's. However, based on the animal studies (and recognizing a. the possibility limited relevance to man and b. the absence of any confirmation of liver cancer in humans) and on this very small amount of information pointing to colorectal tumors, I don't think that this potential carcinogenicity of PCB's can be completely dismissed. I recognize the flimsiness of the evidence, and that a less conservative person could persuasively argue the other way."

The ATSDR Toxicological Profile for PCBs (ATSDR, 2000) summarizes the limitations of the exposure information from Kimbrough *et al.*, (1999a) as follows:

> "PCB exposures were predominantly to Aroclor 1254 from 1946 to 1954, Aroclor 1242 from 1954 to 1971, and Aroclor 1016 from 1971 to 1977. Exposures were qualitatively classified as high, low, or undefinable based on types and locations of jobs and some area measurements. No personal exposure monitoring was performed, although previously reported data on 290 self-selected workers from one of the plants had serum PCBs levels in ranges of 6 to 2,530 and 1 to 546 ppb for lower and higher chlorinated homologs, respectively (Wolff *et al.*, 1982). Workers with high exposure jobs had direct PCB contact (dermal and/or inhalation), workers with lower exposure jobs primarily had inhalation exposure to background levels of PCBs in the plant, and workers with undefinable exposures had exposures that varied depending on whether tasks were performed. Exposure-specific analysis was limited to workers with the greatest potential for exposure (*i.e.*, hourly workers who ever worked in a high exposure job, worked for at least 6 months in a high-exposure job, worked for at least 1 year in a high-exposure job). Workers who exclusively worked in high-exposure jobs could not be analyzed as a separate group due to small numbers (112 males, 12 females)."

The Toxicological Profile for PCBs concluded (ATSDR, 2000):

> "Interpretation of the Kimbrough *et al.* (1999a) findings is complicated by a few study limitations and biases, including some exposure misclassifications related to use of length of employment alone as a surrogate of exposure, potentially insufficient dosage differences between exposed and comparison groups, a degree of selection bias due to the healthy worker effect that may have resulted in an under estimate of SMRs, concern for low statistical power due to the small number of deaths from site-specific cancers in some of the group (*e.g.*, female hourly workers with high exposure and $\geq$ 20 years latency), relatively young age at follow-up, and use of the general population for comparison rather than an internal control group or a group of workers from another company. These issues are discussed by Bove *et al.* (1999), Frumkin and Orris (1999), and Kimbrough *et al.*, (1999b). Some of the limitations are typical of occupational cohort mortality studies, and strengths of the study include its size (the largest cohort of PCB workers ever studied) and essentially complete follow-up of long duration. Unresolved are the puzzling Kimbrough *et al.* (1999a) findings of significantly lower than expected mortality from all cancers among males and the lower number of observed cases of liver and biliary tract cancers among females compared to the smaller cohort studies by Brown *et al.* (1987), a subset of the same study population. These unresolved findings suggest that ascertainment of cancer mortality was not completed in this study. Overall, the study limitations are sufficient to cast doubt on the negative findings for liver and biliary tract cancer and other site-specific cancers."

In light of the information summarized above regarding the limitations of the Kimbrough *et al.* (1999a) study, which are similar to the limitations of other human epidemiological studies, EPA has not changed its Weight of Evidence classification of PCBs as probable human carcinogens.

EPA'S PCB RESEARCH

EPA has conducted significant research on PCBs and the mechanisms of PCB action. Following is a partial list of research conducted by EPA's Office of Research and Development from 1996 to 2000. In addition, EPA has worked with other federal agencies through programs such as the Superfund Basic Research Program (part of the National Institute of Environmental Health Sciences) to fund research on PCB toxicity through grants to a number of Universities (Massachusetts Institute of Technology, State University of New York-Albany, University of Kentucky, etc.) that are evaluating PCB toxicity.

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HUDSON RIVER PCBS SITE

IRIS toxicity values undergo an extensive internal and external peer review process (USEPA, 1996b,c and 1999b) and are thus the preferred toxicity values for use in Superfund risk assessments (USEPA, 1989, 1993, 1996b,c). The use of IRIS data in the evaluation of the toxicity of chemicals at Superfund sites addresses EPA's goal of using consistent toxicity information in risk assessments at Superfund sites across the country.

Consistent with EPA's risk assessment guidance (USEPA, 1989, 1990, 1993), in the HHRA for the Hudson River PCBs Site, EPA evaluated newer studies of PCB toxicity (USEPA, 2000a,b). Based on this review, EPA determined that these newer studies would not change the conclusions of the 1996 PCB Cancer Reassessment (*i.e.*, that PCBs are probable human carcinogens) and that it was appropriate to use the toxicity information and CSFs in IRIS in the Site-specific risk assessment (USEPA, 1996b,c; 2000a-d).

The peer reviewers for the HHRA agreed with EPA's use of the toxicity information in IRIS, but recommended that EPA provide an update of the data to identify recently published studies (ERG, 2000). In response, EPA updated the list of human epidemiology studies in Appendix D of the Revised HHRA (USEPA, 2000a). EPA identified a number of limitations with these newer human epidemiological studies similar to those identified in the IRIS file for PCBs (USEPA, 1999a), including lack of sufficient exposure information, failure to adequately account for co-exposure to other compounds, and inconsistency between study results.

EPA recognizes that environmental processes can alter the congener composition of a PCB mixture in the environment. The CSFs in IRIS are based on studies using a number of different Aroclor mixtures (*i.e.*, the commercial formulation of PCBs including Aroclor 1016, 1242, 1254, and 1260), which together span the range of congeners most frequently found in environmental mixtures (USEPA, 1996b). IRIS provides for using a lower CSFs for risk calculations when congener analysis demonstrates a predominance of the lower chlorinated congeners (*i.e.*, when congener or isomer analysis verifies that congeners with more than four chlorine atoms comprise less than 1/2 percent of the total PCBs). This lower CSF was not used in the HHRA based on congener analysis of Hudson River fish.

Dioxin-like PCBs

Consistent with EPA guidance and procedures (USEPA, 1996b), the revised HHRA (USEPA, 2000a) evaluated cancer risks from exposure to dioxin-like PCBs using the latest scientific consensus on TEFs for dioxin-like PCBs (USEPA, 1996b), as an additional consideration for the risk manager. Risks from dioxin-like PCBs were not combined with non-dioxin-like PCBs, based on EPA's ongoing effort to develop a procedure for combining these cancer risks to avoid potential double counting.

Effect of PCB Exposure on Blood Levels

EPA followed risk assessment guidance and procedures (USEPA, 1989, 1990, 1993, 1996b) to quantify cancer risks to individuals exposed to PCBs at the Hudson River PCBs Site in the HHRA (USEPA, 2000a). The approach used in the HHRA is different than measurement of blood PCB levels in former capacitor workers. First, the HHRA evaluates current and future exposures, while the data on PCB levels in blood integrates past exposure. Second, capacitor workers were primarily exposed through dermal contact and inhalation of PCBs, whereas anglers, which had the highest cancer risks evaluated in the HHRA, would be exposed to PCBs through ingestion of contaminated fish caught in the Hudson River. Third, in the HHRA EPA evaluated cancer risks to the RME individual, whereas for capacitor workers the level of exposure is generally not known. Fourth, the PCB congener profile in the capacitor plant is likely to be different from the congener profile of PCBs that are bioaccumulated in the fish. Lastly, EPA is concerned with potential exposures to the human population including sensitive groups that may include the fetus exposed from mothers who consumed PCB-contaminated fish, infants exposed to PCBs through breast milk, young children, adolescents, adults, and individuals with pre-existing medical conditions (USEPA, 2000a); many of these sensitive groups may not be represented in a healthy worker population. As stated in EPA's 1996 PCB Cancer Reassessment (USEPA, 1996b):

> "people with decreased liver function, including inefficient glucuronidative mechanism in infants, can have less capacity to metabolize and eliminate PCBs (Calabrese and Sorenson, 1977). Additionally, approximately 5% of nursing infants receive a steroid in human milk that inhibits the activity of glucuronyl transferase, further reducing PCB metabolism and elimination (Calabrese and Sorenson, 1977)."

Differences between occupational exposures and exposure through ingestion of contaminated fish were discussed in the 1996 PCB Cancer Reassessment (USEPA, 1996b). Notably, a study of people exposed through eating contaminated fish (Hovinga *et al.*, 1993) suggests that the PCB mixtures in fish can be more persistent than those to which the workers were exposed. From 1977 to 1985, mean PCB serum levels (quantified using Aroclor 1260 as a reference standard) from 111 Great Lakes fish eaters decreased only slightly from 20.5 to 19.0 ppb. This indicates that the rate of decline in the fish eating populations will be slower than that for the workers.

ATSDR's Toxicological Profile (ATSDR, 2000) states that there are no known treatment methods for reducing body burdens of PCBs, concluding that limiting or preventing further exposures appears to be the most practical method for reducing PCB body burdens.

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PCB Non-Cancer Health Effects
362704

No Figures or Tables

WHITE PAPER – PCB NON-CANCER HEALTH EFFECTS

(ID 362704)

ABSTRACT

Non-cancer health effects associated with exposure to PCBs include reduced birth weight, learning problems, and reduced ability to fight infection. The quantification of non-cancer health effects is a Reference Dose, which is a dose below which non-cancer health effects are not expected to occur over a lifetime. EPA has established guidelines for evaluating non-cancer health effects and developing Reference Doses for chemicals. These guidelines were externally peer reviewed. Using these guidelines and associated documents, EPA developed a Reference Dose for Aroclor 1016, which was externally peer reviewed. EPA used the same methodology to develop a Reference Dose for Aroclor 1254, which was internally peer reviewed. EPA's Reference Dose for Aroclor 1254 is consistent with the chronic Minimal Risk Level for PCBs developed by the Agency for Toxic Substances and Disease Registry. EPA is currently updating the non-cancer toxicity information for PCBs contained in the Integrated Risk Information System, which is the Agency's consensus database of toxicity information.

In the Human Health Risk Assessment for the Hudson River PCBs Site, EPA summarized recent studies published since 1994, including studies on developmental/neurotoxic effects, thyroid and immunological effects, reproductive effects, and neurological effects in adults. Based on a review of these studies, EPA determined that it was appropriate to use the current Reference Doses for PCBs in the Human Health Risk Assessment. EPA submitted the Human Health Risk Assessment for external peer review, and the peer reviewers agreed with the toxicity values used in the Human Health Risk Assessment.

INTRODUCTION

The purpose of this paper is to provide an overview of EPA's process for evaluating the non-cancer toxicity of a chemical, development of non-cancer Reference Doses (RfDs) for PCBs, and the application of this toxicity information in the Human Health Risk Assessment for the Hudson River PCBs Site.

This paper is divided into three parts. The first part describes EPA's non-cancer guidelines and background documents for developing reference doses (RfDs) (USEPA, 1986a-b, 1991, 1992, 1993a,b, 1996a, 1998). These documents set forth principles and procedures for evaluating non-cancer toxicity information.

The second part of this paper describes the Agency's evaluation of the non-cancer toxicity of PCBs. It summarizes the important studies regarding PCB non-cancer toxicity, including the critical studies identified for development of the Reference Doses in the Integrated Risk Information System (IRIS), the Agency's consensus database of toxicity information.

The third part describes the non-cancer toxicity information used in the Human Health Risk Assessment for the Hudson River PCBs Site and addresses the Averaging Times and blood PCB levels from occupational studies.

EPA'S NON-CANCER GUIDELINES AND REFERENCE DOSE DEVELOPMENT

EPA's process for evaluating human epidemiological and animal evidence to determine the non-cancer toxicity of chemicals, including PCBs, is set forth in the Agency's guidelines (USEPA, 1986a-b, 1991, 1992, 1993a, 1996a, 1998) and supporting information (USEPA, 1993b; Barnes and Dourson, 1988; Dourson and Stara, 1983). The guidelines cover a variety of health endpoints including developmental toxicity (USEPA, 1991), reproductive toxicity (USEPA, 1996a), neurotoxicity (USEPA, 1998), female reproductive risk (USEPA, 1986a) and male reproductive risk (USEPA, 1986a).

The non-cancer toxicity guidelines were developed within the Agency and published in the *Federal Register* for comment. Periodically, the guidelines have been updated to reflect new scientific understanding regarding toxicity. Prior to being finalized, the guidelines, as updated, are externally peer reviewed by a panel of expert scientists in the various fields associated with non-cancer toxicity including developmental toxicity, neurological toxicity, endocrine effects, who work in universities, environmental groups, industry, labor, and other governmental agencies. EPA responds to comments on the draft guidelines and makes changes based on a review of the comments submitted by these groups or individuals. The guidelines are also submitted for review to EPA's Science Advisory Board, an external scientific review panel.

Reference Dose Development

The quantification of chronic non-cancer health effects is a chronic Reference Dose (RfD), which is defined as an estimate (with uncertainty spanning perhaps an order of magnitude or greater) of an exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime (USEPA, 1989, 1993b).

The procedures used by EPA to develop RfDs are provided in the Background Document on RfD Development available on EPA's IRIS database (USEPA, 1993b; see also www.epa.gov/iris). In general, exposure to a given chemical, depending on the dose, may result in a variety of toxic effects ranging from death to subtle biochemical, physiologic, or pathologic changes. The process for RfD development includes:

- Critical evaluation of the available scientific literature, including human epidemiological and animal toxicity studies. Human data are often useful in qualitatively establishing the presence of an adverse effect in exposed human populations. Human epidemiological studies may be limited in their ability to establish a dose-response relationship between level of exposure and observed health effects, by the degree to which confounders (*e.g.*, other chemicals and lifestyle factors) are controlled.

- For many chemicals, the principal studies are drawn from experiments conducted on non-human mammals, such as the rat, mouse, rabbit, guinea pig, hamster or monkey. These animal studies typically reflect situations in which exposure to the chemical has been carefully controlled and the problems of heterogeneity of the exposed population and concurrent exposures to other chemicals have been minimized.

- EPA uses a weight-of-evidence approach in evaluating the non-cancer toxicity of a chemical, with emphasis on the results from the principal and supportive studies.

- Identification of the critical study(s), critical effect(s) and a dose level (*i.e.*, no observed adverse effect level [NOAEL] or lowest observed adverse effect level [LOAEL]) based on the study(s). The dose level is then divided by uncertainty factors to calculate an RfD. In general, the values used for each uncertainty factor are either 1, 3, or 10 (USEPA, 1993b). The value of 3 is used as a "half" factor and represents the square root (rounded to one significant digit) of the full uncertainty factor of 10, so that two "half" factors yield a full factor of 10 when multiplied together (USEPA, 1994b).

- There are four standard uncertainty factors (ranging from 1 to 10) that can be used when calculating an RfD. These factors account for 1) the variation in sensitivity among members of the human population, 2) extrapolation from animal data to humans, 3) extrapolation from less than chronic NOAELs to chronic NOAELs, and 4) extrapolation from LOAELs to NOAELs. An additional modifying factor (MF), also ranging from 1 to 10, can be applied to the calculation of the RfD. The magnitude of the MF depends upon an assessment of the scientific uncertainties of the study and the database used in deriving the RfD that are not explicitly treated above, such as completeness of the overall database and the number of species tested.

The equation used in the calculation is:

$$RfD = NOAEL / (UF \times MF).$$

NON-CANCER TOXICITY OF PCBs

Based on a weight of the evidence, EPA concluded that PCBs pose a non-cancer health hazard. Non-cancer health effects associated with exposure to PCBs include dermal effects (*e.g.*, chloracne), developmental neurotoxic effects (*e.g.*, learning problems), ocular effects (eye problems), reduced birth weight, and immunotoxic effects (*e.g.*, reduced ability to fight infection). This conclusion is based primarily on animal studies, including monkey studies. Human evidence was also considered.

EPA is not alone in its concern regarding the non-cancer toxicity of PCBs and in using data from studies in monkeys to develop health protective toxicity values. In a joint publication with EPA, ATSDR stated (ATSDR and USEPA, 1996):

> “The findings of elevated PCB levels in human populations, together with findings of developmental deficits and neurologic problems in children whose mothers ate PCB-contaminated fish, have compelling implications. The weight of evidence clearly indicates that populations continue to eat fish containing PCBs

> and that significant health consequences are associated with consumption of large amounts of some fish…Human health studies…indicate that: 1) reproductive function may be disrupted by exposure to PCBs; 2) neurobehavioral and developmental deficits occur in newborns and continue through school-aged children who had in utero exposure to PCBs; 3) other systemic effects (e.g., self-reported liver disease and diabetes, and effects on the thyroid and immune systems) are associated with elevated serum levels of PCBs; and 4) increased cancer risks, e.g., non-Hodgkin's lymphoma, are associated with PCB exposures."

The National Research Council (NAP, 2000) concluded:

> "The Committee's review of recent scientific information supports the conclusion that exposure to PCBs may result in chronic effects (e.g., cancer, immunological, developmental, reproductive, and neurological effects) in humans and/or wildlife. Therefore, the committee considers that the presence of PCBs in sediments may pose long-term public health and ecosystem risks."

Dermal Effects

Several studies document dermal effects in workers exposed to PCBs (Fischbein *et al.*, 1979, 1982, 1985; Maroni *et al.*, 1981a,b; Ouw *et al.*, 1976; Smith *et al.*, 1982). Dermal effects include skin rashes, pigmentation disturbances of skin and nails, thickening of the skin, burning sensations, and chloracne, a severe form of acne that results from exposure to PCBs. Variability in response in more highly exposed individuals suggests that susceptibility varies greatly among individuals (ATSDR, 2000).

Studies in Rhesus monkeys fed diets containing Aroclors for intermediate durations of exposure found effects including facial edema (swelling), acne, folliculitis (inflammation of the hair follicle) and alopecia (hair loss) (Allen and Norback, 1973, 1976; Allen *et al.* 1973, 1974a,b; Barsotti *et al.*, 1976; Becker *et al.*, 1979; Ohnishi and Kohno, 1979; Thomas and Hinsdill, 1978).

Developmental/Neurotoxic Effects

Developmental/neurotoxic effects associated with PCB exposure in animals and identified in human epidemiological studies include reduced birth weight, learning problems, and memory problems.

On September 14 and 15, 1992, EPA convened a Risk Assessment Forum (RAF) Colloquium of expert scientists to evaluate the developmental/ neurotoxic effects of PCB exposure. The Workshop papers discuss the principles and methods for evaluating data from animal and human epidemiological studies (USEPA, 1993a). The report concluded:

> "The sense of the meeting seemed to be that, at least in qualitative terms, the available data are sufficient. In other words, based on an evaluation of the strengths and weaknesses in the data and on the consistency of effects seen in all species tested, including humans, there is sufficient information to indicate that PCBs cause developmental neurotoxicity. Interestingly, the data suggest that

> prenatal exposure to PCBs may be more detrimental than postnatal exposure, even though the level of exposure via breast milk is much greater than that occurring via placental transfer."

Similarly, ATSDRs Toxicological Profile for PCBs (ATSDR, 2000) stated:

> "Studies in humans who consumed high amounts of Great Lakes fish contaminated with environmentally persistent chemicals, including PCBs, have provided evidence that PCBs are important contributors to subtle neurobehavioral alterations observed in newborn children and that some of these alterations persist during childhood…Neurobehavioral alterations have been also observed in rats and monkeys following pre- and/or postnatal exposure to commercial Aroclor mixtures, defined experimental congener mixtures, single PCB congeners, and Great Lakes contaminated fish. In addition, monkeys exposed postnatally to PCB mixtures of congeneric composition and concentration similar to that found in human breast milk showed learning deficits long after exposure had ceased."

Immunotoxic Effects

The immune system is the body's primary defense against infection. Immune effects associated with PCBs include a reduced ability to fight infections.

Several human epidemiological studies evaluated the effects of PCBs on workers and found transient effects on total and differential white blood cell counts (Chase *et al.*, 1982; Lawton *et al.*, 1985; Maroni *et al.*, 1981b; Smith *et al.*, 1982). A number of studies have evaluated the effects of PCBs in specific population groups (*i.e.*, infants, children of mothers who consumed fish, and fish consumers). Immunotoxic effects reported in the Great Lakes populations include increased middle ear and respiratory tract infections in children of exposed mothers (Smith, 1984).

ATSDR (2000) concluded:

> "Findings include increased susceptibility to respiratory tract infections in adults and their children, increased prevalence of ear infections in infants, decreased total serum Immunoglobulin A and Immunoglobulin M antibody levels, and/or changes in T lymphocyte subsets. Overall there is a consistent of effects among the human studies suggesting sensitivity of the immune system to PCBs, particularly in infants expose in utero and/or via beast feeding. However, due to the mixed chemical nature of the exposures and generally insufficient information on exposure-response relationship, the human studies provide only limited evidence of PCB immunotoxicity."

Decreased antibody responses (Immunoglobulin G and Immunoglobulin M) were detected in studies on monkeys (Tryphonas *et al.*, 1989, 1991a,b).

Ocular Effects

Occupational studies have shown eye irritation, tearing and burning among workers exposed to airborne PCBs (Emmett *et al.*, 1988, Ouw *et al.*, 1976; and Smith *et al.*, 1982). Fischbein *et al.* (1979, 1985) found that some capacitor workers had edema of the upper eyelid, congestion of the conjunctiva, eye discharge and enlargement of the Meibomian glands following exposures to various Aroclors in a range of concentrations.

The monkey studies noted ocular exudate (discharge) and inflamed and enlarged Meibomian glands (Arnold *et al.*, 1993a, b).

Reference Doses for Aroclors 1016 and 1254

Using the process summarized above, EPA evaluated both human epidemiological evidence and animal toxicity studies in developing quantitative RfDs for Aroclors 1016 and 1254 (USEPA, 1999a,b).

EPA determined that the human data available for risk assessments of Aroclor 1016 and Aroclor 1254 are useful only in a qualitative manner, noting, "Studies of the general population exposed to PCBs by consumption of contaminated food, particularly neurobehavioral evaluations of infants exposed in utero and/or through lactation, have been reported, but the original PCB mixtures, exposure levels and other details of exposure are not known (Kreiss *et al.*, 1981; Humphrey, 1983; Fein *et al.*, 1984a,b; Jacobson *et al.*, 1984a,b, 1985, 1990a,b; Rogan *et al.*, 1986; Gladen *et al.*, 1988). Most of the information on health effects of PCB mixtures in humans is available from studies of occupational exposure. Some of these studies examined workers who had some occupational exposure, but in these studies concurrent exposure to other Aroclor mixtures nearly always occurred, exposure involved dermal as well as inhalation routes (the relative contribution by each route was not known), and monitoring data were lacking or inadequate (Fischbein *et al.*, 1979, 1982, 1985; Fischbein, 1985; Warshaw *et al.*, 1979; Smith *et al.*, 1982; Lawton *et al.*, 1985)."

A brief summary of EPA's development of the RfDs is provided below.

Aroclor 1016

EPA identified the monkey reproductive studies by Barsotti and van Miller (1984) and neurological studies by Levin *et al.* (1988), and Schantz *et al.* (1989, 1991) as critical studies. The critical effect identified was reduced birth weights. A NOAEL of 0.25 ppm in feed (or 0.007 mg/kg-day) was identified. The IRIS chemical file for Aroclor 1016 summarizes the critical study and effect and describes EPA's evaluation of a number of other studies that provide supporting information for the selection of these studies (USEPA, 1999a; see also www.epa.gov/iris).

As part of EPA's peer review process, on May 24 and 25, 1994, EPA convened an RAF Workshop to assess whether the Reference Dose (RfD) for Aroclor 1016 (USEPA, 1994a) represents a full consideration of the available scientific data and whether that analysis is clearly articulated in the RfD entry on IRIS. The results from this Workshop were used in finalizing the

RfD for Aroclor 1016 (USEPA, 1999a) currently listed on IRIS. The IRIS chemical files for both Aroclor 1016 (USEPA, 1999a) and Aroclor 1254 (USEPA, 1999b) represent the consensus of the Reference Dose/Reference Concentration Workgroup, responsible for reaching consensus on non-cancer toxicity values, which was in existence when the files were completed.

USEPA's applied uncertainty/modifying factors totaling 100 (3 x 3 x 3 x 3 and rounded) to be protective of sensitive human populations that may be exposed *i.e.*, the NOAEL of 0.007 mg/kg-day was divided by a factor of 100 to yield a RfD of 0.00007 mg/kg-day. A summary of the UFs and their basis is provided below:

- A factor of 3 is applied to account for sensitive individuals. The results of these studies, as well as data for human exposure to PCBs, indicate that infants exposed transplacentally represent a sensitive subpopulation.

- A factor of 3 is applied for extrapolation from Rhesus monkeys to human. A full 10-fold factor for interspecies extrapolation is not considered necessary because of similarities in toxic responses and metabolism of PCBs between monkeys and humans and the general physiologic similarity between these species. In addition, the Rhesus monkey data are predictive of other changes noted in human studies such as chloracne, hepatic changes, and effects on reproductive function.

- A factor of 3 is applied because the study duration was considered as somewhat greater than subchronic, but less than chronic; a partial factor of 3 is used to account for extrapolation from a subchronic exposure to a chronic RfD.

- A factor of 3 is applied because of limitations in the database. Despite the extensive amount of animal laboratory data and human epidemiologic information regarding PCBs, the issue of male reproductive effects is not directly addressed and two-generation reproductive studies are not available.

Aroclor 1254

EPA identified the monkey studies by Arnold *et al.* (1993a,b), Tryphonas *et al.* (1989, 1991a,b) as the critical studies. The critical effects were ocular exudate, inflammation and prominent Meibomian glands in the eye, distorted growth of finger- and toenails, and decreased antibody responses (Immunoglobulin G and Immunoglobulin M) based on responses to sheep erythrocytes (USEPA, 1999b). A NOAEL could not be identified so a LOAEL of 0.005 mg/kg-day was identified.

EPA applied uncertainty factors totaling 300 (*i.e.*, 10 x 3 x 3 x 3 and rounded) to the LOAEL of 0.005 mg/kg and calculated an RfD of 0.00002 mg/kg-day. The basis for the UFs are provided below:

- A factor of 10 is applied to account for sensitive individuals such as children, elderly, and others.

- A factor of 3 is applied to extrapolation from Rhesus monkeys to humans. A full 10-fold factor for interspecies extrapolation is not considered necessary because of similarities in toxic responses and metabolism of PCBs between monkeys and humans and the general physiologic similarity between these species. Tilson *et al.* (1990) reported that humans appear to be more sensitive than monkeys or rodents. EPA noted that the differences in species sensitivity may be related to variations in the sensitivity of the testing paradigms used in different species, and/or differences in the toxicity of the various commercial mixtures, or environmental exposures used in various studies" (USEPA, 1993a). Based on similarity in types of effects but dissimilarity in effective doses and NOAELs across test species, EPA concluded that monkeys are not less sensitive than humans with respect to developmental/neurotoxic effects of PCBs (USEPA, 1993a).

- A factor of 3 is applied for the use of a minimal LOAEL since the changes in the periocular tissues and nail bed seen at the 0.05 mg/kg-day are not considered to be of marked severity. The duration of the critical study continued for approximately 25% of the lifespan of Rhesus monkeys, so a factor of 3 is appropriate for extrapolation from subchronic exposure to a chronic RfD.

- A factor of 3 is applied based on the immunologic and clinical changes that were observed but did not appear to be dependent upon duration, which further justifies using a factor of 3 rather than 10 for extrapolation from subchronic to chronic, lifetime exposure.

The Agency for Toxic Substances and Disease Registry issued an updated Toxicological Profile for Polychlorinated Biphenyls following external peer review (ATSDR, 2000). ATSDR (2000) includes Minimal Risk Levels (MRL). The MRL is defined as "an estimate of the daily human exposure to a hazardous substance that is likely to be without appreciable risk of adverse non-cancer health effects over a specified duration of exposure" (ATSDR, 2000). The chronic MRL is developed to be protective over a one-year period or more, and is similar to EPA's RfD, which is developed to be protective over a lifetime. The intermediate MRL is developed to be protective from 15 to 364 days.

ATSDR's chronic MRL is 0.00002 mg/kg/day, based on the study by Tryphonas *et al.* (1989, 1991a,b), which also was used as the critical study for EPA's RfD for Aroclor 1254. The intermediate oral MRL level developed by ATSDR based on monkey studies by Rice (1997, 1998, 1999b) and Rice and Hayward (1997 and 1998) is 0.00003 mg/kg-day, which is slightly higher than the MRL for chronic exposure (ATSDR, 2000). Similar to EPA, ATSDR used a factor of 3 for extrapolating from the monkey studies to humans in developing its MRLs.

HUDSON RIVER PCBs SITE

Consistent with EPA guidance and CERCLA and NCP policies, the PCB non-cancer toxicity information and RfDs that are in IRIS were used in the HHRA (USEPA, 2000a,b). The use of IRIS data in the evaluation of chemical toxicity at Superfund sites addresses EPA's goal of using consistent toxicity information at Superfund sites across the country.

EPA submitted the HHRA (USEPA, 1999c) for external peer review. EPA specifically charged the peer reviewers to evaluate whether use of the IRIS values was appropriate. The peer reviewers for the HHRA agreed with USEPA's use of non-cancer toxicity information from IRIS.

In the HHRA, EPA applied an Averaging Time that is equivalent to the Exposure Duration multiplied by 365 days/year, consistent with USEPA (1989). The peer reviewers of the HHRA agreed with EPA's selection of Averaging Times (USEPA, 2000b) and recommended that EPA evaluate the effects of PCBs to pregnant and nursing women using a shorter exposure duration. The non-cancer hazards to the fetus and infant were addressed qualitatively in the HHRA (USEPA, 2000a), due to the lack of an approved methodology for modeling the effects of PCBs on the fetus and calculating the PCB levels in breast milk based on the mother's body burden.

The HHRA peer reviewers also recommended that EPA also provide a discussion of the more recently published studies on non-cancer endpoints to determine what effect these studies might have on risk estimates. In response, in the Revised HHRA, EPA summarized a number of newly published human epidemiological studies on the non-cancer effects of PCBs (including updates of the neuro-developmental studies in cohorts of children and adults) identified in the IRIS files for Aroclors 1016 and 1254 (USEPA, 2000a). Based on an evaluation of this data, EPA concluded that the toxicity values in IRIS are still appropriate for the HHRA (USEPA, 2000b).

Since 1994, a number of new animal studies and human epidemiological studies and updated studies of the cohorts originally described in 1993-1994 have been published (*e.g.,* Rice 1997, 1998, 1999b, Rice and Hayward, 1997, 1998; Schantz, 1996, Schantz *et al.*, 2001; Jacobson and Jacobson, 1996a,b; 1997; Lanting *et al.*, 1998a,b,c; Patandin *et al.*, 1998, 1999a,b; Koopman-Esseboom *et al.*, 1996; Weisglas-Kuperus *et al.*, 1995, 2000; and Fitzgerald *et al.*, 1995, 1996, 1998, 1999). The studies have been published in a variety of peer-reviewed journals (*e.g.,* Neurotoxicology, New England Journal of Medicine, Science, Lancet, Environmental Health Perspective, Journal of Pediatrics), including a number of public health and epidemiological journals (American Journal of Public Health, Annals of Epidemiology, Epidemiology, American Journal of Epidemiology). In general, as the studies progressed through time, the list of confounders were expanded or reduced as appropriate based on *a priori* information regarding previous studies, consistent with epidemiological practices. A summary of these studies is provided the HHRA (USEPA, 2000a).

Some of these studies found reductions in IQ points (*i.e.*, 3 to 5 points across the various studies) based on prospective studies in children exposed to various sources of PCBs, including fish consumption. At a population level, as well as at an individual level, the potential impacts of the loss of IQ points may be significant, especially among children at the low end of the IQ distribution.

As part of EPA's reassessment of PCB non-cancer toxicity, EPA will critically evaluate this new information (*e.g.*, from human epidemiological studies, animal studies, and mechanistic data) to determine the critical study, critical effect, and appropriate Uncertainty/Modifying Factors necessary to develop a new RfD or reaffirm the current RfD. Documents summarizing the non-cancer toxicology of PCBs will be reviewed within the Agency, and submitted for external peer review. Based on the results of this review, an IRIS chemical file will be developed and undergo internal EPA consensus IRIS review, and will be made available on the IRIS database at the completion of this process.

Effects of PCB Exposure on Blood Levels

EPA followed risk assessment guidance and procedures (see National Contingency Plan; see also USEPA, 1989, 1993c, 1995, 1997) to quantify non-cancer health hazards to individuals exposed to PCBs at the Hudson River PCBs Site in the HHRA (USEPA, 2000a). The approach used in the HHRA is different than measurement of PCB levels in blood of former capacitor workers. First, the HHRA evaluates current and future exposures, while the blood PCB level data integrates past exposure.

Second, capacitor workers were primarily exposed through dermal contact and inhalation of PCBs, whereas anglers, which had the highest cancer risks evaluated in the HHRA, would be exposed to PCBs through ingestion of contaminated fish caught in the Hudson River.

Third, in the HHRA EPA evaluated non-cancer health hazards to the RME individual, whereas for capacitor workers the level of exposure is generally not known. Fourth, the PCB congener profile in the capacitor plant is likely to be different from the congener profile of PCBs that are bioaccumulated in the fish. Lastly, EPA is concerned with potential exposures to the human population including sensitive groups that may include the fetus exposed from mothers who consumed PCB-contaminated fish, infants exposed to PCBs through breast milk, young children, adolescents, adults, and individuals with pre-existing medical conditions (USEPA, 2000a); many of these sensitive groups may not be represented in a healthy worker population. EPA has stated that (USEPA, 1996b):

> "People with decreased liver function, including inefficient glucuronidative mechanism in infants, can have less capacity to metabolize and eliminate PCBs (Calabrese and Sorenson, 1977). Additionally, approximately 5% of nursing infants receive a steroid in human milk that inhibits the activity of glucuronyl transferase, further reducing PCB metabolism and elimination (Calabrese and Sorenson, 1977)."

A study of people exposed through eating contaminated fish (Hovinga *et al.*, 1992) suggests that the PCB mixtures in fish can be more persistent than those to which the workers were exposed. From 1977 to 1985, mean PCB serum levels (quantified using Aroclor 1260 as a reference standard) from 111 Great Lakes fish eaters decreased only slightly from 20.5 to 19.0 ppb (see USEPA, 1996b). Half-life estimates for a mixture can underestimate its long-term persistence (USEPA, 1996b), especially from consumption of fish where changes in PCB blood levels may take longer (Hovinga *et al.*, 1992). This indicates that the rate of decline in the fish eating populations will be slower than that for the workers.

ATSDR's Toxicological Profile (ATSDR, 2000) states that there are no known treatment methods for reducing body burdens of PCBs, concluding that limiting or preventing further exposure appears to be the most practical method for reducing PCB body burdens.

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Relationship Between Tri + and Total PCBs
424694

WHITE PAPER – RELATIONSHIP BETWEEN TRI+ AND TOTAL PCBs

(ID 424694)

ABSTRACT

The historical database of water column, sediment, and fish data generally represent the Tri+ fraction of PCB contamination. In order to estimate the total mass of PCB in these media, a correction factor, the Total PCB to Tri+ ratio, is required. To estimate this ratio, more recent data on sediments, water and fish were examined where all, or nearly all, PCB congeners were represented. This white paper discusses the basis for estimating these ratios in various regions of the river. Data obtained by EPA and GE form the basis for these revised ratios. On the basis of the revised ratios, EPA's estimates of sediment Total PCB inventory, the Total PCB mass remediated and the fraction of the sediment Total PCB inventory remediated will all require adjustment upward. Conversely, Total PCB transport past Waterford actually decreases compared to the previous estimate. The Total PCB release due to dredging increases as well but still remains a minor release relative to the "naturally" occurring sediment releases. However, because PCB contamination in fish tissue is shown to contain almost exclusively Tri+ (98 percent or higher), EPA's forecasts and modeling analyses, which are based on Tri+, require no revision.

INTRODUCTION

EPA has examined the PCB contamination of the Hudson River using two basic measures of concentration, Total PCBs and Tri+ PCBs. Total PCBs represents the entire sum of reported PCB congeners present in a sample. While not including all PCB congeners, this measurement generally represents the major congeners present in the sample and typically includes congeners from all homologue groups. Tri+ PCBs (generally referred to simply as "Tri+") represents only a portion of the PCBs present in the sample. Specifically, this measurement represents the sum of the reported trichloro- to decachloro-homologue groups. As discussed below, many of the historical measurements can be readily transformed into an estimate of the Tri+ concentration.

Both Total PCB and Tri+ PCB values have been used in the assessment of Hudson River PCB contamination. The choice of PCB measure has depended, to a large extent, on the available data. While estimates of Total PCBs are generally preferable, Tri+ represents a measure of PCB contamination that is more accurately represented by historical measurements. That is, historical analytical techniques frequently did not or could not represent the entire spectrum of PCB contamination present in the sample.

The primary means of PCB analysis prior to 1990 involved gas chromatography using a packed column[1] and analytical standards based on commercial Aroclors. These techniques were largely

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[1] A packed column typically consisted of a narrow tube packed with a sorbant material to permit the separation of PCBs from other compounds in the samples. This technology permitted the separation of PCBs into groups of congeners for later identification and quantitation. Current technology uses a capillary tube coated with a sorbant material. Current technology is able to separate PCBs into individual congeners. Application of a capillary tube along with analytical standards for the individual congeners has permitted a much more accurate and complete

incapable of detecting and quantifying many of the monochloro and dichloro PCB congeners present in the environment. In most studies of the Hudson, historical analytical techniques focused on the main Aroclors thought to be discharged by GE; specifically, Aroclors 1242 and 1254. As a result, many analytical programs did not even attempt to examine the less-chlorinated congeners, based in part on the incorrect assumption that a mixture of PCBs in the environment would remain similar to the original mixture discharged to the environment. As extensively documented in the EPA's Phase 2 reports, the original mixtures of Aroclors 1242, 1016, and 1254 have been modified to varying degrees by processes such as dissolution, degradation, dechlorination, bioaccumulation, and partitioning. Thus, in many cases, the mixtures in the Hudson do not closely resemble the GE source from which they were derived. (As shown in the DEIR [USEPA, 1997] and the Ecological Risk Assessment for the Lower Hudson [USEPA, 1999b], sediment mixtures are readily traced to the GE sources when geochemical and dechlorination processes are considered. PCB patterns in the biota were shown to correlate with the GE contamination in the sediments.) Both the lack of analytical technology and the lack of a thorough understanding of PCB geochemistry contributed to the limitations of the historical data.

Early on in the reassessment, EPA recognized the need for the data obtained in Phase 2 to be understood in the context of historical data. In order to accomplish this, it was necessary to convert both the historical and current data sets to a consistent basis so as to permit accurate and meaningful comparisons. As noted above, many of the historical data sets did not represent the entire spectrum of PCB congeners present in the samples. Based on a review of the data available, including data from NYSDEC, USGS, and GE, among others, it became clear that most of the historical data represented analytical techniques that could be translated into a measure of the trichloro and higher homologue fraction. This portion of the PCB mass in a sample, that is, the sum of trichloro to decachloro homologues (or Tri+), became the basis on which historical data could be compared to more recent analyses. Because the more sophisticated techniques employed by EPA and others after 1990 represented the entire (or almost the entire) spectrum of PCBs, the more recent data could be readily converted to a Tri+ basis. Thus, many of the data interpretations performed in Phase 2 were based on the Tri+ parameter. Discussions of the development and application of the Tri+ conversions can be found in the DEIR (USEPA, 1997), the Baseline Modeling Report Responsiveness Summary (USEPA, 2000b) and Butcher *et al.*, 1997.

In particular, the sophisticated computer models developed by EPA required a consistent measure of PCBs across sediments, water, and biota in order to simulate PCB contamination in the Hudson over a long period of time. The extended period of simulation (*i.e.*, the long period of data) provided additional insights into the nature of PCB transport and strengthened the calibration of the model. The Tri+ parameter represents essentially the only metric available that is consistent across all three media and can be derived from nearly all environmental data sets available for the Hudson.

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quantitation of PCBs in environmental samples. This approach was used for the entire set of Phase 2 samples collected by EPA.

Among the more important data sets that required consideration of the Total PCB to Tri+ ratio were the following:

- NYSDEC 1976-1978 Sediment Survey of the Upper Hudson.
- NYSDEC 1984 Sediment Survey of the Thompson Island Pool.
- USGS 1977-1999 Water Samples from the Upper Hudson.
- NYSDEC 1977-2000 Fish Collection from the Upper Hudson.

These data sets, along with the conversion of the EPA Phase 2 and GE data sets, formed the basis for the calibration of the EPA's HUDTOX and FISHRAND models.

As discussed later in this white paper, as well as in the ecological risk assessment reports (USEPA, 1999a, 1999b, 2000c), PCB contamination in fish is almost exclusively comprised of congeners represented by the Tri+ summation. Thus, for the purposes of forecasting future exposures to humans and wildlife via fish, it is only necessary to model the Tri+ parameter. The Tri+ parameter is the main output variable for the EPA's models.

Nonetheless, Total PCBs (*i.e.*, the sum of the entire spectrum of congeners) remains EPA's primary focus for the purposes of remediation, since PCB toxicity is not limited to the Tri+ fraction. Exposure to PCB-contaminated sediments and water results in exposure to the entire spectrum of PCBs, not just Tri+. Additionally, Total PCBs, and not Tri+, form the basis for assessing sediment treatment, shipping, and landfilling. As a result, EPA's analyses have considered both measures of PCB. Tri+ was principally used for modeling, while Total PCBs were considered when dealing with engineering considerations.

Since much of the historical data can only provide an estimate of the Tri+ concentration and since EPA needed estimates of Total PCB in the environmental media, a basis was needed for each medium to translate the measured or forecast Tri+ concentration into an estimate of the Total PCB concentration. The basis for these conversions is described in the next section of this white paper.

CONVERSION OF TRI+ TO TOTAL PCB

The ratio of Total PCB to Tri+ is well defined for Aroclor 1242, based on the analysis of an available standard. Once released into the environment, processes such as dechlorination, partitioning, and bioaccumulation will modify the ratio. The ratio of Total PCB to Tri+ in the analytical standard for Aroclor 1242 is approximately 1.17. Depending upon the media, various factors can work to change this ratio substantially, yielding a different ratio for each media type. In fact, the ratio varies across the Upper Hudson as well, since the factors affecting the ratio are not equal everywhere.

Water

The need for a standard set of factors to convert Tri+ to Total PCB in the water column was recognized early in the model simulation process. Although EPA and GE data are available to quantitate Total PCB and Tri+ through the 1990s in the Upper Hudson, the models employ the Tri+ parameter only, as discussed above; thus, a conversion factor is needed to estimate future

Total PCB loads and concentrations in the Upper Hudson on the basis of the Tri+ forecasts. An examination of the Total PCB to Tri+ ratio at the various Phase 2 monitoring stations showed that the ratio varied with distance downstream from the GE facilities. Thus, station-specific ratios were developed for each of the major water-column monitoring stations. These ratios were applied in the FS in order to forecast Total PCB loads in the Upper Hudson

Ratios for Total PCB to Tri+ were required at two main stations, Thompson Island Dam (TI Dam) and Waterford. The ratio was examined at other stations as well, however, to provide additional support for the analysis. The Phase 2 data set represents the only data set available that can provide an estimate of the Total PCB to Tri+ ratio throughout the Upper Hudson (*i.e.*, from Bakers Falls to Troy). Although the GE data cover a greater period of time, that data set is more limited spatially, extending only to Schuylerville[2]. The GE data are also limited to some degree by a significantly higher detection limit. As a result, the EPA data are used for the ratio estimate of 1.4 at the Waterford station, while the GE values for TI Dam (TID West and PRW2) and Schuylerville were all considered in deriving the value for the TI Dam (2.2). The three stations are considered together in estimating the ratio for the TI Dam, due to the significant variability in water column conditions in this area. The EPA ratios, as well as those developed from the GE data, are given on a station-specific basis in Table 424694-1.

Recognizing the limited number of samples, the Phase 2 results for several stations have been combined to yield the estimated ratios. As shown in Table 424694-1, the GE data yield ratios similar to those developed from the EPA data. Notably, the ratio at TI Dam and Schuylerville obtained from the GE data is about 10 percent higher than that obtained from the Phase 2 data.

Note that the ratio for the GE Rogers Island station is based on the entire period of monitoring. This was appropriate, given the general consistency of the PCB source to this location over time. The ratio at the TI Dam was derived from data limited to the post-1996 period, when the upstream source was better controlled. Prior to this period, the TI Dam sample was subject to fairly large variability, due to the variation in contributions from above Rogers Island relative to those of the sediments of the pool itself. The Phase 2 sampling program was able to avoid some of this variability due to the sampling techniques employed.

As is evident in Table 424694-1, the biggest increase in ratio occurs across the Thompson Island Pool (TI Pool), as the release of PCBs from the sediments greatly increases the total PCB load and changes the proportions among the PCB congeners in the water column. Downstream of Schuylerville, the value declines, possibly as a result of interactions between the water column and the less-dechlorinated PCBs found in the sediments downstream. Preferential losses of the less-chlorinated congeners due to degradation or gas exchange may occur as well (USEPA, 2000d).

The Total PCB to Tri+ ratio developed for the Waterford station was used to estimate the total mass of PCBs delivered to the Lower Hudson under each of the various model runs. Tri+ is directly forecast by the model and is unaffected by the analysis presented here. Conversely, the

[2] Although GE did collect samples downstream of Schuylerville, these samples were collected during the Allen Mill event and ensuing large PCB releases (Sept 1991 to 1992). These samples represented an unusual condition in the Upper Hudson and thus could not be used to characterize the Total PCB to Tri+ ratio for the predominantly sediment-derived releases post-1993.

Total PCB delivered to the Lower Hudson is estimated as the product of the Tri+ load at Waterford and the Total PCB to Tri+ ratio. In the original estimates given in the FS, the Total PCB to Tri+ ratio at Schuylerville of 2.2 (developed from the GE data collected at Schuylerville) was used as the ratio at Waterford. Thus the application of the value estimated for Waterford (1.4) yields load estimates of Total PCB that are one-third lower than originally calculated. Although both the Tri+ and Total PCB loads are estimated, it is important to note that the values estimated for Total PCBs include a higher degree of uncertainty due to this additional step in the estimation process.

Sediments

The estimation of a Total PCB to Tri+ ratio for the sediments of the Upper Hudson was made difficult by the heterogeneous nature of sediment contamination in the Upper Hudson and the lack of a Total PCB data set that could be considered spatially representative of the entire area. The heterogeneity was due to several factors including variable rates of deposition and scour, as well as dechlorination of PCB in the sediments. Dechlorination directly increases the ratio since it produces monochloro and dichloro congeners by converting heavier congeners to lighter ones. As noted in the DEIR (USEPA, 1997), the degree of dechlorination in the sediments of the Upper Hudson is dependent on the concentration of PCBs. In general, the most contaminated sediments typically exhibit the greatest degree of dechlorination. In addition, extensive dechlorination in the Hudson River appears largely limited to sediments above an initial concentration of 30 ppm Total PCB. Thus, sediments with low levels of contamination are expected to have relatively low ratios of Total PCB to Tri+, as compared to highly contaminated sediments[3].

The need for an estimate of the Total PCB to Tri+ ratio for each river section of the Upper Hudson was dependent on the available data and the engineering data requirements. Thus the ratio was estimated for each river section for each sediment subclass on an as-needed basis. The data sets available to provide this information were different in each river section. The derivation of the necessary information for each river section is described below.

River Section 1 (TI Pool)

During the preparation of the FS, it was recognized that no study existed in River Section 1 that could provide a complete description of the Total PCB inventory in the sediments. A ratio was needed to describe Total PCBs for the entire TI Pool. The 1984 data set represented the best coverage for an estimate of the Tri+ concentrations and inventory but was not well suited for the estimate of Total PCB. In the FS, as well as prior EPA reports, the estimate of Total PCB mass for the TI Pool was based on the sum of Aroclors as originally reported by Brown *et al.*, 1988. This approach was considered a low-end estimate since it was recognized that the 1984 results did not capture the monochloro and dichloro fractions well. Upon subsequent review of the most recent GE data (1999 coring data) in conjunction with the existing set of Phase 2 low-resolution cores, Phase 2 high-resolution cores, the 1991 GE composite samples, and the 1998 GE

[3] Evidence suggests that most dechlorination in the Upper Hudson River occurs rapidly after sediment deposition, and subsequent dechlorination is limited. As discussed at length in the DEIR (USEPA, 1997), it is unlikely that historically deposited sediments will undergo further, substantial dechlorination.

composite samples, it was decided that a sufficient amount of data were available to support an independent estimate of the Total PCB to Tri+ ratio for the TI Pool.

To best estimate the Total PCBs to Tri+ ratio, it is important to recognize that this ratio varies almost directly with the degree of dechlorination. This is because the mono- and di-homologue fractions increase and the Tri+ fractions decrease in response to the dechlorination process. Thus, highly dechlorinated mixtures have a high ratio, and vice versa. As extensively documented in the DEIR (USEPA, 1997), the extent of dechlorination in the sediment varies logarithmically with the concentration in the sediment. Thus, the Total PCB to Tri+ ratio should also vary with concentration.

This correlation was best demonstrated by the Phase 2 high-resolution cores (as shown in the first diagram of Figure 424694-1). These core samples represent relatively thin core segments (2- to 4-cm thick), a scale at which sediment concentrations are expected to be relatively homogeneous within the sediment. Thus, the relationship between sediment concentration and dechlorination should be clearest for these samples. The Total PCBs to Tri+ ratio clearly increases with concentration. A weighted curve has been fit to the data to suggest how the mean ratio varies with concentration. The initial value of 1.25, which applies below 10 mg/kg of Tri+, is quite close to the theoretical starting value of 1.17 for Aroclor 1242. This is consistent with the findings of the DEIR, which stated that little dechlorination occurs at low concentrations. As sediment concentrations rise above 10 mg/kg Tri+, the Total PCB to Tri+ ratio increases substantially, reaching a value around 4 above 100 mg/kg Tri+. Clearly, the Total PCB to Tri+ ratio is dependent on the sediment concentration.

This can also be seen in the GE coring data from 1998 and 1999. These data also represent relatively thin core segments and would be expected to yield a similar relationship between the sediment concentration and the Total PCB to Tri+ ratio. This is illustrated in the second diagram of Figure 424694-1. Again a weighted curve has been fit to the data to track the mean ratio. A few outliers were excluded from the weighted curved determination, based on a statistical Mahalanobis analysis (SAS, 1997).

This diagram also shows an initial low value for the Total PCB to Tri+ ratio, rising to a much higher value at higher concentrations. The absolute value of the mean ratio is higher than that obtained from the high-resolution cores. This is attributed to the differences in analytical technique between the two data sets. Part of the difference may lie in the quantitation techniques used by GE. Essentially, the mono and di congeners are analyzed on a congener-specific basis, whereas the Tri+ fraction is tied to an Aroclor standard (Hydroqual, 1997). Thus, the absolute value of the Total PCB to Tri+ ratio for the GE data is strongly dependent on the internal calibration of the two analytical bases. Nonetheless, both the Phase 2 and the GE data sets suggest about a threefold increase in the ratio at high concentrations. In both sets, individual samples can attain ratios nearly double the mean high-end value.

Both data sets demonstrate a strong relationship between Tri+ concentration and the Total PCB to Tri+ ratio. However, both data sets represent small sampling intervals (less than or equal to five cm), much shallower than the 1984 NYSDEC coring data set (nominally 30 cm). As documented in the LRC (USEPA, 1998), the process of collecting thick segments serves to confound ratio-to-concentration relationships, since layers of many different properties are blended into a single analysis. This is evident in the third diagram of Figure 424694-1, where the low-resolution core results are presented. As was seen for the molar dechlorination product ratio (USEPA, 1998), the relationship for the Total PCB to Tri+ ratio to Tri+ concentration is much noisier than that for the high-resolution cores.

However, these samples are closest in collection technique to the 1984 survey, and so are best suited to describe the Total PCB to Tri+ ratio for the 1984 data. Additionally, the conversion of the 1984 data set to a Tri+ basis is founded on the EPA's congener-specific analytical technique. Thus, for both sampling technique and analytical approach, the curve developed for the low-resolution cores is most applicable to the 1984 data set.

A review of the third diagram would not, of itself, suggest a strong relationship. However, the strength of the Total PCB to Tri+ ratio relationship is already well established by the high-resolution core and GE core data sets. As a result, a relationship was developed that parallels the relationship seen in the high-resolution cores (Figure 424694-2).

The upper diagram of the figure shows the weighted mean curves from each of the data sets. Notably, the low-resolution core curve is similar to the GE curve at low concentrations (less than 10 mg/kg), and converges to approximately the same value as the high-resolution cores at high concentrations (greater than 100 mg/kg). The weighted curve developed for the low-resolution curve was then approximated as three segments, as follows:

Tri+ Concentration	**Total PCB to Tri+ Ratio**
Less than 10 mg/kg	2.2
Between 10 and 100 mg/kg	2.2 + log (Tri+Conc / 10)
Greater than 100 mg/kg	3.8

This approximation is shown in the second diagram of Figure 424694-2.

As explained in White Paper – Sediment PCB Inventory Estimates, this curve was applied to the length-weighted average Tri+ concentrations of the TI Pool to estimate the Total PCB concentrations in the sediments. Each individual 1984 core or grab was corrected to estimate the local Total PCB concentration. These results were then integrated over the area of the pool and the volume of sediment to be removed. Based on this integration, the mass-integrated Total PCB to Tri+ ratio for the TI Pool was estimated at 3.1. For the sediments to be remediated under the selected remedy, the ratio was estimated at 3.4. These ratios are summarized in Table 424694-2.

The relationship between Total PCB and Tri+ is such that the most contaminated sediments have the highest ratios. These are also the sediments that are preferentially targeted for removal under the selected remedy. As a result, the estimates for Total PCBs in the areas slated for removal under the selected remedy increased more than the areas to be left untouched. This modification has the effect of increasing the estimate of the fraction of Total PCBs to be removed under the selected remedy. As discussed in Whiter Paper – Sediment PCB Inventory Estimates, the estimate for the *in situ* Total PCB inventory of the TI Pool increased 3-fold to 45 metric tons. Of this inventory, approximately 80 percent, or 36 metric tons, will be removed. The result of the increase in the Total PCB inventory estimate for the TI Pool serves to increase the overall importance of the TI Pool to the PCB inventories of the Upper Hudson. Correspondingly, the mass of PCBs removed for this river section has increased as well. Ultimately, since both estimates increase, this leads to an overall increase in the fraction of PCB removed from the Upper Hudson for the selected remedy.

River Sections 2 and 3 (TI Dam to Waterford)

In these river sections, there is no single synoptic data set of sufficient quality and recent age that can be used to estimate Tri+ or Total PCB concentrations or inventories on a section-wide basis. However, as the data available contain estimates for both Total PCB and for Tri+, there is no need to independently estimate their ratio, as was done for River Section 1. As discussed in White Paper – Sediment PCB Inventory Estimates, the low-resolution coring data set provided a basis for assessing both Total PCB and Tri+ in the areas to be remediated, effectively equivalent to the cohesive sediment areas. For noncohesive sediments, the 1991 GE composite samples were used. The application of these data is discussed in detail in White Paper – Sediment PCB Inventory Estimates. This discussion will focus on the Total PCB to Tri+ ratio for these areas.

Because the relatively recent low-resolution cores and 1991 GE composites samples could be used to estimate inventories and concentrations and because these samples provide direct estimates of Tri+ and Total PCBs, the data could be used to estimate the Tri+ and Total PCB values for the river sections independently. However, it is useful to compute the Total PCB to Tri+ ratio simply for comparative purposes, providing further support for the approach used in River Section 1. These results are summarized in Table 424694-2.

The ratios given in the table agree well with the values found for the TI Pool. Specifically, the values obtained for the sediments targeted for remediation in River Sections 2 and 3 (3.4 and 2.7, respectively) compare well with the value obtained for the sediments targeted for remediation in the TI Pool (3.4). These data support the derivation of a ratio for the TI Pool, as was described above.

Fish

Results for fish samples collected by EPA were used to examine the Total PCB to Tri+ ratio in Hudson River fish. This data set, like the other Phase 2 results, is able to provide an independent estimate of the Total PCB and Tri+ concentration for each sample. The ratio is, therefore, not needed for calculations, but rather to support the modeling and risk-calculation assumptions. In the presentations of the BERA and RBMR, the observation that PCBs in fish are nearly entirely represented by the Tri+ summation has been stated many times but not quantitated. This examination will briefly summarize the results.

The 207 fish samples collected in Phase 2 and analyzed via EPA's congener-specific methodology form the basis for this calculation. Sample replicates of the same species from the same station were averaged prior to inclusion in the region-wide calculation; *e.g.*, five white perch samples from the station were averaged together prior to inclusion in the calculation. In this manner, stations and species were more evenly represented. After this summation, 60 unique species/station samples were available. These were arithmetically averaged together on a regional basis to determine the mean Total PCB to Tri+ ratio. These results are presented in Table 424694-3.

Evident in the table is the very high mass fraction of Tri+, regardless of region. The average value ranges from 98 to nearly 100 percent Tri+. These values translate to a Total PCB to Tri+ ratio of 1.02 to 1, respectively. From these results, it is clear that the assumption used in the RI and FS (*i.e.*, Total PCB in fish is equal to Tri+) is a valid assumption and introduces little additional uncertainty. This has important ramifications for the use of the historical fish data from NYSDEC. As noted in the RBMR (USEPA, 2000a), the NYSDEC fish data can be converted to a Tri+ basis by a relatively small correction factor. The analysis above demonstrates that no further correction is needed to use these data as an estimate of Total PCB in fish tissue.

IMPLICATIONS AND CONCLUSIONS

A focus on the Tri+ fraction of PCBs permitted the use of historic data in the detailed study of PCB contamination in the Hudson. The results and conclusions of the FS are largely based on the Tri+ results due, in large part, to the fact that fish body burdens are almost exclusively Tri+. Thus, future improvements in fish tissue concentrations are inherently tied to declines in the Tri+ PCB concentrations. EPA's selected remedy is specifically designed to reduce Tri+ concentrations.

Improved estimates of the Total PCB to Tri+ ratio were made for the sediments of the TI Pool and the water column at Waterford. Other calculations presented provide further support for these revisions, as well as supporting the assumptions used in the reassessment. The improved estimates for the Total PCB to Tri+ ratio presented here affect only the estimates of Total PCBs. The revised estimates for the ratio specifically affect the following:

- The estimate of Total PCB inventory for the TI Pool will be substantially increased compared to the estimate in the FS. This will increase the estimates of Total PCB removed as well as the increase the fraction of PCB remediated by the selected remedy. (White Paper—Sediment PCB Inventory Estimates).

- The estimated Total PCB load to the Lower Hudson will be decreased from the estimate in the FS. This will apply to all alternatives. (See Chapter 11, Master Comment 337780.)

- The estimate for the amount of Total PCB resuspended by dredging will increase, as more PCB mass will be removed. (See Chapter 10, Master Comment 583 and White Paper – Resuspension of PCBs During Dredging.) This increase will not affect the forecasts for fish tissue concentrations downstream, however, since the increase is strictly limited to the mono- and di-homologue fractions.

- The increase in Total PCB concentrations for the TI Pool will serve to increase the estimate for material requiring TSCA handling. However, the FS estimate also did not account for several important factors that serve to reduce the volume of material requiring TSCA handling. Consideration of these factors yields an estimate for TSCA materials that is similar to that presented in the FS (White Paper – Estimate of Dredged Material Exceeding TSCA Criteria).

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Relationship Between PCB Concentrations in Surface Sediments and Upstream Sources
255353

WHITE PAPER – RELATIONSHIP BETWEEN PCB CONCENTRATIONS IN SURFACE SEDIMENTS AND UPSTREAM SOURCES

(ID 255353)

ABSTRACT

The relative age of the Thompson Island Pool (TI Pool) surface sediments and their associated PCB concentrations is important in determining the fate and transport histories of Upper Hudson River PCBs. Surface PCB concentrations within sediments of the TI Pool have been sampled since 1977. Water column samples have been collected at Rogers Island since 1977 as well. However, for both matrices, the frequency of sampling increased markedly in 1991 shortly after GE began its federally mandated water column monitoring of the Upper Hudson and also began sediment sampling. As discussed elsewhere in the Hudson River PCBs Reassessment reports, the PCB contamination of the sediments of the Upper Hudson is predominantly attributable to releases from the GE facilities at Fort Edward and Hudson Falls. This white paper addresses the relationship between the most recent (post-1990) releases and the sediment contamination of the Upper Hudson.

This analysis presented below finds that the surface concentrations in 1998-1999 sediment samples are far too high to represent PCBs that were deposited after 1996. The finding indicates that surface sediment contamination must be derived from historical PCB stores at least older than 1996. Similarly, an examination of high-resolution cores collected in 1992 and 1998, as well as shallow cores collected in 1998-1999, indicates the absence of a significant increase in surface concentrations over the last 10 years of deposition, despite the occurrence of the Allen Mill event in 1991 to 1993. These two findings collectively lead to the conclusion that surface sediment concentrations have been largely independent of the upstream load for the last 10 years.

By inference, it is the historical deposits of PCBs that control the PCB concentrations in the surface sediments because the concentrations at the surface could only have been derived from older, more-contaminated releases, such as the Fort Edward Dam removal. The corollary to this is that the historical, highly contaminated sediments of the Upper Hudson are not being consistently buried by cleaner sediments with less-concentrated amounts of PCBs. Furthermore, because current and anticipated future loads from the upstream sources are already low and expected to decline further, Total PCB concentrations in future deposition resulting from the upstream sources will be at or below the target residual PCB concentrations. Thus the implementation of the selected remedy does not need to be closely linked to further reductions in PCB releases from the GE facilities. Rather, these remedial efforts can proceed in parallel.

INTRODUCTION

A comparison of the geochemical character of the suspended matter and sediments of the Upper Hudson has the potential to associate or disassociate the upstream water column loads from the surficial sediments. The basic premise is: If current suspended solids, which are relatively clean as compared to historical sediment and suspended solids, are burying the existing contaminated sediments of the Upper Hudson, then the mean surficial sediment PCB concentrations should be comparable to those on the suspended matter. This calculation is limited to the period 1996 to 1999, when the Hudson Falls discharges were largely controlled and concentrations at Rogers Island were consistently low.

For the period prior to 1996, water column loads and suspended-matter concentrations were frequently impacted by discharges from the Hudson Falls facility. Commenters have claimed that these discharges were of sufficient magnitude to recharge the surface concentrations of the TI Pool, thus causing the increases in water column PCB load as the water passes over the sediments of TI Pool. If this is the case, then evidence of this sediment recharge should be visible in the high-resolution cores obtained by EPA and GE in the Upper Hudson. To the extent that no "recharge," expressed as higher sediment concentrations, is evident in the most recently deposited sediments, then little or no recharge has occurred, and the releases from Hudson Falls have been transported through the Upper Hudson with little impact on the sediments there. In turn, this implies that the annual sediment releases are the result of historical discharges, most likely the large release event associated with the Fort Edward Dam removal.

ESTIMATION OF SUSPENDED MATTER CONCENTRATIONS AT ROGERS ISLAND, 1996-1999

Suspended-matter concentrations at Rogers Island can be estimated using the two equilibrium-based partitioning models developed in the DEIR (USEPA, 1997). These models have been applied throughout EPA's modeling analysis. In the DEIR, it was shown that the PCB concentrations in the Upper Hudson are well described by the effective partition coefficients calculated from the transect samples from EPA's Phase 2 investigation. These constants are applied here to estimate the concentrations of PCBs on the suspended matter in the water column at Rogers Island over the period 1996-1999. Both three-phase and two-phase equilibria are examined here, although the two-phase model is used as the primary reference for the concentration on suspended matter.

Three-Phase Equilibrium

PCB equilibrium calculations were performed using both three-phase and two-phase representations of sediment-PCB partitioning. The composite partition coefficients for total PCBs were derived from a mass-weighted average of the available partition coefficients for individual congeners using the total PCB concentration in the water column. Specifically, the entire congener spectrum obtained in EPA's April 1993 (Transect 4) water column sampling was used to weight the congener values. This transect was chosen because it is considered representative of the freshly released PCBs from Hudson Falls and is at a sufficiently high concentration that all important congeners are readily detected.

The recent GE water-column samples collected from 1996-1999 were not used because many of these samples were extremely low in concentration and many congeners were not detected. Because only a small number of congener-specific partition coefficients are available for the three-phase calculations, the entire spectrum was not used in deriving the three-phase coefficient. The three-phase partition coefficient was derived from the two greatest (by weight) congener contributors (BZ#28 – 8.5 percent and BZ#31 – 7.0 percent; TPCB) to the overall PCB signature.

Thus, the three-phase partition coefficient for total PCBs was the average of coefficients estimated for BZ#28 and BZ#31. The three-phase equilibrium assumes total PCB concentration in the water column is equal to the sum of PCBs contained within the dissolved fraction, the suspended solids fraction, and the fraction bound to dissolved organic carbon, such that:

$$\text{TPCB Mass/L} = (C_{SS} * TSS) + (C_{DISS}) + (C_{DOC} * DOC)$$

$$= C_{DISS} (K_{OC} * f_{OC} * TSS + 1 + K_{DOC} * DOC)$$

Where:

TPCB	=	Total PCBs Mass/L
C_{SS}	=	Concentration of suspended solids fraction (ng/L)
TSS	=	Total suspended solids (mg/L)
C_{DISS}	=	Concentration of dissolved fraction (ng/L)
C_{DOC}	=	Concentration bound to dissolved organic carbon fraction (ng/L)
DOC	=	Dissolved organic carbon (mg/L)
K_{OC}	=	Partition coefficient to organic carbon (L/kg)
f_{OC}	=	Fraction of organic carbon in the solid phase
K_{DOC}	=	Partition coefficient to dissolved organic carbon (L/kg)

Two-Phase Equilibrium

The two-phase equilibrium analysis assumes that the PCBs bound to the DOC concentration are small or that the DOC is approximately constant (which is suggested by historical data); therefore, the total PCB concentration in the water column is equal to the sum of an apparent dissolved fraction (truly dissolved and DOC-sorbed PCBs) and the suspended solids fraction, such that:

$$\text{TPCB Mass/L} = (C_{SS} * TSS) + (C_{DISS})$$

$$= C_{DISS} (K_{OC} * f_{OC} * TSS + 1)$$

The following water-quality parameters were used in the equilibrium calculations:

TPCB	=	13.5 ng/L flow-weighted average concentration (at Fort Edward [1/19/96 to 9/30/99]; GE Database)
log K_{OC} (three-phase)	=	5.82 (Optimized with temperature correction to 20 degrees C) (DEIR, USEPA, 1997)
log K_{OC} (two-phase)	=	5.90 (DEIR, USEPA, 1997)
f_{OC}	=	0.175 at avg. discharge (Q) = 4,000 cfs; (RBMR, USEPA, 2000b).
TSS	=	3.23 mg/L (RBMR, Equation 6-13a, USEPA, 2000b)
log K_{DOC}	=	4.28 (Optimized with temp. correction to 20 degrees C) (DEIR, USEPA, 1997).
DOC	=	4.72 mg/L average at Fort Edward (RBMR, Table 6-30, USEPA, 2000b)

The results of the three-phase calculation indicate that the C_{SS} = 1.06 mg/kg; C_{DISS} = 9.22 ng/L; and C_{DOC} = 1.7 ng/L, on average, in response to recent loading from the upstream source. The results of the two-phase calculation indicate that the C_{SS} = 1.30 mg/kg; and C_{DISS} = 9.30 ng/L.

DISCUSSION OF RESULTS

The equilibrium-based calculations yield flow-weighted concentrations of one to two ppm on suspended matter. These values represent the mean concentration expected in sediments derived from the suspended matter. However, as can be seen in the GE core profiles presented in the FS (Appendix D1, Figure 13), the surface concentrations at these locations are substantially higher than these values. Values in the top few centimeters of these cores are frequently more than an order of magnitude higher than the one to two ppm produced at Rogers Island. From the coring results collected in 1998 and 1999, it is clear that at the vast majority of locations studied by GE, principally in *Hot Spots 14 and 16*, little to no recent deposition occurred. Surface concentrations remain well above the concentrations that would be produced within the water column of the river, indicating that, despite the passage of three to four years of relatively low suspended matter concentrations, surface concentrations were largely unaffected.

Among the most recent samples, the 1998 GE sediment sampling (O'Brien & Gere, 1999) shows 129 ppm total PCBs in 0-2 cm sediment at *Hot Spot 14* in the TI Pool (sample BS-14F-200), 90 ppm total PCBs in 0-2 cm sediment at *Hot Spot 10* in the TI Pool (sample BS-10T-100), and 53 and 56 ppm total PCBs in 0-1 cm sediment at *Hot Spot 28* below Lock 6 (samples FS-28-3 and FS-28-3). Among 18 cores collected by GE in 1999, two samples at *Hot Spot 5* showed concentrations of 275 and 586 ppm in the 0-5 cm sediment interval (samples P14-03 and P14-05). These results indicate the absence of any substantial deposition or burial of PCBs in these locations during this period.

Discussion Of High-Resolution Coring Results And The Allen Mill Event

The high-resolution cores obtained by EPA and GE provide a means to examine the degree of "recharge," or replenishment, of surface PCB concentrations by the Allen Mill event. This event delivered high PCB concentrations to the Upper Hudson over the period from September 1991 to late 1995. During this period, water column concentrations were occasionally above 500 ng/L at Rogers Island. These concentrations have the potential to generate high sediment concentrations if deposition occurs. It was noted during this time, however, that water column loads appeared to pass through the Upper Hudson with very little loss, implying little to no deposition of suspended PCBs to the river bottom.

If these loads were responsible for recharging the surface sediments, then evidence for this should be found in the high-resolution cores obtained by GE and EPA. These cores record the levels of PCB contamination in recently deposited sediments. Thus, if the Allen Mill event represented a significant addition to the sediment inventories of the Upper Hudson, this should be evident as an increase in the concentrations of PCBs in the sediments deposited in these cores during this period. That is, if this event was important, the concentrations of deposited sediments should increase during this period and decline afterwards.

Five cores are available to examine this event. The results are presented in Figures 255353-1 and 255353-2. The latter figure is an expanded-scale version of the first figure. Evident in both figures is the absence of any significant increase in sediment concentrations in the most recently deposited sediments, generally the upper 10 to 15 centimeters of the core. These cores indicate that depositing sediments did not respond to the Allen Mill event. Instead, the depositing concentrations were controlled by other processes.

In view of the huge existing inventory of PCBs within the TI Pool and the apparent conservative behavior of the water-column loads during this period, it is likely that the Allen Mill event caused little or no significant recharge. Rather, it is the production and release of PCBs from the sediments of the TI Pool itself that is responsible for the PCB concentration in recent deposition at these coring locations. By the absence of a response in these cores to the Allen Mill event, these cores also indicate that this event was of minor importance relative to the catastrophic release of PCBs that occurred in 1974-1976 after the Fort Edward Dam was removed. The releases associated with the Allen Mill event were also small relative to the direct GE discharges, prior to 1974, as the sediment PCB concentrations from the mid-1990s are much lower than those of the 1960s.

Implications for the Selected Remedy

The current loads from sources above Rogers Island yield surface sediment Total PCB concentrations of about 1 to 2 mg/kg, as discussed above. These values are quite close to the anticipated residual PCB concentrations for the selected remedy prior to backfill. As a result of the on-going and planned remedial efforts at the GE facilities, it is also expected that further reductions in the upstream load will continue to occur over time, with the upstream load declining to a level of 0.0256 kg/day at or around 2005. A load at this level would be expected to yield Total PCB surface concentrations of about 0.25 mg/kg, the target residual concentration after backfill. Thus, the current and anticipated future loads from the upstream sources do not represent significant sources for the maintenance of surface sediment Total PCB concentrations. Additionally, these sources should not be a significant source for the recontamination of surface sediments during or after dredging. They are both low relative to current surface concentrations and at or near the target concentrations for remediation.

As a result, the implementation of the selected remedy will not be directly linked to further control of the upstream sources. While upstream source control is important, the river sediments are the predominant source of PCBs in the fish. Remediation of the sediments is therefore necessary, regardless of when additional upstream source control measures are implemented. As noted in the modeling analyses presented in the RBMR and the FS, the current and anticipated future upstream loads do not control riverine concentrations of Total PCB until 25 to 30 years after the dredging operations begin. Thus, the remediation of the sediments and the continued remediation of the GE facilities can proceed in parallel.

CONCLUSIONS

The data from cores collected in 1998 and 1999 demonstrate that in many locations within the TIP, PCB concentrations in the surface sediments have maintained consistently high concentrations of total PCBs, with concentrations in excess of 50 ppm in the top two cm, despite the occurrence of low PCB concentration on suspended-matter concentrations at Rogers Island. Based on either the two-phase equilibrium or three-phase equilibrium calculations, water-column PCBs bound to suspended solids at Rogers Island are expected to exhibit average PCB concentrations of one to two ppm. These concentrations are well below those seen at the tops of high-resolution sediment cores at several locations in the TI Pool. If significant deposition were occurring on a consistent basis through the TI Pool, the 1 to 2-mg/kg suspended solids that settle out would leave significantly lower concentrations in surface sediments. Based on the sediment-water interface concentrations obtained in 1998-1999, it appears that little to no deposition is occurring within some of the highly contaminated sediment *hot spot* areas of the TI Pool.

Thus, the ongoing release of PCBs to the water column in the Upper Hudson River, documented by both EPA and GE data, is primarily attributable to historical PCB deposits. These deposits remain at the surface in many locations, as shown by both EPA and GE cores, despite the recent low levels of load from upstream. These sediments are not being rapidly sequestered, but instead continue to contaminate the water and fish of the Upper Hudson. As a result, the upstream source controls implemented by GE have had little effect on the net release of PCBs to the water column from the sediments, which has continued relatively unabated for the past 10 years (Master Responses 577 and 633 in Chapter 2 of this RS).

Additionally, the Total PCB concentrations in future deposition resulting from the upstream sources will be at or below the target residual PCB concentrations. Thus the implementation of the selected remedy does not need to be closely linked to further reductions in PCB releases from the GE facilities. Rather, these remedial efforts can proceed in parallel.

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Sediment PCB Inventory Estimates
363334

WHITE PAPER – SEDIMENT PCB INVENTORY ESTIMATES

(ID 363334)

ABSTRACT

Estimates of the PCB inventories in Upper Hudson sediments were revised based on additional data and a subsequent analysis of the relationship between Total and Tri+ PCBs. The amounts of Total and Tri+ PCBs were estimated for each river section based on section-specific data. The major change in the inventories of the Upper Hudson involved the Total PCB inventory of the Thompson Island Pool (TI Pool). Relative to the value given in the FS and previous Reassessment reports, the inventory estimate is three times greater (*i.e.*, 45 metric tons vs. the 15 metric tons given in the FS due to the incorporation of additional data on the ratio of Total PCBs to Tri+ PCBs in sediment samples). Estimates of Tri+ PCB inventories changed only a few percent, resulting from subtle refinements in the data sets used for the estimate. Due to these revisions, the estimate for the percentage of Total PCBs remediated by the selected remedy has increased from about 50 percent to 65 percent of the inventory of the Upper Hudson. Above Schuylerville, the selected remedy will address approximately 80 percent of the in-place Total PCB inventory.

Other Total and Tri+ PCB values were also obtained, including an estimate of surface concentration in the TI Pool and estimates of concentration in the dredged materials. The dredged material estimates account for the overcut in clean material. Therefore, the concentration of Total PCB and Tri+ in the dredged material is up to three times less than *in situ* concentrations.

INTRODUCTION

PCB inventories and average surface concentrations presented for the selected remedy (3/10/Select) in the FS were calculated using various data sets collected from 1977 through 1994. The various data sets provide different perspectives on sediment PCB concentrations due to the various sample collection techniques and subsequent analytical procedures performed during the different investigations. These issues were extensively documented in the Phase 2 reports, *i.e.*, the Data Evaluation and Interpretation Report, or DEIR (USEPA, 1997); the Low Resolution Sediment Coring Report, or LRC (USEPA, 1998); and the Revised Baseline Modeling Report, or RBMR (USEPA, 2000). In addition, the reconciliation of the data sets was summarized in the FS.

In delineating remedial areas, various metrics were employed to determine the extent of PCB contamination. These included surface concentrations; mass per unit area (MPA); length-weighted average (LWA); and maximum concentration. In addition to these PCB-based parameters, physical characteristics of the Upper Hudson River sediments were also considered. Definitions and derivations of the various PCB metrics are presented within the FS.

Since the issuance of the FS, the EPA has continued its review of available data. In particular, new GE data have become available, which permitted a review of the Total PCB to Tri+

relationship in Hudson River sediments. Using this new information, EPA has revised its estimate of the mass of Total PCB in several areas of the TI Pool. The calculations of the Total PCB to Tri+ relationship are described in detail in White Paper – Relationship Between Tri+ and Total PCBs. In this White Paper – Sediment PCB Inventory Estimates, the revisions in the relationship between these parameters are reflected in the revised estimates of the Total PCB sediment inventory.

Additionally, this white paper will serve to clarify some of the concerns about the "correct" estimate of the PCB inventory presented in the FS. Due to the focus of the modeling analysis on the Tri+ PCB (generally referred to simply as “Tri+” in this White Paper) parameter as opposed to Total PCB contamination, the FS was unclear as to the total amount of PCB mass contained in the sediments as well as the fraction of that mass to be remediated. This white paper presents a set of revised inventory estimates for both Tri+ and Total PCBs, as well as revised values for the PCB mass remediated under the selected remedy. In particular, this white paper will reflect the new information concerning the current best estimate of the Total PCB to Tri+ ratio developed for the sediments of the TI Pool. This revision has resulted in a significantly higher estimate of the Total PCB mass in the pool.

The discussion below is organized as follows:

- Calculation Methodology
- PCB Inventory in River Section 1
- PCB Inventory in River Sections 2 and 3
- Comparison of the Extent of Remediation for the Selected Remedy and Full Section Removal
- Surface Sediment Concentrations in the TI Pool
- Estimation of PCB Concentrations in Dredged Sediments
- Uncertainty
- Summary

CALCULATION METHODOLOGY

The process of calculating the PCB inventories for various areas of the Upper Hudson has been discussed at length in several Phase 2 reports, including the DEIR (USEPA, 1997), the LRC Responsiveness Summary (USEPA, 1999), the RBMR (USEPA, 2000) and the FS. In each of these documents, historical and current data were examined to estimate PCB concentrations and inventories that were applicable to the analyses being performed. These calculations used various means to integrate the data. Among the more important of these was the area-weighted and length-weighted averaging calculations, as well as the calculation of mass-per-unit-area, or MPA. These formulas are repeated here as an aid to the reader in the subsequent discussions.

The area-weighted average PCB concentration over the entire thickness of contaminated sediment was determined using the following equation:

$$Area - Weighted\ \ Average\ Concentration = \sum_{i=1}^{n} (Area_i \times LWA_i) / TotalArea \qquad (1)$$

where: Area$_i$ = Area of polygon *i* [based on polygonal declustering (4.2.3, DEIR, USEPA, 1997)].

LWA$_i$ = mean concentration of core *i* as a function of core length (see equation below)

n = number of polygons which represent the entire area (effectively equal to the number of samples contained in the area)

$$LWA_i = \frac{\sum_{j=1}^{no.\,core\,segments} Conc_j * l_j}{\sum_{j=1}^{no.\,core\,segments} l_j} \qquad (2)$$

where: $Conc_j$ = PCB concentration in core segment *j*

l_j = length of core segment *j*

no. core segments = number of core segments in the core at location *i.*

In this calculation, the LWA is used as a means to estimate the average concentration at a given location. The area of the polygons comes from a polygonal declustering analysis (Thiessen polygons), as described in USEPA, 1999. Effectively, it is a nearest-neighbor mapping wherein each location on the map is assigned the properties of the sample nearest to it. This approach avoids over-counting those areas with many samples, since the samples are weighted by the amount of area closest to them. Thus, samples close together are weighted by small polygons whereas samples far apart have large polygons. Further discussion of this technique can be found in Chapter 4 of the DEIR (USEPA, 1997).

