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August 25, 2000

Mr. Douglas Tomchuk
USEPA – Region 2
290 Broadway -19th floor
New York, NY 10007-1866

RE: Environmental Dredging Report

Dear Mr. Tomchuk:

The General Electric Company has made significant efforts to evaluate the true benefits and impacts associated with environmental dredging. We have looked to experience gained by others implementing environmental dredging at other sites across the country. We have shared with you previously a compilation of this information in a comprehensive database of major sediment remediation sites. I am forwarding to you a newly developed report entitled "Environmental Dredging: An Evaluation of its Effectiveness in Controlling Risks". This report was prepared with the assistance of Blasland, Bouck and Lee, Inc. (BBL) and provides a thorough evaluation environmental dredging with a focus on it's ability to achieve meaningful risk reduction. I hope you will consider this valuable information in your analysis of remedial alternatives for the Upper Hudson.

We welcome any comments you may have on this document. Please place a copy of this report into the administrative record. If you have any questions or would like to discuss this further, please do not hesitate to contact me.

Yours truly,

Robert Gibson /bg

Robert G. Gibson

RGG/bg

Enclosure

cc: Mel Haupman - EPA
Alison Hess – EPA
Walt Demmick – DEC
Bill Ports – DEC
Kevin Farrar – DEC
Anders Carlson – DOH
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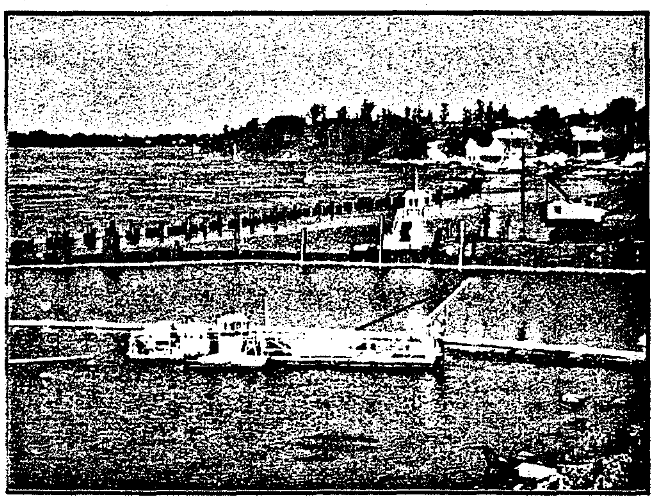
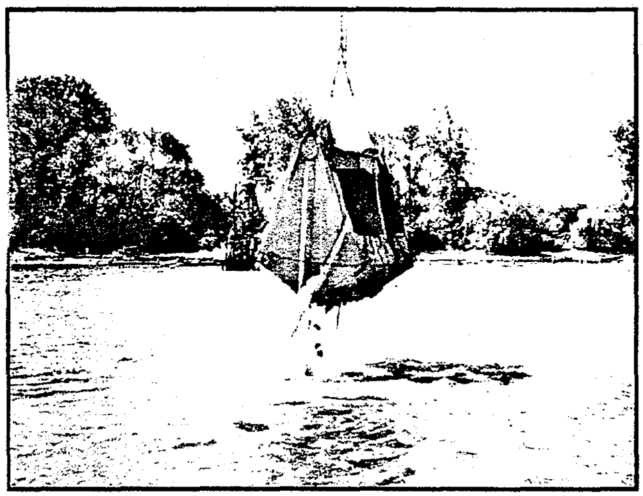
ENVIRONMENTAL DREDGING:

An Evaluation of Its Effectiveness in Controlling Risks





General Electric Company
Corporate Environmental Programs
Albany, New York

Prepared with assistance from:
Blasland, Bouck & Lee, Inc.
Applied Environmental Management, Inc.

August 2000

ENVIRONMENTAL DREDGING:

An Evaluation of Its Effectiveness in Controlling Risks

Introduction

This paper examines the role of environmental dredging in the efforts to reduce risks and protecting human health and the environment from chemicals in sediments. Bioaccumulative chemicals are a particular focus because reduction to levels acceptable to some regulatory agencies requires achieving low residual concentrations in water and sediments in contact with water. Achieving this goal now and in the future is problematic. It warrants careful analysis to determine which portion of the contaminants in sediments is bioavailable and an accurate assessment of the capabilities and limitations of the various remedial technologies, including dredging, to achieve these low levels. Despite increasing reliance upon dredging, to date there has been no systematic evaluation of how effective environmental dredging projects have been in controlling risks from contaminants in sediments. However, a sufficient number of projects have been undertaken that allow such an evaluation to be made, which provides an opportunity to learn what works and what does not.

To that end, this paper reviews major sediment remediation projects undertaken in the United States and summarizes key aspects of these projects, such as the objectives of the sediment remediation projects, the technologies being employed, and the capabilities and limitations of those technologies. Finally, recommendations are provided on needed programmatic change. Supporting documentation and project details are provided in the associated tables and appendices.

The key findings of this paper are:

- Dredging has become the "default" remedy for contaminated sediments.
- The current approach for evaluating the ability of dredging remedies to control risk lacks rigor and is not based on a sound scientific understanding of contaminant dynamics in aquatic systems.
- There has not been a systematic experience-based review of the capabilities and limitations of dredging technology in reducing risks posed by contaminated sediments. Thus, an opportunity exists to apply lessons learned from the current base of experience that can help guide future decision-making.
- Based on an evaluation of projects in the United States, we now have real information on the capabilities and limitations of dredging technology. The data on post-dredging residual contaminant levels in surface sediments, production rates, and costs need to be more rigorously used in the evaluation of dredging technology in sediment remedy decisions.
- While much effort is dedicated to evaluating risk posed by contaminated sediments, there has been no equivalent effort to evaluate risks from implementing remedies. No guidance is available on how to perform such evaluations nor on how to compare the potential benefits of a project to

the impacts. Given the potential impacts to local communities and the aquatic ecosystem, there should be confidence that the risk reduction benefits are real and out-weigh the adverse impacts. In general, risks from site contaminants are often overstated because they are based on conservative assumptions under the guise of the precautionary principle and typically assume unrealistic exposure scenarios for these risks.

The national sediment remediation program needs to incorporate these findings and recognize the technical limitations and inherent disadvantages of dredging. This will require a decisional framework that incorporates the considerations identified and discussed in this paper. It will also require coherent and thorough data collection and analysis. If conditions before and after a remedy are not measured, one cannot tell whether dredging has made conditions better or worse.

Background

Risk to human health and the environment from contaminants in sediments is a concern to both state and federal governments. Approximately 100 of the sites targeted for cleanup under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) involve aquatic-related contamination (NRC, 1997). The U.S. Environmental Protection Agency (EPA) estimates that about 10% of the sediment underlying our waterways, some 1.2 billion cubic yards, is contaminated and may need some form of cleanup or recovery effort (EPA, 1997a).

Dredging, including both wet and dry excavation, for environmental restoration ("environmental dredging") is increasingly used in an attempt to manage the risks posed by contaminated sediments. In contrast, the goal of navigational dredging, which has long been used to maintain waterways for commercial shipping and other maritime purposes, is to remove large volumes of sediments, not to reduce risk.

This paper evaluates current efforts by the government to manage risks from contaminated sediments in waterways, with particular focus on the effectiveness of dredging to control risks to human health and the environment – the method most commonly employed to control those risks. Although government policy states that the goal of sediment remediation is "risk reduction" to protect human health and the environment, this evaluation shows that cleanup decisions rarely contain a clear line of reasoning showing how the selected project will achieve these goals. Further, both government and private parties have failed to assess whether remedial efforts have been successful. Indeed, our review shows no evidence that sediment cleanups performed to date have effectively reduced risks to human health or the environment. Nevertheless, environmental dredging has become the default remedy for contaminated sediments. Most of the decisions appear to be based on the simple, yet largely incorrect, assumption that removing a percentage of the contaminant mass from the sediment will result in a roughly equivalent reduction in risks. This approach is referred to as "mass removal." Our review shows, however, that this approach is substantially flawed. Environmental dredging and the national program that increasingly promotes it, have not produced the risk reduction that is their central goal.

The information underlying this review is taken primarily from the Major Contaminated Sediment Sites (MCSS) Database, which was commissioned by General Electric Company (available at www.hudsonwatch.com). The MCSS database collects, for the first time, available information concerning remedies at the major contaminated sediment sites in the United States and elsewhere. The fact that such information has not been compiled before underscores one of the key points of this paper: in making decisions at contaminated sediment sites, regulatory agencies have evidently failed to examine what has actually been achieved at other sites and have not incorporated that experience into their decisions. This paper offers a review of experiences at several other sites and points to how this experience can be applied to develop a coherent framework for future decision-making based on the goal of effectively reducing risks to human health and the environment.

Understanding the Problem

An accurate understanding of contaminant fate in waterways is essential to devising an effective strategy to reduce risks posed by chemicals in sediment. We begin with a brief overview of how contaminated sediments create potential risks to human health and the environment. This involves two key concepts. First, it is only the contaminants within the biologically active, upper-most layer of the sediment bed that are available for uptake by sediment-dwelling organisms and fish or susceptible to migration downstream. Second, and a direct corollary to the first point, contaminants buried below the bioavailable zone present a risk only if the overlying sediment is subject to significant erosion or other mechanical disturbance, or if groundwater moves the contaminants upward through the sediments, thus creating the possibility that the buried contaminants might make their way to the surface and become bioavailable. Appendix A provides a more detailed review of sediment contaminant dynamics.

Consequently, if a buried chemical mass is stable and is not and will not become available to the water column or biota, the human health and ecological risks at that site will not be reduced by removing that mass. As obvious as this conclusion is, it is frequently overlooked because the greatest mass of contaminants is often found in buried sediments. It is important to remember that most of the contaminants in sediments are the result of waste disposal practices that began 50 to 60 years ago and largely ceased 20 to 25 years ago. The fact that the chemical mass remains buried 25 to 50 years after it entered the sediment is strong evidence that it is associated with stable sediments and is unlikely to migrate to the surficial bioavailable layer in any significant way. This explains why, at many sites, dredging has not been effective in reducing risks. Dredging is effective in removing sediment mass to, for example, clear a clogged navigational channel. However, removing chemicals that are not available to the food chain or the water column does not reduce risks. In fact, removing the surface layers may expose otherwise stable buried sediments with contaminants at higher concentrations, making them bioavailable and thereby increasing risks.

Thus, although targeting sediment deposits with the highest chemical concentration through dredging (mass removal) may intuitively make sense, thorough analysis to test this intuition is critical. When evaluating remedial options, it is necessary to evaluate both the sources of contaminants to the bioavailable surface layers and the capabilities of different technologies to reduce risks posed by contaminated sediments. The analysis begins with the identification of contaminant sources to the bioavailable surface. If the sources are unstable deposits subject to erosion, then the focus should be on finding and remediating these deposits. If the bioavailable surface layer is not receiving contaminants from elsewhere, then methods for accelerating the remediation of the surface layer should occur. If the chemicals in the surficial sediments come from on-shore sources, those sources must be controlled. A particularly important consideration, largely overlooked in previous decisions, is the inability of dredging equipment to achieve low levels of contaminants in the bioavailable surface sediments. Last but not least, one needs to compare the potential benefits from dredging (or any other remedy) against the potential harm to the ecosystem and risks to workers and communities. A large-scale dredging project can have devastating impacts on sensitive ecological habitats, and, like any large construction project, carries with it both significant risks to workers and disruption to local communities.

Only after all of these factors are considered can one make a reasoned, well-informed remedy selection. Unfortunately, our review indicates that regulators are not adequately taking these fundamental considerations into account. The bottom line is that a rigorous analysis of the contaminant source and fate in the aquatic system is required before an effective remedy can be evaluated and selected.

Current Regulatory Approach

Most contaminated sediment sites are subject to one of the federal or state cleanup programs, such as the federal CERCLA, commonly known as "Superfund," the federal Resource Conservation and Recovery Act (RCRA), or comparable state laws. Although differences exist among these laws, they all have the primary goal of ensuring that cleanups manage risks from contaminants so as to protect human health and the environment.

Although risk management is the stated goal of many sediment remedial projects, experience shows that dredging has become the default remedy for managing contaminated sediments, with little apparent consideration given to whether dredging actually reduces risks. The presumption appears to be that the dredging will effectively control risks even though objective analysis is usually not provided to support such a presumption. For example, of the 54 completed projects in the MCSS database (summarized in Table 1), 50 have used dredging or excavation:

Types of Remedies Implemented for 54 Completed Projects

Remedy Implemented	Times Selected
Dredging[1]	26
Wet/Dry excavation	24
Natural recovery/burial[2]	3
Engineered capping[3]	1

1 Includes diver-assisted/hand-held dredging.
2 Three others of the 54 have natural recovery as a component of the overall remedy.
3 Portions of two other sites were capped following removal due to elevated surface sediment concentrations.

For the purposes of this paper, dredging is defined as the underwater removal of sediments using mechanical (e.g., clamshell mounted on a barge) or hydraulic (e.g., cutterhead dredge) means. Diver-assisted dredging, which involves a diver removing sediments using a flexible suction hose connected to a land- or barge-based pump, is included under the dredging category. Wet excavation involves removal of underwater sediments using conventional excavation equipment (e.g., backhoe positioned on a barge or on shore). Dry excavation involves diverting water flow and dewatering the area targeted for removal. Once dewatered, the sediments are removed using conventional excavating equipment (e.g., bulldozers, backhoes).

It is not clear why dredging has become the default remedy at sediment sites because the basis for selecting dredging as the remedy is generally inconsistent and unclear. Table 2 provides a detailed summary of the stated goals, apparent or known basis for decisions, and reported outcomes relative to remedial goals and specific objectives for 25 sites having 10,000 cubic yards or more removed. A review of the MCSS database shows that decisions at sediment sites rarely are based on site-specific, quantitative analysis of risk. Instead, regulators often use default sediment clean up values or seek to remove a large mass of contaminants regardless of whether such approaches will actually reduce risk. The variability and absence of stated goals is symptomatic of the confusion surrounding sediment remediation and the absence of a clear and consistently applied decision-making framework.

Our analysis also shows that the agencies responsible for these decisions and for implementing or overseeing sediment cleanups have not implemented reasonably thorough programs to assess whether cleanup efforts have successfully reduced risks. Several years of high-quality and comparable data before and after remediation are essential to assess the effectiveness of sediment removal in reducing contaminant levels in fish and the associated reductions in contaminant bioavailability, exposure, and risk. An adequate sampling program, database, and evaluation methodology should include the ability to: 1) distinguish the effect of removal from the effects of other processes such as the natural burial, transport, or containment of chemicals, 2) reduce the uncertainties inherent in field sampling of biota, and 3) account for the long biological half-lives of strongly hydrophobic chemicals, such as PCBs, that can delay the response of fish tissue levels to changes in exposure. These important data are simply not available for virtually all of the sediment remediation projects compiled in the MCSS database. Even the relatively limited amount of data that does exist for a subset of projects does not indicate that the projects conducted to date have resulted in an acceptable level of risk control. What is particularly disturbing in light of this are recent claims by EPA regarding the success of dredging projects. In the March 7, 2000 update to an article originally appearing in *Engineering News Record* (Hahnenberg, 1999), it is stated: "Results from recent environmental remediation dredging projects demonstrate significant risk reduction is consistently achieved on environmental projects." (A detailed evaluation of EPA's claims can be found in Appendix C). Quite to the contrary, careful review of the existing data shows that: 1) dredging projects are not being carefully monitored and evaluated with respect to achieving risk reduction goals, and 2) where limited monitoring data are available, risk-reduction goals are not being achieved.