For calculation of mean surface-sediment concentrations, equation 1 above is modified as follows:

$$Area - Weighted\ Surface\ Concentration = \sum_{i=1}^{n} (Area_i \times Conc_i) / TotalArea \qquad (3)$$

where: Area$_i$ = Area of polygon *i* [based on polygonal declustering (see Appendix B of the LRC Responsiveness Summary, USEPA, 1999).

Conc$_i$ = surface concentration of core *i* (*i.e.*, the concentration in the topmost segment)

n = number of polygons which represent the entire area (effectively equal to the number of samples contained in the area)

The Mass-Per-unit-Area (MPA) of PCBs was determined from the following equation:

$$MPA_i = \sum_{j=1}^{no.\ core\ segments} Conc_j * \rho_j * l_j \qquad (4)$$

where:

MPA_i	= PCB mass at location *i*	
$Conc_j$	= PCB concentration in core segment *j*	
ρ_j	= solids-specific weight of core segment *j* (mass of dry solids per unit volume of *in situ* sediment	
l_j	= length of core segment *j*	

Thus, the MPA represents the integration of the PCB content of the core over its length by summing the mass of PCB found in each core segment. To integrate the mass of PCB in an area, each MPA estimate is multiplied by an associated area, based on the polygonal declustering analysis mentioned previously. This is shown in equation 5:

$$Total\ Mass = \sum_{i=1}^{no.\ polygons} MPA_i * Area_i \qquad (5)$$

where:

$Area_i$	=	Area of polygon *i* [based on polygonal declustering (see Appendix B of the LRC Responsiveness Summary, USEPA, 1999).
MPA_i	=	PCB mass at location *i*

Equation 5 can be applied on a pool-wide basis as well as on a sediment-type basis to obtain estimates of inventory for a whole river section or for a more limited area. These five equations form the basis for the inventory estimates to be provided in the remainder of this white paper.

PCB INVENTORY IN RIVER SECTION 1

The original estimates of PCB mass and concentrations were based on the 1984 survey of the TI Pool conducted by NYSDEC (Brown *et al.*, 1988). As discussed in the DEIR, the measure of PCBs represented the sum of Aroclors but neglected the lightest congeners, particularly the mono- and di- homologues. These data provided a good basis for estimating the Tri+ inventory of the TI Pool but were inadequate for estimating the Total PCB inventory. At the time of the preparation of the FS, the estimate for Total PCB was made based on the 1984 data alone, knowing that the value (approximately 15 metric tons) was a significant underestimate. The effective ratio of Total PCB to Tri+ assumed in the DEIR was 1.06. Since the issuance of the FS, EPA has reviewed the original analysis as well as some additional data collected by GE.

There are several data sets collected after 1984 that describe Total PCB and Tri+ concentrations in the Upper Hudson and in particular, the TI Pool. These include the following:

- 1991 GE Sediment Composite Survey (maximum depth 10 in. [25 cm])
- 1992 EPA Phase 2 High Resolution Sediment Coring Program (maximum depth about 3 ft [91 cm])

- 1994 EPA Phase 2 Low Resolution Sediment Coring Program (maximum depth about 4 ft [122 cm])
- 1998 GE Sediment Composite Survey (maximum depth 2 in. [5 cm])
- 1998-1999 GE Sediment Core Collection (maximum depth 3 ft [91 cm], but most limited to 6 in. [15cm])

None of these data sets presented alone are sufficient to describe the inventory of the TI Pool, but taken together they can provide insight as to the relationship between Total PCBs and Tri+. This is discussed at length in White Paper – Relationship Between Tri+ and Total PCBs. For the data sets presented this analysis provided a basis for converting the 1984 Tri+ data to estimates of Total PCBs, which could then be averaged and/or integrated as needed.
In preparing this estimate, several approximations were made that simplified the calculation process but should not significantly impact the estimate, given the age of the 1984 data set and the limitations and uncertainty associated with the Total PCB to Tri+ correction. Specifically, for the estimation of the LWA Total PCB concentration, the relationship between Total PCB and Tri+ was developed on a core-section basis. However, to avoid a lengthy recalculation process, the relationship was applied to the length-weighted Tri+ values directly. This approximation introduces some uncertainty to the calculation, since the ratio developed for the core based on the LWA may differ from the effective one resulting from the correction of the individual core segments. However, this uncertainty is expected to be small relative to the other uncertainties in the calculation.[1]

A second approximation was made in the calculation of the individual MPA values. In this instance, the Total PCB values were obtained as follows:

$$MPA_{TPCB\ i} = \frac{MPA_{Tri+\ i}}{LWA_{Tri+\ i}} * LWA_{TPCB\ i} \qquad (6)$$

where: $MPA_{TPCB\ i}$ = Total PCB mass-per-unit-area at location *i*
$LWA_{TPCB\ i}$ = Total PCB concentration at location *i*
$MPA_{Tri+\ i}$ = Tri+ mass-per-unit-area at location *i*
$LWA_{Tri+\ i}$ = Tri+ concentration at location *i*

The estimation of the Total PCB values for the MPA, like the LWA, is most accurately calculated from the individual core segments. However, the approximation in equation 6 assumes a constant specific weight for the sediment solid throughout the core. As was shown in the LRC (USEPA, 1998), the solid specific weight was correlated with Total PCBs but the relationship was not very strong (correlation coefficient of 0.55). Therefore, this approximation will add some uncertainty, but should not introduce a significant bias in the integration of the MPA values to estimate the Total PCB inventory in the TI Pool. Given the other sources of uncertainty in the calculation, this approximation should not increase the uncertainty significantly[1].

[1] The other sources of uncertainty to the calculation include (among others) the age of the 1984 data set, the accuracy of the conversion factor from the original sum of Aroclors to Tri+, the uncertainty in the Total PCB to Tri+ ratio for thickly sliced sediments, and the failure to obtain the complete inventory of contaminated sediments in many 1984 sampling locations.

Total PCB inventory and concentrations were estimated for the TI Pool using the 1984 data set, the Total PCB to Tri+ ratio relationship developed in this Responsiveness Summary, and the approximations given above. Inventory and concentration estimates were also obtained for the remediated and unremediated sediments and the cohesive and non-cohesive areas of the pool. Equation 5 was used to integrate PCB mass over area. These values are presented in Tables 363334-1 and 363334-2 for Total PCB inventory and Total PCB concentration, respectively. The remediated and unremediated sediment inventories and concentrations are based on the selected remedy for the TI Pool (*i.e.*, greater than 10 mg/kg or 3 g/m^2). In addition to these values, EPA's estimates of Tri+ PCB are also presented. Note that the Tri+ values were developed as described in Appendix B of the LRC Responsiveness Summary. A minor correction has been made to the values originally presented. As a result, the estimate of the Tri+ PCB inventory of the TI Pool was increased by three percent, from 14.1 to 14.5 metric tons.
The revised estimates for the TI Pool Total PCB inventory represent a major change relative to the previous value. The new value (45 metric tons) is over three times greater than the prior estimate, because the calculations in the DEIR underestimated the correct ratio of Total PCBs to Tri+ PCBs. This estimate brings the TI Pool sediment inventory in line with the inventory estimates of River Section 2, discussed below. Although they are both derived independently, the effective Total PCB to Tri+ ratio for remediated sediments in both river sections is 3.4, indicating that dechlorination levels in the sections are comparable, as might be expected. This also places the TI Pool more prominently in the Total PCB inventory of the Upper Hudson, as might be anticipated given its proximity to the GE source areas.

The inventory of Total PCBs in the TI Pool represents 40 percent of the Total PCB mass in all three river sections. The percentage increased from a previous estimate of 20 percent, due to improved estimates of the mono- and di-homologue fractions of PCBs in the sediments. Finally, these revised estimates show the selected remedy to be more effective in removing PCB mass from the river than originally thought. Based on the revised inventory estimate, the selected remedy will remove approximately 80 percent of the Total PCB mass in the TI Pool and about two-thirds (65 percent) of the inventory in the entire Upper Hudson. The basis for the latter conclusion is further discussed below.

Similar levels of change are noted for concentration estimates of Total PCBs in the TI Pool. The overall estimate of the mean Total PCB concentration for this river section also increased threefold, from about 20 to 63 mg/kg, again reflecting the previously uncounted mono- and di-homologue fractions. Total PCB concentration estimates in the cohesive sediments are most dramatically affected, increasing 3.3 fold from about 44 to 145 mg/kg, while non-cohesive sediments increased 2.6 fold from 11 to 29 mg/kg. The difference in the degree of increase is a direct reflection of the dependence of Total PCB to Tri+ ratio on concentration and the lower levels of contamination in non-cohesive sediments relative to cohesive sediments, as described in the White Paper – Relationship Between Tri+ and Total PCBs.

PCB INVENTORY IN RIVER SECTIONS 2 AND 3

In River Sections 2 and 3, the difficulty in estimating the PCB inventories and concentrations does not arise from the lack of Total PCB data in general but, rather, the lack of a completely descriptive data set. Unlike the TI Pool, there has been no synoptic study of the sediment inventory in this area since 1978. The 1978 study is not considered to be useful for current inventory estimates because the river was still undergoing major sediment bed changes due to the removal of the Fort Edward Dam. Additionally, there are significant analytical uncertainties associated with the data, as noted by the peer-review panel (USEPA, 1999b). Subsequent studies have been more limited either vertically (*e.g.,* GE's 1991 composite cores) or horizontally (*e.g.,* EPA's 1994 low-resolution cores).

As a result, it was necessary to combine data sets to estimate the inventories for these areas. To accomplish this, the data were applied as follows. For the most contaminated, generally cohesive sediment areas, the EPA 1994 low-resolution cores were used to estimate PCB inventory and concentration. In River Sections 2 and 3, the LRC program design was to sample the largest, most extensive *hot spots* with the intention of estimating their inventories. These areas generally have vertically extensive inventories as well as high concentrations, which are best represented by these cores.

Estimation of the areas outside the *hot spots* was more problematic. GE's composite cores had two significant limitations for this purpose. First, the cores only penetrated to 25 cm (10 in.); thus, in many instances, the core composites did not capture the entire in-place inventory. Second, the composite samples frequently composited sediments from inside and outside the *hot spots*; thus, the samples did not exclusively represent those areas outside the *hot spots.* More importantly, the samples did not appear to be limited to a single sediment type or general level of contamination. As a result, the estimates of PCB mass outside the cohesive or selected remediation areas would appear relatively uncertain. As will be shown later in this white paper, the overall range in the GE composite data is rather small so that the actual degree of uncertainty is acceptable for the purpose of this analysis.

Both the EPA and the GE data sets provide independent measures of Total PCB and Tri+. Thus, it was not necessary to develop a relationship between the parameters as was done for the TI Pool. In order to obtain a best estimate of PCB mass in the sediments as well as concentration, the inventories of Total PCB and Tri+ in each area were first estimated separately. Mean concentrations were developed later based on these inventories.

In River Sections 2 and 3, the data are not sufficient to prepare separate estimates of cohesive and non-cohesive sediment inventories. In River Section 2, the remediated sediments are predominately cohesive and, therefore, conditions in the remediated sediments that were sampled were assumed to be representative of the unsampled cohesive sediments being remediated; a similar assumption was used for non-remediated and non-cohesive sediments. In River Section 3, cohesive sediments were assumed to be represented by the *hot spot* areas.

Cohesive Sediment/*Hot Spot* Inventories

The mass estimation for both Tri+ and Total PCB in the *hot spots* of River Sections 2 and 3 has been performed based on the MVUE, *i.e.*, a <u>m</u>inimum <u>v</u>ariance <u>u</u>nbiased <u>e</u>stimator of the arithmetic mean. The MVUE represents the best estimate of the arithmetic mean, given that the underlying data distribution is lognormal. The lognormal distribution of the PCB MPA values is documented in the LRC (USEPA, 1998), and therefore not repeated here. The formula for the MVUE is given by Gilbert (1987). The particular formula used here is based on the Psi function

(Gilbert, 1987, equation 13.3), and not the statistically less-rigorous approximation used in the LRC.

Equation 3, given previously, was used to calculate the individual MPA values for Total PCB and Tri+ at each of the low-resolution coring locations. The inventory for each *hot spot* or dredge zone was then estimated by calculating the MVUE based on all the coring locations contained within the zone. This calculation is similar to that performed in the LRC (USEPA, 1998), although a more rigorous approximation is used in these current calculations. The values for the MVUE estimate of the MPA were then multiplied by the total area of the *hot spot* to estimate the Total PCB or Tri+ mass for the area. In this manner an estimate of mass for each of the studied *hot spots* (*i.e., Hot Spots 25, 28, 31, 34, 35, 37* and *39*) was obtained. In the first five *hot spots*, the areas themselves were defined from the side-scan sonar results, which identified cohesive sediments. Areas for *Hot Spots 37* and *39* were based on the original NYSDEC boundaries (Tofflemire and Quinn, 1979).

Since the LRC program in River Sections 2 and 3 focused on *hot spot* areas and, therefore, predominately cohesive sediments, areas consisting of cohesive sediments that were not sampled as part of the program were considered to be similar to those areas that were sampled. Thus, the samples collected from the study areas were considered generally representative of unsampled cohesive sediments. An MVUE of the Total PCB and Tri+ was calculated for the unsampled cohesive sediments using all the samples collected in the river section. Thus samples from *Hot Spots 25* through *35* were used to estimate the MPA for River Section 2 ,and samples from *Hot Spots 37* and *39* were used to estimate the MPA for River Section 3. In River Section 2, the MPA value was multiplied by the area of the additional cohesive sediments in the section, which did not have PCB data, to obtain an estimate of their PCB mass. In River Section 3, the areas of the *hot spots* defined by NYSDEC and the River Section 3 MPA value were used to estimate the PCB mass. Both Total PCB and Tri+ were done in this manner.

In River Section 2, the unsampled areas added approximately 25 percent to the Total PCB and Tri+ inventories. In River Section 3, the unsampled areas were approximately equal to the studied areas in mass.

Non-Cohesive Sediment Inventories

In River Sections 2 and 3, the PCB mass in non-cohesive areas was estimated using GE’s 1991 composite samples. In both sections, only those GE composites falling entirely outside the remediation zones were used in the estimate. In River Section 2, this criterion eliminated all but one of the ten GE composites. In River Section 3, 15 of the 60 composites were excluded by this criterion. The main reason for the higher number of samples accepted in River Section 3 is the proportionately smaller area targeted for remediation in this section.

The identified GE composite samples were then used to create MPA values for each composite line for each parameter. The lines were then arithmetically averaged to obtain the MPA for the section. For River Section 2, four composites were identified as coarse-grained; however, they all included locations within the dredge zones. Despite this concern, these four coarse samples should be characteristic of the unremediated sediments, since it was the fine-grained sediments that were targeted. As it turned out, the one composite entirely outside of the remediation areas in River Section 2 had a LWA value (14.8 mg/kg) that was quite close to the average of the four separate coarse-grained composites in the section (12.1 mg/kg). Thus, the limited data set did not appear to introduce a large amount of uncertainty, since the range is relatively small.

Using the MPA from composite lines entirely outside of remediation areas, the Total PCB and Tri+ inventory in each section for non-remediated areas was then calculated as follows:

$$\text{MASS} = \text{total area (not remediated)} \quad \text{x} \quad \text{average MPA (not remediated)} \qquad (7)$$

For River Section 2, composite rocky areas were excluded from the calculation. For River Section 3, no exclusion for rocky areas could be made due to the lack of data to define these areas. A solid specific weight of 1 g/cm^3, which is close to the average of the cohesive and non-cohesive solid-specific weight values, was used in the MPA calculation. This introduces some potential bias in the calculation since the contaminant mass in non-cohesive areas will be underestimated and the contaminant mass will be overestimated in cohesive sediments. However, given the larger sources of uncertainty involving the extrapolation of the data in general, this correction is not worth further pursuit.

Summary of PCB Inventories in River Sections 2 and 3

Using the approaches described above, the Total PCB and Tri+ inventories were estimated for each river section. These results are summarized in Table 36334-1. The Total PCB inventories given here are the same as those given in Chapter 3, Table 3-4 of the FS, with the addition of the channel dredging. Thus, the 23,600 kg for the *hot spot* remediation in River Section 2 given in Table 3-4 of the FS is 24,300, with the added channel area. Similarly, the previous estimate of 6,700 kg in River Section 3 under the selected *hot spot* removal is replaced with 7,100 kg. The Tri+ calculation presented in Table 363334-1 represents additional inventory estimates for the sections.

River Section 2 is most like the TI Pool in that 80 percent or more of its PCB contamination will be remediated under the selected remedy. As noted in the White Paper – Relationship Between Tri+ and Total PCBs, the relationship between Total PCBs and Tri+ is also similar, with a ratio of 3.4 in the remediated areas. The ratio in River Section 3 is slightly lower, at 2.7, reflecting a lower level of dechlorination due to lower levels of contamination in the section.

In River Section 3, a much more limited PCB removal is anticipated. Thus, its percent removal is only 22 percent. Part of this results from the fact that the section is quite large and much of the area in this section has relatively low levels of contamination, which are, in general, too impractical to consider for remediation.

Estimation of Total PCB and Tri+ Concentrations in River Sections 2 and 3

The section-wide average concentrations of Total PCB and Tri+ were calculated from the mass estimates rather than an independent statistical analysis or via polygonal declustering. Since both the mass of PCB contamination in the sediments and the mass of contaminated sediments were known (the mass of contaminated sediments was known based on the reported length of contaminated core sections and the measured or estimated sediment densities), calculation of the average *in situ* concentration was simply the quotient of these two values converted to mg/kg. These values are given in Table 363334-2. Again, the values for River Section 2 are more similar to River Section 1 than River Section 3, as would be expected.

COMPARISON OF THE EXTENT OF REMEDIATION FOR THE SELECTED REMEDY AND FULL SECTION REMOVAL

There have been several comments that indicate that the PCB mass anticipated to be removed under the selected remedy is not sufficient, and that further remediation is warranted. After modifying the inventories in the sediments and reviewing the areas for remediation, the results indicate that the mass of Total PCBs to be removed represents about 65 percent of the Upper Hudson inventory, a larger fraction than originally reported. Above Lock 5 (*i.e.,* River Sections 1 and 2), the anticipated removal is about 80 percent as measured by Total PCBs or Tri+. Further, for cohesive sediment in the TI Pool, the selected remedy is estimated to remove about 94 percent of the Total PCB and Tri+ inventories.

It is useful to compare these values to those estimated for the most extensive removal alternative (*i.e.,* REM 0/0/3). The estimated masses of Total PCBs and Tri+ remediated under the full-section removal are presented in Table 363334-3. The mean *in situ* concentrations for both Total PCBs and Tri+ for the TI Pool were also estimated for comparison. These values are presented in Table 363334-4. *In situ* concentration is examined in this section to serve as an example of the types of concentration differences between the two alternatives.

As can be seen by comparing Tables 363334-1 and 363334-3, the full-section removal addresses a larger fraction of the PCB inventories. For the entire Upper Hudson the REM 0/0/3 alternative would address 78 percent of the PCB inventory, as opposed to the 65 percent addressed under the selected remedy. For the region above Lock 5, the percent of Total PCBs remediated under the REM 0/0/3 alternative is 96 percent as opposed to 82 percent for the selected remedy. The gain in the fraction of Total PCBs and Tri+ removed comes as the result of a great expansion in the areas affected. Specifically, the number of acres affected under the REM 0/0/3 alternative (964 acres) is nearly double that affected under the selected remedy (493 acres) while the mass of PCBs removed increases by only 20 percent (from 70,000 to 84,000 kg).

The extended areas affected under the REM 0/0/3 alternative are not evenly distributed among the sediment types. For example, in Section 1 the percentage of Total PCB mass removed increases from 82 percent for the selected remedy to 94 percent for the full-section treatment. However, nearly all of this change is due to the addition of a large amount of non-cohesive sediment and associated PCBs. The change in percent remediated for the cohesive sediments is only 3 percent, from 95 to 98 percent. Thus the additional PCB mass in the full-section alternative is the result of the addition of many acres of low PCB concentration, coarse-grained sediment. Similarly, the percentage of remediated Tri+ inventory for cohesive sediments increases from 94 to 99 percent as a result of the more-extensive remediation. Given the extensive volume of cohesive sediments to be removed under either alternative, these differences are only minor.

Changes in the non-cohesive percentage remediated are much more dramatic – the percentage remediated increases from 55 to 88 percent for Total PCBs, and the Tri+ increase is similar.

Thus, the net effect of the full-section alternative for the TI Pool is the addition of extensive areas of non-cohesive sediments. A similar increase between the alternatives would be expected for River Section 2, which achieves an even higher percentage PCB mass removal with a smaller affected area under the selected remedy.

The changes in the mean concentration of the dredged material can be seen by comparing Tables 363334-2 and 363334-4. Essentially, as the level of remediation increases, the PCB concentrations on the material removed approach the PCB concentrations of the material left behind undisturbed. Both values converge to that of the mean *in situ* condition. This is a result of several factors, the most important of which is that for such high removal fractions (better than 80 percent), the conditions of the material removed must approach those of the *in situ* material, since the remediation would remove nearly the entire inventory.

The estimated concentrations of the residual sediments left undisturbed also approach the mean. The cause of this is less clear, as there is no *a priori* reason for it. However, a closer examination reveals that the PCB concentration estimates for much of the unremediated areas are based on the extrapolation of data points over relatively long distances. Few data points are actually in the undisturbed area, in part because much of it is rocky or adjacent to rocky areas and difficult to sample. Thus, as the area left undisturbed gets smaller, the amount of site-specific data pertinent to that area declines and instead the estimates of PCB mass and concentration rely on data points that are outside of the undisturbed area. This is believed to be the cause of the converging sediment concentrations.

SURFACE SEDIMENT CONCENTRATIONS IN THE TI POOL

Estimates of cohesive sediment surface concentrations were required for the revision of the Depth-of-Scour Model (DOSM) calculations. At the request of the State of New York, the impacts of a higher flow rate at Fort Edward were examined (White Paper – Application of the Depth-of-Scour Model [DOSM] in the Thompson Island Pool for Additional Flooding Assumptions). Additionally, the estimate of Total PCB released by a 100-year flood was reassessed using the revised estimates for the TI Pool inventory.

To this end, both (1) the surface core segments and grabs samples obtained by the 1984 NYSDEC survey, and (2) the 1991 GE composite samples were used to estimate Total PCB and Tri+ concentrations in the uppermost sediment layers. The estimate based on the 1984 data provides a reasonable upper bound on the PCB concentration in resuspended sediments during dredging. The depths of the 1984 NYSDEC cores (generally 30 cm or 12 in.) are substantially greater than the predicted depths-of-scour (mean depth is less than 1 cm), and since PCB levels generally increase with depth, the average value obtained from the 1984 data should represent a high-end estimate for resuspension.

The mean surface concentration for cohesive sediment was estimated based on a polygonal declustering analysis using the 1984 data set and the side-scan sonar definitions of sediment texture (USEPA, 1997). The mean surface concentration in the TI Pool was calculated using the Tri+ surface concentrations and Equation 3 (given previously). This yielded a value of 51 mg/kg for cohesive surface sediments. Given that the Total PCB to Tri+ ratio for the TI Pool is approximately 3.4, Total PCBs were not calculated as an area-weighted average but rather by simply multiplying the Tri+ value by the ratio, yielding 170 mg/kg for Total PCB. These values are given in Table 363334-5.

The 1991 GE composite samples were also used to estimate surface concentrations for the DOSM model analysis. These samples are believed to represent a low-end value for several reasons. First, the GE composites appear to contain both cohesive and non-cohesive sediments, thus diluting the more-contaminated cohesive sediment values with the non-cohesive samples. Similarly, they represent a mechanically averaged sediment concentration (via compositing), which tends to suppress higher values. Lastly, since the set of composites represent all areas of the pool (cohesive and non-cohesive), they cannot be easily separated or classified according to sediment type. Because the composites extend across sediments types they generally cannot be classified based on the sediment texture of the sampling location as the cores were.

The concentrations in the 0-5 cm layers of the 1991 GE composite cores were arithmetically averaged to yield mean values for both Total PCB and Tri+. These are also given in Table 363334-5. Notably, the range between the GE- and NYSDEC-based estimates is close to a factor of three for Tri+ and more than a factor of five for Total PCBs. This partially reflects the sensitivity of the Total PCBs estimate to the Tri+ value. The ratio of Total PCB to Tri+ is only 2 for the GE sediments, as opposed to 3.4 for the 1984 samples.

ESTIMATION OF PCB CONCENTRATIONS IN DREDGED SEDIMENTS

In addition to characterizing the PCB contamination in place on the river bottom, estimates of PCB concentration on the dredged materials themselves were also needed as part of several engineering analyses. The nature of the dredging process is such that a significant amount of uncontaminated sediment is expected to be incorporated with the target material. This material will effectively dilute the PCB concentrations on the dredged material, resulting in lower PCB concentrations in the materials undergoing the handling process than in the in-river sediment. In estimating the impacts of the dredging process, it is important to recognize this dilution step, since uncontaminated sediments do not generally pose a risk or dramatically affect the environment.

The estimates of sediment mass and volume to be removed were discussed extensively in the FS as well as in Chapter 4, Master Comments 313219 and 313224, in this RS. The estimation of the mean concentration of Total PCBs and Tri+ on the dredged sediments is given as follows:

$$Conc_{TPCB} = \frac{Mass_{TPCB}}{Vol_{Sed} * r_{Sed}} \tag{8}$$

where:

where:	$Conc_{TPCB}$	=	Concentration of Total PCB in the dredged material for the river section
	$Mass_{TPCB}$	=	Mass of Total PCB contained within the sediments to be dredged
	Vol_{Sed}	=	Volume of sediments to be dredged
	ρ_{Sed}	=	Solid-specific weight of the sediments to be dredged (*i.e.*, the mass of dry solids per unit volume of wet sediment

Using this calculation, the concentration of the dredged material was estimated for each section of the Upper Hudson. Additionally, in the TI Pool it was possible to further separate the calculation by sediment type. These results are presented in Table 363334-6. Both Total PCB and Tri+ concentrations were estimated for each section, as well as for the entire removal operation.

The results show a slightly greater than threefold decline in the Total PCB concentrations of dredged material when compared to *in situ* values for River Sections 1 and 2, which reflects the significant amount of overcut and dredging of low-level contamination from the channel. These sections had extensive data on the depth of sediment contamination and its horizontal extent, as well as bathymetric data. These data provided input for the selection of overcut depth.

Since these data were not available in River Section 3, the actual volume of dredged material and the extent of overcut have much greater uncertainties. Therefore, the overcut in River Section 3 was estimated on a more limited basis and not as rigorously as River Sections 1 and 2. As a result, the concentration estimates for the dredged material in River Section 3 declined from the *in situ* values by only 30 percent. It is important to note that these estimates will undergo extensive refinement as part of the remedial design, when a sampling program will be implemented. The estimates of concentration calculated here are simply intended to provide a basis for estimations involving the transport and processing of the materials and potential hazards or concerns related to dredging. Specifically, the values provided in Table 363334-6 were used as a basis for estimating resuspension losses, gas-exchange losses, and other material transport and processing issues. The issue of TSCA material is discussed separately in White Paper – Estimate of Dredged Material Exceeding TSCA Criteria.

UNCERTAINTY

The nature of the mass estimates provided here does not lend itself easily to quantitative estimates of uncertainty. Although the various data sets could be used to create statistical estimates of uncertainty, there is no simple statistical technique to account for the differing ages of the data, the geochemical changes in the river since the date of collection, the differences in analytical and sampling techniques, and the correction factor (*i.e.*, ratio) between the Tri+ and Total PCB estimates. As a result, there are no presentations of statistical uncertainty. Rather, a professional judgment of uncertainty indicates that most of the inventory estimates probably have an uncertainty of at least $\pm$ 25 percent.

CONCLUSIONS

The reassessment of Total PCB and Tri+ concentrations yielded improved estimates of concentration and total mass for each parameter in each river section. For Tri+, the changes were minor. However, for Total PCBs, the revised inventory of the TI Pool was found to be three times greater than the previous estimate given in the FS. This revision resulted from analysis of the Total PCBs to Tri+ ratio described elsewhere in the White Paper – Relationship Between Tri+ and Total PCBs. As a result of the revisions, the estimate for the TI Pool is in closer agreement to the conditions measured in River Section 2. The revisions also indicate that the TI Pool contains the largest fraction of PCB mass in the Upper Hudson. This follows intuitively, since the TI Pool is closest to the GE facilities and the former Fort Edward dam site. The Total

PCB inventory of the Upper Hudson was estimated to be 110,000 kg (240,000 lbs). Of this, 70,000 kg (150,000 lbs) will be removed as part of the selected remedy.

Estimates were also provided for the percentage of PCB inventory remediated by the selected remedy. Approximately 65 percent of the Total PCB inventory in the Upper Hudson will be remediated under this remedy. This represents an upward revision from the original estimate of approximately half, as a result of a revised estimate of the mono- and di-homologue inventories for the sediments based on recent and earlier data. The estimate of percent PCB inventory removed increases from the previous estimate because the selected remedy targets cohesive sediments. Cohesive sediments in general have higher PCB concentrations and, accordingly, have a higher ratio of mono- and di-homologues than the non-cohesive sediments. The Tri+ inventory estimates presented in the FS were essentially unchanged.

In the TI Pool the selected remedy is estimated to remove 80 percent of the Total PCB inventory. This was contrasted with the 94 percent remediated by the full-section alternative ($>0 g/m^2$) in River Section 1. However, most of this difference results from the addition of low-PCB level, coarse-grained sediment to the remediation volume. The selected remedy addresses about 95 percent of the fine-grained-sediment PCB inventory, as compared to 98 percent under the full-section removal.

Estimates for surface concentrations and concentrations on dredged materials were also derived. Concentrations for surface sediments were obtained from the 1984 NYSDEC sediment survey for cohesive sediment and from the 1991 GE composite samples for the entire TI Pool. These values differed by approximately a factor of five. This range reflects the difference in sampling depths as well as other sampling artifacts, such as compositing. Estimates of concentrations in material targeted for dredging were obtained by integrating the PCB mass and sediment mass slated for removal. In this manner, the impact of the removal of clean sediments along with the contaminated sediments was taken into account. This approach yielded dredged material concentrations that were about three times lower than the *in situ* concentrations.

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Metals Contamination
253002

WHITE PAPER – METALS CONTAMINATION

(ID 253002)

ABSTRACT

Comments included concerns about unacceptable levels of residual heavy metals in river sediments when the selected remedy is implemented. Based on multiple data sets spanning 16 years, the evidence suggests that the historical release of heavy metals found in the Upper Hudson River sediments coincides with the input of PCBs. This co-depositional pattern is clearly evident in Figure 253002-1, which shows the coincidence of maximum metals and PCB concentrations in the early- to mid-1970s (early- to mid-1960s for nickel and zinc) and then a downward trend in concentration in recently deposited sediments. The data indicate that when the selected remedy (REM 3/10/Select) is implemented, not only significant amounts of PCBs will be removed from this river environment, but also much of the heavy metals that were deposited coincidentally with the PCBs. Therefore, residual metals in newly exposed sediments would be at or near background levels. A sampling and evaluation plan for metals will be implemented during remedial design in order to ensure there is no problem with sediment residuals or dredged sediment disposal.

INTRODUCTION

Throughout the course of PCB-related data collection and analyses in the Upper Hudson River, it became apparent that other contaminants were also associated with the same sediments. The first reported data set with respect to heavy-metal concentrations in Upper Hudson sediments is from the 1977 sediment survey (as reported by Tofflemire and Quinn, 1979, and Brown, *et al.*, 1988). Selected samples from this survey were submitted to the NYSDOH for metals analyses. The metals analyzed for included: calcium, chromium, copper, iron, lead, manganese, nickel, potassium, rubidium, strontium, titanium, and zinc. The results of these analyses indicated elevated levels of chromium, copper, lead, manganese, nickel, and zinc, and are summarized in Table 253002-1. For purposes of this report, the term 'elevated levels' refers to concentrations of various metals detected above background levels (Tables 253002-1 through 253002-11) observed in this region. Subsequent data sets, as outlined within the metals data inventory below, were also compiled with respect to Upper Hudson River sediments and metals concentrations. These data have shown that in addition to elevated PCBs concentrations within the sediments, there also are elevated metals concentrations.

The concern was raised that dredging would expose the biota to deeper sediments with elevated metals concentrations. Indeed, removing one contaminant but leaving another would be unacceptable. Another concern focused on the potential mobility of these metals through the process of leaching during the dredging and sediment handling/processing requirements. However, the extraction procedure (EP) toxicity analyses (comparable to Toxicity Characteristic Leaching Procedure [TCLP]) performed on the 1984 and 1986 data indicated that these sediments and their associated metals did not exceed any of the TCLP criteria. In fact, all of the results were at least one order of magnitude below the criteria, and thus, the metals should not be redistributed as a result of leaching.

The data clearly indicate that there is a direct correlation between concentrations of PCBs and metals within these sediments. Furthermore, the removal of PCB-contaminated sediments in target areas will effectively remove much of the metals contamination, and the residual metal levels will be at or near background levels. Therefore, even prior to the placement of clean backfill over dredged areas, the concentrations and availability of metals to the biota will be within acceptable levels.

METALS DATA INVENTORY

Metals data used in this analysis include results contained in the Hudson River PCBs database as well as in other Hudson River reports. These data sets represent sediment, biota, and sediment EP toxicity results, and include the following sources:

- 1977 – Sediment grab data from 1977 and analyzed by NYSDOH, obtained from Tofflemire and Quinn, 1979 (Table 253002-1).

- 1984 – Sediment core data (avg. 80-cm depth) from the 1984 sediment survey, obtained from Brown *et al.*, 1988 (Table 253002-2).

- 1986 and 1987 – Sediment core data (generally one- to two-foot depth) from *Hot Spots 3*, *8*, and *20* collected in 1986 and 1987, obtained from Brown *et al.*, 1988 (Table 253002-3).

- 1983 and 1991 – Sediment core data from *Hot Spot 20* (0- to 44-cm depth) collected at RM 188.5 in 1983 and RM 188.6 in 1991 by Dr. Richard Bopp (Rensselaer Polytechnic Institute [RPI] and Lamont-Doherty Earth Observatory [LDEO] [Tables 253002-4 and 253002-5]).

- 1993 – Sediment data from 0-5 cm collected at RMs 203.3 (baseline station), 194.1, 191.5, 189.5, 189.0, 188.7, and 188.5 for the Ecological Risk Assessment (ERA) by EPA in 1993 (Tables 253002-6, 253002-7, and 253002-8).

- 1984 and 1986 –EP toxicity analysis (comparable to EPA test Method 1311, TCLP) of nine sediment samples from the 1984 survey and six samples from the 1986 survey, obtained from Brown *et al.*, 1988 (Table 253002-9).

- Fish data collected by NYSDEC in 1988 (Table 253002-10).

- Fish data collected in 1997 and 1998, as reported by Sloan, 1999.

The entire sediment data set is summarized in Table 253002-11.

DISCUSSION OF SEDIMENT METALS DATA

Each of the data sets described above provides a slightly different perspective on the levels of inorganic contaminants in Upper Hudson River sediments. No one data set is completely descriptive but combined they provide a fairly complete picture. In this section, the results are combined to describe the likely set of conditions to be encountered under the selected remedy.

The core data obtained by Dr. Richard Bopp provide an important historical perspective (Tables 253002-4 and 253002-5; Figure 253002-1). By the use of radionuclide dating procedures (such as those in Bopp and Simpson, 1988, and USEPA, 1997), it is possible to establish a chronology of sediment deposition and transport. The sediment cores collected by Bopp provide data on both PCBs and metal transport and deposition over the period from 1954 to 1991. These data show that the occurrence of elevated metal concentrations in the sediments is coincident with the highest PCB concentrations. Metal concentrations in the deepest core segments are close to those seen in the baseline samples at RM 203.3. Thus, the removal of PCB-contaminated sediments will also achieve near-baseline levels of metals within the residual sediments.

A second important feature of Dr. Bopp's core data is the decline in metals concentrations in the shallowest core layers. These data indicate levels approaching those seen at the baseline site at RM 203.3 for all metals but chromium. However, even chromium is reduced by an order of magnitude from its peak concentration. These results indicate that areas dredged as a part of the selected remedy will not become recontaminated with elevated metal concentrations after the dredging is complete. That is, since the most recently deposited sediments, represented by the shallowest sediment layers, are at or close to background levels, the river is currently depositing metal concentrations that do not present a contamination problem. Thus, once an area has been cleaned of its metal contamination, the river's depositional processes should not recontaminate it with elevated metals levels. This would not be the case if metals were currently being released to the river or otherwise being re-released from the sediments at a significant rate.

While the Bopp data provide information on the history of metal deposition in the Upper Hudson, the samples are not considered to be spatially representative of the river bottom. To assess the likely metal levels in the sediments to be dredged, the data from all available sources was combined to estimate the mean metals concentrations as well as likely maximums. These results are summarized in Table 253002-11.

The 1977 grab samples analyzed by the NYSDOH (Table 253002-1) were collected at 20 stations within the Thompson Island Pool (TI Pool). The analytical results indicate that chromium, copper, lead, manganese, nickel, titanium, and zinc were detected above background levels (Note: Titanium did not have any detectable background level; however, all reported samples from within the TI Pool had detected concentrations).

The 1984 sediment survey produced over 400 cores for analysis (Table 253002-2). The cores averaged 80 cm (32 inches) in depth from stations within the TI Pool. A subset of these samples was sent for analytical testing of metals based on the results of PCB analyses. The primary purpose of the metals analyses was to characterize sediments within proposed PCB dredging areas with respect to metals concentrations and potential residual levels after remediation. A summary of the results is provided in Table 253002-2. The results indicate elevated levels of arsenic, cadmium, chromium, copper, lead, mercury, and nickel, relative to background. The chromium and lead levels are on the same order as the data from the 1977 samples.

NYSDEC collected six cores at *Hot Spots 3, 8,* and *20* (two from each *hot spot*) in 1986 and again in 1987. A summary of the analytical results is provided in Table 253002-3. The results indicate elevated levels of antimony, arsenic, barium, cadmium, chromium, copper, lead, mercury, selenium, vanadium, and zinc ,relative to background levels.

The 1983 and 1991 cores collected by Dr. Bopp were located at RM 188.5 and RM 188.6, respectively. These cores were on the eastern side of the river within *Hot Spot 20*. The 1983 core was collected to a depth of 40 cm and the 1991 core to a depth of 44 cm. A summary of the analytical results is provided in Tables 253002-4 and 253002-5, as well as Figure 253002-1. Each of the two cores was dated using ^{137}Cs to trace chronological deposition of sediments and their respective contaminants. The analytical results indicate elevated levels of cadmium, chromium, copper, lead, nickel, and zinc, relative to background levels.

As discussed above, the significance of the metals-PCB relationship over time obtained from the Bopp cores is of great importance. Note that the two cores were separated by one-tenth of a mile and collected eight years apart, yet each of the cores depicts a good correlation between metals and PCB concentrations. This observation is relevant when discussing residual metals based on the selected PCB remedy.

Removing PCB-contaminated sediments will also remove nearly all of the harmful metals from the same system (with no additional dredging). Residual concentrations of metals will be at or near background levels. The evidence for this is provided in Table 253002-4. The 1983 Bopp core indicates that a PCB concentration of 1.2 mg/kg is observed at a depth of 36 to 40 cm, which approximately corresponds to the year 1954. If sediment removal occurred to this depth, the residual metal concentrations would be: cadmium (0.2 mg/kg), chromium (11 mg/kg), copper (6 mg/kg), lead (5 mg/kg), nickel (5 mg/kg), and zinc (288 (mg/kg). These residual metal levels, when compared to both surface and background concentrations (Table 253002-4a), indicate that they would be considerably less than current surface concentrations within the TI Pool. All but cadmium would be less than background levels.

However, these concentrations are based on removal to 40 cm (16 inches) (year: 1954), whereas the *hot spot* associated with this core (*i.e., Hot Spot 20*) is targeted for sediment removal to 122 cm (4 feet). Since the targeted depth is set below the zone discussed above, and significantly pre-dates the 1954 sediments and their respective contaminant inputs, it is safe to infer that the PCBs and metal levels will also be significantly less. Thus, the amount of metals removed from this environment is nearly 100 percent, with only trace amounts in the dredging residual.

The 1993 sediment survey for the ERA obtained metals data in sediment. There were five stations in the lower TI Pool (RMs 191.5 – 188.5), one Rogers Island station (RM 194.1), and one background station (RM 203.3). The background station at RM 203.3 was used to determine a baseline level for metals and PCBs concentrations, since its location was sufficiently upgradient of the two GE facilities. Summaries of the results are provided in Tables 253002-6, 253002-7, and 253002-8. The analytical results indicate elevated levels of arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, lead, manganese, mercury, nickel, thallium, and vanadium, relative to background levels observed at the RM 203.3 sampling location.

Extraction Procedure Toxicity Analyses Results

The samples described previously document the level of metals contamination in Upper Hudson River sediments. These data do not provide information on the suitability of the sediments for disposal in a landfill. The test for suitability is the TCLP test. Nine sediment samples from the 1984 survey and six samples from the 1986 survey were analyzed for PCBs and also analyzed using EP toxicity testing, which is comparable to EPA test Method 1311 (the TCLP test). This procedure was done to provide analytical results on the leachability of metals (such as arsenic, barium, cadmium, chromium, lead, mercury, selenium, and silver) within Upper Hudson River sediments. A summary of the analytical results is provided in Table 253002-9. No exceedances of TCLP criteria were found in any sample for any metal. The results indicate that the TCLP results are at least one order of magnitude below the Leaching Toxicity Characteristic Standards (40 CFR § 261.3). Therefore, the potential risks associated with residual metals (post-remedy, in-river) and metals within sediments during processing/transfer will be minimal, due to the low leaching characteristics. The data indicate that disposal of the sediments will not require special restrictions due to leachable metal levels.

NYSDEC FISH DATA

In addition to metals data for sediments, NYSDEC also collected data for fish tissue. In 1988 NYSDEC sampled fish above (RM 201.3) and at two locations (RMs 198.3 and 198.2) adjacent to the Hercules/Ciba-Geigy paint factory to determine concentrations of selected metals in fish. Carp, smallmouth bass, and bass were sampled. Standard fillets for analyses were prepared from all fish; however, liver samples were analyzed from carp only. A total of 14 RM 201.3 background samples (9 fillet, 5 liver), 22 samples from RM 198.3 Hercules/Ciba-Geigy Station 5 (12 fillet, 10 liver), and 19 samples from RM 198.2 Hercules/Ciba-Geigy Station 5 (12 fillet, 7 liver) were analyzed. Samples were analyzed for cadmium, chromium, mercury, nickel, lead, strontium, and vanadium (Table 253002-10).

At RM 201.3, the background fillets had elevated concentrations of mercury, with an average concentration of 0.8 ppm in fillets and 0.6 ppm in liver (Table 253002-10 and Figure 253002-2). Liver samples had an average of 5 ppm cadmium, 0.6 ppm mercury, and 0.2 ppm strontium and vanadium.

At RM 198.3, the fillets had lower concentrations of mercury, with an average concentration of 0.3 ppm in fillets and none detected in liver samples. Strontium was detected at 0.2 ppm in the fillets and at 0.1 ppm in the liver samples. Liver samples also had average concentrations of 11 ppm cadmium, 2 ppm chromium, 1 ppm lead, 0.1 ppm strontium, and 0.2 ppm vanadium.

Just downstream, at RM 198.2, the fillets also had an average concentration of 0.3 ppm mercury, with no mercury detected in liver samples. Liver samples had average concentrations similar to those at RM 198.3, with 16 ppm cadmium, 1 ppm chromium, and 0.3 ppm strontium. The elevated concentrations of cadmium and chromium detected in fish livers at these locations may be due to releases from the paint factory, as these contaminants are used in paint manufacturing.

Contaminants other than PCBs were also analyzed in Hudson River fish collected in 1997 and 1998 (Sloan, 1999). The only metals included as additional contaminants by Sloan were mercury and cadmium. Cadmium concentrations in fish (*i.e.,* yellow perch [3 samples] and pumpkinseed [5 samples]) at all stations within the vicinity of the Hercules/Ciba-Geigy paint station (RMs 201 and 189) were below 0.01 ppm. No livers were analyzed separately in 1997. One of the conclusions of the briefing on 1997 striped-bass results prepared by NYSDEC (Sloan, 1999) was that other contaminants (*e.g.*, DDT, mercury, PAHs, dioxins, and dibenzofurans) are present in the Hudson River, but do not represent as great a problem as PCBs.

RESIDUAL CONCENTRATIONS AND OVERALL CONCLUSIONS

The two cores collected by Dr. Bopp (1983 and 1991) illustrate key aspects of the spatial and chronological relationship between PCBs and metals in the TI Pool sediments (Tables 253002-4 and 253002-5; Figure 253002-1). Table 253002-4 suggests that the maximum levels of PCBs are contained within the same sediment depth horizon as the maximum metals concentrations, or just below (1983 core) the metals maximum concentrations. That is, in the 1991 core, maximum values for Total PCBs and metals occurred at 28-32 cm, while in the 1983 core, maximum metals levels occurred in the sediment layer just below the maximum Total PCBs level. Based on the good correlation between PCBs and metals concentrations, the removal of PCB-contaminated sediments will also achieve near-baseline levels of metals within the residual sediments.

The data from Table 253002-4 suggest that the maximum metals concentrations occur at a maximum depth of 32 cm (12.6 inches). These data generally represent cohesive sediments, where metals contamination is greatest. The minimum depth of removal in cohesive sediments in the TI Pool under the selected remedy is 2 feet (61 cm). The proposed depth of removal in *Hot Spot 20*, where the Bopp cores were collected, is 4 ft. (122 cm). A comparison of the 1983 Bopp core to the 1993 background sample indicates that levels at a depth of 40 cm within the Bopp core are near or below the actual baseline levels (Table 253002-12). Thus, the planned depth of sediment removal in cohesive sediments is at least double the mean depth of metals contamination, as well as more than double the measured depth of contamination in *Hot Spot 20*.

The sediment data from the TI Pool undeniably suggest that the metals were deposited coincident with the PCBs. Based on the depth of removal proposed in the selected remedy, the removal of PCB-contaminated sediments will also remove nearly 100 percent of the metals contamination above background levels. The residual traces within the sediments will be either near or below the baseline metals concentrations. Furthermore, clean backfill placed above dredged areas will reduce aquatic exposure to metals and hence will further minimize the ecological impact.

Based on the fish data, cadmium and chromium may be passed on in the food chain, although tissue concentrations are lower than sediment concentrations (*i.e.*, biomagnification is not occurring). Cadmium has been implicated as the cause of severe deleterious effects on fish and wildlife (Eisler, 1985). Concentrations of cadmium in freshwater above 10 ug/L are associated with higher mortality, reduced growth, inhibited reproduction, and other effects. Effects are most pronounced in waters of comparatively low alkalinity. Adsorption and desorption rates of cadmium are rapid on mud solids and particles of clay, silica, humic material, and other naturally occurring solids.

Chromium toxicity is dependent on speciation, with the hexavalent form considered the most toxic. Although chromium is an essential trace element in many species, at high environmental concentrations it is a health concern (Eisler, 1986).

Cleanup of contaminants to lowest effect levels (LEL) (Persaud *et al.*, 1993; Long and Morgan, 1990) of 0.6 mg/kg cadmium, 26 mg/kg chromium, and 31 mg/kg lead would be protective of aquatic organisms. However, these levels are generally at or close to background levels and would be difficult to achieve. Cleanup based on severe effect levels (SEL) (Persaud *et al.*, 1993; Long and Morgan, 1990) of 9 mg/kg cadmium, 110 mg/kg chromium, and 110 mg/kg lead is unlikely to be adequately protective of aquatic organisms and would be required at only a few areas, as most locations are below these levels. Other sediment quality targets that may be considered for metals are the threshold effects concentration (TEC) (below which adverse effects are not expected to occur), and the probable effects concentration (PEC) (above which adverse effects in sediments are expected to frequently occur), which are consensus-based sediment quality guidelines (SQGs) developed by MacDonald *et al*. (2000). TECs and PECs are 0.99 and 4.98 mg/kg for cadmium, 43 and 111 mg/kg for chromium, and 36 and 128 mg/kg for lead, respectively. A realistic cleanup goal can be considered to fall within the range of these sediment-quality targets, which is likely to be achieved (based on the correlation between PCBs and metals) by the remediation performed for PCBs under the selected remedy (*i.e.*, REM-3/10/Select).

In sum, heavy-metal contamination within the Upper Hudson River sediments will be removed at the same time as the PCBs are removed, based on the dredging plan for the selected remedy. The concentrations of residual metals (post-remedial) are expected to be at or near the baseline (Table 253002-12). No exceedances of TCLP criteria were found in the historical data. The leachability of metals has been found to be at least one order of magnitude below the TCLP criteria (Table 253002-9). A sampling plan for metals will be implemented during the remedial design in order to confirm the conclusion that metal contamination does not pose a significant concern for sediment residuals or dredged sediment disposal.

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Dioxin Contamination
860

WHITE PAPER – DIOXIN CONTAMINATION

(ID 860)

ABSTRACT

Chlorinated dibenzodioxins and dibenzofurans (hereafter simply referred to as dioxins) are present in the sediments of the Upper Hudson River. To the extent that levels exceed regulatory criteria, dioxins may present additional concerns with respect to the handling and disposal of PCB-bearing sediments. Dioxins were detected in four samples from a sediment core collected by Dr. Richard Bopp in 1991 at RM 188.6 in the vicinity of *Hot Spot 20* (Table 860-1). This data set is the most useful set of available data because it was obtained as part of a dated sediment core. Dioxins were also detected in samples collected in 1983 and 1987 (Brown *et al.*, 1988) (Tables 860-2 and 860-3). However, the available dioxin data are not sufficient to determine the full extent of contamination in the Upper Hudson. In order to better understand the spatial and chronological distribution of dioxins within these sediments, more sampling and analyses for dioxins will be conducted as part of the remedial design phase.

INTRODUCTION

Upper Hudson River dioxin data are very limited. The most useful data set is from the Bopp 1991 core at RM 188.6, which was analyzed by the New York State Department of Health (NYSDOH) and the New York State Department of Environmental Conservation (NYSDEC, 2001a). This core was analyzed for PCBs, metals, radiometric dates (^{137}Cs), and, subsequently, dioxins, as well as physical characteristics. A summary of the analytical results for dioxins, furans, and total PCBs is provided in Table 860-1. This core was collected from sediments located at the southern end of the Thompson Island Pool (TI Pool) in the vicinity of *Hot Spot 20*. It was analyzed in a manner similar to the EPA's high-resolution cores (USEPA, 1997). For a more thorough discussion of the core results and coring location, refer to Brown *et al.*, 1988 and McNulty, 1997.

Other dioxin data associated with Upper Hudson River sediments are from Brown *et al.*, 1988. There are two data sets reported – one from 1983, which consists of two cores (one located at RM 188.5 and one located at RM 191.1 [Table 860-2]), and the other from 1987, which consists of composite samples from *Hot Spots 20*, *8*, and *3*.

Based on the limited dioxin data from the Upper Hudson River, relatively little can be asserted about dioxins and their correlation to PCBs within the river. In order to better understand the spatial and chronological distribution of dioxins within these sediments, more sampling and analyses for dioxins will be conducted as part of the remedial design. This sampling can occur prior to as well as during dredging operations, in order to best assess the dioxin concentration of the *in situ* sediments as well as the cumulative concentrations of dredged sediments. The dioxin concentrations of the cumulative dredged sediments are necessary in order to determine the processing and landfill requirements. The dioxin criterion for disposal in a non-TSCA-permitted landfill is less than 0.001 mg/kg.

DISCUSSION OF SEDIMENT DATA

The 1991 Bopp core used for analyses was located near the eastern shore of the Hudson River, at RM 188.6 within *Hot Spot 20*. The core was segmented and analyzed for dioxins in the following depth intervals: 0-2 cm, 20-24 cm, 28-32 cm, and 40-44 cm. The maximum concentration of the sum of tetra-, penta-, hexa-, and hepta- dioxins and furans was found in the 20- to 24-cm depth interval at a level of 180,000 pg/g, or 0.18 ppm (Table 860-1).

Note that this peak concentration lies just slightly above the peak concentration in the core corresponding to total PCBs, and cadmium, chromium, copper, and lead at 28 to 32 cm. Also note that the dioxin peak appears to decrease steadily with depth below the maximum. These data suggest that dioxin levels are largely coincident with PCB levels and that dioxin levels will attain background levels at the same depth as PCBs. However, the number of dioxin samples is not sufficient to confirm this since the core segments were not analyzed all the way to the bottom of the core.

In this core, the measured dioxin maximum lies close to the PCB maximum, suggesting similar discharge and transport histories for the two contaminants. However, the lack of deeper samples or a complete core leaves open the issue of how deep the dioxin contamination extends and its correlation with PCBs. Further sampling as part of the design will be required to clarify these issues. Note that the data set for metals contamination was able to address the depth of contamination issue, since samples are available from the deeper core segments (White Paper – Metals Contamination).

The results from the NYSDEC 1983 data set as reported by Brown *et al.*, 1988, indicate a slight increase in concentration going from 8 – 12 cm to 24 – 28 cm in depth (RM 188.5) with respect to 2,3,7,8 tetrachloro dibenzodioxin (TCDD); total TCDD; and 2,3,7,8 tetrachloro dibenzofurans (TCDF). There was a slight decrease in concentration in total TCDF. The sample recovery was poor for the surface samples at RM 188.5; thus, no samples were analyzed. Based on the limited data from Table 860-2, it is difficult to make any inferences about correlations between dioxins and PCBs. However, all samples fell below the TSCA threshold of 0.001 ppm, or 1 ng/g.

Composite sediment samples were collected by NYSDEC in 1987 from *Hot Spots 20, 8,* and *3* (two from each *hot spot*). The average core length for the individual samples in the composite was 19.2 inches. All of the samples from *Hot Spots 20* and *3* were non-detections (Table 860-3). The composite samples from *Hot Spot 8* had concentrations of 0.17 ng/g and 0.1 ng/g of total TCDF. Again, all samples fell below the TSCA threshold.

LANDFILL CRITERIA

The criterion for dioxin levels in material for disposal in a TSCA-permitted landfill is 0.001 ppm (1 ng/g) per homologue (tetra-, penta-, hexa-, and hepta-). For this determination, dioxins and furans are counted separately. The concentrations of dioxin and furan homologues from the 1991 Bopp core (RM 188.6) can be found in Table 860-1. Surface concentrations (0-2 cm) suggest that surface sediments would not require a TSCA-permitted landfill. However, based on the three other samples at depth (20-24 cm, 28-32 cm, and 40-44 cm), 62.5 percent of the homologue values exceeded the criteria.

By contrast, the results for the NYSDEC 1983 and 1987 samples showed no exceedances. One of the major differences between the Bopp and the NYSDEC samples is the thickness of the sample. Bopp's segments are substantially thinner than those obtained by NYSDEC. The NYSDEC samples can be considered closer to the integrating effects of the dredging process; that is, the dredge operations will remove and homogenize sediments over a one- to two-foot thickness. Thus, the dredged material will tend to have lower concentrations than those obtained from thin core slices.

Combining the results of the three investigations suggests that much of the material to be dredged will not require special disposal based on dioxin levels. It is anticipated that, since PCBs and dioxins have similar geochemistries, high levels of dioxin will tend to be coincident with high levels of PCBs. The Bopp core data suggest such a relationship. It is likely then that any sediments requiring special disposal because of dioxin levels will also require this treatment due to PCB levels. As a result, no additional volume of TSCA material beyond that estimated for PCBs is anticipated at this time. However, the limited data set for dioxins is not sufficient to completely support this conclusion.

Since a well-defined spatial distribution of dioxins cannot be determined from the limited data available, sampling will be required during the remedial design. Once further analyses have been concluded, an estimation of the dioxin concentrations with respect to the cumulative volume of sediments can be made. Thus, after further sampling and evaluation, the necessary processing and landfill requirements can be reassessed.

FISH DATA

Data on dioxin levels in Hudson River fish were obtained by NYSDEC on several occasions. In all, there are 50 Lower Hudson River dioxin samples in the NYSDEC fish database and 25 Upper Hudson River samples. This data set includes two samples from RM 201, above the GE facilities (Table 860-4). Most of the samples were obtained during 1997, but a few were obtained from 1987 to 1991.

Samples were typically analyzed as standard fillets, with a few liver analyses. These fish results have been summarized by homologue in Table 860-4.

In the Upper Hudson River, dioxins were detected in less than 45 percent of the samples for each of the homologue groups (tetra-, penta-, hexa-, and hepta-), except for tetrachlorodibenzofurans, which were detected in about 90 percent of the samples (Table 860-4). In the Lower Hudson River, dioxins were detected in about 50 to 94 percent of the samples for each of the homologue groups.

The highest detected concentrations were for the tetrachlorodibenzofurans. The highest observed concentration for this homologue group in the Upper Hudson River was obtained from a white sucker standard fillet sample at RM 185 at 30 µg/kg. In the Lower Hudson, the maximum value, 26 µg/kg, occurred in a striped bass standard fillet sample from RM 73. In general, average observed concentrations are lower in the Upper Hudson River samples as compared to the Lower

Hudson River. Specifically, for six of the eight homologue groups reported, the Lower Hudson had higher average values.

The fact that dioxin levels in fish were generally higher for the Lower Hudson relative to the Upper Hudson indicates that there are substantive additional sources of dioxin in the Lower Hudson. This is unlike PCBs in the Hudson, which are dominated by the GE source. The fact that the Upper Hudson is not the highest in dioxin levels indicates that any remediation of the Upper Hudson is unlikely to have a substantive impact on dioxin levels in fish in the Lower Hudson.

CONCLUSIONS

Elevated levels of dioxins within the Upper Hudson River sediments appear to be generally coincident with elevated levels of PCBs and metals. Since, geochemically, dioxins behave similar to PCBs within the environment, it is anticipated that the removal of PCB-contaminated sediments will also remove other contaminants, such as metals (White Paper – Metals Contamination) and dioxins, thereby eliminating multiple environmental threats by a single process. The concentrations of dioxins in fish indicate the influence of independent contaminant sources in the lower river; therefore, sediment removal in the upper river is likely to reduce dioxin levels in upper river fish only. However, the limited dioxin data hamper the ability to make inferences over large spatial extents. Since a better data set is necessary to assess the extent of dioxin contamination within these sediments, a sampling and evaluation plan will be implemented during remedial design.

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Model Forecasts for Additional Simulations in the Upper Hudson River
363150

WHITE PAPER – MODEL FORECASTS FOR ADDITIONAL SIMULATIONS IN THE UPPER HUDSON RIVER

(ID 363150)

ABSTRACT

Concerns were expressed that some of the assumptions used to conduct model forecast simulations with EPA's fate and transport (HUDTOX) and bioaccumulation (FISHRAND) models may affect comparisons between various remedial scenarios (*e.g.*, the selected remedy [REM-3/10/Select] versus monitored natural attenuation [MNA]). This paper presents results from additional forecast simulations with the HUDTOX and FISHRAND models that were designed to address these concerns.

The new model forecast simulations examine three specific factors and their effects on long-term predictions of Tri+ PCBs in the water column, surficial sediments, and fish of the Upper Hudson River: a phased implementation of the remedial dredging, with the first year at less than full-scale production, extending the schedule for the selected remedy by one year; reduction of upstream PCB loads to zero as a lower-bound estimate for external source control; and inclusion of sediment Tri+ PCB resuspension losses due to dredging. The results of these additional HUDTOX and FISHRAND forecast simulations:

- Demonstrated that phasing of the implementation schedule for the selected remedy (REM-3/10/Select) by extending it one year does not significantly affect the predicted long-term trajectories for Tri+ PCB levels in the water, sediments, and fish of the Upper Hudson River.

- Demonstrated that elimination of upstream Tri+ PCB loading does not diminish the relative separation between predictions for the selected remedy and MNA scenarios in the water column, surficial sediments, or fish.

- Clarified the significance of sediment-mixing processes and sediment-water interactions in controlling long-term Tri+ PCB levels in surficial sediments, water, and fish as external sources of PCBs entering the river are reduced or perhaps even eliminated.

- Showed that by reducing the upstream Tri+ PCB load to zero, the Remediation Goal of 0.05 ppm PCB in fish tissue (wet weight) will be reached in all river sections during the modeling time frame. The Remediation Goal will be reached much sooner for the selected remedy than for Monitored Natural Attenuation.

- Demonstrated that while dredging-induced resuspension of sediment will likely increase Tri+ PCB levels in the water column (as measured by either concentration or load passing a given location) and in fish, the impacts will largely be confined to the years during which the selected remedy is implemented, regardless of whether a 0.13-percent or a 2.5-percent loss rate is applied. Impacts from dredging-induced resuspension (at either rate) are also predicted to occur in downstream surficial sediments, but the increases in concentration are generally small relative to existing Tri+ PCB contamination levels.

- Showed that application of the 2.5-percent dredging-induced loss rate results in much greater increases in Tri+ PCB levels in water and fish during the active remedy period than does EPA's loss rate estimate of 0.13 percent. However, EPA believes that the 2.5-percent loss rate is unrealistically high and that the 0.13-percent loss rate is a justifiably conservative estimate (Appendix E.6 of the FS report; White Paper –Resuspension of PCBs during Dredging).

These findings are not unexpected, but they do serve to provide a further quantitative assessment of important technical issues raised through comments on the analyses presented in the FS.

INTRODUCTION

Additional model runs were conducted to evaluate issues raised after the release of the Proposed Plan and FS. These additional model runs considered the implications of upstream source-control measures reducing future PCB loading at the upstream boundary to zero, dredging-induced resuspension of PCBs, and a revised schedule of contaminated-sediment removal for the selected remedy (REM-3/10/Select) based on a phased approach.

Descriptions of the assumptions used for the additional modeling runs and the basic results of the HUDTOX and FISHRAND models are examined in this paper. Note that additional discussion of the predicted fish concentrations and associated human health and ecological risks are found in White Paper – Human Health and Ecological Risk Reduction under Phased Implementation.

A lower-bound calculation was conducted for the selected remedy in which it was assumed that future upstream source control completely eliminates Tri+ PCB loading to the river. The comparison of results between this lower-bound calculation and the post-source-control upstream PCB load predicted in the FS better clarifies the influence of the assumed upstream source on the selected remedy, as well as the effectiveness of the selected remedy.

Possible impacts of PCBs being resuspended during dredging have also been incorporated into model simulations for the revised remedy schedule. This was accomplished through inclusion of two different estimates of downstream transport of Tri+ PCB due to dredging-induced resuspension during the dredging season each year. The resuspension loss rates assumed for these simulations were 0.13 percent (from Appendix E.6 of the FS and White Paper – Resuspension of PCBs during Dredging) and 2.5 percent and were taken to represent sediment mass loss to the water column. As discussed in the White Paper – Resuspension of PCBs during Dredging, the 0.13 percent resuspension loss rate is based on the 0.3 percent resuspension rate at the dredge head. The value of 0.13 percent represents a resuspended-mass-weighted average over River Sections 1, 2, and 3 and is a conservative estimate, based on the assumptions employed in determining this value (presented in detail in Appendix E.6 of the FS). These conservative assumptions include:

- No dredging-related settling of resuspended sediment (or PCB mass) beyond 10 meters downstream of the dredge head.

- No use of silt curtains that would reduce downstream sediment transport.

Although EPA believes that 2.5 percent resuspension loss rate is unrealistically high for the dredging equipment and methods that are expected to be used at the site, the Agency decided to run its model using the 2.5 percent loss rate in view of, among other things, the large number of public comments received on the dredging resuspension issue.

The revised schedule extends the remediation effort over six years, versus the five-year schedule originally proposed in the FS, to allow for a phased implementation. The first phase will be the first construction season of remedial dredging. The dredging during that year will be implemented initially at less than full-scale operation and will include an extensive monitoring program of all operations. These monitoring data will be compared to performance standards identified in the Record of Decision or developed during the remedial design with input from the public and in consultation with the State and federal natural resource trustees.

In the ROD, EPA has identified performance standards that address air and noise emissions from the dredging operations and the sediment processing/transfer facilities. Performance standards that will be developed during the remedial design phase will address (but may not be limited to) dredging resuspension, production rates, PCB residuals after dredging (or dredging with backfill, as appropriate), PCB air emissions, and community impacts (*e.g.*, odor). The information and experience gained during the first phase of dredging will be used to evaluate and determine compliance with the performance standards. Further, the data gathered will enable EPA to determine if adjustments are needed to operations in the succeeding phase of dredging or if performance standards need to be reevaluated.

DESCRIPTION OF REVISED SCENARIOS UNDER THE SELECTED REMEDY

As described in Section 6.4.2.2 of the FS, the selected remedy includes expanded *hot spot* removal (nominal mass-per-unit-area (MPA) targets are 3 g/m^2 Tri+ PCBs or greater) in River Section 1, *hot spot* removal (nominal MPA targets are 10 g/m^2 Tri+ PCBs or greater) in River Section 2, and removal of selected areas containing high PCB concentrations that are potentially subject to scour in River Section 3.

The selected remedy also includes sediment removal in the navigation channel as necessary to implement the remediation and to accommodate normal boat traffic on the river. Isolation of residual PCBs in sediments that may remain after dredging is completed through addition of a layer of clean backfill material suitable for replacement of the fish and benthic habitat. No backfill will be placed in the navigation channel. After construction is completed, MNA will be implemented in each section of the river until the remedial action objectives (RAOs) are achieved. The areas to be remediated under the selected remedy are shown in Plate 17 of the FS. The total area of sediments targeted for removal is approximately 493 acres, and the estimated volume of sediments to be removed is 2.65 million cubic yards.

Several HUDTOX model forecasts were conducted to examine various alternative assumptions that could be applied in simulating the selected remedy. Table 363150-1 describes eight HUDTOX model forecast scenarios that are presented in this paper. Three of these scenarios have already been presented in the FS report: No Action, MNA, and REM-3/10/Select (5 years –

no resuspension). The other five scenarios address various alternative assumptions, including an extended implementation schedule, upstream PCB source loading conditions, and possible effects of dredging-induced PCB resuspension.

The REM-3/10/Select alternative (the selected remedy), as presented in the FS, incorporates a five-year dredging schedule with remediation commencing in 2004[1] and being completed by 2008. It has been decided to extend the schedule from five to six years to allow for the phased implementation of the selected remedy. Table 363150-2 contains the percent of each HUDTOX sediment segment that is not dredged and the Tri+ PCB concentration in the portion of each segment that is dredged. It also contains the timing for changing the value (sequencing) and also compares the sequence for the six-year phased-implementation schedule to the five-year schedule.

The removal, capping, and MNA alternatives evaluated in the FS all specify that sediment remediation activity will be performed in conjunction with a separate source-control action in the vicinity of the GE Hudson Falls plant. At this time, it is expected that GE will implement the upstream source control under NYSDEC authority. The upstream source control is characterized in the HUDTOX model by assuming an upstream boundary water column Tri+ PCB load of 0.16 kg/day from 1998 through 2004, followed by a step-down reduction to 0.0256 kg/day on January 1, 2005. In order to further clarify the effect of the upstream boundary conditions on long-term surficial sediment and water column Tri+ PCB predictions, model forecasts for REM-3/10/Select (five-year schedule) and MNA were conducted with a step-down upstream load reduction on January 1, 2005 to zero (*i.e.*, complete elimination of upstream PCB sources).

An evaluation of dredging-induced resuspension effects on model-forecasted water column and sediment Tri+ PCB concentrations for the selected remedy was also conducted in order to address questions regarding this issue in a more comprehensive fashion. Estimated dredging-induced Tri+ loads resuspended to the water column on a daily basis during the active dredging season (May 1 – November 30) for the five-year and six-year scenarios were included in HUDTOX model forecasts for the selected remedy according to the schedules presented in Tables 363150-3 and 363150-4.

Two rates of dredging-induced PCB resuspension were simulated under the six-year remediation schedule: A river section-specific estimate of 0.13 percent mass loss and a public-requested 2.5 percent mass loss. The river section-specific suspension rate estimate is the time-weighted average of the three sections of the river (The resuspension rate for each river section of the river is described in the White Paper –Resuspension of PCBs during Dredging). In addition, the original five-year dredging schedule for the selected remedy was simulated using section-specific estimates for dredging-induced resuspension losses. The estimates of dredging-induced contaminant resuspension were simulated as additional Tri+ PCB loads to specific water column segments in the HUDTOX model. The loading to the HUDTOX water column segments was done in a north-to-south approach. (Note that the north-to-south approach has been assumed for this exercise and that it is expected that such an approach will generally be used. However, the actual location of the first phase of work will not be determined until the remedial design stage.)

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[1] EPA now expects dredging to commence in 2005. Initiating dredging in 2005 would not be expected to significantly affect modeling projections or the comparative analysis of alternatives.

As with all of the remedial alternatives presented in the FS, the scenarios evaluated in this paper, excluding No Action, also rely on institutional controls (such as the fish consumption advisories), and naturally occurring attenuation processes to reduce the toxicity, mobility, or volume of the remaining PCBs in the Upper Hudson River sediments after the construction is completed.

For the selected remedy, target areas in River Section 1 with an MPA target of greater than 3 g/m^2 Tri+ PCBs sediments (cohesive and non-cohesive) are removed. For River Section 2, an MPA target of greater than 10 g/m^2 Tri+ PCBs was selected as the minimum target area criterion, and all target areas with cohesive and non-cohesive sediments in this section are removed. For River Section 3, NYSDEC-defined *Hot Spots 36, 37,* and part of *39* are targeted for removal. This scenario also includes removal of navigational channel sediments as required to implement the remedy. The percent Tri+ PCB mass removed from the sediment is calculated using the polygonal-weighted average method (instead of point-averaged) for River Section 1. For River Sections 2 and 3, the point-averaged method is used to calculate percent Tri+ PCB mass removed.

HUDTOX Implementation for the Selected Remedy

As presented in Appendix D.2 of the FS, the refined engineering sediment-removal remedies are implemented within the HUDTOX model, as follows.

For River Sections 2 and 3, initial average MPA conditions were calculated for a given segment by averaging the MPA of each point within the segment. This approach assumes that each point contributes equally to the initial conditions of the segment; none is more heavily weighted than the others. The average MPA was then recalculated for the segment (assuming removal of those points that fall within the target MPA area) by averaging the MPA of each remaining point. The average calculated MPA was multiplied by the associated area to determine the mass of Tri+ PCBs. One minus the ratio of the recalculated MPA to the initial-condition MPA represents the percent mass removed for the segment during remediation. This calculated percent mass removed is assumed to be representative of the sediment segment. A Tri+ PCB percent mass removed associated with the removal was provided for each sediment segment.

For River Section 1 (TI Pool), Tri+ PCB percent mass removal was calculated as described above (for River Sections 2 and 3) for 15 of the refined engineering model runs. For the remaining model runs, Tri+ PCB mass, mass removed (*i.e.*, Tri+ PCB mass in areas targeted for removal), and mass remaining (*i.e.*, Tri+ PCB mass in areas not targeted for removal) were calculated for each segment by using the Thiessen polygon area-weighted MPAs. The Tri+ PCB mass values were used to calculate the percent mass removed for each sediment segment.

HUDTOX Forecast Simulations for Revised Scenarios under the Selected Remedy

Figures 363150-1 through 363150-14 present comparisons over the 70-year forecast period of predicted HUDTOX Tri+ PCB concentrations in the surficial sediments (cohesive and non-cohesive) and in the water column at various locations throughout the Upper Hudson River for the scenarios presented in Table 363150-1.

The effect of the revised implementation schedule and inclusion of a dredging-induced resuspension load on predicted surficial sediment and water column Tri+ PCB concentrations is largely confined to the six-year active dredging period (2004 to 2009) for the river section-specific estimate (0.13 percent) of dredging-induced Tri+ PCB resuspension (R20RS). Outside of the period of scheduled dredging, impacts on water column Tri+ PCB concentrations are minimal. Inclusion of 2.5 percent for resuspension losses (R20RX) results in significantly higher water-column concentrations during the dredging period and slightly elevated water-column concentrations for five to 10 years in river sections downstream of the TI Pool (Schuylerville to Federal Dam).

Surficial sediment concentrations in sections of the river below the TI Pool are slightly impacted for a number of years beyond the completion of dredging, due to the inclusion of the river section-specific estimate (0.13 percent) of dredging-induced Tri+ PCB resuspension in either the 5-year or 6-year scenarios (R14RS and R20RS). These impacts are not large when compared to selected remedy predictions that do not include resuspension (R14S2). The inclusion of 2.5 percent for resuspension losses (R20RX) results in predictions that show a negligible impact on surficial sediment concentrations in the TI Pool after dredging is completed, but somewhat greater and longer lasting impacts on surficial sediment concentrations downstream. For example, Tri+ PCB concentrations in surficial cohesive sediments at Stillwater remain slightly elevated for approximately 15 to 20 years compared to simulations that included either no resuspension (R14S2) or the river section-specific resuspension loss rate of 0.13 percent (R20RS and R14RS).

The effect of assuming that future upstream source control reduces Tri+ PCB loads to zero is clearly seen in both the water column and surficial sediment predictions (R14S0). This assumption eliminates the upstream boundary as a factor in controlling predicted long-term water column and surficial sediment Tri+ PCB levels. Under this assumption, the interactions between the sediments and water column (*i.e.*, solids dynamics, sediment-to-water PCB fluxes, etc.) largely control the predicted future trajectories. The implications of this result are readily apparent. First, upstream source control to levels below those that are anticipated to occur slightly accelerate the predicted decline of Tri+ PCB concentrations in the surficial sediments as a result of the dilution effect from cleaner solids entering the river. Second, predicted sediment Tri+ PCB concentrations under MNA remain well above levels predicted for the selected remedy, even assuming source control could eliminate upstream PCB loading to the river. Simply put, the degree of separation between the MNA and selected remedy predictions for Tri+ PCBs in the sediments is not diminished when an assumption of complete upstream source control is applied.

The predicted annual Tri+ PCB loads over the Thompson Island Dam, the Northumberland Dam, and the Federal Dam (to the Lower Hudson River) for each of the seven HUDTOX forecast scenarios are shown in Tables 363150-5 through 363150-7. In general, the annual loads for the five- and six-year scenarios (R14RS and R20RS) that incorporate the river section-specific resuspension loss rate of 0.13 percent are not appreciably different from the annual loads for the REM-3/10/Select alternative (the selected remedy) with no resuspension. Differences in the predicted loads to the lower river are largely confined to the five- and six-year active dredging periods, and are largely due to the inclusion of dredging-induced resuspension losses. Application of a 2.5 percent estimate for resuspension losses (R20RX) results in significantly greater predicted Tri+ PCB water column loading during the active dredging period, and continued slightly higher annual loading rates at locations downstream of the Thompson Island Dam for approximately 15 to 20 years after dredging is completed in comparison to other selected remedy scenarios with lower resuspension losses.

The effect of assuming complete control of upstream PCB sources is also seen in the predicted loads entering the Lower Hudson River. Under the assumption of stepping down to zero upstream load, both MNA and the selected remedy (R14S0) predict continued declines in loading to the lower river throughout the 70-year forecast period. However, the load difference between MNA and the selected remedy remains approximately the same, regardless of whether upstream Tri+ PCB loads are reduced to zero or to the 0.0256 kg/day rate applied for the forecast scenarios presented in the FS. The predicted annual Tri+ PCB loads to the lower river for all of the other scenarios, whether assuming a constant upstream load (*i.e.*, No Action) or a reduced load, eventually reach approximately constant values controlled by the upstream boundary load and annual variations in the specified hydrologic conditions.

FISHRAND Forecast Simulations for Revised Scenarios under the Selected Remedy

The FISHRAND model requires surface sediment and dissolved water Tri+ PCB concentrations corresponding to the three river sections as described in the FS. All FISHRAND model parameters were the same as those used in the Revised Baseline Modeling Report (USEPA, 2000); the only differences were the predicted sediment and water concentrations passed from the HUDTOX model to the FISHRAND model.

FISHRAND modeling results show, similar to the HUDTOX modeling, that the revised implementation schedule and inclusion of a dredging-induced resuspension load is largely confined to approximately a 10-year period (2004 to 2014). Results are provided as follows:

- Figure 363150-15 presents a graph of the results for largemouth bass, brown bullhead, and yellow perch just for the period 2004 through 2010 (2015 at RM 154) to highlight the differences in predicted concentrations.

- Figure 363150-16 presents the results for these species through the end of the modeling period (2067).

- Figure 363150-17 presents the results for white perch. This species was only modeled at RM 154, as they are typically not found in the Upper Hudson River above that location.

- Figure 363150-18 presents the species-weighted results for the four different selected remedy model forecasts.

- Figure 363150-19 presents the species-weighted results for the four different selected remedy model forecasts shown in Figure 363150-18, with the addition of the No Action (P3NACW), MNA (P3NAS0), MNA with upstream load decreasing to zero (P3NAS0), and the five-year selected remedy with upstream load decreasing to zero (R14S0).

- Figure 363150-20 presents the species-weighted results shown in Figure 363150-19, with an expanded scale to show only those years during and immediately following dredging.

Predicted fish tissue concentrations for the selected remedy scenarios are within less than 1 percent of each other by approximately 2015 for all species and locations in the Upper Hudson River.

A comparison was conducted of the five-year implementation of the selected remedy without dredging-induced Tri+ PCB resuspension (R14S2), as presented in the FS, with the selected remedy including the section-specific loss estimate (0.13 percent) for dredging-induced Tri+ PCB resuspension (R14RS). The comparison found that the difference in predicted fish body burdens is no more than approximately 15 percent and typically less than 10 percent, for all species and locations in the Upper Hudson River. As stated above, by 2015 the difference between these scenarios is less than 1 percent.

A comparison of predicted body burdens under the five-year versus six-year implementation schedule, both assuming the section-specific loss rate (0.13 percent) for dredging-induced Tri+ PCB resuspension (R14RS and R20RS, respectively), shows slightly higher concentrations during the time frame that dredging is occurring for the six-year scenario. The differences last until approximately 2008 – 2010 at RMs 189 and 184, and until approximately 2015 at RM 154.

Predicted fish body burdens for the six-year implementation schedule with the dredging-induced Tri+ PCB resuspension loss rate of 2.5 percent (R20RX) are higher than all the scenarios (including No Action) at the beginning of the period of dredging, but quickly drop to levels commensurate with the other REM–3/10/Select scenarios by the end of the dredging period.

The differences among predicted fish concentrations are typically greatest for the species more closely associated with the water column. At RM 189, the predicted difference between the five-year and six-year implementation schedules are greater than the predicted difference between the two resuspension assumptions for brown bullhead, a predominantly sediment-associated fish. For largemouth bass, the 2.5 percent resuspension assumption has a greater effect than the difference between the implementation schedules. Yellow perch tend to follow the largemouth bass pattern, although absolute concentrations are slightly lower. At RM 154, the effects of resuspension last slightly longer than at locations further upstream.

The impact of reducing the upstream Tri+ load to zero is seen in the results for the selected remedy (R14S0) and the Monitored Natural Attenuation alternative (P3NAS0). Under the selected remedy (R20RS) (6-years, 0.13 % dredging-induced resuspension, 0.0256 kg/d upstream Tri+ PCB load), the Remediation Goal of 0.05 ppm PCBs (wet weight) in fish tissue is reached only in River Section 3, in 2050. If upstream source control can reduce the Tri+ load to zero, then the selected remedy implemented in 5 years, with no resuspension (RS14S0) is predicted to reach the Remediation Goal of 0.05 ppm PCBs (wet weight) in fish tissue by 2039 in River Section 1, by 2041 in River Section 2, and by 2025 in River Section 3. Under MNA with an upstream load of zero Tri+ PCBs, it would take until 2063 in River Section 1, 2061 in River Section 2, and 2032 in River Section 3. This emphasizes the impact of reducing the upstream PCB load to the greatest extent possible, as well as the need for remediation.