A Proposed Risk-Based Decision Framework

It is evident that a risk-based decision-making framework is needed. Such a framework would build from real-world experience at other sites and from an understanding of how contaminants in sediments have the potential to create risks to humans and the environment. This framework needs to answer the appropriate questions for remedial decision making and must be able to document through measurement whether stated remedial goals are achieved. With these concepts in mind, one can develop a simple and straightforward risk-based framework to guide decision making at sediment sites:

1. *Do chemicals present in bioavailable surface sediments pose an unacceptable risk to human health and the environment?*

2. *Are there active sources that are currently contributing contaminants to the surface sediments in quantities that cause unacceptable risks?* If these sources are not controlled or eliminated they will greatly reduce the likelihood that any remedy directed at contaminants already in the sediments will be successful.

3. *Do the chemicals of concern that are buried below the bioavailable surface sediments have reasonable potential to materially increase contaminant concentrations in the bioavailable surface sediments?* Contaminated sediments that are stable and isolated below the surface sediment and not likely to become exposed during future events, such as flooding, do not warrant active remediation.

4. *If the system and bed are stable, would any active remedial effort (e.g., dredging, capping) materially accelerate natural recovery?* Natural recovery is the benchmark against which remedial options must be measured.

5. *If the answer to 4 is yes, is the accelerated risk reduction outweighed by the potential adverse impacts to human health, the community, and the environment from implementation of the remedy?* Decisions should maximize risk reduction and minimize the negative impacts of remedial technologies on the ecosystem and local communities.

In answering these questions, evaluations of remedial options must be based on a comprehensive scientifically sound analysis:

- Decisions must be based on thorough site assessment that is derived from well-conceived, statistically valid monitoring programs that allow a thorough understanding of chemical sources and fate. Where appropriate, these data should be utilized to construct a quantitative site model that will allow for evaluation of remedial alternatives.

- Decisions must be based on a thorough evaluation of *all* sediment management options. Such evaluations must incorporate experience gained from other sites as to the engineering capabilities and limitations of remedial technologies and fairly evaluate the benefits of natural processes and administrative controls to manage risks.

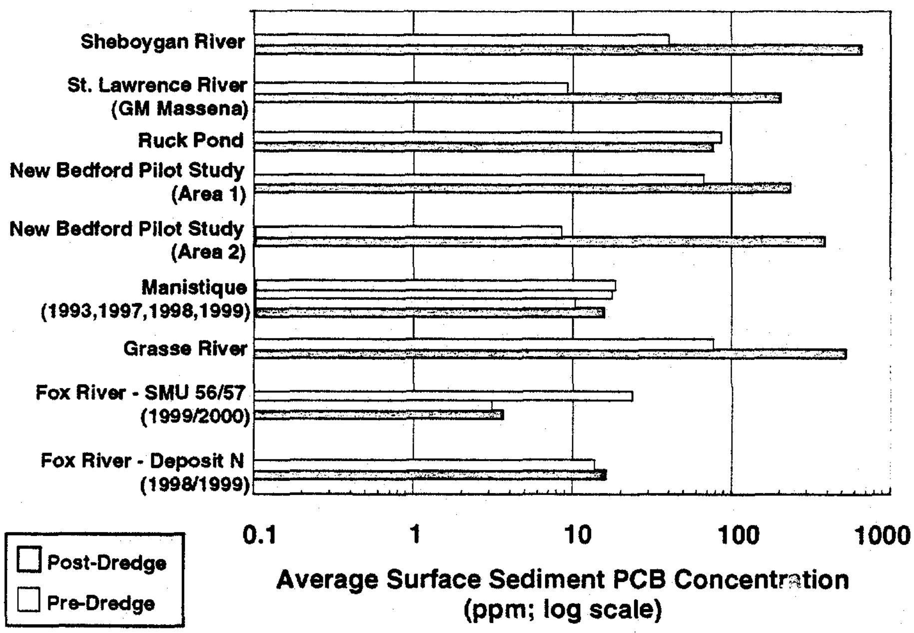
Observations from Environmental Dredging Experience

A review of available information from contaminated sediment sites shows that the environmental dredging projects implemented to date have been relatively small (compared with traditional navigational dredging), costly, and difficult to implement. Moreover, the projects typically have vaguely or inconsistently defined cleanup targets and goals, and their success in achieving risk control has not been documented or demonstrated.

Appendix B provides a summary of results from completed environmental dredging projects that have some post-dredging data available (e.g., contaminant levels in surface sediment, fish, and water). The MCSS database provides additional site information. The primary conclusions drawn from a review of these data are presented below.

1. Environmental dredging has not reduced surface sediment concentrations to acceptable levels.

Cleanup goals and their derivation vary considerably from site to site (i.e., 0.1 ppm to 4,000 ppm for PCBs). However, sediment cleanup goals selected by regulators for bioaccumulative chemicals, such as PCBs, typically are on the order of 1 ppm or less. However, experience has shown that PCB levels of 1 ppm or less have not been consistently achieved through dredging due to the limitations of dredging technologies. Average surface sediment PCB concentrations before and after dredging at several projects are plotted below.



This summary figure shows how dredging has failed to reliably and consistently reduce average surface sediment contaminant levels (PCBs in this case) to typical acceptable levels (i.e., 1 ppm or less). Left at the surface, these contaminants may be available for exposure to biota or movement into the water column.

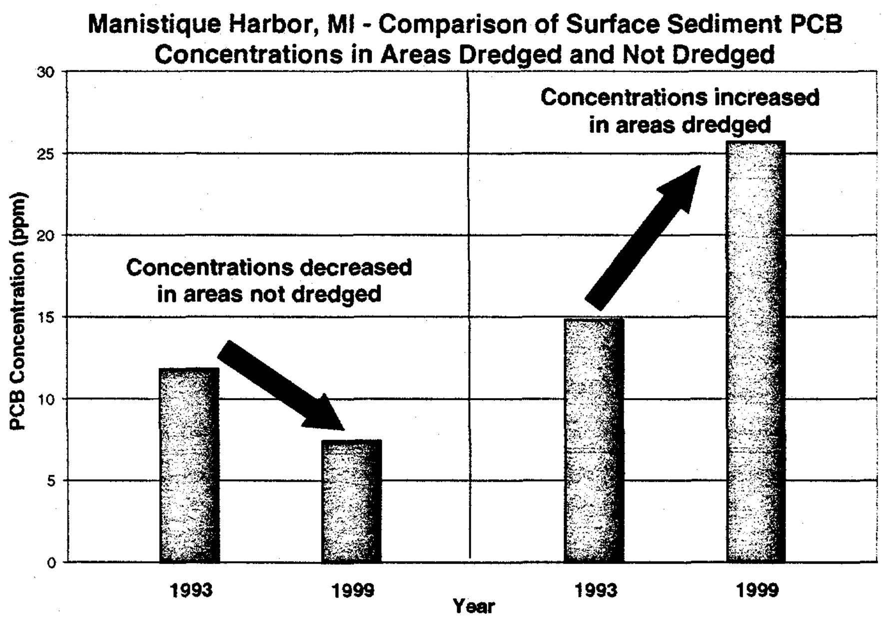
Source: MCSS Database

As can be seen, average PCB levels of 1 ppm or less have not been attained at dredging projects in the United States. At the St. Lawrence River in New York, the 1 ppm cleanup goal was not achieved in all six areas sampled; even though some locations were redredged up to 30 times, the average surface sediment PCB level after dredging was still 9.2 ppm. Similarly, after dredging at the Sheboygan River in Wisconsin and the Grasse River in New York (where the objective was to remove all sediment) average surface sediment PCB levels were 39 ppm and 75 ppm, respectively. At Ruck Pond on Cedar Creek in Wisconsin, the pond was dewatered and excavated "in the dry" in an effort to remove all sediment to the extent practicable. Extensive efforts were employed (e.g., squeegees used on a bulldozer blade, vacuum trucks), yet surface sediment averaged 81 ppm PCBs after removal efforts were finished. Based on the experience to date, it has not been demonstrated that dredging will consistently achieve less than 5 ppm PCBs in surface sediments. The central reasons for these poor results are discussed later in this paper in the section on "Technical Limitations of Environmental Dredging."

2. In some cases, dredging has resulted in increased surface sediment contaminant levels.

As shown in the figures below, dredging at Manistique Harbor in Michigan and the Fox River (SMU 56/57) in Wisconsin resulted in increases in surface sediment contaminant levels. At Manistique Harbor, the increase occurred despite three years of dredging. While the project is apparently not yet complete, it is doubtful that the trend set in motion by dredging (i.e., PCB levels progressively increasing, on average, since 1997) can be reversed by dredging alone. At both sites, conditions before dredging showed lower PCB concentrations at the sediment surface and the highest concentrations were observed in deeper sediment. In essence, dredging has exposed the buried sediments either directly or through sloughing in of the excavation wall, leading to increased surface sediment concentrations.

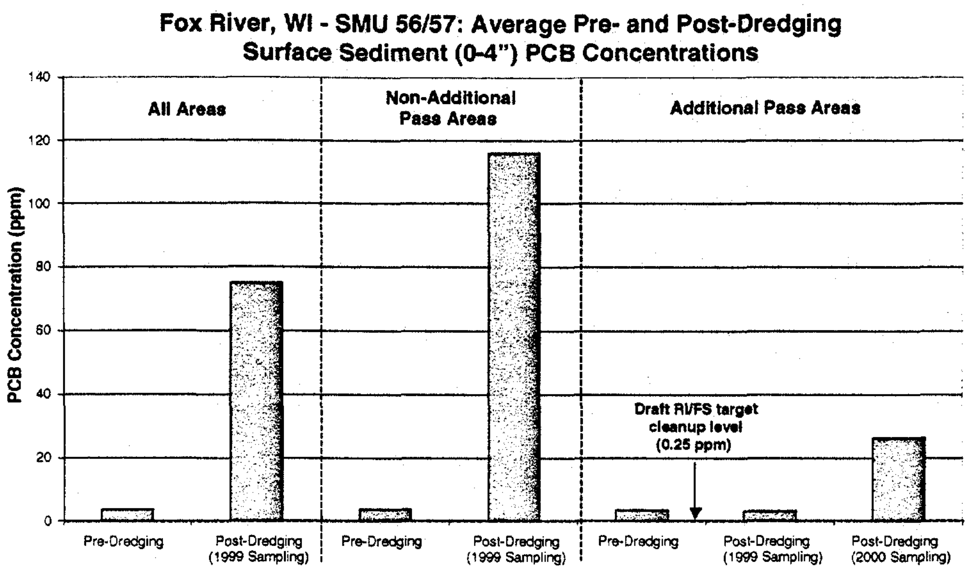
In Manistique Harbor, the average surface sediment PCB levels since 1993 have decreased in areas that have not been dredged, yet increased in areas that were dredged (see figure below and Fox River Group 2000b). This suggests that a natural recovery remedy would have resulted in greater risk reduction than dredging, and that dredging actually has increased potential risks.



At Manistique Harbor, average surface sediment concentrations have declined since 1993 in areas where EPA has not dredged (i.e., data points outside dredging areas), but average concentrations have increased in areas where EPA has dredged since 1997 (i.e., data points within and bordering dredged areas). EPA has returned to the Harbor in 2000 for a fourth season of dredging.

Source:
Fox River Group, 2000b

At the Fox River SMU 56/57 project, executed by the Wisconsin Department of Natural Resources in 1999, average surface sediment PCB concentrations were 3.6 ppm before dredging and 75 ppm after dredging. Due to schedule and budget constraints, only four small subareas were actually dredged "as designed" (i.e., with additional cleanup passes of the dredgehead). Samples obtained shortly after completion of dredging at these subareas showed average surface sediment PCB levels essentially unchanged (i.e., 3.5 ppm before and 3.2 ppm after dredging). However, as shown in the figure below, subsequent sampling conducted two months after completion of dredging (in early 2000) showed 26 ppm as the average surface sediment PCB levels in these areas.



In late 1999, approximately 30,000 cy of PCB-containing sediment were dredged from SMU 56/57 on the lower Fox River. Monitoring data for all areas dredged show that average surface sediment PCB concen-trations rose sharply after dredging. For a short period after dredging in areas where additional passes were used, certain subareas remained at pre-dredging average levels.

Source:
Fox River Group, 2000a

3. Dredging has not been shown to lead to quantifiable reductions in fish contaminant levels.

As noted previously, collection of several years of high-quality and comparable data before and after remediation is critically important to assess the effectiveness of sediment removal in reducing contaminant levels in fish, and the associated reductions in contaminant bioavailability, exposure, and risk. These data are generally not available.

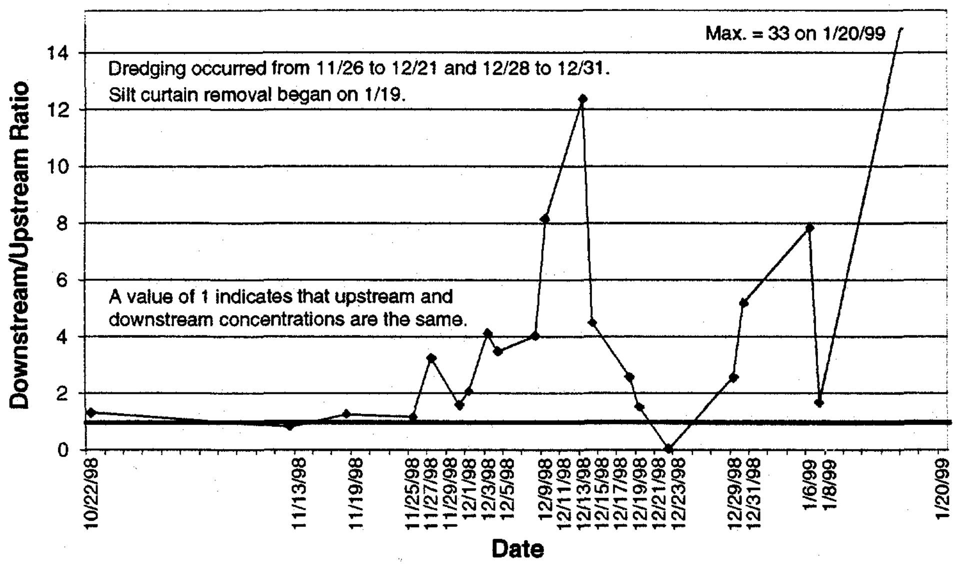
What data do exist are usually inadequate to assess whether dredging has reduced risks from contaminants in sediments. At the Waukegan Harbor site in Illinois, for instance, the pre-remediation fish tissue data consist of one measurement. At the Ruck Pond site, the pre-remediation study included fish cages that were disturbed and one that was lost completely. Pre- and post-dredging data for Ruck Pond are limited to data collected in only one event each. At Waukegan Harbor, where multiple years of carp data are available after dredging, an increasing trend is evident since the harbor was dredged. The uncertainties associated with these minimal monitoring data limit their utility for quantifying, and therefore demonstrating, whether reductions in fish contaminant levels were in fact achieved through dredging.

In addition, at several sites monitoring data collected before dredging indicate that natural processes were already reducing chemical concentrations in fish (e.g., Ruck Pond and Michigan's Shiawassee River), and at some sites other actions such as containment were taken (e.g., Waukegan Harbor, Sheboygan River, St. Lawrence River, Ruck Pond). Distinguishing the effects of these elements on fish levels from dredging is not possible. At the Sheboygan River and Grasse River sites, where several years of fish data are available after dredging, trends in fish levels are not evident in the vicinity of the removal actions. The data do not support the conclusion that dredging reduced fish contaminant concentrations. Appendix C presents additional discussion of this issue.

4. Dredging releases contaminants.

Dredging unavoidably resuspends sediment and releases associated contaminants in the water column. Silt containment systems have been employed at many of the dredging sites in an effort to contain the suspended solids. Although one might think that if suspended solids can be contained, associated contaminants could be as well, this is not always true. Again, there is a paucity of data to evaluate the importance of resuspension and the effectiveness of control. However, there are recent data from projects at the Grasse River and Fox River showing that although silt containment systems generally were effective in containing resuspended solids, increased PCB levels were observed downstream of the dredging (see figure below for Deposit N on the Fox River).

Fox River, WI - Deposit N 1998 Water Column Data: Ratio of Downstream to Upstream Total PCB Concentration



This plot of the ratio between upstream and downstream surface water PCB concentrations in the lower Fox River during the Deposit N project shows that despite the use of silt curtains around the dredging area, PCBs were released to downstream waters during dredging.

Source: Fox River Group, 2000a

In Manistique Harbor, PCB levels in water in the vicinity of the dredging operation were orders of magnitude higher than pre-dredging levels, indicating PCBs were released during dredging (Appendix B).

When released to the water column, the bioavailability of contaminants increases. For example, minnows placed in stationary cages in the Grasse River showed significantly higher PCB uptake during dredging (20 to 50 times higher) and up to six weeks following dredging (2 to 6 times higher) compared with PCB uptake before dredging. These results, combined with the water data, demonstrate increased exposure and potential risks. Given the scarcity of post-dredging data, it is impossible to know how important these releases are in the long term. At a minimum, the release of contaminants will likely delay recovery of the system and therefore must be carefully considered. Further, as project size increases so does project duration, resulting in prolonged impacts.

Contaminants can also be released to the atmosphere during dredging. At the New Bedford Harbor, Massachusetts site, air monitoring documented elevated levels of PCBs downwind of the dredging operation, in some cases exceeding EPA's action level, requiring modifications to the dredge operation.

5. Environmental dredging projects are costly and take a long time to complete.

A common theme observed in evaluating completed projects is that environmental dredging projects generally take longer to complete and cost more than originally anticipated. This is extremely important since cleanup decisions rely heavily on these estimates in weighing and justifying various remedial alternatives. Consequently, *actual* schedule and cost information available from completed projects (see Table 1 and MCSS database) needs to be thoroughly considered when making cleanup decisions. A graphic example of this issue is the Manistique dredging program. In 1995, it was anticipated that the project would take two years to complete at a cost of $15 million. After five years of dredging the harbor and lower river, and expenditures growing beyond $39.2 million, the project is still not complete.

The costs for removal projects cover a wide range as shown in Table 1. Costs are highly variable due to: 1) differences in goals from project to project, 2) differences in production (i.e., removal) rates, which are influenced by a wide variety of site-specific variables such as ease of access, and 3) wide differences in disposal costs, which are influenced by disposal method and location and type of contamination. Average unit costs are summarized below, and a more complete list of factors influencing sediment removal costs is provided in Table 3.

- The average cost for the 22 dredging projects with available volume and cost information is $471 per cubic yard of material removed. The high overall cost is due to two primary factors: low dredging production rates and high costs for disposal. Additional factors that affect the performance of sediment removal are summarized in Table 4. There are a number of uncertainties that also can affect the success of a sediment removal project. Several of the more common uncertainties are also summarized in Table 4, all of which can impact effectiveness, cost, and schedule.

- The average cost for the 19 wet or dry excavation projects with available volume and cost information is $426 per cubic yard of material removed. The high overall cost reflects the low production rates compared with traditional earth-moving projects (using similar equipment) due to difficulties with accessibility and wet terrain, additional water management requirements for maintaining dry conditions, and high costs for disposal.

- Project duration and cost are heavily influenced by the effective production rates of environmental dredging (i.e., how quickly sediment can be removed). While the production rate is influenced by numerous site-specific factors, a review of completed projects shows that typical production rates of only 3,000 to 7,500 cubic yards per month have actually been achieved. These production rates are extremely low in comparison to navigational dredging, and extrapolation to large-scale projects involving hundreds of thousands of cubic yards of sediment indicate that such projects are likely to be decadal in duration.

6. There is limited environmental dredging experience in large rivers.

Almost all of the projects completed to date have covered limited areas and had relatively straightforward access. Of the 26 dredging projects in the MCSS database (i.e., not including wet/dry excavation projects), the largest project was at Bayou Bonfouca and involved only 169,000 cubic yards. In fact, two-thirds of the 26 projects involved removal of 40,000 cubic yards or less. In many of these smaller projects, access and space were available at a responsible party's property in close proximity to the areas to be dredged. This simplifies the implementation by eliminating the need to obtain access to unrelated properties, minimizing transport of sediment, and reducing the schedule and quantities that need to be removed, processed, and disposed of. In fact, projects where access to third-party properties has been required have experienced significant delays in implementation (i.e., Town Branch Creek in Kentucky and the Sheboygan River). For example, barges transporting removed sediment on the Sheboygan River had to travel relatively long distances between the removal areas and the limited number of available land-based access points. Also, shallow water limited the movement of equipment, making the operation inherently slow. In contrast, there is no experience with large-scale environmental dredging projects on extended rivers. With these larger projects, the access, waste management, and disposal issues are likely to be much more problematic. This means that experience at smaller rivers (in terms of ease of implementation) may not apply to larger projects.

7. Advances in dredging technology have been limited.

Specialty dredges, designed to overcome some of the shortcomings of conventional navigational dredges when applied to environmental dredging have their own limitations with respect to remediating large contaminated sediment sites. Japan and the Netherlands have been leaders in developing specialty dredging systems suitable for removing fine-grained contaminated material from harbor and lake bottoms with minimum resuspension. The availability of foreign-made specialty dredges is limited both by law (e.g., the Jones Act) and demand in the United States. Furthermore, their production rates are low compared with production rates of conventional hydraulic dredges. Also, specialty dredges typically have narrow or shrouded dredgehead openings that are particularly susceptible to plugging by debris or vegetation.

Actual production rate data for specialty dredges are sparse, and available data are poorly documented with respect to site conditions and dredge operating parameters. Further, specialty dredges are subject to the same inefficiencies and logistical difficulties as are conventional dredges for environmental dredging.

Of the specialty dredges listed in the table below, the Cable Arm environmental bucket has been used on three major environmental dredging projects in the United States, but it is relatively light-weight, and the absence of "digging" teeth limits its use to unconsolidated (soft) sediments only. In addition, as noted in the table, although minimizing resuspension is an intended feature, actual experience has shown that sediment resuspension with the Cable Arm bucket is still a concern. For the major environmental dredging projects implemented in the United States to date, conventional hydraulic cutterhead and horizontal auger dredges or mechanical clamshells have traditionally been used but with inconsistent results.

Features of Several Specialty Dredges

Dredge Type	Feature
Matchbox Cleanup Refresher	Shielded auger or cutterhead to reduce resuspension
Soli-Flo Versi AgEm	High solids, underwater pump located at dredgehead to shorten suction line and allow passage of large solids/objects
Cable Arm Watertight Dry DREdge	Environmental bucket to maximize percent solids and minimize resuspension upon impact and minimize losses to water column upon removal
Pneuma Oozer	Compressed air piston/cylinder pump to minimize resuspension and maximize percent solids

Technical Limitations of Environmental Dredging

Several technical limitations are inherent in environmental dredging. These limitations restrict the effectiveness of sediment removal in reducing contaminant levels in surface sediments. Although dredging can remove significant volumes of sediment and associated contaminant mass, dredging inevitably leaves behind residual materials at the sediment surface. These residuals are attributed to "missing," "mixing," and "messing," which are described below. In addition, dredging introduces new risks to the ecosystem and community.

Missing: Dredging cannot remove all targeted sediment and contaminants.

Even with careful operations, experience has shown that sediments are unavoidably left behind after dredging. According to the Army Corps of Engineers, "No existing dredge type is capable of dredging a thin surficial layer of contaminated material without leaving behind a portion of that layer and/or mixing a portion of the surficial layer with underlying clean sediment" (Palermo, 1991). Because surface sediments play a central role in transferring contaminants to fish and the wider food web, any action that leaves contaminants at the biologically-active sediment surface is unlikely to achieve risk-based goals requiring low part-per-million concentrations of chemicals.

Dredging's inability to reliably remove all sediments and contaminants and create a clean sediment surface results from various factors, including: 1) incomplete spatial coverage in dredged areas as evidenced by cratering of the sediment bed from the action of a mechanical clamshell or creation of windrows and furrows between swaths of a hydraulic dredge; 2) inaccessibility of sediments located in shallow waters where barges and hydraulic dredging equipment cannot operate effectively, located adjacent to or under boulders and debris that cannot be removed, or resting on an irregular hardpan or bedrock bottom; and 3) performing work underwater and out of sight of the operator.

Mixing: Dredging unavoidably mixes sediment targeted for removal with underlying materials.

To remove sediments, a dredge must cut into the sediment bed, which mixes sediments targeted for removal with other sediments either above or below the targeted material. Whether higher-concentration sediments are present at depth and cleaner sediments are present at the surface, or vice versa, the mixing caused by dredging inevitably leaves behind contaminated sediment on the new sediment surface created by the dredge. Many sediment sites have lower concentrations of the target chemical in surface sediments than at depth. This is often due to previous implementation of source controls and ongoing natural recovery through sedimentation and burial. Thus, dredging mixes the lower concentration surficial sediments with deeper, higher-concentration sediments, which can result in elevated residual concentrations at the new sediment surface. This is particularly problematic at sites with stable sediments because dredging does what nature cannot, bringing contaminants once sequestered in deep sediments to the surface and exposing them to biota and the water column. It also is problematic at sites where deeper, more contaminated sediment rests on bedrock because one cannot overcut into cleaner sediments beneath the contaminated layers. For example, this underlying bedrock condition exists at the Manistique Harbor site.

Messing: Dredging resuspends and releases contaminants into the water column.

The physical mixing action of the dredge inevitably stirs up sediments, releasing both suspended and dissolved contaminants to the water column. Although there are devices to reduce resuspension and the dredge operator can modify certain operating parameters such as production rate, no dredging method has totally eliminated local sediment resuspension. Sediment resuspended during dredging will eventually settle on the surficial layer of the area dredged or be transported and redeposited outside or downstream of the removal area. Thus, for contaminants with an affinity for binding to sediments, surface sediments both within and outside the removal area may become more contaminated than before dredging.

The transport of suspended sediments outside the removal area along with increased turbidity can cause a variety of adverse effects in fish, including interference with gill function, enhanced fungal infections of fish embryos, and reduced resistance to disease. In addition, certain chemicals that may be acutely toxic to local biota (e.g., metals, ammonia) may be released during dredging or result in anoxic conditions. Other chemicals released when the sediment bed is disturbed (e.g., nitrogen compounds, phosphorous) may degrade water quality by stimulating algal blooms.

To reduce the negative impacts of downstream sediment transport, environmental dredging areas are typically isolated from the rest of the waterway by a silt curtain or other containment barrier. These systems do not effectively control the transport of dissolved contaminants, and experience shows contaminants (especially in dissolved-phase) typically migrate outside the containment system and downstream (see examples in Appendix B). Once contaminants are dissolved in the water, they also are more apt to volatilize into the atmosphere.[1] Further, the more effective the barrier system is in containing resuspended sediment, the more contaminated sediment will resettle within the removal area. If sediments

[1]This situation was encountered at the New Bedford Harbor site where, according to EPA (1997b), "control of airborne PCB emissions did contribute to a slower rate of dredging and thus a longer project duration."

migrate outside the removal area, they can resettle over a larger surface area.[2] Chemicals in this resettled/residual sediment will be bioavailable, and the sediments will generally be more susceptible to scour than the pre-existing surface sediment since any natural armoring that may have occurred over time is removed during the dredging operation.

The impacts of resuspension are generally considered a short-term effect of dredging since most environmental dredging projects performed to date have been of limited duration. However, for large-scale, long-term dredging projects, the cumulative effect of these "short term" impacts could be substantial and must be considered in remedial decision-making.

Dredging introduces new risks to the ecosystem and community.

In 1995, EPA posed the question, "How can dredging affect the environment?" The Agency's response was that "impacts can include benthic disturbance, water quality degradation, impacts on aquatic organisms, and water and soil contamination from disposal of dredged materials" (EPA, 1995). EPA was right. Environmental dredging operations bring with them a myriad of risks and impacts not directly related to what is happening at the sediment surface. For example, dredging can destroy important ecological features of a site, such as vegetation, the benthic environment, and various fish spawning and nursery habitats, not to mention the communities of biota that inhabit the removal areas. Although some reconstruction of habitat can be attempted, impacts are typically observed until recolonization occurs, which may take years. As observed by Suter (1997), "the ecological risks related to remedial activity must be balanced against risks associated with the contaminant to the ecosystem components and against often hypothetical health risks." Unfortunately, these impacts are seldom evaluated with any rigor on environmental dredging projects despite the fact that they are carefully analyzed on proposals for navigational dredging projects.

In addition, environmental dredging operations, on-shore sediment handling and processing equipment (e.g., dewatering, treatment), and transportation of materials (via pipeline. barging, conveyance, trucking) to treatment or disposal facilities are inherently dangerous processes. Environmental dredging operations invariably cause normal commercial shipping and recreational boating near a site to become more hazardous and difficult or restricted. Indeed, large-scale environmental dredging projects could take decades and severely impair portions or all of a waterway during active operations. Such disruptions can have devastating economic impacts on a local community's use of the waterway for tourism or other commercial purposes. Again, the impacts from these types of projects in terms of injuries to workers and community members are real, not hypothetical.

As part of the planning process for all types of dredging projects, the Army Corps of Engineers evaluates the potentially detrimental effects of dredging on habitat to ascertain whether dredging must be confined to specific time periods to minimize its adverse environmental impacts. The most persistent concerns are: 1) disruption of avian nesting activities and destruction of bird habitat, 2) sedimentation and turbidity issues involving fish and shellfish spawning, 3) disruption of anadromous fish migrations, 4) entrainment

[2] Studies of the Yazoo and Yalobusha Rivers in Mississippi indicated that turbidity plumes extended up to one-half mile downstream of dredging activities, even when containment measures were utilized (Wallace, 1992). Similar evidence was noted at the New Bedford Harbor site as discussed in Appendix B.

of juvenile and larval fishes, 5) burial and physical destruction of protected plants, and 6) disruption of recreational activities (Dickerson et al., 1998). It is sensible and prudent to consider and weigh the potential damage to habitat and disruption to ecosystem structure and functioning against whatever environmental benefits might accrue from removal of contaminated sediments.

Clear guidance is needed on the evaluation of actual risks to ecological resources and communities resulting from implementation of environmental dredging projects and how to balance these risks and impacts relative to any benefits achieved in risk reduction. Currently, detailed guidance does not exist on how to evaluate objectively and quantitatively the negative consequences of sediment remediation projects.

Final Observations and Recommendations

Dredging has historically been used to remove bulk sediments from shipping channels and harbors. It is effective for that purpose. Dredging to reduce risks posed by contaminated sediments is relatively new, and its effectiveness has not been demonstrated. When viewed in the context of risk reduction, there is no sound justification for dredging stable, isolated sediments that contain contaminants that are not and will not migrate to the bioavailable surface sediment layer in any meaningful way. Decision makers often have not recognized the technical limitations of dredging and its potential for adverse ecological and community impacts. If this does not change, the contaminated sediment program will fall short of its goal of effectively reducing risks to human health and the environment. A number of conclusions can be drawn based upon our review of sediment remediation projects undertaken in the United States.