Calculations of human health and ecological risk corresponding to the model simulations discussed in this paper are found in the White Paper – Human Health and Ecological Risk Reduction under Phased Implementation. The human health risk calculations used species-weighted concentrations in the estimates of exposure-point concentrations. The species-weighted predicted fish body burdens (47 percent largemouth bass, 44 percent brown bullhead, and 9 percent yellow perch) are presented in Figures 363150-18 through 363150-20. These figures show that the differences in predicted species-weighted concentrations are comparable to the differences in the individual species.

A comparison of short-term risk between the selected remedy scenarios found essentially no differences between the selected-remedy scenarios. See White Paper – Human Health and Ecological Risk Reduction under Phased Implementation.

CONCLUSIONS

Comments received expressed concerns that some of the assumptions used to conduct forecast simulations with the EPA fate and transport (HUDTOX) and bioaccumulation (FISHRAND) models may affect comparisons among various remedial scenarios. Several additional HUDTOX and FISHRAND forecast simulations were conducted to address these concerns, the results of which:

- Demonstrated that phasing of the implementation schedule for the selected remedy (REM-3/10/Select) by extending it one year does not significantly affect the predicted long-term trajectories for Tri+ PCB levels in the water, sediments, and fish of the Upper Hudson River.

- Demonstrated that elimination of upstream Tri+ PCB loading does not diminish the relative separation between predictions for the selected remedy and MNA scenarios in the water column, surficial sediments, or fish.

- Clarified the significance of sediment mixing processes and sediment-water interactions in controlling long-term Tri+ PCB levels in surficial sediments, water, and fish as external sources of PCBs entering the river are reduced or perhaps even eliminated.

- Showed that by reducing the upstream Tri+ PCB load to zero, the Remediation Goal of 0.05 ppm PCB in fish tissue (wet weight) will be reached in all river sections during the modeling time frame. The Remediation Goal will be reached much sooner for the selected remedy than for Monitored Natural Attenuation.

- Demonstrated that while dredging-induced resuspension of sediment will likely increase Tri+ PCB levels in the water column (as measured by either concentration or load passing a given location) and in fish, the impacts will largely be confined to the years during which the selected remedy is implemented, regardless of whether a 0.13 percent or a 2.5 percent loss rate is applied. Impacts from dredging-induced resuspension (at either rate) are also predicted to occur in downstream surficial sediments, but the increases in concentration are generally small relative to existing Tri+ PCB contamination levels.

- Showed that application of the 2.5 percent dredging-induced loss rate results in much greater increases in Tri+ PCB levels in water and fish during the selected remedy period than does EPA's loss rate estimate of 0.13 percent. However, EPA believes that the 2.5 percent loss rate is unrealistically high and that the 0.13 percent loss rate is a justifiably conservative estimate (Appendix E.6 of the FS; White Paper – Resuspension of PCBs during Dredging).

The additional HUDTOX and FISHRAND model forecast simulations presented in this paper provide a quantitative assessment of important technical issues raised through comments on the analyses presented in the FS.

REFERENCES

USEPA. 2000. Phase 2 Report, Further Site Characterization and Analysis, Volume 2D, Revised Baseline Modeling Report (RBMR), Hudson River PCBs Reassessment RI/FS. Prepared for USEPA and US Army Corps of Engineers Kansas City District by Limno-Tech, Inc., Menzie-Cura and Associates, Inc. and TetraTech, Inc. January.

Application of the Depth of Scour Model (DOSM) in the Thompson Island Pool for Alternative Flooding Assumptions

407426

WHITE PAPER – APPLICATION OF THE DEPTH-OF-SCOUR MODEL (DOSM) IN THE THOMPSON ISLAND POOL FOR ALTERNATIVE FLOODING ASSUMPTIONS

(ID 407426)

ABSTRACT

An important question in the Hudson River PCBs Reassessment was whether there are contaminated sediments now buried that are likely to become "reactivated" following a major flood, possibly resulting in an increase in contamination of the fish population. To address this question, a Depth-of-Scour Model (DOSM) was developed to provide estimates of sediment erodibility in response to large floods. In the Revised Baseline Modeling Report (RBMR) (USEPA, 2000a), the DOSM was applied to a 100-year flood peak flow estimated to be 47,330 cubic feet per second (cfs), based on available historical data. Concern was expressed that this flow estimate could be too low and that a more appropriate value for a 100-year peak flow was 61,835 cfs. To address this concern, the DOSM application was repeated for this new estimated upper-limit peak flow.

The results of the DOSM reapplication indicate that the average flow and bottom shear stress for the upper-limit flood peak is greater than for the previous flow estimate. The resulting average erosion depth in cohesive sediments for the upper-limit flood peak is more than twice the estimate for the lower flow, but is still less than 1 cm (0.719 cm versus 0.317 cm). Although a flow of 61,835 cfs could erode between 120 to 650 kg of Total PCBs, this mass represents only 0.2 to 3 percent of the mass inventory estimated to reside in the cohesive sediments of the Thompson Island Pool (TI Pool).

The re-application of the DOSM for the upper-limit estimate of 61,835 cfs for the 100-year flood peak flow found that a major flood would not scour a large portion of the PCB inventory in the sediment of the TI Pool. The upper-limit flood peak would result in a mass of PCBs resuspended that is only slightly higher than the amount expected during typical annual high flow events.

INTRODUCTION

One of the principal questions in the RBMR (USEPA 2000a) was whether there are contaminated sediments now buried that are likely to become "reactivated" following a major flood, possibly resulting in an increase in contamination of the fish population. To address this question, the Depth of Scour Model (DOSM) was developed to provide spatially refined information on sediment erodibility in response to high-flow events, such as a 100-year flood. The DOSM is a two-dimensional, sediment-erosion model that was applied to the TI Pool. The DOSM is linked with a hydrodynamic model (RMA-2V) that predicts the two-dimension pattern of flow velocities and water depths within the TI Pool, providing the information necessary to compute bottom-shear stress (force of the water acting on the sediment surface) during high flows. In the RBMR, the DOSM was applied for a 100-year flood peak-flow value estimated to be 47,330 cfs at Fort Edward. This estimate was based on an analysis of available historical flow records and operational practices for the Sacandaga Reservoir, as presented in the Appendix of the Baseline Modeling Report Responsiveness Summary (USEPA, 2000b).

The New York State Department of Environmental Conservation (NYSDEC) raised questions regarding uncertainties in development of the 100-year flood peak-flow value used in the RBMR. Concern was also expressed regarding the potential for the NYSDEC relicensing agreement with Orion Power in operating the Sacandaga Reservoir to increase maximum Hudson River flood flows. In order to evaluate these concerns, an estimated upper limit for the 100-year flood peak flow of 61,835 cfs was developed for NYSDEC (NYSDEC, 2001). To estimate potential "reactivation" of buried sediments under this new condition, the RMA-2V and DOSM were reapplied for a peak flow of 61,835 cfs, and results were compared with the application of these same models for the peak flow of 47,330 cfs in the RBMR.

The following sections in this paper present the results of:

- Reapplication of the RMA-2V model for a flow of 61,835 cfs.
- Reapplication of the DOSM for cohesive sediment areas at bottom-shear stresses corresponding to a flow of 61,835 cfs.
- Comparison of RMA-2V and DOSM results for the RBMR flow of 47,330 cfs and the alternate flow of 61,835 cfs for all TI Pool cohesive sediment areas and at the locations of five USEPA high-resolution core sites.

HYDRODYNAMIC MODEL REAPPLICATION

The development and application of the RMA-2V hydrodynamic model is fully described in Chapter 3 of the RBMR (USEPA, 2000a). The RMA-2V model is a two-dimensional finite-element model capable of simulating dynamic fluid flow in the horizontal plane. The version of the model that was applied was RMA-2 Version 4.35, developed in August 1995 by the US Army Corps of Engineers.

The upper limit estimate of the 100-year flood flow evaluated in this paper is based on the assumption that the Sacandaga Reservoir is full during an extreme flood and will not act as an effective flood-control structure (NYSDEC, 2001). In effect, this assumption infers that the Sacandaga drainage basin will behave similarly to the Upper Hudson River during an extreme event, and that Hudson River flows at Hadley may be prorated (based on drainage area differences) to estimate an upper limit for the 100-year flood flow at Fort Edward.

Using Hudson River at Hadley flow data collected from 1922 to 1998, Bopp (NYSDEC, 2001) estimated an upper limit 100-year flood flow of 61,835 cfs (assuming a log-Pearson Type III distribution). EPA does not endorse this estimated upper-limit flow as being appropriate for evaluating the potential effects of PCB remobilization from the sediments due to a 100-year flood event, since the probability of maximum releases from Sacandaga Reservoir coinciding with a 100-year flood flow in the mainstem Hudson River is less than one. Thus, the actual recurrence interval for the assumptions that were applied to generate the upper limit flow is greater than 100 years. However, EPA concurs that it is reasonable and appropriate to examine the estimated sediment scour and PCB remobilization response for this event, since there is significant uncertainty in estimating maximum flood flows based on data for a period of record that is significantly less than the recurrence interval.

The two principal boundary conditions for the model are the upstream flow, designated as 61,835 cfs for the upper limit 100-year flood peak flow, and a downstream water surface elevation. The downstream water surface elevation was estimated based on water surface elevation and flow data from NYSDOT barge canal data for Gage 118, along with the FEMA-estimated 100-year flow of 52,400 cfs and downstream elevation of 126.5 feet (Zimmie, 1985; FEMA, 1982). A second-order polynomial curve was fit to this data and extrapolated out to a flow of 61,835 cfs to estimate the corresponding water surface elevation of 127.6 feet.

Comparative results from application of the RMA-2V hydrodynamic model for average velocity, depth, and shear stress for cohesive and non-cohesive sediment areas are presented in Table 407426-1. As illustrated in the table, the average velocities and water depths are higher for the 61,835 cfs peak flow. Even with the increased water depth, the velocities are elevated such that the resulting average shear stresses for the 61,835 cfs flow are also higher than for the 47,330 cfs flow. Also, for both peak flows, the average values of all three parameters are higher for the non-cohesive areas than for the cohesive areas.

DEPTH-OF-SCOUR MODEL REAPPLICATION

The development and application of the DOSM is fully described in the RBMR (USEPA, 2000a). The DOSM calculates cohesive sediment scour on the basis of site-specific measurements of resuspension properties for TI Pool cohesive sediments. Model inputs include sediment dry-bulk density (*i.e.*, grams of dry solids per cm^3 of sediment) and flow-induced bottom-shear stress. The model outputs mass of solids eroded by the shear stress and the depth of erosion. Given a concentration of PCBs in the sediment solids, the mass of PCBs eroded can also be calculated. The DOSM was used to estimate an average poolwide depth-of-scour and to estimate sediment erosion at five locations where EPA collected high-resolution sediment cores in 1992.

The DOSM results are given in terms of probabilistic estimates. These probabilities range from 0 to 100 percent, and represent the likelihood that less than or equal to a given amount of solids will be eroded by a given shear stress. From these calculated probabilities, values corresponding to the 5^{th} percentile, the 50^{th} percentile (median), and the 95^{th} percentile erosion depths are reported, as well as the mean (arithmetic average) erosion depth. DOSM probabilities of erosion reflect the range of variability actually observed in site-specific measurements of erosion made using annular flume and shaker devices (HydroQual, 1995).

Reapplication of the DOSM was done in the same manner as the RBMR application, with the only change being the new bottom-shear stresses corresponding to the upper-limit peak flow of 61,835 cfs. The same map of cohesive sediment areas in TI Pool was used for both applications. For poolwide estimates, these cohesive sediment areas were subdivided into polygons of constant shear stress and dry-bulk density by intersecting coverages (maps) for these properties in a Geographical Information System (GIS). The Monte Carlo technique was employed to calculate the depth of scour and the solids-mass scoured for each such polygon, as a function of randomly varied parameters in the resuspension equation. The extent of random variation was set to reflect the underlying variability in the observed data upon which the resuspension equation is based.

Poolwide results for total mass scoured were obtained by summing the results at all locations, while an area-weighted average was calculated as the mean depth-of-scour. The calculation was repeated 3,000 times to get a valid statistical distribution of results. Monte Carlo calculations were performed with the Crystal Ball 2000® computer program (Decisioneering, Inc., 2001). Results at specific high-resolution coring sites made direct use of the DOSM output, without Monte Carlo analysis. Depths of scour at these sites made use of estimates of dry-bulk density interpolated from the reported core densities at depth.

In addition to the assumptions in the RBMR for application of the DOSM to the original peak flow of 47,330 cfs, the following factors must be given consideration when interpreting the predictions generated by the DOSM to the upper-limit peak flow of 61,835 cfs. First, the 61,835 cfs flow extends the hydrodynamic model application further beyond the available calibration data, resulting in increased uncertainty of computed bottom-shear stress values used to drive the DOSM.

Second, the higher shear-stress values predicted in this model reapplication increase the degree of extrapolation from the site-specific experimental measurements used to derive the sediment resuspension relationships in the DOSM, resulting in additional uncertainty beyond that associated with the hydrodynamic model application.

Finally, the passage of time between the RBMR application and this reapplication results in increased uncertainty about TI Pool physical and chemical properties.

Apart from the above factors, a new source of uncertainty has been discovered that affects the DOSM application to both the original peak flow of 47,330 cfs in the RBMR and the upper-limit peak flow of 61,835 cfs. The sediment resuspension measurements conducted by GE with Upper Hudson River sediments used a shaker device and an annular flume (HydroQual, Inc., 1995). Experimental artifacts have been discovered with these techniques (Lick *et al.*, 1998; Jones and Lick, 1999), and it is now believed that resuspension data acquired with these devices may underestimate the potential for sediment scour, especially under high-shear stresses. In recent years, newer techniques have been developed to measure sediment resuspension (McNeil *et al.*, 1996; Jepsen *et al.*, 1997; Lick *et al.*, 1998; Jones and Lick, 1999; Ravens and Gschwend, 1999); however, transport and fate models capable of using experimental data from these techniques remain the subject of ongoing research (*e.g.*, Jones, 2000) and have not yet been fully tested. Furthermore, no measurements using these newer techniques have been conducted with TI Pool sediments. Consequently, given the absence of additional site-specific data and a compatible modeling framework, it is currently not possible to accurately assess the significance of these new findings with regard to predicting flood-driven erosion of TI Pool cohesive sediments.

COMPARISON OF MODEL RESULTS

At the high-resolution core locations, model results are compared on the basis of predicted erosion depth, which in turn depends on predicted flow-induced bottom shear stress. For the poolwide Monte Carlo modeling, the applications are compared on the basis of erosion depth and mass of solids scoured. Using the same approach as in the original DOSM application, PCB erosion is estimated using an assumed constant cohesive sediment PCB concentration. Two different PCB concentrations for the surface sediment are used, based on different sets of data. The first one is based on 1984 data, using Thiessen polygons and side-scan sonar results. On this basis, the area-weighted average Tri+ PCB concentration is 51.49 mg/kg (White Paper – Sediment PCB Inventory Estimates). For purposes of the calculations presented in this paper, this Tri+ PCB sediment concentration has been rounded off to two significant digits, or 51 mg/kg, in acknowledgment of the uncertainty associated in determining this value. This number (51 mg/kg) is about 17 percent higher than the number used in previous calculations (43.7 mg/kg) that were presented in the Baseline Modeling Responsiveness Summary Report [RBMR RS] (USEPA, 2000b). The new value is based on a revised analysis of the 1984 data (White Paper – Sediment PCB Inventory Estimates). The estimates of PCB mass eroded using the 1984 data are thought to be conservative – that is, the poolwide average PCB concentration in the cohesive surface sediments is now lower than in 1984, thus the present estimates of PCB mass eroded are higher than would actually occur for a given mass of solids eroded.

The second PCB concentration is based on the more recent data collected in 1991 by GE. Using these data, the TI Pool surface sediment Tri+ PCB concentration is 16 mg/kg and the Total PCB concentration is 32 mg/kg. A summary of the cohesive surficial sediment concentrations for Tri+ and Total PCBs is listed in Table 407426-2.

Table 407426-3 lists estimated shear stresses at the high-resolution core locations being compared. Despite the higher flow in the model reapplication, estimated shear stresses decreased at two of the five locations, HR-25 and HR-26. The reason for this relates to greater submergence of the floodplain in the vicinity of these near-shore core locations for the higher flow. In effect, a significant portion of the increased flow is predicted to move over the floodplain, resulting in lower local velocities and, subsequently, lower bottom-shear stresses in some areas.

In addition, the side channel and floodplain along the eastern portion of Rogers Island are predicted to have increases in storage under this higher flow condition, resulting in greater depths and somewhat lower velocities within the channel as compared to the lower flow of 47,330 cfs in the RBMR. Bottom-shear stress depends on both depth and velocity, and it decreases for greater depths and smaller velocities. Given the hydraulic behavior in this region of the TI Pool, the decreases in predicted bottom-shear stresses for cores HR-25 and HR-26 under a higher flow condition are plausible. However, as noted above, the uncertainty in the hydrodynamic model predictions likely increases with the further extrapolation beyond available data for the downstream boundary condition and for model calibration.

Table 407426-4 compares depth to peak PCB concentration at the high-resolution core locations with low (5^{th} percentile), median (50^{th} percentile), and high (95^{th} percentile) estimates of erosion depth for the original RBMR application (47,330 cfs) and the upper-limit peak flow (61,835 cfs).

Table 407426-5 compares expected values (*i.e.*, arithmetic averages of the 3,000 Monte Carlo iterations) for the original RBMR application (47,330 cfs) and the upper-limit peak flow (61,835 cfs). Tables 407426-6 and 407426-7 list the expected values of Tri+ and Total PCBs eroded in comparison to the existing PCB inventory in the cohesive sediment of the TI Pool.

Although the estimated mass of Tri+ PCB resuspended for the upper-limit peak flow (60 and 190 kg based on 1991 and 1984 data, respectively) is greater than that resuspended for the lower flow used in the RBMR analysis (30 and 89 kg based on 1991 and 1984 data, respectively), these values are small relative to the estimated Tri+ PCB mass reservoir in the TI Pool. The resuspended Tri+ PCB mass due to the 100-year peak flow is only 0.3 to 2.4 percent of the Tri+ PCB inventory in the cohesive sediment of the TI Pool. The previous estimated mass of Tri+ PCB resuspended reported in the RBMR RS – 76 kg – falls in the range of the revised estimated mass of 30 to 89 kg.

The estimated mass of Total PCB for the upper-limit peak flow ranges from 120 to 650 kg based on 1991 and 1984 data, respectively. The estimated mass of Total PCB for the lower 100-year flood peak flow (47,330 cfs) ranges from 60 kg (using 1991 sediment data) to 300 kg (using 1984 sediment data). Similar to the Tri+ PCB, the estimated mass of Total PCBs resuspended during the 100-year flood is small (0.3 to 3 percent) relative to the estimated Total PCB mass inventory in the cohesive sediment of the TI Pool.

Two factors suggest that use of 1984 sediment PCB concentrations would overestimate Total PCB mass scour. First, a poolwide average of less than one cm of surface sediment is predicted to scour in the TI Pool. Application of a sediment PCB concentration that represents the upper 30 cm of sediment would include highly contaminated subsurface layers that are not predicted to scour during a 100-year peak flow.

Second, surface-sediment PCB concentrations have declined substantially since 1984 (USEPA, 2000a; Appendix D.1 of the FS report) and do not represent present conditions. Therefore, PCB mass scour calculations based on the 1984 sediment data should be considered high-end estimates. The 1991 sediment data are likely more representative for estimating PCB remobilization based on DOSM-predicted scour depths, since these data are more recent and represent conditions in the top five cm of sediment. However, the average PCB concentrations based on these data incorporate composite samples that represent both cohesive and non-cohesive sediment areas. Non-cohesive sediment areas generally have lower levels of PCB contamination than cohesive areas; therefore, PCB mass scour calculations based on an average of the 1991 sediment data may be characterized as low-end estimates. In summary, the 1984-based PCB mass scour estimates are larger than the 1991-based estimates due both to likely declines in concentration and the substantial difference in the vertical resolution of surficial sediment that each of these datasets are capable of representing (*i.e.*, 5 cm for 1991 versus 30 cm for 1984).

Although the DOSM predicts substantially greater scour of solids mass from cohesive sediments within the TI Pool for the upper-limit peak flow of 61,835 cfs, the poolwide average scour depth is still less than one cm. Additionally, the 95th percentile maximum scour depth is less than the depth of peak PCB concentrations at four of the five high-resolution core locations that were analyzed for the RBMR. The fifth location, HR-25, has a predicted 95th percentile scour depth that is below the measured peak PCB concentration for both the upper-limit peak flow (61,835 cfs) and the lower peak flow (47,330 cfs) used for the RBMR analysis of the 100-year flood. It should be noted that the peak PCB contamination in this core is just five cm below the sediment-water interface, suggesting that this location is subject to periodic scour, as predicted by the

DOSM in the RBMR analyses and for the reapplication of the model to an upper-limit 100-year peak flow.

The DOSM reapplication for the estimated upper-limit 100-year flood peak flow of 61,835 cfs does not alter the major findings that were presented in the RBMR. The major finding related to the 100-year flood analyses was that "results of the 100-year peak flow simulation show that a flood of this magnitude would result in only a small additional increase in sediment erosion beyond what might be expected for a reasonable range of annual peak flows."

CONCLUSION

The results of the DOSM reapplication indicate that:

- Average bottom shear stress on cohesive sediments increases by 27 percent for the upper-limit flood peak versus the lower flow analyzed in the RBMR (21.3 dynes/cm^2 versus 16.8 dynes/cm^2).

- Average erosion depth in cohesive sediments is approximately 2.27 times greater for the upper-limit flood peak flow than for the lower flow analyzed in the RBMR (0.719 cm versus 0.317 cm).

- The 95th percentile maximum scour depth for the upper-limit 100-year flood peak flow is less than the depth of peak PCB concentration at the same four out of five high-resolution sediment core locations in the TI Pool that were examined for the lower flow in the RBMR.

- Using 1991 surficial sediment data representing average PCB concentrations in the top five cm of the sediments across the entire TI Pool, a low-end estimate of PCB mass scour can be made. This estimate indicates that an additional 30 kg of Tri+ PCBs would be scoured for the upper-limit flood peak flow than for the lower flow analyzed in the RBMR (60 kg versus 30 kg). On a Total PCB-basis the increase in estimated scour for the upper-limit flood peak would be 60 kg (120 kg versus 60 kg).

- Using 1984 sediment data representing average PCB concentrations in the top 30 cm of the cohesive sediments in TI Pool, a high-end estimate of PCB mass scour can be made. This high end estimate indicates that an additional 101 kg of Tri+ PCBs would be scoured for the upper limit flood peak flow than for the lower flow analyzed in the RBMR (190 kg versus 89 kg). On a Total PCB-basis the increase in estimated scour for the upper limit flood peak would be 350 kg (650 kg versus 300 kg).

- The predicted mass of PCBs resuspended, on either a Tri+ or Total PCB basis, for the upper-limit peak flow is small, ranging from 0.2 to 3 percent of the mass inventory estimated to reside in the cohesive sediments of the TI Pool.

In summary, the major RBMR findings related to the assessment of flood-induced sediment PCB remobilization are not significantly altered based on the results from reapplication of the DOSM to the upper-limit estimate of 61,835 cfs for the 100-year flood peak flow in the TI Pool. Based on a conservative estimate of the mass of PCBs resuspended under a 100-year peak flow, the 100-year flow will result in only a slightly larger amount of PCBs resuspended than may be expected during typical annual high-flow events.

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Trends in PCB Concentrations in Fish in the Upper Hudson River
312627

WHITE PAPER – TRENDS IN PCB CONCENTRATIONS IN FISH IN THE UPPER HUDSON RIVER

(ID 312627)

ABSTRACT

EPA presented an analysis of trends in PCB concentrations in fish in the Upper Hudson River in Appendix D.1 of the FS. That analysis demonstrated that the decline in fish tissue concentrations observed in the 1970s and 1980s had tapered off, with a period of increase (accompanying the increased PCB releases at the GE Hudson Falls facility) followed by very slow rates of decline in the 1990s. Even after the releases at the Hudson Falls facility were largely controlled (*i.e.*, since 1995), fish tissue PCB concentrations in the Thompson Island Pool (TI Pool) and near Stillwater appeared to remain approximately constant, at levels well above acceptable concentrations.

In addition, the monitoring results suggest the possibility that the EPA and GE models predict too rapid a rate of decline in fish tissue PCB concentrations under current conditions. Since the FS was written, additional, more-recent fish monitoring data have become available, requiring an update of EPA's analysis.

This white paper contains two sections. The first section is a revised version of Appendix D.1, Section 3.1, of the FS, which presents EPA's analysis of trends in PCB concentrations in fish in the Upper Hudson River. This update is needed to incorporate the new data, most notably the year 2000 fish sampling results, which were not available at the time the FS was prepared. Including the newer data reinforces and confirms the conclusions presented in the FS.

GE has provided an alternative analysis of trends in PCB concentrations in fish, submitted as Appendix H to GE's comments. GE questioned a number of the assumptions used by EPA in its analysis, and presented conclusions regarding recent trends in PCB concentrations in fish that differ from those of EPA.

The second section of this white paper provides a detailed commentary on GE's alternative analysis of trends. The technical analyses conducted by GE are, in general, valid; however, the conclusions that are drawn from these analyses are not supported by the data. In fact, GE's reanalysis of fish concentration trends does not contradict EPA's analysis. Instead, the GE analysis confirms the observation of a lack of clear declining trends in PCB concentration in recent years for most of the fish species sampled.

INTRODUCTION

Data on PCB concentrations in fish are important for two reasons: First, PCB concentrations in fish tissue represent the primary route of exposure to humans in the Hudson River. Second, because PCBs accumulate in fatty tissue, observations of PCBs in fish integrate exposure over time and provide an important indicator of trends in river conditions that is not strongly influenced by transient water-column concentrations.

The FS examined PCB concentrations in fish through the 1998 data, which were the most recent analytical results available at the time. Since release of the FS, NYSDEC has provided the results of the 1999 and 2000 fish data collections. This white paper examines recent PCB trends in fish up through 2000. It also evaluates an alternative analysis of fish tissue concentration trends provided in comments by GE.

EPA'S ANALYSIS OF PCB CONCENTRATION TRENDS IN FISH

Concentration trends in fish potentially provide one of the most rigorous tests of the joint performance of the HUDTOX and FISHRAND models, as the fish response integrates over many geochemical and biological processes. Long time-series of concentrations in various species at multiple locations are available from NYSDEC, and these biotic concentrations should integrate or smooth out short-term or spatial variability seen in other media.

Several caveats should, however, be noted. Most importantly, changes in analytical methods over time may serve to introduce apparent step changes into the fish concentration record. This problem is reduced by attempts to convert the NYSDEC data to a consistent Tri+ PCB basis, although the conversions themselves are subject to uncertainty. In addition, concentrations in fish in a given year may be influenced by factors such as weather, food availability, and the distribution of age and sex in a given sample.

It is also important to remember that calibration of the FISHRAND model was conducted using environmental concentration estimates from HUDTOX as the forcing function. Thus, any shortcomings in HUDTOX will also propagate into the FISHRAND calibration. Trends in brown bullhead should generally follow HUDTOX-predicted trends in surface sediment concentration, while trends in pumpkinseed should generally follow predicted trends in water column concentration (particularly summer concentrations), and largemouth bass should depend on both sediment and water (Table 6-7 in the Revised Baseline Modeling Report, or RBMR; USEPA, 2000).

Concentration trends in fish are evaluated here as lipid-based concentrations, on the assumption that conversion to a lipid basis better reflects actual uptake processes and helps to smooth out some of the year-to-year and sample-to-sample variability.

A comparison of FISHRAND median predictions to observed (corrected) Tri+ PCB data in fish lipid is shown for three species in the lower TI Pool and the Stillwater reach in Figures 312627-1 and 312627-2. These results use actual (observed) upstream boundary conditions for the 1998–1999 validation period, while model results for 2000 are from the model forecast of future conditions conducted for the Revised Baseline Modeling Report (USEPA, 2000).

Figure 312627-1 shows results for fish collected by NYSDEC near Griffin Island at RM 189 in the TI Pool. While the general fit seems acceptable, there are some discrepancies between model and data. For largemouth bass, the model appears to under-predict recent concentrations (*i.e.*, 1998 through 2000). High concentrations observed in 1990–1991 were also not predicted by the model.

For brown bullhead in the TI Pool, the general model trend appears to be a better fit than that for largemouth bass. It is noted, however, that the model predicts a gradual decreasing trend from 1995–2000, while the data show what appear to be nearly constant concentrations, with a slight increase in 1999.

As EPA's FISHRAND bioaccumulation modeling indicates a strong relationship between brown bullhead tissue PCB concentrations and concentrations in surface sediment, this result suggests that the modeled trend in surface sediment concentrations for this period might differ from the trend in sediment-driven exposure experienced by the sampled fish. This could occur either because the modeled trend is incorrect or because the fish's exposure occurs at a local spatial scale that is smaller than that simulated by the model, in which sediment concentration trends differ from the reach-averaged trend. Alternatively, the trend in the observed fish data may be obscured by random variability in the sample results.

Pumpkinseed body burdens should provide a diagnostic of model ability to reproduce summer water-column concentration trends. The FISHRAND model fits the general trend for pumpkinseed in the TI Pool. Notable here, however, is the failure to predict elevated concentrations in 1989 – which could, in turn, be a source of the elevated concentrations seen in largemouth bass in 1990 and 1991. The year 1989 is one in which data to characterize the upstream boundary loads are very sparse, which could indicate a failure to capture pulse loading from upstream and consequent underestimation of summer water-column concentrations.

The 1995–2000 data from the TI Pool suggest that the models could be predicting a rate of decline in fish tissue concentration that is more rapid than seen in the environment since the upstream source was largely controlled. Small changes in trend at this end of the distribution could have large effects on the rate of natural decline during the forecast period.

The interpretation of the TI Pool results must be made with caution, due to the locations used for sampling. The fall samples of yearling pumpkinseed are generally collected on the east side of the main channel, opposite Griffin Island and just south of *Hot Spot* 14. The spring samples of largemouth bass and brown bullhead are, however, collected in the backwater channel *behind* Griffin Island (because this is an area in which the bass congregate in the spring). Because this channel is somewhat isolated from the main river, the relevance of trends in these data to overall conditions in the lower TI Pool is uncertain. (During remedial design, EPA will work with the State and federal natural-resource trustees to design a comprehensive fish-monitoring program to evaluate the effectiveness of the remedy, and may modify the sites used in the existing fish-monitoring program.)

The model and data for the Stillwater reach (Figure 312627-2) are generally in closer agreement for brown bullhead and largemouth bass in the 1990s relative to the TI Pool. The pumpkinseed calibration misses the error bars on observed lipid-based concentrations in most years up through 1993, which could indicate that summer water-column concentrations are different than those predicted by HUDTOX.

More notable at this location is a divergence between the model and observations for the period between 1977 and 1982. For all three species, the data suggest that initial concentrations were higher, with a more rapid decline, than is indicated by the FISHRAND model. For this period, the data to constrain water-column concentrations in the modeling are very sparse. There are also significant uncertainties regarding the interpretation of analytical methods for the earlier data.

Table 312627-1 summarizes half-life data for the three species discussed above, plus limited data for yellow perch. The consistent Tri+ data includes both Aroclor-based data reported by NYSDEC and direct estimates of Tri+ from homologue-based analyses conducted for GE by Northeast Analytical (NEA). For 1998 through 2000, only the homologue results from NEA are used, as provided in NYSDEC's database for 1998–1999, and in GE's database for 2000. For this period, an analysis for conversion of the Aroclor data to consistent Tri+ results is not available.

Across the period 1985-2000, trends in model and data (expressed as consistent Tri+ PCBs) are generally quite close. This reflects the fact that FISHRAND is calibrated to data that span this period, and the general fit of the model is quite good. For the 1985-1991 period of declining concentrations, model and data are again close in the Stillwater reach; however, the data-based trends in the TI Pool show both largemouth bass and pumpkinseed increasing, whereas the model predicts declines.

In general, the model does a good job of reproducing observed fish concentrations over the period of record when examined as an annualized lipid-based average concentration. However, the model does not seem to reproduce the trend in observed concentrations since 1995.

For the 1995-2000 period following substantial control of the upstream source, trends in the model and data appear to diverge. The model predicts continuing steady declines in fish concentrations in the TI Pool, but the data show either increasing or very slowly decreasing concentrations in the pool. For brown bullhead, the data show essentially no downward trend for the period 1995-2000, while the decline predicted by the model is approximated by a half-life of 3.8 years. The largemouth bass concentrations have a data-based half-life of 11.6 years versus a model estimate of 3.6 years. The rate of decline predicted by the model for the Stillwater reach also appears to be more rapid than that observed for brown bullhead and largemouth bass.

In evaluating these trends it is important to keep in mind that the observed data are variable and subject to uncertainty. Reported trends are based on annual means. The 95-percent confidence limits on the observed means in the TI Pool for 1995-2000 are consistent with half-lives as short as 5.4 years for brown bullhead and as short as 4.0 years for largemouth bass.

The FISHRAND output provides 1995-2000 half-lives that are outside (shorter than) the range estimated from observed data for both brown bullhead and largemouth bass, suggesting that the model did not accurately predict realized concentrations; however, the cause of this is uncertain at this time.

COMMENTARY ON GE'S APPENDIX H: TRENDS IN PCB LEVELS IN HUDSON RIVER FISH

Appendix H to GE's comments is a document entitled "Trends in PCB Levels in Hudson River Fish." This document presents conclusions that differ in some respects from EPA's analysis. Most importantly, the document contends that fish in the TI Pool and at Stillwater exhibit a continuing decline in concentration that is consistent with rates of decline in fish and water and sediment exposure concentrations predicted by the GE model. This section of the white paper examines the reasons for the difference in conclusions between GE’s and EPA’s analyses, as revised to incorporate the year 2000 data.

GE’s analysis is based on essentially the same data as used by EPA. The data have, however, been subjected to selective filtering by GE, some of which appears to be inappropriate. In addition, GE sets up a false comparison to EPA’s effort that reflects a misunderstanding of the purpose of the analyses presented in Appendix D of the FS.

Introduction

In Section 1 of Appendix H, GE states that EPA "claims that the PCB concentrations in Upper Hudson River fish have not declined in recent years or are declining very slowly," and that "EPA's conclusions about these trends is based on an unsound analysis."

EPA neither contends nor believes that PCB concentrations in Hudson River fish have stopped declining – when viewed at the scale of reach averages. EPA's fate and transport and bioaccumulation models predict a continuing decline, as documented in the FS. However, reach-averaged trends do not necessarily hold across all individual locations and it is, therefore, essential to point out that fish concentrations have not exhibited a consistent decline in recent years for a number of species and at a number of the locations historically sampled by NYSDEC.

The fact that a decline is not verified in the monitoring data is reason for concern. As described in Appendix D.1 of the FS, this lack of decline may reflect localized sediment-exposure concentrations that are not declining at the reach-averaged rates predicted by the model.

The "bounding calculation" developed by EPA is intended to provide a reasonable estimate of elevated fish tissue concentrations in localized areas that exhibit continuing, slowly declining sediment-exposure concentrations. It is not a prediction of average biotic concentrations in the river as a whole. However, it should be noted that the NYSDEC sampling locations in the TI Pool and at Stillwater were selected because they are areas in which fish are plentiful. Thus, significant populations of fish are associated with areas of the Hudson River in which the rate of decline in PCB concentration is less than is predicted by the models at the reach-averaged scale. This is an important factor that should be taken into consideration as part of the remedial decision to use the model predictions alone, or to use several analytical tools and factual databases to evaluate the need for remediation of contaminated sediment.

Data Interpretation

Section 2 of GE’s Appendix H contains a detailed discussion of "factors that affect interpretation of the data." GE's interpretation of these factors is reviewed in detail.

Fish Lipid Content

Because PCBs are more soluble in oils than in water, they are stored primarily within fats, or lipids, in fish. EPA agrees with GE's conclusion that the fish PCB concentration data should be evaluated on a lipid basis to remove potential bias and uncertainty in PCB concentrations due to changes in lipid content. That is why EPA's analysis was conducted on a lipid basis. EPA also agrees that the relationship between PCB concentration and lipid content is likely to be somewhat nonlinear and may be due to the interaction between elimination rate and growth dilution.

Fish Weight and Age

EPA agrees that PCB concentrations in fish are likely to increase with age. Unfortunately, age estimates are not available for the vast majority of Hudson River fish samples. Available measures of length and weight are imprecise surrogates for age.

Potential dependence of fish PCB body burden on weight and/or length was evaluated in detail during the development of the bivariate BAF model (USEPA, 2000), but no statistically significant explanatory power was found for these variables. GE presents lipid-based PCB concentration versus weight in Appendix H, Figure 2.2-2, and claims to find a relationship between weight and PCB concentration. The figure does not support this statement. In fact, the large majority of the weight-class PCB concentrations have 95-percent confidence limits that overlap one another. There is possibly an increasing trend in lipid-normalized PCBs across the several lightest-weight classes in largemouth bass and brown bullhead, but no clear trend in other weight classes.

GE also claims that variance in PCB concentration increases with weight, but this does not seem to be correct: The larger uncertainty associated with the heavier weight classes instead seems to reflect smaller sample size.

In sum, the data do not support GE's assertion "that body weight impacts bioaccumulation" to an extent sufficient to require subsetting the data.

Nevertheless, GE chose to analyze trends in brown bullhead and largemouth bass based only on data from a restricted weight range. This excludes 24 percent of the data for largemouth bass and 20 percent of the data for brown bullhead. No pumpkinseed data were excluded, as NYSDEC samples almost exclusively yearling pumpkinseed. EPA believes that this paring of the data is not justified, based on available evidence in the monitoring.

Extreme Values

GE further limited the data set by excluding extreme value outliers that deviated from local mean conditions. Given the relatively small size of fish samples (typically 20 analyses per species/location/year), one should be extremely careful in rejecting outliers as anomalous or unrepresentative.

In fact, because the Hudson River is not a homogenous environment, and each individual fish sampled may have a somewhat different history of exposure, it can be argued that no outliers should be rejected unless there is strong evidence of analytical error. EPA concluded that there was not compelling evidence to reject any individual data points, and therefore worked with the complete data set.

Variation in Exposure Location

In Section 2.4 of Appendix H, GE states, "Fish may be exposed to sediment and water column PCB concentrations characteristic of some sub-reach-scale location…" This statement is worth highlighting, as, in direct comments, GE has stated that a focus on the sub-reach scale "lacks any basis in fact or reason."

GE's Appendix H also misrepresents the purposes of the bounding calculation, stating that "EPA inappropriately uses these brown bullhead data to estimate a rate of recovery for the cohesive sediments of the entire pool." This is incorrect: EPA uses the NYSDEC TI Pool brown bullhead data to make inferences about trends at fine-grained sediment locations similar to the NYSDEC sampling location, not over the whole TI Pool.

Laboratory Methods

GE provides an analysis of split-sample comparisons between NYSDEC contract-laboratory results and capillary-column analysis by NEA, including new results for the Year 2000 data. EPA generally agrees with GE's analysis regarding uncertainties in the interpretation of the later packed-column data to a consistent Tri+ basis. For this reason, the revised analysis presented in the previous section of this paper relies on NEA's capillary-column results.

As did GE, EPA used NYSDEC packed-column results in the trend analysis up through 1997, as few capillary-column results are available. The earlier version of EPA's analysis, presented in the FS, also used NEA capillary-column results preferentially, when these were available.

Interruptions to Ongoing Decline

EPA generally agrees with GE's analysis, presented in Section 2.6 of their Appendix H, of temporal interruptions to ongoing declines in the TI Pool.

Analysis of Temporal Trends

Section 3 of GE's Appendix H presents results of their temporal trend analysis. This section commences by setting out several incorrect characterizations of EPA's effort.

First, GE states that it presents trends for largemouth bass, pumpkinseed, and brown bullhead, which "contrasts with EPA's analysis." This is incorrect, as EPA also analyzed trends in all three species, as well as for yellow perch.

The introduction to GE's Section 3 also claims that "EPA did not include values measured by GE since 1998." This is incorrect: EPA used all the GE/NEA analyses that were available in the database provided by GE at the time the FS was written. For split samples where both NYSDEC packed-column and NEA capillary-column results were available, the GE/NEA capillary-column results were judged more reliable and were thus used to estimate Tri+ PCBs. GE's 2000 results were not available at the time the FS was written; however, they are now available and have been incorporated into the analysis (the previous section of this white paper), with little change in results.

GE presents trend results for a variety of time intervals, including, for the TI Pool, results from start of monitoring up through 1991 and results for 1994-2000, and, for Stillwater, results for 1984-1991, 1984-2000, and 1994-2000.

Of most interest are the recent trends. A key difference from EPA's approach is that GE chose 1994 as the starting point for the evaluation of recent trends. This is significant, because 1994 results were higher than 1995 results for all three species in the TI Pool; thus, starting with 1994 results in faster estimates of decline. EPA believes it is inadvisable to start the trend analysis in 1994, as fish samples were collected in the first half of the year and likely still showed the effects of 1993's high-concentration releases from the Hudson Falls facility.

In presenting results for the TI Pool, GE again mischaracterizes EPA's effort, stating that EPA "used data for one species, brown bullhead, in one location, Thompson Island Pool, over four years, 1995 to 1998." In fact, the analyses cover four species, in both the TI Pool and Stillwater, over a five-year period, 1995 to 1999. These results have now been extended through 2000, as shown in the previous section of this white paper.

It is true that EPA based its bounding calculation primarily on the brown bullhead results. This was done intentionally; of monitored species, bullhead are believed to be those most closely tied to sediment food-chain pathways. Therefore, it is the bullhead that are most likely to be affected by elevated localized sediment concentrations that are not declining at the reach-averaged rate, and the bullhead that are most appropriate to establish an upper-bounding calculation.

GE's results for the TI Pool for 1994 onward suggest that largemouth bass, brown bullhead, and pumpkinseed concentrations are all declining, at rates of from 7 to 16 percent per year. EPA finds slower rates of decline, with increases for pumpkinseed (Table 312627-1 and Figure 312627-1). The difference is due primarily to GE's choice of 1994 as a starting period for the trend analysis, with some additional impact due to the selective rejection of data.

As is clear from Figure 312627-1, strong decreases in brown bullhead and largemouth bass concentrations in the TI Pool occurred between 1994 and 1995, and inclusion of the 1994 data automatically results in a faster estimated rate of decline. However, no decrease in bullhead concentrations are evident from 1995 onward in either GE's or EPA's analysis of the data.

GE's analysis for Stillwater presents results for 1984-1991, 1994-2000, and 1984-2000. GE's focus on the 1984-2000 results is inappropriate, as fish at Stillwater clearly were affected by the Allen Mill event (see brown bullhead and pumpkinseed results for 1992 in Figure 312627-2), thus rendering the long-term trends crossing this event meaningless. As GE itself notes, observed trends are not reliable predictors of the future.

The fact that a strong decline occurred at Stillwater between 1984 and 1991 does not negate the lack of trend in recent data. GE results for 1994-2000 (with corrections for analytical differences) show increases in both largemouth bass and brown bullhead at Stillwater from 1994 on – results that are generally in agreement with EPA's analyses.

GE's text states that "EPA errs in its interpretation of these data, concluding that trends have flattened out in the 1990s" (p. 3-10 of Appendix H); however, GE had already made the same conclusion a page earlier: "For the period from the mid-1990s to 2000, levels exhibited no trend."

In sum, the re-analysis of fish concentration trends developed by GE does not contradict EPA's analysis that provides evidence for lack of strong declining trends in most species in the TI Pool and at Stillwater in recent years. Rather, it merely demonstrates that, for the TI Pool, faster declining trends can be obtained if earlier data in close proximity to the loading events at Hudson Falls are included, coupled with selective rejection of outliers.

For Stillwater, GE’s analysis shows essentially the same lack of trends as EPA’s analysis of recent data. Therefore, GE’s analysis is consistent with EPA’s finding that fish tissue PCB concentration data collected since 1995 do not demonstrate a continuing steady decline.

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Relative Reduction of Human Health and Ecological Risks in the Mid- and Lower Hudson River
313699

WHITE PAPER – RELATIVE REDUCTION OF HUMAN HEALTH AND ECOLOGICAL RISKS IN THE MID- AND LOWER HUDSON RIVER

(ID 313699)

ABSTRACT

For the five remedial alternatives evaluated in detail in the Feasibility Study (No Action, Monitored Natural Attenuation, CAP-3/10/Select, REM-3/10/Select, and REM-0/0/3), cancer risks and non-cancer hazard indices were calculated for the Mid-Hudson River (Federal Dam at Troy to just south of Poughkeepsie) and ecological toxicity quotients were calculated for the Lower Hudson River (Federal Dam to River Mile 33.5)). The risks, hazards and toxicity quotients are greatest for the No Action and Monitored Natural Attenuation alternatives and are similarly reduced for the active alternatives, with the most extensive remedial alternative, REM-0/0/3, offering the greatest risk reduction. Thus, the cancer risks, non-cancer hazard indices and ecological toxicity quotients show the same pattern of relative risk reduction among the remedial alternatives as was shown for the Upper Hudson in the Feasibility Study. In addition, the modeling shows the same overall pattern among the five remedial alternatives with respect to meeting remediation goals and other target concentrations of PCBs in fish.

For the selected remedy, REM-3/10/Select, three scenarios were examined: 1) a five-year dredging schedule with no resuspension (same as Feasibility Study and Proposed Plan); 2) a six-year dredging schedule with 0.13 percent resuspension loss (base case); and 3) a six-year dredging schedule with 2.5 percent resuspension loss. Cancer risks, non-cancer health hazards, and ecological toxicity quotients for the three scenarios are essentially the same. In addition, the modeling shows essentially no difference among the scenarios of the selected remedy with respect to meeting remediation goals and other target concentrations of PCBs in fish. These results indicate that implementing the selected remedy in two phases over six years does not change the relative human health and ecological risk reductions of the selected remedy from the risk reductions presented in the Feasibility Study and Proposed Plan.

INTRODUCTION

This white paper presents the risks to human health (cancer and non-cancer) in the Mid-Hudson, modeled from Federal Dam (River Mile [RM] 153.5) south to the salt front (approximately RM 63.5), and ecological receptors (river otter and mink) in the Lower Hudson, modeled from RM 153.5 to RM 33.5. Risks are presented for the five remedial alternatives evaluated in detail in the Feasibility Study (No Action Alternative, Monitored Natural Attenuation, CAP-3/10/Select, REM-3/10/Select [selected remedy], and REM-0/0/3).

For the selected remedy, REM-3/10/Select, this paper presents the risks to human health (cancer and non-cancer) and ecological receptors (river otter and mink) under three scenarios. The three scenarios evaluate the effects on risk reduction in the Mid-and Lower Hudson of implementing the selected remedy in the Upper Hudson in two phases over six years rather than in a single phase over five years as described in the Feasibility Study and Proposed Plan. The scenarios also show the effect of different rates of PCB loss due to resuspension on the calculated risk reductions. See White Paper – Resuspension of PCBs during Dredging for additional information on resuspension. Specifically, the three scenarios and their model run designations (e.g., R14S2) are as follows:

- Five-year dredging schedule, no resuspension, same as Feasibility Study and Proposed Plan (R14S2).
- Six-year dredging schedule, 0.13 percent resuspension, base case (R20RS).
- Six-year dredging schedule, 2.5 percent resuspension (R20RX).

Similar to the approach used for baseline conditions summarized in the Feasibility Study, output from HUDTOX, EPA's fate and transport model for PCBs in the Upper Hudson River, was used as an external input for modeling of the Mid- and Lower Hudson River. Future concentrations of PCBs in river water, sediment and white perch were derived from the Farley PCB fate, transport, and bioaccumulation model (Farley *et al*., 1999, as updated by Cooney, 1999). Future concentrations of PCBs in brown bullhead, largemouth bass, yellow perch, and spottail shiner were derived from FISHRAND, EPA's bioaccumulation model for the Upper Hudson River. The calculations for all remedial alternatives (No Action, Monitored Natural Attenuation, CAP-3/10/Select, REM-3/10/Select [the selected remedy; all three scenarios] and REM-0/0/3) are based on the top layer of sediment in the Farley model (i.e., 0-2.5 cm) rather than the top two layers (i.e., 0-5 cm) used in the Revised Human Health Risk Assessment and Revised Baseline Ecological Risk Assessment. The Farley model does not have a particle exchange process between sediments layers so that deeper layers are effectively isolated from the surface and any surface interactions with the biota. Additionally, the Farley bioaccumulation model was calibrated using only the 0-2.5 cm layer. Thus the use of the top layer only for the EPA forecasts yields the most representative forecast of surface concentrations for the Lower Hudson River. This is also consistent with the author's original model design (Farley, 2001).

Risks were calculated with exposure durations (*e.g.*, 40 years for evaluating cancer risks to the reasonably maximally exposed (RME) adult angler, 7 years for evaluating non-cancer health hazards to the RME adult angler, and 25 years for river otter and mink) beginning one year after the year in which dredging will be completed in the Upper Hudson River. Thus, risks were calculated beginning in 2010 (five-year scenarios) and 2011 (six-year scenarios), the years in which dredging will be completed over the entire 40-mile stretch of the Upper Hudson River.

All other risk assumptions, locations, toxicity values, receptors and fate, transport and bioaccumulation models (i.e., HUDTOX, FISHRAND, and Farley) used in this White Paper are the same as those used for baseline conditions in the Revised Human Health Risk Assessment, the Revised Baseline Ecological Risk Assessment and the Feasibility Study.

See White Paper – Model Forecasts for Additional Simulations in the Upper Hudson River for additional information regarding modeling of PCB concentrations in fish in the Upper Hudson. See White Paper – Human Health and Ecological Risk Reduction under Phased Implementation for risk calculations for the selected remedy and modeling of PCB concentrations in fish in the Upper Hudson.

HUMAN HEALTH RISK REDUCTION

The species-weighted annual average PCB concentrations in fish fillet used in the human health calculations are shown in Table 313699-1 averaged over the entire Mid-Hudson (*i.e.*, RMs 152, 113, and 90).

Cancer Risks

For the Mid-Hudson, the cancer risks for the RME adult angler for the five remedial alternatives follow the same overall pattern of relative risk reduction as was shown for the Upper Hudson in the Feasibility Study. The cancer risks are greatest for the No Action and Monitored Natural Attenuation alternatives and are similar for the active alternatives, with the most extensive remedial alternative, REM-0/0/3, offering the greatest risk reduction. Specifically, the cancer risks are 1.6E-04 for No Action, approximately 8E-05 for Monitored Natural Attenuation, and range from 5E-05 to 7E-05 for the active alternatives (CAP-3/10/Select, REM-3/10/Select, and REM-0/0/3) (see Table 313699-2). The cancer risks for the central tendency (CT), or average, adult angler are lower than those for the RME adult and also show a similar pattern among the remedial alternatives.

Cancer risks to the RME and CT adult for the three scenarios of the selected remedy are essentially the same (*i.e.*, 6 to 8E-5 for RME and approximately 2E-6 for CT), with the highest risks for the six-year, 2.5 percent resuspension scenario, as shown in Table 313699-2.

Non-Cancer Health Hazards

Similar to the cancer risks, the non-cancer Hazard Indices (HIs) for the RME adult angler for the five remedial alternatives follow the same overall pattern of relative reduction in HIs as was shown for the Upper Hudson in the Feasibility Study. The non-cancer HIs are greatest for the No Action and Monitored Natural Attenuation alternatives and are similar for the active alternatives, with the most extensive remedial alternative, REM-0/0/3, offering the greatest reduction. Specifically, the HIs are approximately 10 or 11 for No Action, approximately 6 to 8 for Monitored Natural Attenuation, and range from 4 to 6 for the active alternatives (see Table 313699-3). The HIs for the CT adult angler also show a similar pattern among the remedial alternatives, but are all less than an HI of 1.

Non-cancer HIs for the RME and CT adult for the three scenarios of the selected remedy are essentially the same (i.e., 6 to 8 for RME and 0.4 to 0.6 for CT), with the highest HIs for the six-year, 2.5 percent resuspension scenario).

Time to Reach Human Health Risk-Based Concentrations in Fish

The human health risk-based remediation goal (RG) and other target concentrations of PCBs in fish show the same overall pattern among the remedial alternatives with respect to the years in which such goals and target concentrations are met as was shown for the Upper Hudson in the Feasibility Study (see Table 313699-4).

Specifically, the RG of 0.05 mg/kg PCBs (wet weight) in species-weighted fish fillet is not met for the any of the remedial alternatives by 2046, which is the extent of the modeling period. The 0.2 mg/kg target concentration is not met for the No Action alternative, is met in 2017 for the Monitored Natural Attenuation alternative, is met in 2015 for CAP-3/10/Select and REM-3/10/Select, and is met in 2013 for REM-0/0/3. The 0.4 mg/kg target concentration is met in 2016 for the No Action alternative, is met in 2009 for the Monitored Natural Attenuation alternative, is met in 2008 for CAP-3/10/Select and REM-3/10/Select, and is met in 2007 for REM-0/0/3 (see Table 313699-4).

Among the three scenarios of the selected remedy, the years in which the human health risk-based RG and other target concentrations of PCBs in species-weighted fish fillet are met in the Mid-Hudson are essentially the same, with the longest times for the six-year, 2.5 percent resuspension scenario. Specifically, the 0.05 mg/kg RG is not met by 2046, the 0.2 mg/kg target concentration is met in 2015-2016, and the 0.4 mg/kg is met in 2008-2012 (see Table 313699-4).

ECOLOGICAL RISK REDUCTION

Annual PCB concentrations in largemouth bass, which were used in the ecological calculations of risk to the river otter, are shown in Table 313699-5 averaged over the Lower Hudson (i.e., RMs 152, 113, 90, and 50). Annual PCB concentrations in spottail shiner, which were used to calculate risk to the mink, are shown in Table 313699-6 averaged over the Lower Hudson.

Toxicity Quotients

The Toxicity Quotients (TQs) for the river otter in the Lower Hudson for the remedial alternatives follow the same overall pattern of relative reduction in TQs as was shown in the Feasibility Study for the Upper Hudson. The TQs are greatest for the No Action and Monitored Natural Attenuation alternatives and are similar among the active alternatives, with the most extensive remedial alternative, REM-0/0/3, offering the greatest reduction. Specifically, the TQs are about 5 (lowest-observed-adverse-effect [LOAEL] basis) to 50 (no-observed-adverse-effect [NOAEL] basis) for No Action, 3 (LOAEL basis) to 30 (NOAEL basis) for Monitored Natural Attenuation, and are approximately 2 (LOAEL basis) and 20 (NOAEL basis) for the active alternatives (see Table 313699-07). The TQs for the mink in the Lower Hudson also show a similar pattern among the remedial alternatives.

For the three scenarios of the selected remedy, ecological TQs for the river otter in the Lower Hudson are essentially the same (*i.e*, 2 or 3 [LOAEL basis] and 22 to 26 [NOAEL basis]), with the longest times for the six-year, 2.5 percent resuspension scenario (see Table 313699-7). The TQs for the mink are lower (*i.e*., less than one [LOAEL basis] and about 5 [NOAEL basis]) than those for the river otter and also show similar patterns for the three scenarios of the selected remedy (see Table 313699-7).

Time to Reach Ecological Risk-Based Concentrations in Fish

The ecological risk-based concentrations of PCBs in fish show the same overall pattern among the remedial alternatives with respect to the years in which risk-based concentrations are met as was shown in the Feasibility Study (see Table 313699-8). Specifically, the risk-based NOAEL concentration of 0.03 mg/kg in largemouth bass based on the river otter is not met for the any of the remedial alternatives by 2046, which is the extent of the modeling period. The risk-based LOAEL concentration of 0.3 mg/kg target concentration is not met for the No Action alternative, is met in 2033 for the Monitored Natural Attenuation alternative, is met in 2023 for CAP-3/10/Select and REM-3/10/Select, and is met in 2019 for REM-0/0/3. The risk-based NOAEL concentration of 0.07 mg/kg in spottail shiner based on the mink is not met for the No Action alternative or the Monitored Natural Attenuation alternative, is met in 2030 for CAP-3/10/Select, is met in 2027 for REM-3/10/Select, and is met in 2023 for REM-0/0/3. The risk-based LOAEL concentration of 0.7 mg/kg in spottail shiner is met in 2004 for all alternatives.

As shown in Table 313699-8, there is little difference among the three scenarios of the selected remedy in the time to reach ecological risk-based concentrations in fish. In the Lower Hudson, the ecological remediation goal (RG) range of 0.03 to 0.3 mg/kg in largemouth bass (whole body), based on the river otter, is met between 2023 to 2026 for all scenarios, with the longest time for the six-year, 2.5 percent resuspension scenario.

The ecological RG is considered to be protective of all the ecological receptors evaluated because it was developed for the river otter, the piscivorous mammal calculated to be at greatest risk from PCBs. An additional range of 0.07 to 0.7 mg/kg PCBs in spottail shiner (whole fish) was developed based on the NOAEL and LOAEL for the mink, which is a species known to be sensitive to PCBs. As shown in Table 313699-8, there is no difference among the three scenarios in the time to reach the range developed for protection of the mink; all scenarios of the selected remedy meet the range in 2004.

CONCLUSIONS

Cancer risks, non-cancer hazard indices, and ecological toxicity quotients show the same overall pattern of risk reduction in the Mid-Hudson (human health) and Lower Hudson (ecological receptors) for the five remedial alternatives evaluated in detail in the Feasibility Study. The risks, hazard indices and toxicity quotients are greatest for the No Action and Monitored Natural Attenuation alternatives and are similarly reduced for the active alternatives, with the most extensive remedial alternative, REM-0/0/3, offering the greatest risk reduction. The modeling shows the same overall pattern among the five remedial alternatives with respect to meeting risk-based concentrations in fish.

For the three scenarios of the selected remedy, REM-3/10/Select, cancer risks and non-cancer hazard indices in the Mid-Hudson and ecological toxicity quotients for river otter and mink in the Lower Hudson are essentially the same. These results show that implementing the selected remedy in two phases over six years rather than in a single phase over five years does not materially change the risk reduction in the Mid- and Lower Hudson provided by the selected remedy.

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Resuspension of PCBs During Dredging
336740

WHITE PAPER – RESUSPENSION OF PCBs DURING DREDGING

(ID 336740)

ABSTRACT

Numerous comments were received on the resuspension and contaminant transport estimates in Appendix E.6 of the Feasibility Study (FS). Some comments concurred with the estimates while others suggested the estimates are unrealistically low. Most of the comments suggesting higher resuspension rates based on their position on information from other environmental dredging projects. Dredging projects in the Fox River, Manistique River and Harbor, and Grasse River were presented as examples with data seemingly contrary to the FS estimates. GE comments also suggested inconsistency between the source strength models discussed in the FS and those recommended in other publications such as the recent NRC report (NRC, 2001). Several other questions related to the basis for the source strength estimates were raised.

This paper addresses these comments, focusing particularly on how the initial estimates were made, the use of models in the FS, and the applicability of data from these other dredging projects to the Hudson River. Careful analysis shows that the original estimates are still valid for the dredging operations proposed in the FS. Observed rates of resuspension at several sites are compared with model-based estimates for the Hudson.

After carefully considering the comments received on dredging, EPA concludes that the information contained in or referred to by those comments does not justify modifying the basis for the sediment resuspension estimates contained in Appendix E.6 of the FS. The estimates of resuspension at the dredgehead of 0.3 percent for mechanical dredges and 0.35 percent for cutterhead dredges are well supported by the above discussion. The associated downstream transport estimates of 0.13 and 0.065 percent, respectively, represent conservative estimates of the potential releases due to dredging and are consistent with direct observations made on several sites.

INTRODUCTION

Dredging is fundamentally a sub-aqueous earth moving action. Just as ground-based earth moving operations generate dust, dredging results in sediment particles being released into the water column. Just as air currents spread dust from a construction site, ambient water currents transport resuspended sediments downstream. And, resuspended sediments with particulate-associated PCBs increase water column PCB concentrations just as contaminated dust particles result in airborne contaminants. The key to estimating these impacts is an accurate estimate of the mass of sediment placed into the water column by the mechanical actions of the dredge itself. Even hydraulic dredges utilize mechanical actions to loose sediment and guide it into the hydraulic intake. Since each dredgehead and dredge type applies distinctly different mechanical forces to move sediment from the river bottom, the most important factor in estimating the rate of sediment resuspension is the type of equipment being used and its operation. Resuspension rates for one type of dredge are entirely irrelevant to resuspension that might be expected from another dredge type.

It is important to note that resuspension of sediments occurs as a result of the movements of the dredge head and its appendages along the river bottom (and in the case of the mechanical dredge, upward through the water column). These disturbances serve to suspend a portion of the sediment in the water column in the immediate vicinity of the dredge head. However, only those materials that can remain in the water column (i.e., silts and clays) represent potentially important releases to the water column and downstream areas. The process of moving the sediments from the river bottom to the water column is referred to in the following discussion as “resuspension.” The downstream movement of resuspended material away from the dredge head is referred to in this discussion as “downstream transport.” Notably, it is the amount of downstream transport and not simply the amount of resuspension which ultimately determines the impacts to areas downstream.

Because the processes of resuspension and downstream transport are so closely linked, the term “resuspension”: is used throughout the main body of this responsiveness summary to refer to the entire process of resuspension and downstream transport.

A brief summary of related FS assessments is followed by a discussion of the relationship between suspended sediment concentrations and downstream contaminant transport. The role of silt curtains in this transport process is also addressed. Available field data including the data referenced in a number of FS comments are summarized. Existing resuspended sediment source strength models are addressed in light of these data.

RELATED FS ASSESSMENTS

Appendix E.6 provides a comprehensive assessment of sediment resuspension, PCB release, and downstream transport estimates for the proposed remedial dredging operations. This section summarizes the procedures followed and the resulting estimates.

The remedial dredging operations of the selected remedy are substantial in scope. Selecting dredging equipment capable of successfully completing the project is critical, particularly since not all dredges can satisfactorily accomplish the project goals under the given Site conditions and project constraints. In particular, some of the smaller hydraulic dredges have inadequate pumping capacity and removal effectiveness. Significantly higher water quality impacts have been observed around these dredge types (USACE 1990; FRRAT 2000; USGS 2000).

The FS identified a conventional hydraulic cutterhead dredge and environmental bucket dredges as the most appropriate existing equipment for the project. Further, the FS suggested that a hydraulically closed bucket such as the horizontal profiler might have advantages, especially in areas where debris could be a problem. Thus, Appendix E.6 provided estimates for a conventional 12-inch diameter cutterhead suction dredge and 4-cy and 2-cy environmental bucket dredges; although a brief discussion of horizontal profiler buckets was included, only estimates for environmental buckets were used since no data on the horizontal profiler bucket were available.

For the conditions expected in the Hudson River, suspended sediment loss estimates were 0.35 percent for a conventional cutterhead dredge and 0.3 percent for an “environmental bucket.”

Since resuspension characteristics are so closely tied to equipment characteristics and operation, these estimates are applicable only to these two dredge types; similarly, one should not expect any relationship between observed sediment resuspension rates from other dredge types and the selected dredge types.

The suspended sediment loss rate of 0.35 percent[1] for a conventional cutterhead dredge reflects conditions in the immediate vicinity of the dredging operation (within a few meters). The estimate results from the combination of several analyses, but is primarily based upon available field data for similar dredging operations.

First, empirical near-field source strength models developed by Hayes *et al.* (2000b) and Wu and Hayes (2000) were used to estimate the sediment resuspension rate based upon anticipated operational conditions. These models were developed solely on available water quality data from hydraulic cutterhead suction dredging operations. Hayes *et al.* (2000b) includes ranges of applicability for all parameters and clearly states that these models should not be used outside of these ranges; the models are well behaved within these ranges. However, because the operational conditions are still uncertain, it was concluded that these estimates should be verified using field data from two conventional 12-inch hydraulic cutterhead suction dredging operations; these data are described in detail below. A statistical analysis showed the highest observed instantaneous rate for this dredge type and size was 0.35 percent. Most of the observed resuspension rates were less than 0.2 percent.

Source strength models are not available for environmental bucket dredging operations. Thus, data from more conventional bucket dredging operations were the basis for estimating a 0.3 percent sediment resuspension rate. Properly sealed and operated environmental buckets should result in lower resuspension rates.

The estimates above are for fine sediment particles at the point of dredging. Some silt resuspended by the dredging operation will resettle in the immediate vicinity of the dredging operation. Published transport models for dredging operations (Kuo *et al.* 1985; Kuo and Hayes 1991) were used to estimate downstream transport of the resuspended material. Settling rates and lateral dispersion coefficients from the literature were used in the modeling efforts as described in Appendix E.6 of the FS. These models were used to estimate the mass flux of suspended sediment particles at 10 meters from the dredging operation. The flux at 10 meters was conservatively assumed to remain in suspension in perpetuity[2] and used in all other estimates of downstream suspended sediment and PCB transport. Assuming 100 percent efficiency, the production rates are 95 cy/hr and 270 cy/hr for the environmental bucket and cutterhead dredges, respectively. The downstream TSS flux for different sections of the river can be found in Table 336740-1. The fluxes are summarized for both the conventional enclosed bucket and cutterhead dredge. Mass-weighted averages of these flux rates in the three reaches result in an estimated

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[1] All sediment resuspension loss values are of the fine-sediment fraction only. Sands and larger particles will resettle immediately, even if resuspended, and not be transported downstream

[2] In reality, reductions in PCBs in the water column will occur at greater distances downstream due to further settling of resuspended particles. For discrete load gain calculations, no further settling was assumed. For model calculations, no settling due to the dredging process was included. Some settling is inherent in the model algorithms.

downstream transport of 0.13 percent of the total sediment dredged for an environmental bucket dredge and 0.065 percent for a conventional hydraulic cutterhead dredge. The mass of PCB resuspended and transported downstream based on the revised estimate of total PCBs to be removed by the selected remedy (150,000 lbs) would be 200 lbs (91 kg) for environmental bucket dredges and about half that amount for a hydraulic cutterhead dredge.

TOXIC CONSTITUENT LOSSES AND TRANSPORT

Although the complexities of contaminant interactions and transformation of specific constituents are not completely understood, the basic theories associated with toxic constituent transport in surface waters are well developed. As described in Appendix A of the comments by GE (QEA *et al*., 2001), the area in the immediate vicinity of the dredging operation can likely be approximated as a well-mixed tank, often referred to as a continuous flow stirred tank reactor or (CFSTR). Suspended sediment and toxic constituent concentrations in the well-mixed water volume that can be approximated as a CFSTR can be approximated by:

Solids: $$V_{nf} \frac{dm}{dt} = qm_{in} - qm - v_s A_h m + \dot{M}_R$$

Toxics: $$V_{nf} \frac{dc}{dt} = qc_{in} - qc - kV_{NF} c - v_v A_{ws} F_d c - v_s A_h (1 - F_d) c + 10^{-6} \dot{M}_R c_{sed}$$

where:

V_{nf} = volume of the near-field area (m^3),
t = elapsed time (sec),
m_{in} = TSS concentration of flow entering the near-field volume (g/m^3),
m = TSS concentration in the near-field volume (g/m^3),
q = flow through the near-field volume (m^3/sec),
v_s = settling velocity of suspended particles in near-field volume (m/sec),
$\dot{M}_R$ = rate of mass resuspension into the near-field area due to dredging (g/sec),
c = toxic constituent concentration within the near-field volume (g/m^3),
c_{in} = toxic constituent concentration flowing into the near-field volume (g/m^3),
k = contaminant transformation rate (1/sec),
v_v = volatilization mass-transfer coefficient (m/sec),
A_{ws} = horizontal area of the near field exposed to the water surface (m^2),
A_h = horizontal area of the near field of the near-field volume (m^2),
F_d = fraction of contaminant mass in dissolved form (unitless), and
c_{sed} = contaminant concentration on bottom sediments (mg/kg).

Note that the term qm in the solids transport equation equates to the resuspension loss rate as defined by Hayes *et al* (2000b) as m_R. In most cases, some of these parameters can be neglected. For example, volatilization need be considered only when the near-field area extends to the water surface. Under steady-state conditions with minimal background concentrations and short retention times in the near-field zone allows the equations to be simplified to:

$$\dot{m}_R = \dot{M}_R - v_s A_h m \qquad \text{and} \qquad \dot{c}_R = 10^{-6} \dot{M}_R c_{sed} - v_s A_h F_p c_{sed}$$

Although the general approach is the same as described by QEA *et al.* (2001), Appendix A of the GE comments assumes that there is no suspended sediment transport out of the well-mixed volume. This is contrary to the basic assumption of a well-mixed water volume; if the energy is sufficiently low for all for the fine solids to settle, the volume is unlikely to be well-mixed. Observations of many dredging operations by Hayes have led to the conclusion that the well-mixed areas around active dredging operations are very small with correspondingly low detention times; flux out of this volume varies both temporally and spatially. Given a short retention time and the preferential association of toxic constituents with fine particles, it is reasonable to further assume that settling in this zone is not a major influence on contaminant water column concentrations. If so, these can be further simplified to:

$$\dot{m}_R \cong \dot{M}_R \qquad \text{and} \qquad \dot{c}_R = 10^{-6} \dot{m}_R c_{sed}$$

where $\dot{c}_R$ = mass flux of toxic constituent out of the near-field volume (g/sec).

Downstream transport is more complex, because spatial and temporal variability in current velocity and direction result in incompletely mixed transport. The general transport equations are:

Solids: $$\frac{\partial m}{\partial t} = -\nabla(um) + \nabla^2(Em) - \frac{m v_s}{h} + \dot{m}_R$$

Toxics: $$\frac{\partial c}{\partial t} = -\nabla(uc) + \nabla^2(Ec) - kc - \frac{v_v F_d c}{h} - \frac{v_s F_p c}{h} + \dot{c}_R$$

where: u = current velocity (m/sec), E = rate of diffusive transport (m^2/sec),
and $\dot{m}_R$ = mass flux of toxic constituent into the water column.

Common simplifications to this equation include steady-state conditions and neglecting either diffusive or advective transport, depending upon the value of the estuary number. Suspended solids transport models have been applied to dredging operations by several authors; Cundy and Bohlen (1980) is a classic example. Kuo *et al*. (1985) and Kuo and Hayes (1991) developed more general solutions for specific dredge types.

If concerns related to toxic transport are primarily associated with particulate concentrations, water column contaminant can be estimated from suspended solids concentrations by applying equilibrium partitioning. Given a water column TSS concentration of m, the contaminant concentration associated with the sediment mass, c_{sed}, and an associated partitioning coefficient, K_d (m^3/g), the dissolved and particulate fractions can be estimated as:

$$F_d = \frac{1}{1 + K_d m} \quad \text{and} \quad F_P = \frac{K_d m}{1 + K_d m}$$

or $c_d = F_d c_{sed}$ and $c_p = F_p c_{sed}$. Although this approach involves a number of simplifying assumptions, the particulate-associated concentrations should be reasonably accurate, albeit conservative. Recent observations of elevated dissolved constituent concentrations, however, suggest that a more rigorous analysis of contaminant transport, especially the dissolved component, may be needed. Dissolved constituents are of greater concern because silt curtains do not impede their transport and they are generally more bioavailable to fish and other aquatic organisms.

The approach described by QEA *et al.* (2001) assumes that no solids escape the well-mixed zone, thus downstream transport of suspended solids is unnecessary. The result, however, is that the PCBs in the water column are completely distributed in the dissolved phase since no suspended solids are available to adsorb hydrophobic PCBs.

The result of the modeling discussion in Appendix A of GE's comments proves that toxic constituent loss cannot exceed suspended sediment loss. This deduction is correct. Dissolved toxic constituent mass is of primary interest, however, since particulate-associated contaminants will predominantly resettle to the river bottom while dissolved contaminants are transported downstream and pose a much greater risk to fish and other aquatic biota. Low water-column TSS concentrations, as observed downstream of most dredging operations, result in a larger fraction of the toxic constituent mass being in the dissolved phase. Monitoring results from recent environmental dredging demonstration projects in the Fox River (WI) at Deposit N and SMU 56/57 seem to show substantial dissolved PCB loss downstream without corollary suspended sediment losses (USGS, 2000). In fact, USGS (2000) reports higher suspended sediment concentrations existed upstream from SMU 56/57 dredging operations than downstream. However, a review of the contaminant distribution equations in light of ambient suspended solids concentrations in the Fox River suggests this is quite unlikely.

It is useful, however, to review the conditions that might yield high dissolved toxic constituent losses to the water column. The basic dissolved constituent loss equation is:

$$R_{dissolved} = \frac{100R}{1 + K_d R h}$$

where

$$h = \frac{Q_d C_d}{q}$$

where: $R_{dissolved}$ = fraction of *in situ* toxic constituent mass loss to the water column in a dissolved form,

Q_d = dredge flowrate (m^3/sec), and

C_d = solids concentration of sediment during dredging (g/m^3).

It should be pointed out that this approach assumes the mass rate of sediment removal during dredging and the mass flow through the dredging operation are essentially equivalent. Since residuals certainly exist, the result is conservative. Figure 336740-1 shows that the highest dissolved releases occur when a combination of low production rates (resulting from low solids concentrations in the slurry and low-flow rates), significant flow through the area, and high resuspension rates. This probably explains, at least partially, the higher dissolved constituent observations during horizontal auger dredging operations. It also points to the need for caution in applying operational controls that substantially reduce dredge production rates.

Role of Silt Curtains

Silt curtains and silt screens are common appurtenances in environmental dredging operations. These consist of sections of either permeable or impermeable fabric hanging down into the water column from a floating boom. The fabric may extend into the water column as little as a meter or for the entire water depth. Sections can be connected together to form long, mostly continuous barriers. Under ideal conditions, silt curtains can contain much of the sediment resuspended by the dredging operation. QEA *et al.* (2001) imply that dissolved contaminants flow directly through the curtain, thus yielding an imbalanced contaminant distribution beyond the curtain. Although data have not been collected to prove the result, observations show that very little flow actually passes through the screen material. Instead, headloss associated with flow through small mesh results in redirection of most of the flow either around or underneath the curtain. Securing a silt curtain to the bottom is almost impossible, except in unusual circumstances. Headloss associated with flow through the curtain usually exceeds the maximum anchoring weights that can be dealt with in open water environments; as a result, the curtain lifts, redirecting much of the flow between the lifted curtain and bottom sediment. As illustrated in Figure 336740-2, even a slight current can generate enough force to lift the curtain above the bottom and allow the turbidity plume to escape.

NRC (2001) implies that when properly deployed under proper conditions, suspended sediments within the silt curtain can be considered to be at a uniform concentration and the toxic constituents at equilibrium much like a CFSTR. These conditions are mostly likely to exist when the volume within the silt curtain is small and the curtain is securely attached to the bottom sediments and completely encircles the dredging operation. However, such small enclosures require more intensive management and frequent repositioning. Thus, silt curtains are more frequently deployed in larger circles. Incomplete mixing and significant variations in suspended sediment concentrations within the curtain itself usually characterize these larger volumes.

EVALUATION OF FIELD STUDY DATA – HYDRAULIC DREDGES

Hydraulic dredges utilize mechanical action to feed sediment into a suction pipe that carries it to the surface and eventually some point of discharge. Most bottom sediments are too viscous to pump directly; thus, ambient water mixes with the sediment in the suction pipe to form a pumpable slurry.

Hydraulic dredges are quite distinguishable by the combination of mechanical actions used to guide sediment into the suction pipe. These include conventional basket-type cutterhead dredges, dustpan dredges, horizontal auger dredges, and a variety of specialized dredgeheads designed for specific purposes. For example, the dustpan dredge was designed for use in the lower Mississippi River where it could dredge a pit in the channel, then remove sediment from the pit as the river deposited it there. This minimizes dredge movement and is a very efficient sand-dredging operation in this environment. Horizontal auger dredges were initially designed for removing sewage sludge from lagoons.

These differences result in vastly different machinery and dictate that resuspension and production characteristics be considered independently for each hydraulic dredge type. This section attempts to summarize the available data by dredge type and draw conclusions related to the use of these data for estimating water quality impacts in the proposed Hudson River dredging operations.

CUTTERHEAD FIELD STUDIES

Resuspension data for cutterhead dredges have been presented by a number of authors. Hayes *et al.* (2000b) and Wu and Hayes (2000) present almost 400 observations of resuspension rates from five field studies. The characteristics of these studies are summarized in Table 336740-2.

Observed Resuspension Rates

Table 336740-2 summarizes the resulting resuspension rate values and their statistical characteristics for each field study. The results are consistent with expectations. The highest resuspension rate is from Lavaca Bay – Phase II. The combination of a small dredge with relatively low horsepower removing highly consolidated, sticky clay in a dynamic environment would be expected to be a poor combination. Small particle sizes and a relatively low production rate exacerbate the problem. New Bedford (Acushnet River) observations were also elevated (not accounting for background PCB levels) because of low dredge production, light sediments, and extensive debris. The DUBUQUE operated under almost ideal conditions in Calumet Harbor, and the resuspension rate reflects that the operation was quite effective.

The 18-inch cutterhead dredges used in the Back River (Savannah, GA) and James River (Norfolk, VA) are far larger vessels than one might deduce, based simply on their descriptive sizes. Generally, these larger dredges carry powerful hydraulic pumps capable of dredging much greater depths and transporting the sediments much larger distances. Thus, under normal conditions, the intake velocities are substantially greater; one would expect this fact alone to result in less resuspension, and these data generally support that conclusion.

Modest resuspension rates were observed, especially considering that the CLINTON (Back River) undercut a 20-foot bank (which often collapsed) using very aggressive operational tactics. As expected, the more cautious operation used by the ESSEX (James River) (McLellan, *et al.* 1989) yielded lower resuspension rates. James River sediments were likely more vulnerable to resuspension because of their high *in situ* moisture content[3] (186 percent), especially considering that they are greater than the liquid limit (120 percent). Although sediment data are not available, the *in situ* moisture content of the Back River sediments were almost certainly between the plastic limit and liquid limit, or there probably would have a more significant difference.

Statistical Evaluation

Ideally, one would match site-specific conditions, dredging equipment, and operational methods with similar projects for which observed resuspension rates exist. The limited data currently available do not allow such a direct comparison. However, an adequate number of observations exist to draw general conclusions of observed resuspension rates for cutterhead dredging operations. Thus, it is useful to evaluate the range and frequency of observed resuspension rates.

Figure 336740-3 shows a frequency histogram of the 388 observations listed in Table 336740-2. Observed resuspension rates range from near 0 to 0.51 percent with the preponderance of values between 0 and 0.1 percent. The data have a mean of 0.11 percent with a standard deviation of 0.11 percent. Most of the observations, 282, are from the Phase II pilot study in Lavaca Bay, which has a strong influence over the data set. The data for all of the projects except Lavaca Bay have an average of 0.05 percent and standard deviation of 0.07 percent.

Observed resuspension rates used in this paper are from dredges ranging in size from 10-inch to 18-inch. The data imply that the average and range of sediment resuspension rates do not vary consistently with dredge size, except that both are lower for the larger dredge. This is probably due to higher vacuum pressures near the intake, due to large pump horsepower. Despite this seeming consistency between dredge sizes, care should be exercised in attempting to apply these resuspension values to dredge sizes outside of this range. In particular, the increase in resuspension rate for the smaller dredges is likely to be more exacerbated for other dredges. The Ellicott 370 used in the New Bedford study is more adequately powered than other types of similarly sized hydraulic dredges. For example, the Ellicott 370 has a 360 HP engine, as compared to the 175 HP engine used in the 8-inch horizontal auger dredge used later in the study. Many of the smaller specialty dredges were initially designed for dredging sewage sludge, which is much more fluid than sediments.

Water quality evaluations often focus on the possibility of exceeding regulatory criteria. These analyses require one to look at a cumulative probability distribution of observed resuspension rates. For the data presented here, a sediment resuspension rate of 0.31 percent is exceeded only 5 percent of the time ($R_{.05}$); a resuspension rate of 0.46 percent is exceeded only 1 percent of the time ($R_{.01}$). It would seem that these values should represent approximate maximums for similar cutterhead dredging operations.

It was also observed that the data fit a log-normal distribution quite well. While this is not utilized here, it provides the possibility to extend the current analysis to a risk-based assessment.