- There is no consistent framework for making cleanup decisions at contaminated sediment sites. The goal of any program should be to effectively control risks. There is a need for a clear, simple-to-apply, risk reduction decision framework. This paper proposes such a framework, which is based on an understanding of sediment dynamics using sound scientific principles.

- Appropriate data-collection programs to acquire the data necessary to measure the effectiveness of remedial techniques in adequately reducing risks at sediment sites have not been developed. As a result, substantial experience cannot be properly incorporated into remedial decisions. This paper and the MCSS database should help fill this gap.

- The limited available data clearly show the limitations of environmental dredging technology:

  - Dredging has not reliably and consistently removed all sediment, restored a "clean enough" sediment surface, or decreased the bioavailability of contaminants. Dredging is unable to reliably and consistently achieve low residual concentrations typically sought in surface sediments, even after repeated passes with the dredging equipment. The residuals left behind after dredging may be at a higher concentration and more bioavailable than before dredging, resulting in increased risk.

  - While environmental dredging typically employs controls to prevent resuspension and release of contaminants during operations, such releases to water, biota, and air occur.

These releases can create unacceptable long-term risks due to redeposition of resuspended sediment and are particularly problematic at large projects, where such releases may occur over a multi-year implementation period.

- Dredging removes material that must then be handled and processed, typically on shore. This can increase the complexity of remediation. Dredging is inherently dangerous, a fact verified by insurance statistics and poses serious short-term risks to workers and the community, and long-term risks to the extent the material must be permanently managed in a disposal facility. Dredging will disrupt or destroy the habitat and biota in the areas in which it is applied. These very real impacts and risks imposed by the remedy need to be balanced against the hypothetical risks posed by the sediment itself.

- Environmental dredging projects are costly and take a long time to complete.

Decision makers should select remedial alternatives that are protective, technically feasible, and cost-effective. Other options can be more effective than dredging with fewer negative impacts. Based on the evidence presented in this paper and supporting documents, we offer the following recommendations regarding how environmental dredging should be viewed in managing risk:

- Regulators need to reaffirm that risk reduction is the proper goal of any remedial action.

- How contaminants move in the aquatic system must be evaluated during risk analysis and remedy selection. Risk reduction in aquatic systems is directly linked to a remedy's ability to decrease the probability that fish and other biota are actually or potentially exposed to sediment-bound contaminants. The first step is to control or eliminate active sources of contaminants to the surficial bioavailable sediments. The second step is to evaluate sediment deposit stability to assess whether normal erosion or some extreme events (e.g., high flows, flooding) could mobilize otherwise isolated contaminants being currently buried, thus moving non-bioavailable chemicals into the surface sediment layer. The final step is to evaluate methods to reduce surface concentrations of the contaminants now and in the future so as to minimize their bioavailability. Fair consideration must be given to less disruptive risk controls like natural recovery and administrative controls (e.g., fish consumption advisories).

- Regulators must recognize the technical limitations of dredging that result in the inability of dredging to reliably and consistently achieve low residual contaminant concentrations in surface sediments. They must consider the new and potentially higher risks that might occur from increases in contaminant concentrations in surface sediment, the water column, and ultimately fish tissue concentrations.

- Regulators must consider the real environmental and human impacts of environmental dredging projects. These impacts must be weighed against any hypothetical reduction in risk that might be achieved. Comprehensive policy and guidance in this area are needed.

- The experience at completed projects needs to be considered in making future decisions. Adequate monitoring data and formal plans for pre- and post-remediation evaluation of risk reduction are essential elements in sediment remediation projects. These types of essential data can reduce uncertainty and allow one to draw sound conclusions regarding the relative effectiveness of remedial activities.

- Regulators must thoroughly consider *actual* schedule and cost information available from completed projects and incorporate this into their decisions. Experience shows that projects completed to date generally have taken longer to complete and cost more than originally anticipated.

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Tables

Table 1
Summary of Remediated Contaminated Sediment Sites

Project	USEPA Region	Setting	Contaminant Of Concern	Methods of Remediation and Disposal	Volume Removed (cy)	Total Cost (millions)	Total Unit Cost ($/cy)
Baird & McGuire, MA	I	3-mile sector of the Cochato River and several tributaries	As, DDT, chlordane, PAHs	Dry/wet excavation; on-site incineration[1]; natural recovery	4,712	$0.9[1]	$191
Bayou Bonfouca, LA	VI	Turning basin and 4,000 ft. of bayou	PAHs	Mechanical dredging; on-site incineration	169,000	$115	$680
Black River, OH	V	Two hotspots totaling 8 acres	PAHs	Hydraulic dredging and mechanical dredging; on-site landfill	60,000	$5	$83
Bryant Mill Pond, MI	V	22-acre 2,500 ft. long Bryant Mill Pond area of Portage Creek	PCBs	Dry/wet excavation; on-site former dewatering lagoons	165,000	$7.5[4]	$45
Cherry Farm, NY	II	Approximately 1,600 ft. of shoreline (full length of site) extending about 150 ft. into river	PAHs	Hydraulic dredging; on-site existing disposal pond	42,445	$2.2	$52
Convair Lagoon, CA	IX	10-acre embayment	PCBs	Engineered three-layer cap over 5.7 acres	N/A	$2.75[3]	N/A
DuPont Newport Plant, DE	III	1.5-mile sector of the Christiana River	metals (Pb, Cd, Zn); solvents	Mechanical dredging; on-site landfill disposal	10,000	Not available	—
Duwamish Waterway, WA	X	Slip	PCBs	Divers (hand-held dredging techniques); pneumatic dredging; off-site disposal ponds	10,000	Not available	—
Eagle (West) Harbor, WA	X	Puget Sound Embayment comprising about 200 acres of West Harbor	mercury, PAHs	Mechanical dredging, wet excavation, thin-layer capping, and enhanced natural recovery; nearshore CDF, commercial landfill, and *in situ* capping	3,000	$3	$1,000
Ford Outfall, MI	V	2.6 acre nearshore area (about 750 ft. long by 150 ft. wide)	PCBs	Mechanical dredging; on-site landfill	28,500	$5.65	$198
Formosa Plastics, TX	VI	1.1 acres (about 150 ft. by 350 ft.) in corner of an active turning basin	EDC	Mechanical dredging; commercial landfill	7,500	$1.4	$187
Fox River, WI (SMU 56/57)	V	9-acre depositional area in river	PCBs	Hydraulic dredging; commercial landfill	31,000	$9	$290
Fox River, WI (Deposit N)	V	approximate 3-acre depositional area	PCBs	Hydraulic dredging; commercial landfill	8,175	$4.3	$525
Gill Creek, NY (DuPont)	II	250-ft. sector of Gill Creek near its confluence with Niagara River	PCBs, PAHs	Dry/wet excavation; commercial landfill	8,020	$12[3]	$1,496
Gill Creek, NY (Olin Industrial Welding Site)	II	About 1,800 ft. in length of Gill Creek bed	BHCs, PAHs, mercury	Dry/wet excavation; use as on-site fill material	6,850	not available	—
GM (Massena), NY	II	11-acre, 2,500 ft. long nearshore area in the St. Lawrence River	PCBs	Hydraulic dredging, wet excavation, and capping; commercial landfill[1]	13,250	$10[1]	$755
Gould (Portland), OR	X	3.1-acre East Doane Lake remnant, a shallow impoundment	PAHs	Hydraulic dredging; on-site landfill	11,000	$3	$273
Grasse River, NY	II	1-acre nearshore hot spot in river	PCBs	Hydraulic dredging, wet excavation, and diver-assisted; on-site landfill	3,000	$4.9	$1,633
Hooker (102nd Street), NY	II	25 acres in an embayment in the Niagara River	VOCs, metals	Dry/wet excavation; on-site landfill	28,500	not available	—
Housatonic River, MA	I	550-foot sector of the river	PCBs	Dry/wet excavation; commercial landfill	6,000 sediment and banks	$4.5	$750
James River, VA	III	81-mile long estuary; 0.6 to seven miles in width	Kepone	In situ; natural recovery	N/A	N/A	N/A

11.1402

Table 1
Summary of Remediated Contaminated Sediment Sites

Project	USEPA Region	Setting	Contaminant Of Concern	Methods of Remediation and Disposal	Volume Removed (cy)	Total Cost (millions)	Total Unit Cost ($/cy)
Lake Jarnsjon, Sweden	N/A	62-acre lake (bank-to-bank removal)	PCBs	Hydraulic dredging; on-site dedicated landfill	196,000 cy	not available	$33
Lavaca Bay, TX	VI	One deep and one shallow bay area comprising about 7 acres	mercury	Hydraulic dredging; on-site existing disposal ponds	80,000	not available	—
LCP Chemical, GA	IV	13-acre tidally-influenced marsh area; one-half mile of an outfall channel; a separate natural drainage channel	PCBs; mercury	Wet excavation; bucket-ladder dredge; commercial landfill	25,000	$10	$400
Lipari Landfill, NJ	II	18 acres of Alcyon Lake; 5 acres of Chestnut Branch Marsh; Chestnut Branch Stream	multiple organics, inorganics	Dry/wet excavation; some thermal desorption and beneficial reuse; some stabilization and placement	163,500	$50	$306
Loring AFB, ME	I	>2,500 ft. long Flightline Drainage Ditch; 15-acre Flightline Drainage Ditch Wetland (about 2,000 ft. by 400 ft.); >2,500 ft. long East Branch Greenlaw Brook	PCBs, PAHs	Dry/wet excavation; on-site landfill	162,000	$13.85	$85
Love Canal, NY	II	About 10,000 linear ft. of Black and Bergholtz Creeks	TCDD	Dry/wet excavation; commercial incineration[1]	31,000	$14[1]	$452
LTV Steel, IN	V	3,500 ft. of intake flume (width ranges from 96-467 ft.)	PAHs, oils	Hydraulic dredging and diver-assisted removal; commercial landfill	109,000	$12	$115
Mallinckrodt Baker, NJ (formerly J.T. Baker)	II	Nearshore hotspot (about one-half acre) in the Delaware River	DDT	Dry/wet excavation; on-site landfill	3,750[2]	$1.2	$320
Manistique River, MI	V	One 2-acre hot spot in dead-end and back water area; two other hot spots: one of 2 acres in the river and one of 15 acres in the 97-acre harbor	PCBs	Hydraulic dredging; commercial landfill	97,050	$35.9	$370
Marathon Battery, NY	II	200 acres of open cove and a small cove in the Lower Hudson River	cadmium	Hydraulic dredging and mechanical dredging; natural recovery; commercial landfill	77,200	$10[3]	$130
Marathon Battery, NY[5]	II	340 acres of backwater marshes and sheltered cove	cadmium	Dry/wet excavation; commercial landfill	23,000	not available	—
National Zinc, OK	VI	5,300 ft. of the north tributary (unnamed) of Eliza Creek	PCBs	Dry/wet excavation; commercial landfill	6,000	not available	—
Natural Gas Compressor Station, MS	IV	2-mile length of Little Conehoma Creek	PCBs	Dry excavation; commercial landfill	75,000 (includes floodplain soils)	not available	—
New Bedford Harbor, MA	I	Five acres of hot spots in the estuary	PCBs	Hydraulic dredging; commercial landfill[1]	14,000	$20.1[1]	$1,436
Newburgh Lake, MI	V	105-acre man-made lake	PCBs	Dry/wet excavation; commercial landfill	588,000	$11.8	$20
N. Hollywood Dump, TN	IV	40-acre man-made lake adjacent to the Wolf River	pesticides	Hydraulic dredging; on-site burial in an isolated oxbow	40,000	$2.4	$60
Ottawa River (Unnamed Trib.), OH	V	Unnamed tributary about 975 ft. long and 90 ft. wide at its mouth, and tapering to 10 ft. wide at its origin	PCBs	Dry/wet excavation; commercial landfill	9,692	$5	$516

11.1403

Table 1
Summary of Remediated Contaminated Sediment Sites

Project	USEPA Region	Setting	Contaminant Of Concern	Methods of Remediation and Disposal	Volume Removed (cy)	Total Cost (millions)	Total Unit Cost ($/cy)
Pettit Creek Flume, NY	II	One-acre cove in the Durez Inlet of the Little Niagara River	DNAPLs (VOCs and semi-volatiles)	Diver-assisted dredging; portion to commercial hazardous waste landfill	2,000	not available	—
Pioneer Lake, OH	V	200 ft. x 240 ft. (depth: 0.5 to 3 ft.) area of southern lake	PAHs	Hydraulic dredging; commercial landfill	11,100	$2.5	$225
Queensbury NMPC, NY	II	An area of the Hudson River extending 180 ft. offshore and 800 ft. downstream from site	PCBs	Dry/wet excavation; commercial landfill	4,750[2]	$3.5	$737
Ruck Pond, WI	V	800-1,000 ft. long by 75-100 ft. wide impoundment in Cedar Creek	PCBs	Dry/wet excavation; commercial landfill	7,730	$7.5	$970
Sangamo-Weston, SC	IV	7-mile sector of Twelvemile Creek and 730 acres of Lake Hartwell	PCBs	In situ; enhanced sedimentation and natural recovery	N/A	N/A	N/A
Selby Slag, CA	IX	Nearshore area of about 17 acres (fronting on 61.5 acres of shoreline and extending into the water about 280 ft.)	lead	Mechanical dredging; on-site disposal as fill	101,000[2]	$2.1	$21
Sheboygan River, WI	V	17 small hot spot areas in the upper 3.2 miles of river immediately downstream of the PRP site	PCBs	Mechanical dredging, wet excavation, and capping; on-site storage (temporary)	3,800	$7[1]	$1,842
Shiawassee River, MI	V	A 1.5 mile stretch of the South Branch of the Shiawassee River	PCBs	Dry/wet excavation; commercial landfill	1,805	$1.3	$720
Starkweather Creek, WI	V	About 1 mile upstream of the confluence of the east and west branches of Starkweather Creek	mercury (primary); also lead, zinc, cadmium, and oil and grease	Dry excavation; on-site disposal in former dewatering lagoons	15,000	$1.0	$67
Tennessee Products, TN	IV	2.5-mile sector of the Chattanooga Creek	coal tar	Dry/wet excavation; off-site fuel source and commercial landfill	24,100	$12	$498
Town Branch Creek, KY	IV	3.5-mile sector of the Town Branch Creek	PCBs	Dry/wet excavation; commercial landfill	17,000 (sediment and banks); 76,000 (floodplains)	$11	$118
Triana/Tennessee River, AL	IV	11-mile stretch of two tributaries of the Tennessee River	DDT	Rechannelization and *in-situ* burial	N/A	$30	N/A
United Heckathorn, CA	IX	Lauritzen Channel ~1,600 ft long by 200 ft wide; Parr Canal about 1,000 ft. long by 70 ft. wide	DDT	Mechanical dredging; commercial landfill	108,000	$7.5[4]	$69
Velsicol Chemical (Pine River), MI	V	3-acre hot spot in St. Louis Impoundment	DDT, HBB, PBB	Dry excavation following stabilization; commercial landfill	35,000	$7.8	$246
Waukegan Harbor (Outboard Marine), IL	V	10 acres of 37-acre harbor; abandoned boat Slip #3; and a North Ditch which flowed directly into Lake Michigan	PCBs	Hydraulic dredging; Nearshore CDF	38,300	$15	$392
Willow Run Creek, MI	V	Edison and Tyler Ponds - 21 acres combined; Willow Run Sludge Lagoon	PCBs	Dry/wet excavation; nearby new on-site landfill	450,000	$70	$156

Table 1
Summary of Remediated Contaminated Sediment Sites

Project	USEPA Region	Setting	Contaminant Of Concern	Methods of Remediation and Disposal	Volume Removed (cy)	Total Cost (millions)	Total Unit Cost ($/cy)
ROUNDED TOTALS					2,774,430[6]	$522.3[6]	$462[6] (MEAN)

1. Does not include disposal cost. Several years delay to determine disposal method.
2. Final volume is a range; midpoint is listed.
3. Cost is a range; midpoint is listed.
4. Cost listed is a midpoint; actual not determined.
5. Listed twice since both dredging and dry excavation were used.
6. Does not include sites without either volume or cost data.