[3] Moisture content is calculated here as the ratio of the mass of water per unit volume of wet sediments to the mass of solids per unit volume of wet sediment. Thus a sediment that was half water and half solids by weight would have a moisture content of 100 percent.

Summary of Cutterhead Data

Although more data would be helpful and the five field studies do not cover all possibilities, the presented values represent a reasonable range of sediment resuspension rates for different cutterhead dredge sizes and operating conditions. By matching dredging project characteristics with these field studies, one should be able to reliably estimate resuspension rates for similar dredging equipment.

OTHER FIELD STUDIES

Several other dredging resuspension studies provide additional information. This section examines four of them – the Fox River demonstration dredging operations at SMU 56/57 and Deposit N, the New Bedford *hot spot* dredging, and GE dredging at Hudson Falls.

Fox River Dredging Demonstration Studies

Many commenters mentioned the recent Fox River SMU 56/57 and Deposit N demonstration dredging operations as providing resuspension and contaminant transport data contrary to the FS estimates. Suspended sediment and PCB concentrations in the water column were measured upstream and downstream of the dredging operations as part of a larger PCB mass balance study of the projects. Both studies were over three months long and generated impressive data sets. Composite suspended solids samples were taken at four to five stations across cross sections upstream and downstream of the dredging area; equal volumes of water at 20 percent and 80 percent depths were composited from each station to form a single sample later analyzed for TSS concentration.

A single PCB composite sample for the entire cross section was obtained by compositing equal volumes from the same depths at all locations; *i.e.*, 8 or 10 equal volume samples were combined to obtain a single PCB composite sample for the cross section. The resulting data set included 22 data pairs (TSS and PCB) from Deposit N and 36 data pairs from SMU 56/57 during dredging operations. The average of the Deposit N data pairs show a TSS *loss* across the area of 1.7 percent and a PCB *gain* of 10.6 percent (FRRAT 2000). USGS (2000) reports that similar results from SMU 56/57 show a TSS *loss* across the area (a specific rate is not mentioned) and a PCB *gain* of 2.2 percent.

These results initially seem contrary to those estimated in Appendix E.6. Closer investigation, however, shows apparent limitations in the Fox River studies. These are listed below:

- The load-gain estimate is based on a cross-section that is located too close to the dredging area. The cross-section is also located in an area that is a likely backwater (it is in a turning basin, with a nearby coal boat canal). It should be noted that sampling activities during boat activity showed higher PCB concentrations and were included in estimates of releases. Thus, flows through the cross-section are unlikely to be consistent. The proximity of the cross-section to the dredging area also increases the likelihood that the sampling will not be representative of the total load, since the input from dredging will be poorly mixed (see Figure 336740-4).

- The sample compositing strategy, designed to reduce the number and cost of PCB analyses, was contrary to the mass flux analysis attempted. The equal volume composites do not allow consideration of flow variation across the cross-section. USGS (2000) states that stagnant areas and even reversed flows were observed during sampling operations, confirming the errors associated with the composite PCB samples. The TSS sample composites induce less error and provide a more accurate estimate of downstream TSS flux, yet they showed an unexplained decrease in suspended sediment across the dredging operation. The decrease is almost certainly an artifact associated with compositing equal volume samples from 20 percent and 80 percent depth. Even though it has long been established that velocity measurements from these depths represent the average velocity in an open channel, there is no justification for suggesting that a composite sample from these depths represents the average concentration along the profile. This is particularly true in deeper water where the two samples represent 25 feet or more of water depth.

- The method of PCB collection is not documented, but it appears that the method represents the dissolved and suspended matter fractions inaccurately, based on the lack of change in PCB pattern across the dredging area (see Figure 336740-5). The load gain is attributed to a large gain in dissolved PCBs, but this is inconsistent with the PCB congener pattern. A large dissolved-phase PCB contribution from the sediments, either by porewater displacement or sediment-water exchange, should yield a gain whose pattern is similar to the filter supernatant (see Figure 336740-6). The fact that the congener pattern is unchanged across the study area would suggest a direct sediment addition. Yet the suspended solids data documents no increase in suspended sediments.

- Similarly, the total PCB concentration of the suspended matter doubles, yet there is no change in the suspended matter loading (see Figures 336740-7 and 336740-8). Given the proximity of the downstream sampling cross-section to the source area, it is unlikely that the majority of the TSS in the river could be directly affected by dredging induced resuspension.

- A review of the PCB loading over the dredging period shows that PCB loads were relatively low for the first 2.5 months of operation, when dredging took place at the more upstream end of the targeted area (see Figure 336740-9). During this period, the estimated release was only 3 kg or about 1.2 kg/month. This changed dramatically during the last month of operation, when the loading rate increased to about 13.5 kg/month. During this latter period, the dredging took place at the downstream end of the targeted area, very close (the closest station less than 80') to the sampling cross-section, near areas with higher PCB concentrations. Another significant factor, as discussed in the USGS paper, that may have caused elevated PCB concentrations in the downstream profile was increased water flow velocities. Proximity of dredging to the deposit or water flow could have been significant contributing factors for increased PCB concentrations observed in the downstream profile. To conclude that observed increases are only related to dredging fails to consider these and other potential influences. Additionally, a lack of comparable transect data for PCB water column concentrations for pre-dredging (*i.e.*, “natural”), and during dredging also contributes to the uncertainty evaluating dredging surface water contributions.

- The fact that significant loss of PCBs only occurred when the dredging area was close to the sampling cross-section suggests that settling of any resuspended matter occurs within a short distance of the dredging operation. Only when the monitoring location was close to the dredging could this signal be found. This suggests that the loads obtained by this study do not represent PCB released for long-distance transport. Rather, the PCBs appear to be quickly removed from the water column a short distance downstream. As such, it is inappropriate to use these results to estimate downstream transport from a dredging site.

Although substantial data sets resulted from the Fox River dredging demonstration projects, the sampling approach and compositing strategy mask the results. A close review shows the study results can only be considered inconclusive and should not be used as the basis for estimating resuspension from any future dredging operations.

More importantly, however, the dredges used in the Fox River studies are entirely different from those proposed for the Hudson River remedial dredging. An 8-inch Moray Ultra dredge was used in the Deposit N demonstration project and for 7 to 14 days during the SMU 56/57 project. A horizontal auger dredge completed the SMU 56/57 dredging. The equipment characteristics and operation of both dredges are substantially different from those proposed for the Hudson River remedial dredging. There is no reason to believe that their resuspension characteristics are in any way related to those of either a conventional hydraulic cutterhead suction dredge or an environmental bucket dredge. In fact, the New Bedford pilot study compared the sediment resuspension characteristics of a horizontal auger dredge with a conventional hydraulic cutterhead suction dredge and found a disparity similar to that observed between the Fox River data and the estimates in the FS.

New Bedford Harbor Hot Spots

Under Superfund, the sediments of New Bedford Harbor were identified as a significant source of PCBs to the environment as such remediation via dredging was selected as the selected remedy. In 1994-1995, a portion of these sediments, identified as hot spots were removed from the harbor and stored in an upland facility (see Figure 336740-10). Approximately 14,000 cubic yards of sediment were removed. Waterborne PCB concentrations recorded during the outgoing tide at a downstream location (Coggeshall Bridge) served as a measure of the mass of PCBs released to the bay as a result of dredging. This location was sufficiently far away from the dredging area such that water column concentrations of re-released PCBs were probably homogeneous. Water column measurements were made throughout the dredging period. It was estimated that 57 kg of PCB escaped the inner harbor area. This included background PCB levels as well as any PCBs resuspended from the dredging. Measurements of the removed sediments themselves were also performed as part of treatability studies. These results indicate that the dredged sediments had an approximate PCB concentration of 5,000 mg/kg, or about 0.5 percent PCB by weight.

Using these results, it is possible to estimate a dredging “loss rate” by taking the ratio of PCBs lost to PCBs removed. The estimated PCB mass removed was 43,733 kg; thus, the estimated loss was 57/43,733 or 0.13 percent. This is substantially less than the 2.2 percent estimated from the Fox River work. It is also likely that this value is more representative of long-distance transport since the monitoring location was sufficiently far from the dredge area that any rapidly settling particles were not captured by the monitoring samples.

The conditions at this location were also more extreme than those found on the Hudson, thus suggesting that the New Bedford Harbor results might represent an upper-bound estimate for dredging on the Hudson. Specifically, sediments were substantially finer than those of the Upper Hudson (silts and clays at New Bedford *vs.* fine sands in the Hudson). Additionally, the sediments of New Bedford were approximately two orders of magnitude more contaminated than those on the Hudson; thus, small spills or leaks at New Bedford have the potential to re-release substantially more PCBs than a similar-sized spill on the Hudson. In addition, it should be restated that the calculation for New Bedford did not account for background PCB flux.

The New Bedford Harbor resuspension rate coincidentally is identical to the resuspension calculations done for the FS. Therefore, based on the updated estimate of total PCBs removed by the selected alternative (150,000 lbs), approximately 91 kg (200 lbs) of Total PCB would be released. Table 336740-3 summarizes the calculations for this analysis of dredging losses.

GE Dredging at Hudson Falls

The last dredging study discussed here involves the removal of Hudson River sediments around the pump house near Hudson Falls. Sediments around this structure were shown to contain percent levels of PCBs, as well as pure PCB oil. Based on a series of cores collected from the area prior to dredging, the sediment concentration of PCB is estimated at 3,670 mg/kg, or 0.367 percent PCB by weight. Dredging was accomplished by diver-directed suction hoses over a total period of about seven months (Oct.-Dec. 1977 and Aug.-Nov. 1998).

During this period, GE conducted its regular monitoring at Bakers Falls and Rogers Island. A map of the area is given in Figure 336740-11. These data can serve to estimate the net release of PCBs from the dredging effort. The supporting calculations are outlined in Table 336740-4.

This effort resulted in the removal of 1,067 tons of sediment at 0.365 percent total PCB by weight. This yields approximately 3,900 kg (8,600 lbs) of total PCB removed. Monitoring data collected by GE at Rogers Island during this period shows little direct evidence of PCB additions to the water column. However, a net contribution from dredging was estimated by estimating the gain in PCB transport between Bakers falls and Rogers Island after correcting for the load gain seen prior to the start of dredging. Based on this, approximately 14 kg of PCBs were released as a result of GE's operations. Dividing this value by the total mass removed yields a PCB mass loss rate of about 0.36 percent. At this rate the removal of 150,000 lbs of PCB would re-release approximately 540 lbs. As noted above, the FS estimate is considered conservative, erring toward a higher value than is likely to occur.

While the dredging technique used by GE is different from that selected for the Hudson by EPA, it is unlikely that the technique was radically different (*i.e.,* substantially cleaner) than that proposed by EPA. Water column concentrations (see Figure 336740-12) inside the silt curtains were frequently higher than 1,000 ng/L and hit over 2,000 μg/L on several occasions (note unit change). Yet water column concentrations at Rogers Island increased little more than 15 ng/L relative to Bakers Falls during this period; thus, little impact was seen downstream despite creating extremely high PCB concentrations within the dredging area. Similar levels of control are anticipated for the removal selected by EPA.

Like New Bedford Harbor, the material dredged by GE had a concentration nearly two orders of magnitude more contaminated than that in the selected remedy. This again suggests that future EPA releases will be substantially smaller, since the river sediment will be less contaminated. Additionally, the GE operation took place within the moving river, just above the dam at Bakers Falls, thus documenting the ability to use silt curtains in portions of the Hudson River.

Summary of Other Field Studies

These three recent dredging projects were examined to determine a dredging loss rate. While this approach is strictly empirical, it offers some potential bounds to the issue. Monitoring at one of the dredging operations (Fox River) was considered inappropriate (*i.e.*, too close to the source) for estimating the true re-release from the sediments. The latter two dredging operations yielded similar rates of PCB release that were also similar to those calculated from EPA's dredging resuspension model. In conclusion, the empirical results from two recent dredging projects provide evidence in support of EPA's FS estimates for PCB loss during dredging.

EVALUATION OF FIELD STUDY DATA – BUCKET DREDGES

Similar amounts of data are available from bucket dredging studies, but these data have not been as extensively evaluated as those from cutterhead dredges. The proximity of the data to the source is also not as convenient as for the cutterhead dredging operations; the operation of bucket dredges make it difficult to get data in the immediate vicinity of the source. There are, however, sufficient data to develop representative resuspension rate values for bucket dredging operations. Since all data are away from the immediate vicinity of the dredging operation, it is assumed that all particles larger than 74 μm have already settled (see equation (1), below). Thus, the resuspension rate was not adjusted for this fraction.

Standard (Open) Clamshell Buckets

A number of field studies have used standard clamshell buckets; these are often referred to as "open" buckets to distinguish them from buckets that are fully enclosed in an attempt to reduce turbidity. These data have been reported and analyzed by a number of authors. Table 336740-5 summarizes the studies used in this paper to estimate resuspension rate values.

Kuo and Hayes (1991) used average sediment-loss rates from the Thames River, St. Johns River, and Black Rock Harbor to calibrate their transport model for bucket dredging operations. Sediment loss rates for these studies are shown in Table 336740-5. Sediment loss rates for the Thames River and Black Rock Harbor are the same as those presented by Kuo and Hayes (1991). Sediment loss rates for the St. Johns River, however, were adjusted for what appears to be an error in the initial concentration used by Kuo and Hayes. Collins (1995) estimates the source strength to be 0.45 kg/sec rather than the 0.31 kg/sec published by Kuo and Hayes. Since an earlier version of Collins' report was the source of this value, it is assumed to be in error. This increases the sediment loss rate to 0.16 percent, more in line with the other studies.

A study of open clamshell dredging in the Calumet River (Hayes *et al.*, 1988) also included scow overflow. Collins (1995) calculated a sediment loss rate of 243 g/sec for the Calumet River field study. Although a production rate is not provided, assuming a full bucket and 50 cycles per hour, the production rate would be 380 m^3/hr. Assuming that the sediment characteristics are the same as those found in the Calumet Harbor field study (*in situ* concentration of 920 kg/m^3), the resulting loss is 0.25 percent.

All of these dredging operations included scow overflow; that is, the sediment scow was filled beyond the initial filling to displace supernatant liquid with sediment and increase the economic load. The supernatant overflows the barge and discharges solids into the water column, thus increasing TSS concentrations in the water column; once in the water column, these solids are not distinguishable from resuspension due to mechanical actions of the dredge.

Hayes *et al*. (2000a) present results from a dredging study in Boston Harbor conducted during 1999. Scow overflow was not allowed during these dredging operations; thus, measured sediment resuspension values result from dredging actions only. The conventional 26-cy bucket removed about two feet of silt plus a foot or so of virgin clay from the 38-ft bottom. The production rate is assumed to be about 1,530 m^3/hr, based upon the dredge operation and bucket capacity. TSS observations during dredging yield a depth-averaged TSS concentration above background of 201 mg/L. The width of the plume was not measured. Considering the short distance between the bucket and sampling location, it is unlikely to be more than twice the bucket width of about three m. Assuming that concentration occurs across a six-m width in a current velocity of 0.17 m/sec, the source strength is about 2.4 kg/sec. Assuming an *in situ* sediment concentration of 844 kg/m^3, the sediment lost to resuspension was 0.66 percent.

All of these studies show higher resuspension rates than the cutterhead dredge studies described previously. Resuspension rates range from 0.16 to 0.66. The results for the Boston Harbor field study are surprising in that they are among the highest value, even though barge overflow was not allowed. The other values seem to be in a reasonable range, particularly considering that barge overflow was included. If overflow accounts for 50 percent of the suspended sediments, the remaining resuspension rates are not substantially different from those for the cutterhead dredges.

The apparent increase in resuspension rate for Boston Harbor may result from the samples being collected much closer to the actual dredging location (within two to seven meters) than in the other studies. TSS concentrations at the source for the other studies were extrapolated from samples collected farther downstream. A substantial amount of the TSS in the Boston Harbor study was near the bottom; without that value, the average TSS concentration and source strength would have been reduced by 30 percent, yielding a resuspension rate of about 0.47. This is much more in line with the other studies. It is likely that these additional solids would have settled in the near vicinity of the dredging operation and not been measured in downstream samples as taken in the other studies.

Resuspension rate values from the open clamshell bucket dredges show a strong relationship with water depth (Figure 336740-13). This substantiates previous theories that sediment erosion from the top of the bucket as it moves upward is a primary resuspension mechanism for standard clamshell buckets.

Enclosed Clamshell Buckets

Data are available for two bucket dredging studies that used enclosed clamshell buckets. The first study was conducted in the St. Johns River at the same location and under the same conditions as the open bucket dredging study described above. Collins (1995) did not estimate source strength for the enclosed bucket operation in the St. Johns River, but did report an estimated TSS concentration at the bucket location of 150 mg/L. The estimated TSS concentration at the open bucket was 285 mg/L; since the conditions are the same, the resuspension rate is proportional. Thus, the representative resuspension rate for the enclosed bucket during the St. Johns River study was 0.27 kg/sec and a sediment loss rate of 0.10 percent. The resulting resuspension rate is 1,000 and includes bucket overflow.

The most recent data were collected in Boston Harbor in August 1999 (Hayes *et al*., 2000a) during the operation of a 39-cy enclosed bucket. The enclosed bucket was a conventional 26-cy bucket converted to an enclosed bucket with a 39-cy capacity. The bucket removed about 2 feet of sediment from the 38-ft bottom with an observed depth-averaged TSS concentration of 50 mg/L. Assuming that concentration occurs across a six-m width in a current velocity of 0.17 m/sec the source strength is about 0.66 kg/sec. The dredge production was about 2,000 cy/hr. Assuming an *in situ* sediment concentration of 844 kg/m^3, the sediment lost to resuspension is 0.22 percent. The associated resuspension rate is 0.22.

EVALUATION OF SOURCE MODELS

Nakai's Source Models

Nakai (1978) proposed the popular TGU method. Nakai's initial formulation and variable definitions were:

$$TGU = \frac{KW_O}{Q_S} = \left(\frac{R_{74}}{R_O}\right)\frac{W_O}{Q_S} = KCg \tag{1}$$

where: W_o = total quantity of turbidity generated by dredging (tons),
C = coefficient depending upon dredge type, soil conditions, etc.,
W_s = total quantity of dredged materials (tons),
TGU = turbidity generation unit, tons/m^3,
Q_s = volume of dredged materials (m^3),
ã = specific weight of dredged materials (tons/m^3),
$K = R_{74}/R_O$,
R_{74} = fraction of particles with a diameter smaller than 74 m, and
R_O = fraction of particles with a diameter smaller than the diameter of a particle whose critical resuspension velocity equals the current velocity in the field.

Since the immediate interest is in using Nakai's approach to estimate the source strength, the appropriate equation form is:

$$W_O = \frac{TGU(Q_S)}{(R_{74} / R_0)} = TGU\left[\left(\frac{R_O}{R_{74}}\right)Q_S\right] \tag{2}$$

At this point, Nakai redefined W_O as the rate of turbidity generation in kg/sec rather than the units of tons as he did in the previous equation. This requires Q_s also be redefined as the volumetric rate of sediment removal (m^3/sec). Although easy to use, the R_0Q_S/R_{74} term has fundamental problems. First, there is the issue of incompatibility between the weight-based fractions R_0 and R_{74} and the volumetric flowrate Q_S. While troublesome, the gross nature of what is trying to be accomplished minimizes its impact. The term R_0Q_S, as defined by Nakai, represents the sediment mass (or volume) with a settling velocity sufficiently low that they will theoretically stay in suspension forever. While there are difficulties with the practicality of defining R_0, the concept is theoretically sound. However, the $1/R_{74}$ term increases as the average particle size increases (*i.e.* R_{74} decreases), thereby adjusting the rate of resuspension in the wrong direction.

Nakai determined W_O during dredging operations by measuring TSS along laterals normal to flow at 30 m and 50 m downstream from the dredging operation; the original manuscript describes the approach in detail, but does not provide details of the dredging projects investigated. He calculated the total mass of turbidity as:

$$W_O = C_{avg} BHU \tag{3}$$

where: C_{avg} = average concentration of TSS (kg/m^3),
B = width (m),
H = water depth (m), and
U = water velocity (m/sec).

Empirical Source Models

Several authors have developed empirical source strength models for cutterhead dredges that consider dredge-operating parameters (Hayes 1986; Crockett 1993; Hayes, *et al* 2000b; and Wu and Hayes 2000). The latest versions of these models, based upon 387 observations from a number of dredging sites, are:

$$DM: \qquad \hat{g}(\%) = \frac{(C_S t_C)^{0.676} V_S^{\ 2.008}}{10^{3.647} L_S^{\ 13.899}} \left(\frac{A_E}{d_C}\right)^{14.575} \left(\frac{Q}{D^2}\right)^{0.805} \tag{4}$$

$$NDM: \quad \hat{g}(\%) = 10^{-3.3293}\left(\frac{A_E}{L_S d_C}\right)^{13.503}\left(\frac{Q}{D^2 V_S}\right)^{0.388} \tag{5}$$

where: $\hat{g}$ = predicted rate of sediment suspended by the cutter and transported away from the dredging operation as a fraction of sediment mass dredged (%),
C_S = *in situ* sediment concentration (g/L),
t_C = thickness of cut (m),
A_E = cutter surface exposed to free water (m^2),
V_S = swing velocity at the tip of the cutter (m/sec),
d_C = diameter of cutter (m^2),
Q = volumetric flow rate through dredge (m^3/sec),
L_s = dredge stepping distance (m); and
D = sediment inlet pipe diameter (m).

The modified *DM* model, which is based upon the individual variables that affect dredging operations, resulted in an R^2 value of 0.588. An R^2 value of 0.470 was determined for the modified *NDM* model, which is based upon non-dimensional groups of the same variables. Although these models are empirically sound, they have several substantial drawbacks: a) they apply only to conventional cutterhead suction dredges, b) the forms of the empirical equations do not allow reliable extrapolation beyond the range of data used to develop them (12-inch to 20-inch dredges), and c) the equations require more knowledge of the dredging operation than is usually known prior to the initiation of dredging. Most readers trying to apply the models lack the knowledge of dredging operations to make reasonable estimates of the operating parameters.

Collins (1995) developed models to estimate the dredging-induced resuspended sediment concentrations near the dredge as a function of the dredge, dredge operation characteristics, and sediment properties. An approach similar to the empirical models shown in equations 4 and 5 was used to develop models for cutterhead and bucket dredging operations. However, these models also require considerable knowledge of the dredging operation and Collins described them as preliminary, unverified models.

OPERATIONAL CONTROLS

Operational controls are popular for environmental dredging projects. Hydraulic dredges, in particular, often have limits on swing speed, cutter rotation speed, and cutting depth imposed. Controls on mechanical dredges are often in the form of limits on bucket fall and raise velocities and total cycle speeds. Both types of restrictions result in lower production rates as a tradeoff in an attempt to reduce water quality impacts. Both are based upon research showing that these operational factors influence sediment resuspension and, probably, toxic constituent releases.

A closer look at the research shows that the concerns arise from extreme operating parameters and that normal operational ranges do not typically result in disproportionate increases in sediment resuspension. It is also not clear that such controls result in an overall decrease in toxic constituent release. For example, increasing the raising speed of a dredge bucket increases the acceleration force applied to sediments in the bucket. Leakage of sediment-laden water from the bucket likely increases due to this acceleration. However, the leakage occurs for a shorter period and it is possible that a longer raising time could result in more mass release into the water column. In essence, an inappropriate operational control for bucket dredges could increase the total mass released during a removal operation. Operational controls for hydraulic dredges tend to reduce the concentration of sediment being pumped from the site. Reduced production rates decrease the productivity of the dredging operation, increase the water that must be treated, and, as shown in Figure 336740-1, may increase the dissolved contaminant release.

Operational controls are an effective means to ensure that careless dredge operation does not lead to excessive losses. Operational controls should focus on avoiding extreme conditions and encouraging "typical" operations that are more efficient.

SUMMARY

After carefully considering the comments received on dredging, EPA concludes that the information contained in or referred to by those comments does not justify increasing or decreasing the sediment resuspension estimates contained in Appendix E.6 of the FS. The estimates of resuspension at the dredgehead of 0.3 percent for mechanical dredges and 0.35 percent for cutterhead dredges are well supported by the above discussion. The associated downstream transport estimates of 0.13 and 0.065 percent, respectively, represent conservative estimates of the potential releases due to dredging and are consistent with direct observations made on several sites.

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Human Health and Ecological Risk Reduction Under Phased Implementation
363176

WHITE PAPER – HUMAN HEALTH AND ECOLOGICAL RISK REDUCTION UNDER PHASED IMPLEMENTATION

(ID 363176)

ABSTRACT

Risks to human health (cancer and non-cancer) and ecological receptors (river otter and mink) were calculated for the selected remedy, REM-3/10/Select, assuming four scenarios: 1) a five-year dredging schedule with no resuspension (same as Feasibility Study and Proposed Plan); 2) a five-year dredging schedule with 0.13 percent loss of PCBs due to resuspension; 3) a six-year dredging schedule with 0.13 percent resuspension loss (base case); and 4) a six-year dredging schedule with 2.5 percent resuspension loss. Cancer risks, non-cancer health hazards, and ecological toxicity quotients are essentially the same for all four scenarios. In addition, the modeling shows essentially no difference among the four scenarios with respect to meeting risk-based remediation goals and other target concentrations of PCBs in fish are met. These results indicate that implementing the selected remedy in two phases over six years does not change the relative risk reduction of the selected remedy from that presented in the Feasibility Study and Proposed Plan. The modeling also shows that, for the remediation goals and other target concentrations that are achieved by 2067 (the extent of the modeling period), the concentrations in fish are reached one or two years later assuming 2.5 percent resuspension loss, compared to assuming no resuspension or 0.13 percent resuspension loss.

INTRODUCTION

This white paper presents the risks to human health (cancer and non-cancer) and ecological receptors (river otter and mink) for the selected remedy, REM-3/10/Select, under four different scenarios. These scenarios evaluate the effects of EPA's decision to implement the selected remedy in two phases over six years rather than in a single phase over five years as described in the Feasibility Study and Proposed Plan. The scenarios also show the effect of different rates of PCB loss due to resuspension on the calculated risk reductions. See White Paper – Resuspension of PCBs during Dredging for additional information on resuspension. Specifically, the four scenarios and their model run designations (e.g., R14S2) are as follows:

- Five-year dredging schedule, no resuspension, same as Feasibility Study and Proposed Plan (R14S2).
- Five-year dredging schedule, 0.13 percent loss of PCBs due to resuspension (R14RS).
- Six-year dredging schedule, 0.13 percent resuspension, base case (R20RS).
- Six-year dredging schedule, 2.5 percent resuspension (R20RX).

The risk assumptions, locations, toxicity values, and receptors used in this White Paper are the same as those used in the Feasibility Study. Risks were calculated with exposure durations (*e.g.*, 40 years for evaluating cancer risks to the reasonably maximally exposed (RME) adult angler, 7 years for evaluating non-cancer health hazards to the RME adult angler, and 25 years for river otter and mink) beginning one year after the year in which dredging will be completed in each river section. Thus, under the modeling assumption that dredging would begin in 2004, risks were calculated beginning in 2008 (five-year scenarios) and 2009 (six-year scenarios) for River Section 1 (RM 189), beginning in 2009 (five-year scenarios) and 2010 (six-year scenarios) for River Section 2 (RM 184), and beginning in 2010 (five-year scenarios) and 2011 (six-year scenarios) for River Section 3 (RM 154). (Note: EPA now expects dredging to commence in 2005. Initiating dredging in 2005 would not be expected to significantly affect modeling projections or the comparative analysis of alternatives.) Exposure durations for the Upper Hudson River as a whole begin in 2009 (five-year scenario) and begin in 2010 (six-year scenario), the mean of the years in which dredging will be completed over the entire 40-mile stretch of the Upper Hudson River.

See White Paper – Model Forecasts for Additional Simulations in the Upper Hudson River for additional information regarding modeling of PCB concentrations in fish in the Upper Hudson. See White Paper – Relative Reduction of Human Health and Ecological Risks in the Mid- and Lower Hudson River for relative risks below the Federal Dam at Troy.

Human Health

Table 363176-1 presents the annual average PCB concentration in species-weighted fish fillet for River Sections 1, 2 and 3 and for the Upper Hudson River (length-weighted) for the modeling period (i.e., 1998-2067). The exposure point concentrations for species-weighted fish fillet used in the human health calculations are shown in Table 363176-2 (Upper Hudson average and by river section).

Cancer Risks

As shown in Table 363176-3, there is essentially no difference among the four scenarios in the cancer risks for the RME adult angler. The cancer risks are either 2.0×10^{-4} or 2.3×10^{-4} in River Section 1 for all scenarios, are either 1.6×10^{-4} or 2.0×10^{-4} in River Section 2 for all scenarios, and range from 4.3×10^{-5} to 5.5×10^{-5} in River Section 3 for all scenarios. Cancer risks for the entire Upper Hudson River range from 8.2×10^{-5} to 9.9×10^{-5} for all scenarios.

The cancer risks for the central tendency (CT), or average, adult angler are lower than those for the RME adult and also show essentially no difference among the four scenarios (see Table 363176-3).

Non-Cancer Health Hazards

As shown in Table 363176-4, there is very little difference among the four scenarios in the non-cancer health hazards for the RME adult angler. The Hazard Indices (HIs) are either 13 or 15 in River Section 1 for all scenarios, range from 13 to 17 in River Section 2 for all scenarios, and range from 3 to 5 in River Section 3 for all scenarios. The HIs are either 6 or 8 averaged over the entire Upper Hudson for all scenarios.

The HIs for the CT adult angler are lower than those for the RME adult and also show essentially no difference among the four scenarios (see Table 363176-4).

Time to Reach Human Health Risk-Based Concentrations in Fish

As shown in Table 363176-5, there is essentially no difference among the four scenarios in the time to reach human health risk-based concentrations of PCBs in fish. In River Sections 1, 2, and averaged over the entire Upper Hudson, the remediation goal (RG) of 0.05 mg/kg PCBs (wet weight) in species-weighted fish fillet is not met by 2067, which is the extent of the modeling time period, for all scenarios due to the continuing upstream source of PCBs. In River Section 3, the RG of 0.05 mg/kg is met in 2051 for all scenarios. Note that the scenarios presented here and in the Feasibility Study all assume an upstream boundary water column Tri+ PCB load of 0.16 kg/day from 1998 to 2004, followed by a step-down reduction to 0.0256 kg/day on January 1, 2005. Should the upstream load be reduced to zero, the RG of 0.05 mg/kg PCBs in fish fillet would be met in each river section within the model forecast period (see White Paper – Model Forecasts for Additional Simulations in the Upper Hudson River).

The 0.2 mg/kg target concentration is not met by 2067 in River Section 1 for all scenarios, is met in 2040 in River Section 2 for all scenarios, is met in either 2014 or 2015 in River Section 3 for all scenarios, and is met in 2024 averaged over the entire Upper Hudson for all scenarios. The 0.4 mg/kg target concentration is met in 2025 in River Section 1 for all scenarios, is met in 2024 in River Section 2 for all scenarios, is met in either 2010 or 2011 in River Section 3 for all scenarios, and is met in 2012 averaged over the entire Upper Hudson for all scenarios (see Table 363176-5).

Reduction in Short-Term Risks

Short-term risks to humans will be reduced through a Site-specific health and safety plan, appropriate monitoring, and institutional controls such as fish consumption advisories and fishing restrictions. A qualitative comparison among the four different scenarios of the selected remedy with respect to short-term cancer risks and non-cancer health hazards from ingestion of fish can be made based on the modeling of PCB concentrations in fish (species-weighted fish fillet) performed to estimate long-term effects. The short-term period begins in 2004 and ends one year after the year in which dredging will be completed in each river section. Specifically, the short-term period ends in 2008 (five-year scenarios) and 2009 (six-year scenarios) for River Section 1 (RM 189), ends in 2009 (five-year scenarios) and 2010 (six-year scenarios) for River Section 2 (RM 184), ends in 2010 (five-year scenarios) and 2011 (six-year scenarios) for River Section 3 (RM 154), and ends in 2009 (five-year scenarios) and 2010 (six-year scenarios) for the entire Upper Hudson.

As shown in Table 363176-5, the modeling predicts essentially no difference among the four scenarios in the short-term period. Only the 0.4 mg/kg target concentration in River Section 3 is met in the short term for all scenarios. The 0.2 mg/kg target concentration and the 0.05 mg/kg RG are not met in the short term for all the scenarios in each river section or averaged over the entire Upper Hudson.

Ecological Receptors

The PCB concentrations in largemouth bass (whole fish), which were used to calculate risk to the river otter, are shown in Table 363176-6 by river section and averaged over the Upper Hudson.

The PCB concentrations in spottail shiner that were used to calculate risk to the mink are shown in Table 363176-07 by river section and averaged over the Upper Hudson.

Ecological Toxicity Quotients

As shown in Table 363176-8, there is essentially no difference among the four scenarios in the ecological risks to the river otter and mink. The Toxicity Quotients (TQs) for the river otter are 5 (lowest-observed-adverse-effect-level [LOAEL] basis) and 50 or 52 (no-observed-adverse-effect-basis [NOAEL] basis) in River Section 1 for all scenarios; 3 (LOAEL basis) and range from 28 to 30 (NOAEL basis) in River Section 2 for all scenarios; and less than 1 (LOAEL basis) and range from 8 to 9 (NOAEL basis) in River Section 3 for all scenarios. The TQs for the river otter are 2 (LOAEL basis) and 17 or 18 (NOAEL basis) averaged over the entire Upper Hudson.

The TQs for the mink are lower than those for the river otter and also show essentially no difference among the four scenarios (see Table 363176-8).

Time to Reach Ecological Risk-Based Concentrations in Fish

As shown in Table 363176-9, the modeling predicts essentially no difference among the four scenarios in the time to reach ecological risk-based concentrations of PCBs in fish. In River Sections 1 and 2, PCB concentrations in largemouth bass (whole body) are not within the ecological remediation goal (RG) range of 0.03 to 0.3 mg/kg (based on the NOAEL and LOAEL for the river otter) by 2067, which is the extent of the modeling time period for all scenarios, due to the continuing upstream source of PCBs. In River Section 3, PCB concentrations in largemouth bass are within the RG range in 2019 or 2020 for all scenarios. Averaged over the entire Upper Hudson, all scenarios of the selected remedy are within the RG range of 0.03 to 0.3 mg/kg in 2035.

As noted above, the four scenarios of the selected remedy evaluated in this white paper assume that the upstream boundary water column Tri+ PCB load is reduced to 0.0256 kg/day on January 1, 2005. Should the upstream load be reduced to zero, the RG range of 0.03 to 0.3 mg/kg PCBs in whole fish would be met in each river section within the model forecast period (see White Paper – Model Forecasts for Additional Simulations in the Upper Hudson River).

The ecological RG of 0.03 to 0.3 mg/kg PCBs in largemouth bass (whole body) developed for the river otter, the piscivorous mammal calculated to be at greatest risk from PCBs, is considered to be protective of all the ecological receptors evaluated. In addition, a range of 0.07 to 0.7 mg/kg PCBs in spottail shiner (whole fish) was developed based on the NOAEL and LOAEL for the mink, which is a species known to be sensitive to PCBs. As shown in Table 363176-9, the modeling predicts essentially no difference among the four scenarios in the time to reach the range developed for protection of the mink.

Reduction in Short-Term Risks

Short-term, temporary impacts to aquatic species and wildlife habitat of the Upper Hudson will be reduced through appropriate mitigation measures and monitoring. A qualitative comparison among the four scenarios of the selected remedy with respect to short-term ecological risks can be made based on the modeling of PCB concentrations in fish (largemouth bass and spottail shiner) performed to estimate long-term effects. The short-term period begins in 2004 and ends one year after the year in which dredging will be completed in each river section, as described above.

As shown in Table 363176-9, there is essentially no difference among the four scenarios in the short-term period. The RG range of 0.03 to 0.3 mg/kg in largemouth bass (whole body) based on the river otter is not met in the short term for all the scenarios in each river section and averaged over the entire Upper Hudson. The range of 0.07 to 0.7 mg/kg PCBs in spottail shiner (whole body) based on the mink is met in the short-term for all scenarios in River Sections 2 and 3 and averaged over the entire Upper Hudson.

CONCLUSIONS

Cancer risks, non-cancer hazard indices, and ecological toxicity quotients show very little or no difference among the four scenarios of the selected remedy. Similarly, the modeling shows essentially no difference among the four scenarios with respect to meeting remediation goals and other target concentrations of PCBs in fish. These results indicate that implementing the selected remedy in two phases over six years rather than in a single phase over five years does not change the relative risk reduction provided by the selected remedy from that presented in the Feasibility Study and Proposed Plan. The modeling also shows that, for the remediation goals and other target concentrations that are achieved by 2067 (the extent of the modeling period), the concentrations in fish are reached one or two years later assuming 2.5 percent resuspension loss, compared to assuming no resuspension or 0.13 percent resuspension.

ENGINEERING FEASIBILITY

Example Sediment Processing/Transfer Facilities
253216

WHITE PAPER – EXAMPLE SEDIMENT PROCESSING/TRANSFER FACILITIES

(ID 253216)

ABSTRACT

In order to reduce the transportation cost and meet the disposal requirement for moisture content, the construction of sediment processing/transfer facilities at suitable sites is necessary to process the dredged sediment.[1] In this white paper, conceptual facility layouts for processing the mechanically dredged sediment and the hydraulically dredged sediment are discussed.

For mechanically dredged sediment, processing will include barge unloading, treatment of excess water, removal of large debris, chemical stabilization, transfer to rail car loading area, and rail load-out. The water treatment would consist of coagulation/flocculation, filtration, and granular activated carbon (GAC) treatment.

For hydraulically dredged sediment, the slurry will be equalized before the dewatering process. Vibrating screens and hydrocyclones will remove the debris and large sandy particles, and the coagulation/flocculation process will remove the fine particles from the slurry stream. The settled solids will be dewatered and the supernatant will be treated in a water treatment unit before discharge to the river. All the solids would be loaded onto rail cars for off-site disposal. The preliminary design of unit processes is also included in this paper.

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[1] It is important to note that EPA has not yet determined the location(s) of sediment processing/transfer facilities necessary to implement the selected remedy. For purposes of the Feasibility Study, example locations were identified from an initial list of candidate sites based on screening-level field observations which considered potential facility locations from an engineering perspective. In the Feasibility Study, it was necessary to assume the locations of sediment processing/transfer facilities in order to develop conceptual engineering plans, analyze equipment requirements, and develop cost estimates for the remedial alternatives. For this purpose, two example locations were identified: one at the northern end of the project area in the vicinity of the Old Moreau Dredge Spoils Area, and one at the southern end of the project area near the Port of Albany. Each of these example locations fulfills many of the desired engineering characteristics for such a facility to support the remedial work, and is representative of reasonable assumptions with regard to distance from the dredging work and cost. Other locations, both within the Upper Hudson River valley and farther downstream, are possible.

The example facility locations presented in the Feasibility Study have also been used in the Responsiveness Summary in order to clarify material presented in the Feasibility Study and Proposed Plan and in connection with additional noise, odor and other analyses that were performed in order to respond to public comments. EPA will not determine the actual facility location(s) until after EPA performs additional analyses and holds a public comment period on proposed locations and considers public input in the final siting decision. Thus, all information provided in this Responsiveness Summary relative to potential impacts of the sediment processing/transfer facilities on communities, residents, agriculture, the environment and businesses should likewise be considered representative and illustrative. Further specific assessment of and, as necessary, mitigation of, potential impacts will be addressed during design.

INTRODUCTION

The construction of sediment processing/transfer facilities is necessary to process the dredged sediments and ship them off-site for disposal. For the purpose of this study, two example locations were identified: a northern sediment processing/transfer facility (NTF) and a southern sediment processing/transfer facility (STF).

The NTF site is, for the sake of this analysis, assumed to be located in the vicinity of the Old Moreau Dredge Spoil Area. The existing site features are shown in Figure 253216-1. While this site has existing rail access, it may require access dredging for barge traffic. The site may be used for mechanical as well as hydraulic sediment processing/transfer operations. The nearest residential properties were identified using aerial photographs.

The STF is, for the sake of this analysis, assumed to be located in the industrial area near the Port of Albany. The site may only be considered for mechanical dredging because of its location (*i.e.*, about 40 miles from Thompson Island Pool [TI Pool]). The site has existing rail and barge access facilities. Site features are shown in Figure 253216-2. The nearest residential properties surrounding the site were identified using aerial photographs.

Both mechanical and hydraulic dredging were evaluated in the FS. Therefore, considering the different characteristics of dredged sediment, the layout of sediment processing/transfer facility will be discussed.

Mechanical Dredging Option

Hopper and/or deck barges will be used to transport the mechanically dredged sediments to the processing/transfer facilities. Barges delivering dredged sediments to the processing/transfer facilities will be secured at an existing or newly constructed wharf or dock. Material in the barges will be unloaded by an excavator. Prior to unloading barges, excess water that has accumulated above the incoming sediments will be pumped off, treated, and discharged back to the river. The water treatment process will include a series of filtration units for solids removal and activated carbon adsorption columns for dissolved PCB removal. It is expected that most of the excess water (*i.e.*, water entrained during dredging operations) will be recovered and treated by this means.

Once the dredged material has been off-loaded, it will be discharged into a hopper through a series of racks and screens that remove larger debris. The dredged material will then be blended with Portland cement (or other similar stabilizing agent) in a pug mill. Stabilized sediments will then be placed into a temporary staging area prior to being loaded into trucks (if necessary) by either conveyors or front-end loaders for delivery to the rail car loading area. At the rail car loading area, the sediment will be placed on the concrete pad and then loaded into the rail cars by front-end loader. It is possible that some in-storage residence time will be required before the sediment’s handling properties improve sufficiently to allow rail load-out. Stabilized sediments will be hauled off-site in covered rail gondolas.

Preliminary Design of Unit Processes

Assuming mechanical dredging (*i.e.*, 20 percent water by volume) is utilized to remove all PCB-contaminated sediments, the necessary 120,000 gallon/day water treatment plant and the associated rail transfer facility will require about 10-15 acres of land area (Montgomery, 1985). The process layout at potential northern and southern sites are shown in Figures 253216-3 and 253216-4, respectively. Assuming a water flow rate of 85 gallons per minute (gpm), the proposed processing/transfer facility will consist of:

a. Filtration

Hydraulic Loading Rate = 2 to 10 gpm/ft^2 (Vesilind, 1997)
Assume 2 gpm/ft^2 for the units
Required filter area = 85/2 = 42.5 ft^2
Assume (5 ft X 10 ft) filter beds
Number = 1

(Additional units will be provided for reliability and backups)

b. Granular Activated Carbon Treatment (GAC)

Hydraulic loading 2 to 10 gpm/ft^2 (Vesilind, 1997)
Required filter area = 85/2 = 42.5 ft^2
Assuming 5-ft diameter columns, the resulting area
$= (\pi/4) * 5^2$
$= 19.63$ ft^2
Number of GAC columns = 42.5/19.63
= 3 units

(Additional units will be provided for reliability and backups)

Hydraulic Dredging Option

The PCB-contaminated dredged sediment will be pumped from the dredging site to the sediment processing/transfer facility, where it will arrive in slurry form. As noted in the FS, the solids content of the dredged slurry is typically 10 – 20 percent by *in-situ* (or, cut) volume. Hydraulic dredging operations produce highly variable slurry flow rates and solids concentrations; therefore, direct dewatering of dredged slurries may not be suitable and temporary storage in a tank or lagoon may be necessary to equalize the flow and the solids concentration prior to the dewatering process.

The primary goal of the processing/transfer facility is to improve the material's handling properties and to reduce shipping and disposal quantities by reducing water content and, hence, weight and volume. A schematic diagram of the slurry treatment process at the potential NTF is shown in Figure 253216-5. The slurry passes through a vibrating screen to remove cobbles, large rocks, and gravel. The slurry then enters a series of hydrocyclones at high velocity and pressure through the feed port and swirls downward toward the apex. The flow reverses near the apex into an upward direction and leaves the hydrocyclone through the overflow. Coarse particles settle rapidly toward the walls and exit at the apex through a nozzle. Fine particles are carried with the fluid to the axial overflow port.

The slurry stream, which now contains mostly fine materials, is then pumped into a coagulation/flocculation process. Chemicals, such as alum, iron salts, or polymers are added to coagulate/flocculate the suspended particles. The settled solids from this process are then dewatered using mechanical dewatering systems. Mechanical dewatering systems have been used extensively to dewater hydraulically dredged materials. A high solids-removal efficiency is desired, because solids that escape from the process represent a route for contaminant loss. However, some solids loss is inevitable; therefore, the effluent stream must be treated for further solids removal. Generally, the mechanical systems can increase the solids content up to 70 percent by weight[2], which will be adequate for subsequent transport and disposal. Dewatered solids can then be loaded onto rail cars for off-site disposal. The supernatant from the coagulation/flocculation process will be pumped to treatment (filtration) units for treatment prior to discharge to the river.

Granular activated carbon columns will be used to remove dissolved PCBs from filtered water. The treatment plant will be sized to handle the entire incoming slurry flow, as well as any additional wastewater incidental to site operation.

Solids generated by the solids separation and water treatment systems will be hauled to off-site disposal facilities. Since Hudson River PCB contamination has been associated with fine-grained sediments (predominantly silts), the coarser fraction of the slurry materials, separated by physical methods as described above, is expected to be relatively free of contamination and may be suitable for beneficial use without further processing. The viability of beneficially using this fraction will be determined during the project's design phase.

Preliminary Design of Unit Processes

As indicated in the FS, assuming hydraulic dredging is utilized to remove PCB-contaminated sediments, the necessary 8.7 MGD sediment- and water-treatment plant (Montgomery, 1985) and the rail transfer facility will require about 15 to 20 acres of land area. The conceptual process layout at the NTF is shown in Figure 253216-5. The processing/transfer facility consists of:

a. Vibrating Screen

Three vibrating screens with a total area of 216 ft^2 are selected
Slurry flow rate = 9,000 gpm
Water velocity through the screen = 9,000/(60 x 7.48 x 216) = 0.1 ft/sec

b. Hydrocyclones

18-inch diameter cyclones: Approximately 700 gpm capacity

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[2] It should be noted that reductions in solids water content are characterized on a percent weight basis in the materials-processing discipline. However, dredging productivity is characterized on the basis of *in-situ* (or, cut) volume, as shown elsewhere in the white paper.

80 percent overall solids removal

Slurry flow = 9,000 gpm
Number of hydrocyclones = 9,000/700 = 13 hydrocyclones

(Additional units will be provided for reliability and backups)

c. Coagulation/Flocculation

The flowrate, Q = 8.7 MGD = 362,500 gph.

Assuming a one-hour flocculation time, the flocculation tank volume:
= 362,500 gallons X ft^3/7.48gallons
= 48,462 ft^3

Assuming a six-foot depth, the area required:
= 48,462/6
= 8,077 ft^2

(Additional units will be provided for reliability and backups)

d. Filter Press

Belt filter system
Throughput = 75 *in situ* cy/hr = 1,800 cy/day

or, Daily throughput = 75 *in situ* cy/hr * 24 = 1800 cy/day
Required volume = 2.65 X 10^6 / (5 * 180)
= 2,944 cy/day
= 3,000 cy/day (approximately)
Required number of units = 3

(Additional units will be provided for reliability and backups)

e. Filtration

Hydraulic loading rate = 2 – 10 gpm/ft^2 (Vesilind, 1997)
Assume 2 gpm/ft^2 for the units
Required filter area = 9,000/2 = 4500 ft^2
Assume (20 ft X 20 ft) filter beds
Number = 12

(Additional units will be provided for reliability and backups)

f. Granular Activated Carbon Treatment (GAC)

Hydraulic loading 2 – 10 gpm/ft^2 (Vesilind, 1997)

Required filter area = 9,000/2 = 4500 ft^2
Assuming 12-ft diameter columns, the resulting area
$= (\pi/4) * 12^2$
$= 113\ ft^2$
Number of GAC columns = 4,500/113
= 40 units

(Additional units will be provided for reliability and backups)

CONCLUSIONS

This paper presents the existing features of two example locations that may be considered as potential sites for the sediment processing/transfer facilities. Preliminary conceptual design and process layouts are presented for both mechanical and hydraulic dredging options at the NTF and for mechanical dredging at the STF.

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Dredging Productivity and Schedule
253090

WHITE PAPER – DREDGING PRODUCTIVITY AND SCHEDULE

(ID 253090)

ABSTRACT

In response to several comments, this white paper addresses the matters of project schedule and dredging productivity. Included in the discussion is a general description of the principal activities that will occur prior to remediation and, as well, an overview of the time frames available to accomplish those tasks. The activities addressed herein include project design, contracting for remedial services, and mobilization by the selected contracting team. It is concluded, that adequate time is available to accomplish the activities that precede remedial work.

Also addressed herein are questions that have arisen concerning production rates achievable by the selected dredging technologies. The commenters' principal criticism has been that production rates planned for the Hudson River have not been attained elsewhere. EPA has reviewed the projects referenced by commenters and has concluded either that production rates achieved at these other sites are consistent with those expected on the Hudson River or that site specific differences make comparisons inappropriate. In light of the assessment of other remedial projects, EPA concludes that production rates expected from the dredging technologies selected for the Hudson River are reasonable.

INTRODUCTION

Several commenters questioned the viability of the schedule on which EPA proposes to accomplish the selected remedy. Issues raised in this regard include the time needed to complete various pre-construction (pre-remedial) activities and the plausibility of achieving targeted dredging rates with the selected equipment technologies. Both pre-remedial activities and dredging productivity are discussed herein; related information may also be found in the White Paper –Delays and Downtime.

There are a number of factors that can influence implementation of the selected remedy, including resolving the matter of who will actually undertake the project. Once EPA issues a Record of Decision (ROD), it is possible for either a potentially responsible party (PRP) or the government to conduct the remedial work. At most Superfund sites, remedial work is accomplished by a PRP.

There can be schedule-related advantages to having the PRP undertake the work. In general, PRPs are able to minimize the complicated procedures that federal agencies must follow to secure construction services. Private parties need not follow the same competitive bidding procedures, labor regulations, and contractor-selection processes that are mandated for federal agencies. In addition, in some cases a PRP may have greater flexibility to divide the project into phases that allow for early initiation of some work, and for modifying contractual relationships as work proceeds through various phases.

Another advantage to the PRP handling the project is that a large corporate entity may have resources on-hand that are directly applicable to accomplishing the selected remedy. For instance, the PRP may have property available to set up processing/transfer facilities and may have staff knowledgeable in several aspects of the remedial work. In addition, a large corporate entity may have direct familiarity with transportation systems in the vicinity of the Superfund site, since it would have engaged those systems for its ongoing commercial needs.

It is not possible to elaborate, herein, on the approach that a PRP would take to implement the selected remedy. The resources and options available to the PRP are not publicly known and, therefore, it would be speculative to discuss a PRP scheduling strategy. Should a PRP conduct the work, EPA would monitor construction progress and assess the PRP's conformance with the agreement reached or order issued for conduct of the project. Among the matters over which EPA would maintain careful oversight is the PRP's adherence to the overall implementation schedule approved or ordered by EPA.

The discussion that follows is premised on the assumption that the government will implement the selected remedy. In that case, the US Army Corps of Engineers (USACE), acting on behalf of EPA, will develop a final design, contract for construction and disposal services, and manage the overall program. The following section presents an assessment of the interrelationships between various activities that must be accomplished before dredging of river sediments can begin. Also presented is an outline of the overall schedule for the principal pre-remedial activities. For purposes of this discussion, it is assumed that EPA issues a ROD on December 31, 2001.

PRE-REMEDIATION ACTIVITIES (AGENCY-MANAGED REMEDIATION)

Sediment Sampling and Analysis

It is assumed that actual in-river removal work will be initiated during the 2005 Champlain Canal operating season and that the work will be carried out over six successive seasons (2005 through 2010), with the first year being a phase-in period. In order to initiate remediation by 2005, a number of pre-construction activities will need to occur, including conducting various project investigations, establishing final sediment cut lines, preparing contract documents, selecting contractors, and contractor mobilization, among other matters. If removal is planned to start during the 2005 construction season, there will be over 36 months to accomplish these pre-remedial activities.

Perhaps the most extensive single investigation that would be accomplished before dredging begins is that associated with identifying removal or cut lines. As proposed in the design support section of the Hudson River Monitoring Plan in the FS, Appendix G, several thousand sediment samples will ultimately be collected for this purpose. However, a number of items should be noted with regard to this particular investigative program. The first is that EPA views the delineation of target removal areas presented in the FS (Plate 17) to be representative of final removal requirements, the reason being that fine-grained sediments (as mapped by side-scan sonar) are the principal source of PCBs to the water column and contain the highest PCB concentrations in the Upper Hudson. Thus, when additional sediment analysis becomes available, it is expected that fine-grained materials will continue to be targeted as detailed in the FS, Appendix D.1.

Secondly, it is not necessary, in EPA's view, to complete the entire sediment-sampling program prior to either selecting a contractor or initiating remedial work. Dredging can be initiated on the basis of partial information from the program while the overall sampling effort is continued and completed. It is, of course, vital to have the latest information in-hand for those areas that are actually undergoing remediation. Finally, it may be possible to initiate some removal work without the benefit of up-to-date sediment chemical analysis. This circumstance would apply to areas being targeted only for navigational purposes and not because of their contribution to the river's PCB problem. Cut lines for these areas can be determined on the basis of bathymetric data. It should be noted that navigational dredging has been included in the project's volumetric removal estimates.

Other key elements of the design will include identification of backfill sources, development and implementation of a public involvement program, siting and design of dewatering facilities, establishment of performance standards and the attendant monitoring program, and development of contractor selection criteria including specifications for the work to be performed. It is expected that design will begin shortly after issuance of the Record of Decision. As mentioned in the following paragraphs, it is also likely that some design activities will continue past the first year of construction. This would be accomplished in a manner that design information specific to a construction season's work would always precede initiation of that work by a sufficient amount of time to ensure proper quality control and adequate time for mobilization.

Construction Contracting

Federal contracting for remedial work can follow any one of several processes. For purposes of this discussion, it is assumed that the government will pre-qualify contracting teams and then distribute bid documents to those found qualified. Of the approximate 30 months available to conduct pre-remedial activities, it is expected that at least half of that time will be expended on the process of selecting a contractor. EPA believes that there is sufficient time in the project schedule for EPA to prepare and issue Requests for Qualifications (RFQs), review contractor qualifications, to prepare and issue bid packages, review responses, and to then select a contractor.

While it is not yet certain as to how the RFQ will be structured, it is likely that EPA will request contracting teams to identify, among other matters, the specific dredging equipment they intend to use to implement the selected remedy, as well as their general requirements for processing facility sites. The bid package provided to qualified organizations will identify cut lines for those areas that are to be dredged during the initial phase of project work (*e.g.*, navigational and remedial dredging during the 2005 canal season). In addition, the bid package will specify performance requirements that must be met by the contractor, as well as various environmental and other constraints that will be imposed on contractor activities (*e.g.*, no trucking of backfill with the Upper Hudson River area). The bid documents will reflect the fact that final cut lines have not yet been established for the entire remedial program and that contract adjustments will, therefore, be necessary when final removal quantities and locations are established.

Dredging Technologies and Processing/Transfer Facility Siting

Selection of a preferred dredging method (mechanical or hydraulic) will depend on EPA's continuing evaluation of technologies, site conditions, and disposal options, as well as other information. The choice of technologies will be made before a contract is awarded. It is expected, however, that work in River Section 3 will be accomplished by mechanical methods irrespective of the dredging method chosen for River Sections 1 and 2, due to the potentially large distances between areas targeted for dredging in River Section 3 and a Southern Transfer Facility.

In order to begin dredging during the 2005 canal season, it will be necessary to have one functioning sediment processing/transfer facility on-line. The principal site development work needed to set up a processing/transfer operation is expected to include rail line improvements, construction of a stabilization facility, and the establishment of a water treatment plant.

Mobilization

In order to start phased dredging operations during the 2005 canal season, it will be necessary for the contractor to assemble a large array of equipment, design the materials handling operation, and arrange for sediment transportation and disposal. Given the scale and complexity of this project, it would be advantageous to provide up to 12 months for the mobilization phase. Assuming 12 months were available for mobilization, then EPA would have approximately 24 months (36 less 12) to select and enter into an agreement, via the USACE, with the preferred remedial contractor.

DREDGING PRODUCTIVITY

Some commenters suggested that actual dredging work, as required to implement EPA's selected remedy, cannot be accomplished in a five-year period. A concern has been expressed that removal will take twice as long – or longer – to accomplish, depending on the dredging technology employed. The response to such comments differs depending on whether mechanical or hydraulic dredging methods are selected for the bulk of the removal work in River Sections 1 and 2 (as already mentioned, it is anticipated that River Section 3 will be dredged mechanically irrespective of which dredging technology is selected for River Sections 1 and 2). Thus, the discussion that follows is specific to each potential dredging technology.

In addition, EPA has decided to pursue a phased approach for implementing the selected remedy. The phasing would involve removal of about 150,000 to 300,000 cubic yards of targeted sediments during the 2005 canal operating season. After this phase, full-scale removal operations would occur during the 2006 through 2009 canal season, then the remaining targeted sediment would be extracted during the 2010 canal season.

The information that follows concerning the productivity that is expected from the selected mechanical and hydraulic dredging technologies is directly relevant to removal rates during the 2005 to 2010 canal seasons. EPA's goal for these years is comparable to that presented in the FS.

Mechanical Dredging – General

EPA has identified a hydraulic excavator fitted with appropriate auxiliary equipment as the preferred technology, if mechanical dredging is utilized, for Upper Hudson River remediation. FS Section 5 details the principal attributes of the excavator-based system that led to its initial selection, and also describes various recent technical innovations that enhance that system's productivity and reduce its environmental impacts. It is recommended that the reader refer to FS Section 5.2.2.1 for additional information on the principal characteristics of excavators and recent advances in that technology.

Estimates of Productivity

Several principal factors that establish overall productivity of a mechanical dredging system have been presented in the FS and are summarized in the following table:

Bucket Capacity:	4 cubic yards	2 cubic yards
% filled per cycle:	80%	80%
Cycle time:	2 minutes	3 minutes
Production rate per hour:	82 cubic yards/hour	27 cubic yards/hour
Fraction of time productive work accomplished:	47 percent of available hours per week	47 percent of available hours per week
Hours of productive work per week:	78 hours	78 hours
Weeks of productive work each season:	30 weeks	30 weeks

The productivity information shown above reflects a range of site-specific factors such as sediment characteristics, river geometry, in-river transportation systems, and environmental constraints. It is worth noting that no commenters questioned the parameters in the above table but, rather, applied information obtained from other Superfund sites to critique the hourly and weekly production rates that were presented in the FS. It is EPA's view that productivity information obtained from one location needs to be fully evaluated before it is applied to another project. Minus that careful analysis of site specific circumstances, comparisons between sites can become an academic exercise and not a serious technical assessment.

In setting production rates for the Upper Hudson, EPA was aware that mechanical excavators are capable of significantly greater outputs than those presented in the FS, particularly in situations where there are few or no environmental constraints. For instance, excavators can readily attain one-minute cycle times, or less, if concerns over resuspension are not reflected in the work. The key point is that there are no inherent equipment-related, mechanical limitations that require cycle times of two or more minutes.

In addition, excavators are able to generate substantial digging force and thereby avoid some of the difficulties that have plagued some crane-mounted, bucket-on-rope systems at other locations. The force generated by these machines is expected to result in greater bucket utilization than has been achieved at several Superfund sites where conventional equipment was employed. Furthermore, the precise positioning possible with excavator-based systems also leads to improved productivity, since less overlap is needed between bites to attain target removal elevations.

Another factor embedded in EPA's productivity estimates that supports the overall FS schedule is the fraction of the work week assumed available for productive work. For purposes of the FS it was assumed that mechanical dredges would function productively 78 hours per week, or about 47 percent of the time. Conversations with contractors have suggested that greater productive use of equipment may be possible with proper planning of work.

Productivity Comparisons with Other Sites

As mentioned above, one commenter compared EPA's productivity estimates to those actually attained at other Superfund sites. While such comparisons may be an interesting academic exercise, it must be noted that either site-specific conditions or the type of equipment actually employed at other sites will often render such comparisons without technical validity. For instance, for a project that entails removal of 50,000 cubic yards of sediment, there would be no purpose to attain the production levels specified for the Hudson River. Also, as has often happened in the past, productivity can be influenced by sediment in-river transport and processing systems, as well as by the dredging equipment itself. It is important that the actual causes of low productivity be identified when comparisons are made between dredging projects.

Saginaw River

Some effort has been expended, by one commenter, to reference productivity attained at the Saginaw River Superfund site in calendar year 2000 and superimpose that same value on the Upper Hudson. It appears that this commenter's entire conclusion on the productivity achievable on the Upper Hudson is based on the outcome of dredging work accomplished at Saginaw during calendar year 2000 and a somewhat narrowly focused assessment of that project.

By one estimate, the mechanical system employed at Saginaw (a conventional crane-mounted clamshell) was able to remove, on average, 981 cubic yards of sediment daily, or about 41 cubic yards per hour. Assuming this estimate is accurate, it should be noted that only one dredging unit was employed for all work at this site. This can be compared to the four excavators, with varying capacities and characteristics, proposed to work simultaneously on the Upper Hudson. In addition, beyond utilizing four excavators, debris removal on the Hudson would be accomplished by yet another piece of equipment, so that work by the main dredges will proceed unimpeded to the greatest extent possible. It is clear that productivity at Saginaw would have been considerably greater if a separate piece of equipment had been dedicated to pulling piles, an operation that did not contribute to sediment removal but did consume time that could have otherwise been used for dredging work (William Rito, pers. comm., June 2001).

Another inefficiency related to having one dredge on the Saginaw site was that apparently six different-sized buckets were employed there (4, 5, 8, and 10 cubic yard conventional buckets and 6 and 16 cubic yard cable-arm buckets). Thus, every bucket changeover that occurred resulted in a complete shutdown of in-river production. Under EPA's Hudson River approach, loss of a single unit would not result in complete shutdown, since three other pieces of equipment would continue to function.

Finally, as mentioned above, the prediction by one commenter that Upper Hudson productivity (mechanical dredging) would be half that estimated by EPA appears to be largely based on averaging the production rate across all buckets used at the Saginaw site. If one were to perform the same calculation for the equipment EPA proposes to use on the Upper Hudson, the result would be an average production rate of 54.5 cubic yards per hour ([82+27]/2) – the 41 cubic yards per hour calculated for Saginaw. When one considers the lack of redundancy at Saginaw (one dredging machine) and the inefficiencies noted above, it is evident that EPA has not been optimistic in its estimates of Hudson River mechanical dredging productivity based on the outcome at Saginaw.

Early Action Assessment

One commenter compared EPA's productivity estimates in the FS to those presented in EPA's early action assessment (USEPA, 1999). The comparison is inappropriate for several reasons. The early action assessment was a relatively quick study that considered, among other alternatives, three dredging options (removal of 238,000 cubic yards, 120,000 cubic yards, and 59,000 cubic yards). To simplify comparisons, the same dredging- and material-management approach was presented for each alternative; *i.e*., use of a single, small mechanical dredge (two cubic yard) for all removal work, and transportation of dredged sediments in shipping containers to final disposal facilities.

It should be fairly evident that a system fashioned to handle 59,000 cubic yards would not be particularly efficient for removal of 238,000 cubic yards. The application of one dredge and shipping containers in the early action assessment is most applicable to the 59,000-cubic yard alternative, but is not considered efficient for larger removal programs.

Finally, it should be noted that the output of the small mechanical dredge described in the FS (the small dredge would be fitted with a two-cubic-yard bucket) was estimated to be 27 cubic yards per hour, which translates to somewhat over 60,000 cubic yards per dredging season, assuming 13 hours of production per working day and 180 working days per dredging season. Thus, from this perspective, the early action assessment and the FS are consistent.

New Bedford Harbor

Results of dredging programs at New Bedford Harbor, a Superfund site in Massachusetts, have also been used to draw conclusions with regard to Hudson River equipment productivity by at least one commenter. Until calendar year 2000, all demonstration and production work at New Bedford had been conducted with hydraulic dredges. However, in calendar year 2000, a mechanical system underwent testing to determine its viability for use during the next and largest phase of site cleanup. The mechanical system consisted of a hydraulic excavator fitted with a European-style profiling bucket that generates a relatively large, flat cutting profile. This technology is discussed in Chapter 5 of the FS and demonstrates that dredging equipment has evolved in reaction to environmental constraints imposed at Superfund sites.

Initial reports from the calendar year 2000 demonstration (Lally and Ikalainen, 2001) suggest that dredge performance was significantly improved over prior demonstrations at New Bedford. Production rates of 50-60 cubic yards per hour were achieved with the 4.5-cubic yard profiling bucket. In addition, it was concluded that with further pre-planning (*e.g.,* debris removal), the mechanical system could be expected to attain production levels of 75 cubic yards per hour or greater. The productivity estimates used in the FS are consistent with findings of the New Bedford mechanical demonstration program.

Hydraulic Dredging – General

EPA has proposed a hydraulic dredge system for the Hudson River remedy that is unique (FS, Appendix H). It will be built specifically for the work required under the selected remedy. Its hull dimensions, swing width, hoist speeds, spud and anchor handling, and cutter head arrangement will be fitted to the contours of the river and specific cut depths unique to the Upper Hudson River. Its subsystems will be designed and built to specification for this particular job.

Many dredging projects use off-the-shelf systems. This approach is taken largely because of economic realities and time constraints. Compromises are made on operating equipment that is not specifically designed for the task at hand. The equipment works well, but may not be absolutely ideal for project circumstances. Some systems are pieced together based on equipment availability rather than strictly on performance specifications.

Due to these compromises, the dredge must be adapted to work in the specific job environment. For example, a cutter head may be designed for use in a deep 20- to 45-foot channel, but may need to work in a shallower river environment. Dredges and pumps may also not be tailored to the specific project but rather be reflective of available equipment. Production efficiency may suffer somewhat as the project crew adapts their off-the-shelf system to the individual project. On the other hand, the system proposed for the selected remedy will be designed with actual Hudson River requirements in mind and, therefore, will operate with maximum design efficiency and minimal impact.

For work in the Hudson River, components of the hydraulic dredge, including a swing, ladder, and spud and anchor hoists, will be designed for optimal performance. Anchor booms will be provided to facilitate shifting of swing anchors in the Hudson's shallow water. The proposed cutter head design will allow the dredge to fit optimally to the contours of the Hudson River banks and to the relatively shallow river depth. This precise cutting technology maximizes the operating efficiency of the overall dredge. With minimal movement of the dredge to fit the river location, more production can result with each placement of the dredge head. The dredge can accomplish more in less time simply because it spends more time doing the actual work.

In addition, the hydraulic dredging processing subsystems will be designed for the unique requirements of this particular project. Inadequate materials-processing facilities can limit the productivity levels achieved by the overall project. Therefore, it is critical that the entire dredging/transport/processing system be designed to handle the operational requirements of the project.

Estimates of Productivity

Hydraulic dredging productivity is a function of the rate of dredge advance that is itself largely based on the swinging and stepping and anchoring characteristics of the dredge, and limited by its pumping capacity. Pumping capacity can be modeled using known theoretical values. These time-tested models use values that take into account many variables including friction loss, pump horsepower, material characteristics, and length of pipe.

These factors are analyzed against known historical data, and a theoretical pumping capacity value is found. This value produces a model for a particular project, in this case the Hudson River selected remedy, with a pumping capacity of 300-500 cy/hour. Even accounting for less-than-optimal conditions and for the uniqueness of the relatively shallow Upper Hudson River, the desired production rate of 275 cy/hour is achievable. This productivity estimate is described in greater detail in the FS, Appendix H.

Productivity Comparisons with other Sites

Commenters on EPA's proposed remedy indicated that the proposed dredging schedule would be unachievable. Commenters use a series of claims to support their argument based on dredging production at other sites. One commenter references four different environmental dredging projects as representative of hydraulic dredging projects and as proof of their argument that EPA's schedule is unachievable.

As stated previously for mechanical dredging, production rates at different sites can not simply be compared by the average production rate achieved at a site. It is necessary to take into consideration the type of dredging equipment employed at the site, quantity of equipment pieces, dredging pattern, and many site-specific conditions. Table 253090-1 illustrates missing essential data that was not presented in the commenter's analysis. The commenter's argument uses little information regarding the pipeline and only a smattering of information concerning the dredge and pump. There is no commonality in the hydraulic dredge type implemented at the various sites. Some are auger type; some are cutter head type. All of these variables are integral to an accurate assessment of dredge productivity and comparison to the proposed dredging rate suggested in the FS.

The same four environmental dredging projects noted earlier (Table 253090-1) are used by the commenter to make the argument that the EPA-proposed effective daily dredging hours are unachievable. The reality of dredge production is that certain days may have little or no production because of equipment delays due to maintenance and other operational constraints. At other times, production may continue unimpeded for days, weeks, or longer. EPA uses the concept of the average project day in its production analysis.

This average day analysis incorporates all sources of delay into a loss-of-production factor. In addition, the hydraulic dredge was not assumed to work full time or 24 hours per day, 7 days per week. Allowances were given in the typical dredge-operating week for downtime associated with equipment malfunctions, dredge repositioning, etc. For purposes of the FS, it was assumed that the hydraulic dredge would operate productively 102 hours per week, on average, or approximately 61 percent of the time.

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Delays and Downtime
313398

WHITE PAPER – DELAYS AND DOWNTIME

(ID 313398)

ABSTRACT

This white paper addresses comments regarding various factors that can cause dredging delays and thereby impact the planned remedial schedule. Comments related to delays and downtime can be placed into three categories:

- Constraints in canal operations.
- Adverse weather conditions.
- Equipment-related delays.

The implications of delay-inducing factors are discussed below. It is concluded that most of the conditions that can lead to project delays and downtime can be managed through proper planning of the remedial work. For instance, physical limitations associated with operating within the Champlain Canal system can be overcome by planning vessel movements for evening and other off-hour periods. In addition, the availability of spare equipment would reduce downtime associated with either canal constraints or mechanical malfunctions.

Finally, it does not appear that the potential for weather-related delays is significant and, in fact, based on EPA's review of historic data, it may be possible to extend the working season beyond the planned 30 weeks per year. Additional supporting information related to sources of potential delays and downtime may be found in the White Papers – Dredging Productivity and Schedule, River Traffic, and Rail Operations.

INTRODUCTION

Several commenters raised questions concerning constraints that the Champlain Canal will place on project operations. These canal-related matters include:

- The seasonal and daily operating schedule.
- Vessel navigation interferences.
- Lock-specific delays.
- Debris removal.

Information regarding these matters is presented below.

Canal Operating Schedule

The canal operates approximately 29 weeks per year; for purposes of the FS, it was assumed that the canal will operate 30 weeks per year. EPA plans to consult with the Canal Corporation throughout the term of remedial work and may request that the canal operating season be extended so that seasonal dredging productivity targets can be attained or exceeded. It is expected that the practical limits to dredging each season will be based on weather conditions and the need for the Canal Corporation to conduct maintenance work on the system. These matters are discussed in following sections.

In addition to seasonal constraints, the Canal Corporation also imposes daily limits on passage through the system of locks that form the Champlain Canal. A typical daily lock schedule, over an entire season, is as follows:

7:00 am to 5:00 pm –	May 7 to May 23
7:00 am to 10:30 pm –	May 24 to October 3
8:00 am to 6:00 pm –	October 4 to October 27
7:00 a.m. to 5:00 pm –	October 28 to November 4

However, the Canal Corporation also provides around-the-clock access to commercial operators when adequate notice is given to the canal traffic agent. Since movements of loaded hopper barges to a potential southern transfer facility (STF) and return of empty hoppers to the dredging site may occur on a 24-hour-per-day basis, it will be necessary to request the Canal Corporation to operate the lock system on an extended daily schedule when dredging is occurring.

In-Transit Delays of Barges

One commenter suggested that project delays would be encountered when barges are not available for loading at the dredging site because of in-transit delays. Perhaps the most important project-related river movement is loaded hopper barges being towed to an STF and then returning empty to the work site for reloading. In the FS it was estimated that multiple hoppers would be needed to support operation of each large dredge under the mechanical dredging scenario. Under the hydraulic dredging scenario, hoppers would be used to haul dewatered sediments from a northern transfer facility (NTF) to an STF.

Should the contractor experience difficulty with hopper barge schedules, it would be possible to have additional equipment on-hand that could be placed into service whenever in-transit delays are being encountered. The additional barge(s) could be secured at an STF when not operating on the river. In this context, the availability of additional barges would not be for the purpose of increasing the daily output of dredged material but, rather, would be for the purpose of achieving the planned level of sediment removal for completing the remedial effort in six working seasons.

Delays Due to Lock Congestion

For information regarding congestion at the Champlain Canal System locks, refer to White Paper – River Traffic, which concludes that there is some potential for traffic congestion to occur at Locks 5 and 6 during the months of July and August when pleasure-craft traffic is highest. However, it is expected that a portion of the project's most time-critical movements of hopper barges can occur during off-peak periods (including nighttime hours when the canal locks do not operate for pleasure craft), and thereby limit or avoid congestion delays.

Delays Due to Canal System Maintenance and Other Activities

The Canal Corporation reports that navigation may be impeded by the need to conduct maintenance on canal locks and other facilities. Maintenance of locks is subdivided into several categories, including winter rehabs, navigation season repairs, and preventive maintenance. Major repairs and overhaul of canal systems (winter rehab) and preventive maintenance are performed outside the navigational season, and therefore outside the planned dredging season. Repairs during the operating season are performed on a maintenance cycle and are largely limited to above-water work. Occasionally, emergency maintenance (*e.g.*, removal of logs and debris resulting from high water) may occur during the operating season (USACE, 1990). Also, at times, events such as boat parades and land-based emergencies may impede navigation.

Delays Due to the Presence of Debris

As presented in the FS, the remedial concept calls for debris removal ahead of dredging work so as to limit the impact of debris on productivity. Results of a geophysical survey, conducted in November 1999 and presented in Appendix H of the FS, indicate that most debris that would interfere with dredging work can be detected by various electronic systems and removed prior to the start of dredging in any particular area.

ADVERSE WEATHER CONDITIONS

Commenters suggested that adverse weather may be the cause of delays in the dredging schedule. Adverse weather conditions could include low temperatures, high winds, and precipitation- or runoff-induced high river-flow rates and flooding.

Weather Delays – Temperature

Low temperatures leading to ice formation on both the river and canal locks place a practical constraint on the canal's operating season. From the perspective of dredging and sediment processing operations, low temperatures will primarily impede the transfer and processing of dredged sediments. Operations at the transfer facilities will become difficult as temperatures drop; also barges cannot transit the river when ice blockage occurs.

To evaluate the effect of temperature on seasonal work, data were obtained from the meteorological stations at the Albany and Glens Falls Airports for the years 1991 through 2000 (National Weather Service, 2001). The data are presented in Table 313398-1, which shows the dates of earliest daily average thaw and freeze for the 10-year period. Also presented in the table is the earliest seven consecutive day period when average daily temperatures were above freezing on the approach to spring and the earliest seven consecutive day period when average daily temperatures fell below freezing on the approach to winter.

These data support the possibility of extending dredging operations beyond the assumed 30-week dredging season. For instance, it would appear that, based on temperature, remedial work could occur throughout the entire month of April in most years (assuming other factors did not intervene). It may also be beneficial to the remedial effort to extend the canal's seasonal closing by one or two weeks some years when moderate temperatures are occurring. EPA has had initial discussions with the Canal Corporation regarding this issue. From the perspective of weather limitations, it would appear that productive work could occur for 33 or 34 weeks each season. It should be noted, however, that the actual ability to extend the season will also depend on the Canal Corporation's plans for off-season maintenance.

High Winds

Although high winds cause turbulence and waves in open waters where there is a significant fetch, the Hudson River is relatively sheltered and is not prone to wave formation. Therefore, it is not expected that significant wind-related delays will be experienced during the remedial work.

Delays Due to High Water

The Canal Corporation reports, in published-memo form, instances wherein the system has been closed as a result of either flooding or high river flows. Between the years 1997 and 2001, the corporation issued one Memo to Mariners indicating that the canal system, between Lock C-1 and Lock C-4, would be closed for a few days until water receded to a safer level and debris could be removed (Canal Corporation, 2000a). A subsequent Memo to Mariners indicated that the canal had reopened for navigation (Canal Corporation, 2000b).

Delays Due to High River Flow Rate

To assess the possibility that high river flows (over 10,000 cfs) may halt dredging work, data from the USGS Fort Edward gauging station were examined. As illustrated in Table 313398-2 (data for the years 1978 to 2000), river flow rates in the range of 10,000 cfs are only encountered a fraction of time during the canal operating season. During the May through November season, river flows approach the level where delays could occur only seven times, on average, over the period of record.

As noted, the actual number of delay days is about 5.3 percent of the total available and not 10 percent, as had been estimated by one commenter who had included data for the month of April in the analysis. It should also be noted that dredging work may be possible when flows exceed 10,000 cfs, but the precise flow constraint will need to be established during the project's design phase.

EQUIPMENT-RELATED DELAYS

Several commenters contend that significant downtime will occur from equipment failures. They cite projects such as the first year of the Saginaw River work, where major and minor repairs caused 12 percent downtime.

Saginaw River and other Dredging Projects

The Saginaw River project experienced delays associated with using a single dredge to conduct non-dredging tasks such as debris and pile removal. Considerable production time was lost each time the dredge bucket was switched to accommodate a change in dredging conditions. The situation at Saginaw is further described in White Paper – Dredging Productivity and Schedule.

A summary of dredging projects relevant to the Hudson River project is presented in Appendix A of the FS. In addition to the Saginaw River project, two other dredging projects that are applicable, Bayou Bonfouca and Ford Outfall, have been reviewed. For these two mechanical dredging projects, delays associated with mechanical difficulties were reported to be minimal. During the Ford Outfall project, the reported site-specific difficulties included problems associated with the sediment processing equipment; the problems were quickly resolved when the contractor implemented design modifications to the processing facility. There were no delays associated with mechanical problems reported for the Bayou Bonfouca project.

With regard to hydraulic dredging, it can be noted that at the Fox River site, although only one hydraulic dredge was in use at any given time, three dredges were available to conduct the work. Redundancy will similarly be applied to the Hudson River remediation to minimize downtime due to equipment problems.

Silt Curtains

One respondent refers to problems associated with the use of silt curtains during the Saginaw River project (silt curtains are discussed in Appendix E.5 of the FS). For the Hudson River project, it is expected that use of silt curtains will be in shallower, low-velocity portions of the river where these systems are particularly effective. In addition, as presented in Appendix A of the FS, silt curtains were also used during the Bayou Bonfouca, Cherry Farm, Fox River, Grasse River, Manistique River/Harbor, Sheboygan River/Harbor, Sever Sound, Georgian Bay, Welland River, Thunder Bay, and Collingwood Harbor dredging projects. Few silt curtain-related problems were reported for these projects. The Ford Outfall project experienced silt curtain damage from a passing freighter traveling through a portion of the river that was closed to commercial traffic, with a resulting delay of approximately one to two dredging days.

Processing Bottlenecks

One commenter expressed concern that land-based bottlenecks will occur when dredged material arrives at the processing/transfer facility. Although problems encountered at other Superfund sites related to solids processing and water treatment have been documented, the main problems encountered at these other locations relate to selection of a less-than-optimal processing train and undersizing various processing-system components.

For the Hudson River project, EPA can avoid processing-related difficulties that have been encountered at other sites in several ways. Experiences at numerous other Superfund sites will be given substantial consideration during the design stage of the Hudson River project. Information is routinely shared for such purposes among EPA regional staff, so that experiences at one location can benefit projects occurring elsewhere. Qualified contractors will be expected to have operated similar materials processing/transfer facilities either at other Superfund sites or under conditions comparable to those occurring at Superfund sites.

Finally, all processing systems and components selected for use on the Hudson River PCBs Site will likely have been demonstrated at other locations. By using these and other methods, EPA will be able to avoid the principal processing-related difficulties encountered at other dredging sites.