Table 2
U.S. Sediment Remediation Projects Implemented (>10,000 cy) – Primary Goal versus Outcome

Project	Primary Goal	Basis for Primary Goal	Sediment Remedial Target	Relationship of Target to Goal	Remediation Method	Achievement of Remedial Target	Achievement of Primary Goal
Bayou Bonfouca, LA[1] (169,000 cy)	Reduce PAH human contact risk to $<10^{-4}$ and minimize threat to aquatic biota.	Human health risk assessment	Depth horizon to achieve <1300 ppm PAHs	Direct	Mechanical dredging followed by fill	Depth horizon achieved; no analytical verification	Likely accomplished, particularly since fill was added to the dredged areas. However, post-monitoring consists of the state annual monitoring program for water, sediment, and fish and seems hit or miss. Also, it is unclear if targeted surface PAH levels were achieved since a sediment contact and swimming advisory is still in effect due to PAHs in sediment samples exceeding EPA guideline values, but not verified.
Manistique River, MI[1] (97,050 cy)	Reduce PCB in fish levels, reduce carcinogenic and noncarcinogenic risks to $<10^{-4}$ and <1, respectively, except for high-end subsistence and some high-end recreational exposure from fish consumption.	Human health risk assessment	10 ppm PCBs	Default level after using biota to sediment accumulation factor (BSAF) to estimate a target sediment level, then increasing the estimate to 10 ppm PCBs, which EPA justified based on cleanup levels at other EPA projects, the likelihood of achieving <10 ppm, and future natural burial	Hydraulic dredging	In progress; consistent achievement of 10 ppm or less proving difficult	Too soon to tell. Remediation still in progress in Year 5. No postmonitoring program defined as of yet.
LTV Steel, IN[1] (109,000 cy)	Remove all oil-contaminated sediments from a 3,500-foot man-made intake channel.	Clean Water Act Consent Decree	Depth horizon (removal down to original bottom)	Direct	Hydraulic dredging and diver-assisted removal	Depth horizon achieved; no analytical verification	Likely accomplished, but not verified.
United Heckathorn, CA[1] (108,000 cy)	Achieve EPA marine chronic water quality criteria of 1 part per trillion (ppt) DDT; achieve human health surface water criteria of 0.6 ppt DDT; achieve the National Academy of Sciences action levels for DDT in fish to protect fish-eating birds.	Ecological risk assessment	Remove all "young bay mud" to achieve <0.59 ppm DDT.	Indirect (calculated in the ecological risk assessment)	Mechanical dredging	Depth horizon (penetration into "old bay mud") achieved; 20 samples for chemical analysis collected from top 6 inches of final dredged surface for informational purposes (several exceeded 0.59 ppb DDT)	Too soon to tell; post-monitoring in progress.

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Table 2
U.S. Sediment Remediation Projects Implemented (>10,000 cy) – Primary Goal versus Outcome

Project	Primary Goal	Basis for Primary Goal	Sediment Remedial Target	Relationship of Target to Goal	Remediation Method	Achievement of Remedial Target	Achievement of Primary Goal
Marathon Battery, NY[1] (102,000 cy)	Eliminate adverse ecological impacts by achieving 100 ppm cadmium in sediment in East Foundry Cove (EFC) Marsh and 10 ppm cadmium in other areas; allow natural recovery in over 300 acres of adjacent cove/marsh.	Ecological assessment based on "weight of evidence," bioassay tests, and comparison with ambient water quality standards	Remove top 1 foot of sediment in areas targeting 10 ppm cadmium; remove to <100 ppm cadmium in EFC marsh; allow natural recovery in over 300 acres.	None other than 95% cadmium mass removal predicted	Hydraulic and mechanical dredging; dry excavation	Removed more than top 1 foot; decided to take verification samples for analysis in some areas; achieved an average of 25 ppm cadmium in EFC Marsh; achieved an average of <10 ppm cadmium in EFC and near pier	Post-monitoring in progress. Two years of reported results are inconclusive.
Black River, OH[1] (60,000 cy)	Remove all PAH- and metal-contaminated sediments.	Clean Air Act Consent Decree	Depth horizon (removal down to "hard bottom" or "bedrock")	Direct	Hydraulic and mechanical dredging	Depth horizon achieved; no analytical verification	Likely accomplished, but not verified.
Cherry Farm, NY[1] (Niagara River)(42,445 cy)	Reduce PAH-related risks to benthic aquatic life and fish.	Ecological and biotoxicity testing; literature review for ecotoxicity of PAHs	Depth horizons based on characterization data to achieve 20 ppm PAHs in the top 1 foot; 50 ppm PAHs below 1 foot	Vague; target levels set by negotiation and by comparing prevailing PAH levels to upstream background levels	Hydraulic dredging	Achieved depth horizons based on bathymetry; no analytical verification	Unknown; post-monitoring program being negotiated.
N. Hollywood Dump, TN[1] (40-acre lake)(40,000 cy)	Restore the pesticide-contaminated fishery in the lake so that it is suitable for human consumption.	Human health risk assessment	Remove or isolate pesticide-contaminated surface sediments.	Direct	Fish harvesting first, then part hydraulic dredging/part direct burial	Achieved	Too soon to tell; long-term bi-annual fish and sediment sampling in progress.
Outboard Marine, IL (Waukegan Harbor)[1] (38,300 cy)	Eliminate PCB flux from the harbor into Lake Michigan.	Hydrodynamic modeling	50 ppm PCBs in the harbor; 500 ppm PCBs in Slip #3	Direct for the harbor; unknown for the 500 ppm target in Slip #3	Hydraulic dredging	Unknown. No analytical verification. Dredged to a predefined depth in the harbor to the reportedly uncontaminated sand layer.	Unknown. Some limited analysis of surface samples at undefined locations in the harbor over four years after dredging exhibited 3 to 9 ppm PCBs. PCB levels in harbor fish are trending downward.
Ford Outfall, MI (River Raisin)[1] (28,500 cy)	Reduce PCB levels in fish.	Risk analysis by EPA	10 ppm PCBs after removal down to the native clay layer	Direct	Mechanical dredging	Partially achieved. Removal to refusal was accomplished. Verification by field test kits, then 14 samples (one per quadrant) for laboratory analysis; seven quadrants had insufficient sediment to collect; four quadrants exhibited 0.5 to 7 ppm PCBs; three quadrants exhibited 12 to 20 ppm PCBs.	Unknown. No formal post-monitoring program identified. Results of fish samples and caged fish studies from a monitoring program performed by MI Department of Environmental Quality (MDEQ) are not yet available. Two post-removal sediment core samples taken by MDEQ from the dredged area exhibited 60 and 110 ppm PCBs.

Table 2
U.S. Sediment Remediation Projects Implemented (>10,000 cy) – Primary Goal versus Outcome

Project	Primary Goal	Basis for Primary Goal	Sediment Remedial Target	Relationship of Target to Goal	Remediation Method	Achievement of Remedial Target	Achievement of Primary Goal
New Bedford Harbor, MA[1] (14,000 cy)	Remove PCB mass at an optimum "residual concentration to volume removed" ratio and reduce PCB flux to the water column (interim measure).	Mass removal calculations; flux modeling studies conducted by PRPs; water column data	4,000 ppm PCBs in five acres of hot spots	Direct	Hydraulic dredging	Achieved based on a limited number of verification samples (15 composite samples ranging from 67 to 2,068 ppm PCBs)	Achieved mass removal. Water column data post-dredging (if collected) not obtained. PCBs in surface sediment samples in the Upper Harbor increased 32% on average, following hot spot dredging.
GM (Massena), NY[1] (13,800 cy)	Reduce PCB levels in fish.	Human health risk assessment	Achieve 1 ppm PCBs and remove as much sediment as technically feasible.	Vague; 0.1 ppm PCBs desired, but 1 ppm selected based on technical feasibility	Hydraulic dredging	Not achieved. Average residual PCB levels at completion in six dredged quadrants across 11 acres ranged from 3 to 27 ppm with a maximum of 90 ppm	Two annual post-dredging fish monitoring programs completed. No discernible trends other than a slight increase in fish PCB concentration in Year 2 versus Year 1.
Gould (Portland), OR[1] (11,000 cy)	Vague; apparently protect from direct contact risk and remove lead-contaminated surface (0 to 2 feet) sediments that exceed the extraction procedure (EP) toxicity concentration.	Applicable or relevant and appropriate requirement (ARAR)	5 ppm lead	Not identified	Hydraulic dredging followed by filling in the 3.1 acre lake	Achieved based on verification sampling	Apparently achieved.
Newburgh Lake, MI[2] (588,000 cy)	Restore 105-acre lake depth and restore fishery.	Not identified.	Depth horizon which will both restore depth and remove the detectable PCBs	Direct	Dry excavation supplemented by hydraulic dredging in undrained bypass channel through the lake	Depth horizon achieved; no analytical verification	Achieved, but no analytical verification. Fish harvested and restocked. Post-monitoring not identified.
Willow Run Creek, MI[2] (450,000 cy)	Eliminate adverse ecological impacts.	Ecological assessment based on ecological ingestion modeling, then feasibility and compliance with MI Environmental Response Act 307	Removal to 21 ppm or 1 ppm PCBs below waterline depending on locale; removal to 21 ppm or 2.3 ppm PCBs above waterline	Direct	Dry excavation	Achieved based on verification sampling	Unknown. No formal post-monitoring is planned.
Lipari Landfill, NJ[2] (163,500 cy)	Reduce human health risk from direct contact with or air exposure to targeted VOCs to below 10^{-6}.	Human health risk assessment	Depth horizon 6 inches into the underlying Kirkwood Clay layer to achieve nondetect for bis(2-chloro-ethyl)ether	Direct	Dry excavation	Depth horizon achieved except in areas where no Kirkwood Clay was encountered, in which instances excavated 18 inches below a level extrapolated from adjacent contiguous clay layers; no analytical verification.	Apparently achieved, particularly since clean fill was also placed. No post-monitoring identified.

Table 2
U.S. Sediment Remediation Projects Implemented (>10,000 cy) – Primary Goal versus Outcome

Project	Primary Goal	Basis for Primary Goal	Sediment Remedial Target	Relationship of Target to Goal	Remediation Method	Achievement of Remedial Target	Achievement of Primary Goal
Bryant Mill Pond, MI (Kalamazoo River) [2](165,000 cy)	Mitigate the public health threat posed by direct human and wildlife contact and mitigate threats posed to aquatic life and wildlife by ongoing releases (i.e., source control) to the Kalamazoo River.	Ecological risk assessment along with direct observation of continuing releases by erosion and sloughing from banks	10 ppm PCBs	Unknown	Dry excavation	Reportedly achieved based on verification sampling. Sample results not obtained or reviewed.	Unknown and probably too early to tell since removal was completed in June 1999. However, as stated in the Action Memorandum, "the nature of the removal is, however, expected to minimize the need for post-removal site control, at least in the Bryant Mill Pond area."
Loring AFB, ME[2](162,000 cy)	Reduce human health risk to below 10^{-6} and below a hazard index of 1 and eliminate adverse ecological impacts.	Human health and ecological risk assessments	Various for specific contaminants (e.g., 1 ppm Aroclor 1260, 35 ppm total PAHs)	Direct	Dry excavation	Apparently achieved based on verification sampling for PCBs and less rigorous testing for five other indicator compounds	Too soon to tell. A long-term environmental and wetlands monitoring plan was finalized in late 1998.
Love Canal, NY[2](31,000 cy)	Reduce human health risk from direct contact and from fish consumption.	Evaluation of various health advisories for dioxin from multiple sources such as NY Department of Health, Canadian agencies, and FDA	1 ppb 2,3,7,8-TCDD (CDC action level)	Direct	Dry excavation	No details obtained	Probably achieved, but no details obtained.
Hooker (102nd Street), NY[2](28,500 cy)	Vague; apparently reduce risk from fish ingestion to below 10^{-4} to 10^{-6} and a hazard index of 1 and reduce water concentrations to below state water quality standards.	Human health risk assessment and environmental endangerment assessment	Remove out to a "clean" boundary line and to a depth horizon dictated by characterization data	Vague	Dry excavation	Areal and depth horizon achieved; no analytical verification	Too soon to tell. One foot of fill added to remediated areas. No post-monitoring identified.
Tennessee Products, TN[2](24,100 cy)	Remove visual coal tar material from several thousand feet of the creek (interim measure).	Non-time critical removal action	Remove all visual coal tar material	Direct	Dry excavation	Achieved. Visual confirmation only.	Achieved. Visual confirmation only.
Town Branch Creek, KY[2](17,000 cy)	Reduce PCB in fish levels to <2 ppm FDA limit.	State environmental agency evaluation and Circuit Court Judgment	0.1 ppm PCBs	Direct	Dry excavation	Achieved sediment removal to extent practical but not always 0.1 ppm in 30% of 3.5 miles of creek so far. Work on remaining 2.5 miles on hold pending resolution of access issues.	Too soon to tell. Post-monitoring planned after all of remediation is completed.
Triana/Tennessee River, AL[2](no removal)	Reduce DDT in fish levels to <5 ppm FDA limit.	Negotiated agreement and Consent Decree to restore the fishery	Rechannelization and direct burial of the two isolated tributaries (2.5 miles) containing an estimated 93% of the DDT mass	Vague, basically a "try it and see what happens" approach	Stream diversion, direct burial, and some natural recovery	Achieved	Substantial progress. One target species reached the 5 ppm standard in the 10-year attainment period, two species did not but they exhibit 80 to 90% DDT reductions in the 10 years. Annual monitoring continuing.

Table 2
U.S. Sediment Remediation Projects Implemented (>10,000 cy) – Primary Goal versus Outcome

Project	Primary Goal	Basis for Primary Goal	Sediment Remedial Target	Relationship of Target to Goal	Remediation Method	Achievement of Remedial Target	Achievement of Primary Goal
James River, VA[3](no removal)	Allow natural recovery of fish and biota to below FDA limit for Kepone (0.3 ppm in fish and 0.4 ppm in blue crabs).	Technical impracticability of achieving FDA limits in fish by remediation	None	N/A	Natural recovery	N/A	Natural burial by clean sediments is continuing to decrease the bioavailability of Kepone. Crab/oyster Kepone levels dropped from 0.8 to 0.1-0.2 ppm from 1976 through 1985. The commercial fishing ban was lifted in 1988; only a subsistence fish eating advisory remains.
Sangamo-Weston, SC[3](no removal)	Reduce PCB in fish levels to <2 ppm FDA limit by natural recovery.	Technical impracticability of achieving risk-based concentrations in fish by remediation; existence of an ARAR (the FDA limit); and the voluntary nature of fish consumption	1 ppm PCBs	Default level, per the Record of Decision: "The time for two to eight year old largemouth bass to achieve 2 ppm for the range of sediment cleanup goals was compared to a baseline. It was determined that fish PCB levels decline at about the same rate regardless of sediment cleanup goal. Therefore, 1 ppm was selected based on technical feasibility…"	Natural recovery; modeling predicts 2 ppm levels in fish will be reached by 2004	Too soon to tell	Too soon to tell. Annual monitoring in progress. No reports yet available for review.

Notes:

1. True dredging projects
2. Dry excavation projects
3. Natural recovery projects

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TABLE 3

COST FACTORS ASSOCIATED WITH SEDIMENT REMOVAL

A. Extent of Sediment Subject to Removal

- Larger Extent = Larger Costs
- Economies of Scale Advantages Significantly Diminish with Larger Projects

B. Dredge production rate which is primarily dependent upon:

- unique site conditions such as access, water depth, debris/vegetation, and free oil
- the targeted sediment depth or cleanup level
- limitations in land-based water management facilities
- operational controls imposed to limit resuspension
- whether or not verification sampling is performed during dredging

C. Disposal cost which is dependent upon type of contaminant, and type and location of disposal facility. Commercial disposal facilities tend to be more costly, but may be appropriate for smaller projects or may be required under regulation (e.g., RCRA, TSCA)

- The disposal methods for 50 completed removal projects were: offsite landfill or pond (26); onsite landfill, pond/CDF, or burial (15); offsite thermal treatment (2); onsite thermal treatment (3); other, such as stabilization and beneficial reuse (4); disposal method not selected or unknown (2). (Note: Two of the projects used a combination of 2 disposal methods)

D. Access: Availability of upland areas for staging, sediment processing, and disposal (if on-site) can significantly affect cost and the absence of such areas in fact makes a project infeasible. Limited access can result in higher costs due to:

- More extensive river-based transport of sediment
- Costs to obtain access from property owners
- More extensive land-based transport of sediment

E. Presence of Rocks, Vegetation, and Debris: The presence of obstructions not only impacts dredge selection, but may require multiple equipment types to be used, which will increase costs.

TABLE 4

PERFORMANCE FACTORS ASSOCIATED WITH SEDIMENT REMOVAL

A. Performance Metrics – Primary Risk-Based Measurements of the Effectiveness of Removal

- Bioavailable Surface Sediment Characteristics Before and After Removal
  - Chemical Contamination Levels
  - Organic Carbon Levels
  - Physical Characteristics (Affecting Mobility)
  - Density
  - Geotechnical (Cohesion, etc.)
  - Bathymetry (verify amount removed and geometry)

- Biota Concentrations Before and After Removal
  - Resident Fish
  - Other Site-Specific Species
  - Caged Fish (Controlled Study Bioavailability Indicator)
    - Can Also Be used During Removal

- Water Column Data Before, During, and After Removal
  - Chemical Contamination Levels
  - Total Suspended Solids (TSS)
  - Turbidity (sometimes an indicator of TSS)

- Ambient Air Concentrations Before, During, and After Removal
  - Need for measurement is Chemical and Site-Specific

B. Factors Affecting Performance of Sediment Removal

- Aquatic Environment Characteristics
  - Water Body Type (Lake, River, Harbor, Estuary, Bay)
  - Water Level Fluctuations (Tides, Seiche, etc.) – Can affect accessibility to sediment
  - Water Velocities – Will affect selection and performance of dredge equipment and resuspension controls
  - Water Depth – Will affect accessibility and equipment selection

- Sediment Characteristics
  - Presence of Debris (rock, timber, man-made objects) – will require removal or will limit effectiveness of removal; even with removal may create cavities which may limit removal of remaining sediment
  - Sediment Depth – Deeper sediment removal drives multiple dredge passes, more likely to leave furrows/windrows and higher removal volumes to account for side sloughing
  - Subbottom Characteristics (Below Contamination) – Bedrock, hard pan, and irregularity all act to reduce effectiveness of removal by inherently leaving material behind
  - Sediment Type (Sand, Gravel, Silt, Clay) – Fines will tend to be resuspended and either migrate, desorb contamination, and/or settle (in the removal area or elsewhere in system); also clays tend to clog hydraulic dredges
  - Type of Contamination – Highly sorptive chemicals will tend to stay with solids; less sorptive compounds more likely to be released to water column
  - Chemical Concentration Profile – Higher contamination at depth will have a tendency to result in higher concentrations remaining after removal

- Removal Equipment Selected – dredging (or removal through water column) inherently limits capability to accurately remove sediment since operator can't see sediment to be removed
  - Hydraulic dredges – (Numerous Types Available)
    - Resuspension inevitable, although generally less than mechanical removal

    - Material left behind due to "furrowing", irregular subbottom, settling, or resuspended material
    - Releases with transport pipeline malfunctions/breaks
  - Mechanical Dredges (Primarily Clamshells)
    - Resuspension inevitable; recent innovations (Cable Arm, Bonacavor) claim to reduce, but can't eliminate
    - Material left behind due to "cratering", sloughing, irregular subbottom, settling of resuspended material
  - Excavation in "Dry" Conditions
    - Air emissions (dust, chemical) may need to be controlled
    - Material left behind due to irregular subbottom, "smearing', equipment tracking, wet slurry conditions from infiltration

- Resuspension Control System – Suspended silt curtains, sheetpiling typically used to minimize migration of inevitable sediment resuspension. None are watertight, so releases are inevitable. The higher degree of containment will act to allow reuspended sediment to settle within removal area, less containment will allow material to settle outside removal area.

- Disposal Method
  - Onsite (landfill, confined disposal facility) vs. offsite commercial facilities
  - The method of disposal will affect the dredge technology selection, and limit sediment removal rates (due to dewatering and water treatment requirements)

- Predisposal Processing – This factor is primarily defined by the disposal method and may include
  - Primary settling
  - Dewatering
  - Stabilization/Solidification
  - Water Treatment
  - The extent of pre-processing required will drive the need for space, affect dredge selection, affect production rates (may increase project duration), and increase risk of contaminant release (more unit processes)

C. **Uncertainties Associated with Sediment Removal**

- Unpredictability of Sediment Concentration After Removal
- Bioavailable Surface Sediment Concentration Affects Biota Levels and Water Column concentrations
- Highly Variable Results Achieved Elsewhere (see Table 1 and 2)
- Numerous Variables Involved (see Table 3) which Essentially Prohibit Prediction of Results at a Given Site
- This Uncertainty Must be Recognized Before Embarking on Sediment Removal Project
- Site Conditions Never Entirely Predictable
- Underwater Environment Compounds This Common Uncertainty at All Contaminated Sites
- Surprises Are Inevitable
  - Volumes Tend To Increase
  - Debris Tends to Be More Extensive
- Project Schedule and Cost (refer to Cost Factors in Table 3)
- Weather Unpredictability Can Affect Schedule and Cost
- Extent of Winter Weather Affects Overall Schedule
  - Freeze-up Significantly Reduces or Prohibits Removal Productivity and Interferes with Land-Based Water Handling and Treatment
- Items A and B above Also Impact Schedule and Cost

Appendices

Appendix A

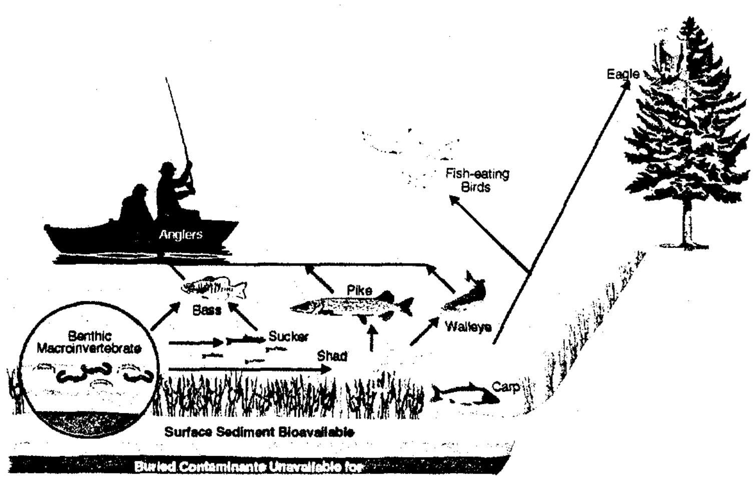
Surface Sediments Play Key Role in Driving Risk

APPENDIX A - Surface Sediments Play Key Role in Driving Risk

Contaminants accumulate in sediments if they possess chemical properties that cause them to associate preferentially with the particulate matter that forms the sediment. These same properties tend to cause such contaminants to accumulate in biotic tissue and to become more concentrated as they are transferred through the food web. As a result, ingestion of fish is typically the prevailing human and ecological exposure pathway at contaminated sediment sites.[1]

The transfer of a contaminant from sediment to fish is initiated by direct transfer from sediments to benthic animals or by the flux of contaminant from the sediment to the water column and the transfer from water to animals living in the water column. Either way, the sediments involved in the transfer are those close to the sediment-water interface. Sediments buried below the surface "mixed" layer subject to disturbance by hydrodynamic forces or inhabited by benthic animals typically provide almost no contribution to the transfer process. This is so because the contaminant's propensity to associate with the sediment particulate matter greatly inhibits its ability to migrate from below the mixed layer into the mixed layer.

At most sites, the primary route of exposure for people or wildlife is consumption of fish that have accumulated contaminants from the surface of the sediment bed. Contaminants located at the sediment surface, as shown in the adjacent diagram, are "bioavailable" and thus prone to transfer up the food chain from benthic organisms to fish and on to higher-level receptors such as fish-eating birds and mammals.



The size of the surface mixed layer depends on the nature of the sediment particles, the magnitude of the forces placed on the sediments by currents and waves and the depth to which infaunal benthic animals mix sediments in a process termed "bioturbation." In most cases, bioturbation is the controlling factor. Studies have shown that depths can range up to about 20 centimeters, but are typically on the order of 10 centimeters or less in sandy substrate (Palermo et al., 1998). Below this hydrologically and biologically active surface layer, contaminants may be locked in the consolidated deeper sediments and, according to the IJC (1997), "once buried in deep sediment, particles are often considered lost to the system" and thus unavailable for transport or exposure. In these cases, newer sediments with continually lower concentrations deposit on the surface and gradually bury those older sediments having higher concentrations representative of past discharges. These long-buried contaminated sediments remain unavailable

[1] Major transport mechanisms include downstream migration of contaminated fine-grained materials that are suspended within the overlying water column (carried as a portion of bed load); partitioning to dissolved organic matter; or available as dissolved-phase in the water column (Paris, et al. 1978; Valsaraj et al., 1997).

for biological exposure and therefore pose no appreciable associated risks. In the words of a guidance document from EPA's Assessment and Remediation of Contaminated Sediments (ARCS) Program (EPA, 1998):

> Humans, aquatic organisms and wildlife will generally only be exposed to sediment contaminants in the uppermost active layer of the sediment deposits. Hence, contaminated sediments separated from the overlying water by a surface layer of relatively clean sediments may not represent an ongoing risk to humans, aquatic organisms or wildlife. [I]n fact, as ARCS and other coring studies have shown, the most contaminated sediments may be located well below the surface sediment (i.e., in older sediments)."

These factors combine to suggest that in order for dredging (or any other remedy) to be effective in reducing exposure and associated risks, it must "break the link" between the surface sediment source of contaminants and the fish and other receptors within the system's food webs. If remediation can effectively reduce surface sediment concentrations, bioavailability will be reduced and subsequent exposure to all receptors along the food chain from benthic organisms to fish and on to humans and wildlife also will be reduced. Remedial actions that do not address these linkages will not be effective in reducing bioavailability, exposure, and potential risks (IJC, 1997). Thus, any action that fails to create a sufficiently clean sediment surface will not be effective in achieving the desired risk reduction.

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Appendix B

Environmental Dredging – Site Profiles

APPENDIX B – Environmental Dredging – Site Profiles

This appendix summarizes several case-study examples of dredging.

Among the many sites referenced or mentioned in this paper, the following sites are reviewed in greater detail within this appendix:

- Grasse River, NY
- St. Lawrence River, NY
- Sheboygan River, WI
- Lake Järnsjön, Sweden
- Fox River, WI (2 projects)
- Duwamish Waterway, WA
- River Raisin, MI
- Manistique River/Harbor, MI
- Shiawassee River, MI
- Ruck Pond, WI
- Waukegan Harbor, IL
- New Bedford Harbor, MA

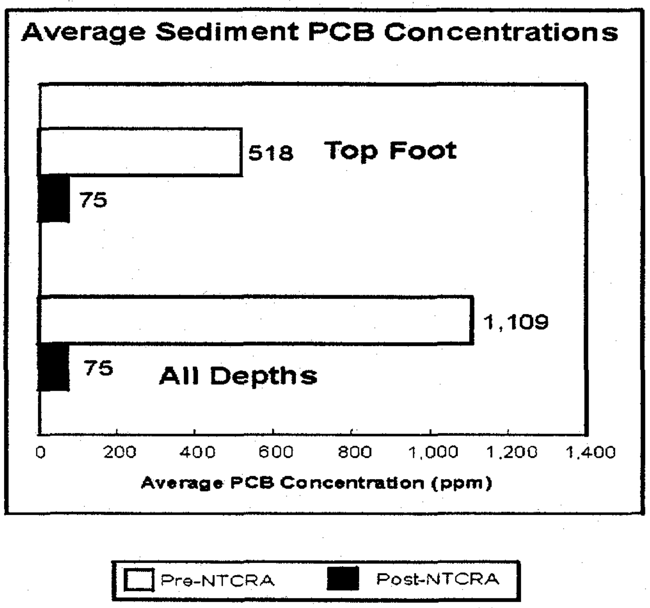
Compared to navigational dredging, environmental dredging is in its infancy. Through 1999, only about 50 sediment removal projects have been completed, compared with the many hundreds of navigational dredging projects completed over many decades. These 50 projects largely exclude small projects [i.e., less than 3,000 cubic yards (cy)], since these smaller projects typically represent spill cleanups, interim measures, or "hot-spot" removal actions that are much less representative of larger-scale dredging. Monitoring data at these 50 sites is typically lacking and sporadic. Indeed, the International Joint Commission (IJC) (1999) notes that for 38 remediation projects in the Great Lakes region, "only two currently have adequate data and information on ecological effectiveness." Further, the IJC suggests that "much greater emphasis be placed on post-project monitoring of effectiveness of sediment remediation," that "a high priority be placed on monitoring ecological benefits and beneficial use restoration," and that "additional research is essential to ... forecast ecological benefits and monitor ecological recovery and beneficial use restoration in a scientifically defensible and cost effective fashion" (IJC, 1999). Of the 50 completed projects, 25 are polychlorinated biphenyl (PCB) sites (see Table 1), and of these 25, 13 have some data that are useable for asssessing how effective dredging has been. Each of these sites are discussed below.

As described in Appendix A, the level of PCBs accumulated by fish depends on the concentration of PCBs found in surface sediment and the water column. Although PCB concentrations in fish may be the most important source of potential risks to humans and wildlife, it can take years for PCB concentrations in fish to respond to a dredging project. In addition, there are limited fish data available for completed environmental dredging projects. Thus, PCB concentration in residual surface sediment provides a more immediate and the most important measurement of the effectiveness of dredging in reducing human and ecological risks. This appendix discusses the available data for residual PCB concentrations in surface sediment, the water column, and fish tissue for several environmental dredging projects. A more thorough evaluation of fish data at many of these sites is provided in the paper titled "Effectiveness of Sediment Removal: An Evaluation of EPA Region 5 Claims Regarding Twelve Contaminated Sediment Removal Projects" (FRG, 1999), which is included as Appendix C. Additional information on these sites and other sediment removal projects can be found in the Major Contaminated Sediment Sites (MCSS) database.