CONCLUSIONS

Delays and downtime associated with weather, equipment problems, and/or canal constraints are not expected to significantly impact implementation of the selected remedy. Proper planning and design by a qualified team of designers and contractors, and taking maximum advantage of experiences and lessons learned at other Superfund sites, will be expected to minimize the impacts of factors that can cause delays.

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Post-Dredging PCB Residuals
312663

No Figures or Tables

WHITE PAPER – POST-DREDGING PCB RESIDUALS

(ID 312663)

ABSTRACT

Sediment removal depths for the selected remedy were calculated assuming removal of all sediment with Tri+ PCB concentrations of 1 mg/kg or more, plus an overcut into deeper sediment with less than 1 mg/kg (*i.e.*, “clean” sediment). Based on this approach, EPA expects that the residual PCB levels will be less than 1 mg/kg Tri+ PCBs. In addition, in areas that are backfilled with clean material, PCB concentrations are expected to approach non-detectable levels. Based on an evaluation of other environmental dredging projects (*i.e.*, Grasse River, GM Massena, Manistique, Fox River SMU 56/57, Cumberland Bay, New Bedford, Marathon Battery, Black River, Lake Jarnsjon), and an analysis of conditions within the Hudson River, EPA expects that the selected remedy will reduce PCB concentrations in targeted areas of the Upper Hudson by approximately 98 percent.

INTRODUCTION

This paper addresses expected residual PCB concentrations after completion of dredging. Commenters have questioned whether the 1 mg/kg Tri+ PCB residual for the selected remedy can be achieved, citing higher residual concentrations found at other sites. This white paper explains why EPA believes that a 1 mg/kg Tri+ PCB residual is achievable and discusses the site-specific conditions that may have led to higher residuals at some other projects.

As stated in the White Paper – Sediment PCB Inventory Estimates, the mean length –weighted average Tri+ PCB concentration in sediment in Section 1 is 27.2 mg/kg and in Section 2 is 43.1 mg/kg. Using the expected 1 mg/kg residual concentration of Tri+ PCBs, the selected remedy will leave a residual concentration in the target areas of between 96 and 98 percent less than the original concentration of PCBs.

Under the selected remedy, essentially all PCB-contaminated sediments within targeted areas will be removed. Although the goal is to remove all PCBs, EPA conservatively estimated that a sediment veneer containing 1 mg/kg of Tri+ PCBs would remain after dredging due to, for example, redeposition of sediments suspended by the dredge. To isolate these remaining PCBs and to provide a substrate suitable for benthic and fish habitat, clean backfill material will be placed in the dredged areas, as appropriate.

RESIDUAL SEDIMENT PCB CONCENTRATION

Sediment deposition rates in the Hudson River are relatively low and the majority of the PCB contamination occurs within a foot or so of the sediment surface. In most of the Upper Hudson, the PCB-contaminated sediment overlies older sediments that predate the deposition of PCBs. Therefore, by overcutting, it is possible to remove virtually all the PCBs in targeted areas.

The approach taken to estimate dredging depths, which is described in the FS, is conservative in that it overestimates the quantity of material to be removed based on contamination depth from individual Hudson River samples (complete and incomplete cores). If a sample core is complete (*i.e.*, it extends past the depth of 1 mg/kg Tri+ PCB concentration), the 1 mg/kg Tri+ PCB depth is assumed to be the depth of contamination for that area. If the core is not complete (PCB concentration greater than 1 mg/kg at the bottom of the core), one of two approaches is used. For cores with a peak PCB concentration of 100 mg/kg or more, the contamination depth is estimated as the length of the core plus 1.5 ft. For cores with a peak PCB concentration less than 100 mg/kg, the contamination depth is estimated as the length of the core plus 1 ft.

Within an individual target area, the dredging depth was determined by the sample with the greatest depth of contamination. In addition, minimum cut depths were assigned to further ensure that essentially all contamination within that target area would be removed. Based on the average depth of contamination, a minimum dredge cut of 2 ft was assigned to the 3 g/m^2 Tri+ PCB mass per unit area (MPA) and 2.5 ft for the 10 g/m^2 Tri+ PCB MPA target areas.

In addition to using conservative estimates of dredging depths, EPA included overcutting in the calculation of dredging volumes in River Sections 1 and 2. For these river sections, dredging volume was estimated based on removing sediment to a specific elevation or surface and then defining a trapezoidal cross-section of sediment to be removed (Master Comment 313224, Chapter 4 of this RS). Essentially, the area between two contour lines (*e.g.*, 105 and 106 ft above datum) formed a removal trapezoid. For each removal trapezoid, dredging volume was based on the deeper of two adjacent contours. Thus, on the average, in every removal trapezoid, the overcut depth is approximately 0.5 ft.

After dredging, clean backfill will be placed on many targeted areas. To estimate the effective PCB concentration after backfill, EPA assumed that backfill materials would be mixed uniformly with residual sediments. This is a conservative assumption. More likely is that only the bottom layer of backfill will mix with the residual sediment below. Therefore, in areas that are backfilled with clean material, PCB concentrations are expected to approach non-detectable levels. Backfill is not planned for the channel areas (*i.e.*, water depths of 12 feet or greater).

Residual PCB Concentrations at Other Dredging Projects

Results achieved at several environmental dredging projects are presented here. Site-specific circumstances that are not directly applicable to the Upper Hudson appear to be the principal reason for higher residual concentrations for some of these projects.

Grasse River

The Grasse River dredging operation was essentially a hot-spot excavation. The goal of the project was to remove 25 to 30 percent of the total PCB mass from the Grasse River study area; there was no specific goal on the residual PCB level (Alcoa, 1995). The combination of a mechanical excavator and a horizontal auger dredge was selected for use in one area (Area A). In addition, three-inch suction hoses manifolded directly to the horizontal auger pump were used to remove sediment within a second area (Area B). After dredging, an overall arithmetic average of 75 mg/kg was detected within the remaining sediment located in Area A, with concentrations ranging between 1.1 mg/kg and 260 mg/kg. In addition, up to 130 mg/kg PCBs (average PCB concentration of 108 mg/kg) was detected in Area B. Relative to original conditions, these results represent an overall decrease in average PCB concentration for both areas (*i.e.*, 93 percent reduction in Area A and 64 percent reduction in Area B) (Alcoa, 1995). Post-dredging bathymetric and bed-sampling measurements demonstrated high solids and PCB mass removal, which indicated that the targeted PCB removal was achieved (Thibodeaux, 2000). The presence of boulders significantly interfered with and reduced the efficiency of removal operations. It was concluded that the rocky nature of the river bottom created a less-than-ideal environment for the dredge and, quite frequently, the auger would deflect off obstructions, which limited its ability to remove sediment (Alcoa, 1995).

Project goals and site conditions differed significantly from those of the Hudson River. In the selected remedy, high PCB concentrations in Upper Hudson River sediments are associated with recent deposits of silty materials. Rocky areas were not selected as target areas based on side-scan sonar data and other information. Therefore, problems caused by the presence of boulders and outcrops will be avoided. The post-removal PCB concentrations observed in the Grasse River project include PCB-laden sediments that were targeted for removal but were left because they could not be withdrawn by the dredge's suction. These conditions are not considered directly relevant to the Hudson River, given the manner in which areas are targeted for removal under the selected remedy.

GM Massena Project

The GM Massena project had a cleanup goal of 1 mg/kg total PCBs. Although over 99 percent of the contaminated PCB sediment mass was removed from the St. Lawrence River at the GM site, the 1 mg/kg goal was not met in some areas. In five of six quadrants, the average post-dredging concentration was 3 mg/kg, and in Quadrant 3 the average post-dredging concentration was 27 mg/kg (General Motors Powertrain, 1996). But when considering the relatively high pre-dredging concentrations within these sub-areas (208 mg/kg and 2,170 mg/kg), the reduction can be estimated at 98.6 percent for five quadrants and 98.8 percent for Quadrant 3 (Kelly, 2001). Similar to Grasse River, the inability to reach the cleanup goal in some areas was attributed to the presence of a hard till layer beneath the targeted sediments, which limited the ability to overcut into clean material.

The percent reduction expected for the selected remedy for the Hudson River is very similar to that achieved in the GM Massena project. The ability to overcut soft sediments in the Hudson River, which was not possible at GM Massena, increases the likelihood that a PCB percent removal of 96% to 98% will be achieved. In addition, dredging technology has improved since the GM Massena removal in 1995.

Manistique River

At the Manistique River site, PCB concentration increased with sediment depth, with levels of 16.5 mg/kg in the top 3 inches, 77.5 mg/kg in the 3- to 24-inch range, and almost 200 mg/kg in samples deeper than 24 inches (Thibodeaux, 2000). This differs from the Hudson, where most PCB mass is within the top 9 inches of sediment. The presence of extensive contamination at depth increased the difficulty of attaining a low residual at Manistique River, and also increased the likelihood of mixing of contaminants with residuals.

The cleanup goals for this project were removal of 95 percent PCB mass and an overall residual layer concentration of less than 10 mg/kg at the conclusion of dredging (Bolen, 2001). Therefore, it should not be expected that residuals approaching 1 mg/kg would be attained here, since the project goal was a substantially higher residual than 1 mg/kg.

Fox River SMU 56/57

Three hydraulic dredges were employed at this site to remove the PCB-contaminated sediments. Target dredge elevations were developed for the sediment bed to correspond to post-dredging PCB concentrations in the surface sediments of less than 1 mg/kg. However, 1 mg/kg was not specified as a hard target (Fort James Corporation, *et al*., 2001). The cleanup objectives were:

- Achieve a surficial sediment concentration of 1 mg/kg PCBs or less, where possible.

- Achieve an average post-dredging surficial sediment concentration of 10 mg/kg or less.

- 90 percent of each subunit should achieve surficial sediment concentrations of 10 mg/kg or less, and the surface sediment concentration should not exceed 25 mg/kg in any subunit.

- Place six inches of clean sand over all subunits that did not attain a surficial sediment concentration of 1 mg/kg or less.

The average post-dredging PCB concentration in the top 4 inches of sediment before backfill with sand was 2.2 mg/kg (the range was from non-detect to 8.5 mg/kg), showing that site-specific goals were achieved. Of 38 post-dredging samples, 11 had concentrations less than 1 mg/kg (Fort James Corporation, *et al*., 2001).

Cumberland Bay

This 75-acre site consisted of underwater areas that contained PCB-contaminated sludge from paper mill operations. The specified target for the project was complete removal of the sludge bed down to the underlying clean sand layer. No specific goal was set for residual concentration. Two-horizontal auger dredges were used and the depth of dredging was based on visual observations: If fine sludge was observed, dredging would continue; if only sand was observed, dredging would stop. The pre-dredging PCB concentrations ranged from 10 mg/kg to 3,000 mg/kg, according to the NYSDEC project manager (Dolata, 2001). Based on further information provided by NYSDEC (Ports, 2001), it was estimated that the average pre-dredging concentration was 135 mg/kg PCBs.

After dredging, 115 confirmation cores were collected. Analysis was not performed for 73 of the 115 cores as a result of either the collection point being located on shore (5 cores) or the core materials being visually verified to contain only sand (68 cores). The remaining 42 cores yielded 51 samples that were analyzed for PCBs. The results ranged from 0.04 mg/kg to 18.0 mg/kg and averaged 5.87 mg/kg. If sand cores were included, the average residual concentration could be as low as 2.5 mg/kg (assuming the PCB concentration in the sand cores is 0 mg/kg).

New Bedford

A mechanical dredging demonstration project with a cleanup goal of 10 mg/kg residual PCBs was performed at New Bedford during the summer of 2000. Based on information obtained from the US Army Corps of Engineers site representative, pre-dredging depth-averaged sediment PCB concentrations were 857 mg/kg for 0 to 1 ft, 147 mg/kg for 1 to 2 ft, and 26 mg/kg for 2 to 3 ft (Simeone, 2001). Based on post-demonstration samples, the average PCB concentration was reduced to 29 mg/kg in the top 1-ft layer. The reduction was calculated to be 96.5 percent using the 0 to 1 ft pre-dredging concentration and 91.5 percent using the average concentration of 0 – 3 ft. Similar to Manistique, at New Bedford the project goal for residual contamination was substantially higher than 1 mg/kg. While a direct comparison to the Hudson River is not possible, the percent reductions achieved at New Bedford are similar to those expected for the selected remedy.

Marathon Battery

The Marathon Battery Superfund site is located along the Lower Hudson River near the city of Cold Spring, New York. The site includes a former nickel-cadmium battery plant (operating from 1952 to 1979), the Hudson River in the vicinity of the City of Cold Spring pier, and a series of backwater areas. The site is composed of three study areas. Remediation of Areas I and III included environmental dredging.

The primary cleanup goal for Area I was dredging sediments with greater than 100 mg/kg cadmium (Tames, 2001). In Area I, the average cadmium sediment concentration was 27,799 mg/kg. Post-remediation results for Area I indicated a mean residual sediment concentration of about 12 mg/kg, or a 99.6 percent reduction from average pre-dredge concentrations.

The goal for Area III was a 95 percent removal of cadmium, with a target goal of 10 mg/kg. To achieve this Area III target, the necessary removal depth was determined to be 1 foot. In Area III, the average concentration was 179 mg/kg. The mean post-dredging concentration of cadmium was 10.9 mg/kg, in Area III which represented a 92 percent reduction from pre-dredging levels. The percent reductions achieved at Marathon Battery suggest that EPA’s expected PCB reduction for Hudson River (96% to 98%) is reasonable.

Black River

The Black River, Ohio was the site of a remedial dredging project in 1989 that focused on removal of sediments contaminated with polynuclear aromatic hydrocarbons (PAHs). The primary cleanup goal was to remove all PAH-contaminated sediment in the main stem of the Black River to “hard bottom.” Overall, this dredging project successfully met the target goals of removing PAH-contaminated sediments to “hard bottom” (Wisconsin Department of Natural Resources, 2001).

Lake Jarnsjon

The primary constituents of concern at this site were PCBs with a maximum detected sediment concentration of 30.7 mg/kg on a dry weight basis (average 5 mg/kg). The cleanup remedy was to dredge from 0.4 to 1.6 meters of sediments from the lake bottom and dispose of the dewatered contaminated materials in a nearby landfill.

The average post-dredging sediment concentration was 0.06 mg/kg. Although not specified as a target goal, this relatively low post-dredging concentration equates to attainment of a 99 percent reduction in sediment PCB levels. Also, PCB concentrations in fish decreased from 34 mg/kg extractable fat before dredging to 16 mg/kg after dredging. Based on data collected for the two-year period following dredging, declines were observed in PCB levels in sediment, lake water, and fish.

The PCB residual concentrations and percentage of PCB reduction in sediments achieved at Lake Jarnsjon are consistent with those expected for the selected remedy for the Hudson River.

CONCLUSIONS

An average residual of 1 mg/kg Tri+ PCBs, within targeted dredged areas, is a reasonable goal for the Hudson River, considering the river's sediment PCB profile and the approach taken by EPA to establish dredging targets under the selected remedy. The percent reduction in PCB concentrations expected to be achieved for the Hudson (approximately 96 to 98 percent) is approximately at the range attained at other sites, some with more difficult environmental conditions. In addition, placement of backfill material will provide a substantial barrier between residual contamination and the water column, thereby providing effective isolation of dredged areas.

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Estimate of Dredged Material Exceeding TSCA Criteria
424851

WHITE PAPER – ESTIMATE OF DREDGED MATERIAL EXCEEDING TSCA CRITERIA

(ID 424851)

ABSTRACT

The volume of sediment exceeding the Toxic Substances Control Act (TSCA) criteria for disposal in a non-hazardous-waste landfill was reexamined in response to the revised estimate for the Total PCB inventory of the Thompson Island Pool (TI Pool). This reexamination also took into account the volume of overcut (overcut refers to the volume of uncontaminated sediments that are slated for removal in order to ensure attainment of the target residual PCB concentrations). After accounting for both factors, the anticipated percentage of dredged material from the TI Pool to be treated as TSCA-regulated wastes was reduced from 28 percent (as presented in the FS) to 20 percent. Overall, this reduced the percentage of dredged material for the entire remediation exceeding the TSCA threshold from 42 to 38 percent. Actual classification of material for disposal will occur once sediments have been removed from the river and sampled.

INTRODUCTION

A revised estimate of the volume of dredged materials exceeding TSCA criteria (and thereby precluded from disposal in a non-hazardous-waste landfill) was created in response to the revision of the estimates of the mass of Total PCBs to be removed by dredging from the TI Pool (White Paper – Relationship Between Tri+ and Total PCBs and White Paper – Sediment PCB Inventory Estimates). As discussed in these white papers, estimates of both the Total PCB inventory of the in-place sediments of the TI Pool, as well as the Total PCB inventory in the sediments to be removed, have been revised upward. These changes reflect an improved understanding of the relationship between the original PCB measurements conducted by NYSDEC in 1984 for the TI Pool and the likely Total PCB content of the sediment.

Appendix E-12 ("Distribution of Sediment Volume by PCB Concentration, Range in the Thompson Island Pool and Below Thompson Island Dam") of the FS describes the original approach used to estimate the volume of sediments requiring handling as TSCA-regulated wastes. TSCA-regulated wastes, because of their higher levels of contamination, involve substantially greater costs in handling and landfilling; thus, it was important to determine a reliable estimate of the volume of these materials from the available data. This estimate was calculated individually for each river section. As noted in White Paper – Sediment PCB Inventory Estimates, the estimates for River Sections 2 and 3 for the Total PCB inventories and concentrations have remained largely unchanged between the release of the FS and the release of this RS. As a result, no substantive change in the TSCA estimates for the sediments from these river sections is required. However, as also noted in the White Paper – Sediment PCB Inventory Estimates, this is not the case for the TI Pool.

Based on the improved understanding of the relationship between Total and Tri+ PCBs, the estimates of inventory and concentration of Total PCBs in the TI Pool were increased about three-fold. Since Total PCB concentration was directly affected by the revised relationship, the TSCA materials analysis presented in the FS, which depends directly on the Total PCB concentration, required some review. The analysis presented in the FS for the TI Pool was based effectively on the Tri+ value; thus, the estimates for the concentrations of Total PCBs did not include the mono- and di-homologue fractions.

Since the preparation of the FS, EPA has conducted additional analyses of the relationship between Tri+ and Total PCBs for the TI Pool. These analyses are presented in White Paper – Relationship Between Tri+ and Total PCBs. The revised relationship between Tri+ and Total PCBs for the TI Pool is not a linear one and has warranted a reanalysis of the Total PCB inventory of the TI Pool sediments (White Paper – Sediment PCB Inventory Estimates), as well as the volume of TSCA material to be processed.

ASSUMPTIONS

For both this analysis and the FS estimate of TSCA material for the TI Pool, the 1984 NYSDEC sediment-sampling data were used to represent conditions at the time of dredging. No corrections for PCB losses from the sediments during the intervening time were made. Since losses have been documented (USEPA, 1998) and continue to occur, this approach is assumed to provide an upper bound on the actual amount of TSCA material to be generated, based on the assumption that the volume of TSCA material has decreased due to PCB release from the sediments since 1984.

In addition to this assumption, the calculation presented in the FS made several other conservative assumptions, some of which are not repeated in the current calculations. These assumptions were useful during the FS since it was known that the estimates of Total PCBs, based on the reported sum of Aroclors for the TI Pool, represented an underestimate of the Total PCB concentrations for the pool. Thus, the use of conservative assumptions helped ensure that a useful estimate of the TSCA volume for the TI Pool was created. These assumptions included the following:

(a) The length-weighted average concentration in the contaminated sediments at each location was assumed to apply over the entire depth to be dredged. This assumption ignored the effect of the planned overcut intended to ensure attainment of the residual target contamination levels. The additional volume should have served to reduce the TSCA volume, since the contaminated sediments and the clean overcut material are largely removed and mixed together during the dredging process. This approach provides an upper-bound estimate of the concentration to be compared with the TSCA criteria.

(b) TSCA material was defined as any sediment having a length-weighted average concentration greater than 32 mg/kg. This value provides a sufficient margin of safety for the landfills accepting non-TSCA-regulated materials; *i.e.*, the chances of a non-hazardous-waste landfill accepting TSCA wastes are substantively reduced, since the criteria is actually 50 mg/kg.

(c) The percentage of material meeting TSCA criteria was then based on the fraction of 1984 sampling locations included in the areas to be dredged that exceeded the TSCA criteria.

This assumption weighted each sampling location based on the volume associated with it (*i.e.,* the product of its area as associated by the Thiessen polygon analysis and the depth of contamination at the sampling location). In this manner, the TSCA percentage was based on the distribution of the volumes of contaminated sediment exceeding the criteria and not simply the number of samples.

In the calculations presented here, assumption (a) is not used, since it has been possible to better estimate the Total PCB concentrations within the sediments. As a result, the estimate of the volume of sediments exceeding TSCA criteria should be more accurate.

Assumption (b) reflects the reality of the classification process, in that non-hazardous-waste landfills typically do not accept materials over this concentration. This threshold criterion has been retained for the current calculation, although the revised calculation includes the material captured by the overcut. Similarly, assumption (c) was also used in this revision.

REVISED ESTIMATE OF THE VOLUME OF SEDIMENTS EXCEEDING TSCA CRITERIA

As stated above, the approach used for the FS provided an upper bound on the actual amount of TSCA-regulated material to be generated and does not account for the overcut volume. To accomplish the current estimate, the revised Total PCB inventory for the TI Pool was used as a basis. The calculations presented in White Paper – Sediment PCB Inventory Estimates provide an estimate of the local PCB mass at each sampling point in the 1984 survey of the TI Pool. These estimates of mass were converted to an estimate of the local concentration in the dredged material as follows:

- An estimate of the concentration of Total PCBs in the contaminated sediments was generated for each sampling location and corresponding polygon using the length-weighted average Tri+ concentration and multiplying by the appropriate conversion factor (White Paper – Relationship Between Tri+ and Total PCBs). As described in the white paper, the factor varied from 2.2 to 3.8.

- The Total PCBs concentration was converted to a mass-per-unit-area (MPA) value in g/m^2 by multiplying the concentration by the depth of contaminated sediment and the mean density of the core.

- The Total PCB mass contained within the polygon was estimated by the product of the polygon area and the MPA.

- Finally, the concentration of Total PCBs in the material to be removed (contaminated sediment plus overcut) was estimated as the quotient of the Total PCBs mass and the mass of sediments to be removed from the polygon. The mass of sediments to be removed was determined from the bathymetric data and the assigned dredging elevation (Chapter 4 of this RS, Master Comment 313219), thus representing the entire mass of sediment to be removed from the location.

The results of this calculation are presented in Figure 424851-1. This figure shows the cumulative fraction of the sediment volume as a function of concentration. The data are represented in log-scale, with a vertical line corresponding to the threshold value of 32 mg/kg. Approximately 80 percent of the sediment to be dredged falls below this value (to the left in the figure), suggesting that, for the TI Pool, only 20 percent of the material to be removed would require handling as TSCA-regulated waste.

This revised estimate is lower than the upper-bound value of 28 percent originally presented in the FS. This is the result of the revised relationship of Total PCB to Tri+ (which serves to increase the volume of TSCA-regulated material) and correcting for the additional overcut material (which serves to reduce the volume exceeding the TSCA criteria). The net result shows that the original, conservative estimate of 28 percent TSCA-regulated material for the TI Pool was a good estimate of the upper bound, as intended in the FS. The revised value of 20 percent represents a best estimate of the volume exceeding the TSCA criteria. The results for this revision, along with the original estimates presented in the FS, are summarized in Table 424851-1.

The table also contains the original estimates of material exceeding the TSCA criteria for Sections 2 and 3. These are included to facilitate the recalculation of the total fraction and volume of materials for the entire remediation exceeding the TSCA criteria. Note that for both Section 2 and Section 3, the calculations were performed on a Total PCB basis directly. However, due to the limited nature of the data in these sections, it was not possible to estimate a distribution of Total PCB concentrations incorporating the volume of overcut. This limitation was derived from the use of dredge-zone-wide mean values, rather the use of polygonal declustering to estimate inventories. As a result, the extent of overcut could not be incorporated in these sediment estimates and the values represent an upper bound on the volume of material exceeding the TSCA criteria. Note that the volumes in Table 424851-1 represent the entire dredge volume, including overcut. The calculation was limited only in that it was not possible to independently assess the impact of the overcut on the distribution of the Total PCB concentrations.

As a point of comparison, if the Total PCB inventory of the TI Pool is integrated without the additional volume from overcut, the TSCA volume rises to 52 percent of the total, or more than twice that of the estimate given above. A similar effect would be expected for River Sections 2 and 3 volumes exceeding the TSCA criteria. This would bring the values presented in Table 424851-1 for these sections more in-line with those of the TI Pool (*i.e.*, it would reduce them by one half or more). The inability to do this directly limits the current estimate of an upper bound on the amount of TSCA material.

Using upper-bound estimates for all three river sections, the original estimate placed the fraction of dredged sediments exceeding the TSCA criteria at 42 percent of the total. The inclusion of the revised TI Pool estimate reduces this fraction to 38 percent. This value is still considered an upper bound for the entire effort, but it is less conservative than the original value.

CONCLUSION

The revised estimates of Total PCB inventory in the TI Pool were incorporated in the estimate of the volume of sediments to be classified as TSCA-regulated wastes. The estimate of TSCA-regulated wastes for the TI Pool incorporated both the new values for Total PCBs as well as the volume of uncontaminated sediments to be removed as overcut. These revisions led to a decrease in the estimate of TSCA-regulated wastes for the TI Pool.

The revised estimate for the TI Pool (20 percent of the dredged material) is considered to be a best estimate, whereas the prior estimate represented an upper bound. Combining the revised percentage for the TI Pool with the previous upper-bound estimates for River Sections 2 and 3 yields a slightly lower estimate (38 percent) of the volume of TSCA-regulated wastes.

In light of the small scale of this revision, the FS cost estimate has not been adjusted. It is expected that the costs of remediation would decrease comparably (less than 10 percent) if this revision were reflected in the cost estimate.

REFERENCES

USEPA. 1998. Further Site Characterization and Analysis. Volume 2C-A Low Resolution Sediment Coring Report (LRC), Addendum to the Data Evaluation and Interpretation Report, Hudson River PCBs Reassessment RI/FS. Prepared for USEPA Region 2, New York by TAMS Consultants, Inc., Gradient Corporation, and TetraTech, Inc. July.

Rail Operations
312991

No Figures or Tables

WHITE PAPER – RAIL OPERATIONS

(ID 312991)

ABSTRACT

Commenters questioned whether or not the rail infrastructure of the Upper Hudson River region has adequate capacity to support the proposed remedy. Concerns were raised with regard to rail line capacity, rail yard capacity, rail line safety, rail car availability, and rail operations, among other matters.

The principal rail concerns raised by commenters are addressed below. Input to the responses provided herein was obtained, in part, from representatives of the Canadian Pacific Railroad (CPR), the principal rail system that serves the Upper Hudson River valley. Based on Site inspections conducted by EPA and information provided by CPR and other sources, it has been concluded that the existing rail infrastructure has the capacity to support the selected remedy and that rail operations can be conducted safely and efficiently.

INTRODUCTION

For purposes of the analysis described herein and to allow a detailed evaluation of the current rail line capacities and their ability to support the selected remedy, it was necessary for EPA to assume two example locations for the sediment processing/transfer facilities.[1] These locations were assumed at the northern limit of the project in the vicinity of the Old Moreau Dredge Spoils Area and at the southern limit of the project in the vicinity of the Port of Albany. Please refer to the White Paper – Example Sediment Processing/Transfer Facilities for a discussion of processing/transfer facility site attributes.

RAIL LINE CAPACITY

Commenters suggested that CPR's Fort Edward/Albany rail corridor does not have adequate capacity to handle the traffic that would be generated by the proposed remedy. As presented in the FS, it is anticipated that once remedial operations have reached a steady state (full-production levels), approximately 16 rail car loads of stabilized sediments will be generated per day at the northern sediment processing/transfer facility (NTF) and approximately 29 rail car loads at the

[1] It is important to note that EPA has not yet determined the location(s) of sediment processing/transfer facilities necessary to implement the selected remedy. For purposes of the Feasibility Study, example locations were identified from an initial list of candidate sites based on screening-level field observations which considered potential facility locations from an engineering perspective. In the Feasibility Study, it was necessary to assume the locations of sediment processing/transfer facilities in order to develop conceptual engineering plans, analyze equipment requirements, and develop cost estimates for the remedial alternatives. For this purpose, two example locations were identified: one at the northern end of the project area in the vicinity of the Old Moreau Dredge Spoils Area, and one at the southern end of the project area near the Port of Albany. Each of these example locations fulfills many of the desired engineering characteristics for such a facility to support the remedial work, and is representative of reasonable assumptions with regard to distance from the dredging work and cost. Other locations, both within the Upper Hudson River valley and farther downstream, are possible.

southern sediment processing/transfer facility (STF), based on a six-day work week. Thus, approximately 90 rail car loads of sediments would need to be hauled to offsite disposal facilities three times each working week.

Conversations with the railroad's representatives indicated that CPR had acquired the Delaware & Hudson (D&H) (the historical rail operator in the Hudson valley) rail assets with the intention of expanding service along the former D&H corridor. One commenter stated that there are five daily passenger trains and two daily freight trains using CPR's Fort Edward/Albany corridor at this time. Furthermore, this commenter indicated that Amtrak passenger service occurs during daylight hours on the Fort Edward/Albany corridor and that freight traffic is relegated to nighttime hours. Finally, this commenter stated that the CPR/Amtrak relationship was one of sharing existing trackage. Based on these statistics, the commenter concluded that the Fort Edward/Albany rail corridor might have inadequate capacity to serve the project.

During a telephone conversation regarding the conclusions drawn by this commenter, CPR indicated that the Ft. Edward/Albany corridor does have adequate rail capacity (CPR, May 17, 2001). The following items summarize CPR's assessment of the current rail situation in the Upper Hudson River valley:

- There are six passenger trains using the Fort Edward/Albany corridor at this time. In addition, up to 14 freight trains (through and local) move along the corridor on a typical day. The freight train movements currently occur during both daytime and nighttime hours. There is no nighttime "window" applicable to this corridor as is the case, for instance, along the Hudson East corridor south of Albany. CPR believes that the commenter may be confusing the Hudson East rail situation (Amtrak and CSX) with the Fort Edward/Albany corridor (Amtrak and CPR). CPR also indicated that it owns the rail line between Fort Edward and Albany (which extends northward to Canada and southwestward to Binghamton) and does not share this line with Amtrak, but leases its use to Amtrak as part of an overall business agreement.

- Depending on details of the project, it is likely that activities at the example NTF will generate one or, at most, two trainloads of sediment each week. Initially, this material would be hauled, on a daily basis, to the Fort Edward Yard. CPR suggested that project rail cars could either be attached to through freight trains or be assembled into complete trainloads at Fort Edward; also, blocks of project rail cars could be assembled and hauled to the Saratoga Springs Yard for assembly into complete trainloads. CPR expressed the view that best shipping economics would be obtained if regular, complete trainloads were assembled at its yards.

- CPR indicated that the commenter's conclusion that there may be inadequate capacity on the Fort Edward/Albany corridor to handle disposal of dredged sediments generated from the project was inaccurate. CPR indicated that its goal is to expand the use of this line, thereby increasing its profitability. It was concluded by the rail company's representative that this is the reason CPR purchased the rail corridor. In addition, CPR has expended considerable resources improving conditions on the line and is now planning additional upgrades (including new sidings) to improve competitiveness.

Safety – Rail Crossings

It was stated by one commenter that there are 26 at-grade rail crossings between Saratoga and Fort Edward. It was indicated that 19 of the 26 crossings do not have electronic controls or signals; as a result, the commenter expressed concern with safety. CPR stated that the 19 uncontrolled crossings are located on private roads (*i.e.*, on farm property with no general public access) (CPR, 2001b). Furthermore, CPR indicated that all public rail crossings between Fort Edward and Saratoga are equipped with automatic highway warning devices consisting of gates, flashers, and bells.

Sediment Processing/Transfer Facility Site Rail Operations

According to the FS, the NTF will load out approximately 16 gondola rail cars each day. Commenters stated that freight trains would not be expected to move any fewer than 32 loaded rail cars and, as a result, the NTF would have to be capable of storing up to 32 gondola rail cars. However, it was indicated by CPR that they would pick up the daily quota of 16 loaded rail cars and subsequently place 16 empty rail cars at the NTF, as stated in the FS. The 16 loaded rail cars would either be stored at the Fort Edward rail yard or hauled to the Saratoga Springs rail yard and assembled into a complete train.

The FS did not identify on-site storage at the NTF for 32 loaded rail cars and 32 empty rail cars as suggested by commenters. Due to the close proximity of the Fort Edward rail yard to the NTF, the FS assumed that a storage track with a capacity for 16 rail cars would be constructed at the NTF, with daily pickup and drop-off by CPR from their Fort Edward facility. This addresses the concerns expressed by commenters with regard to storage space needed at the NTF for 64 gondola rail cars.

Commenters also questioned the availability of room for an additional 90 project-related gondola rail cars at the Albany and Fort Edward rail yards, given the space currently available at these yards. CPR representatives indicated that various reviewers of the FS had asked them "how much storage would be needed to support the remedial operation." The answer provided by CPR was that "storage for about 90 cars at each location would be sufficient to support the proposed plan." Providing storage for 90 rail cars was not intended to be an upper-bound limit of what is available but rather to be a reflection of what would be appropriate for a project of this magnitude.

Safety, Spills, and Contamination

One commenter stated that the railroad requires their customers clean the exteriors of rail cars and wheels to avoid injuries to their employees. CPR representatives stated that there was no specific policy in this regard and they were unaware of where this information was obtained. CPR did state that, in instances where rail cars visibly have stabilized sediment adhering to their exteriors, they would need to be cleaned (pneumatically or hydraulically) before exiting the NTF. The CPR representative stated that the approach to car cleaning would be an individual customer preference and not a railroad requirement.

The railroad currently manages its operations in accordance with a spill control and response plan, since they presently handle hazardous wastes as part of their daily business operations, and have done so historically. The railroad's spill control and response plan will continue to be applicable and will be followed during implementation of the selected remedy.

Gondola rail cars leaving the NTF or STF with stabilized materials will be covered. Spills are not expected to occur from these cars; however, in the event of an accident, it is not anticipated that the stabilized sediment will disperse from the location of the spill since stabilized sediment has qualities similar to those of soil. Therefore, in the remote event of a release or spill, discharged sediment can readily be addressed by following precautions and procedures outlined in the spill control and response plan.

Rail Car Movement/Turnaround Time

One commenter indicated that a four- or six-week turnaround time would be experienced for the movement of TSCA materials to a TSCA landfill located in Andrews, Texas. Note that this rail car turnaround time impacts the number of gondolas that will be needed to run the project. The six-week turnaround was an estimate provided to the commenter by the Andrews, Texas landfill operator. The CPR representatives suggested that a four-week turnaround time was reasonable, and that this time may be decreased if unit trainloads were being handled. In addition, if a non-TSCA facility is selected within New York State or the surrounding area, a two-week turnaround time for gondola rail cars can be expected.

As previously indicated, project-related freight train movements can occur during day or night hours. Rail movements are not currently limited on the Fort Edward/Albany corridor. Project-related rail movements will be dependent upon the schedule agreed upon with the railroad during the project design phase, as well as in accordance with the rate of sediment processing at either the NTF or STF and subsequent loading of processed sediments into rail cars.

Rail Car Availability and Leasing

Commenters suggested that about 1,100 rail cars would be needed for disposal purposes and an additional 200 rail cars needed for hauling backfill. In addition, commenters indicated that 800 gondolas would be needed for mechanical dredging operations and 1,300 for the hydraulic dredging operations. It was also stated by commenters that CPR has 3,000 gondolas on-hand and that dedicating rolling stock to the project could strain the railroad's ability to serve other customers.

During the phone conversation held with CPR regarding these concerns, CPR representatives agreed that the selected remedy demands a large number of gondolas and could burden the railroad's own car resources. However, CPR also stated that the availability of gondolas depends on economic conditions at the time the equipment is needed. Gondolas are not currently in great demand and are instead being scrapped. At this time, it would be possible to lease gondolas for $350 per month. According to CPR, a viable strategy for the project would be to lease gondolas and not rely on CPR-supplied equipment. Therefore, EPA's contracting team would need to obtain rolling stock from various organizations that specialize in rail car leasing.

Assuming a lease arrangement that averages $400 per car per month, the resulting rail car leasing charge to the project would be about five dollars per shipped ton (assuming an average rail car turnaround time of three weeks). However, actual leasing costs to the project may be less, since it should be possible for others to utilize the project's rolling stock of gondola cars during the winter season when dredging operations are in abeyance. Based on the uncertainties inherent in predicting future market conditions, it is EPA's view that the transportation costs presented in the FS are reasonable for the project's feasibility stage.

Shipping Two Commodities

Comments were also received concerning the logistical complexity inherent in managing two different commodities (TSCA and non-TSCA materials). With respect to this issue, CPR representatives stated that the need to manage rail cars loaded with two different commodities would complicate operations at their various yard facilities. In particular, it will result in considerable additional switching activity at the yards, particularly if it were assumed that the two materials were arriving in a random fashion. Active coordination between the in-river work and the railroad would be needed to reduce inefficiencies that could otherwise occur under these circumstances.

A comment was also received to the effect that there would be a need to increase the number of gondolas leased if the output of TSCA and non-TSCA materials is not relatively uniform. This is due to the fact that there is an expected four-week turnaround for cars going to the TSCA landfill in Texas and only a two-week turnaround for cars going to non-TSCA landfills. The commenter may be technically correct in this regard; however, the number of cars necessary will be influenced by many factors (*e.g.*, final disposal destinations, applicability of the unit train concept, the composition of the contractor's team) that are not known at this time.

Rail Car Loading

One commenter stated that EPA has provided no information or engineering analysis for rail car loading at the sediment processing/transfer facility sites. A possible arrangement of transfer site facilities is provided in the White Paper – Example Sediment Processing/Transfer Facilities, which describes concept arrangements for transfer and processing equipment at two of the several possible facility locations. In the case of the example NTF and STF, possible arrangements for rail facilities are also shown on the conceptual drawings. It can be expected that, based on detailed engineering, the actual layouts will differ from those provided conceptually, since detailed engineering analyses and designs are not normally generated, and are not required, at the FS stage of a remedial project.

REFERENCES

Canadian Pacific Railroad (CPR). 2001a. Personal communication via telephone between Ed Fitzgerald of CPR and John Szeligowski of TAMS Consultants, Inc. May 17.

CPR. 2001b. Personal communication via e-mail between Ed Fitzgerald of CPR and John Szeligowski of TAMS Consultants, Inc. May.

Off-Site Disposal of Processed Sediments
253477

No Figures or Tables

WHITE PAPER – OFF-SITE DISPOSAL OF PROCESSED SEDIMENTS

(ID 253477)

ABSTRACT

This white paper addresses the viability of EPA's plans for disposal of sediments removed from the Upper Hudson River, as described in the FS. Commenters raised questions concerning the availability of non-TSCA landfills with adequate capacity, and the effects of non-uniform flows of TSCA and non-TSCA materials.

Based on communications with landfill operators during preparation of the FS (which are documented in the Administrative Record), it is concluded that adequate commercial landfill capacity is available for disposal of Hudson River sediments. Finally, it is expected that, with proper planning, the selected contractor can manage the flow of dredged material so as not to overburden either sediment transfer logistics or transportation systems.

NON-TSCA LANDFILLS – CAPACITY AND RAIL ACCESS

Sediments with PCB levels below 50 ppm do not have to be disposed in landfills permitted pursuant to TSCA, but can be disposed in conventional landfills approved pursuant to RCRA Subtitle D and State regulations. A comprehensive search was conducted to identify landfills that have both the capacity and adequate rail access to manage Hudson River sediments (Table 4-14 of the FS).

Given that overall project costs are particularly sensitive to transportation factors, it appeared reasonable to attempt to identify facilities in New York State that would be suitable for disposal of non-TSCA sediments. However, the search identified only one landfill that would be a plausible candidate in New York: The BFI Waste Systems of North America, Inc., Niagara Falls landfill (formerly the CECOS landfill). Given the limited capacity at this location, the evaluation of non-TSCA landfill alternatives was then expanded beyond New York State to include nearby Canada, Atlantic region states, and states in the Midwest.

From the expanded search it was determined that most landfills capable of handling Hudson River sediments did not have direct rail access. However, in several situations, the landfill operator indicated that rail-delivered sediment could be off-loaded near the landfill and trucked onto the site for final disposal. If such a landfill were proposed, EPA would require the demonstration of the viability of constructing the rail/truck transfer facility in conformance with applicable environmental regulations.

Finally, several full-service waste management companies that operate disposal facilities in various regions of the country were contacted for the purpose of identifying additional suitable disposal options for non-TSCA dredged sediments. A potential advantage of using a full-service waste management organization is that such a company may have access to a number of different facilities authorized to receive processed dredged material. Thus, they may be well positioned to organize and coordinate a disposal scenario for relatively large projects such as the Hudson River remedy. Additional disposal sites with rail access were, in fact, identified as a result of discussions with national waste management organizations. As a result of these conversations and the numerous other contacts indicated in Table 4-14 of the FS, EPA believes that commercial disposal sites will be found that have both the capacity and transportation infrastructure needed to manage the non-TSCA fraction of Hudson River sediments.

WASTE FLOW

One commenter raised a question concerning characterization of Hudson River sediments for disposal purposes. The question is based on the fact that, in effect, two commodities will be generated during the remedial work: TSCA material (+50 ppm PCB) and non-TSCA material (<50 ppm PCB). It should be noted, however, that since some commercial landfills have selected more conservative TSCA thresholds than those mandated by federal regulation, a concentration of 32 ppm PCB has been used in the FS to discriminate between TSCA and non-TSCA material.

In principle, there are several ways that the issue of waste characterization for disposal can be resolved. One approach is to let the competitive bidding process reach its intended and logical conclusion; *i.e.*, the contractor with the most efficient overall strategy will be selected to perform the work. In order to be successful, contractors will need to weigh the benefits of various sampling and testing strategies against the uncertainty and costs of not knowing where particular carloads of sediment will be transported until those cars are loaded. In selecting their particular strategies, contractors will be able to consider results of EPA's pre-construction sampling program, as well as the entire Hudson River database. Thus, the contractor will be evaluating both an expanding database (as work proceeds) and the risk of some incoming material exhibiting unexpected disposal characteristics.

Alternatively, EPA may categorize some sediments as TSCA materials on the basis of the pre-construction sampling program. This approach will have the benefit of pre-characterizing certain sediments (undoubtedly the most contaminated) as TSCA materials, thereby reducing uncertainties in materials handling and transportation logistics. The contractor will still need to sample each rail car load of non-designated sediment, but the risks of finding TSCA material will be significantly reduced, as will the associated logistical difficulties. Should a limited number of carloads of non-designated material occasionally prove to exceed the TSCA threshold, these loads would need to be segregated and routed for disposition at a TSCA facility.

IMPACT ON RAIL OPERATIONS

One commenter suggested that there will be times when the flow of dredged material will be such that only TSCA-grade sediments will be arriving at the processing/transfer facilities. At these times, there will be a need for additional rail rolling stock because the distance to TSCA landfills is expected to be at least twice that of non-TSCA landfills. There are two matters to consider in this regard:

- As detailed in the previous section, pre-construction sampling data will be used to characterize sediments before removal actually occurs. These data and other information derived from contractor experience will provide a reasonable basis to plan the project's logistical requirements, including the number of rail cars needed to haul sediments to TSCA and non-TSCA landfills.

- Additional rail stock will be required to transport processed sediment when only TSCA materials are being removed from the river. As long as the contractor has some ability to forecast trends in sediment quality, it will be possible to lease additional gondolas for those periods during which only TSCA sediments are expected to be generated. Thus, the contractor would commit to lease a basic number of cars for the full term of the project and then would locate additional rolling stock for those periods when longer-range movements to the more remote TSCA landfills are necessary.

Additional Technology Evaluation
255314

WHITE PAPER – ADDITIONAL TECHNOLOGY EVALUATION

(ID 255314)

ABSTRACT

A number of vendors have submitted additional information on PCB remediation for evaluation during the public comment period. The vendors claimed that their respective technologies, if applied, will be most effective in remediating the PCBs in the Upper Hudson River. Their main concern was that EPA did not consider their technologies during preparation of the FS. The objective of this white paper is to evaluate the applicability of these specific technologies for remediation of PCBs in the Upper Hudson River.

Based on an evaluation of the technical documents presented by the vendors, there is insufficient information to indicate that these three technologies are feasible for remediation of sediments of the Upper Hudson River and that they will achieve RAOs/PRGs within a reasonable time frame.

INTRODUCTION

During the public comment period on the FS, a number of vendors submitted treatability study reports or additional technical information on their PCB remediation products for consideration. The objective of this white paper is to evaluate the applicability of these technologies for remediation of PCBs in the Hudson River (either *in situ* or on-site), as well as post-dredging remediation of the residual mass of PCBs in the river.

The processes may be categorized as proprietary *in-situ/ex-situ* bioremediation technologies. Although these types of technologies were evaluated for their applicability in Chapter 4 of the FS, a detailed evaluation of the submitted material is presented in this white paper. The remediation methods and the vendor names are as follows:

Process Name	Vendor Name
Aerobic/Anaerobic Bioremediation/Encapsulation Technology	Huma-Clean, LLC
Cogen V	TAMCO/KATHER
Anaerobic Bioremediation Technology	Geovation

METHOD

The criteria used to evaluate the technologies are described in Chapter 4 of the FS. Personal communication with the vendors or technical experts and the following databases were used to obtain and search for additional information on the processes:

- Superfund Innovative Technology Evaluation (SITE) program (USEPA, 1999).
- USEPA Hazardous Waste Cleanup Information (CLU-IN) Web site (USEPA, 2000a).
- USEPA Remediation and Characterization Innovative Technologies (USEPA REACH IT) database (USEPA, 2000b).

- Remediation Technologies Network (RTN) Remediation Information Management System (RIMS, 2001) database.

The unavailability of additional information in these databases indicates that the processes have never been independently evaluated by EPA. They are considered on the basis of vendor-supplied information in this white paper, and have been incorporated into Table 255314-1 (updated version of FS Table 4.3).

DESCRIPTION OF TECHNOLOGY

Aerobic/Anaerobic Bioremediation/Encapsulation Technology

The process utilizes a proprietary biological culture (H-101) to mineralize the PCBs in soil. The vendor claims that the culture contains a mixture of humic/fulvic acids and other proprietary substances and attacks the PCB molecules by generating aerobic and anaerobic reactions. The vendor also lists a number of projects successfully completed within a very short time (six months) in which site contaminants included pesticides, BTEX compounds, cyanide, phenols, nitrogen compounds, and organics. Although the vendor stated that their technology can treat PCBs in the Hudson River, no technical information is presented involving PCB remediation.

Efforts were made to obtain additional information by contacting the vendor's company consultant. The consultant confirmed that the process has never been applied to PCB-contaminated sediments and the degradation mechanism outlined in the letter is only based on speculation. She also indicated that the application of Huma-Clean might be a problem under riverine conditions.

Cogen V

The Cogen-V treatment consists of injecting bio-treatment fluids containing viable microorganisms and a wide spectrum of multiple hydrocarbon-oxidizing enzymes into the sub-surface sediments, thus allowing direct contact between sediment and the microbial slurry. A network of PVC pipes and a pumping system would be used to deliver the biofluids to the sediments.

Efforts were made to obtain additional information by contacting the vendor's company consultants. The consultants indicated that the company is working on a similar PCB-contaminated river site in Germany and they expect to publish a report on the results.

Anaerobic Bioremediation Technology

The process uses BioGeoCheMix, a pelletized solid-chemical composition, for the anaerobic bioremediation of organic contaminants such as PCBs, organochlorine pesticides (*e.g.*, DDT, toxaphene) and chlorinated solvents (*e.g.*, PCE, TCE), and inorganic contaminants (*e.g.*, arsenic-based pesticides) and oxidized forms of heavy metals (*e.g.*, hexavalent chromium). BioGeoCheMix pellets are designed to sink in water and into aqueous sediments for the remediation of difficult-to-treat contaminated environments such as river sediments, lake bottoms, and lagoons.

The vendor provided a pilot-scale treatability study report entitled "Anaerobic Bioremediation of PCB-contaminated Sludges and Sediments," which addresses the effectiveness of BioGeoCheMix. The study was funded by the New York State Energy Research and Development Authority (NYSERDA). The project consisted of applying Geovation's BioGeoCheMix (solid pellet) and N-Blend (liquid) products to PCB-contaminated sludge in a landfill cell. The cell was divided into three sub-areas (triplicate). A five-gallon plastic bucket buried in one of the sub-areas was used to serve as an untreated control. The experimental plots were covered with geotextile fabric, capillary mattings, and plastic sheets. Initial and subsequent sludge samples (day 34 and day 242) were collected and analyzed using EPA Method 8082 (congeners and Aroclors) and the Green Bay Method.

The data analyzed by Method 8082 showed a small initial decrease, followed by an increase in PCB concentration in the sludge with time. Although the method is widely used for quantitative analysis of PCB, the author attributed the data anomaly to matrix-related inefficiency of Method 8082 PCB-extraction procedures. This statement was not supported by additional laboratory experiments or literature citations; rather, the laboratory confirmed that there is essentially no difference in extraction process between EPA Method 8082 and the Green Bay Method (Kotas, 2001).

The weight percent of individual congeners did not show any decrease/trend with time that indicates that composition of PCBs (*i.e.*, relative proportion of individual congeners) did not change during the experiment. The PCB concentration in the control increased by about 83 percent between day 34 and day 242 – this increase indicates serious reliability concerns associated with the experimental method and subsequent sampling conducted by Geovation staff. The authors did not provide any control PCB concentration data at the start of the test, which is a standard procedure for such treatability studies. Method 8082 PCB Aroclor data from both the test and control samples also showed a similar increasing trend with time. Therefore, based on Method 8082 PCB analysis data, no meaningful performance evaluation of Geovation products could be made.

The PCB concentrations in the sludge were also measured by the Green Bay Method. The high percent-reduction values of the process were calculated using PCB concentrations from the control; however, the control showed a large variation from day 0 to day 34. The homologue distribution as measured by this method did not show any significant changes. The claim that "a slight shift from higher-molecular weight congeners towards lower molecular weight congeners was observed from baseline event to subsequent events" was not evident from the data and may be attributed to an analytical artifact.

The report did not address the effectiveness of BioGeoCheMix to remediate contaminated sediments, as stated in the report title. The product has been applied to "sanitary" sludge that contains far more organic matter and viable anaerobic organisms than would river sediment.

It is also not clear how the vendor envisions the application of a solid pellet and liquid reagent for *in-situ* bioremediation, as would be used in the Hudson River. For *ex-situ* application, assuming a 20-month PCB half-life (per Green Bay Method data), 12-inch thickness of sediment and a remediation goal of 1 ppm, it would require approximately 1,650 acres of land area for six years in the vicinity of Hudson River Site to treat the sediments.

Given the inconclusive demonstration of the effectiveness of the product to remediate PCB-contaminated river sediment, this technology does not meet the criteria used to evaluate the technologies in Chapter 4 of the FS.

TECHNOLOGY EVALUATION

It is clear from the technology description and the supporting documents presented that the processes mainly rely on indigenous or amended organisms and other nutrients/enzymes/desorption chemicals. Although all three vendors claim that their particular product will be the most effective to fully remediate the PCBs in the Hudson River sediments, they could not provide any relevant data or results in support of their claim. Some of the technology descriptions and PCB degradation mechanisms as reported in the documents were found to be confusing, and contradict the current peer-reviewed literature.

It should be noted that an *in-situ* PCB bioremediation study was conducted by GE on sediments in the Upper Hudson River (Harkness *et al.*, 1993). Six self-contained steel caisson reactors were driven into the river bottom in the vicinity of Fort Edward. The sediments within the caissons were mixed and subjected to varying additions of oxygen, inorganic nutrients, a co-metabolite, and known PCB-degrading bacteria. Test results showed increases in the numbers of indigenous PCB-metabolizing microorganisms in all experimental caissons. In addition, PCB concentrations, normalized to total organic carbon (TOC) content, decreased by about 50 percent in all caissons. However, biodegradation of the remaining PCBs was not observed, which was attributed to slow desorption kinetics. Garvey and Tomchuk (1997) used the molar dechlorination product ratio (MDPR) to demonstrate the infeasibility of complete PCB biodegradation.

CONCLUSION

Based on the PCB remediation literature reviewed in the FS and an evaluation of the technical documents presented by the vendors, there is insufficient information to indicate that these three technologies are feasible for remediation of sediments of the Upper Hudson River and that they will achieve RAOs/PRGs within a reasonable time frame.

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POTENTIAL IMPACTS OF THE SELECTED REMEDY

Potential Impacts to Water Resources
312851

WHITE PAPER – POTENTIAL IMPACTS TO WATER RESOURCES

(ID 312851)

ABSTRACT

Commenters have expressed concern that implementation of EPA's selected remedy will adversely impact Hudson River water quality due to the release of suspended solids, metals, polycyclic aromatic hydrocarbons (PAHs), nutrients and pathogens. Also, some commenters are concerned that oxygen demand exerted by project discharges may reduce river dissolved oxygen (DO) levels. Finally, it is thought that there may be impacts on nearby groundwater resources from project-related activities. The purpose of this white paper is to evaluate the potential water quality impacts of implementing the selected remedy.

Construction of sediment processing/transfer facilities is not expected to generate water quality impacts since construction activities will be strictly controlled and monitored pursuant to EPA's contract specifications. Operation of sediment stabilization/dewatering processes and water treatment systems at the processing/transfer facilities will conform with substantive federal and State requirements and, therefore, no impacts to water quality are expected. Using conservative calculations it has been determined that water column increases of conventional and trace pollutants will not contravene NY State water quality standards. It is concluded that, overall, work on the selected remedy will not significantly impact either the Hudson River's surface water resources or adjacent groundwater resources.

INTRODUCTION

The selected remedy will involve construction of one or more sediment processing/transfer facilities within the Upper Hudson and removal (dredging) of approximately 2.65 million cubic yards of contaminated sediment by either mechanical or hydraulic dredging methods. Several components of EPA's selected remedy have water quality implications:

- Construction of sediment processing/transfer facilities.
- Operation of sediment processing/transfer facilities.
- Removal of targeted sediments by either mechanical or hydraulic dredging equipment.

Each of these components is evaluated below from the perspective of potential water quality impacts. The water quality parameters that are considered in the following sections are those mentioned by various commenters. It should be noted that PCB water quality impacts are addressed in the Response to Master Comment 365942 in Chapter 9.

SEDIMENT PROCESSING/TRANSFER FACILITIES

Construction

As stated in the FS, one or more facilities will be needed to process sediments that have been dredged from targeted areas of the Upper Hudson. Construction of these sediment processing/transfer facilities will be accomplished by conventional construction methods using readily available construction machinery. Based on the selected remedy, the following structures/systems may need to be erected at the sediment transfer/processing sites (it should be noted that any particular site may already support one or more of the components listed here):

- Wharf for unloading sediment-laden barges.
- Various sediment dewatering and stabilization systems.
- Water treatment system.
- Rail storage track and loading area.
- Administrative facilities including offices, laboratory, etc.

EPA will require the sediment transfer/processing facility construction specifications to contain a number of conditions focused on controlling construction-related water quality impacts. In fact, such specifications are now routinely incorporated in contract documents for all significant developmental work. Examples of environmental clauses or conditions that may be included in the construction documents are as follows:

- Clauses that require preparation of a construction-period best management practices document.
- Conditions to control soil erosion.
- Conditions for stormwater management.
- Requirements for proper storage of fuels, lubricants, and any other chemicals.

The above list of contract requirements is by no means exhaustive but, rather, is suggestive of the approach EPA will take to avoid mobilization-period water quality impacts. It should be noted that EPA will directly monitor sediment processing/transfer facility construction to confirm that development of the facilities is actually occurring in conformance with project specifications. Given the fact that the contract documents will specifically address the matter of construction-related impacts on water quality, and that EPA will oversee contractor activities, it is not expected that development of the sediment processing/transfer facilities will result in significant water quality impacts.

Operation

Operation of the sediment processing/transfer facilities will entail a number of activities that have the potential to impact water quality without adequate management. As presented in the FS, it is expected that dredged sediments will be removed from barges at these facilities and then processed in one of several ways to improve the handling characteristics of the material. Several activities at these facilities can potentially impact water quality:

- Conveyance of sediments to the sediment processing/transfer facilities.
- Temporary storage of stabilized or dewatered sediments.
- Loading of sediments into rail cars.
- Discharges from water treatment systems.

Basically, from a water quality perspective, sediment processing/transfer activities can be grouped into two categories: Those that involve direct discharges to surface waters and those that may result in indirect discharges. Point sources such as water treatment system discharges are an example of the first category. It is expected that a water treatment system will be constructed at the processing/transfer sites to manage excess water removed from dredged sediments. In the case of mechanical dredging technology, the water treatment system will be relatively modest, while in the case of hydraulic dredging, the treatment complex may approach 10 mgd throughput.

Direct or point-source discharges are normally regulated via a discharge permit issued pursuant to Section 402 of the Clean Water Act and State law. The discharge permit system has been specifically designed to protect the water resources of locales where discharges are occurring. Discharge permits are not required for facility operations, although point sources at these facilities will comply with substantive requirements of federal and State permits that would otherwise be required. Thus, it is reasonable to conclude that direct or point-source releases from the sediment processing/transfer facilities will not significantly impact water quality in their vicinity. For further information on water treatment systems, refer to White Paper – Example Sediment Processing/Transfer Facilities.

Indirect or non-point source discharges may occur at the sediment processing/transfer facilities in the form of stormwater runoff from processing, storage, and loading areas. If controls are not placed on runoff, it could be possible for sediment-borne contaminants, entrained in stormwater, to enter the river. EPA's approach to controlling non-point sources will be to require the contractor to develop and implement an operating-phase stormwater management plan that will include detailed best management practices that the contractor will follow to control indirect discharges. The stormwater management plan:

- Will require the contractor to direct runoff away from sediment handling and storage areas.
- Will require that contaminated runoff be directed to sediment basins.
- May mandate that certain activities be covered to avoid precipitation and runoff.
- Will require contaminated stormwater to be processed via the same treatment processes that handle excess water removed from incoming dredged material.

Based on EPA's approach to managing indirect discharges, which will also include Agency oversight of the contractor's efforts, it is not expected that indirect discharges from the sediment processing/transfer facilities will significantly impact water quality.

REMOVAL OF TARGETED SEDIMENTS

The process of removing targeted sediments has the potential to introduce sediment-borne contaminants into the Hudson River in the vicinity of work sites. The analysis presented below addresses potential water quality impacts resulting from the introduction of the following conventional and trace constituents into the water column during dredging operations:

- Total suspended solids.
- Nutrients (nitrogen and phosphorous).
- Metals.
- PAHs.

- BOD/COD.
- Pathogens.

Total Suspended Solids

Release of particulate matter to the water column (*i.e.*, total suspended solids, or TSS) during dredging operations has been estimated in White Paper – Resuspension of PCBs during Dredging. A series of mathematical models were presented in the white paper that were used to predict the mass of sediment expected to be resuspended during dredging operations and the extent to which the resuspended material will settle and disperse downstream of the dredge. In addition to TSS load to the water column, the White Paper – Resuspension of PCBs during Dredging also provides an estimate of the PCB mass that would be remobilized during dredging operations.

Water column TSS levels (mg/L) for the hydraulic and mechanical dredging technologies have been estimated in White Paper – Resuspension of PCBs during Dredging and are presented in Table 312581-1. It should be noted that the results presented in Table 312581-1 are in two forms: The "near-field" concentrations are represented as plume TSS concentrations estimated to occur 10 meters downstream of the dredging work. These concentrations do not reflect complete mixing of resuspended sediment into the entire flow of the Upper Hudson. Complete mixing can be expected to occur at some point further downstream of the dredging operations. The fully mixed, "far-field" concentrations are also presented in Table 312581-1. In this instance, the TSS load 10 meters downstream of the dredge head is simply homogenized across the entire river. This calculation represents an upper-bound TSS estimate for the well-mixed flow, since it assumes no subsequent settling occurs beyond 10 meters. TSS levels are calculated for both hydraulic and bucket dredges at similar production rates (*i.e.*, one hydraulic dredge or three bucket dredges in operation). Note that the TSS concentration varies among the river sections due to the variations in sediment type in the section-specific target areas.

In comparison to the levels shown in the table, spring runoff produces increased flows and increased TSS concentrations throughout the entire river. USGS data for Fort Edward show average levels of 13 mg/L in April over the period 1978-1995; Schuylerville averaged 21 mg/L in April from 1977-1989; Stillwater averaged 27 mg/L in April from 1977-1996; and Waterford averaged 40 mg/L in April from 1976-1996 (USGS, 2001). Thus, normal spring runoff produces far greater TSS levels than any increase estimated from dredging operations. Additionally, spring TSS loads encompass the entire river, while dredging operations are projected to increase levels by less than 1.5 mg/L within 10 meters of the dredge. Beyond 10 meters, the incremental TSS levels would be further reduced as water column mixing continues.

Throughout the rest of the year, riverine TSS levels are considerably lower than those observed in springtime, with average concentrations running between 2 mg/L and 3 mg/L. Resuspension from dredging would increase these levels, but only moderately. Given the relatively modest area of influence of the dredging-induced sediment plume, as well as the fact that mixing will further reduce dredging-induced increases, it can be expected that the river will be minimally impacted by sediment resuspension. Note, as well, that subsequent settling of resuspended sediment has not been accounted for here and will also serve to further reduce the impact of dredging. Thus, it is not expected that any downstream users of Hudson River water are likely to be impacted by dredging-related sediment resuspension, and municipal water treatment facilities designed for year-round operation will readily handle the TSS increases associated with dredging.

Nutrients

Nutrients in Hudson River sediments are the results of historic and current point-source (*e.g.*, wastewater treatment plants) and non-point source (*e.g.*, agricultural runoff) discharges. Potential releases of two nutrients, nitrogen and phosphorous, are evaluated in the following sections.

Nitrogen

High-resolution core samples (USEPA, 1997, 1998) and USGS data (Allen, 2001) were used to investigate the nitrogen profile in Upper Hudson River sediments and the water column, respectively. Figure 312581-1 shows the typical total nitrogen profile in the high-resolution core samples in relation to the total PCB profile as a function of depth. The total nitrogen peak occurs in the same or a more-shallow layer than the total PCB peak. Since the project's dredging plan was developed to remove essentially all PCBs within a target area, most of the nitrogen-bearing compounds in targeted sediments will also be removed.

The average sediment release rate, as predicted in the White Paper – Resuspension of PCBs during Dredging, was used to estimate total nitrogen load to the water column. As detailed in that white paper, the average sediment release rate is estimated at approximately 0.13 percent of the overall mechanical dredging rate and 0.07 percent of the overall hydraulic dredging rate at 10 meters downstream of the dredge head. The maximum, median, and average nitrogen concentration (as N) found in 163 high-resolution core samples was 7,300, 2,800 and 2,600 mg/kg, respectively. Based on this data, the average total nitrogen concentration in targeted sediments may be approximated at 3,000 mg/kg. Table 312581-2 presents the estimated water column nitrogen increase, in comparison to background levels.

Based on these calculations, no water quality impacts are anticipated from the resuspension of nitrogen during dredging, since the estimated increase of this nutrient, above background levels, will be negligible (less than 1 percent).

Phosphorous

Dredging-related water column phosphorous levels were estimated using USGS data (Allen, 2001); from that data, the total phosphorous concentration in sediment is estimated to be 1,500 mg/kg. Table 312581-3 presents an analysis similar to that provided for nitrogen. As shown in Table 312581-3, there is not expected to be a concern over water-column phosphorous during dredging operations. Although the percentage increase is greater than that projected for nitrogen, the net increase of two to three percent is still minor, compared to normal river concentrations.

METALS

As discussed in the White Paper – Metals Contamination, dredging will capture most of the trace metals in targeted sediments along with PCBs. However, metals adsorbed to sediment may enter the water column as a result of sediment resuspension during dredging. For each metal considered herein, an estimate has been made of the increase in water column concentration during dredging operations, under NYSDEC-specified low-flow conditions. The increase is calculated using the TSS flux 10 meters downstream of dredging operations; similar to the TSS analysis, far-field losses due to settling have not been considered in this analysis.

Metals identified in the White Paper – Metals Contamination as being of concern and having water quality criteria are evaluated here. The analysis presented here compares dredging-induced water column concentration to NYSDEC standards without consideration of ambient levels. Estimated water-column fluxes 10 meters downstream of the dredge head, for each river section and dredge type, are presented in Table 312581-4.

Knowing the metal flux at 10 meters, it is possible to estimate dredging-induced water column concentrations by applying NYSDEC-specified low-flow conditions (seven-day minimum event once in 10 years - 7Q10, and 30 day minimum flow once in 10 years - 30Q10). This analysis has been conducted for the mechanical dredging system, since fluxes are somewhat higher for this system. Resultant water column concentrations have then been compared to the most stringent NYSDEC water quality criteria applicable to the Hudson River (Table 312581-5).

Except for mercury and lead, estimated metal increases are predicted to be less than 1 percent of the NYSDEC water quality standard at the NYSDEC-specified low flow. Lead is predicted to increase under low-flow conditions to about 5.5 percent of the standard. Mercury would approach 87 percent of the NYS WQ standard for fish consumption (although there is currently a fishing advisory for the study area), 0.1 percent of the standard for protection of aquatic life, and 24 percent of the standard for protection of wildlife. However, since dredging-induced increases are transient and the calculations presented here applied a number of conservative assumptions (settling, flow rate, etc), it is not expected that dredging-induced increases in lead or mercury levels will have a impact on fishery resources.

Although elevated levels of titanium have been reported in Hudson River sediments, NYSDEC does not currently maintain a water quality standard for this constituent.

PAHs

Polycyclic aromatic hydrocarbons (PAHs) are formed mainly as byproducts of combustion, such as fossil-fuel power generation, numerous industrial processes, and forest fires. PAH concentration patterns in sediment may be affected by geographic location, proximity to point sources, and/or the character of nearby land uses.

Data detailing the distribution of PAHs in the Upper Hudson River sediment are limited. USGS' National Ambient Water Quality Assessment (NAWQA) Program conducted a study from 1992 to 1995 in the Hudson River Basin and collected sediment samples for PAH analysis near Stillwater and Waterford. Twenty-seven organic compounds identified as PAHs were found in sampled sediments. Concentrations were significantly higher at Waterford than at Stillwater.

Dredging-induced water column levels of these organic compounds may be conservatively estimated using the available data. Again the analysis focuses on increases that may occur immediately downstream of the dredging activity (10 meters). Tables 312581-6 and 7 provide results for nine trace organics, together with the NYSDEC surface water standards. Although there is no ambient water column data available for PAHs, it is reasonable to conclude that given the predicted low-level increase of these contaminants, there will be no significant impact to water quality during dredging operations.

Dissolved Oxygen

Dissolved oxygen (DO) is a critical water-quality parameter for aquatic organism survival. BOD/COD-bearing constituents such as organic carbon, ferrous iron, and sulfide, found in Hudson River sediments, have the potential to consume water column oxygen.

Both high-resolution core samples (USEPA, 1997) and low-resolution core samples (USEPA, 1998) were evaluated to estimate the total organic carbon (TOC) content in Upper Hudson River sediment. The highest, median, and average TOC concentrations for a total of 188 samples were 11.5 percent, 4.8 percent, and 4.0 percent, respectively. Using these TOC concentrations, and the highest estimated TSS release to the water column during dredging, TOC levels 10 meters downstream of dredging operations have been estimated and presented in Table 312581-8. To estimate the maximum oxygen consumption related to TOC, it was assumed that all the carbon will be instantaneously oxidized stoicheometrically (*i.e.*, 2.67 oxygen to carbon ratio). As shown in Table 312581-8, the oxygen consumption/decrease is predicted to be 0.4 mg/L 10 meters downstream of the working dredge.

No ferrous iron or sulfide data are available for Upper Hudson River sediments. Total iron levels from EPA's ecological sediment samples (USEPA, 1999) and total sulfur in sediment reported by USGS for Upper Hudson River were used to conduct an upper-bound estimate of released ferrous iron and sulfide during dredging. The highest, median, and average iron concentrations for the 31 ecological sediment samples were 2.8 percent, 1.8 percent, and 1.8 percent, respectively. Using the highest concentration, the release of ferrous iron to the water column is estimated to be 0.035 mg/L for mechanical dredging and 0.038 mg/L for hydraulic dredging. Assuming that all ferrous iron is oxidized by the river's dissolved oxygen, the maximum oxygen consumption is calculated to be 0.005 mg/L on the basis of the following reaction:

$$Fe^{2+} + \frac{1}{4}O_2(g) + H^+ \rightarrow Fe^{3+} + \frac{1}{2}H_2o$$

Sediment total sulfur averaged around 0.15 percent in USGS samples. On this basis the maximum possible release of sulfide to the water column is estimated to be 0.002 mg/L (for either the mechanical or hydraulic dredging alternative). Based on the following reaction, the highest oxygen DO reduction is estimated to be 0.004 mg/L.

$$S^{2-} + 2O_2 \rightarrow SO_4^{2-}$$

Surface waters of the Upper Hudson River are designated as Classes A, B, and C by NYSDEC, with an applicable daily average DO standard of 5 mg/L. Additionally, the standard requires that the waters never fall below a value of 4 mg/L. USGS historical data indicate that DO concentrations in the Upper Hudson River are usually above 7.2 mg/L. Therefore, the conservatively estimated net mean DO reduction of 0.2 mg/L resulting from resuspension-derived TOC, ferrous iron, and sulfide releases will not affect maintenance of acceptable DO levels.

Pathogens

The sources of pathogens in sediments can include animal feed lots, dairy farms, combined sewer overflows (CSOs), wastewater treatment facilities, and stormwater runoff, among others. Figure 312851-2 shows the locations of CSOs and point-source discharges along the Upper Hudson and the sole water supply intake at Waterford.

Once introduced or resuspended into the river, pathogens undergo loss ("die-off") due to various natural processes including settling, thermal stress, solar radiation, and predation by zooplankton. Hence, pathogen levels in the water column decrease with distance downstream from the source.

The only water supply intake in the study area is at Waterford, at approximately RM 157, although a second water intake is currently being planned for Halfmoon, NY, also in the vicinity of RM 157. The intake is more than four miles downstream from the most southerly dredging target area. In fact, most of the dredging will occur between RMs 195 and 183, more than 25 miles upriver from the Waterford intake. Given the distance between Waterford and the dredging operation, and the probability that dredging-induced pathogen increases are not likely to be significant, the Waterford water supply will not experience significant pathogen influx. In addition, the Waterford water treatment facility (which includes operations for coagulation, prechlorination, flocculation, two-stage settling, filtration, and post-chlorination) has been designed to remove such organisms.

GROUNDWATER UNDER THE INFLUENCE OF THE HUDSON RIVER

A concern has been expressed about the possible impact of dredging on groundwater resources under the influence of the Hudson River. Studies have shown that in general the Hudson River is a point of groundwater discharge (i.e., groundwater flows into the river). However, it is possible for flow reversals to occur when seasonal water levels are low or near a dam. It is expected that the process of removing targeted sediments will result in an overall reduction in the potential risk of groundwater contamination, since the primary source of PCB contamination will be removed. Given that the typical flow direction is from groundwater into the river, it is not likely that any short-term increase of contaminants in river water generated by dredging operations will impact groundwater resources.

CONCLUSIONS

In summary, Upper Hudson River water quality is not expected to be significantly impacted by implementation of EPA's selected remedy for the following reasons:

- Construction of sediment processing/transfer facilities will follow best management practices, and will include requirements to control construction-related water quality impacts.

- Operations at the sediment processing/transfer facilities will conform with the substantive requirements of otherwise applicable point-source and stormwater discharge permits.

- Based on an analysis of potential releases of various sediment constituents during dredging operations, it has been determined that either water quality standards will not be contravened or that project releases will represent a minor fraction of applicable NYSDEC regulatory levels. Mercury is the one possible exception, since its release may approach regulatory levels for fish consumption on the basis of the conservative analytical methodologies applied herein.

- Groundwater resources are expected to be unaffected during the implementation of EPA's selected remedy.

- Dredging-induced water quality impact will be insignificant in the Upper Hudson River; therefore, no impact is expected on the water quality of the lower river.

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Coastal Zone Management
253238

WHITE PAPER – COASTAL ZONE MANAGEMENT

(ID 253238)

ABSTRACT

Several comments have asked whether or not the selected remedial activities for the Hudson River PCBs Superfund Site will be consistent with New York State's Coastal Zone Management (CZM) policies. One commenter expressed concern that the selected remedy may have adverse effects on the coastal zone as a result of remedial activities that are performed south of the Federal Dam (such as a potential sediment processing/transfer facility[1] near the Port of Albany), or indirect impacts such as resuspension of PCBs during dredging in the Upper Hudson that may enter the Lower Hudson. The coastal zone includes the Hudson River from the Federal Dam in Troy to New York City. The following is a brief summary of the State's CZM Program and the relationship of EPA's selected remedy to the State's CZM policies.

INTRODUCTION

The federal Coastal Zone Management Act (CZMA) requires federal agencies that conduct or support activities that directly affect a coastal use or resource to undertake those activities in a manner that is consistent, to the maximum extent practicable, with State CZM programs that have been approved by the National Oceanographic and Atmospheric Administration (NOAA).

Pursuant to the CZMA, New York State developed and obtained federal approval for its coastal zone program, which is structured in the form of 44 coastal policies. New York State Department of State (NYSDOS) has developed and manages the New York CZM program. In accordance with the CZMA and CERCLA, EPA will ensure that on-site response activities to implement the

[1] It is important to note that EPA has not yet determined the location(s) of sediment processing/transfer facilities necessary to implement the selected remedy. For purposes of the Feasibility Study, example locations were identified from an initial list of candidate sites based on screening-level field observations which considered potential facility locations from an engineering perspective. In the Feasibility Study, it was necessary to assume the locations of sediment processing/transfer facilities in order to develop conceptual engineering plans, analyze equipment requirements, and develop cost estimates for the remedial alternatives. For this purpose, two example locations were identified: one at the northern end of the project area in the vicinity of the Old Moreau Dredge Spoils Area, and one at the southern end of the project area near the Port of Albany. Each of these example locations fulfills many of the desired engineering characteristics for such a facility to support the remedial work, and is representative of reasonable assumptions with regard to distance from the dredging work and cost. Other locations, both within the Upper Hudson River valley and farther downstream, are possible.

The example facility locations presented in the Feasibility Study have also been used in the Responsiveness Summary in order to clarify material presented in the Feasibility Study and Proposed Plan and in connection with additional noise, odor and other analyses that were performed in order to respond to public comments. EPA will not determine the actual facility location(s) until after EPA performs additional analyses and holds a public comment period on proposed locations and considers public input in the final siting decision. Thus, all information provided in this Responsiveness Summary relative to potential impacts of the sediment processing/transfer facilities on communities, residents, agriculture, the environment and businesses should likewise be considered representative and illustrative. Further specific assessment of and, as necessary, mitigation of, potential impacts will be addressed during remedial design.

ROD, including the dredging and the sediment processing/transfer facilities, will comply, to the maximum extent practicable, with substantive requirements of the State's CZM program, including 19 Local Waterfront Revitalization Programs (LWRPs) that are part of the New York State CZM program.

NYSDOS and New York State Department of Environmental Conservation (NYSDEC) believe it would be premature to perform a CZM consistency determination on this project before the ROD is issued. The State believes that such a determination should be performed after the ROD is signed, but before the remedial design is finalized. Nevertheless, EPA has reviewed the selected remedy against applicable State CZM policies, and has determined that the dredging component of the selected remedy will be consistent with New York State coastal management policies. A consistency analysis for the sediment processing/transfer facilities will be done after the locations of those facilities are determined, but before the remedial design is finalized. EPA's CZM review of the dredging is presented below.

Activities in the Coastal Zone

EPA's selected remedy will primarily involve activity north of Federal Dam, beyond the New York State-delineated coastal zone, with the possible exception of a sediment processing/transfer facility that may be located south of the Federal Dam. However, EPA will not select location(s) for sediment processing/transfer facilities until after a public comment period that will be held on the siting issue after the ROD is signed. EPA will further evaluate the consistency of the sediment processing/transfer facilities with the State's coastal management program during the remedial design period. Consequently, the discussion that follows does not address consistency issues related to construction and operation of sediment processing/transfer facilities, but is strictly concerned with effects to the New York State-designated coastal zone associated with the dredging of PCB-contaminated sediments in the Upper Hudson River north of the Federal Dam.

Benefits of the Selected Remedy on the Lower River

EPA’s modeling projects that the coastal Lower Hudson River will experience significant benefits associated with the selected remedy (Chapter 6 of this RS). Human health cancer risks and non-cancer health hazards, as well as risks to ecological receptors, will be reduced as a result of remedial dredging. Implementation of the selected remedy will achieve a 42 to 56 percent reduction in non-cancer health hazards and a 56 to 65 percent reduction in human health cancer risks compared the No Action Alternative. Similarly, reductions in risks to ecological receptors are expected. For example, implementation of the selected remedy will result in a reduction in the toxicity quotients between 43 and 63 percent for mink in the lower river compared to the No Action Alternative.

In addition, the effects of implementation of the selected remedy on Tri+ PCB concentrations in the water column of the Lower Hudson and the subsequent effect of these concentrations on PCB body burdens in fish tissue have been modeled (Response to Master Comment 313787, Section 2.4.3 of the RS). The modeling results show significant reductions in water column Tri+ PCB concentrations and, subsequently, in fish tissue after completion of the selected remedy. Model forecasts show that implementation of the selected remedy will reduce Tri+ PCB concentrations in the water column of the lower river by 70 percent by 2029, compared to the No Action

Alternative. Since there is a strong correlation between PCB levels in the water column and those observed in fish tissue, a significant reduction in PCB body burdens in fish in the lower river is expected to occur in about the same amount of time.

CONCLUSION

The coastal resources of the Hudson River have been exposed to PCB contamination for many years, resulting in fishing advisories and the closure of commercial fisheries once maintained in the Hudson River south of the Federal Dam. In addition, PCB contamination in the Hudson River has been a major factor in the continuing public perception of the Hudson as a "polluted" river. EPA has determined that there will be no significant short-term effects on water quality in the coastal zone as a result of the dredging, and that in the long term the selected remedy will have a beneficial impact on the coastal zone because the remedy will reduce the water-column PCB load to the coastal zone. Further, EPA believes that this remedial action is consistent with the State's efforts to revitalize the environment of the Hudson River, which is expected to not only safeguard State economic, social, and environmental interests, but advance them.

The following evaluation addresses the consistency of the dredging component of the selected remedy with the 44 NYS coastal management policies. In addition, 19 LWRPs have been reviewed as part of this consistency evaluation. Figure 253238-1 depicts the locations of these cities, towns, and villages. Consistency for each LWRP is presented, as appropriate, below.

Policy 1: Restore, revitalize, and redevelop deteriorated and underutilized waterfront areas for commercial, industrial, cultural, recreational, and other compatible uses.

Not Applicable (N/A) for the dredging component of the selected remedy.

Policy 2: Facilitate the siting of water-dependent uses and facilities on or adjacent to coastal water.

N/A for the dredging component of the selected remedy.

Policy 3: Further develop the State’s major ports of Albany, Buffalo, New York, Ogdensburg, and Oswego as centers of commerce and industry and encourage the siting in these port areas, including those under the jurisdiction of State public authorities, of land use and development which is essential to, or in support of, the waterborne transportation of cargo and people.

N/A for the dredging component of the selected remedy.

Policy 4: Strengthen the economic base of smaller harbor areas by encouraging the development and enhancement of those traditional uses and activities which have provided such area with their unique maritime identity.

N/A for the dredging component of the selected remedy.

Policy 5: Encourage the location of development in areas where public services and facilities essential to such development are adequate.

N/A for the dredging component of the selected remedy.