Grasse River – Massena, New York

Between July and September 1995, Alcoa, Inc. removed approximately 3,000 cy of sediment and boulders/debris from two areas of the Grasse River due to elevated levels of PCBs (up to 11,000 mg/kg). The removal areas covered approximately 1 acre of the Grasse River (i.e., a river area and adjacent outfall structure). The goal of the removal action was to remove all sediment within these areas to the extent practicable. Nearly 400 cy of boulders were removed from a "boulder zone" with a mechanical long-stick excavator (with a specialized perforated bucket) mounted on a barge. The sediments were removed using a horizontal auger hydraulic dredge. Sediments were dewatered and disposed with the boulders and debris in an on-site landfill (BBL, 1995b). Sediments within the outfall structure were removed using small manually directed plain-suction hydraulic hoses.

Sediment Data:

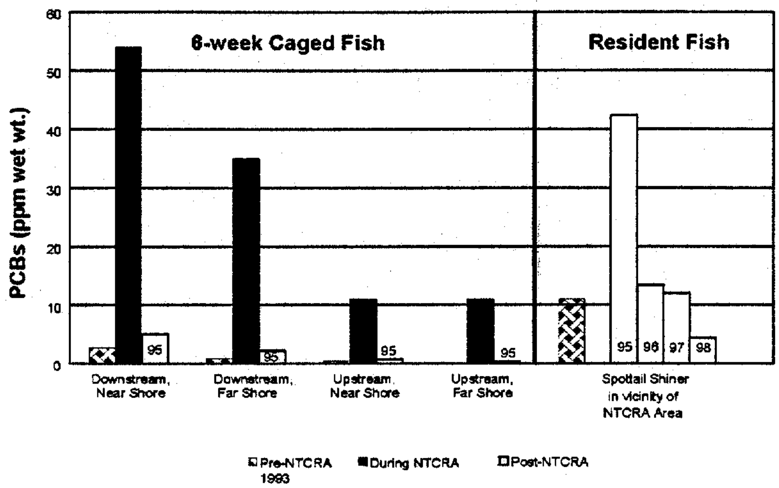


As shown on the figure at left, pre-removal PCB surficial sediment concentrations (i.e., top 12 inches in this case) ranged from 12 to 1,780 parts per million (ppm) (average of 518 ppm). After hydraulic dredging was completed in an effort to remove all sediment, an average sediment depth of 4 inches (up to a maximum of 14 inches) remained even after multiple dredge passes. Based on these results, U.S. Environmental Protection Agency (EPA) and its representatives, Alcoa, and the contractors determined that sediment had been removed to the extent practicable (BBL, 1995c). Conditions such as the rocky nature of the river bottom and the presence of hardpan reduced the dredge's effectiveness in removing sediment. It was estimated that approximately 84% of the sediments were removed (along with 27% of the PCB mass in the lower Grasse River). Following removal, residual (surficial) PCB concentrations ranged from 1.1 to 260 ppm (average of 75 ppm). Moreover, at 30% of post-removal sample locations, residual surface sediment PCB concentrations increased relative to pre-removal concentrations (BBL, 1995c). Even in the outfall structure, where operators were able to manually direct vacuum hoses to remove sediment, surface sediment remained with PCB concentrations of 108 ppm (388 ppm PCBs in surface sediment before removal).

Water Data:

During removal activities, a triple-tiered silt curtain system was used in an attempt to contain suspended PCB-containing sediments. The curtains were quite effective in containing suspended sediments, with only one action level exceeded for total suspended solids (TSS) and turbidity. However, elevated PCB water column concentrations were observed; that is, PCBs were present in 88% of the samples collected at a location 2,300 feet downstream of the removal area, while PCB were detected only once at the upstream location. Also, two of the downstream fixed-station filtered samples had quantifiable PCB levels, whereas quantifiable levels were never observed at this location in the pre-removal monitoring.

Fish Data:
In addition to water column PCB level increases during removal, increases in fish levels also were noted during removal. The figure to the right shows both caged fish and spottail shiner data before, during, and after removal. Although limited data are available before removal, it is obvious that sediment removal increased PCB levels in fish during removal, and levels remained elevated for several years following removal.



Other resident fish (i.e., brown bullhead and smallmouth bass) also were collected and analyzed for PCBs as part of pre- and post-removal monitoring (through 1998) of the Grasse River project. Review of the post-removal monitoring results reveal that there was generally no reduction in potential long-term risks to human health and the environment as a result of these dredging activities. For example, resident fish collected in 1995 immediately following removal exhibited an increase in PCB concentrations. PCB concentrations in resident smallmouth bass and brown bullhead samples collected prior to the removal activities are similar to those collected in 1997 and increased slightly in 1998. Overall, the apparent negative effect of the removal was greater for smallmouth bass than for brown bullhead and was most significant for spottail shiners, with the most significant differences observed in the vicinity of the removal area.

St. Lawrence River - Massena, New York

Between May 8 and December 22, 1995, General Motors (GM) removed approximately 13,250 cy of PCB sediment and associated boulders/cobbles from an approximate 11-acre area of the St. Lawrence River. These materials were dewatered and stockpiled at the GM Powertrain facility for subsequent off-site disposal.

EPA selected a 1 ppm sediment cleanup goal in the St. Lawrence River because it believed it was achievable and provided an acceptable measure of human health protection. In doing so, EPA, believed it had balanced its desire for a very low cleanup level to minimize residual risk with the constraints posed by the limitations of dredging as a means of removing sediment (in Turtle Creek, an applicable or relevant and appropriate (ARAR) cleanup level of 0.1 ppm was set). However, EPA recognized that technical limitations may preclude removal of sediments to this level (EPA, 1990b).

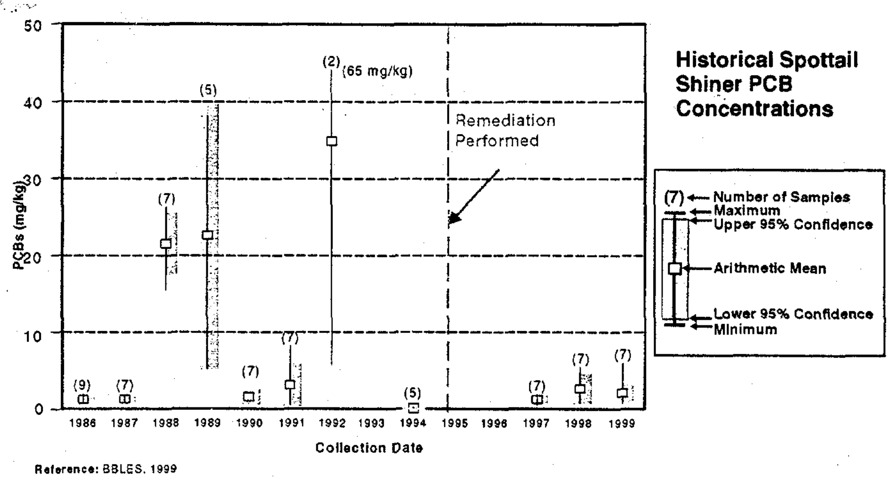
After efforts to utilize a silt curtain containment system failed (due to excessive water velocities), a sheetpile wall was installed around the removal area as a suspension containment measure. Prior to sediment removal, the initial footprint of the sheetpile wall was modified to exclude a cobble and boulder zone. It was agreed by the EPA and GM that the removal of sediment from this area was technically impractical because of large boulders and the potential for slope failures. Within the removal area, boulders and debris were removed mechanically prior to hydraulic dredging.

Sediment Data:
Pre-removal surficial sediment PCB concentrations ranged from non-detect to 4,430 ppm (average of 200 ppm) (ERM, 1993).

Even after significant passes with a hydraulic dredge were performed (up to 15 to 30 passes in some areas), residual surface sediment in all six removal quadrants remained above the cleanup goal of 1 ppm, with an overall average PCB-concentrations of 9.2 ppm (average PCB concentrations were up to 27 ppm in one quadrant). EPA determined that sediments were removed to the maximum extent possible. Consequently, EPA "determined that installation of a cap over Quadrant 3, effectively isolating this area from the rest of the river, was the only remaining technically practicable remedial alternative." This area was subsequently capped with a multi-layer granular cover (BBLES, 1996a).

Water Data:
Early on in the sediment removal process, turbidity action levels were exceeded due to turbid water escaping over the top of low sheetpiling sheets. The low sheets were installed according to the design and assured stability of the containment system during storms and high waves from passing ships. To compensate for the low sheets, the contractor installed filter fabric over the low sheets and installed short steel sheets over some of the low sheetpiles. At one point during sediment removal activities, elevated water column turbidity and PCB levels were reported outside of the sheetpile wall. Due to the high concentrations, a silt curtain was installed along the inside of the sheetpile wall. PCBs were also released via air as PCBs were detected at levels exceeding the project action level at the closest downwind sample location.

Fish Data:
The figure below shows total PCB concentrations in spottail shiner (the only species monitored) whole-body composite samples collected from the GM site. PCB levels may have decreased since the late 1980s, but comparison of the pre- and post-remediation data are complicated by factors such as fish sizes, lipid contents, species, mobility, and uncertainties about sampling locations (especially the 1988-89 and 1992 data relative to all other years). Previous sampling locations are important for data comparability over time. Note that remediation occurred in 1995.



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The annual monitoring reports describe an anomaly to the apparent general downward trend since the late 1980s: two spottail shiner samples collected by New York State Department of Environmental Protection (NYSDEC) in 1992. The wide difference in concentrations for these two samples (total PCB concentrations of 5.7 mg/kg and 65 mg/kg) is difficult to explain. Similar variability, although not as great, is also evident in the data collected by the Ontario Ministry of the Environment (OME) in 1989. The variability of the data may be due to several factors, including differences in sampling locations, fish lengths and sizes, fish lipid content, or species mobility. In fact, discussions with both NYSDEC and OME regarding sampling locations indicate that the specific sampling locations cannot be determined. This is extremely important given the relative size of the St. Lawrence River [about 2,000 feet wide 250,000 cubic feet per second (cfs)] compared to the area dredged (about 200 feet wide in an embayment). Post-dredging sampling locations are well documented, but without pre-dredging location details, one cannot consider the data truly comparable. Regardless, the variability of the data precludes a more detailed evaluation and interpretation of the overall spottail shiner data. As such, the monitoring reports conclude that the significance of the 1997, 1998 and 1999 PCB data, and any apparent trends, will need to be more thoroughly evaluated following the collection of additional data over the next several years.

Sheboygan River – Sheboygan Falls, Wisconsin

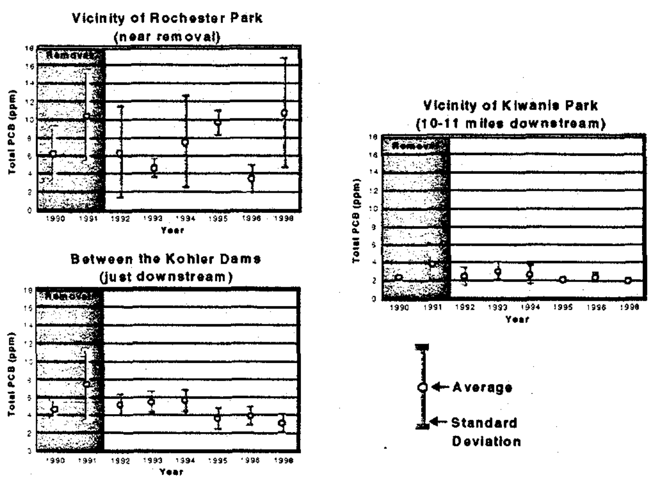
Approximately 3,800 in-situ cy of PCB-containing sediments were removed from the Sheboygan River by Tecumseh Products Company (Tecumseh), the only participating potentially responsible party (PRP), from 17 discrete sediment deposits in the Upper River from 1989 through 1991 using a modified "sealed" clamshell mechanical dredge. Dredging was performed within the confines of a silt containment system comprised of an internal geotextile silt screen and external geomembrane silt curtain. In general, a minimum of two dredge passes (and up to four passes in some areas) were performed in each area followed by sampling and analysis. The first dredge pass was performed in an effort to remove as much sediment as possible (i.e., to hard subgrade material). Following the first pass, the resuspended sediment within the silt containment system was allowed to settle, and a second dredge pass subsequently followed. Additional dredge passes were utilized if post-dredging sampling results exhibited elevated PCB levels (BBLES, 1992; BBL, 1995a, 1998).

Sediment Data:
Pre-removal surficial sediment concentrations ranged from 0.2 to 4,500 ppm (average 640 ppm) in 1987. Post-removal surficial sediment concentrations ranged from 0.45 to 295 ppm (average 39 ppm). Following four dredge passes, one sediment deposit exhibited residual PCB concentrations up to 295 ppm. The EPA and Wisconsin Department of Natural Resources (WDNR) agreed that the sediment had been removed to the extent practicable and directed Tecumseh to cap and armor the deposit to contain the sediment and residual PCBs (BBL, 1995a). At another Upper River deposit, pre-removal surficial sediment PCB concentrations ranged from 2.6 to 8.2 ppm (average of 5 ppm) with 1.6 to 1,400 ppm (average of 376 ppm) present in subsurface sediment. Following several removal passes, up to 136 ppm remained in a portion of this deposit. Again, the EPA and WDNR directed that that portion of the deposit be capped/armored. Two other deposits also required capping and armoring to contain elevated residual PCB concentrations following dredging. Removed sediments remain in on-site facilities pending final disposal.

Water Data:
Water-column monitoring activities were conducted before, during, and after sediment removal activities by measuring total suspended solids (TSS) and/or turbidity and PCBs. Monitoring data indicated an increase in PCB concentrations in the water column during dredging. As a result, dredging was halted several times during the project due to increased turbidity, PCB water-column concentrations, or visual observations of sediment migration. Specifically, PCBs were detected in one or more fixed downstream sampling stations during 19 of 29 sampling events, with the highest measured concentration of 0.47 ppb detected at a location approximately 500 feet downstream of removal activities. No PCBs were detected at the upstream location during that sampling round. Typical causes of elevated PCB or turbidity levels included water disturbances from boats, breaking ice, barges in motion upstream of the sample locations, damaged silt curtains due to high flows, etc. In addition, PCB concentrations within the silt control system were as high as 8.3 ppb (measured 11 days after dredging activities were completed) (BBL, 1995a).

Fish Data:
The figure at left shows the smallmouth bass data collected during and after removal activities. Note that no pre-removal data are available due to a laboratory problem. There is no apparent downward trend, and therefore no apparent risk reduction, in the Rochester Park vicinity (area where removal activities were concentrated), despite removal of over 95% of the PCB mass from the targeted deposits and 70% overall mass removal from the Upper River. In addition, although a slight downward trend is evident between the Kohler Dams and in the vicinity of Kiwanis Park, after sediment removal, both locations show an increase in 1991, possibly a result of removal activities.