Policy 6: Expedite permit procedures in order to facilitate the siting of development activities at suitable locations.

N/A for the dredging component of the selected remedy.

Policy 7: (Albany LWRP 7, 7a; Beacon LWRP 7, 7a; Croton-on-Hudson LWRP 7, 7a; Esopus LWRP 7, 7a, 7b, 7c, 7d; Kingston LWRP 7, 7a, 7b; Lloyd LWRP 7, 7a, 7b; New York City LWRP 7; Nyack LWRP 7; Ossining LWRP 7, 7a; Piermont LWRP 7, 7a; Poughkeepsie LWRP 7; Red Hook LWRP 7, 7a; Saugerties LWRP 7, 7a; Schodack/Castleton-on-Hudson LWRP 7, 7a, 7b; Sleepy Hollow LWRP 7, 7d; Stony Point LWRP 7, 7a, 7b, 7c, Tivoli LWRP 7, 7a): Significant coastal fish and wildlife habitats will be protected, preserved, and, where practical, restored so as to maintain their viability as habitats.

Current PCB levels measured in the river water and sediment of the Hudson River exceed criteria and guidelines established for the protection of the environment (USEPA, 2000c). Therefore, fish and wildlife in the Upper and Lower Hudson River are currently being exposed to hazardous levels of PCBs being released from contaminated sediments located primarily north of the Federal Dam. The selected remedial action will remove a significant amount of the most highly contaminated sediments from the Upper Hudson River. This will drastically reduce the amount of PCBs flowing into the New York State-designated coastal zone, where they are potentially adversely impacting wildlife.

An analysis of the potential for adverse water-quality impacts as a result of the implementation of the selected remedy has shown that dredging-related concentrations of key water-quality parameters will not contravene water quality standards (White Paper – Potential Impacts to Water Resources, White Paper – Resuspension of PCBs during Dredging). Dredging-related increases in suspended sediments are also not expected to adversely affect coastal fish and wildlife. Currently, concentrations of total suspended solids (TSS) measured in the Hudson River vary significantly throughout the course of a year (Figure 253238-2). TSS varies spatially, as well (*i.e.*, areas near tributaries have greater TSS concentrations). Peak concentrations occur in the spring along with the highest water flows from the spring runoff. Spring is also the spawning season for many fish species and a time during which many aquatic organisms are at sensitive life stages (*e.g.*, larvae, juveniles). Elevated TSS concentrations in the Hudson River resulting from the spring runoff are typically observed throughout March and April.

Any increases in TSS concentrations resulting from the selected remedy are expected to be a small fraction of the Hudson River’s natural variation, and as such should not adversely affect fish and wildlife (Figure 253238-2, FS Chapter 9, White Paper – Potential Impacts to Water Resources). In addition, settling and dilution will limit effects experienced in the coastal zone, since dredging work will occur largely in River Sections 1 and 2 (RMs 194.6 – 183.4), located approximately 29 miles north of the Federal Dam at Troy, where the New York State-designated coastal zone begins. Consequently, EPA believes that implementation of the selected remedy will have a beneficial effect on resources of the coastal zone of New York State. NYSDEC supports active remediation of contaminated sediments in the Upper Hudson River (NYSDEC, 2000). The federal Trustees for Natural Resources also favor an environmental dredging remedy for the Hudson River PCBs Site (USDOI and NOAA, 2001).

There are 34 State-designated Significant Coastal Fish and Wildlife Habitats in the Hudson River south of Federal Dam. In addition, five sites have been identified as containing important animal and plant communities (NYSDOS and The Nature Conservancy, 1990). A discussion of each significant habitat is provided in Table 253238-1. It should be emphasized that all sediment removal associated with this project would occur well north of these habitats. River Section 1, located approximately 34 river miles north of the New York State-designated coastal zone, would account for approximately 60 percent of the total sediment volume to be removed (USEPA, 2000c). In addition, no significant short-term adverse water quality effects or significant increases in resuspended sediment or PCB concentrations are expected in the coastal zone due to environmental dredging. Furthermore, in the long-term, EPA expects that the selected remedy will have a beneficial impact on Significant Coastal Fish and Wildlife Habitats south of Troy, since the selected remedy will reduce the exposure of these habitats to PCBs. Consequently, EPA does not anticipate any adverse impacts to these Significant Coastal Fish and Wildlife Habitats as a result of the implementation of this remedy.

Compliance with this policy also requires that habitats that support rare and endangered species will be protected, preserved, or, where practical, restored to maintain their viability. To determine whether habitat supporting any federally listed or proposed endangered or threatened species exists in the project area, EPA consulted with the US Fish and Wildlife Service (FWS) and the National Marine Fisheries Service (NMFS). In an August 17, 2001 letter to EPA, the FWS identified one threatened species (bald eagle) that is known to occur in the project area, as well as two endangered species (Karner blue butterfly and Indiana bat) that may be found in the project area. The FWS recommended that EPA's project documents include an evaluation of potential direct, indirect, and cumulative effects of project-related activities on the bald eagle, Karner blue butterfly, and Indiana bat, and their habitats.

In a letter dated May 7, 2001, the NMFS notified the EPA that the federally listed endangered shortnose sturgeon is found in the Hudson River south of the Federal Dam at Troy, and requested that an assessment of the project's impacts to the shortnose sturgeon be prepared.

Consequently, EPA will conduct biological assessments for the bald eagle and shortnose sturgeon, as they have been identified as being in the project area. Moreover, once the locations for the sediment processing/transfer facilities and other necessary land-based infrastructure have been established, EPA will evaluate those locations to determine if they contain habitat suitable to support the Karner blue butterfly or Indiana bat. If suitable habitat is found, additional biological assessment work will be conducted for those species. Any completed biological assessments will be submitted to the FWS or NMFS for review and a final determination of effect. Further, the final remedial design will reflect appropriate measures to protect these species.

Policy 8: (Albany LWRP 8; Beacon LWRP 8; Croton-on-Hudson LWRP 8; Esopus LWRP 8; Kingston LWRP 8; Lloyd LWRP 8; New York City LWRP 8, North Greenbush LWRP 8; Nyack LWRP 8; Ossining LWRP 8; Piermont LWRP 8; Poughkeepsie LWRP 8; Red Hook LWRP 8; Rensselaer LWRP 8; Saugerties LWRP 8; Schodack/Castleton-on-Hudson LWRP 8; Sleepy Hollow LWRP 8; Stony Point LWRP 8; Tivoli LWRP 8): Protect fish and wildlife resources in the coastal area from the introduction of hazardous wastes and other pollutants which bioaccumulate in the food chain or which cause significant sub-lethal or lethal effects on those resources.

Currently, PCB levels measured in the Lower Hudson River exceed standards and guidelines established to be protective of the environment. PCBs are known to bioaccumulate, and their current and future levels are sufficient to pose a risk to piscivorous birds and mammals in the Hudson River (USEPA, 2000a, and USEPA, 2000b). The implementation of the selected remedy will serve to reduce the concentrations of PCBs that fish and wildlife are exposed to in the Hudson River, both within and outside of the coastal zone, and is therefore expected to have a beneficial impact on the entire Hudson River. PCB resuspension associated with the selected remedy is not expected to have long-lasting adverse effects on PCB concentrations in the water column or PCB body burdens in fish in the selected sediment-removal areas, or the portion of the Hudson River south of the Federal Dam.

Policy 9: (Albany LWRP 9; Beacon LWRP 9; Croton-on-Hudson LWRP 9; Esopus LWRP 9; Kingston LWRP 9; Lloyd LWRP 9; New York City LWRP 9; North Greenbush LWRP 9; Nyack LWRP 9; Ossining LWRP 9; Piermont LWRP 9; Poughkeepsie LWRP 9; Red Hook LWRP 9; Rensselaer LWRP 9; Saugerties LWRP 9; Schodack/Castleton-on-Hudson LWRP 9; Sleepy Hollow LWRP 9; Stony Point LWRP 9; Tivoli LWRP 9): Expand recreational use of fish and wildlife resources in coastal areas by increasing access to existing resources, supplementing existing stocks, and developing new resources.

There are no recreational fishing bans for the Hudson River south of Federal Dam. However, the New York State Department of Health (NYSDOH) recommends eating none of the fish in the Upper Hudson River, and that children under the age of 15 and women of childbearing age eat none of the fish in either the Upper or Lower Hudson River, and that the general population eat none of most species of fish caught between the Federal Dam at Troy and Catskill, New York. NYSDOH, 2001. Implementation of the selected remedy will help facilitate the relaxation or termination of these advisories, and as a result could increase recreational utilization of the Hudson River south of Federal Dam. In addition, potential adverse ecological effects to fish and wildlife in the Hudson River due to PCB contamination will be reduced.

Policy 10: (Beacon LWRP 10; Croton-on-Hudson LWRP 10; Esopus LWRP 10; Kingston LWRP 10; Lloyd LWRP 10, 10A; New York City LWRP 10; Nyack LWRP 10; Ossining LWRP 10; Piermont LWRP 10; Red Hook LWRP 10; Sleepy Hollow LWRP 10; Stony Point LWRP 10; Tivoli LWRP 10): Further develop commercial finfish, shellfish, and crustacean resources in the coastal area by encouraging the construction of new or improvement of existing on-shore commercial fishing facilities; increasing marketing of the State's local seafood products; maintaining adequate stocks; and expanding aquaculture facilities.

In 1976, PCB contamination led to the closure of essentially all of the commercial fishery (except for Atlantic sturgeon greater than four feet, American shad, goldfish, and baitfish) once maintained in the coastal Hudson. While the commercial fishery has since been opened for a limited number of species, it remains closed for several species, including striped bass, pumpkinseed, brown bullhead, American eel, white perch, and white catfish (USDOI and NOAA, 2001; NYSDEC, 2000). Implementation of the selected remedy will return the river to a more pristine condition, and may allow the resumption and possible expansion of commercial fishing opportunities in the Hudson River south of the Federal Dam.

Policy 11: Buildings and other structures will be sited in the coastal area so as to minimize damage to property and the endangering of human lives caused by flooding and erosion.

N/A for the dredging component of the selected remedy.

Policy 12: Activities or development in the coastal area will be undertaken so as to minimize damage to natural resources and property from flooding and erosion by protecting natural protective features including beaches, dunes, barrier islands, and bluffs.

N/A for the dredging component of the selected remedy.

Policy 13: The construction or reconstruction of erosion protection structures shall be undertaken only if they have a reasonable probability of controlling erosion for at least 30 years as demonstrated in design and construction standards and/or assured maintenance or replacement programs.

N/A for the dredging component of the selected remedy.

Policy 14: Activities and development, including the construction or reconstruction of erosion protection structures, shall be undertaken so that there will be no measurable increase in erosion or flooding at the site of such activities or development, or at other locations.

N/A for the dredging component of the selected remedy.

Policy 15: (Albany LWRP 15; Beacon LWRP 15; Croton-on-Hudson LWRP 15; Esopus LWRP 15; Kingston LWRP 15; Lloyd LWRP 15; New York City LWRP 15; North Greenbush LWRP 15; Nyack LWRP 15; Ossining LWRP 15; Piermont LWRP 15; Poughkeepsie LWRP 15; Red Hook LWRP 15; Rensselaer LWRP 15; Saugerties LWRP 15; Schodack/Castleton-on-Hudson LWRP 15; Sleepy Hollow LWRP 15; Stony Point LWRP 15; Tivoli LWRP 15): Mining, excavation, or dredging in coastal waters shall not significantly interfere with the natural coastal processes which supply beach materials to land adjacent to such waters and shall be undertaken in a manner which will not cause an increase in erosion of such land.

N/A for the dredging component of the selected remedy.

Policy 16: Public funds shall only be used for erosion-protective structures where necessary to protect human life and new development which requires a location within or adjacent to an erosion hazard area to be able to function or existing development; and only where the public benefits outweigh the long term monetary and other costs including the potential for increasing erosion and adverse effects on natural protective features.

N/A for the dredging component of the selected remedy.

Policy 17: Non-structural measures to minimize damage to natural resources and property from flooding and erosion shall be used whenever possible.

N/A for the dredging component of the selected remedy.

Policy 18: (Albany LWRP 18; Beacon LWRP 18; Croton-on-Hudson LWRP 18; Esopus LWRP 18; Kingston LWRP 18; Lloyd LWRP 18; New York City LWRP 18; North Greenbush LWRP 18; Nyack LWRP 18; Ossining LWRP 18; Piermont LWRP 18; Poughkeepsie LWRP 18; Red Hook LWRP 18; Rensselaer LWRP 18; Saugerties LWRP 18; Schodack/Castleton-on-Hudson LWRP 18; Sleepy Hollow LWRP 18; Stony Point LWRP 18; Tivoli LWRP 18): To safeguard the vital economic, social, and environmental interests of the State and of its citizens, proposed major actions in the coastal area must give full consideration to those interests and to the safeguards which the State has established to protect valuable coastal resource areas.

Since EPA's 1984 interim No Action decision for PCB-contaminated sediments in the Upper Hudson River, PCB concentrations have remained elevated in Hudson River sediments, water, and fish. Some changes have occurred during this period, but in general conditions have not improved substantially for the last five years (USEPA, 2000c). The coastal resources of the Hudson River have been exposed to PCB contamination for many years, resulting in fishing advisories and the closure of the commercial striped bass fishery, among others, in the Lower Hudson River. In addition, PCB contamination has been a major factor in the continuing public perception of the Hudson as a "polluted" river. Implementation of the selected remedy will help to remediate PCB-contaminated sediments, which will ultimately help to reduce exposure of the Hudson's coastal resources to PCB contamination. Further, this remedial action will continue New York State's efforts to revitalize the environment of the Hudson River, which EPA believes will not only safeguard State economic, social, and environmental interests, but advance them.

Policy 19: Protect, maintain, and increase the level and types of access to public water-related recreation resources and facilities.

N/A for the dredging component of the selected remedy.

Policy 20: Access to the publicly owned foreshore and to lands immediately adjacent to the foreshore or the water edge that are publicly owned shall be provided and it shall be provided in a manner compatible with adjoining uses.

N/A for the dredging component of the selected remedy.

Policy 21: (Albany LWRP 21; Beacon LWRP 21; Croton-on-Hudson LWRP 21; Esopus LWRP 21; Kingston LWRP 21; Lloyd LWRP 21; New York City LWRP 21; North Greenbush LWRP 21; Nyack LWRP 21; Ossining LWRP 21; Piermont LWRP 2; Poughkeepsie LWRP 21; Red Hook LWRP 21; Rensselaer LWRP 21; Saugerties LWRP 21; Schodack/Castleton-on-Hudson LWRP 21; Sleepy Hollow LWRP 21; Stony Point LWRP 21; Tivoli LWRP 21): Water-dependent and water-enhanced recreation will be encouraged and facilitated and will be given priority over non-water-related uses along the coast.

Remediation of PCB-contaminated sediments in the Hudson River north of the Federal Dam will do much to foster a public perception that the river is now "safe." In addition, a reduction in PCBs entering the coastal zone may facilitate the relaxation or lifting of current health advisories recommending against eating fish in the Hudson River south of the Federal Dam at Troy. An increase in water-dependent and water-enhanced recreation (*e.g.*, recreational boating and fishing) in the portion of the Hudson River located in the New York State-designated coastal zone may result from the remedial action.

Policy 22: Development, when located adjacent to the shore, will provide for water-related recreation, whenever such use is compatible with reasonable anticipated demand for such activities and is compatible with the primary purpose of the development.

N/A for the dredging component of the selected remedy.

Policy 23: Protect, enhance, and restore structures, districts, areas, or sites that are of significance in the history, architecture, archaeology, or culture of the State, its communities, or the nation.

N/A for the dredging component of the selected remedy.

Policy 24: Prevent impairment of scenic resources of Statewide significance.

N/A for the dredging component of the selected remedy.

Policy 25: Protect, restore or enhance a natural and man-made resources which are not identified as being of Statewide significance but which contribute to the overall scenic quality of the coastal area.

N/A for the dredging component of the selected remedy.

Policy 26: (Esopus LWRP 26; Lloyd LWRP 26; Red Hook LWRP 26; Schodack/Castleton-on-Hudson LWRP 26; Tivoli LWRP 26): Conserve and protect agricultural lands in the State's coastal area.

EPA has determined that dredging-related concentrations of key water-quality parameters such as TSS, nutrients (phosphorus, nitrogen), polycyclic aromatic hydrocarbons (PAHs), dissolved oxygen (DO), metals, and pathogens will represent only a fraction of their ambient levels. Additionally, resuspended sediment concentrations resulting from dredging operations and

subsequent PCB water-column concentrations are within the Hudson River's natural range of variation. As a result, farmers who use Hudson River water for irrigation purposes will not be adversely affected by implementation of the selected remedy.
See the discussion under Policy 30 for further information.

Policy 27: Decisions on the siting and construction of major energy facilities in the coastal area will be based on public energy needs, compatibility of such facilities with the environment, and the facility's need for a shorefront location.

N/A for the dredging component of the selected remedy.

Policy 28: Ice management practices shall not interfere with the production of hydroelectric power, damage significant fish and wildlife and their habitats, or increase shoreline erosion or flooding.

N/A for the dredging component of the selected remedy.

Policy 29: Encourage the development of energy resources on the outer continental shelf, in Lake Erie, and in other water bodies and ensure the environmental safety of such activities.

N/A for the dredging component of the selected remedy.

Policy 30: (Albany LWRP 30; Beacon LWRP 30; Croton-on-Hudson LWRP 30; Esopus LWRP 30; Kingston LWRP 30; Lloyd LWRP 30; New York City LWRP 30; North Greenbush LWRP 30; Nyack LWRP 30; Ossining LWRP 30; Piermont LWRP 30; Poughkeepsie LWRP 30; Red Hook LWRP 30; Rensselaer LWRP 30; Saugerties LWRP 30; Schodack/Castleton-on-Hudson LWRP 30; Sleepy Hollow LWRP 30; Stony Point LWRP 30; Tivoli LWRP 30): Municipal, industrial, and commercial discharges of pollutants including, but not limited to, toxic and hazardous substances into coastal waters, will conform to State and national water quality standards.

EPA has performed detailed analyses on the potential for adverse water quality impacts resulting from sediment removal operations in the Upper Hudson River, including the potential for PCB resuspension and the discharge of other sediment-borne contaminants.

PCB concentrations entering the Hudson River from above Rogers Island currently exceed federal and State criteria. Consequently, the federal Ambient Water Quality Criterion, the New York State standard for protection of wildlife, and the New York State standard for protection of human consumers of fish will continue to be exceeded during implementation of this remedy.

In order to reduce the potential for resuspension of PCBs during the sediment removal process, EPA will utilize environmental dredges and engineering controls. However, it is possible that there will be a temporary increase in PCB concentrations caused by sediment resuspension during dredging. EPA estimates that a maximum 61 kgs per operating season of total PCBs will be released in one operating season as a result of implementation of the selected remedy, which is only a small fraction of the average annual release (calculated for the last 10 years) of 272 kgs that would persist under the No Action Alternative. Further, the coastal zone is located 10 river miles downstream from the southernmost remedial dredge site, well beyond the area predicted to experience impacts from resuspended sediment (Figure 253238-1).

In addition, EPA has evaluated the effect of implementation of the sediment removal on other water quality parameters to determine if the process of removing sediments has the potential to introduce other sediment-borne contaminants into the Hudson River in the vicinity of the work sites. EPA has performed analyses on conventional and trace water quality parameters including suspended solids (TSS/Turbidity), nutrients (*i.e.*, nitrogen and phosphorus), PAHs, DO, selected metals, and pathogens to address potential adverse water-quality impacts that could result from implementation of the selected remedy. Each contaminant listed was evaluated with respect to existing river background levels and published State and federal standards.

Water column TSS levels (mg/L) have been estimated for both the mechanical and hydraulic dredging alternatives. The results indicate that, upon mixing, water column TSS levels are expected to be only a fraction of that experienced during normal spring runoff. In addition, the plumes of TSS that will result from dredging would only extend a short distance from the dredge, and not river-wide. Even for times of the year when river TSS levels are relatively low, overall TSS concentrations resulting from implementation of the selected remedy are expected to be elevated only marginally in comparison to the spring conditions. This, coupled with the small area affected, leads to the conclusion that the river will be minimally impacted by resuspension from dredging.

Project-related concentrations for the remaining conventional and trace parameters were evaluated using the modeled TSS release rate. This analysis determined that:

- Increased nutrient concentrations in water will be negligible compared to the normal concentrations carried by the river. It has been estimated that the increase in nitrogen will be less than 1 percent and phosphorus will see a net increase between 2 and 3 percent above background levels.

- Metal-related water quality impacts of dredging operations will not be significant. The estimated water column increases of cadmium, chromium, copper, manganese, and nickel in the 'near-field' (*i.e.*, within 10 meters of a dredge site) due to dredging are relatively minor and do not represent a significant concern relative to regulatory levels. Lead is conservatively estimated to increase about 5.5 percent of the regulatory value, which should not be a concern. Mercury is estimated to be close to the regulatory criterion (87 percent) for consumption of fish, but since a fishing ban is in place for PCBs and will remain so during the dredging operations, this does not represent a new regulatory issue. Mercury is not expected to exceed the regulatory values for the protection of wildlife and aquatic organisms; thus, no ecological risks are anticipated.

- Estimates of increases in PAH concentrations resulting from dredging are minor. Therefore, it has been concluded that there will be no significant adverse water quality effects during dredging resulting from the introduction of trace organics to the water column.

- DO concentrations will continue to comply with water quality standards.

Accordingly, dredging-related concentrations of various water quality parameters will not contravene NYSDEC standards and applicable or relevant and appropriate State and federal water quality standards. Moreover, there are no known pollutant discharges expected as part of this project that would impact the New York State-designated coastal zone, and there will be no significant short-term impacts on water quality in the coastal zone as a result of the dredging. Further, in the long-term, the selected remedy will have a beneficial impact on the coastal zone, since the water column PCB load to the coastal zone will be reduced, resulting in improved overall water quality in the Hudson River. In the event that project-related discharges become necessary, all of these discharges would be in compliance with all applicable or relevant and appropriate State and federal water quality standards.

Policy 31: State coastal area policies and management objectives of approved local waterfront revitalization programs will be considered when reviewing coastal water classifications and while modifying water quality standards. However, those waters already overburdened with contaminants will be recognized as being a development constraint.

N/A for the dredging component of the selected remedy.

Policy 32: Encourage the use of alternative or innovative sanitary waste systems in small communities where the costs of conventional facilities are unreasonably high, given the size of the existing tax base of these communities.

N/A for the dredging component of the selected remedy.

Policy 33: Best management practices will be used to ensure the control of stormwater runoff and combined sewer overflows draining into coastal waters.

N/A for the dredging component of the selected remedy.

Policy 34: (Albany LWRP 34; Beacon LWRP 34; Croton-on-Hudson LWRP 34; Esopus LWRP 34; Kingston LWRP 34; Lloyd LWRP 34; New York City LWRP 34; North Greenbush LWRP 34; Nyack LWRP 34; Ossining LWRP 34; Piermont LWRP 34; Poughkeepsie LWRP 34; Red Hook LWRP 34; Rensselaer LWRP 34; Schodack/Castleton-on-Hudson LWRP 34; Sleepy Hollow LWRP 34; Stony Point LWRP 34; Tivoli LWRP 34): Discharge of waste materials into coastal waters from vessels subject to State jurisdiction will be limited so as to protect significant fish and wildlife habitats, recreational areas, and water supply areas.

Discharge of waste materials into coastal waters from project-related vessels could be in the form of sanitary and/or thermal wastes. These discharges will be in compliance with applicable or relevant and appropriate State and federal standards, including those of the Coast Guard and the NYSDEC.

Policy 35: Dredging and dredge spoil disposal in coastal waters will be undertaken in a manner that meets existing State dredging permit requirements and protects significant

fish and wildlife habitats, scenic resources, natural protective features, important agricultural lands, and wetlands.

N/A for the dredging component of the selected remedy.
Policy 36: (Albany LWRP 36; Beacon LWRP 36; Croton-on-Hudson LWRP 36; Esopus LWRP 36; Kingston LWRP 36; Lloyd LWRP 36; New York City LWRP 36; North Greenbush LWRP 36; Nyack LWRP 36; Ossining LWRP 36; Piermont LWRP 36; Poughkeepsie LWRP 36; Red Hook LWRP 36; Rensselaer LWRP 36; Schodack/Castleton-on-Hudson LWRP 36; Sleepy Hollow LWRP 36; Stony Point LWRP 36; Tivoli LWRP 36): Activities related to the shipment and storage of petroleum and other hazardous materials will be conducted in a manner that will prevent or at least minimize spills into coastal waters; all practicable efforts will be undertaken to expedite the cleanup of such discharges; and restitution for damages will be required when these spills occur.

It is possible that barge transport of petroleum, specifically diesel fuel, will be necessary to support project operations occurring north of the Federal Dam at Troy. Petroleum products have been transported and stored in the coastal zone as a commodity moving through the Port of Albany (POA, 2001). Best management practices will be established for the safe transport of petroleum products. In order to address the unlikely event of a spill, remediation contractors will be required to have a spill response plan in place. Consequently, EPA activities related to the shipment and storage of petroleum and other hazardous materials associated with this remedial project will be conducted in a manner that will prevent or at least minimize spills into coastal waters

Policy 37: (Albany LWRP 37; Beacon LWRP 37; Croton-on-Hudson LWRP 37; Esopus LWRP 37; Kingston LWRP 37; Lloyd LWRP 37; New York City LWRP 37; North Greenbush LWRP 37; Nyack LWRP 37; Ossining LWRP 37; Piermont LWRP 37; Poughkeepsie LWRP 37; Red Hook LWRP 37; Rensselaer LWRP 37; Saugerties LWRP 37; Schodack/Castleton-on-Hudson LWRP 37; Sleepy Hollow LWRP 37; Stony Point LWRP 37; Tivoli LWRP 37): Best management practices will be utilized to minimize the non-point discharge of excess nutrients, organics, and eroded soils into coastal waters.

The potential for increased nutrient levels to occur in the Upper Hudson as a result of the selected remedy has been analyzed. EPA has determined that elevated nutrient levels are unlikely to result from implementation of the selected remedy. Sediments with high nutrient concentrations generally occur concurrently with the PCB-contaminated sediments targeted for removal. Therefore, sediments with higher nutrient concentrations will be removed and not left exposed to the water column. Increases in nutrient concentrations resulting from resuspension caused by sediment-removal activities are expected to be within State and federal standards. Since adverse effects associated with elevated nutrient concentrations are not anticipated in the Hudson River north of the Federal Dam, EPA does not believe that there will be adverse effects in the coastal section of the river south of the Federal Dam.

Policy 38: (Albany LWRP 38; Beacon LWRP 38; Croton-on-Hudson LWRP 38; Esopus LWRP 38; Kingston LWRP 38; Lloyd LWRP 38; New York City LWRP 38; North Greenbush LWRP 38; Nyack LWRP 38; Ossining LWRP 38; Piermont LWRP 38; Poughkeepsie LWRP 38; Red Hook LWRP 38; Rensselaer LWRP 38; Saugerties LWRP

38; Schodack/Castleton-on-Hudson LWRP 38; Sleepy Hollow LWRP 38; Stony Point LWRP 38; Tivoli LWRP 38): The quality and quantity of surface water and ground water supplies will be conserved and protected, particularly where such waters constitute the primary or sole source of water supply.
Dredging-related concentrations of key water-quality parameters, such as TSS/Turbidity, nutrients (nitrogen and phosphorous), PAH, DO, metals, and pathogens will not contravene water quality standards in the Upper Hudson River. Therefore, it is not expected that contaminant resuspension associated with sediment removal activities will adversely impact water quality in the New York State-designated coastal section of the Hudson River. See the discussion under Policy 30 for further information on the effect of the selected remedy on surface water quality.

In order to protect primary water supplies in the Lower Hudson River, EPA intends to establish a notification system for municipal water suppliers located downstream of the active remedial areas. In the highly unlikely event of an observed release of sediments, municipal water suppliers will be alerted, so they can take action with regard to their river intakes. In addition, when appropriate, ongoing sampling results will be made available to municipalities so that they can assess the need for any actions necessary to protect their water supplies.

Dredging-related increases in water column TSS levels will be marginal, when compared to existing springtime conditions. Thus, water treatment facilities designated for water use during the spring would easily be able to handle the minor TSS increases associated with dredging. It should be noted that in the last 10 years, even during release events that have resulted in PCB concentrations in the water column that are much higher than those expected from dredging, there were no reported exceedances of PCB standards in water supplied by municipalities to residential and commercial users in any of the water supply districts currently obtaining their water from the Hudson River. EPA does not expect there to be any adverse effects on groundwater supplies as a result of the dredging (Chapter 9 of this RS).

Policy 39: The transport, storage, treatment, and disposal of solid wastes, particularly hazardous wastes, within coastal areas will be conducted in such a manner so as to protect groundwater and surface water supplies, significant fish and wildlife habitats, recreation areas, important agricultural land, and scenic resources.

N/A for the dredging component of the selected remedy.

Policy 40: Effluent discharged from major steam electric generating and industrial facilities into coastal waters will not be unduly injurious to fish and wildlife and shall conform to State water quality standards.

N/A for the dredging component of the selected remedy.

Policy 41: Land use or development in the coastal area will not cause national or State air quality standards to be violated.

N/A for the dredging component of the selected remedy.

Policy 42: Coastal management policies will be considered if the State reclassifies land areas pursuant to the prevention of significant deterioration regulations of the federal Clean Air Act.

N/A for the dredging component of the selected remedy.

Policy 43: Land use or development in the coastal area must not cause the generation of significant amounts of acid rain precursors: nitrates and sulfates.

N/A for the dredging component of the selected remedy.

Policy 44: (Beacon LWRP 44, 44a; Croton-on-Hudson LWRP 44, 44a; Esopus LWRP 44; Kingston LWRP 44; Lloyd LWRP 44; New York City LWRP 44; North Greenbush LWRP 44; Nyack LWRP 44; Ossining LWRP 44; Piermont LWRP 44, 44a; Poughkeepsie LWRP 44; Red Hook LWRP 44; Rensselaer LWRP 44; Saugerties LWRP 44, 44a; Schodack/Castleton-on-Hudson LWRP 44; Sleepy Hollow LWRP 44; Stony Point LWRP 44; Tivoli LWRP 44): Preserve and protect tidal and freshwater wetlands and preserve the benefits derived from these areas.

Wetlands found in the New York State-designated coastal zone will not be directly affected by dredging. The majority of the dredging will occur in River Sections 1 and 2, located approximately 29 miles north of the coastal zone (Figure 253238-1). In the event that a sediment processing/transfer facility is located in the coastal zone, potential impacts to wetlands will be evaluated during remedial design.

With respect to water quality-related indirect impacts to coastal wetlands, significant adverse water quality effects in the coastal zone are not expected to result from the selected remedy, as has been stated throughout this document. Analyses of key water quality parameters such as nutrients (nitrogen and phosphorus), PAH, DO, metals, and pathogens indicate that dredging-related concentrations will likely represent only a fraction of their ambient levels. Implementation of the selected remedy is expected to be beneficial to coastal wetlands as it will reduce the current PCB loads being introduced to the coastal section of the Hudson River.

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PCB Releases to Air
253202

WHITE PAPER – PCB RELEASES TO AIR

(ID 253202)

ABSTRACT

Emission of PCBs from dredging and operation of sediment processing/transfer facilities[1] and potential impact on ambient air quality was one of the concerns expressed by the public and various organizations during the public comment period. This white paper briefly describes the methods used for estimating the PCB flux, and provides comparison to relevant air quality standards. In addition, the white paper evaluates potential risks to residents in the vicinity of environmental dredging operations, to residents outside the processing facility boundary, and to processing facility workers.

The modeled PCB concentrations in air within the facility are compared to existing occupational standards developed by the Occupational Safety and Health Administration (OSHA). All predicted PCB concentrations in air within the facility boundary, ranging from 1-hour to annual average periods, are well below these standards. As noted in the FS, off-site modeled concentrations (*i.e.*, exposures to children, adolescents, and adults) are compared to the New York State Annual Guideline Concentrations (AGC). The modeled annual concentrations outside the facility boundary are 10 or more times lower than the AGC value. In addition to these comparisons, a risk analysis was performed to evaluate the human health cancer risks and non-cancer human health hazards to the sediment processing/transfer facility workers and those living outside the facility (*i.e.*, child, adolescent, and adult residents) using reasonable maximum exposure (RME) assumptions. The risk analysis shows that modeled PCB concentrations do not pose an unacceptable risk to human health.

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[1] It is important to note that EPA has not yet determined the location(s) of sediment processing/transfer facilities necessary to implement the selected remedy. For purposes of the Feasibility Study, example locations were identified from an initial list of candidate sites based on screening-level field observations which considered potential facility locations from an engineering perspective. In the Feasibility Study, it was necessary to assume the locations of sediment processing/transfer facilities in order to develop conceptual engineering plans, analyze equipment requirements, and develop cost estimates for the remedial alternatives. For this purpose, two example locations were identified: one at the northern end of the project area in the vicinity of the Old Moreau Dredge Spoils Area, and one at the southern end of the project area near the Port of Albany. Each of these example locations fulfills many of the desired engineering characteristics for such a facility to support the remedial work, and is representative of reasonable assumptions with regard to distance from the dredging work and cost. Other locations, both within the Upper Hudson River valley and farther downstream, are possible.

The example facility locations presented in the Feasibility Study have also been used in the Responsiveness Summary in order to clarify material presented in the Feasibility Study and Proposed Plan and in connection with additional noise, odor and other analyses that were performed in order to respond to public comments. EPA will not determine the actual facility location(s) until after EPA performs additional analyses and holds a public comment period on proposed locations and considers public input in the final siting decision. Thus, all information provided in this Responsiveness Summary relative to potential impacts of the sediment processing/transfer facilities on communities, residents, agriculture, the environment and businesses should likewise be considered representative and illustrative. Further specific assessment of and, as necessary, mitigation of, potential impacts will be addressed during remedial design.

Air monitoring, engineering controls, appropriate worker personal protection equipment, and standard safety procedures will be used as appropriate to protect the sediment processing/transfer facility workers. EPA will implement an air monitoring program to prevent unacceptable exposures to the community associated with dredging and processing of the PCB contaminated sediment. Air sampling results will also be used to confirm the modeled predictions.

INTRODUCTION

Numerous studies have documented the volatilization of PCBs from contaminated water, sediment, and soil, both in the laboratory (Larsson, 1985; Okla and Larsson, 1987; Chiarenzelli *et al.*, 1996; Chiarenzelli *et al.*, 1997; Bushart *et al.*, 1998) and in the environment (Cirpka *et al.*, 1993; Achman *et al.*, 1993; Commoner *et al.*, 1999; Harza, 1990, 1992a, 1992b; WDNR, 2001). Volatilization is an important phenomenon controlling the fate and transport of PCBs in the environment (Andren, 1983).

Review of PCB Concentrations Near Hudson River and Other Dredging Projects

The 1984 Hudson River Feasibility Study (NUS, 1984) reported air sampling in farm fields near the river in 1981. The results showed an average total PCB concentration of about 0.005 $\mu g/m^3$. General Electric sampled air-phase PCBs in the Upper Hudson River during construction of the caps for the remnant deposits between Hudson Falls and Fort Edward (Harza, 1990, 1992a, 1992b). The pre-construction samples contained no detectable levels of PCBs at a detection limit of 0.02 – 0.04 $\mu g/m^3$. During construction, samples were collected within several feet of the heavy machinery moving contaminated dredged sediments; PCB levels as high as 2.77 $\mu g/m^3$ were detected. The post-construction positive results ranged from 0.02 to 0.13 $\mu g/m^3$.

As part of the *hot spot* remedial action at New Bedford Harbor Superfund site (1997), an air monitoring program was implemented. Six monitoring stations were in the dredging area, six in the confined disposal facility (CDF) area, and four at off-site locations around the CDF. Of the 4,041 total samples, 1,063 (26 percent) exceeded the 50 ng/m^3 level, 49 exceeded the 500 ng/m^3 level and 10 exceeded the 1,000 ng/m^3 level. All but one of the 10 exceedances of the 1,000 ng/m^3 occurred at CDF monitoring stations. The one dredge area exceedance of the 1,000 occurred at the closest dredge area station, approximately 30 ft away from the most-contaminated *hot spot* area during initial deployment of various dredging related equipment. As discussed above, the calculated PCB concentrations in Tables 253202-5 and 6 compares well with those measured in the field.

The mechanism of PCB volatilization from environmental dredging and subsequent sediment associated with the selected remedy processing may be described as a two-step process. It involves desorption of PCBs from the solid into the liquid phase, followed by volatilization of the solubilized PCBs from the liquid to the air phase. Desorption of PCBs from the sediment is linear and is related to organic carbon content of the dredged sediment, particle size, and octanol/water partition coefficient. Subsequent PCB volatilization flux from liquid to air may be estimated using mass transfer correlations from the scientific literature and the concentration gradient between the liquid and the air phase.

The dredging and the sediment processing/transfer facilities are identified as the potential PCB release areas for the selected remedy. The quantity of PCBs released to the air is a function of the total surface area available for PCB release and flux of PCBs from the surface. The total PCBs released from the dredging or the sediment processing/transfer facilities may be used as input to air dispersion models for estimating the air-phase PCB concentrations at specific receptor locations. The PCB concentration values may be used to quantify the risk associated with inhaling PCBs released to air from the dredging and the sediment processing/transfer facilities. Therefore, the objective of this white paper is to present the details of a conservative estimate of:

- PCB mass transfer coefficient.

- PCB concentration estimates at nearby receptor location with highest possible impact.

- Risks associated with the sediment processing/transfer facility and the dredging site.

THEORETICAL CONSIDERATIONS

The mass transfer of a constituent across the surface microlayer can be described by the two-phase resistance model (Liss and Slater, 1974; Burkhard, 1995). The equation is given as:

$$F = K_{OL} (C - C_A) \qquad (1)$$

where:

F = constituent flux,

K_{OL} = gas transfer coefficient,

C = concentration of the dissolved constituent in the bulk water phase, and

C_A = concentration of the constituent in air expressed as the water concentration in equilibrium with the air.

The mass transfer coefficient is affected by, among other factors, the thickness of the microlayer, which is in turn affected by the turbulence in the medium. For the purposes of this analysis, the PCB concentration in Hudson River water is assumed to be much larger (orders of magnitude) than the surrounding air-phase concentration and so the flux equation can be written as:

$$F = K_{OL} * C \qquad (2)$$

Equation 2 can be applied to estimate the PCB flux across the air-water interface.

Liquid-Air Interface Mass-Transfer Coefficient, K_{OL}

The magnitude of K_{OL} is dependent on the physical and chemical properties of the compound as well as environmental conditions. The reciprocal of K_{OL} is the total resistance to transfer expressed on a gas RT/Hk_a and liquid $1/k_w$ phase basis as (Achman, 1993):

$$1/K_{OL} = 1/k_w + RT/Hk_a \qquad (3)$$

where:

k_W = the water side gas transfer velocity in m/day,

k_a = the air side gas transfer velocity in m/day,

R = the universal gas constant (8.2057×10^{-5} atm-m^3/mol K),

H = the Henry's Law constant in atm-m^3/mol, and

T = the absolute temperature in ^oK.

In the above equations, k_W and k_a are estimated from published data for PCBs as discussed below.

Gas Film Coefficient

Schwarzenbach *et al.* (1992) proposed the following equation to estimate the k_a value for an organic chemical:

$$k_{a,\, organic} = k_{a,\, H2O} * [D_{organic}/D_{H2O}]^{0.61} \qquad (4)$$

where:

$k_{a,\, organic}$ = the gas film transfer coefficient for the constituent in cm/sec.

$k_{a,\, H2O}$ = the gas film transfer coefficient for water vapor in cm/sec.

$D_{organic}$ = the molecular diffusivity of the constituent in air in cm^2/s.

D_{H2O} = the molecular diffusivity of water in air in cm^2/s.

The mass transfer coefficient for water can be estimated by the following equation (Schwarzenbach *et al.*, 1992):

$$k_{a,\, H2O} = 0.2 * u_{10} + 0.3 \qquad (5)$$

where:

u_{10} = the wind speed at a reference height of 10 m in m/s.

Using the average wind speed of 3.52 m/sec obtained from the National Climatic Data Center (NCDC, 1996) for the month of July (approximately the middle of dredging period) for the Albany Airport, eq (5) can be used to calculate the k_a for water vapor as:

$$k_{a,\,H2O} = 0.2 * u_{10} + 0.3$$

$$= 0.2 * 3.52 + 0.3$$

$$= 1.0 \text{ cm/sec}$$

Molecular diffusivities for various congeners of PCB in air were calculated by Achman *et al.* (1993); an average molecular diffusivity of PCB of 0.0527 cm^2/sec was used here. Using air-water diffusivity of 0.26 cm^2/sec (Cussler, 1984), k_a for PCBs transfer can be calculated as:

$$k_{a,\,organic} = k_{a,\,H2O} * [D_{organic}/D_{H2O}]^{0.61}$$

$$= 1.0 * [0.057/0.26]^{0.61}$$

$$= 0.396 \text{ cm/sec}$$

Liquid Film Coefficient

Liss and Marlivat (1986) proposed the following relationship based on laboratory and field data to estimate the liquid film mass transfer coefficients:

$k_w = 0.17 * u_{10}$ for $u_{10} < 3.6$ m/sec
$k_w = 2.85 * u_{10} – 9.65$ for $3.6 < u_{10} < 13$ m/sec
$k_w = 5.9 * u_{10} – 49.3$ for $u_{10} > 13$ m/sec

In the above equations, the k_w has the units of cm/hr and the relationships have been normalized to CO_2 at 20^oC. Assuming 13 m/sec wind speed to represent the mixing conditions during dredging and handling, the k_w can be calculated as:

$$k_w = 5.9 * u_{10} – 49.3$$
$$= 5.9 * 13 – 49.3$$
$$= 27.4 \text{ cm/hr}$$
$$= 0.0076 \text{ cm/sec}$$

Liss and Marlivat (1986) also suggested the following Schmidt number (Sc) relationship for estimating the k_w for other compounds:

$$k_{w,CO2} / k_{w,organic} = [Sc_{CO2} / Sc_{organic}]^n \qquad (6)$$

where:

n = -2/3 for $u_{10} < 3.6$ m/sec, and
n = -1/2 for $u_{10} > 3.6$ m/sec.

Using Schmidt numbers for CO_2 and PCBs as 600 and 2,400 (Achman *et al.*, 1993), respectively, in eq. (6), the k_w for PCBs can be calculated as:

$$k_{w,CO2} / k_{w,organic} = [Sc_{CO2} / Sc_{organic}]^n$$

or,

$$\begin{aligned} k_{w,\,organic} &= k_{w,\,CO2} * [Sc_{CO2} / Sc_{organic}]^{-0.5} \\ &= 0.0076 * [2400/600]^{-0.5} \\ &= 0.0038 \text{ cm/sec} \end{aligned}$$

Mass Transfer Coefficient

As stated earlier in eq. (3), the mass transfer coefficient is given by:

$$1/K_{OL} = 1/k_w + RT/Hk_a$$

where

$$k_w = 0.0038 \text{ cm/sec} = 3.28 \text{ m/day and}$$

$$k_a = 0.396 \text{ cm/sec} = 342.14 \text{ m/day}$$

Using an average Henry's Law constant of 0.00025 atm-m^3/mol for PCBs at 25^{o}C (Brunner *et al.*, 1990) and substituting, the mass transfer coefficient can be estimated as:

$$\begin{aligned} 1/K_{OL} &= 1/k_w + RT/Hk_a \\ &= 1/3.28 + 8.2057 \text{ X } 10^{-5} * 298 / (0.00025 * 342.14) \\ &= 0.59 \end{aligned}$$

or,

$$\begin{aligned} K_{OL} &= (1/0.59) \text{ m/day} \\ &= 1.695 \text{ m/day} \\ &= 0.0019 \text{ cm/sec} \end{aligned}$$

The estimated value of the PCB volatilization mass transfer coefficient is consistent with the values reported in the scientific literature and EPA reports (Stumm and Morgan, 1981; EPA, 2000b).

Aqueous-Phase PCB Concentration in the Dredging and Sediment Processing/Transfer Facilities

The following two assumptions were made for calculating the PCB flux from the liquid to the atmosphere (USEPA, 2000a):

- Average sediment PCB concentration is about 31.2 mg/kg (31.2 ppm).
- Organic fraction (f_{oc}) of sediment is approximately 4 percent.

Using the organic carbon partition coefficient of PCBs (K_{oc}) as 5.3 X 10^5 mL/g, K_d, the distribution coefficient for PCBs can be calculated as:

$$K_d = K_{oc} * f_{oc} \quad (7)$$

$$= 5.3 \text{ X } 10^5 * 0.04$$

$$= 21{,}200 \text{ mL/g}$$

Assuming a linear isotherm that relates the concentration of PCBs in the aqueous phase (C) and the solid phase (S), the aqueous phase concentration can calculated as (LaGrega, 2001):

$$S = K_d * C \quad (8)$$

or:

$$31.2 = 21200 * C$$

or:

$$C = 1.47 \text{ X } 10^{-3} \text{ mg/L } = 1.47 \text{ µg/L}$$

Equilibrium Air-Phase PCB Concentration

$$\text{Henry's Constant,} \quad 0.00025 = P \text{ (atm)} / C(\text{mol/m}^3) \quad (9)$$

or:

$$P = 0.00025 * [(1.47 \text{ X } 10^{-6}) \text{ g/L}] * [1/(240 \text{ g/mole})] * [1000 \text{ L/m}^3]$$

$$= 1.53 \text{ X } 10^{-9} \text{ atm}$$

Using the Ideal Gas Law:

$$PV = nRT \quad (10)$$

or:

$$6.26 \text{ X } 10^{-8} *1 = n * 8.2057 \text{ X } 10^{-5} * 298$$

or:

$$n = 1.00 \text{ X } 10^{-7} \text{ mole/m}^3$$

Therefore,

$$C_{air} = (6.26 \text{ X } 10^{-8} \text{ mole/m}^3) * (240 \text{ X } 10^{-6} \text{ µg/mole})$$

$$= 15.02 \text{ µg/ m}^3$$

PCB Flux from the Air-Water Interface

Substituting K_{OL} and C in eq. (2), the PCB flux from water at 25°C can be estimated as:

$$
\begin{aligned}
F &= K_{OL} * C \\
&= (1.695 \text{ m/day})*(1.47\ \mu\text{g/L})*(1000 \text{ L/m}^3) \\
&= 2491.65\ \mu\text{g/m}^2\text{-day} \\
&= 2.49 \text{ mg/m}^2\text{-day}
\end{aligned}
$$

ESTIMATION OF POTENTIAL PCB EXPOSURE LEVELS

The dredging sites and the sediment processing/transfer facilities for both the mechanical and hydraulic dredging operations are areas that may release PCBs into the atmosphere.

Mechanical Dredging

As stated in the FS, sediment removed by the mechanical environmental dredges would be placed into barges and towed to the processing/transfer facility. For the purpose of this analysis, it was assumed that the example northern processing/transfer facility (NTF) would be utilized to its maximum processing and transfer capacity of 1,460 tons/day (one-half of the daily amount, which equals 3,000 cy/day); the remaining dredged sediment would be processed at the southern processing/transfer facility (STF).

The preliminary design of the processing/transfer facility is presented in White Paper – Example Sediment Processing/Transfer Facilities. The system consists of a pug mill for stabilizer addition, temporary staging area, railcar loading area and a water treatment plant. The estimated surface area of each component at the processing facility for mechanical dredging is given in Table 253202-1.

It should be noted that due to the heat generated during the cement hydration reaction, the processed sediment from the pugmill would be at a higher temperature. An estimate (Kosmatka and Panarese, 1988) of the adiabatic temperature rise can be made using the following equation:

$$T = CH/S \qquad (11)$$

where:

T = temperature rise in °F of the concrete due to heat generation of cement under adiabatic conditions

C = proportion of cement in the concrete, by weight

H = heat generation due to hydration of cement, Btu/lb/°F

S = specific heat of concrete, Btu/lb/°F

Assuming the specific heat of sediment and the heat of hydration of cement are 0.2 Btu/lb/^oF and 100, respectively in eq (11), the change in temperature can be calculated as:

$$T = (0.08*100)/0.25$$

$$= 32^oF$$

$$= 17.8\ ^oC$$

Considering the heat dissipation associated with pugmill operation and subsequent handling of the sediment, the final processed sediment temperature may be estimated to be 30^oC (assuming average sediment temperature = 18^oC).

Hydraulic Dredging

The hydraulic dredging alternative proposes the use of a 12-inch hydraulic cutterhead dredge for remediation of all targeted areas in River Sections 1 and 2 and environmental mechanical dredging equipment in River Section 3. The mechanically dredged sediments will be barged to the STF and all hydraulically dredged contaminated sediments will be pumped to the NTF for processing (approximately two-thirds of the volume). Due to the estimated 1,600 ton per day shipping limit at the NTF, half of the processed river sediments will be transported to the STF for rail car loading and final disposal. Therefore, the PCB volatilization potential from STF, under the hydraulic dredging option, will essentially be the same as the mechanical dredging option.

The preliminary conceptual design of the processing/transfer facility is presented in White Paper – Example Sediment Processing/Transfer Facilities. The main components of the system include coagulation/flocculation and belt filter press for solids processing and filtration and granular activated carbon columns for water treatment. The estimated surface area of each component at the processing/transfer facility for hydraulic dredging is given in Table 253202-2.

Dispersion Model and Meteorological Data Used

Dispersion Model Used

The Industrial Source Complex Short Term (ISCST3) model, the USEPA-approved refined air quality dispersion model for simple terrain, was used in PCB concentration modeling analysis. ISCST3 is a steady-state Gaussian plume dispersion model and is used to assess pollutant concentrations for industrial facilities and to calculate concentrations for several different averaging periods such as 1-hour, 8-hour, and annual. The main Gaussian dispersion equation for calculating pollutant concentration at a given distance and height is expressed as:

$$\chi = QKVD * \exp\,[0.5\,(y/\sigma_y)^2]\,/\,(2\pi\, u_s\, \sigma_y\, \sigma_z) \qquad (12)$$

Where:

χ = pollutant concentration,

Q = pollutant emission rate,

K	=	a scaling coefficient to convert calculated concentrations to desired units,
V	=	vertical dispersion term,
D	=	decay term,
y	=	distance between the source and the receptor,
σ_y, σ_z	=	standard deviation of lateral and vertical concentration distribution, and,
u_s	=	mean wind speed at release height.

The parameters in the above equation are estimated using site specific conditions and guidelines provided in the *User's Guide for the Industrial Source Complex (ISC3) Dispersion Model - Volume II* (USEPA, 1995). A summary of the ISCST3 modeling inputs used in the PCB concentration calculations is presented in Table 253202-3.

Dredging Period

The dredging period is assumed to be from May to November each year for six years.

PCB Emission Rate

Liquid-Air Interface PCB Flux

The scientific literature indicates that the rate of PCB volatilization is highly dependent on temperature (Okla and Larson, 1987; Chiarenzelli *et al.*, 1996, 1997; Bushart, 1998), however, the availability of data required to quantify the effect of temperature on the rate of volatilization is limited. The results reported by Okla and Larson (1987) and others may be used to predict the effect of temperature on the rate of PCB volatilization; a 26.6 percent increase in volatilization flux may be assumed for each 10^oC rise in temperature. Therefore, the following PCB emission rates were used for model input based on the expected pre- and post-cement addition temperature of the dredged sediment.

Temperature (^oC)	PCB Flux (mg/m^2-day)
18	2.02
30	2.82

An unit area emission rate of 2.02 mg/m^2-day was applied to PCB transfer areas under the normal temperature condition including:

- Two barge areas along each side of the river under mechanical dredging conditions.

- Areas in each sediment processing/transfer facility under the hydraulic dredging scenario. These areas were grouped into three areas as unloading, processing, and loading areas in the model.

- Unloading barges and water treatment areas in each sediment processing/transfer facility under the mechanical dredging scenario.

An unit area emission rate of 2.82 mg/m^2-day was applied to post cement mixing areas in each sediment processing/transfer facility under a mechanical dredging scenario.

PCBs Associated with Suspended Particles

As indicated in White Paper – Air Quality Evaluation, the total suspended particles (TSP) emission rate from the NTF during sediment handling process was estimated to be 8 g/hour. Using an average overall sediment PCB concentration of 31.2 mg/kg, the PCB flux associated with TSP in sediment handling would be about 0.0016 mg/m^2-day. This value is about three orders of magnitude lower than the volatilization flux, and therefore, was not considered in the PCB dispersion modeling or risk calculation since these risks are below levels of concern.

LOSS OF PCBS BY VOLATILIZATION FROM SEDIMENT PROCESSING/ TRANSFER FACILITY

The loss of PCBs by volatilization from the barge (during loading/unloading) and various treatment processes can be estimated using PCB flux and the total transfer area. The PCB loss from the northern and the southern transfer facility for both mechanical and hydraulic dredging options are presented in Table 253202-4. The volatilization loss represented as a fraction of total PCB removed by the selected remedy is also presented.

Sources Modeled

Processing/Transfer Facility

The sources at each facility were conservatively modeled as ground level-release area sources with zero exit velocity. Given the limited design specifics, the modeled area sources are assumed to be located at three areas (unloading, processing and loading areas) with each area covering the whole potential exposure area. For example, at the unloading area, the size of source is equal to the entire barge area assuming each unit area on the barge would emit the same amount of PCB. This approach would result in conservative estimates of PCB levels.

Dredging Process

The same area source modeling approach described above was employed to two dredging barges (one along the river west bank and one along the river east bank) simulated under the worst-case condition in a mechanical dredging scenario. Each barge has approximately 6,000-ft^2 of exposure area and was placed 100 ft from each side of the shoreline.

Receptors

Processing/Transfer Facility

- Discrete receptors were assumed at typical residential locations adjacent to each facility. The distance between the center of facility to the typical receptor location was assumed to be approximately 300 meters.

- Discrete receptors were assumed within the processing facility that were 1 and 10 meters from each side of each modeled area source.

Dredging Process

A series of hypothetical residences along both sides of the river were modeled. Two Cartesian receptor grid systems were placed along the river. Each grid system started 50 ft from each shoreline and covers an area 150 ft wide and 6,000 ft long that is close to a typical dredging site. This 6,000-ft distance is equivalent to the length of river that would be dredged by a single dredge within a year, under a mechanical dredging scenario.

Meteorology

The most recent and complete five consecutive years of meteorological air data were used in the ISCST3 model. The data included:

- 1993-1997: Five consecutive years of Glens Falls surface air data were used for NTF and in-river dredging.
- 1995-1999: Five consecutive years of Albany surface data were used for STF and in-river dredging.

Averaging Period Modeled

For the evaluation of human health cancer risk and non-cancer health hazards, the appropriate time-frame for averaging chronic impacts during the remediation is an annual period. In order to examine possible short-term PCB concentration in air, shorter averaging periods (*e.g.*, 1-, 8-, 24-hour and monthly) were considered in the PCB modeling. Within each modeled period, both PCB sources and receptor locations were assumed to be stationary. For example, in in-river dredging impact modeling, the predicted 8-hour level was calculated assuming the dredging equipment would be operated at the same spot for 8 hours. In the same way, it was assumed that processing facility workers would not move during the 8-hour modeled period.

Methodology for Predicting Annual Average Level from Dredging Activities

Modeling potential annual average impact from moving dredging activities along the river is extremely difficult, especially using actual meteorological data. Therefore, an equivalent stationary source approach was developed and a moving source PCB release impact at a given receptor location on an annual basis was simulated as an average of a stationary source release impacts on a series of receptors along the river. These receptors cover a distance that takes a moving barge one year to cross.

Since dredging activities would move down the river from time to time over the entire six-year dredging period, the potential PCB emission impact on a given receptor location from dredging barges due to volatilization along the river would vary from time to time, unlike a stationary source. At a given receptor location along the river, PCB concentration levels would be expected to be higher when a barge is close and lower when it moves away. Therefore, even though the actual PCB release from a barge is constant, PCB concentration levels would vary with time at a given receptor. This moving effect can be approximated as the effect on a moving receptor from a stationary barge. Based on this analogy, the dispersion model was used to predict concentrations from two stationary barge area sources (each along one side of the river) at a receptor grid that consists of 61 receptor locations placed between 3,000 ft downstream and 3,000 ft upstream with 100-ft spacing. These receptors cover a distance (6,000 ft range along the river) that is equal to the distance that a dredge would progress in a year.

An annual average PCB level from two stationary barges was predicted at each of 61 stationary grid receptors. The average of the annual levels at these 61 receptor locations was then calculated and this level is considered equivalent to the annual concentration at a given receptor location from moving barge operations. Moreover, since the number of mechanical dredging hours on an annual basis would be approximately 47 percent of the total hours within a 30-week working period, a factor of 0.47 was used to calculate the annual average level near a typical dredging site in either the Albany or the Glens Falls area.

Modeled PCB Concentration Levels

The modeled PCB concentrations are summarized in Tables 253202-5 and 6 for processing/transfer facility operations and dredging activities in the river, respectively. Based on the assumptions described above, the predicted PCB levels may be considered to be conservative.

RISK ANALYSIS

Cancer risks and non-cancer health hazards due to possible inhalation of PCBs in air as a result of the selected remedy were evaluated for adult workers and residents (adult, adolescent, and child). Note that because a non-cancer inhalation toxicity factor does not exist for PCBs, the oral RfD was applied to the inhaled dose of PCBs. While there is uncertainty with this approach, it is recognized in EPA guidance (USEPA, 1993), and it was done here in order to avoid neglecting the consideration of potential adverse non-cancer health effects. In general, EPA's position is that, "the potential for toxicity manifested via one route of exposure is relevant to considerations of any other route of exposure, unless convincing evidence exists to the contrary." (USEPA, 1993). PCBs are absorbed through ingestion, inhalation, and dermal exposure, after which they are transported similarly through the circulation. This provides a reasonable basis for expecting similar internal effects from different routes of environmental exposure. Information on relative absorption rates suggests that differences in toxicity across exposure routes may be small (USEPA, 2000a).

With this one additional consideration of applying the oral RfD to inhaled doses of PCBs, this risk analysis is consistent with the revised baseline Human Health Risk Assessment (HHRA) using the same toxicity factors, as well as reasonable maximum exposure (RME) assumptions.

For the inhalation pathway, intake is calculated as:

$$Intake_{inhalation}\ (mg/kg - d) = C_{air} * IR * EF * ED / (BW * AT) \qquad (13)$$

where:

C_{air} = Concentration of the chemical in air (mg/m^3),

IR = Inhalation rate (m^3/day)

EF = Exposure frequency (days/yr)

ED = Exposure duration (yrs)

BW = Body weight (kg)

AT = Averaging time (days)

- *PCB Concentration in Air (C_{air}).* Predicted PCB concentrations are summarized in Tables 5 and 6. The annual predicted PCB concentrations were used for the adult worker and the adult, adolescent, and child residents. As stated earlier, for the evaluation of possible human health cancer risks and non-cancer health hazards, the appropriate time-frame for averaging chronic impacts during the remediation is an annual period.

- *Inhalation Rate (IR).* For adult workers and residents, an inhalation rate of 20 m^3/day was used, which is the recommended value for long term exposure assessments for Superfund risk assessments (USEPA, 1991). The inhalation rate for young children (8.3 m^3/day) and adolescents (13.5 m^3/day) are current recommendations in the 1997 Exposure Factors Handbook for long term exposures (USEPA, 1997). These values are consistent with those used in the revised baseline HHRA.

- *Exposure Frequency (EF).* A full work year (250 days/year) was assumed for the adult worker. Residents may be exposed to PCBs in air when performing activities outside their homes as well as when they are inside (through outside air exchange); therefore, a RME scenario assuming exposure for 350 days a year was used (which assumes two weeks away from the residence).

- *Exposure Duration (ED).* The entire duration of the selected remedy (six years) was evaluated in this analysis.

- *Body Weight (BW).* Age-specific body weights were used. The mean BW for children aged 1 to 6 is 15 kg, for adolescents aged 7 to 18 is 43 kg, and for adults (over 18 years old) is 70 kg (USEPA, 1991).

- *Averaging Time (AT).* A 70-year averaging time of 25,550 days is used for cancer evaluations (365 days/year × 70 years), while a six-year averaging time (*i.e.*, the entire duration of the selected remedy) of 2,190 days is used for non-cancer evaluations (365 days/year × 6 years).

The evaluation of non-cancer health effects involves a comparison of average daily intake levels with RfDs. The hazard quotient is calculated by dividing the estimated average daily intake estimate by the RfD as follows (USEPA, 1989):

$$\textit{Hazard Quotient (HQ)} = \textit{Intake(mg/kg-day)/RfD(mg/kg-day)} \quad (14)$$

The evaluation of carcinogenic risks involves the evaluation of lifetime average daily intake levels with cancer slope factors (CSFs) as follows (USEPA, 1989):

$$\textit{Cancer Risk} = \textit{Intake (mg/kg-day)} * \textit{CSF (mg/kg-day)} \quad (15)$$

The annual predicted PCB concentrations in air were used and RME exposures were assumed for screening purposes. The calculated outside facility boundary cancer risk and non-cancer health hazard for both mechanical and hydraulic dredging options are presented in Tables 253202-7 and 8, respectively. The calculated cancer risk and non-cancer health hazard for inside facility boundary workers are presented in Tables 253202-9 and 10, respectively. Using assumptions for an RME, the cancer risks and non-cancer health hazards to processing/transfer facility workers and outside boundary adult, adolescent, and child residents are *de minimis* (*i.e.,* several orders of magnitude below a cancer risk of 1 in 1,000,000 and a hazard index of 1.0).

Comparison of Modeled PCB Concentrations with Other Guidelines

The modeled PCB concentrations in air within the facility are compared to existing occupational standards developed by the Occupational Safety and Health Administration (OSHA). For PCB workplace exposures, based on an eight-hour time-weighted average (TWA), the standards are 1,000 $\mu g/m^3$ for Aroclor 1242 and 500 $\mu g/m^3$ for Aroclor 1254 (29 CFR 1910.1000). The National Institute for Occupational Safety and Health (NIOSH) guideline concentration, based on a 10-hour work shift, is also presented in Table 253202-5. All predicted PCB concentrations in air within the facility boundary, ranging from 1-hour to annual average periods, are well below these standards (Table 253202-5).

The PCB concentrations outside the facility boundary (*i.e.*, exposures to children, adolescents, and adults) near the processing/transfer facility and the dredging site were also calculated and presented in Table 253202-5 and Table 253202-6, respectively. The concentration values are compared to the New York State Annual Guideline concentrations (AGCs) in NYSDEC's Air Guide 1 – Guidelines for the Control of Toxic Ambient Air Contaminants. The AGC value for PCBs (Aroclors 1248 and above) is 0.002 $\mu g/m^3$. The modeled annual concentrations outside the facility boundary were found to be a factor of two to several orders of magnitude lower than the AGC values. It should be noted that the AGC values are derived by New York State on the basis of continuous (daily) inhalation of PCBs over a lifetime (70 years), while remediation is expected to take six years.

In addition to the comparisons described above, a risk analysis was performed to evaluate the human health cancer risks and non-cancer human health hazards to the sediment processing/transfer facility workers and those living outside the facility (*i.e.*, child, adolescent, and adult residents) using reasonable maximum exposure (RME) assumptions. The risk analysis shows that modeled PCB concentrations do not pose an unacceptable risk to human health (*i.e.*, human health cancer risks are below 1 in 1,000,000 and the non-cancer human health hazards are significantly less than 1). It should be noted that the calculated PCB flux associated with the total suspended particles (TSPs) is about three orders of magnitude lower than those due to volatilization. Therefore, the PCB exposures due to TSP are below a cancer risk of 1 in 1,000,000 and the non-cancer health hazards are significantly less than 1 and are, therefore, not quantitatively evaluated in the risk assessment.

CONCLUSIONS

This white paper describes the methods used for estimating the PCB flux from the dredging and the sediment processing/transfer sites, and the potential risks to processing/transfer facility workers and nearby residents. The cancer risks and non-cancer health hazards due to inhalation of volatilized PCBs in air by processing/transfer facility workers and residents living near the river or the sediment processing/transfer facility were found to be below *de minimis* levels of regulatory concern (*e.g.* below a cancer risk of 1 in a million, and below a non-cancer hazard index of 1.0). PCB exposure associated with the suspended particles were found to be about three orders of magnitude lower than the volatilized PCB and, therefore, do not pose an unacceptable cancer risk or non-cancer health hazard.

Air monitoring, engineering controls, appropriate worker personal protection equipment, and standard safety procedures will be implemented to protect the processing/transfer facility workers. EPA will conduct air monitoring and establish engineering controls to prevent unacceptable exposures to nearby residents associated with implementation of the remediation. EPA will also conduct a detailed analysis to quantify the exposure potential of PCBs from the dredging and the sediment processing/transfer facilities and implement a comprehensive air monitoring and health and safety program to address community concerns as appropriate. Air monitoring will be used to confirm the model predictions.

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Air Quality Evaluation
313846

WHITE PAPER – AIR QUALITY EVALUATION

(313846)

ABSTRACT

Commenters raised concerns regarding potential air quality impacts from the proposed dredging and/or operation of the sediment processing/transfer facilities.[1] This white paper discusses the potential pollutant-emission sources associated with various activities, provides an impact analysis methodology, estimates emission rates for each National Ambient Air Quality Standards (NAAQS) criteria pollutant, and predicts concentration levels at the potential worst-case residential locations. The results of these analyses show that predicted criteria pollutant concentration levels, including background levels, do not exceed the NAAQS. Therefore, the project would not significantly impact air quality and appropriate monitoring will be implemented to confirm these results.

INTRODUCTION

The potential air quality impacts from the selected remedy may be divided into two categories:

- Long-term impacts, such as:
  - In the neighborhood of the sediment processing/transfer facilities, due to operation of those facilities.
  - Along transfer routes, due to the transfer and disposal of processed sediment via barge or railroads.
  - Along the river, due to booster-pump operation during hydraulic dredging.

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[1] It is important to note that EPA has not yet determined the location(s) of sediment processing/transfer facilities necessary to implement the selected remedy. For purposes of the Feasibility Study, example locations were identified from an initial list of candidate sites based on screening-level field observations which considered potential facility locations from an engineering perspective. In the Feasibility Study, it was necessary to assume the locations of sediment processing/transfer facilities in order to develop conceptual engineering plans, analyze equipment requirements, and develop cost estimates for the remedial alternatives. For this purpose, two example locations were identified: one at the northern end of the project area in the vicinity of the Old Moreau Dredge Spoils Area, and one at the southern end of the project area near the Port of Albany. Each of these example locations fulfills many of the desired engineering characteristics for such a facility to support the remedial work, and is representative of reasonable assumptions with regard to distance from the dredging work and cost. Other locations, both within the upper Hudson River valley and farther downstream, are possible.

The example facility locations presented in the Feasibility Study have also been used in the Responsiveness Summary in order to clarify material presented in the Feasibility Study and Proposed Plan and in connection with additional noise, odor and other analyses that were performed in order to respond to public comments. EPA will not determine the actual facility location(s) until after EPA performs additional analyses and holds a public comment period on proposed locations and considers public input in the final siting decision. Thus, all information provided in this Responsiveness Summary relative to potential impacts of the sediment processing/transfer facilities on communities, residents, agriculture, the environment and businesses should likewise be considered representative and illustrative. Further specific assessment of and, as necessary, mitigation of, potential impacts will be addressed during remedial design.

- Short-term impacts, such as:

  - Construction of the sediment processing/transfer facility and associated buildings, roads, parking lots, etc.
  - Dredging and backfilling activities, which would be moving downstream along the river.

Potential air pollutant emissions will result from:

- Fuel-burning diesel engines, such as used by non-road equipment, trucks, locomotives, tug boats, etc.

- The sediment-handling process.

Detailed design plans and locations for the sediment processing/transfer facilities and disposal/transfer routes have not been finalized. Therefore, the assessment of air quality effects from on-road/in-river mobile-source operations cannot be quantitatively evaluated for truck, boat, and train operations at this time. However, these mobile-source operational impacts will not be significant, given the small number of daily trips (13 truck trips when applicable, two train trips, and a few tug boat trips) generated from operations.

Based on the conceptual design plan at each potential processing facility site, a quantitative air quality impact analysis was performed at the likely "worst-case" site, the Northern Sediment Processing/Transfer Facility (NTF) (White Paper – Example Sediment Processing/Transfer Facilities), for the following reasons:

- The facility would use the greatest amount of equipment.

- An on-site unpaved roadway associated with truck transfer operations would contribute dust emissions under the mechanical dredging scenario.

- Sensitive receptors are adjacent to the facility.

The long-term impact analysis was also performed for a stationary booster-pump operation near the river.

Given the differences in impact duration between long- and short-term activities, the short-term impacts from the selected remedy would be similar to impacts from a typical construction project; *i.e.*, impacts would occur only for a short period of time at any given receptor location. The air quality effects of short-term activities are generally of less concern than those from continuing operations (long-term effects) and are generally evaluated qualitatively in most environmental analyses (*e.g.*, environmental assessments and environmental impact statements). However, for this analysis, potential impacts on any given receptor location near the river from short-term dredging activities, is compared with concentration levels predicted from long-term stationary booster-pump operation. In general, the following sections mainly discuss long-term air quality effects due to the selected remedy.

NATIONAL AMBIENT AIR QUALITY STANDARDS (NAAQS)

Under the requirements of the 1970 Clean Air Act (CAA) as amended in 1977 and 1990, EPA establishes NAAQS for six contaminants, referred to as criteria pollutants (40 CFR 50). The criteria pollutants are:

- Ozone (O_3).
- Carbon monoxide (CO).
- Sulfur dioxide (SO_2).
- Nitrogen dioxide (NO_2).
- Particulate matter (PM10).
- Lead (Pb).

The NAAQS include primary and secondary standards. The primary standards (Table 313846-1) were established at levels sufficient to protect public health with an adequate margin of safety. The secondary standards were established to protect public welfare from the adverse effects associated with pollutants in the ambient air. The NYS Department of Environmental Conservation (NYSDEC) adopts the EPA's NAAQS as the statewide ambient air quality standards.

O_3 is a regional concern and is usually not addressed on a project-by-project basis. Lead emissions from mobile sources are not significant and have declined in recent years through the phased out use of lead in gasoline. Therefore, the localized air-quality concentrations from the sediment processing/transfer facilities, and stationary booster-pump operations would result from emissions from CO, SO_2, PM10, and NO_2.

The analysis presented below compares the modeled concentrations for each of the relevant pollutants with the corresponding NAAQS averaging periods to evaluate the potential air quality impacts.

It should be noted that for potential non-criteria pollutants, such as metals, a limited discussion is included in this paper based on the lack of detailed emission factors, or emission rates for each potential source at this time. Emissions estimates will be conducted during the design phase. In addition, when the project is implemented, EPA will conduct ambient air monitoring for a series of sediment-related air pollutants, as necessary, to address community concerns and confirm modeled results.

ANALYSIS SCENARIOS AND SOURCES ANALYZED

The project would involve: 1) continuous sediment processing operations at fixed locations and stationary booster operations under a hydraulic dredging process at a given location; and 2) dredging activities that would move along the Hudson River over a six-year period (It is important to note that EPA has not yet determined the locations of sediment processing/transfer facilities).

Unlike the continuous stationary operations that will occur at the sediment processing/transfer facilities, dredging activities will result in only temporary air quality effects at any given receptor location, as the dredging will occur in individual areas for brief amounts of time. Dredging activities are similar to construction activities, and were not considered quantitatively in this evaluation.

Over a six-year period, operations at the sediment processing/transfer facilities will consist of continuous loading, processing, and unloading for 20 hours per day, six days per week, from May 1 to November 30 of each year. Each facility would process 80 tons per hour of sediment, plus eight tons per hour of cement materials to be mixed with the dredged materials. The sediment moisture content is expected to be about 96 percent (assuming 20 percent excess water) in the barge before cement addition, and about 72 percent after decanting excess water. All emission sources can be mainly characterized into two types: Equipment engine exhausts and dust emissions from the sediment handling process.

Each sediment processing/transfer facility can be constructed for either mechanical or hydraulic dredging processes. For purposes of this RS, the two example facilities are being used: the Northern Sediment Processing/transfer Facility (NTF) and the Southern Sediment Processing/transfer Facility (STF)

Long-Term Operational Activity at the Sediment Processing/transfer Facilities

NTF

A mechanical dredging facility will have the following potential emission sources:

- Two front-end loaders in the enclosed, temporary staging building to load sediment onto dump trucks.
- Three dump trucks near the unloading dock.
- Two front-end loaders in the semi-open loading area, loading sediment onto rail cars.
- Two dump trucks near the loading area.
- One locomotive running for 30 minutes during the daytime period (between 7 am and 10 pm), twice a day.
- A 350-hp tug boat running continuously for 30 minutes each trip. For each 24-hour period, a total of six trips will occur during the daytime hours (between 7 am and 10 pm).
- A 350-hp tug boat running continuously for 30 minutes each trip. For each 24-hour period, a total of three trips will occur during the nighttime hours (between 10 pm and 7 am).
- Five round trips each hour along the on-site roadways.
- One electrical-powered materials handler (CAT 375 MH). Note: The emissions from such a machine are minimal compared to a diesel excavator of the same capacity (Stonehocker, June 15, 2001); the materials handler is, therefore, omitted from the air quality analysis.
- Pug mill, conveyor, and other necessary small generators and compressors will be electrical-powered to the extent possible, resulting in negligible emissions. Therefore, they are not considered in this analysis.

A hydraulic dredging facility will have the following potential emission sources:

- One locomotive running continuously for 30 minutes, twice a day, during the daytime periods (between 7 am and 10 pm).
- One 1,000-hp tug boat running for 60 minutes, three times a day, during daytime periods (between 7 am and 10 pm)
- Other necessary small horsepower generators, auxiliaries, and boosters potentially used are minor sources with negligible emissions; therefore, they are not considered in this analysis.

STF

The STF would include one more excavator than the NTF and use a 1,000-hp tug boat for two trips per day, compared to 350-hp tug boat for nine trips per day for the NTF. However, the STF would eliminate loader operations at the loading area and all truck operations. Overall, this facility is expected to emit fewer pollutants than the NTF. Unlike the NTF, no sensitive receptor locations are adjacent to the facility site. Therefore, the STF is not considered the worst-case scenario and an air quality impact analysis was not conducted, since the ambient air concentrations are less than those found at the NTF, which meets all standards.

Long-Term Stationary Booster Pump Operations along the River

Under a hydraulic dredging scenario, one 1,000-hp booster pump would be installed and operated 17 hours per day per each 10,000-ft dredging-distance increment.

Emission Rate Estimates

Emission Factor References

The following EPA-published emission factor references and models were used in determining emission rates at the NTF for both mechanical and hydraulic dredging scenarios and for a stationary booster near the Hudson River:

- AP-42 (January 1995).
- Non-road Engine and Vehicle Emission Study Report (November 1991).
- Exhaust Emission Factors for Non-road Engine Modeling – Compression-Ignition (April 2000).
- Median Life, Annual Activity, and Load Factor Values for Non-road Engine Emissions Modeling (June 15, 1998).
- Procedures for Emission Inventory Preparation – Volume IV – Mobile Sources (July 1992).
- Commercial Marine Emissions Inventory for EPA Category 2 and 3 Compression Ignition Marine Engines (August 1998).
- Final Emissions Standards for Locomotives (December 1997).
- Guideline on Air Quality Models (July 1, 1999).
- Mobile 5b Model (September 1996).

- Part5 Model (February 1995).

The peak hourly and daily average hourly emission rates were estimated based on the operational conditions for each identified emission source and the emission factors (provided in the above references) for the criteria pollutants for the two main source categories – equipment engine exhaust and the sediment handling process.

Engine Exhaust Emission Rates

Non-Road Diesel Equipment

The selected remedy will use a series of non-road diesel equipment and trucks to process and transfer dredged sediment. The EPA has developed a database for non-road engine and vehicles emission factors as a function of the type and size of the equipment, and has provided guidance for developing emission inventories for these engines. Since the selected remedy is not expected to occur before year 2005, emission factors for the Tier 1 type of diesel engines were used in the estimates. It should be noted that emission factors for particulate matter (PM) are developed for total particulates (*i.e.*, all particle sizes); therefore, using PM emission factors for the PM10 evaluation is a highly conservative approach.

The EPA recommends the following formula to calculate hourly emissions from non-road engine sources:

$$M_i = N \times HP \times LF \times EF_i$$

where:

M_i	=	mass of emissions of ith pollutants during inventory period
N	=	source population (units)
HP	=	average rated horsepower
LF	=	typical load factor
EF_i	=	average emissions of ith pollutant per unit of use (*e.g.*, grams per horsepower-hour)

Based on the equipment operation schedule, emission rates were calculated as: 1) peak hourly rates used for a short-term averaging period (a less-than-24-hour average period) concentration prediction, and 2) daily average hourly rates for the long-term (24-hour and annual average periods) concentration calculation.

The estimates of peak and daily average hourly non-road diesel engine emission rates are presented in Table 313846-2. It should be noted that in predicting the annual average NO_x concentration level from an in-river stationary booster pump operation, an annual average hourly emission rate was used. This emission rate (2,618 grams/hour as compared to the peak hourly rate of 4,292 grams/hour in Table 313846-2) was calculated based on the projected percentage of booster usage hours over the total annual working hours, which is 61 percent.

On-Site Truck

Under the mechanical dredging scenario, on-site truck operations will be part of the sediment transportation plan. Criteria pollutants will be emitted from the truck engines at the site, as well as from the on-site roadway.

Based on the processing capacity of the facility, a total of 10 one-way truck trips (five two-way trips) per hour, from a total of five trucks, would occur between the unloading and loading areas. The five truck operations were modeled as continuously running vehicles, as follows:

- Two hours of idling at the unloading area.
- Two hours of idling at the loading area.
- One hour of running at a speed of five miles per hour along the on-site unpaved road.

Therefore, all five trucks are conservatively assumed to be running without a break during each operating hour. Truck-engine exhaust emission rates were calculated using EPA's Mobile5b emission factor model, and the dust emissions from the on-site roadway were estimated using EPA's Part5 model. Based on EPA's emission-factor document for various industrial operations (September 1988 and AP-42, January 1995), a water-suppression method can be expected to result in dust-control efficiency of more than 50 percent. Therefore, given: 1) the high water content (more than 50 percent) in the sediment to be transferred by trucks, and 2) periodically spraying the on-site unpaved roadway surface, which will be performed using water hoses or water trucks, a 50-percent dust-control efficiency was used in unpaved roadway PM emission estimates. Truck operation-related emission rates are presented in Table 313846-3.

Locomotive

The locomotive engine emission rates (Table 313846-2) were conservatively estimated based on the emission standards established by the EPA (December 1997) for the switch duty-cycle type of Tier 1 engine. The average power of each locomotive was assumed to be 3,000 hp. It should be noted that emission factors for PM are developed for total particulates (*i.e.*, all particle sizes); therefore, using PM emission factors for the PM10 evaluation is a highly conservative approach.

Tug Boat

The emission rates (Table 313846-2) for tug boats were estimated based on a unit fuel consumption rate of 0.06 gal/hp-hr provided by Gahagen and Bryant Associates, Inc. (GBA) (Thomas, June 1, 2001) and the emission factor for an uncontrolled diesel industrial engine in EPA's AP-42. It should be noted that emission factors for PM are developed for total particulates (*i.e.*, all particle sizes); therefore, using PM emission factors for the PM10 evaluation is a highly conservative approach.

Sediment Handling and Storage Pile Dust Emissions

Dust emissions (Table 313846-4) due to a drop-type operation will occur at several stages in the storage and transport cycle, with the most emissions occurring under a mechanical dredging scenario. Stages at which such emissions could occur are when:

- An excavator unloads sediment from a barge and onto a conveyer.
- The conveyer transfers the dredged sediment onto a pug mill to be mixed with cement materials.
- The materials are transferred to a staging area.

- A loader places processed sediment onto a truck.
- A truck is moving sediment along the on-site unpaved roadway to the loading area.
- A loader places sediment onto a rail car.

Three sediment-dropping operations will occur before dewatering (loading from barge by an excavator, unloading to a conveyer, and then dropping to a pug mill). Three sediment-dropping operations will occur after dewatering but before transporting to the loading area (dropping onto a conveyer from a pug mill, dropping to a staging area from a conveyer, and loading onto a truck). Two sediment-dropping operations would occur in the loading area (unloading from a truck and loading onto a rail car). The quantity of unit-dust emissions generated by each of the above dropping operations was estimated based on the weight of sediment and the water content of sediment transferred and the following EPA AP-42 empirical equation:

$$\underset{\text{(lb/ton)}}{EF} = k * (0.0032) * (u/5)^{1.3} / (M/2)^{1.4}$$

where:

EF = emission factor

k = particle size multiplier (0.35 for PM10 and 0.74 for total suspended particles [TSP]);

U = mean wind speed (mile/hour), in this case, an average mean wind speed of 5 miles per hour was used

M = material moisture content (defined as the ratio of weight of water to weight of solids)

The weights of sediment materials transferred would be 80 tons per hour before dewatering and 88 tons per hour after dewatering and cement mixing. The moisture (water) content in sediment was conservatively assumed to be 50 percent in emission estimates. The estimated uncontrolled sediment handling emission rates are summarized in Table 313846-4. The total suspended particle (TSP) emission rate estimates were further used in determining the potential impacts from dust-adhered PCBs (White Paper – PCB Releases to Air).

Sediment Stabilization and Solicitation Dust Emissions

Fugitive dust emissions may potentially be released during the sediment mixing and processing period. However, stabilization and solidification of sediment will be designed to occur within enclosed areas in each sediment processing/transfer facility. These enclosures will include areas for the cement mixing operations, the sediment staging locations, etc. Therefore, sediment stabilization and solicitation PM emissions would likely be negligible and are not addressed further in this paper.

Other Pollutant Emissions

In addition to the criteria pollutants discussed above, certain non-criteria pollutants with the potential to be emitted from a stationary source are also regulated under the Title V of the Clean Air Act and the NYSDEC Part 201 regulation. Non-criteria pollutants, such as volatile organic compounds (VOCs), certain metals, hydrogen sulfide (H_2S), etc. have the potential to be emitted from stationary sources in the sediment processing/transfer facility. Typical stationary sources may include aboveground storage tanks (ASTs), pug mills, fugitive dust sources, etc.

Based on the forecasted fuel-consumption data for the selected remedy (Chapter 8 of this RS), the EPA TANKS model (Version 4.09) was used to estimate VOC emissions associated with the diesel fuel requirements for operations at both the NTF and STF. It was assumed that the AST needed would have an external floating roof with a 20-ft diameter and would be refilled once a week, for a total of 30 times, between May and November of each year. The fuel consumption data and estimated annual VOC emissions are presented in the following table:

Aboveground Storage Tank Annual VOC Emission Rate

Facility	Dredging Condition	Fuel Consumption (gallons/yr)	Annual Emission Rate (lbs/yr)
NTF	Hydraulic	104,950	14.5
	Mechanical	91,670	14.4
STF	Hydraulic	160,150	15.2
	Mechanical	145,850	15.0

Additionally, certain metals may adhere to fugitive dust emissions. Due to the lack of detailed emission factors or emission rate data for each potential source, the metals emissions cannot be quantified at the present time but will be addressed during the design phase.

IMPACT DISPERSION MODELING

The dispersion modeling techniques used in this paper are consistent with the *Guideline on Air Quality Models* (EPA, 1999). The most recent version of the EPA-approved numerical air dispersion model was used. Air quality impacts for the modeled criteria pollutants were determined at receptor locations adjacent to the NTF and a potential booster pump near the river's shoreline for the applicable averaging periods. Results were then compared to the ambient air quality standards established by the EPA and NYSDEC in order to determine the compliance of the proposed operations with these regulations .

ISCST3 Dispersion Model

For this analysis, ISCST3, an EPA-approved refined air quality dispersion model, was used. ISCST3 is a steady-state Gaussian plume dispersion model designed for a simple terrain analysis (*i.e.*, terrain elevation below the stack height). This model is used to assess pollutant concentrations for industrial facilities and to calculate impacts for several different averaging periods, ranging from one-hour and annual concentrations. A summary of the ISCST3 modeling inputs for this study is provided in Table 313846-5.

Modeled Sources

The sources at the NTF were conservatively modeled as ground level-release area sources with zero exit velocity. The area sources are located at unloading, processing, and loading areas, as appropriate. It is noted that this source modeling approach is very conservative, given the limited design information available at this phase of the project.

A point source with an exit velocity of 1 m/sec was conservatively modeled at a hypothetical location along the river, such as where a stationary booster barge will be installed every 10,000 ft to support a hydraulic dredging activity. The point source was assumed to be 12-ft high with a 6-inch diameter, which is similar to a typical diesel-engine truck exhaust stack, and was placed 150 ft from either shoreline of the Hudson River. Potential impacts from stationary booster operations were modeled within two areas, Glens Falls and Albany.

Meteorology

In predicting potential impacts from the NTF operations using the ISCST3 model, the most recent and complete five consecutive years of meteorological data (1993-1997) were used. The data included hourly surface observations from the Glens Falls weather service station and coincidental daily mixing heights from the Albany, New York National Weather Station (NWS). The Glens Falls station is the closest weather station to the NTF and the data collected at this station are considered representative of the facility site.

Potential booster pump operational impacts were modeled for: 1) the Albany area, using five consecutive years of both surface and upper air data (1995-1999) collected in Albany; and 2) the Glens Falls area, based on the same data used for the NTF impact modeling.

Receptors

A total of five sensitive receptors were modeled near the NTF including four residential houses and one club that are close to the facility. Since emission sources were modeled as ground-level area sources with no plume rise at the NTF, the greatest potential pollutant impact from continuous operations would occur at those sensitive receptors that are adjacent to the facility site. This would also be true for those receptors near both shorelines when a stationary booster is nearby, given that the source is only 12 ft high with an exit velocity of only 1 m/s.

A series of hypothetical residential houses along both sides of the river were modeled for a nearby stationary booster operated 150-ft from either shoreline. Two Cartesian receptor grid systems were placed along the river. Each grid system started 50 ft from each shoreline and extended to cover an area 150 ft wide and 2,000 ft long that is close to a potential stationary booster pump when hydraulic dredging is occurring.

Impact Results

As indicated above, many conservative assumptions were used in estimating emission rates, which consequently resulted in higher concentration forecasts. Use of the ISCST3 modeling approach to simulate all emission releases as either ground-level area sources or near-ground-level point sources with minimal exhaust velocity also resulted in conservative concentration calculations.

For the applicable averaging time periods, the worst-case concentrations at modeled sensitive receptor locations were identified for each pollutant for each dredging scenario at the NTF (Tables 313846-6 and 7). The model also simulated the potential impact along the river near a stationary booster pump in both the Glens Falls and Albany areas (Tables 313846-8 and 9). A 75-percent conversion factor recommended by the EPA (1999) was applied in converting NO_X levels (for which the emission estimates were conducted) to NO_2 levels in the concentration calculations.

In order to estimate the maximum expected total impact at a given receptor location, the facility impact was added to a representative background level that accounts for existing pollutant concentrations contributed from other sources in the region. The background levels in the project area can be characterized based on air quality data collected at State air-monitoring stations and published by NYSDEC (June 2000). Data collected at the closest monitoring location to each project site are considered representative of the background levels for the northern and southern project areas.

Total “worst-case” concentrations at the NTF are presented in Tables 313846-6 and 7. No exceedances of the NAAQS are predicted. Therefore, even using conservative assumptions and modeling approaches, the air quality impacts from continuous operations at the NTF would not be significant, and the facility operations would be in compliance with the applicable ambient air quality standards. The same conclusion is expected to be applicable for the STF, as the NTF is the worst-case site.

Total worst-case concentrations at theoretical residential houses along both sides of the Hudson River near the stationary booster pump under the hydraulic dredging scenario are presented in Table 313846-8 for the Glens Falls area and Table 313846-9 for the Albany area. No exceedances of the NAAQS were predicted; therefore, the operation of stationary booster pumps would not be expected to have a significant air quality impact.

Unlike the stationary booster pump operation, mobile dredging activities would have short-term effects on any given residential house along the river, which are of less concern. Furthermore, the combined engine horsepower from two dredge machines (one along each side of the river) is less than the engine size from one booster pump. Based on the concentration levels predicted from a stationary booster pump operation, it can be concluded that short-term dredging activities would not have significant air quality impact near the river.

Metals Associated with Airborne Suspended Particles

The sediment metal-concentration data reported by various parties have been discussed and summarized in the White Paper – Metals Contamination. A review of the risk factors (*i.e.*, sediment concentration and toxicity values [either the cancer slope factor or 1/RfD value for those metals for which they exist]) revealed the following four metals to have the highest risk associated with them:

- Lead.
- Chromium.
- Cadmium.
- Titanium.

The emission rate of total suspended particles (TSPs) from the processing/transfer facilities was estimated to be 8 g/hour during the sediment handling process (Table 313846-4). Using the average sediment metal concentrations (White Paper – Metals Contamination) and the suspended particle emission rate, the metal flux associated with TSPs from sediment-handling facilities were calculated (Table 313846-10). The annual and 8-hr maximum metal concentrations at the receptors were calculated using the ISCST3 results, and also are presented in Table 313846-10. The airborne metal concentration values were predicted to be several orders of magnitude lower than the corresponding applicable standards (these ambient and OSHA standards are also presented in Table 313846-10 for comparison). Therefore, the release of metals associated with the suspended particles is not expected to cause any adverse health effects.

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Odor Evaluation
255361

WHITE PAPER – ODOR EVALUATION

(ID 255361)

ABSTRACT

This white paper evaluates the potential to generate nuisance odors during implementation of the selected remedy. Based on the analysis of diesel air emissions from construction equipment that is presented in White Paper – Air Quality Evaluation and experience at numerous construction sites and the fact that new equipment must comply with rigorous air emission standards, no nuisance odor at nearby residences is expected due to the use of diesel-engine equipment. The potential for nuisance odor due to release of hydrogen sulfide (H_2S) or ammonia was also evaluated. The available data for the relevant parameters (sediment sulfate, nitrogen, organic carbon, pH, dissolved oxygen, oxidation-reduction potential, etc.) were reviewed; however, most of the required information is unavailable for the Hudson River. Therefore, porewater data from Mississippi River sediments were used to estimate the potential for odor associated with dredging in the Upper Hudson River.

Based on this analysis, there is a hypothetical potential for short-term episodes of occasional H_2S odor outside sediment processing/transfer facility boundary and near dredging locations if no mitigation measures are employed. However, the likelihood of odor problems is believed to be small based on experience at other sediment sites. During remedial design, Site-specific porewater H_2S concentration data will be collected and the odor generation potential of the sediment will be evaluated in greater detail. A number of mitigation measures may be employed to address potential H_2S odor generation, if necessary. The predicted H_2S concentrations inside and outside the facility boundaries as well as at dredging locations, were projected to comply with applicable ambient and workplace standards. Based on un-ionized ammonia levels, concentrations of ammonia in the air at receptor locations were found to be several orders of magnitude lower than applicable ambient and workplace standards. In addition, the occurrences of odor problems associated with a number of river-dredging projects were investigated. Project personnel indicated no significant odor problems from dredging or sediment processing facilities.

INTRODUCTION

Several commenters were concerned that implementation of the selected remedy will generate nuisance odors at the dredging locations and the sediment processing/transfer facilities.[1] The

[1] It is important to note that EPA has not yet determined the location(s) of sediment processing/transfer facilities necessary to implement the selected remedy. For purposes of the Feasibility Study, example locations were identified from an initial list of candidate sites based on screening-level field observations which considered potential facility locations from an engineering perspective. In the Feasibility Study, it was necessary to assume the locations of sediment processing/transfer facilities in order to develop conceptual engineering plans, analyze equipment requirements, and develop cost estimates for the remedial alternatives. For this purpose, two example locations were identified: one at the northern end of the project area in the vicinity of the Old Moreau Dredge Spoils Area, and one at the southern end of the project area near the Port of Albany. Each of these example locations fulfills many of the desired engineering characteristics for such a facility to support the remedial work, and is

possible sources of odors are operation of construction equipment, the dredged material itself, and processing operations. This white paper evaluates the potential that nuisance odors may be generated during the construction and operational phases of the selected remedy.

CONSTRUCTION EQUIPMENT

Implementation of EPA's selected remedy will entail the use of diesel-driven construction equipment both during project mobilization and actual in-river dredging operations. The equipment that will be used on the Upper Hudson River is typical of conventional construction machinery that is seen operating on most land-based and marine work sites. Examples of diesel equipment that may be employed on the Upper Hudson River are as follows:

- Backhoes, graders, and front-end loaders during construction of the sediment processing/transfer facilities.
- Mechanical and hydraulic dredges.
- Towboats and survey vessels.
- Sediment processing/transfer facility vehicles and machinery.

Equipment such as that identified above typically does not generate nuisance odor complaints. Diesel-engine emissions are regulated by both federal and State agencies, and manufacturers of new construction equipment are required to comply with increasingly stringent emission standards. Thus, based on experience at numerous construction sites and the fact that new equipment must comply with rigorous air emission standards, there will be little likelihood that diesel-driven systems will generate nuisance odors at nearby residences.

An air quality analysis has been conducted for EPA's selected remedy (White Paper – Air Quality Evaluation). Among other matters addressed in that analysis are the impacts of project-related diesel emissions on ambient conditions. On the basis of the analysis, EPA has concluded that diesel emissions from project construction equipment will not generate a significant ambient impact.

SEDIMENT PROCESSING/TRANSFER FACILITY CONSTRUCTION

One commenter raised the possibility of odors that may be generated if an existing landfill, such as Moreau, is selected as the location of a sediment processing/transfer facility. As has been

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representative of reasonable assumptions with regard to distance from the dredging work and cost. Other locations, both within the Upper Hudson River valley and farther downstream, are possible.

The example facility locations presented in the Feasibility Study have also been used in the Responsiveness Summary in order to clarify material presented in the Feasibility Study and Proposed Plan and in connection with additional noise, odor and other analyses that were performed in order to respond to public comments. EPA will not determine the actual facility location(s) until after EPA performs additional analyses and holds a public comment period on proposed locations and considers public input in the final siting decision. Thus, all information provided in this Responsiveness Summary relative to potential impacts of the sediment processing/transfer facilities on communities, residents, agriculture, the environment and businesses should likewise be considered representative and illustrative. Further specific assessment of and, as necessary, mitigation of, potential impacts will be addressed during remedial design.

stated elsewhere in this Responsiveness Summary, processing/transfer facility sites will be determined during remedial design; thus, the site identified by the commenter may not be selected.

However, in the event that the Old Moreau Dredge Spoils Area is considered and subsequently selected, it should be noted that dredged sediments from the Hudson River were deposited at this location more than 30 years ago. These sediments had migrated from above the Fort Edward Dam to the channel near Rogers Island, from which they were mechanically dredged and hauled by truck to the Moreau location. It is expected that, given the limited excavation that would be likely to occur in this area in order to construct a sediment processing/transfer facility, the age of the materials in the Old Moreau Dredge Spoils Area, and EPA's experiences at the Remnant Deposits, it is not expected that odors would be generated by construction work at this location.

DREDGED MATERIAL

Hydrogen Sulfide

Hydrogen sulfide (H_2S) gas is produced from the microbial decomposition of sulfur-containing organic matter and sulfate (SO_4^{-2}) under anaerobic conditions (Water Environment Federation [WEF], American Society of Civil Engineers [ASCE], 1995). An analysis of the equilibrium speciation of sulfur is presented by Stumm and Morgan (1981). Any significant reduction of SO_4^{-2} to H_2S requires a strong reducing environment; therefore, the availability of H_2S in the porewater of dredged sediment will depend on the sediment characteristics and the *in-situ* redox condition.

Mass Transfer Coefficient

The mechanism of H_2S transfer from the dredged sediment/water mixture to air is the same as aqueous-phase PCB transfer to air. The water-to-air PCB transfer has been analyzed in detail in White Paper – PCB Releases to Air. The H_2S transfer flux from the sediment/water mixture may be estimated by multiplying the mass transfer coefficient and the concentration gradient between the liquid and the air phase. The same mass transfer correlations may be used as those presented in White Paper – PCB Releases to Air; the equation numbers in the following paragraphs refer to the equation numbers in that white paper.

The diffusion coefficient of H_2S in air is estimated using the empirical equation of Fuller, Schettler, and Giddings (Geankoplis, 1982) to be 0.175 cm^2/sec. The diffusion coefficient may be substituted in the equation below to calculate the $k_{a,H2S}$ as:

$$
\begin{aligned}
k_{a,H2S} &= k_{a,\,H2O} * [D_{H2S} / D_{H2O}]^{0.61} \\
&= 1.0 * [0.175/0.26]^{0.61} \\
&= 0.767 \text{ cm/sec} \\
&= 662.68 \text{ m/day}
\end{aligned}
$$

The liquid diffusivity of H_2S is reported to be 0.000141 cm^2/sec (Cussler, 1984); therefore, the Schmidt number (Sc) for H_2S is estimated to be 709. The Sc may be substituted in the equation below to calculate the $k_{w,H2S}$ as:

$k_{w,H2S}$ $= k_{w,\,CO2} * [Sc_{CO2} / Sc_{H2S}]^{-0.5}$
$= 0.0076 * [709/600]^{-0.5}$
$= 0.007$ cm/sec
$= 6.048$ m/day

Using an average Henry's Law constant of 0.01086 atm-m^3/mol for H_2S (Metcalf and Eddy, 1991) and substituting the mass transfer coefficient can be estimated as:

$1/K_{OL}$ $= 1/k_w + RT/Hk_a$
$= 1/6.048 + 8.2057 \times 10^{-5} * 298 / (0.01086 * 662.68)$
$= 0.1689$
or: K_{OL} $= (1/0.1689)$ m/day
$= 5.92$ m/day
$= 0.0068$ cm/sec

Air and Aqueous-Phase H_2S Concentration

In the Upper Hudson River sediments, the organic carbon content is between 0.13 and 0.15 percent and the sulfur content is between 1 and 4 percent (USGS, 2001). These organic carbon and sulfur contents in the river sediments indicate that there is potential for microbial production of H_2S in the anaerobic sediment. However, no data for H_2S concentrations in porewater of the Upper Hudson River sediments was found in the published scientific literature. A review of related literature found that only one recent study has been conducted to evaluate porewater characteristics (*i.e.*, for Mississippi River sediment [Dwyer *et al.*, 1997]).

It is possible that the results from the Mississippi River may be comparable to porewater characteristics of the Upper Hudson River. The organic carbon content of Mississippi River sediment ranged from 0.2 to 5.2 percent and the sulfide levels ranged from 0.005 µmoles/g (0.000016 percent) to 63.0 µmoles/g (0.216 percent). These values are comparable to those reported by USGS (2001) for the Upper Hudson. Assuming that the sediment porewater characteristics, (including pH, dissolved oxygen, and oxidation-reduction potential) of the Mississippi River are similar to those in the Upper Hudson River, the H_2S concentration in the porewater may be assumed to be the same in the Upper Hudson, as a preliminary estimate. The reported mean ammonia and H_2S content of Mississippi River sediment porewater (ranges in parentheses) are as follows: un-ionized ammonia 0.007 (0.000 to 0.025) mg/L and hydrogen sulfide 0.023 (0.000 to 0.569) mg/L.

Assuming 20 percent excess water by volume and 40 percent porosity of the mechanically dredged material, the resulting liquid-phase H_2S concentration is:

$(0.023 * 0.32)/(0.32 + 0.2)$
$= 0.014$ mg/L

Substituting K_{OL} and C in eq. (2), the H_2S flux from water can be estimated as:

F $= K_{OL} * C$

$$= (5.92 \text{ m/day})*(0.014 \text{ mg/L})*(1000 \text{ L/m}^3)$$
$$= 82.88 \text{ mg/m}^2\text{-day}$$

One-hour and annual maximum H_2S concentrations at the receptor locations outside the facility boundary and at the dredging location were calculated using the ISCST3 model and are presented in Table 255361-1. The H_2S concentrations (8-hour) inside the facility boundary were also calculated and are presented in Table 255361-1. The relevant New York State and OSHA standards are also presented in Table 255361-1 for comparison. The predicted airborne H_2S concentration values were found to be lower than the corresponding applicable State or OSHA standards; therefore, the release of H_2S is not expected to cause regulatory exceedances or adverse health effects.

The H_2S recognition threshold level (WEF and ASCE, 1995) relates to the minimum H_2S concentration required for a typical person to perceive and recognize its odor and is also presented in Table 255361-1. Based on porewater H_2S data from other riverine sites (*i.e.*, Mississippi), the predicted short-term airborne H_2S concentrations outside the facility boundary and near dredging locations indicate the hypothetical possibility of brief episodes of occasional H_2S odor if no mitigation measures are taken. Site-specific porewater H_2S concentration data will be collected during remedial design and the odor generation potential of the sediment will be evaluated in greater detail. A number of mitigation measures, including oxidation of H_2S (using chlorine or hydrogen peroxide) or use of covered tanks followed by air treatment, for example, may be employed to address potential H_2S odor generation, if necessary. However, it must be emphasized that the evaluation described above is hypothetical as no Site-specific data are currently available. The likelihood of odor problems is believed to be small based on experience at other sites.

Air and Aqueous-Phase Ammonia Concentration

Ammonia may be released during dredging if the porewater pH is high (above 8) and there is sufficient ammonia present in the porewater. The nitrogen data for Hudson River sediments indicate that average total nitrogen concentrations are typically 0.3 percent (White Paper – Potential Impacts to Water Resources). Conditions in the water column of the Hudson River are not favorable for ammonia formation, which typically forms at or above pH 8. The Hudson River has a pH range of 6 to 8.

Using the same mass transfer correlations from the White Paper – PCB Releases to Air and the aqueous-phase NH_3 concentration from the Mississippi River sediment (Dwyer *et al.*, 1997), the un-ionized ammonia flux from the sediment/water mixture may be estimated to be 45.36 mg/m^2-day. One-hour, 8-hour and annual maximum NH_3 concentrations were calculated using the ISCST3 results and are presented in Table 255361-2. The New York State, OSHA standards and recognition threshold value are also presented in Table 255361-2 for comparison. The predicted airborne NH_3 concentration values were found to be several orders of magnitude lower than the corresponding applicable standards or threshold value; therefore, the air phase NH_3 release is not expected to cause any odor problems or adverse health effects.

ANECDOTAL EVIDENCE

Several commenters provided information on other contaminated sediments projects gathered from agencies and Potential Responsible Parties (PRPs). It is noteworthy that the information submitted provides little mention of negative odors at other locations.

For this RS, EPA has queried participants in several dredging projects that involved the removal of contaminated sediments. A brief synopsis of odor-related comments from these discussions follows:

- Dredging of sediments near Rogers Island in the mid-1970s apparently generated no significant issues with regard to odor, based on the recollection of one participant (Thomas, 2001). It was also noted that Hudson River sediment-sampling work completed during the early and mid-1990s did not generate a noticeable odor.

- Work at Fox River's Sediment Management Unit (SMU) 56/57 in Wisconsin raised no odor issues and there was virtually no odor noted (Bories, 2001). It is important to note that this dredge site was located three miles downstream of two wastewater treatment plant outfalls; thus, the sediments removed at SMU 56/57 could be considered representative of sediments in the Upper Hudson with respect to sulfur compounds since there are also wastewater treatment outfalls in the Upper Hudson. Since no odor problems were noted at SMU 56/57, no odor problems are anticipated for sediment removal in the Upper Hudson.

- Recent demonstration dredging (mechanical) at New Bedford Harbor in Massachusetts generated only minor, musty, marine-type odors from sediments.

- Dr. G-Yull Rhee of the New York State Department of Health is conducting a study of odors in water supplies. Dr Rhee stated that, other than some low-level musty odors, the Hudson River sediments will not cause an odor nuisance (Rhee, 2001, pers. comm.) Dr. Rhee also stated that where higher levels of organic compounds (including PCBs) are encountered, odor-generating potential does exist. However, targeted Hudson River sediments contain about 31.2 ppm PCBs on average and do not meet the criteria for higher levels of organics suggested by Dr. Rhee.

- At the Pine River site in Michigan, noticeable odor levels were when dredging occurred in areas heavily contaminated with DDT. In addition, targeted areas located in vegetation produced a strong vegetative decomposition odor; however, this odor was described as a musty odor, not a hydrogen sulfide odor, according to the US Army Corps of Engineers staff person (Bories, 2001). This experience is not considered relevant to the Upper Hudson River due to substantially different site-specific conditions. For instance, the main contaminants at Pine River were pesticides (specifically, DDE and DDT); the sediments also contained various solvents and total petroleum hydrocarbons at 1 percent levels, which emit odors. In contrast, the main contaminants at the Hudson River PCBs Site are PCBs, which are considered odorless.

SUMMARY

Nuisance odor associated with operating diesel-engine construction equipment and dredging, barging, and processing PCB-contaminated sediment in the Upper Hudson River is expected to be of little or no impact to the surrounding community. Due to the low organic matter content, as well as the low nitrogen and sulfur concentrations in the Hudson River sediments, nuisance odor associated with H_2S gas and ammonia is not expected. Experiences at other contaminated sediment dredging sites support this conclusion. Based on data obtained from the Mississippi River, there is a hypothetical potential for short-term episodes of occasional H_2S odor outside the sediment processing/transfer facility boundary and near dredging locations if no mitigation measures are employed. During remedial design, Site-specific data of H_2S concentrations in sediment porewater will be collected and the odor generation potential of the sediment will be evaluated in greater detail. A number of mitigation measures may be employed to address potential H_2S odor generation, if necessary.

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Noise Evaluation
312685

WHITE PAPER – NOISE EVALUATION

(ID 312685)

ABSTRACT

Concerns were raised regarding the potential for nuisance noise from the dredging and operation of the sediment processing/transfer facilities.[1] This white paper provides discussions of potential noise sources associated with various activities, discussions of analysis methodology, and prediction of noise levels at the worst-case receptor location. Noise effects were evaluated through comparisons with appropriate guidelines, with the finding that predicted noise levels would not exceed guidelines for most activities. One exception would be noise levels from stationary booster pump operations at potential receptor locations under the hydraulic dredging scenario. However, if such a situation arose, a series of noise mitigation measures can be applied to reduce stationary booster pump noise to acceptable levels.

During the remedial design phase of the project, EPA will monitor existing noise conditions to better assess the impact of potential noise increases resulting from the selected remedy. EPA will also consider noise-mitigation measures to minimize potential noise impact to communities.

INTRODUCTION

Potential noise impacts from the selected remedy can be divided into two categories:

- Long-term impacts, such as would result from:

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[1] It is important to note that EPA has not yet determined the location(s) of sediment processing/transfer facilities necessary to implement the selected remedy. For purposes of the Feasibility Study, example locations were identified from an initial list of candidate sites based on screening-level field observations which considered potential facility locations from an engineering perspective. In the Feasibility Study, it was necessary to assume the locations of sediment processing/transfer facilities in order to develop conceptual engineering plans, analyze equipment requirements, and develop cost estimates for the remedial alternatives. For this purpose, two example locations were identified: one at the northern end of the project area in the vicinity of the Old Moreau Dredge Spoils Area, and one at the southern end of the project area near the Port of Albany. Each of these example locations fulfills many of the desired engineering characteristics for such a facility to support the remedial work, and is representative of reasonable assumptions with regard to distance from the dredging work and cost. Other locations, both within the Upper Hudson River valley and farther downstream, are possible.

The example facility locations presented in the Feasibility Study have also been used in the Responsiveness Summary in order to clarify material presented in the Feasibility Study and Proposed Plan and in connection with additional noise, odor and other analyses that were performed in order to respond to public comments. EPA will not determine the actual facility location(s) until after EPA performs additional analyses and holds a public comment period on proposed locations and considers public input in the final siting decision. Thus, all information provided in this Responsiveness Summary relative to potential impacts of the sediment processing/transfer facilities on communities, residents, agriculture, the environment and businesses should likewise be considered representative and illustrative. Further specific assessment of and, as necessary, mitigation of, potential impacts will be addressed during remedial design.

- Equipment operation at sediment processing/transfer facilities.
- The transfer of processed, dredged materials via barge or railroad.
- Booster pump operation along the river (under the hydraulic dredging scenario).

- Short-term impacts, such as would result from:

  - Construction of the sediment processing/transfer facility and associated buildings, roads, parking lots, etc.
  - Dredging and backfilling activities, which would be constantly moving along the river.

As the location and detailed design of the sediment processing/transfer facility sites have not been finalized, the assessment of noise effects from long-term, on-road, mobile-source operations cannot be accurately and quantitatively performed for either truck or train operations at this time. However, these mobile-source operational impacts will not be significant, given the projected number of trips (13 daily truck trips and two daily train trips, when applicable) that would occur even with seasonally continuous operations. The analysis presented in this white paper includes a quantitative evaluation of long-term noise from the sediment processing/transfer facility operations and booster-pump operations.

Given the difference in duration between long- and short-term activities, the short-term impacts from the selected remedy would be similar to impacts from a typical construction project; *i.e.*, they would only occur for a short period of time at any given receptor location. The noise effects of short-term activity are generally of less concern than those from long-term seasonally continuous operations and are generally handled qualitatively in most environmental documentation (*e.g.*, Environmental Assessments [EAs], Environmental Impact Statements [EISs], and Records of Decision [RODs]). However, a quantitative noise evaluation for dredging activities was performed for the selected remedy and is discussed herein.

Noise impact is typically assessed in two ways: the absolute noise level compared to the applicable noise criteria, and the net change in noise levels compared to existing conditions.

Because the locations of the sediment processing/transfer facilities have not yet been determined, noise surveys to establish existing ambient noise conditions at potentially impacted areas cannot be conducted at this time. Potential noise impacts based on the net change in noise levels resulting from the selected remedy will be assessed after an on-site noise monitoring study is performed. In this future noise increment analysis, the New York State Department of Transportation- (NYSDOT) recommended increment impact criterion would be used as the impact assessment guideline for long-term continuing operations.

This white paper deals only with potential noise effects related to absolute noise levels from seasonally continuous sediment processing/transfer facility operations and stationary booster-pump and dredging operations moving along the Hudson River.

NOISE FUNDAMENTALS

One way of describing fluctuating sound is to describe the fluctuating noise, heard over a specific time period, as if it had been a steady, unchanging sound. For this condition, a descriptor called the equivalent sound level, or L_{eq}, can be computed. The L_{eq} descriptor is the constant sound level that, in a given situation and time period (*e.g.*, one-hour L_{eq} [$L_{eq}(1)$], or 24-hour L_{eq} [$L_{eq}(24)$]), conveys the same sound energy as the actual time-varying sound. Statistical sound-level descriptors such as L_1, L_{10}, L_{50}, L_{90}, and L_x are also sometimes used to indicate noise levels which are exceeded 1, 10, 50, 90, and x percent of the time, respectively. These terminologies can be found from many noise reference books or noise analysis guidance, such as the New York City Environmental Quality Review Technical Manual (1993).

The $L_{eq}(1)$ descriptor is in standard use by various agencies as the most appropriate metric for estimating the degree of nuisance or annoyance that would occur from increased noise levels occurring during a typical peak hour.

It is often useful to account for the difference in response of people in residential areas to noises that occur during sleeping hours as compared to waking hours. A descriptor, the day-night noise level (L_{dn}), is defined as the A-weighted average sound level in decibels during a 24-hour period, with a 10-decibel (dBA) weighting applied to nighttime sound levels. The 10-dBA weighting accounts for the fact that noises at night are more perceptible to humans because there are fewer background sounds to obscure the noise. The L_{dn} descriptor has been proposed by the US Department of Housing and Urban Development (USHUD), the EPA, and other organizations as one of the appropriate criteria for estimating the degree of nuisance or annoyance that increased noise levels would cause in residential neighborhoods.

Given the characteristics of the selected remedy, $L_{eq}(1)$ and L_{dn} have been selected as the noise descriptors to be used in the noise impact analysis. $L_{eq}(1)$ is used to evaluate peak-hour noise conditions and L_{dn} is used to evaluate the nighttime noise component.

Human response to changes in noise levels depends on many factors, including the quality of the sound, the magnitude of the change, the time of day at which the changes take place, whether the noise is continuous or intermittent, and the individual's own ability to perceive the changes. The average ability of an individual to perceive changes in noise levels is presented in the table below.

Average Ability to Perceive Changes in Noise Levels

Change in Decibels (dBA)	Human Perception of Sound
2-3	Barely perceptible
5	Readily noticeable
10	A doubling or halving of the loudness of sound
20	A "dramatic change"
40	Difference between a faintly audible sound and a very loud sound
Source: FHWA, June 1995.	

Generally, a 3-dBA or smaller change in noise level would be barely perceptible to most listeners but a 5-dBA level would be readily noticeable. A 10-dBA change is normally perceived as a doubling (or halving) of noise levels. These thresholds permit estimation of an individual's probable perception of changes in noise levels.

Noise Standards and Criteria

There are a number of standards and guidelines adopted by federal and State agencies for assessing noise impacts. These regulations and standards are useful to review in that they provide both a characterization of the quality of the existing noise environment as well as a measure of project-induced impacts.

Federal Highway Administration (FHWA) (23 CFR 772)

The FHWA noise regulations contain noise abatement criteria (NAC) that the FHWA considers to be the acceptable limits for noise levels for exterior land uses and outdoor activities and for certain interior uses (Table 312685-1). While the selected remedy is not a highway project, the FHWA noise regulations offer measures that can be evaluated in the context of the selected remedy. According to the FHWA NAC, if noise levels from highway traffic at an impacted receptor location exceed the corresponding L_{eq} or L_{10} criteria listed in Table 312685-1, abatement measures, such as the installation of noise barriers, if feasible or reasonable, need to be considered. Although it is EPA's expectation that the facilities well be located in an industrial or commercial area, the determination of which NAC will apply will depend on where the sediment processing/transfer facilities are sited.

US Department of Housing and Urban Development (USHUD) Environmental Criteria and Standards

As shown in the table below, USHUD Site Acceptability Standards, USHUD has adopted environmental standards, criteria, and guidelines for determining acceptability of federally assisted projects and has proposed mitigation measures to ensure that activities assisted by USHUD will achieve the goal of a suitable living environment. These guideline values are strictly advisory.

USHUD Site Acceptability Standards

Noise Zone	Day/Night Sound Level (L_{dn})
Acceptable	Not exceeding 65 dB
Normally Unacceptable	Above 65 dB but not exceeding 75 dB
Unacceptable	Above 75 dB
Source: 24 CFR Part 51	

USHUD funding assistance for the construction of new noise-sensitive land uses is generally prohibited for projects with "unacceptable" noise exposure (as defined in the table above) and is discouraged for projects with "normally unacceptable" noise exposure, without suitable mitigating measures.

This policy applies to all USHUD programs for residential housing, college housing, mobile home parks, nursing homes, and hospitals. It also applies to USHUD projects for land development, new communities, redevelopment, or any other provision of facilities and services that is directed toward making land available for housing or noise-sensitive development.

Sites falling within the "normally unacceptable" zone require implementation of sound mitigation measures: 5 dB if the L_{dn} is greater than 65 dB but does not exceed 70 dB, and 10 dB if the L_{dn} is greater than 70 dB but does not exceed 75 dB. If the L_{dn} exceeds 75 dB, the site is considered "unacceptable" for residential use.

USHUD encourages noise attenuation features in new construction or in alterations of existing structures. The USHUD-mandated or recommended design mitigation measures to eliminate or minimize "unacceptable" or "normally unacceptable" levels, respectively, include well-sealed double-glazed windows, forced-air ventilation systems (which permit windows to remain closed in summer), acoustic shielding, and insulation.

New York State Department of Transportation (NYSDOT) Construction Guidelines

Given its temporary effects on any given receptor location, construction activity (similar to dredging or backfill operation for the selected remedy) noise is normally considered less critical than noise from a continuing operation (most noise criteria were established for continuing operations). Construction noise impact guidelines have been developed by NYSDOT for transportation projects. Relevant to the selected remedy is the guideline for temporary construction noise, which defines "impact" as occurring at levels exceeding $L_{eq}(1) = 80$ dBA (NYSDOT, 1998).

ANALYSIS SCENARIOS AND SOURCES ANALYZED

Again, it is important to note that EPA has not yet determined the locations of the sediment processing/transfer facilities necessary to implement the selected remedy. For purposes of the FS, example locations were assumed to include a northern transfer facility (NTF) and a southern transfer facility (STF).

The project would involve 1) seasonal sediment processing/transfer operations at fixed locations and 2) a dredging process that would move along the Hudson River over a six-year period. Unlike the operations at a sediment processing/transfer facility, the dredging process, much like construction activity, would result in only temporary noise effects at any given receptor location.

For purposes of the noise analyses, the sediment processing/transfer facility operations would include seasonal operations at loading, processing, and unloading areas for approximately 20 hours per day, including a total of five nighttime hours. Each sediment processing/transfer facility could be constructed for either mechanical or hydraulic dredging processes.

In identifying the type of noise source that would likely contribute the most to noise levels, the basic noise fundamental concept considered in this analysis is that a small noise source contributes negligible noise emissions if a large noise source exists. For example, if a 100-hp generator has a noise reference level of 65 dBA and a 1,000-hp generator has a noise reference level of 90 dBA, the total combined noise level of these two generators would be 90.01 dBA (*i.e.*, the small source only contributes 0.01 dBA increase of the total noise level compared to the large source alone). Since a 3-dBA difference is a barely perceptible noise change, small noise source contributions were omitted in this analysis in order to simplify the study. The sample calculation is shown below by using an equation for calculating the combined L_{eq} as $L_{eq} = 10 \log \Sigma 10^{Leq(i)/10}$:

$$10 \log \Sigma 10^{Leq(i)/10} = 10 \log (10^{90/10} + 10^{65/10}) = 90.01 \text{ (dBA)}$$

A complete project equipment list can be found in White Paper – River Traffic. However, it should be noted that, for noise analysis purposes, the following noise sources and associated operational conditions are not necessarily described in the same way as in other white papers. For example, three tug boats to be operated 30 minutes each in a non-overlapped way in one day are considered, in the noise analysis, as one tug boat to be operated 30 minutes for three times in a day.

Long-Term Operational Activity at Sediment Processing/Transfer Facilities

Northern Transfer Facility

Mechanical Dredging Scenario

With respect to operation under the mechanical dredging scenario, the NTF would include the following potential noise sources:

- Two front-end loaders in the enclosed, temporary staging building loading sediment onto dump trucks.
- Three dump trucks near the unloading dock.
- Two front-end loaders in the semi-open loading area loading sediment onto rail cars.
- Two dump trucks near the loading area.
- One locomotive running for 30 minutes during the daytime period (7 am – 10 pm) twice a day.
- A 350-hp tugboat running continuously for 30 minutes each trip, with a total of six daytime (7 am – 10 pm) trips needed per day.
- A 350-hp tugboat running continuously for 30 minutes each trip, with a total of three nighttime (10 pm – 7 am) trips needed per day.
- Five round trips each hour along on-site roadways.
- One electric-powered material handler (CAT 375 MH). However, the noise from such a machine is substantially lower than a diesel-powered excavator of the same capacity

because of the lack of diesel engine noise (Stonehocker, June 15, 2001); this machine is, therefore, omitted from the noise analysis.

- Pug mill, conveyor belts, and other necessary small generators would not likely generate a noticeable difference in noise levels from the large noise sources listed above. Furthermore, most of these small sources would be electric-powered or installed in an enclosed room or building, as they would be in a typical water-treatment plant, so the exterior noise from these small sources would not be significant. Therefore, they are not considered in the evaluation.

Hydraulic Dredging Scenario

With respect to operation under the hydraulic dredging scenario, the NTF would include the following potential noise sources:

- One locomotive running continuously for 30 minutes, twice a day, during daytime periods (7 am – 10 pm).

- One 1,000-hp tugboat running for 60 minutes, three times a day, during daytime periods (7 am – 10 pm).

- Other necessary small-horsepower generators, auxiliaries, and boosters would not be likely to generate a noticeable difference in noise levels from the large noise sources listed above. Furthermore, most of these small sources would be electric-powered or installed in an enclosed room or building, as they would be in a typical water-treatment plant, so the exterior noise from these small sources would not be significant. Therefore, they are not considered in the evaluation.

Southern Transfer Facility

Mechanical Dredging Scenario

With respect to operation under the mechanical dredging scenario, the STF would include the following potential noise sources:

- Three front-end loaders within an enclosed and temporary staging building.

- Two diesel-powered excavators (CAT345).

- One locomotive running continuously for 30 minutes twice a day during daytime periods (7am-10pm).

- One 1,000-hp tugboat running continuously for 60 minutes three times a day during daytime periods (7am-10pm) and once a day during nighttime periods (10pm-7am).

- Pug mill, conveyor belts and other necessary small generators would unlikely generate a noticeable difference in noise levels from the above large noise sources. Furthermore, most of these small sources would be electrical-powered or installed in an enclosed room or building, as they normally would be in a typical water treatment plant, so that the exterior noise from these small sources would not be significant. Therefore, they are not considered in the evaluation.

Hydraulic Dredging Scenario

An STF configured for hydraulic dredging would involve operation of fewer noise sources than a mechanical-dredging facility. Therefore, the potential noise effects would be less than from a mechanical dredging facility, and were not analyzed in this white paper.

Long-Term Booster Pump Operations

Under the hydraulic dredging scenario, one stationary 1,000-hp booster pump would be installed along the river at each 10,000-foot increment of dredging distance. For purposes of this noise analysis, it is assumed that each booster pump would be operated 17 hours per day, including two nighttime hours.

Short-Term Dredging Activity

Mechanical Dredging

Short-term mechanical-dredging activities would include the following potential noise sources:

- One large excavator (CAT 375) for deep dredging 100 ft from the west bank shoreline.

- One small excavator (CAT 345) for shallow dredging approximately 400 ft from the west bank shoreline.

Hydraulic Dredging

Short-term hydraulic-dredging activities would include the following potential noise sources:

- One 600-hp dredge machine operating approximately 150 ft from the west bank shoreline.

- One 1,000-hp booster operating approximately 150 ft from the west bank shoreline and trailing 2,000 ft behind the dredge machine.

Equipment Noise Reference Levels

Reference noise levels (measured at 50 ft) for each major noise source to be used for the project are identified below.

Excavators

The noise emission reference levels for the specific excavator models proposed for the mechanical dredging activities (*e.g.*, CAT 345 and CAT 375) were obtained from the manufacturer (Foley Inc., 2001). Based on the levels for various operational movements for each excavator, the highest tested level was conservatively used for the $L_{eq}(1)$ in the analysis (79 dBA for the CAT 375 model and 74 dBA for the CAT 345 model).

Hydraulic Dredge Machine

In a hydraulic dredging scenario, the $L_{eq}(1)$ reference level of 77 dBA at 50 ft is assumed in this analysis for the proposed 600-hp dredge machine. This level was the highest level measured on June 11, 2001, based on a series of on-site measurements approximately 50 ft away from an active dredging spot near Cape Cod. The dredge machine on the site was an Ellicott Wheel Dragon B890 model with a 624-hp pump plus a 210-hp auxiliary pump. The dredging process and capacity monitored are similar to that proposed for the selected remedy.

Booster

In a hydraulic dredging scenario, a 91 dBA $L_{eq}(1)$ reference level was used for the 1,000-hp booster, based on the manufacturer-provided noise reference level for a 1,020-hp diesel engine (Thomas, February 2, 2001).

Front-End Loader

A typical front-end loader can result in a peak noise level of 84 dBA at 50 ft (Table 312685-2). It should be noted that the noise levels summarized in Table 312685-2 are the peak levels for each piece of equipment. The average noise levels would likely be 2 dBA lower than the peak levels, as suggested by FHWA (FHWA, 1976).

Locomotive

An 80.4 dBA $L_{eq}(1)$ noise reference level was provided in the US Federal Transit Administration (FTA) guidance (April 1995).

Tugboat

Tugboat noise levels were considered to be comparable to a diesel engine with a similar operational capacity. However, since a tugboat is normally operated at less than 60 percent of its rated power (Thomas, June 1, 2001), especially during low-speed barge loading and unloading processes, and furthermore, the boat's engine noise is shielded to a certain degree by either the tugboat deck or an engine room, the average noise from a tugboat with a rated horsepower of

1,000 hp is expected to be less than that from a 600-hp diesel engine. Therefore, a reference level of 77 dBA $L_{eq}(1)$ was assumed for the 1,000-hp tugboat, based on the 600-hp hydraulic dredge machine reference level measured at Cape Cod and discussed above. Subsequently, a 74 dBA $L_{eq}(1)$ reference level was used for the 350-hp tugboat, due to the same load factor considerations discussed above. The level is based upon a full-powered 286-hp excavator (CAT 345).

Truck

Noise reference levels for an idling truck engine during loading and unloading process are conservatively assumed to be 80 dBA under the worst-case (full throttle) condition (FHWA, 1998).

ANALYSIS METHODOLOGY

Equipment Noise

Estimated equipment noise levels were derived from recommendations provided in *Highway Construction Noise: Measurement, Prediction and Mitigation* (FHWA, 1976) as per the following:

$$EL(i) = L(i) + EF$$

$$L_{eq}(i) = EL(i) - 20 \log D(i)/D_0$$

$$L_{eq} = 10 \log \Sigma\, 10^{Leq(i)/10}$$

where:

EL(i) is the average cycle noise emission level for equipment i;

L(i) is the peak noise emission level of i equipment obtained in Table 312685-2 or from manufacturer;

EF is the equivalency factor to adjust peak noise level to average equipment cycle noise level. Here an average EF is about -2 dBA for equipment at material processing facilities and 0 for dredging excavators;

D(i) is the distance from receptor to construction equipment i;

D_0 is the reference distance at which L(i) is measured (*e.g.*, D_0 = 50 ft for the level identified in Table 312685-2 for a front end loader);

$L_{eq}(i)$ is the sound level resulting from operation of equipment i; and

L_{eq} is the cumulative sound level from all equipment.

Truck Running Noise

Heavy truck-running noise along the truck route between loading and unloading areas within the NTF was modeled using the FHWA's Traffic Noise Model (TNM), assuming an average travel speed of 25 miles per hour.

Nighttime Background Noise

In order to determine L_{dn}, a 40-dBA background-noise level was assumed for those nighttime hours in which no operations would occur. This level is typical for a quiet suburban nighttime background condition (NYCDEP, 1993) and would result in a conservative L_{dn} level in the calculation if the area were rural. Noise levels for the remaining nighttime hours were the same as the daytime levels as a result of the project's long-term operations, plus a 10 dBA nighttime noise penalty.

RECEPTORS

The sensitive noise receptor locations analyzed include:

- In the long-term sediment processing/transfer facility impact evaluation – The residences that are closest to each facility site (a camp location near the NTF was also considered).

- In the short-term dredging impact evaluation – Typical residences approximately 50, 100, and 200 ft from the west bank shoreline along the river near any dredging site. This includes a total of 49 homes located within 200 ft or less of the river's shoreline. Of these 49 homes, 8 are located within 50 ft, 12 are located within 50 to 100 ft, and 29 are located within 100 to 200 ft.

ANALYSIS ASSUMPTIONS

Sediment Processing/Transfer Facility

A conservative building noise attenuation of only 10 dBA was applied to noise that would be generated from operation of equipment inside the enclosed staging area at both the NTF and STF. This attenuation level is equivalent to the typical noise reduction from a normal residence with open windows (FHWA, June 1995).

A 3-dBA noise shielding is applied to the noise levels to be generated from equipment operation at the semi-enclosed structure at the loading area at both the NTF and STF. This level is considered achievable, even with a long-but-discontinuous wall that blocks 40-65 percent of the area between a source and a receptor (FHWA, 1978). Since the structure at the loading area would be constructed as an enclosed building (except that it will have the built-in flexibility of either end being capable of opening for entering and exiting rail cars), a more-than 3-dBA reduction is expected from this building.

Stationary Booster Operation

For purposes of this analysis, it was assumed that, under the hydraulic dredging scenario, each needed stationary booster would be installed at a distance greater than 400 ft from the closest receptor location near the river.

Short-Term Dredging Activity along Hudson River

Mechanical Dredging

Deep dredging activities would move along a river path approximately 100 ft from the shoreline and shallow dredging activities would be stationed approximately 100 ft from the opposite shoreline.

Based on an estimated rate of movement of 2.5 ft per hr, the mechanical dredge would take 407 hours, or approximately five weeks, to move a distance of 1,000 ft. Thus, the mechanical dredge would be in a zone of 1,000 ft that is close to any given receptor along the river for a 10-week period.

Hydraulic Dredging

The hydraulic dredge machine, trailed by a booster pump 2,000 ft behind it, would move along a river path approximately 150 ft from the shoreline. Each additional 10,000 feet of moving distance would require the addition of a booster pump. Therefore, any affected residents would first hear the noise from the hydraulic dredge working and, once work is accomplished and the dredge moves farther downstream, they would then hear the noise produced from the trailing booster pump.

The hydraulic dredge and the booster pump would move down the river at an estimated rate of 6.5 ft per hr and would operate seasonally for 17 hrs per day, six days per week. Based on these parameters, the hydraulic dredge would be in a zone of 1,000 ft that is close to any given receptor along the river for a three-week period.

When the dredge machine moves away and the trailing booster pump moves closer to any given receptor, the booster pump would then dominate temporary noise effects. In order to evaluate the worst-case noise impact range from a moving booster-pump operation, a 3,000 ft worst-case zone, in which the trailing booster would stay for a total of nine weeks, was used for noise evaluation.

ANALYSIS RESULTS

Based on the methodology and the assumptions described above, the potential noise effects from both sediment processing/transfer facility and dredging activities were estimated at the worst-case receptor locations.

Long-Term Operational Noise Levels

The long-term predicted noise levels from the sediment processing/transfer facilities and a stationary booster are summarized and compared to the appropriate noise guidelines in Tables 312685-3 and 4, respectively. Overall, the sediment processing/transfer facility operational noise levels would be below the applicable guidelines at any existing receptor locations. However, noise effects from a stationary booster pump would be significant for receptors along the Hudson River within approximately 1,000 ft of the booster.

Short-Term Dredging Noise Levels

Mechanical Dredging

At the worst-case receptor (50 ft off the shoreline) within the worst-case 10-week period (1,000-ft dredging zone), the $L_{eq}(1)$ would begin at 57 dBA level, then reach a peak level of 70 dBA at the end of the fifth week, and return to 57 dBA after a 10-week period as the dredging operations move down the river. These worst-case dredging noise levels (Table 312685-5) would not exceed the NYSDOT construction-noise impact guideline of 80 dBA.

Hydraulic Dredging

During the worst-case nine weeks (in which the trailing booster would move in a zone from 3,000 ft upstream to 3,000 ft downstream) when a dredge machine and a trailing booster pump are close to any given receptor location along the river, the noise levels (Table 312685-6) at the closest receptor location (50 ft off the shoreline) would vary as follows:

- $L_{eq}(1)$ would start from 57-dBA level and approach mid-peak of 66 dBA within two weeks, when the dredge machine is at the nearest point.

- $L_{eq}(1)$ would remain at mid-60s levels during the following four weeks, before the trailing booster becomes a dominant noise source.

- $L_{eq}(1)$ would reach a peak level of 79 dBA in the middle of the fifth week, when the trailing booster is at the nearest point to the receptor.

- $L_{eq}(1)$ would drop to the level of 56 dBA from the peak of 79 dBA within the next four weeks after the booster reaches the downstream worst-case zone boundary (3,000 ft downstream), at the end of nine-week period.

Overall short-term noise levels during the worst-case nine-week dredging period would not exceed the available NYSDOT construction-noise impact guideline of 80 dBA at any existing receptor locations.

DISCUSSION AND MITIGATION CONSIDERATIONS

The above noise levels were predicted using various conservative assumptions such as:

- Each piece of equipment, truck, booster, etc., was assumed to run continuously during the identified operational time period. For example, a truck within the NTF loading area was assumed to run continuously for 60 minutes per hour and 20 hours per day without a break. In fact, all equipment operations would need maintenance work periodically and thus lower noise levels would occur compared to predicted levels.

- Each piece of equipment, truck, booster, etc., except for tugboats, is assumed to run continuously under the full capacity during all phases of operations. For example, a 1,000-hp booster pump to be stationed in the river under the hydraulic dredging scenario is assumed to run continuously for 60 minutes per hour and 17 hours per day at the maximum load condition. In fact, all equipment would be operated at full capacity only occasionally, rather than continuously as assumed in the analysis. Thus, in reality, lower noise levels would occur compared to predicted levels.

- For equipment operations that have an operational cycle with different movements or functions under various power-settings (such as digging, moving, and dumping movements during an excavator cycle), the maximum reported L_{eq} level was conservatively assumed as an average level for the equipment.

The results of the noise analyses using the conservative assumptions discussed above are summarized in the following sections.

Long-Term Operational Effects from Sediment Processing/Transfer Facilities

- Noise levels generated from the sediment processing/transfer facilities at receptor locations near each facility would not exceed the FHWA NACs for Category B and C. As indicated above, it is EPA’s expectation that the facilities well be located in an industrial or commercial area. The determination of which NAC will apply will depend on where the sediment processing/transfer facilities are sited.

- Day-and-night noise levels generated from the sediment processing/transfer facilities at receptor locations near each facility would not exceed the USHUD acceptability guidelines for housing.

- Sediment processing/transfer facility noise effects from hydraulic dredging would be generally less than for mechanical dredging.

- The greatest noise effects would occur at the NTF under the mechanical dredging scenario but would be still below applicable guidelines.

Long-Term Operational Effects from Stationary Boosters

- Noise levels generated by a 1,000-hp stationary booster at residences within a 1,000-ft radius of the booster would be significant.

- Mitigation measures can include:

  – Enclosure of the booster operation.
  – Using an electric-powered booster.

- – Carefully selecting booster location as far away as possible from the nearest receptor.
- – Reducing nighttime booster operational hours to the extent possible.

Short-Term Effects of Dredging Noise

- Noise levels under both mechanical and hydraulic dredging scenarios would not exceed the NYSDOT short-term construction impact guideline.

- Noise effects from mechanical dredging would be less than those from hydraulic dredging.

General Noise Mitigation Consideration

Even though most of the absolute noise levels summarized for various activities would not exceed the applicable guidelines, a perceptible noise increase may occur, especially during nighttime operations. As indicated in the beginning of this paper, the potential noise increase will be assessed after an extensive on-site noise monitoring study is performed during remedial design.

However, as per a series of conversations with the company providing noise-control measures for typical diesel equipment (MacDonald, August 2, 2001), a 10-dBA reduction of equipment noise can be readily achieved through special design considerations. These noise-reduction measures include utilizing insulation, silencers, etc. Therefore, the noise levels summarized in this white paper can be reduced, when specific equipment is considered during the project's design phase.

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Project-Related Traffic
253245

WHITE PAPER – PROJECT-RELATED TRAFFIC

(ID 253245)

ABSTRACT

Commenters raised concerns with regard to the potential for impacts to local communities from project-related traffic. While implementation of the selected remedy would generate additional truck and auto trips mainly in the vicinity of the sediment processing/transfer facilities,[1] it has been suggested that the project will also create congestion on adjoining roadways at these locations, thus increasing the need for road maintenance and resulting in a higher occurrence of accidents. EPA has estimated the traffic that will be generated by the proposed activities and evaluated it in the context of current area road capacities and volumes.

Using available traffic data, an analysis has been conducted for select roadways in the vicinity of the Thompson Island Pool (TI Pool) (assumed for the purpose of this study to be the location of a northern transfer facility [NTF]). It is concluded from the analysis that traffic generated by the project will not be disruptive to local communities in the TI Pool vicinity, as the volume increase on nearby roadways will be minor. Also, since the increase in road usage is relatively small, it is not likely that there will be a need for significant road maintenance as a result of the selected remedy. Impacts for a southern transfer facility (STF) site were not evaluated, as that area, assumed to be near the Port of Albany, is highly industrialized and experiences much greater vehicular activity than would be generated by project operations.

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[1] It is important to note that EPA has not yet determined the location(s) of sediment processing/transfer facilities necessary to implement the selected remedy. For purposes of the Feasibility Study, example locations were identified from an initial list of candidate sites based on screening-level field observations which considered potential facility locations from an engineering perspective. In the Feasibility Study, it was necessary to assume the locations of sediment processing/transfer facilities in order to develop conceptual engineering plans, analyze equipment requirements, and develop cost estimates for the remedial alternatives. For this purpose, two example locations were identified: one at the northern end of the project area in the vicinity of the Old Moreau Dredge Spoils Area, and one at the southern end of the project area near the Port of Albany. Each of these example locations fulfills many of the desired engineering characteristics for such a facility to support the remedial work, and is representative of reasonable assumptions with regard to distance from the dredging work and cost. Other locations, both within the Upper Hudson River valley and farther downstream, are possible.

The example facility locations presented in the Feasibility Study have also been used in the Responsiveness Summary in order to clarify material presented in the Feasibility Study and Proposed Plan and in connection with additional noise, odor and other analyses that were performed in order to respond to public comments. EPA will not determine the actual facility location(s) until after EPA performs additional analyses and holds a public comment period on proposed locations and considers public input in the final siting decision. Thus, all information provided in this Responsiveness Summary relative to potential impacts of the sediment processing/transfer facilities on communities, residents, agriculture, the environment and businesses should likewise be considered representative and illustrative. Further specific assessment of and, as necessary, mitigation of, potential impacts will be addressed during design.

TRUCK AND AUTO GENERATION

Mobilization Phase

The construction and mobilization phase of the project will involve developing the sediment processing/transfer facilities. Facility expansion will require movement of employees and materials to and from the project sites. It should be noted that the areas discussed herein are only for analytical purposes.

Northern Transfer Facility

As indicated in the FS, construction costs to develop the NTF have been estimated at approximately $15.1 million for construction-related labor, materials, and equipment. For purposes of this discussion it is assumed that construction of the NTF will take nine months and that traffic will be generated as a result of the need to bring labor and material onto the site on a regular basis. It is further assumed that 40 percent of the construction cost is for labor (about $6 million), 40 percent is for materials, and about 20 percent is for equipment costs.

With respect to generation of traffic, it can be expected that the busiest phase of construction will be the site preparation phase, when grading materials are being brought in and concrete is being placed for foundations and structures. It is assumed that this phase of the construction work will take about three months and that construction of the remainder of the facility (*i.e.*, dewatering facilities, water treatment systems, etc.) will require about six months. Since the first phase of construction involves importing relatively low-priced commodities (concrete, grading materials, etc.), it is estimated that about 25 percent ($1.5 million) of the materials cost will be for these commodities (with about $755,000, or 50 percent, for grading materials and $755,000 for concrete).

Traffic associated with bringing in the grading materials and concrete may then be estimated on the basis of the cost of these commodities ($20 per ton for grading materials and $95 per yard for concrete) and the quantity that can be hauled in each truckload (20 tons per load for grading materials and 15 tons per load for concrete). Using these parameters, it is estimated that about 34 truckloads per day of low-cost commodities will arrive at the NTF site during the first three months of construction. The material that arrives over the remaining six months will consist of relatively costly commodities (on a per-unit weight basis) such as pumps, conveyors, valves, pipes, electrical gear, etc. It is estimated that up to five truckloads of these materials will arrive at the NTF site each day over the remaining six months of construction.

Neither the 34 truckloads per day of low-value materials nor the five truckloads per day of higher-valued commodities will create congestion on local roadways. As discussed in the next section of this white paper, there is substantial existing capacity on the roadways that are assumed to be used for project deliveries. In addition, material deliveries are expected to occur throughout the workday and not during peak commuting hours when roads are most congested. Please note that the volume of trucks addressed here is for the mobilization phase, and is not representative of the truck deliveries estimated for the operational phase.

Construction employment is expected to average 50 workers over the nine-month construction period for the NTF. This number was estimated on the assumption that average labor costs (including benefits, other overheads, and contractor profit) will be $80 per hour and that work occurs five days per week for nine months. Since construction phase employment at the processing/transfer sites is less than that expected during the project's operational phase, the estimate of operational phase traffic impacts that follows is applicable to the construction phase as well. As will be noted from the information provided below, vehicular movements associated with commuting employees will only have a minor impact on nearby roadways.

Southern Transfer Facility

The assumed siting of the STF in an industrial zone in the Port of Albany area will limit the impacts of traffic associated with the construction of this sediment processing/transfer facility. The Port of Albany is a very active industrial waterfront area that is served by excellent road and highway connections, including a component of the interstate system. In addition, materials deliveries can be readily accomplished by either rail or barge, thereby avoiding the roadway system entirely. Thus, a specific analysis of traffic impacts at this location is not presented herein since no significant impacts are expected.

Operational Phase

Northern Transfer Facility

During routine operations, trucks will deliver supplies to the NTF site, including fuel, stabilization agents, water treatment supplies, equipment/lubricants, and office/cafeteria supplies, and remove trash. It should be noted that EPA does not intend to haul dredged material by truck but rather to move this commodity by rail to suitably permitted landfills, thus avoiding local roadways. In addition, EPA has committed to moving backfill materials within the Upper Hudson River area either by rail or in river barges. Thus, movements of neither dredged material nor backfill will contribute to traffic near the NTF.

In addition to the routine delivery of supplies, additional vehicular movements will occur during the project's operational phase as a result of employees commuting to work each day. Employees will arrive at the NTF site to operate the sediment processing/transfer facilities, as well as to support in-river operations such as dredging. An evaluation of the potential for project operations to cause roadway congestion follows.

- **Stabilization Agent:** A stabilization agent is required to improve the handling properties of mechanically dredged sediments. It has been estimated that 112 tons per day of agent will be required for sediment processing. Assuming the use of 20-ton delivery trucks, six trucks per day will required at the NTF. In the case of hydraulic dredging, there is no need for stabilization agent, but additional materials are required to support the large slurry processing and water treatment plant. The number of trucks needed to deliver processing materials is also estimated to be six per day.

- **Fuel:** The total diesel fuel requirements for each dredging scenario are addressed in Chapter 8 of the RS. Assuming the use of 5,000-gallon delivery trucks, the fuel deliveries required per week for each dredging scenario are outlined in Table 253245-1.

- **Other:** Trucks will also be needed to deliver general supplies, chemicals, equipment/lubricants, and office/cafeteria supplies and for trash removal. The frequency of the delivery will depend on the type of material. A summary of the truck delivery schedule for the NTF is contained in Table 253245-2.

  The number of deliveries required at the NTF, assuming a six-day workweek, will be approximately 13 trucks per day: seven large (*i.e.*, 20-ton trucks for stabilization agent and 5,000-gallon diesel tanker trucks) and six small/medium (*i.e.*, standard parcel-delivery trucks) vehicles. It is not expected that trucks will make their deliveries at peak commuter hours. As a result, they are not expected to contribute to roadway congestion, which, as explained below, is most likely to occur at peak commuting hours.

- **Employees:** The number of operational-phase employees was evaluated based on the nature of activities occurring at the sediment processing/transfer facility. It is estimated that there will be 34 employees per day shift, 32 employees per night shift, and 10 visitors per day at the sediment processing/transfer facility. In addition, there will be 12 employees per day per shift to support the dredging equipment, 6 employees per day per shift for the towboats, and 14 employees per day per shift to support workboat operations. It is assumed that visitors will not arrive during peak-hour times, and so they are not included the congestion analysis. Therefore, it is expected that 130 employee auto movements could occur during peak traffic conditions when sediment processing/transfer facility work shifts change (for example, between 5:00 pm and 6:00 pm).

- **Estimating Traffic Impacts:** Annual average daily traffic (AADT) volumes were obtained from the NYS Department of Transportation (DOT) Web site (NYSDOT, 2001) for several roads in the Upper Hudson River valley. The AADT represents the number of vehicles traveling in both directions over a designated section of highway in a 24-hour period. Each roadway section represents an area where volumes are approximately equal. The AADT values were used to determine the impacts, generally, of traffic generated by project activities. An industry standard assumption is that nine percent of the AADT occurs during the peak hours.

  As stated in the previous section, the potential exists for 130 employee trips to occur during a shift change at the NTF. If it is assumed that these additional vehicular movements were to occur at the time local roadways experienced peak traffic flows, it is be possible to estimate project impacts under these relatively conservative conditions.

  In order to complete the calculation it is also necessary to assume a directional flow of traffic leaving and arriving at the NTF. For this purpose it is assumed that no employee vehicles move south along West River Road, that 50 percent of the movements are along Route 197 to the west, and the remainder are along Route 197 to the east. On this basis, the percentage increase in vehicular movements under peak-hour conditions has been estimated for several local roadway segments, as shown in Table 253245-3.

As an example, Route 197 between Route 32 and the Washington County line has 713 vehicles traveling along this road at the peak hour (in both directions). It is estimated that the project could add 65 cars to this section of highway, with a resulting 9 percent increase in traffic under peak-hour conditions. It also should be noted that County Road 197/Reynolds Road and Route 9 are two-lane roadways that typically have a maximum capacity of approximately 1,800 cars under peak-hour conditions. Thus, these roadways are currently operating at well below capacity, and the additional project-related traffic will not substantially change the performance of the roads.

In addition to the AADT, the design-hour volume to the rated capacity (*i.e.*, volume-to-capacity ratio) was reviewed. Project-related traffic will not adversely affect the volume-to-capacity ratio of roads near the NTF, further indicating that there is sufficient capacity in the vicinity of the NTF to accommodate project-related traffic. However, it should be noted that a level of service (LOS) analysis is required in order to fully understand traffic impacts. A LOS analysis, which measures the operating conditions within a traffic system and how those conditions are perceived by drivers, will be performed during the project's design phase.

Southern Transfer Facility

There will be an increase in employee vehicular activity and truck deliveries when transfer/processing operations are initiated at the STF. Approximately six trucks per day will be required to support project operations. Employee commutation-related traffic is estimated at 102 auto trips per day. The impacts of project-related traffic are expected to be negligible due to the industrial character of the Albany area, where the sediment processing/transfer facility is assumed to be for the purpose of this study. Thus, a general roadway congestion analysis has not been conducted for this location.

CONCLUSIONS

Project-related traffic in the vicinity of the NTF is not anticipated to be disruptive to local communities. The principal roadways expected to be used by project employees and for deliveries are not currently operating at capacity, and the additional project traffic will increase vehicular flows by four to nine percent under peak-hour conditions. This additional traffic load is ascribed to employees traveling to and from the NTF under typical commuting conditions. The STF is not expected to impact traffic congestion because of the existing industrial nature of the area and the presence of both interstate highways and alternative modes for materials delivery, such as rail and barge.

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River Traffic
337804

WHITE PAPER – RIVER TRAFFIC

(ID 337804)

ABSTRACT

Several commenters suggested that implementation of the selected remedy will create untenable vessel traffic congestion on the Upper Hudson River. Commenters particularly assert that the congestion would take the form of bottlenecks at various locks along the Champlain Canal and interference with the routine passage of vessels along the canal's navigational channel. Based on the analyses presented in this white paper, it is concluded that there may be some interference with other vessels passing through the canal; however, it is expected that any such impacts can be controlled with proper management of remedial work and that, overall, project-related interferences will not be significant.

INTRODUCTION

The navigable waterway within the Upper Hudson River valley, the Champlain Canal, is managed by the New York State Canal Corporation, a subsidiary of the New York State Thruway Authority. The canal typically operates from early May to late November, depending on weather conditions. The canal's navigable channel is authorized for a depth of 12 feet, and its width varies from 75 feet (land cuts) to somewhat over 200 feet (in-river reaches) (J. Dergosits, pers. comm., February 9, 2000). Hudson River water levels and navigational access are managed through a series of locks and dams located throughout the system. The river pools upstream of each lock and dam, thereby allowing a more or less constant water depth to be maintained. In relation to normal pool elevations, the canal provides a 15.5-foot headroom clearance, with the current exception of the railroad bridge located north of Lock 3. To obtain full clearance at this bridge, the pool level must be lowered by lowering flashboards at the Lock 3 Dam.

Lock dimensions are the principal limitations on the size of vessels able to use the canal. Typical lock dimensions are 328 feet by 45 feet, and the actual available horizontal clearance for vessels passing through the locks is 300 feet by 43.5 feet (J. Dergosits, pers. comm., February 9, 2000). The locks are operated on an as-needed basis during regular hours of operation, which are from 7 am until 10:30 pm. However, with advance notice, commercial users may pass through the locks 24 hours per day (J. Dergosits, pers. comm., February 9, 2000).

Passage time through a lock (called a "lockage") is approximately 30 minutes (J. Dergosits, pers. comm., February 9, 2000). For commercial traffic, one barge can lock-through at one time. Under normal situations, a barge is moved into the lock by its attendant towboat. In the case of specialty barges of approximately 300 feet in length, the barge is pulled through the lock with a winch and the attendant tug passes through in a separate lockage. The situation with smaller pleasure craft differs, however. According to Canal Corporation staff, there have been instances wherein as many as 20 pleasure craft have moved through a lock at one time (J. Dergosits, pers. comm., May 25, 2001).

Presently, there is essentially no commercial traffic that involves shipment of industrial commodities on the Champlain Canal. Commercial traffic currently consists of cruise ships and tour boats. In the recent past, the Champlain Canal was used to transport petroleum products and jet fuel to Plattsburgh Air Force Base (AFB). After closure of the base in 1994, bulk commodity traffic dwindled to zero (J. Dergosits, May 25, 2001). Table 337804-1 lists commercial traffic moving along the Hudson River prior to the Plattsburgh AFB closure in 1994. It should be noted that the number of barge loads shown on the table was computed assuming approximately 1,500 tons of commodity per barge.

As shown in Table 337804-1, there were approximately 150 commercial barge loads moving along the Hudson in 1989. Since it is likely that most of these barges moved through all locks on the canal, each of those locks would have experienced 150 lockages as loaded barges passed through and 150 lockages as empty barges returned, for a total of 300 lockages due to movements of industrial commodities in 1989.

CURRENT CONDITION OF THE CHAMPLAIN CANAL

Historically, the Canal Corporation routinely dredged the canal to maintain the 12-foot water depth. However, no dredging has occurred along the canal between Locks 1 and 7, the area designated as River Sections 1, 2, and 3 in the FS, since 1979, with an exception in the area where the Hoosic River discharges coarse-grained materials between Locks 3 and 4 (J. Dergosits, February 9, 2000). Annual sweeps, or depth measurements, are conducted by the Canal Corporation to determine where the river has shoaled, creating potential vessel-clearance limitations. The sweeps, conducted largely by manual methods, are recorded as feet of sediment accumulated above nominal bottom elevation. After completing the canal sweeps, the Canal Corporation publishes a Notice to Mariners identifying current water depths and areas considered to be navigational hazards due to sediment accumulation. The notice indicates numbered buoys where the shoal condition occurs and the depth of water across the channel at that location.

A recent Notice to Mariners, published in April 2001 and based on the year 2000 canal sweeps, identified maximum sedimentation at buoy number R160, north of Lock 5 and south of the Route 4 Bridge. This location is within the proposed work zones. Here, the sweeps identified only 4 feet of water on the west side of the channel, 7 feet in the center, and 12 feet on the east side of the channel. Shoaling also occurs within many other sections of the river, but is not as severe as at buoy R160.

IN-RIVER CONGESTION DUE TO EQUIPMENT REQUIREMENTS

Mechanical Dredging

As stated in the FS, excavators fitted with two-cubic-yard and four-cubic-yard buckets would conduct the bulk of removal operations, should mechanical dredging be selected as the preferred dredging technology. Sediment removed by the larger excavators would generally be placed into hopper barges that would be loaded with about 1,000 tons of material, and the hopper barges then towed to the southern transfer facility (STF). Material removed by the smaller dredge would be placed into deck barges loaded to about 200 tons, which would then be towed to the northern transfer facility (NTF).

In order to meet the removal target volume of 2.6 million cubic yards in six construction seasons, approximately 3,000 cubic yards of sediment must be removed each working day. To accomplish this it is assumed, for purposes of this analysis, that four mechanical dredges, six hopper barges, and six deck barges will be needed. If backfill material were transported to the remediation area in hopper barges, either one or two barge loads per day will be needed. Two hopper barges and two tow boats will be needed to support backfill operations, assuming that backfill is obtained from sources beyond River Sections 1 and 2. Table 337804-2 identifies the vessels and other in-river equipment that may be required to support mechanical dredging operations under the selected remedy.

As shown in the table, an estimated total of 39 vessels will be required to support the project. It is important to note that the equipment will be dispersed over 40 miles of river. There will be barges located at the dredge site, barges and towboats in transit, barges secured at sediment processing/transfer facilities, and other supporting equipment in various river sections conducting surveys and performing other work. It is expected that the worst-case situation for in-river congestion would occur when four dredges and associated supporting equipment are located in the Thompson Island Pool (TI Pool) (River Section 1).

Of the 39 project vessels shown in Table 337804-2, 24 are likely to be actively involved at TI Pool work sites (dredging, restoring shorelines, backfilling, planting) at any one time under this scenario. The other 15 pieces of equipment, not actively engaged in River Section 1, would be located throughout the 40-mile length of river. For instance, three hopper barges could be in transit with three large towboats moving toward or away from the STF. Two deck barges could be moored at the NTF for unloading and the associated towboats could be in transit**.** The second backfill barge and towboat could be situated at a bulk-materials transfer facility beyond River Section 1, outside the project area. The second fuel barge could also be at either the NTF or STF for loading purposes. Lastly, two shoreline restoration and habitat replacement vessels could be either in transit or moored for restocking with supplies.

Given that the length of the TI Pool (River Section 1) is approximately 6.3 miles, it is not expected that 24 vessels actively involved in remedial work will generate either an actual or perceived congestion problem. Several factors support this conclusion:

- Much of the work will occur off-channel in shallower sections of the river. Thus, movements of pleasure craft and tour boats in the channel will not be impacted by much of the working equipment.

- Some of the working vessels are similar in scale to tour boats and pleasure craft already using the river. This is particularly the case for various survey vessels and possibly for vessels engaged in restoration activities.

- Major pieces of equipment will tend to work in clusters and, therefore, the number of possible interactions between working vessels and other river traffic will be less than otherwise expected. The equipment clusters will include dredges and associated barges, debris collectors and associated barges, and backfill equipment and associated barges.

Most importantly, the work will be conducted in a way that limits the potential for interference with other river traffic. The contractor will be required to maintain sufficient clearance in the navigation channel for other river users to move through areas where work is in progress. In addition, movements of dredges, barges, and other vessels associated with the remediation will be directed to favor off-peak hours to avoid inconveniencing other canal users.

Hydraulic Dredging

The hydraulic dredging scenario proposes the use of a 12-inch hydraulic cutter head dredge for remediation of all targeted areas in River Sections 1 and 2, and mechanical dredging equipment in River Section 3 to perform both remediation and navigational dredging in this section. All hydraulically dredged contaminated sediments will be pumped to the NTF for processing. Due to the estimated 1,600-ton-per-day (TPD) shipping limit at the NTF, three barge loads per day, on average, of processed river sediments will be transported to the STF for rail car loading and final disposal. In addition, if backfill material were transported to the remediation area in hopper barges, either one or two barge loads per day would be needed. Two hopper barges and two tow boats would be needed to support backfill operations.

Table 337804-3 identifies the vessels and other in-river equipment required to support hydraulic dredging operations under the selected remedy.

As shown in the table, an estimated total of 39 vessels and other in-river equipment will be required to support the project. As noted with the mechanical dredging scenario, it is important to note that these vessels will be dispersed over 40 miles of river. It is expected that the worst-case situation for in-river congestion would occur when the hydraulic dredge and all supporting equipment are located in the TI Pool (River Section 1). As indicated in Table 337804-3, of the estimated 39 project vessels, 18 are likely to be actively involved at TI Pool work sites (dredging, restoring shorelines, backfilling, planting) at any one time under this scenario.

The other 21 pieces of equipment would be either located in River Section 3 (spread out over 30 miles) or situated outside the remediation area. For instance, three hopper barges could be in transit, with three large towboats moving toward or away from the STF. The second backfill barge and towboat could be situated at a bulk-materials transfer facility beyond River Section 1, outside the remediation area. The second fuel barge could also be at either the NTF or STF for loading purposes. Lastly, one shoreline-restoration vessel and one habitat-replacement vessel could be expected to be either in transit or stationary somewhere, being restocked with supplies.

Given the 6.3-mile length of the TI Pool (River Section 1), it is not expected that 18 vessels actively involved in remedial work will generate either an actual or perceived congestion problem. Factors supporting this conclusion are the same as those previously presented under the mechanical dredging scenario.

In addition, in the case of hydraulic dredging, EPA will require the remedial contractor to develop a work plan that limits the potential for interference with other river traffic. As with the mechanical dredging scenario, the contractor will be required to maintain sufficient clearance in the navigation channel for other river users to move through areas where work is in progress. Movements associated with the remedial work will be directed to favor off-peak hours to avoid inconveniencing other canal users. Finally, the contractor will be required to distribute both the equipment and its movements so as to minimize the potential for project-related congestion

LOCK CONGESTION

Current Traffic on the Champlain Canal

Data is available from the Canal Corporation (J. Dergosits, pers. comm., May 25, 2001). on the number of commercial (mostly tour boats) and pleasure vessels that utilize the locks each season on a monthly basis. For the 1999 canal season, 1,361 commercial vessels and 14,298 pleasure craft traveled through Locks 1 through 6, for a total of 15,569 vessels during the period from May through November 1999. Commercial and pleasure traffic was most active at Lock 5 (Schuylerville) during the month of July 1999. Lock 1 (Waterford) had the second-highest usage in July 1999, with Lock 6 (Fort Miller) being heavily used by pleasure vessels at this time as well. The second-busiest month in relation to lock usage was August 1999 at Lock 4 (Stillwater) and Lock 1 (Waterford).

The greatest potential project-related congestion may occur from Locks 1 (Waterford) through 6 (Fort Miller). Although Lock 7 is located in the vicinity of the remedial work, it is not expected that significant project-related traffic will move through that lock. It is possible that some backfill material may come by barge from the lower Lake Champlain region, but this would involve only one, or at most two, daily movements. Since, as discussed below, arrival of backfill is not a time-critical activity, it is not expected that these one or two movements will have a significant congestion impact on Lock 7.

Table 337804-4 presents vessel traffic at Locks 1 through 6 during the July and August 1999 canal season.

Impacts Associated with Lock Passage and Lock Capacity

For purposes of this analysis it was assumed that only one commercial vessel (cruise or tour boat) could be locked through at any one time. Note that a lockage represents vessel movement through a lock in one direction. Recreational traffic consists of personal boats used for cruising, water skiing, and fishing. As discussed, Canal Corporation staff stated that up to 20 pleasure vessels have been observed being locked through at one time. Therefore, lockages associated with pleasure craft traffic were analyzed for three situations: (1) each pleasure craft is locked through individually; (2) two pleasure craft are, on average, locked through simultaneously; and (3) three pleasure craft are, on average, locked through simultaneously. These assumptions represent a range from worst-case to more probable situations likely to be encountered at the locks.

Table 337804-5 presents the canal operating schedule for the 2001 season and an estimate of the number of total lockages (both upstream and downstream) that will be available in 2001. It is assumed that each movement through the locks requires an average of 30 minutes.

The remedial project may require use of the lock system on a 24-hour-a-day basis. Table 337804-6 identifies the available lockages for the canal operating season.

Current Daily Lockages at Lock 5

As mentioned previously, Lock 5 was the busiest lock in the month of July during the 1999 canal season. This suggests that project movements through this lock may potentially cause a congestion-related impact. Table 337804-7 compares the available lockages at Lock 5 (for calendar year 2001 operating hours) to those that were actually utilized during the 1999 canal season. Usage during 1999 was derived from the three scenarios previously described, since neither specific data indicating actual vessel lockages nor data on actual lock-operating cycles were available at this time.

The table indicates that presently, assuming the worst-case scenario (Case 1), Lock 5 capacity is fully utilized during normal operating hours. This implies that any additional traffic generated by remedial work will have to move off-hours. Assuming that Case 2 more accurately portrays current Lock 5 usage, there are 10 available lockages not used during normal operating hours, implying that there would be capacity to support project-related movements during the daytime. Lastly, if Case 3 accurately portrays lock operations, there would be considerable capacity for project-related movements during the normal canal-operating day.

A similar analysis was conducted for Lock 6 (Table 337804-8), where Case 1 would suggest that Lock 6 is almost fully utilized during normal canal operating hours and, therefore, project traffic would be relegated to off-hours. Cases 2 and 3 have results similar to those for Lock 5, suggesting that project traffic could pass through Lock 6 during normal canal operating hours.

DAILY LOCKAGES ASSOCIATED WITH THE SELECTED REMEDY

Mechanical Dredging

The potential for the project to generate congestion at various locks will vary, depending on the location of the dredging operations. It is expected that sediment removed from River Section 3 will be barged southward to the STF, resulting in impacts to Locks 4 through 1. The equipment estimated to be traveling through these locks would be three hopper barges and supporting towboats (barge and towboat pass through lock simultaneously) generating requirements for six lockages at each lock in a 24-hour period. The addition of six lockages is not expected to cause congestion, since these locks are not utilized at full capacity (based on 1999 data) during the normal canal operating hours.

Removal operations in the TI Pool (River Section 1) will impact Locks 1 through 6. However, additional lockages required will be minimal, since the vessels traveling south would be three hopper barges and supporting tow boats, requiring six lockages in a 24-hour period. Additionally, one backfill barge and tow boat would move through either Locks 1 through 6 per 24-hour period or through Lock 7 depending on the source of backfill. Remaining support equipment would be traveling to and from the NTF, with no lock passage.

The worst-case congestion scenario appears to occur when two dredges operate in River Section 1 and two dredges operate in River Section 2. With this setup, project equipment (barge and tow boat combinations) is required to move both north and south through Locks 5 and 6, resulting in the greatest impact to these locks, which are the Canal's busiest.

To evaluate this particular situation, several assumptions have been made concerning the working strategy that may be adopted by the contractor. It is assumed that smaller craft (survey vessels and small work boats) will either move through the locks together with pleasure craft or will find windows of opportunity when the locks are not otherwise being used. Also, it is assumed that barges hauling backfill materials need not utilize the locks during normal operating hours, since their arrival at the work site is not likely to be time-critical. Thus, the principal project-related movements that are likely to occur during normal operating hours are those associated with moving loaded barges to the transfer facilities and those associated with returning empty barges to the work area. Table 337804-9 portrays project barge and towboat movements that have the potential to generate lock congestion.

As shown in Table 337804-10, when dredging is occurring in both River Sections 1 and 2, a total of 18 daily lockages are assumed at Lock 6 and a total of 6 daily lockages are assumed at Lock 5. It should be noted that these are lockages associated with time-critical movements; *i.e.*, movements that cannot readily be delayed to an off-hour. The table also suggests that Lock 6 will incur the most project-related traffic. The table compares current and project-related activity at Lock 6 to the available capacity at that location.

Results shown in Table 337804-10 suggest that when the worst-case scenario is assumed (one pleasure craft per lockage), there would be a potential for lock congestion at this location even assuming 24-hour operation. However, under Case 2 and 3 assumptions, considerable lock capacity would exist and there would be sufficient flexibility to allow efficient management of project traffic.

Table 337804-11 compares current and project-related activity at Lock 5 to the available capacity at that location. The analysis shown in the table indicates that the situation at Lock 5 is similar to that at Lock 6, particularly with regard to lock usage during normal canal operating hours under Case 1 conditions. However, when consideration is given to off-hour use of the lock, Lock 5 shows somewhat better performance under Case 1 conditions than Lock 6. For Case 2 and 3 assumptions, there appears to be adequate lock capacity to manage project-generated traffic efficiently.

Hydraulic Dredging

Assuming hydraulic dredging is conducted within River Sections 1 and 2, the greatest impact to locks would result at Lock 6, from small craft such as vessels associated with survey, sampling, habitat restoration, and backfill operations. It is expected that sediment dredged from River Section 3 will be removed with mechanical dredging equipment. Once removed, the dredged sediments will be placed onto hopper barges and sent to the STF, resulting in impacts to Locks 4 through 1. The equipment transiting through Locks 4 through 1 would be six hopper barges and supporting towboats, resulting in a total of 12 lockages at each lock in a 24-hour period. The addition of 12 lockages is not expected to cause congestion, since these locks are not utilized at full capacity (based on 1999 data) during the normal canal operating hours.

Removal operations in the TI Pool (River Section 1) will impact Locks 6 through 1. However, additional lockages will be minimal since the only vessels traveling south would be three hopper barges and supporting tow boats, requiring six lockages in a 24-hour period, and one backfill barge and tow boat per 24-hour period coming from either the north (Lake Champlain) or south (downstream of Lock 1). Remaining supporting equipment would be traveling to and from the NTF with no lock passage.

The worst-case lock congestion scenario would be created when the hydraulic dredge is operating in River Section 2. With this setup, project equipment (barge and tow boat combinations) is required to move both north and south through Lock 6. This setup will result in the largest impact to the busiest lock (Lock 6). Smaller craft (survey vessels and small work boats) will either move through the locks together with other vessels using the canal, or these craft will find windows of opportunity when the locks are not otherwise being used. Also, it is assumed that barges hauling backfill materials need not utilize the locks during normal operating hours, since their arrival at the work site is not likely to be time-critical.

Thus, the principal project-related movements that are likely to occur during normal operating hours are those associated with moving loaded barges to the sediment processing/transfer facilities and those associated with returning empty barges to the work area. Table 337804-12 portrays barge and towboat movements that have the potential to generate lock congestion at Lock 6 under the hydraulic dredging scenario.

Results shown in the table suggest that when the worst-case scenario is assumed (one pleasure craft per lockage), there would be a potential for lock congestion at this location under normal operating hours. However, assuming the project utilizes the locks 24 hours per day, it is anticipated that congestion at Lock 6 would not occur. In addition, under Case 2 and 3 assumptions, considerable lock capacity would exist and there would be sufficient flexibility to allow efficient management of project traffic both during normal canal operating hours and over a 24-hour-per-day operating period.

SUMMARY

Results of the analysis presented lead to the conclusion that, if appropriately managed, the proposed mechanical and hydraulic dredging alternatives will not result in navigational or lock congestion. Project vessels will be located largely shoreward of the channel during dredging operations. When work is occurring in the channel, the contractor will be required to provide adequate clearance to allow non-project traffic to move through the work area. In addition, many of the craft associated with the remedial work will be survey vessels similar in scale to other vessels that routinely navigate the Hudson.

The proposed dredging alternatives potentially cause the largest impact for Lock 6, where mechanical dredging would result in 18 additional vessel movements over a 24-hour period and hydraulic dredging would result in six additional vessel movements over a 24-hour period. These movements are not anticipated to impact current vessel traffic at this lock, and the analysis presented above indicates that the addition of project-related vessel movements to current canal traffic at locks will not exceed the capacity of the locks.

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Socioeconomics
313617

WHITE PAPER – SOCIOECONOMICS

(ID 313617)

ABSTRACT

This white paper addresses the major potential socioeconomic impacts of implementing the selected remedy, as identified by business organizations and other members of the public in the upper Hudson region. It first reviews the scale of the dredging of PCB contaminants along limited sections of the 40-mile reach of the Hudson River between the Federal Dam at Troy and Fort Edward, concluding that there is little credibility to charges that the dredging will create a regional economic disruption and stigma. It is additionally concluded that the region's economy can easily absorb the anticipated stimulus. This economic impact is estimated using the US Bureau of Economic Analysis RIMS-II economic model for the five-county upper Hudson region. Conservatively applying only 38 percent of total expenditures as expended in the region, the model forecasts $576 million in additional output for the region, $126 million in additional earnings, and more than 500 jobs per year over the six-year dredging operation.

This socioeconomic study continues with a review of some key sectors of the river-oriented economy – in particular, navigation (this section of the river is part of the New York Canal System), commercial and recreational fishing, and tourism. It is noted that this region of the Upper Hudson River valley appears not to have shared in the growth typically identified with such tourist activities elsewhere in upstate New York. Navigation will be much improved by the dredging, and short-term impacts will be minimal on existing canal traffic. Recreational fishing and wildlife observation are identified as highly valued recreational activities whose economic benefits to the region will be enhanced by the PCB cleanup. The cleanup will also substantially improve the long-term potential for commercial fishing on the Hudson River.

The white paper concludes with an examination of the potential for impacts on property values. The short-term impacts of a temporary dredging operation are not considered sufficient to generate discernable property-value losses. In fact, evidence indicates that river-property values in the Upper Hudson River valley have been depressed, compared to the value of property elsewhere in the region. The cleanup of PCBs offers the prospect of increasing property values both in the Upper Hudson River valley and along the entire river.

Property in close proximity to the sediment processing/transfer facilities[1] may be subject to some depreciation in value. The professional literature on property values and proximity to hazardous

[1] It is important to note that EPA has not yet determined the location(s) of sediment processing/transfer facilities necessary to implement the selected remedy. For purposes of the Feasibility Study, example locations were identified from an initial list of candidate sites based on screening-level field observations which considered potential facility locations from an engineering perspective. In the Feasibility Study, it was necessary to assume the locations of sediment processing/transfer facilities in order to develop conceptual engineering plans, analyze equipment requirements, and develop cost estimates for the remedial alternatives. For this purpose, two example locations were identified: one at the northern end of the project area in the vicinity of the Old Moreau Dredge Spoils Area, and one at the southern end of the project area near the Port of Albany. Each of these example locations fulfills many of the desired engineering characteristics for such a facility to support the remedial work, and is representative of reasonable assumptions with regard to distance from the dredging work and cost. Other locations, both within the Upper Hudson River valley and farther downstream, are possible.

materials, landfills, and other heavy industrial uses has been studied, but the literature is often inconsistent and indicates that impacts are affected by a variety of factors. However, the literature does appear to confirm that losses are typically recouped following the remediation. The sediment processing/transfer facilities will be carefully sited and operated to minimize potential off-site impacts so that their usage will have few negative impacts on property values. It is anticipated that the long-term benefits of the selected remedy will represent the means to significantly improve property values throughout the region.

INTRODUCTION

Several of the "umbrella" groups for the region (*i.e.*, the Adirondack Regional Chambers of Commerce [ARCC], the New York Farm Bureau, Inc., Farmers against Irresponsible Remediation [FAIR], and Scenic Hudson [along with their consultant KLIOS, Inc.]), have made substantive comments on the potential social and economic impacts of the selected remedy. These comments range widely, are sometimes highly specific, but are often of a generic nature. This white paper attempts to capture the essence of many of these comments under broad categories of concern, including impacts on tourism, fishing, navigation, and property values, and respond to these concerns. It begins with an overall review of comments and potential impacts on the regional economy and then provides reviews of the specific socioeconomic sectors.

OVERALL IMPACTS ON REGIONAL ECONOMY

Comments from ARCC assert the dredging would result in uncertainty for business, a stigma to the region, and difficulty attracting labor or new businesses. Examples of these comments are:

> "*…the uncertainty from the proposed dredging project has had, and will continue to have, a significant chilling effect on local business activity*." (ARCC, p 22.)
>
> "*Business may find it difficult to attract new employees…A limited labor pool could cause the overall wages to increase and local businesses would suffer. A massive dredging project could also consume most of the*

The example facility locations presented in the Feasibility Study have also been used in the Responsiveness Summary in order to clarify material presented in the Feasibility Study and Proposed Plan and in connection with additional noise, odor and other analyses that were performed in order to respond to public comments. EPA will not determine the actual facility location(s) until after EPA performs additional analyses and holds a public comment period on proposed locations and considers public input in the final siting decision. Thus, all information provided in this Responsiveness Summary relative to potential impacts of the sediment processing/transfer facilities on communities, residents, agriculture, the environment and businesses should likewise be considered representative and illustrative. Further specific assessment of and, as necessary, mitigation of, potential impacts will be addressed during design.

> *available rail transportation in the region. Some businesses may be forced to incur higher operating costs as they switch to truck transportation."* (ARCC, p 22.)

Stigma

Impacts of the remediation program will be related to dredging operations on sections of the 40.9 miles of the Hudson River/Champlain Canal between the former dam at Fort Edward and the Federal Dam at Troy, and to proximity to the sediment processing/transfer facilities (which will export the stabilized dredged material by rail). Historically, navigational dredging was a familiar feature in this section of the river/canal, and continues to be along other sections of the Hudson River and the Erie Canal, until it was constrained by the presence of concentrated PCBs in the late 1970s.

The dredging operations, in brief, involve a cluster of barges working over a period of six years, six days per week, up to 14 hours per day, 30 weeks per year. (For more-detailed information on the dredging operations, see White Paper – Dredging Productivity and Schedule.) The dredging will be targeted to three particular sections of the river:

- River Section 1 is the northernmost, between Fort Edward and the Thompson Island Dam (6.3 miles).
- River Section 2 is between Thompson Island Dam and the Northumberland Dam (5.1 miles).
- River Section 3 is between the Northumberland Dam and the Federal Dam at Troy (29.5 miles).

River Section 1 will undergo the most-concentrated dredging activity, with most of its six-mile length subject to dredging. Of River Section 2's five miles, about two miles will be subject to dredging, typically near only one bank. Of River Section 3's 29.5-mile length, a total of only 1.75 miles, comprising three locations on one bank of the river, will be dredged, although additional isolated areas (total of about one mile) will be dredged for navigational purposes. Thus, in linear terms, only 11 of the 40.9 miles of the upper river will experience dredging activity. The remaining 73 percent of the upper river will not be subject to dredging; in terms of surface area, only 493 (or 13 percent) of the 3,900 acres of the Upper Hudson River will be remediated.

In terms of the major urban centers along the river (Fort Edward, Stillwater, Mechanicville, Pleasantdale, Waterford, Lansingburgh, and Troy), dredging will only be adjacent to Fort Edward and Stillwater (at Mechanicville, there will be some navigational dredging on the other side of the island separating the navigational channel from this town). Although low-density residential areas are scattered all along the riverbanks, the dredging will not occur adjacent to the majority of residences comprising the study area. Proximity effects of dredging, therefore, will be substantially limited by the geography of the targeted dredging, as well as by the relatively brief duration when dredging activity will be proximate. The dredging will only be directly in front of a particular residence in a targeted area for about one week, with proximity effects, such as noise, perceptible for only about one or two weeks longer where hydraulic dredges are used, or up to six weeks longer when the mechanical dredges are used.

Once the dredging and backfill operations have passed by, the only remediation-associated activity will be ongoing monitoring activities in the river. At the sediment processing/transfer facilities, work will continue over the planned six-year period. These facilities are assumed to operate 24-hour days during the 30-week annual operating season.

For those properties along the river, the proximity impacts will be very temporary and far less than if, for example, a new building were to be built nearby. For those properties in proximity to the sediment processing/transfer facilities, there will be impacts that last for the six-year period of active operation. The sediment processing/transfer facility locations will be carefully selected to minimize potential impacts, with access to rail and water transportation for the movement of materials; they are likely to be areas with an industrial land-use history and are likely to be substantially screened and buffered from residential and other sensitive land uses (White Paper – Example Sediment Processing/Transfer Facilities). Potential impacts from the facilities will thus be relatively limited and apply only to areas of close proximity.

It is difficult, however, to predict the degree of impact that the sediment processing/transfer facilities will have on nearby property values, because such effects are dependent on many factors, ranging from the degree of odor, noise, traffic, etc.; negative media publicity; and the relative value of homes to the presence of off-setting amenities (employment centers, parks, etc.). The kind of facilities proposed here have not been studied in the scientific literature on property-value impacts, which typically deals with land uses such as landfills, incinerators, or contaminated sites. In reality, the proposed sediment processing/transfer facilities will have operational characteristics more akin to a quarry or small concrete-manufacturing plant, rather than a landfill or incinerator. Later sections of this white paper address impacts on property values and provide a review of the literature on these effects.

As indicted in the White Paper – Project-Related Traffic and White Paper – Rail Operations, the region's rail system (largely operated by the Canadian Pacific Railroad) will be able to handle the additional rail traffic without displacing existing users or creating congestion delays that will adversely affect business in the region.

The potential for the selected remedy to create a regional and long-lasting stigma is quite remote, given the temporary and limited impacts on particular locations along the 40 miles of river, and the finite operations (six years) of the sediment processing/transfer facilities that will rely on water and rail transportation. In fact, it is the selected remedy that offers the potential for removing a long-lasting stigma to the region associated with the existence of PCBs in the river.

Regional Economy

The ARCC comments present an overview of the local economy, noting that the region is finally recovering after decades of economic stagnation, with a growth in population, decline in unemployment, and increases in personal income. ARCC is correct in noting the economic gains following the recession of 1990-92 (Figure 313617-1). However, it should also be recognized that the region (whether the focus is on the four counties along the affected area – Albany, Rensselaer, Saratoga, and Washington – or if adjacent Warren County is included) experienced lower economic impacts, in terms of unemployment, than the State as a whole. For example, in

1992, the unemployment rate for New York State was 8.6 percent, whereas in the four counties it was 33 percent less, at 5.8 percent. While this favorable regional position has continued, the region's advantage has narrowed, so that in April of 2001, the State unemployment rate was 4.1 percent and the four-county region was 30 percent less, at 2.9 percent (NYS Department of Labor, 2001). Contrary to ARCC's assertion, the unemployment data indicate that the region has in fact enjoyed an advantaged position in recent economic terms.

Over the period from 1989-2001 (April), the four-county region experienced a 9 percent increase in the number of employed and an 8.4 percent increase in the labor force (NYS Department of Labor, 2001). However, the patterns of employment have significantly shifted among industries. More-detailed industry-level data, available through 1998, indicate declines in farm employment (-41 percent), manufacturing (-24 percent), construction (-17 percent), and State and local government (-7 percent) (US Bureau of Economic Analysis, 2001a). These losses were offset by gains in services (21 percent), finance, insurance, and real estate (FIRE) (14 percent), and transportation and utilities (8 percent).

ARCC acknowledges the significance of service industries in the new regional economy, especially of tourism, and goes on to claim that the dredging will adversely impact these activities (tourism is addressed later in this white paper). One of the dredging impacts ARCC claims would be the difficulty of employers to find new employees and, with a limited labor pool, dredging would cause labor costs to rise and local businesses to suffer.

It is true that the resurgent regional economy has seen a decline in the number of unemployed (Figure 313617-2). Nonetheless, in April 2001, the numbers of unemployed in the region remain substantial, with 10,500 unemployed in the four-county region and 11,700 when Warren County is added. This is a substantial pool of labor actively seeking work. The direct and indirect employment estimated to be generated by the dredging is 533 for each of the six years of the project's duration (employment impacts are discussed in greater detail elsewhere in this white paper). Moreover, to assume that the alleged stigma of dredging would inhibit recruiting new employees to the region is hardly credible, given the limits of the affected area (a narrow swath along 11 miles of river in a five-county region of almost 3,700 square miles) and the duration of impacts (which, except for the sediment processing/transfer facility sites, will pass by any particular location on the river in a matter of weeks).

The pattern of earnings and employment generated directly and indirectly by the selected remedy is discussed in greater depth elsewhere in this white paper, where an input-output (I/O) model for the region is developed. The key findings from this I/O model indicate that, of the total 3,200 jobs that will be generated in the five-county region over the six years, the construction sector will account for 25 percent (almost 800 jobs) of the employment created by the expenditures on dredging. Construction employment is followed by various business services, with about 670 jobs (21 percent), and transportation, with about 560 jobs (17 percent). A variety of other services account for the bulk of the remaining projected employment, notably in retail, health services, and eating and drinking places.

Employment in construction in the region has experienced a slower rebound than in other economic sectors, with 11.8 percent (or 2,898) fewer employed in 1998 than in 1989 (US Bureau of Economic Analysis, 2001a). It is likely, therefore, that the selected remedy, with its demand upon the labor pool, will, in fact, be a welcome addition to this economic sector of the region. Figure 313617-3 shows the recent historical trends in construction employment in the region.

The combined direct and indirect increase in employment in the region that will be generated from the dredging operation is estimated at 3,200 jobs over the six years or, if the expenditures were evenly distributed over the period, an average of 533 jobs per year. As a percent of current employment, this represents an increment of 0.14 percent to the April 2001 employed population of 386,000 in the five-county region (NYS Department of Labor, June 2001). Of the presently unemployed population of 11,700, the 533 jobs will represent 4.5 percent, or, if all the employees are drawn from this unemployed pool, the unemployment rate will be reduced from 3 percent to 2.8 percent. Therefore, the expected scale of the dredging employment is not sufficient to create discernable labor shortages or wage pressures that will adversely impact business.

REGIONAL ECONOMIC MODEL

The comments from ARCC on the regional economic impacts of the proposed PCB dredging that were noted and discussed earlier in this white paper are also relevant to this section, which continues the broad regional economic focus and also responds to comments from Scenic Hudson and Sloop Clearwater's consultant, KLIOS Inc., which estimates the positive economic impacts associated with the dredging.

KLIOS applies an I/O econometric model developed for the region by Regional Economic Models, Inc. (REMI). This model makes several input and output assumptions that are different from the US BEA RIMS II model adopted here (US BEA, 2001b). These differences include KLIOS' consideration of only a two-county region (Saratoga and Washington), the assignment of $225 million in dredging expenditures to the region (compared to $262 million in a five-county region adopted here), and their model purports to identify not only direct and indirect earnings and employment (considered in the RIMS II model), but also induced, dynamic, and structural changes in the local economy. As a result, KLIOS comments that:

> *"The overall impact of the proposed cleanup project on the regional economy between 2003 and 2008 is as follows:*
>
> - *3,543 new direct jobs.*
> - *1,028 new indirect and induced jobs.*
> - *4,571 total new jobs.*
> - *$88.5 million new direct wages.*
> - *$141 million new indirect and induced wages.*
> - *$229.5 million total new wages.*
> - *$800 million Gross Regional Product."* (KLIOS, p.19).

These estimates are greater than those generated by the RIMS II model adopted here, even though KLIOS was considering only a two-county region. The remainder of this section discusses the findings of the RIMS II model, comparing them as appropriate with the KLIOS findings.

Direct and Indirect Economic Impacts of Dredging Expenditures

The estimated total expenditure for dredging and related disposal and monitoring activities (in year 2000 dollars) is $461.9 million (FS, Table 8-11b). Of these dollars, $262.2 million, or 38 percent, are conservatively assigned as expenditures within the five-county region (Albany, Rensselaer, Saratoga, Warren, and Washington Counties). This is accomplished by a line-by-line consideration of the type and location of expenditures. For example, 10 percent of the costs associated with the transportation of sediments to Texas (if this is the ultimate disposal location) are allocated to the region, or 50 percent of the costs of barging are assigned to the region because a substantial amount of this equipment will be imported to the region. The US BEA RIMS II I/O model is used to predict the direct and indirect economic impacts of these expenditures in the region. Projections of output, earnings, and employment are possible from this model, which US BEA customized for the analysis of this specific five-county region.

The allocations of dredging and disposal expenditures were compiled into the appropriate US BEA industrial codes, of which five were used as the industries receiving direct changes in final demand (*i.e.*, an increase in these industries' total expenditures for goods/services from all other industries and households). These projected increments represent the "inputs" to the model. The five industries assigned these expenditures are: industrial-buildings construction; other new construction; rail transportation; water transportation; and engineering, architectural, and surveying services (Table 313617-1). The US BEA model creates "final demand multipliers," which are used from the 38-by-490 industry matrices to compute the total dollar output, earnings, and employment generated by the dredging activities in the five-county region.

The expenditures for dredging, disposal, and monitoring were assigned to each of the input industries, as shown in Table 313617-1. These data represent the changes to final demand in each of the industries in time-adjusted dollars (year 2000 dollars). The $262.2 million represent *direct* changes in output in the identified industries; the model also permits identification of the *indirect* economic effects generated by these inputs.

Outputs

The RIMS II I/O model identifies total changes in output in the five-county region, projected as $576.2 million, which is $314 million (or 120 percent) more than the $262.2 million in direct expenditures associated with dredging in the region. The $314 million represent additional indirect or secondary effects, as the original dollars expended on labor and materials circulate in the local economy, in turn creating increased demand in a host of other industries, as well as in the original input industries. Of the 38 industries for which the model computes outputs, those representing one percent or more of the total increase in output are identified in Table 313617-2.

Table 313617-2 indicates that the greatest share of output is allocated to households (as wages), at $126.8 million, or 22 percent of the total. The increases in construction, transportation, and business services represent a total of $309.3 million, $47 million more than the $262.3 million in direct changes in these industries, thus representing additional indirect or secondary increases in demand for these industries. Other indirect beneficiaries range from real estate ($22.4 million, or 3.9 percent) to printing and publishing, at $6 million (1 percent). KLIOS estimates an increase of

$800 million in regional product, about 39 percent more than the $576 million in output projected by the RIMS II model.

Earnings

The same approach described for projecting increases in total output is used for projecting total earnings generated by the dredging activity. The RIMS II model projects that the $262.2 million in direct expenditures will generate total earnings (wages) of $126.8 million in the five-county region. The distribution of these earnings among the different industries is shown in Table 313617-3. As would be expected, the industries representing the inputs (business services, construction, and transportation) account for the major portion of total earnings at $91.9 million, or 72.5 percent. Other industries sharing in the secondary economic impacts range from health services at $6.5 million (5.1 percent), to lodging/recreation at $1.2 million (1 percent). KLIOS estimates $229.5 million in total new wages, or 80 percent more than the RIMS II model.

Employment

The RIMS II model also projected the total (direct and indirect) employment from the dredging and disposal activities. A similar approach to that described for output and earnings is applied, except that the dollars expended in the region must be further adjusted to the 1997 benchmark data upon which labor inputs to the model are based. The discounting adjustment reduces the $262.2 million to $248.2 million, in 1997 dollars.

Table 313617-4 shows that a total increase of 3,214 jobs is predicted over the period of the dredging and subsequent monitoring operations in the five-county region. Some of these jobs will be created for the whole period, others only for a particular phase. If the new employment were evenly spread over six years, it would imply the creation of 533 jobs for each of the six years. The great majority of these jobs (63 percent) will be in those industries where the direct expenditures occur: construction (24.7 percent), business services (20.9 percent), and rail and marine transportation (17.3 percent). As deduced from Table 313617-2, about 20 percent of the jobs in these direct-input industries will be generated from the secondary, or indirect, effects and 80 percent will be created directly. Additional indirect employment will occur in a wide variety of service industries, ranging from 277 jobs in retail (8.6 percent) to 40 jobs in banking (1.2 percent).

KLIOS does not attempt to break out the industry sectors in which the new employment will occur, but estimates a total of 4,571 new jobs, which is 42 percent more than the RIMS II model predicts. The REMI model used by KLIOS is a proprietary model whose internal assumptions remain private. Perhaps a reasonable approach is to recognize that the RIMS II model presents a baseline estimate of potential impacts and that the REMI model represents an upper boundary of the positive economic impacts that the direct expenditures on dredging will generate in the immediate region. Still greater, albeit often more intangible, economic benefits will accrue to the much larger region of the entire Hudson Valley south of Fort Edward. As KLIOS suggests, these will relate to waterside economic development, commercial and recreational fishing, tourism, navigation, and property values. These subject areas are addressed as separate sections of this white paper.

IMPACTS ON TOURISM

Comments from ARCC assert the dredging will have a deterrent effect on tourism in the region. In contrast, comments from Scenic Hudson and their consultant KLIOS (Appendix A) claim that the dredging will improve the long-term potential for tourists in the region.

Short-Term Impacts

ARCC comments that the dredging operations would "*choke the canals with dredging equipment and barge traffic. The roadways will be heavily congested from increased truck and rail transportation. Tourists would avoid the area because of the noise, odors, light pollution, visual nuisances, and the threat of PCB resuspension from the project*" (ARCC, p. 19-20).

Remedial dredging operations are estimated to take six years to complete along the targeted sections of the Hudson River. Under one scenario, the plan calls for the simultaneous use of four dredges and associated barges. PCB dredging in or near the navigation channel is very limited and will be organized so that river navigation will continue to function during the day, with the possible exception of short-term restrictions when maneuvering in limited areas is required. The dredging operation's impacts on tourism, each of which will be addressed in turn, will be primarily limited to:

- Traversing the section of the river where dredging will be in operation.
- Tourists staying at a fixed location along the river (*e.g.*, an inn).
- Limited delays at locks when loaded barges transit during daylight hours.

It is clear that, at any one time, the dredging operation will cover much less than one percent of the affected length of the Upper Hudson River, and in total, over the six years, will affect only 27 percent of the upper river's length. The tourist experience on the river will remain substantially unaffected in those areas away from the dredging operation.

Travelers on the river or moving along adjacent roadways will pass through areas where dredging is in progress in a matter of minutes. For these individuals, project-generated noise, odor, and visual intrusion will be of little consequence once they are beyond the immediate work area. In these situations the impacts will be quite minimal and travelers on the Upper Hudson River, in particular, will find 99 percent of the river unaffected by the physical presence of the dredging barges. Noise impacts and the potential for odor generation are detailed in White Paper – Noise Evaluation and White Paper – Odor Evaluation; noise levels are projected to be below NYSDOT construction-impact guidelines at a relatively short distance from the dredging equipment, and odor is not anticipated to be an issue.

For those tourists that will be non-mobile (*e.g.*, staying at an inn on the river), the dredging operations will be slowly moving into proximity and then receding. The rate of movement will depend on the amount of dredging targeted at that location; however, on average, the dredging operation will be adjacent to a given location for about one week. Assuming the river has no bends, islands, or other obstructions close to the hypothetical inn, the operation will be audible for about two to six weeks. It is true that during this relatively brief period, the river will lose some of its aesthetic attraction for tourists staying at the inn; however, there is also the possibility that the dredging work will engender some interest for tourists, as it is a high-profile, if temporary, activity with a unique and historical environmental objective.

Sixteen hotels, motels, and bed-and-breakfast inns have been identified in the communities along the 40-mile section of the Upper Hudson River from Fort Edward to the Troy Dam; these are addressed in more detail elsewhere in this white paper.

Canal Navigation Congestion

With respect to the potential congestion of the river and canal locks, the operational demands and lock capacities are discussed in White Paper – River Traffic. The conclusions of the analysis are that, based on 1999 use patterns by pleasure vessels and projections of dredging operations, under all reasonable scenarios there will continue to be excess lock capacity with no congestion for pleasure vessels at locks. Consequently, few adverse impacts are anticipated for recreational boaters during the proposed remediation. Moreover, a significant portion of the dredging is oriented to navigational dredging that, when completed, will provide an expanded and safer capacity for recreational and commercial use of the river.

The following is a list of identified marinas located between Fort Edward and the Troy Dam with a brief assessment of their potential for any direct impacts from the proposed dredging, where possible. In general, potential impacts have to do with proximity to the proposed dredging site and its associated activities.

- West River Road Marina in Fort Edward is situated in the immediate proximity of a prospective dredging site. This marina could also be in the area of the proposed northern processing/transfer facility (NTF), depending upon where it is ultimately sited. For these reasons, this marina and its activities could at least temporarily be adversely affected by the remediation. Although dredging operations are likely to occur for only a few weeks near this location, its proximity to a potential sediment processing/transfer facility may generate additional longer-term impacts during the facility’s anticipated six-year life, depending on the facility’s actual location and design. However, it now appears that this marina may no longer be in operation, so potential adverse impacts may not be an issue.

- Coveville Marina in the vicinity of Schuylerville is more than 3,000 feet north of a small section of the river targeted for navigational dredging. Thus, the activities of this marina are unlikely to be adversely affected by the remediation operations.

- Schuyler Yacht Basin is a marina in Schuylerville. The closest dredging site is about a mile upstream, above the Northumberland Dam. Thus, the activities of this marina are unlikely to be adversely affected by the remediation operations.

- Admiral’s Marina is located in Stillwater. Proposed dredging sites are almost a mile from this facility. There is a proposed dredging site situated slightly less than a mile upstream from this marina, and a navigational dredging site about a mile downstream from it. Thus, no adverse effects on the marina are likely to ensue as a result of the proposed dredging.

- Mechanicville Terminal is a public marina in Mechanicville. There is a PCB dredging site located about 4,000 feet upstream, adjacent to Riverside. There is also a navigational dredging site located about 2,500 feet downstream. Thus, no direct adverse impacts on the marina are anticipated.

- Lock 1 Marina is situated by Lock 1 in Waterford. This marina and its activities are unlikely to be affected by the proposed PCB remediation because of its distance (over five miles) from any prospective dredging sites.

- Van Schaick Island Marina in Cohoes is, similar to Lock 1 Marina, unlikely to be affected because all prospective dredging is more than five miles away upstream.

- Troy Town Dock and Marina, again, is unlikely to be influenced by the dredging activities because it is located significantly far downstream of any prospective dredging.

- Troy Motor Boat and Canoe Club is a marina in Lansingburgh. Again, this is unlikely to be affected by dredging activities because all prospective dredging is more than five miles away upstream.

- Albany Yacht Club in Rensselaer is also unlikely to be affected by the PCB cleanup operations, because it is far downstream from any proposed dredging sites.

- Waterford Harbor Visitors Center is a free-access marina in Battery Park in Waterford. This marina is unlikely to be affected by the dredging operations, because it is located more than five miles downstream of the nearest proposed dredging sites.

This review of the identified marinas and their proximity to the proposed dredging sites indicates that only one marina, the West River Road Marina near Fort Edward, which may no longer be in operation, will be proximate to any proposed dredging. The proximity of dredging here, were the marina to remain open, may require the temporary closure of this marina when dredging is undertaken at its entrance. However, the duration of the dredging activities will be relatively brief (one or two weeks) and may be timed to interfere minimally with the peak boating season. As stated, proximity to the proposed NTF operations, including barge unloading, may result in adverse impacts to the marina.

Existing Tourism

ARCC, in its comments, presents an overview of tourism-related activity in the economy of the upper Hudson region, citing Warren, Washington, and Saratoga Counties. The four counties that are actually proximate to the dredging are Albany, Rensselaer, Washington, and Saratoga. It appears rather arbitrary, therefore, to omit two counties and include Warren, which is not directly affected by the selected remedy and whose tourism is more oriented to the Adirondacks and Lake George than it is to the Hudson River. Also important to note is that Saratoga's tourist attractions are much more oriented to I-87 and Saratoga Springs than they are to the Hudson River. Washington County is much more representative, as a whole, of the stretch of river that will be affected (*i.e.*, more rural and with relatively few built tourist amenities).

ARCC cites a US Census Bureau (County Business Patterns) study on the number of hotels in Warren and Saratoga Counties, citing 556 hotels and lodging places in Saratoga County and 755 amusement and recreation establishments (US Census Bureau, 1999). However, in fact, 1999 data from this source cites 52 accommodation establishments in Saratoga County and 62 amusement and recreation establishments. ARCC states equivalent data are not available for Washington County; in fact, equivalent data are available for Washington County and there are no accommodation establishments and 15 amusement and recreation establishments reported (however, the discussion on river-oriented accommodations below identifies one motel in Fort Edward, Washington County, that was open in 1999, implying that the data source may not be completely inclusive).

Analysis of US BLS ES-202 data over 1988-99 using a detailed list of tourism and recreation-oriented activities[2] reveals a more complete perspective on trends in tourism in the region (US BLS, 2001). Comparison of the counties in the Upper Hudson region with others outside the region in upstate New York that have freshwater recreation resources (*i.e.*, Herkimer, Cayuga, and Seneca Counties) provides a reasonable control sample for the counties in the upper Hudson. The data reveal that Washington County's small tourist-oriented employment has declined since 1988, whereas Cayuga County's has grown 63 percent over the period 1988-99. Table 313617-5 shows the changes for each of eight counties. If, for example, this sector had grown in Washington County at the same rate as in Herkimer County (23 percent), it would have added 500 tourism-oriented jobs or, similarly, Albany County would have added 3,266 such jobs.

The census bureau's county business patterns report records employment in the tourist-relevant categories of arts, entertainment, and recreation (North American Industrial Coding System [NAICS] 71) and accommodations and food service (NAICS 72). In 1999, these categories accounted for 7.3 percent of all employment in Washington County; this compares to 11.3 percent in Saratoga, 10.6 in Cayuga, 9.5 percent in Herkimer, and 8.9 percent in Seneca Counties.

With Washington County so far behind the tourism growth of other counties, both in the upper Hudson region and elsewhere in the State, it is quite apparent that the county that typifies the target area for dredging has not shared in this important growth industry. The image of one of its key tourist amenities, the Hudson River, as contaminated with PCBs may well have contributed to this poor performance.

Potential Impacts on Existing Parks, Festivals, and Tourist Accommodations

Parks and Festivals

A number of parks (federal, State, and local) and other recreational attractions are located on or near the waterfront in the area between Fort Edward and the Federal Dam at Troy. This

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[2] The list of SIC code industries was: Fish hatcheries and preserves; hunting & trapping; water transportation of passengers; marinas; other water services; arrangement of passenger transportation; eating and drinking places; real estate operators; real estate managers; real estate developers; hotels/motels; rooming and boarding houses; camps; organized hotels/lodging; auto rental; auto repair; auto services; motion picture theaters/distribution; amusement & recreation; museums & art galleries.

subsection contains a list of these parks, identifying their location and proximity to dredging. Depending upon the distance of each facility from proposed dredging sites, the type of dredging (hydraulic or mechanical), and whether or not the facility is available for use at night, there could be short-term impacts at some locations that EPA will seek to mitigate, if possible. EPA has examined the potential for impacts to the community in various white papers and in responses to master comments throughout this document. These sources should be consulted for further discussion on the likelihood and magnitude of potential impacts.

- *Champlain Canal Scenic Byway*, stretching for 64 miles from Waterford to Whitehall, is a part of the National Scenic Byways program initiated by the US Department of Transportation and Federal Highway Administration. The canal's virtue as a scenic byway is that it provides attractive views to the motorists driving alongside it (for example, on Route 4). Whether or not the passing motorists will continue to be able to enjoy the canal's scenery during dredging depends on a number of factors, including season, location of the dredging equipment, and characteristics of the surrounding landscape that may, for example, partially conceal dredging equipment and facilities. The tract of the canal designated as a scenic byway also features picnic parks and trails at the 11 locks located between Waterford and Whitehall. (Two of the parks by the canal locks, as well as some other parks adjoining the Champlain Canal Byway, are discussed in greater detail below.) Parks, boating facilities, and other amenities of the canal are open May through November; the parks operate from dawn until dusk. The section of the Champlain Canal Byway between Waterford and Fort Edward includes a number of the proposed PCB dredging sites. Dredging will occur at the time of year and during the hours when parks and boating facilities along the Champlain Canal Byway are open. In all, about 11 miles of the 40.9 river miles between Fort Edward and the Federal Dam are targeted for some amount of dredging activity over six years. The first six miles south of Fort Edward is almost entirely targeted. Thereafter, targeting is sporadic, with long reaches of the river unaffected and with many of the areas being targeted only for navigational dredging. Of the seven locks, three (Locks 2, 3, and 5) will require dredging immediately to their north, and Lock 7 in Fort Edward will see dredging 300 feet to its north and south. Locks 1 and 3 (Stillwater) will experience no proximate dredging.

- *Rogers Island Visitors Center* is a historical attraction on Rogers Island in Fort Edward. The center has recently undergone large-scale renovations, including installation of a professionally designed exhibit covering 5,000 years of history of Rogers Island and the surrounding area, from prehistoric time to the Civil War. Archeological artifacts from the region are incorporated into the exhibit. The visitor's center also features a gift shop and is open all week in the summer season. Dredging operations are proposed in the eastern channel for about 2,000 feet alongside the southern section of the island, and in the midsection of its western channel for about 1,000 feet. During the several weeks dredging will be in operation here, enjoyment of this historical site might be at least somewhat impacted. EPA will work with the community to mitigate these potential impacts. The center could also be relatively close to a potential site for the NTF, but because the actual facility has not yet been sited, it is premature to attempt an assessment of potential impacts on the center from processing/transfer operations. Further, if a nearby site were to be selected, design factors may be able to substantially buffer the facility and minimize discernable impacts on the visitors' center.

- *Bradley Beach* is a recreational area along the Hudson River in Fort Edward that offers facilities for picnicking. Bradley Beach is obscured from the river by Rogers Island and is upstream from the nearest proposed dredging sites, which are over 2,000 feet to the south.

- *Fort Edward Yacht Basin* is a public park and dock located in downtown Fort Edward, over 1,000 ft north of proposed dredging. The facility has recently benefited from a $2 million renovation and it now hosts a variety of community events, such as summer concerts and Fort Edward Heritage Days. Shallow depths (three to five feet in the summer) presently inhibit the full potential of this facility. Dredging this channel to the east of Rogers Island will enhance the site's attraction to recreational boating.

- The *French and Indian War historic site* in the vicinity of Fort Edward is a historical attraction that has yet to be constructed along the river. The proposed facility's exact location is still unclear.

- *McIntyre Park, Feeder Canal, and the adjacent bike trail* in Fort Edward are all located near the Champlain Canal, away from the Hudson River.

- *Fort Miller Recreation Area* is situated alongside the Hudson River in the historic village of Fort Miller. The recreation area is still under construction; a ball field, playground equipment, and a picnic area are to be installed there, in addition to already-existing facilities for sports fishing. Dredging is proposed about 1,000 feet south of Lock 6 and about 2,000 feet to the north around a bend in the river.

- *Starks Knob Scientific Reservation* can be defined as an educational resource, an "open-air museum" of natural and historical significance. At this point, Starks Knob Scientific Reservation, located near Northumberland, is undergoing transformations to improve and facilitate public access to the site. Since the site overlooks the Hudson River, certain visual impacts of the dredging activities north of the Northumberland Dam are possible.

- *Fort Hardy Park* in Schuylerville is almost a mile downstream of the nearest proposed dredging site, and adjoins historic Fort Hardy. The park features a beach and a recreation center. The park also hosts some activities of the Schuylerville community, most notably Family Day in January and Scared in Schuylerville (a variety of Halloween festivities) in October.

- Another *waterfront park near Schuylerville* is located at Lock 5. This park features picnicking equipment. The nearest dredging site is north of the Northumberland Dam, almost a mile further upstream.

- *Lock 4 parks* are located in Stillwater. These parks are a mile south of, and around a bend of the river from, the nearest PCB dredging site. A site designated for navigational dredging is located approximately 4,000 feet south of the parks.

- *Saratoga National Historical Park* is the site of the Saratoga Battle of 1777; thus, most of the activities that take place in this four-square-mile park are linked to this event. The majority of the organized events in the park occur by the park's interpretive center, which is about three miles away from the river. A small section of the river in proximity to the park is targeted for navigational dredging, likely to have a duration of about a week or two; no PCB dredging is proposed near the park.

- *Blockhouse Park and Museum* in Stillwater are represented by a Revolutionary War-era timber structure surrounded by the riverfront park, which offers some picnicking facilities. The proposed dredging sites are over a mile from this park.

- There are *two golf courses* across the Hudson River from Mechanicville in Hemstreet Park, one of which borders the river. Sections designated for PCB and navigational dredging are north of Lock 3, about 2,000 feet from the golf courses. The temporary effect of dredging on these golf courses will depend on the visibility of the dredging equipment, the extent of noise and odor, and duration of the operations.

- *Peebles Island State Park* is at the confluence of the Hudson and Mohawk Rivers in the vicinity of Waterford. The visitors to the park can enjoy a variety of activities, such as hiking, jogging, picnicking and fishing in summer, and cross-country skiing in winter. The nearest dredging sites are about five miles upstream from the park.

- *Soldiers and Sailors Park* in Waterford overlooks the Hudson and features the Soldiers and Sailors Monument. Waterford is about five miles downstream of the nearest proposed dredging.

- *Battery Park* in Waterford is also about five miles downstream of the nearest proposed dredging.

- *Button Park, Old Champlain Canal Park and bike trail, Waterford Flight of Locks, RiverSpark Lock 2 Park*, and *Waterford Village Canal Promenade and Docks* in Waterford are also well downstream of the dredging operations.

- *Riverfront Park* in downtown Troy occupies 4.10 acres of land substantially south of any proposed dredging, and is the newest public park in the city. The park was dedicated in 1982, and since then, it has hosted a variety of the community activities and gatherings.

- *Burden Park* along the bank of the Hudson in South Troy opened in 1919. This park occupies the grounds of the old steel works, again substantially downstream of any proposed dredging.

- *Herman Melville Memorial Park* in Lansingburgh is located in the riverfront section of land that was sold by the Troy City Council to the Lansingburgh Historical Society in 1972. Like other parks in Troy and its vicinity, this park is substantially downstream of any proposed dredging.

- *Bikeway 9* is a proposed bike trail to be introduced along the Champlain Canal. No definite information is available at this time as to the trail's length, its exact location, or the date when it will be opened.

A series of events and festivals is associated with the parks and/or the waterway. A number of annual events were identified based on the 2001 calendar. Depending on the location of a particular event in relation to the proposed dredging sites, the type of dredging (mechanical or hydraulic) that occurs at a specific site, and whether the event occurs in the daytime or nighttime, there may be some potential short-term noise, odor and aesthetic impacts. EPA will work with the communities involved to mitigate impacts to the maximum extent practicable. EPA has examined the potential for impacts to the community in various white papers and in responses to master comments throughout this document. These sources should be consulted for further discussion on the likelihood and magnitude of potential impacts.

- *Third Annual Tugboat Roundup* – A tugboat parade from Albany to Waterford, more than five miles south of any area to be dredged (September 7 – 9, 2001).

- *Fourth Annual Canal Cruise and Trek* – Boating and cycling along the canal system (Fort Edward – Waterford, July 11, 2001). This event, sponsored by the Canal Corporation, traverses the study area during one day of the trip from Whitehall to Buffalo. If deemed appropriate, arrangements could be made in advance to avoid any potential conflict with dredging operations.

- *CanalFest* – A festival in Waterford, more than five miles from any proposed dredging site, that includes fireworks display, vendors, and live entertainment (May 12, 2001).

- *Memorial Day Parade in Waterford*, five miles downstream of any proposed dredging site.

- *Christmas Parade in Waterford* – This event is outside the dredging season and substantially downstream of any proposed dredging (December 2, 2001).

- *Family Day* – Skating and games on the canal in Fort Hardy Park in Schuylerville (January 21, 2001). The activities associated with Family Day take place immediately on the canal; however, dredging operations are not scheduled during winter, and thus, the event will not be affected.

- *Victorian Stroll* – A history-themed festival taking place in Troy in December. This event is outside the dredging season and substantially downstream of any proposed dredging.

- *Riverfront Arts Fest, Troy* – Art exhibits and entertainment (Father's Day weekend in June). This riverfront park is located substantially south of where any dredging will occur and will not, therefore, be affected.

- *Carama 2001, Waterfront Park, Troy* – Caribbean carnival (July 11, 2001). As noted above, this park is located south of where any dredging will occur and will not be affected.

- *Waterfront Farmers Market, Hedly Park Place, Troy* – As noted above, this park is located south of where any dredging will occur and will not be affected (every Sunday from June 23 to October 27).

- *Scared in Schuylerville* – Halloween festivities and costume judging in Fort Hardy Park in Schuylerville. This event is almost a mile away from the proposed dredging site in the vicinity of the Northumberland Dam, and thus is unlikely to be affected by the dredging activities (October 28, 2001).

- *Summer Concerts in the Park, Hudson Falls-Kingsbury* – One night per week during July and August. These events are held upstream from where the dredging will occur.

- *Antique Auction and Country Fair, Fort Edward* – Arts and crafts vendors. This event is held alongside the Champlain Canal. It is premature to know whether the timing of this one-day event may coincide with the period when dredging may occur nearby, along the east bank of Rogers Island (July 29, 2001)

- *Summer Concerts in the Yacht Basin, Fort Edward* – Concerts one night per week in August. As with the previous event, it is premature to know whether the timing of this series may coincide with the period when dredging may occur nearby.

- *Fort Edward Heritage Days* – If this is an annual event, as with the previous events noted in Fort Edward, it is premature to know whether the timing may coincide with the period when dredging may occur nearby (July 7 – 8, 2001).

- Saratoga National Historic Park hosts a series of events each year, including:

  Stillwater Heritage Day (October 7, 2001)
  History Hikes (every Sunday in July and August)
  24th Regiment Encampment (August 4 – 5, 2001)
  August Lunch Series (every Tuesday in August)
  18th Century Weekend (August 4 – 5, 2001)
  Colonial Concert (July 29, 2001)
  Happy Birthday NPS! (August 25, 2001)

  Most of these events occur at or near the park's interpretative center some three miles from the river, rather than anywhere close to the river. Moreover, no PCB dredging is proposed in proximity to the park and its waterfront sections, although a small section (less than 2,000 feet) is proposed for navigational dredging in this general area of the river.

Thus, of these identified events and festivals, only the Canal Cruise and events at Fort Edward could potentially be affected by the dredging. Although the brief duration of the dredging at any particular location minimizes potential conflict with any of these annual events, EPA plans to work with the community if conflicts arise to mitigate impacts to the extent practicable.

Lodgings and Accommodations

Following is a list of the hotels, motels, and bed and breakfast inns that have been identified in the communities along the 40-mile section of the Hudson River from Fort Edward to the Federal Dam at Troy and which are located close to where dredging will occur. Included is a brief description of each of these accommodations and their distances from proposed remedial dredging sites. Depending upon the distance from the dredging sites and the type of dredging that occurs there (mechanical or hydraulic), there could be some short-term nighttime noise impacts at some of these locations that will be mitigated as much as practicable. EPA has examined the potential for impacts to the community in various white papers and in responses to master comments throughout this document. These sources should be consulted for further discussion on the likelihood and magnitude of potential impacts.

- Victorian Motel, at 215 Broadway in Fort Edward, is located within 500 feet of the Hudson River at a location where PCB dredging is proposed. It is likely, therefore, that during the one or two weeks when dredging operations will be underway here that business at the motel might be adversely impacted. Possible mitigation of these impacts will be to attempt to schedule dredging to avoid the peak season.

- Inn on Bacon Hill, at 200 Wall Street in Schuylerville, is a bed-and-breakfast inn. This inn is relatively distant from the river (about 4,000 feet away). Additionally, the closest proposed dredging site, which is in the vicinity of the Northumberland Dam, is about a mile away from the Inn on Bacon Hill.

- Marshall House, at 136 Route 4 North in Schuylerville, is a bed-and-breakfast inn located within 200 feet of the Hudson/Champlain Canal at Lock 5. The inn, also about one mile downstream of the proposed dredging site near Northumberland Dam, is situated near a long downstream reach of the river (over five miles) without proposed dredging sites.

- Burgoyne Motor Inn, at 220 Broad Street in Schuylerville, is an 11-room motel. This motel is located within 500 feet of the river. The nearest dredging site is over a mile upstream, in the vicinity of Northumberland Dam.

- Dovegate Inn, at 184 Broad Street in Schuylerville, is a three-room bed-and-breakfast. Dovegate Inn is located within 1,000 feet of the river; the nearest prospective dredging site is over a mile away upstream, in the vicinity of Northumberland Dam.

- Empress Motel, at 177 Broad Street in Schuylerville, features 12 rooms and is located within about 1,000 feet of the river. The nearest prospective dredging site is over a mile away upstream, in the vicinity of Northumberland Dam.

- Lee's Deer Run, at 411 County Road 71 in the Stillwater area, is a bed-and-breakfast inn. This inn is located over a mile away from the river and there are no proposed PCB dredging sites near this section of the river.

- River's Edge B & B, at 90 Wrights Loop near Saratoga National Historical Park in Stillwater, is a bed-and-breakfast inn consisting of two rooms and a guesthouse. As the name implies, the inn is situated at the riverfront; however, there are no PCB dredging sites within several miles of the accommodation. There is a navigational dredging site about a mile downstream, around a bend in the river.

- Wolff's Diner and Motel, on Route 4 near Saratoga National Historical Park in the Stillwater area, is situated within about 1,000 feet of the river. There is a navigational dredging site about 3,000 feet upstream of the motel. Additionally, there is a PCB dredging site about a mile downstream.

- Grace Guest Home, at 122 North Main Street in Mechanicville, is a bed-and-breakfast inn of five rooms. It is located within 200 feet of the river and there is a proposed PCB dredging site about 2,500 feet upstream at Lock 3. There is also a navigational dredging site about 3,500 feet downstream.

- Waterford Inn, the Olde Judge Mansion in Troy, Park Hotel in Green Island, the Schuyler Inn in Menands, and the Super 8 Motel and the Best Western Rensselaer Inn in Troy are situated more than five miles downstream of any proposed dredging sites.

Thus, of the parks, festivals, and accommodations at or near the river, very few are in close proximity to any proposed dredging; most of the potentially affected facilities are located near Fort Edward. Proposed dredging activities are substantial in this area; however, their duration and impacts will be relatively brief, and EPA will work closely with the community specifically to mitigate impacts during that period.

Long-Term Prospects for Tourism in the Hudson Valley

Scenic Hudson comments:

> *"Tourism is a major industry, today. Many sources regard it as the fastest growing industry in the world. The economic potential for tourism stems from amenities and services in combination with image and marketing. Tourism has always been important along the Hudson River Valley… At present, the stigma of Superfund site designation clouds the Hudson River Valley…. Without dredging, the full economic benefits to be reasonably expected from such a world class resource as the Hudson River will never come close to being realized."* Scenic Hudson Comments, April 17, 2001, p. 39-40.

Scenic Hudson's consultant, KLIOS, Inc., in Appendix A of its comments, addresses the economic potential of tourism in the Hudson Valley, noting a variety of literature sources on the value of recreational tourism. Among these:

- Recreational boating on the Ottawa River (Hushak, 2000).

- Recreational boating in Maryland (Lipton, 1995).

- Scenic corridor tourism on the coast of New Hampshire (Robertson, 1997).

- Value of coastal theme festivals on southern Lake Michigan (Wicks, 1992).

- A profile of Columbia River Gorge tourism (Anderson, 1988).

The authors noted, for example, that the value of recreational boating in Maryland exceeded $1 billion in 1993, that Lake Michigan festivals in 1992 grossed revenues of $51 million, and that recreational boating on the Ottawa River added $14 million in sales to the local economy. The authors note that specific studies on the value of tourism in the Hudson Valley are not available.

Tourism involves a wide variety of recreational activities. Among those most relevant for the upper Hudson valley will be outdoor recreation, as compared to visiting museums or movie theaters. A common way of differentiating among outdoor recreation activities is to classify them as “user-oriented” or “resource-based” activities. User-oriented activities, such as team sports, are not usually dependent on any natural resources other than space, whereas resource-based activities, such as bird watching and fishing, depend on the existence and quality of supporting natural resources (fishing is examined as a separate section of this white paper).

A national survey of recreation and the environment conducted in 1994 and 1995 (Outdoor Coalition of America, 1997), a follow-up to a similarly extensive survey conducted 10 years earlier, reports high participation rates for activities relevant to the upper Hudson valley. For the U.S., these include the following annual data:

- Viewing/studying (76.2 percent, or 152.6 million persons).

- Visiting beach/waterside (62.1 percent, or 124.4 million persons)

- Sightseeing (56.6 percent, or 113.4 million persons).

- Freshwater fishing (24.4 percent, or 48.8 million persons).

- Boating (29 percent, or 58.1 million persons).

Those activities that saw major increases in participation over the decade were bird watching (an increase from 21.2 million to 54.1 million persons) and hiking and backpacking (an increase from 33.5 million to 63 million persons).

Another key database on this type of tourist activity is the National Survey of Fishing, Hunting. and Wildlife-Associated Recreation (US Fish and Wildlife Service [FWS], 1998). This source reported 62.9 million people participated in watching wildlife, generating associated expenditures of $29 billion for the US. Of these participants, 23.7 million took trips away from home to participate in this recreation. This same source cites 3.3 million wildlife-watching participants in New York State, of whom 1.173 million were nonresidential. Total expenditures for wildlife watching in New York were almost $1.3 billion, of which trip-related expenditures were $139.7 million, with $1.1 billion for equipment and other expenditures.

It is clear that the upper Hudson River valley ought to be a major participant in these outdoor, nature-oriented modes of recreation (the economic potential of recreational boating and fishing are discussed as separate sections of this white paper). The available data on the economic significance of these activities points to their substantial scale; however, the riverside communities of the four counties near the targeted PCB dredging appear to share much less than would be expected, especially given the world-class resource that the river provides in its own right along this reach of the river, and as a connector to other magnificent resources such as Lake Champlain and the St. Lawrence River, and the Mohawk River/Erie Canal to the Great Lakes. With the remediation of the PCBs, the river will have a much greater likelihood of securing these tourism and recreational benefits.

IMPACTS ON FISHING

Comments from Scenic Hudson and their consultant KLIOS (Appendix A) claim that the dredging would substantially improve the long-term potential for recreational and commercial fishing in the region. Comments from the ARCC assert that resuspension of PCBs from dredging would keep anglers from the region.

Short-Term Impacts

ARCC comments that the National Survey of Fishing, Hunting and Wildlife-Associated Recreation (US FWS, 1996) reported that $3.9 billion was spent in New York State in 1995 and, although the study does not break out expenditures by region, ARCC states: *"Nonetheless, given the interest in hunting, fishing and wildlife watching in the upper Hudson River, the expenditures in this region are no doubt substantial"* (ARCC, p 10). ARCC's position is that dredging would adversely impact this activity (ARCC, p 21).

Commercial Fishing

Commercial fishing was banned in the Hudson River in February 1976; this ban remains in effect with the exception of baitfish, American shad, and Atlantic sturgeon over four feet. Thus, there will be few or no short-term impacts on commercial fishing as a result of the proposed remedial dredging.

Recreational Fishing

After the 1976 ban, catch-and-release sport fishing in the upper Hudson was not reinstated until 1995; advisories against eating Hudson River fish remain in effect. It is, however, the presence of PCBs that has caused the bans/advisories and only remediation of the pollution will return commercial and much of the potential recreational fishing to the river south of Fort Edward.

It is interesting to note that the New York Canal Corporation, which is responsible for the entire New York canal system, including the Erie Canal and the Champlain Canal (of which the upper Hudson is part), markets the waterways as a major tourist/recreational resource, with fishing as one of the key activities (New York Canal Corporation, Recreationway Plan, 1995). Its 2001 season Web page, *Get Hooked on Fishing*, notes:

> *"The most common species caught along the Canal System include small and large mouth bass, walleye, panfish, northern pike, blueback herring and Coho Salmon. One can also find yellow and white perch,*

> *pumpkinseed, channel catfish and crappie at various locations along the system.*
>
> *"Fishing is available almost entirely throughout the 524 miles of water, except on the Champlain Canal south of Fort Edward. Fisherman say that areas outside of locks provide one of the best areas to make a prize catch. In addition, as the gateway to the Great Lakes and rivers of western and upstate New York, the Canal System will lead you to some of the country's best trout and salmon fishing."*

It is the PCB contamination that prevents the Champlain Canal south of Fort Edward from joining this world-class recreation resource. While no reliable data are maintained on existing fishing in this section of the waterways, it is safe to say that recreational fishing is only a faint shadow of what its full potential might be.

If the proposed remediation operations were to inhibit the present limited fishing, as ARCC claims, it will be from one or more of the following:

- The proximity of the dredging barges.
- The resuspension of PCBs.
- The destruction of fish habitat.

In the case of proximity to the dredging, such operations will occupy less than one percent of the 40-mile reach of the river such that, at any particular time, anglers will be able to find alternate sites to fish where the dredging and backfill operations are not proximate.

In the case of the resuspension of PCBs, the threat of contamination will be only marginally greater than at present and will be closely monitored to assure that this remains so; it should not affect catch-and-release fishing.

The destruction of habitat will be temporary and will affect only certain species over the short term. Some species of fish are likely to return sooner than others but ultimately, the dredged waterway is predicted to return to conditions that will support a major recreational fishery.

Further downstream, the effective removal of PCBs will enable the removal or relaxation of bans and advisories on fishing and fish consumption and so restore the Hudson River to its full potential for both recreational and commercial fishing.

Long-Term Impacts

As noted in the introduction to this section on fishing, comments from Scenic Hudson claim that the PCB dredging would substantially improve the long-term potential for recreational and commercial fishing in the region.

Commercial Fishing

Before 1976, commercial fishing was allowed on the Hudson River south of the Federal Dam at Troy. In February 1976, all commercial fishing was banned; this ban remains in effect for several species, including striped bass and American eel. The result of this ban is a diminished commercial fishing industry employing a small fraction of Hudson River commercial fishermen compared to the period before the 1976 ban. Also, the number of commercial fishermen currently employed in other states with important freshwater, estuarine, and marine fisheries is far greater than the number employed in the remaining Hudson River commercial fisheries. KLIOS, Inc. states that commercial and recreational fishing, including striped bass, American eel, and shad, were valued at $40 million annually before they were closed down in 1976 (KLIOS, p 21-22).

Commercial fishing, both estuarine and marine, at other East Coast locations has experienced varying trends over the period since 1988. Massachusetts' fisheries, for example, experienced a decline of over 50 percent in their employment over the period 1988-99, whereas Cape May County, New Jersey, has been more or less stable over the same period and Washington County, Rhode Island, experienced modest growth over the 1990s. In 1999, commercial fishing based in Cape May County supported 204 jobs with wages of $6.2 million, and Washington County supported 65 jobs with annual wages of $2.9 million. These activity levels are small compared to, for example, Bristol County, Massachusetts, which in 1999 supported 857 jobs with wages of $50.7 million (US BLS, 2001). It should also be recognized that many fishing enterprises are sole proprietorships operating on a cash basis and are thus not included in State labor department ES-202 data.

In addition to the direct wages associated with commercial fishing, the activity generates a variety of support, trade, and indirect economic activity that add a substantial multiplier to these direct earnings. US BEA I/O models typically produce employment multipliers in the 2.6 to 2.8 range for commercial fishing. Consequently, in Cape May County, for example, we might expect total direct and indirect employment on the order of 550 jobs. KLIOS estimates that a fully functioning Hudson River fishery would support 450 direct and indirect jobs with annual earnings of $18 million. This is not an unreasonable estimate.

Recreational Fishing

As KLIOS and Scenic Hudson comment: "*The region surrounding the Hudson River does not obtain the full economic benefit that would accrue if restrictions on commercial and recreational fishing were eased or removed*" (Scenic Hudson, p 39).

Recreational fishing along the river is likely to be a more significant economic activity than its commercial counterpart. The National Survey of Fishing, Hunting, and Wildlife-Associated Recreation indicates that there is a population of recreational anglers in the US of 35.2 million, who spend $38 billion per year (US FWS, 1998). Of this population of anglers, 84 percent are freshwater fishermen.

In New York State, the survey reports 1,493,000 anglers who are New York residents (over age 16) and spend an average of 18 days at this activity with average annual expenditures of $942 each. Non-New York residents add to the fishing days in New York by 11 percent, with a total spent in the State (after excluding expenditures by New York residents out of State) of $1.3 billion. Of New York's resident anglers, 996,000 (or 67 percent) fished in freshwater (excluding the Great Lakes), making 13.5 million trips covering 16.2 million days. In 1996, of the $1.3 billion fishing expenditures in New York, trip-related expenditures, including food, lodging, transportation, and boat rentals, came to $601 million; each angler spent an average of $353 on trip-related costs.

In addition to these direct expenditures by anglers, there are secondary economic benefits as these dollars circulate in the local economy, generating additional indirect jobs and earnings. Employment multipliers from service activities in the upper Hudson region, such as hotels and eating/drinking establishments, are on the order of 18-28 jobs per million dollars expended (US BEA, 2001b). Thus, if, for example, the Upper Hudson were to generate a direct increment of $100 million of expenditures in these service industries important to anglers, another 1,800 to 2,800 new jobs will be created.

It is important to recognize that with the bans and advisories in effect, the communities along the Hudson River south of Fort Edward have a limited participation in these huge recreational expenditures, despite some of the most magnificent scenery and fishing opportunities in the State. It is also appropriate to note that the benefits of recreational fishing are hardly limited to economics. The social, physical, psychological, and educational benefits of intimate contact with nature, while intangible, provide significant opportunities for personal renewal and reflection, accounting for much of fishing's broad popularity.

EPA's remedy offers the long-term prospects of a renewed and enhanced recreational fishing industry. This will generate a range of positive benefits that include a substantial boost to local economies and, indirectly, a greater sensitivity to preservation of the natural environment, an intrinsic quality of recreational fishing.

IMPACTS ON PROPERTY VALUES

Comments from ARCC assert the dredging would result in declining property values on both sides of the river and near the sediment processing/transfer facilities. Comments from Scenic Hudson and their consultant KLIOS (Appendix A), citing EPA and other studies, claim that the proximity to Superfund sites generates a negative impact on property values of two to eight percent and declines with distance from the site. However, successful remediation is seen to restore or increase property values, on a situation-dependent basis (*e.g.*, values may exceed pre-contamination levels when use of the site provides for neighborhood enhancements such as parks).

Existing Values

Data on property values along the Hudson River and Champlain Canal were studied in detail in the New York Canal Corporation's study of economic benefits of operation on flood damages (New York Canal Corporation 1990). The corporation collected and refined data on property values for 1,592 residential properties along the river's floodplain in the following municipalities: Village of Waterford, Town of Waterford, Mechanicville, Schaghticoke, Village of Stillwater, Town of Stillwater, Schuylerville, Fort Miller, and Fort Edward. While the purpose of the study was to assess potential flood damages on different types of residences (*e.g.*, with and without basements, second floors, trailers, etc.), it also allows an identification of the average values of the residences near the water. For those in Saratoga County, the average value in 1990 was $61,218; for those in Rensselaer it was $68,331; and for those in Washington County it was $50,406.

The 1990 census records median owner-occupied values for these counties of $107,600, $92,500, and $69,900, respectively for Saratoga, Rensselaer, and Washington Counties. While *average* and *median* values as indicators of centrality may not be identical, the differences between values along the river's floodplain and the rest of the county are very substantial. In Saratoga County those in the floodplain were 43 percent less than the county as a whole; in Rensselaer County 26 percent less; and in Washington County, 28 percent less. Exactly what specific factors account for this variation in values are uncertain, but the potential that the PCB issue was a contributing factor in lowering these values must be considered. Only if these data were reversed, and property along the river was valued more highly, could proximity to the PCB contamination be discounted.

One former property broker in Saratoga County, Michael Burns, who provided testimony at the EPA public hearing in Queensbury in February 2001, stated: "I got used to out-of-towners saying, please don't show us anything near the Hudson River, we don't want to live there." Another broker in Saratoga County noted that despite a very active practice with property elsewhere in the county, she had experienced no demand for buying riverfront property (Merling, 2001). While such views remain anecdotal, they indicate the exact opposite of what would ordinarily be a major property amenity and attraction, capable of commanding a substantial price premium.

Proximity Effects on Property Values

Riverfront Property

As noted in previously, existing property values along the river may have already suffered because of their proximity to PCB contamination. ARCC, however, comments that it is the proposed remedial dredging that would depress property values. The discussions presented earlier in this white paper on the likely scale and operational patterns of the dredging are relevant to this section also.

The dredging scenario presents a remediation effort that will involve a cluster of working barges and their support vessels steadily moving along the river for six years. The pace of the barges will be such that its adjacency to most locations will be limited to only a few weeks. At other times, loaded and empty barges and supply vessels are likely to pass by a few times a day. These are patterns of operation that are reminiscent of the 1970s and earlier, when numerous commercial barges were using the canal and navigational dredging was a regular requirement.

The operational characteristics of the proposed PCB dredging, the brief duration of the dredging at any particular location, and the targeted dredging of only 13 percent of the river bottom on sections that extend for only 27 percent of the river length between Fort Edward and the Federal

Dam at Troy, are highly unlikely to generate any significant or permanent adverse impacts on the adjacent waterfront properties. Over the longer term, after the PCB remediation, owners will enjoy the prospect of substantially enhanced property values. Similarly, owners along the entire Hudson River south of Troy will obtain an increased amenity from the cleanup of the river that could translate into substantial gains in aggregate property value.

Property Proximate to Sediment Processing/Transfer Sites

Description of the Sediment Processing/Transfer Facilities

The remediation program requires a sediment processing/transfer facility site or sites where the dredged material is dewatered and stabilized and from which the stabilized material will be transported by rail to sites situated well beyond the Hudson Valley. These facilities will operate for approximately six years and operations will occur around the clock when dredging is in progress so that a relatively smooth flow of processed sediments will be generated for loading onto rail cars.

Should mechanical dredging be selected as the preferred removal technology, stabilization of sediments will occur in a facility that looks somewhat similar to a concrete batch plant. Incoming sediments will be reclaimed from barges at the site wharf, placed into a receiving hopper, and then conveyed to a pug mill that blends cement (or some other stabilizing agent) into the dredged material stream. Stabilized sediments will then be placed into a small surge or storage area or, alternatively, directly into rail cars. The processing section of the site will encompass two to five acres and the rail yard may require another 10 acres of property.

Should hydraulic dredging be selected as the preferred technology, incoming sediment slurry will be processed through a series of hydrocyclones, where coarse and fine fractions will be separated. The finer-grained material will then be pumped to a processing facility that incorporates several stages of treatment including coagulation, sedimentation, and mechanical dewatering. Overall, the processing site, under the hydraulic dredging scenario, will appear as a medium-sized processing complex supporting a number of tanks, pumps, considerable piping, and a mechanical dewatering plant covering, in total, approximately 10 acres. The additional significant feature on the site will be the rail car loading area.

As discussed in the white papers that address matters such as PCB volatilization and general air quality, no significant hazards are likely to be associated with operations of the sediment processing/transfer facility sites (*e.g.*, White Paper – PCB Releases to Air). The principal contaminant of concern for the Hudson River is PCBs, which bioaccumulate in aquatic ecosystems and then pose a risk to humans as fish consumers. The handling of PCBs at the sediment processing/transfer facility sites will not pose an undue risk to nearby communities since EPA will impose strict operating controls on the contractor and will then monitor site operations to confirm adherence to the project's technical specifications. It is expected that the overall perception of the sediment processing/transfer facility sites will be similar to that of modest industrial complexes that operate for several years and are then recycled for other uses or dismantled.

Other factors such as noise, lights, odor, and traffic will also have a bearing on the manner in which the sediment processing/transfer facilities are perceived and received. These issues are discussed in several white papers and chapters of the Responsiveness Summary. The general conclusions of those documents are as follows:

- *Traffic* – The sediment processing/transfer facilities will generate traffic both during the project's mobilization phase and during the six-year period of dredging operations. However, the expected level of vehicular activity is not expected to generate a significant impact on roadways near the sediment processing/transfer facilities.

- *Odor* – Activities at the sediment processing/transfer facility sites will not be a source of odor to nearby communities.

- *Noise* – Operations at the sediment processing/transfer facilities will generate low, though perceptible, levels of noise in their immediate vicinity.

- *Lighting* – Nighttime operations at the sediment processing/transfer facilities will require lighting for worker safety reasons. It is expected that site lighting can be designed so as to avoid nuisance impacts to nearby residential land uses.

Despite the careful design and selection of sites for these facilities to minimize their potential for adverse impacts, there remains the potential for temporary adverse impacts to property values in close proximity.

The literature of empirical studies on the negative effects of Superfund sites on property values does not examine any facility such as that proposed. Rather, the professional literature typically deals with unremediated sites, where hazardous materials have penetrated aquifers or generate hazardous air emissions. Property value impacts from such sites examined in the literature generally ranges from 2 to 8 percent, with such negative effects declining with distance from the site. A variety of factors appear to influence the level of effect, amongst which a very powerful influence can be negative publicity by the media, in the mode of a "self-fulfilling prophecy." A review of the literature follows.

Literature Review

As noted, the literature does not address a project of the kind proposed here; however, a review of the professional literature on the proximity effects of undesirable land uses such as hazardous sites, landfills, and incinerators illustrates a range of discernable impacts, often with little consistency. However, some basic patterns tend to be that negative impacts decline with distance from the site, values are likely to rebound after remediation or cessation, adverse media publicity can play a marked role in depreciating prices, and other amenities or facilities may play a positive role. A synopsis of the literature follows, first on hazardous materials sites and then on non-hazardous landfills, incinerators, and other industrial facilities. Table 313617-6 presents a matrix indicating the general conclusions of these studies.

Hazardous Waste Sites

The presence of a hazardous waste site may exert a significant negative impact on surrounding property values, as documented by the body of research on the subject (Thayer, Albers, & Rahmatian 1992; Ketkar 1992; Reichert 1997; McClelland, Schulze, & Hurd 1990; Kohlhase 1991; Smolen, Moore, & Conway 1992; Kiel 1995; Greenberg & Hughes 1992; Greenberg & Hughes 1993; McCluskey & Rausser 1999a; McCluskey & Rausser 1999b; Dale, Murdoch, & Waddell 1997).

As a rule, values of the properties in the immediate proximity of a hazardous waste site suffer the most, while the depreciating effect of a site diminishes as distance from the site increases (Smolen, Moore, and Conway, 1992; Reichert, 1997; Kohlhase, 1991). Greenberg and Hughes (1993) note that in the judgment of 28 percent of tax assessors (out of 150 comprising their sample) property values within 0.25 miles of a hazardous waste site suffer as a result of such proximity. However, it must also be noted that 26.7 percent of the participants of the same sample opine that hazardous waste sites in their communities do not damage or otherwise affect property values.

A number of factors associated with the hazardous waste facilities have been noted to influence property depreciation. Clearly, among the most notorious ones are perceptual cues, such as noxious odor emanated by the sites (Kiel 1995). On the other hand, McClelland, Schulze, and Hurd (1990) found no relationship between the odor emitted by a hazardous waste site and property values in the surrounding community. Their findings link property depreciation to *ungrounded* beliefs shared by the community residents concerning health risks posed by the facility in question.

Attempts to gauge a relationship between actual risks posed by a site and the surrounding property values also show inconclusive results. Thayer, Albers, & Rahmatian (1992) indicate a clear correlation between the level of potential danger posed by the site and property depreciation rates. According to this study, property value depreciation near a hazardous waste site constitutes 2.7 percent, as compared to 0.7 percent depreciation for properties near a non-hazardous landfill. However, other studies (Greenberg & Hughes 1992; Kohlhase 1991) suggest that property values do not depend on a site’s toxicity level.

Media coverage of a hazardous waste site may be an important factor in determining property value trends (Reichert, 1997; McCluskey and Rausser, 1999a). However, Dale, Murdoch, Thayer, and Waddell (1997) failed to confirm their findings regarding media coverage.

The impact of the hazardous waste sites on property values tends to fluctuate over time (Kiel 1995; Kohlhase 1991; McCluskey & Rausser 1999b; Dale, Murdoch, Thayer, & Waddell 1997). Kiel, and McCluskey and Rausser documented property value depreciation becoming harsher with the passage of time, pointing to the existence of stigma. On the other hand, Dale, Murdoch, Thayer, and Waddell observed a property value rebound coinciding with the remediation of the site. The results of research by Ketkar (1992) support this conclusion. Namely, her findings suggest that if the number of hazardous waste sites in any given municipality included in her sample (consisting of 64 New Jersey municipalities) was reduced by one, an increase of property values by two percent will be likely to ensue. One study of several Superfund sites in Houston, Texas (Kohlhase, 1991), found that property values rebounded fairly quickly following completion of cleanup activities.

Real estate appraisal theory and practice supports the premise that if the displeasing aspects of a site are corrected, property values are likely to recover. However, a perceived stigma may attach to an area and reduce post-clean-up recovery of property values. Little empirical evidence is available on this issue. Markets generally take time to adjust to new information and time patterns vary from site to site. McCluskey (1999b) hypothesizes that permanent stigma effects are usually related to a change in the demographic composition of neighborhood during the site's discovery, investigation, and remediation, when high-income households move out and are replaced by low-income households.

It is, however, rare that Superfund sites are found in high-income communities; moreover, the price effect (2 to 8 percent) is usually not enough to motivate such demographic recomposition of neighborhoods. Another factor is that some sites are rezoned, permitting less valuable uses to occupy the land and thereby permanently depreciating values. The case of Love Canal in Niagara, New York, is cited where, despite the enormous negative publicity, homes in the remediated neighborhood went from zero value to within 10 to 15 percent of comparable values in the area.

Non-Hazardous Landfills, Incinerators, and Other Industrial Facilities

Non-hazardous landfills may negatively affect surrounding property values, as well. In this case, the magnitude of the impact appears to depend on the amount of activity occurring at the site (Nelson, Genereux, and Genereux, 1992; Cartee, 1989). Reichert, Small, and Mohanty (1992) reported a 5.5 percent negative effect on the real estate values in the surrounding affluent community, attributed to the nearby landfill, which increased to 7.3 percent for the properties within sight of the facility; however, surprisingly, the landfill in their study also appeared to exert a positive impact on properties in other, less affluent communities they investigated. The findings of Nelson, Genereux, and Genereux (1997) indirectly support those of Reichert, Small, and Mohanty. Nelson, Genereux, and Genereux (1997) found that values of more expensive homes suffer more as a result of the proximity to a landfill. A number of other investigators found no statistically significant impact of sanitary landfills on surrounding property values (Zeiss & Atwater 1989; Bleich & Findlay 1991; Cartee 1989). Moreover, Cartee points out that in one case discussed in his literature review, property values appeared to increase due to the presence of a landfill, and in two other cases, since the construction of a landfill, host neighborhoods became more residential.

As another type of a waste treatment facility, incinerators may also influence property values in host communities, as exemplified by Kiel and McClain (1995 and 1996). For instance, according to Kiel and McClain (1995), an operating incinerator reduced property values by 5.13 percent; however, as residents adjusted to the presence of the facility, the negative impact of incinerator on real estate values diminished to 4.19 percent. Zeiss (1990) found a weak negative impact of an incinerator in Oregon on marketability of the surrounding properties; namely, for every kilometer (.625 miles) of distance closer to the facility, a property's time on the market increased by two days. However, as reported by Zeiss and Atwater (1989), the incinerator studied by Zeiss (1990) did not have any apparent effects on property values per se.

Even though industrial facilities such as plants and factories may be hypothesized to exert an impact on surrounding property values, the findings in this area are also mixed. Kiel and McClain (1996), for example, found an 8 percent negative impact on property values attributed to a brick plant, and 4.9 percent effect ascribed to a recycling plant. On the other hand, Bui and Mayer (1999) found no discernible effect of the toxic emissions reported by manufacturers on housing prices in Massachusetts.

Conclusion

By and large, hazardous waste sites do appear to depress surrounding real estate values, albeit to varying degrees that typically range from 2 to 8 percent; such effects may be aggravated if a great deal of media attention is paid to the facility and its potential problems. On the other hand, remediation of hazardous waste sites does appear to moderate stigma and its effect on prices. In general, the influence of non-hazardous landfills and incinerators on property values is more limited in scope. The research regarding the correlation between industrial facilities and property values yields rather inconclusive results.

In light of the operational characteristics of the proposed PCB remediation, with active dredging proximate to any particular location for only a matter of weeks, it is not likely that properties along the river will suffer any significant or permanent loss of value. Rather, the remediation of the PCB problem is much more likely to see a rebound in these properties' value, to levels that are more appropriate for waterfront property in the region. Moreover, property values along the entire Hudson River south of Fort Edward are similarly likely to see some enhancement of value once the remediation has been completed. No attempt is made here to value the enhancement that a river clean of PCBs will generate for property owners along its banks.

For those properties more proximate to the sediment processing/transfer facilities, impacts on their value will be likely to depend on several key factors. Among these will be the proximity of any sensitive receptors (residences, schools, churches, etc.), the degree of any substantive adverse emissions, such as noise, odor, light, dust, or health risks (all of which can be effectively controlled by design and operational programs), or the less tangible effects of negative media publicity and the creation of self-fulfilling prophecies of property value losses.

While the potential for property value losses exists, the extent of these will certainly be ameliorated by careful siting decisions, effective buffering, the location of processing activities within structures, the use of rail and barge for transportation of materials, and other design elements that will be applied. Moreover, the six-year design life of these sediment processing/transfer facilities places their effects within a relatively short-term horizon that will generate less significant impacts on property values and is more likely to see a quick rebound from any potential for adverse impacts.

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