Sheboygan River - Smallmouth Bass Mean Total PCB Concentrations (1990 - 1996, 1998)

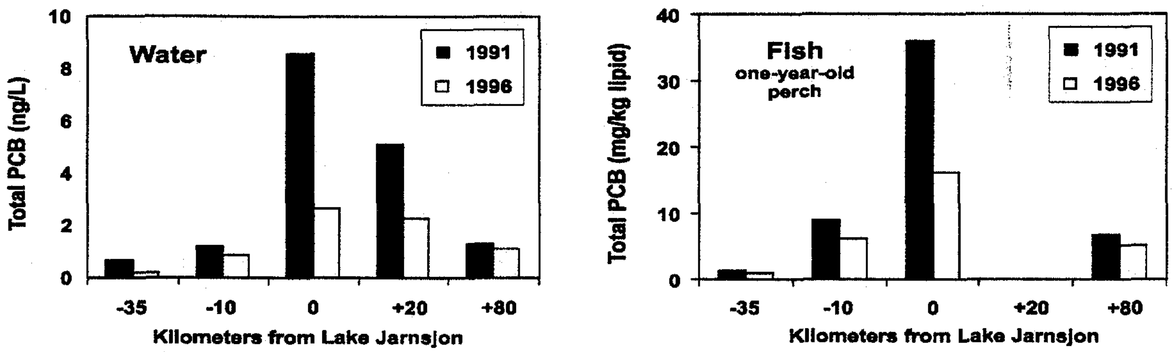


Lake Järnsjön - Sweden

Lake Järnsjön is a 62-acre lake located 72 miles upstream of the mouth of the Emån River in Sweden. In 1993/1994, approximately 196,000 cy of PCB sediments were removed from the lake.

Sediment Data:
Pre-removal PCB concentrations in sediment in 1990 and 1992 ranged from 0.4 to 30.7 ppm (average 8.1 ppm) in the top 1.3 feet and 0.18 to 2.9 ppm (average 1.5 ppm) in the top 0.1 foot (Bremle, Okla, and Larsson, 1998). Sediment remained following dredging with post-removal concentrations ranging from 0.01 to 0.85 ppm (average 0.13 ppm) from the top 0.66 feet (Bremle, Okla and Larsson, 1998).

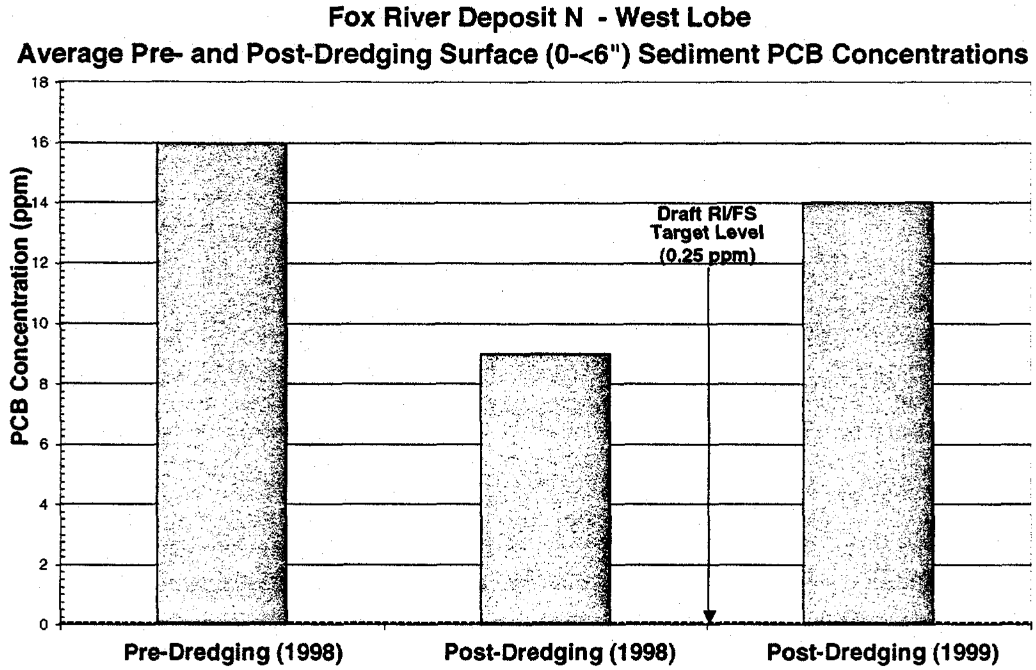
Water and Fish Data:
Although this project appears to have been successful in reducing surficial sediment PCB concentrations, review of the fish data indicate that PCBs in the lake continue to influence fish concentrations.



The two graphs shown above depict total lipid-normalized PCB concentrations in fish (one-year-old perch) and water from the Emån River, comparing 1991 pre-remediation levels with 1996 post-remediation levels. Spatial trends are also apparent and indicate that while PCB concentrations decreased by approximately 50% in Lake Järnsjön, upstream and downstream concentrations were also on the decline likely due to ongoing system-wide natural recovery processes. Finally, it is apparent that even after dredging an estimated 97% of PCB mass from the entire bottom of Lake Järnsjön, lake sediments remain a dominant source of PCBs to fish and the water column (FRG, 1999).

Fox River Deposit N – Kimberly, Wisconsin

Sediment Data:
Approximately 8,200 cy of sediment was removed from a 3-acre area at Deposit N [Note: This volume includes 1,000 cy of sediment from a nearby sediment area (Deposit O)] in the Fox River located near Little Chute and Kimberly, Wisconsin beginning in November 1998 as part of a demonstration project. The project specification for the demonstration project was to remove the majority of the contaminated sediments from the 3-acre area deposit efficiently and in a cost-effective manner, realizing that a thin layer of sediment would be left behind due to the presence of bedrock and the limitations of dredging (Foth & VanDyke, 2000). The sediment volume targeted for removal was approximately 65% of the 11,000 cy present in Deposit N (Foth & VanDyke, 2000). Two rounds of dredging were conducted at Deposit N, the first during November and December 1998 and the second between August and October 1999, since dredging could not be completed in 1998. Subsequent to the removal of approximately 7,200 cy of sediment from Deposit N, funds and good weather allowed the removal of approximately 1,000 cy from Deposit O in October and November 1999. The overall cost of the demonstration project was $4.3 million, which equates to unit cost of $525 per cy (Foth & VanDyke, 2000).

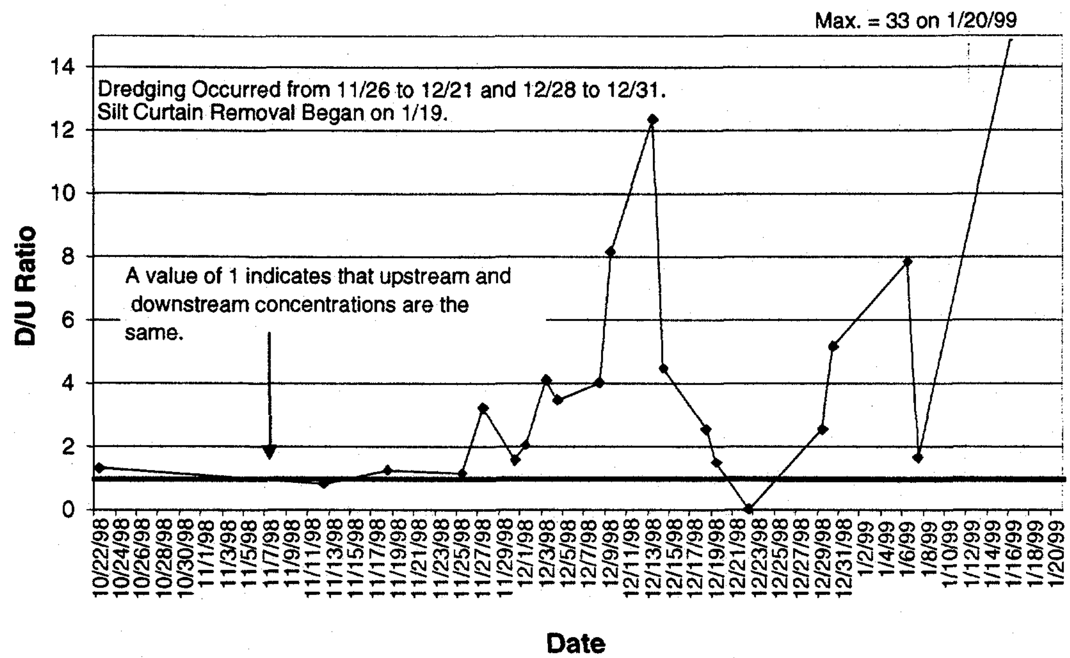


As shown on the above figure, the pre-dredge average surface sediment PCB concentration for Deposit N in 1998, was 16 ppm (BBL, 2000). The 1998 post-dredge average surface PCB concentration was calculated by BBL to be approximately 9 ppm. The 1999 post-dredge average surface PCB concentration is 14 ppm as reported by Foth & Vandyke (2000). Independent calculations by BBL result in a 1999 post-dredge average surface PCB level of 21 ppm.

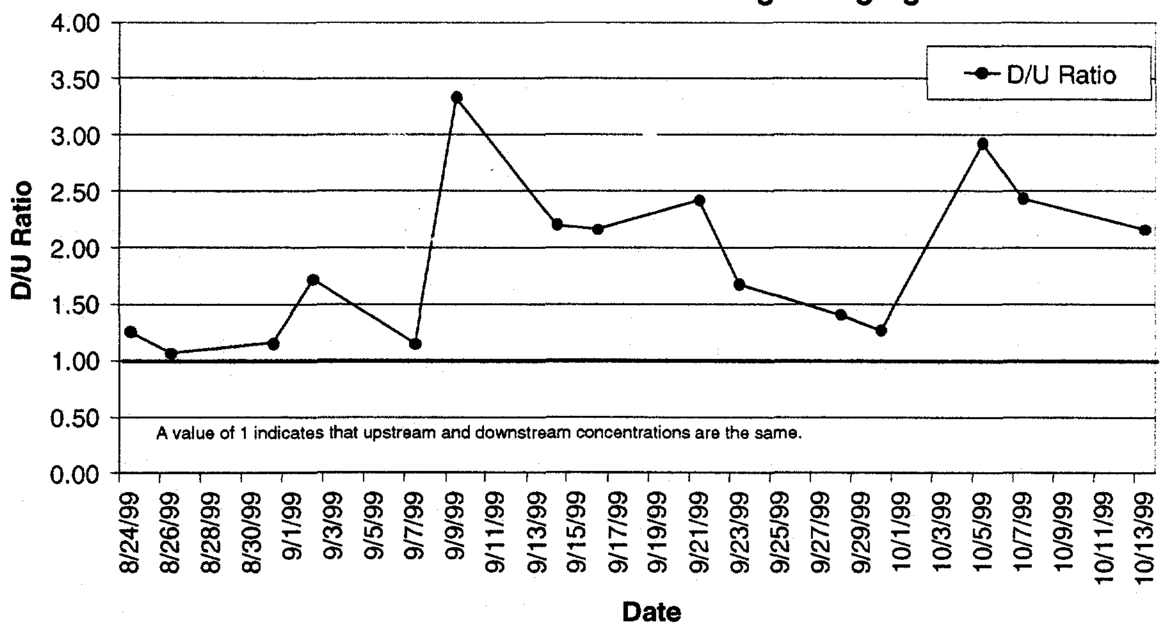
The pre-dredging average sediment thickness was 2 to 3 feet over fractured bedrock in water depths of approximately 8 feet (Foth & VanDyke, 2000). Shallow bedrock at the site prevented over cutting beneath the sediment and resulted in residual sediment left behind. Post-dredge 1999 probing data collected from the west lobe of Deposit N showed that an average of 5 inches of PCB-containing sediment remained, with as much as 15 inches remaining in one portion of the deposit.

Resuspension Data:
Two rounds of dredging were conducted at Deposit N, the first during November and December 1998 and the second between August and October 1999. In 1998, the dredging area was surrounded by a silt containment system including an 80-mil high density polyethylene (HDPE) flexible plastic barrier and a silt curtain. In addition, two deflection barriers were used to direct water around the local paper mill water intake. No turbidity barrier was used during the 1999 dredging. However, a silt curtain was placed approximately 150 feet or less downstream of the dredge (Foth & VanDyke, 2000). Generally speaking, data from both Deposit N dredging events indicate higher PCB concentrations downstream of the dredging site during dredging, while pre-dredging upstream and downstream PCB concentrations are similar.

1998 Water Column Data - Ratio of Downstream To Upstream Total PCB Concentration



1999 Water Column Data - Ratio of Downstream to Upstream Total PCB Concentrations During Dredging



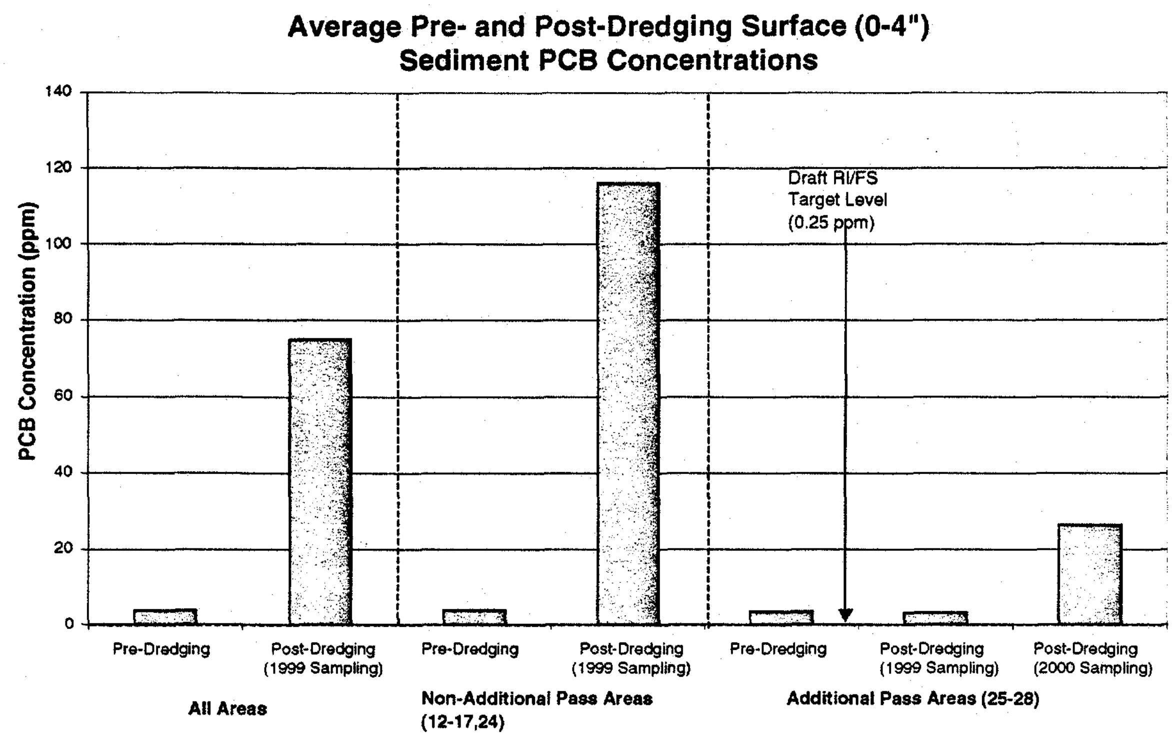
In 1998, the pre-dredging PCB concentrations in upstream and downstream samples were similar, averaging 15 nanograms per liter (ng/L) upstream and 15 ng/L downstream. As indicated in the above figures, evaluating the changes in the downstream to upstream PCB concentration (D/U ratio) indicates that downstream PCB concentrations during dredging exceeded upstream concentrations in both 1998 (by a factor of 1.5 to 12.4) and 1999 (by a factor of 1.1 to 3.3) (BBL, 2000). This trend was not evident in the pre-dredging samples. On average, downstream PCB concentrations were 4.3 times higher than upstream PCB concentrations during 1998 dredging and 1.9 times higher during 1999 dredging (BBL, 2000).

Fox River Sediment Management Unit 56/57 – Green Bay, Wisconsin

Sediment Data:

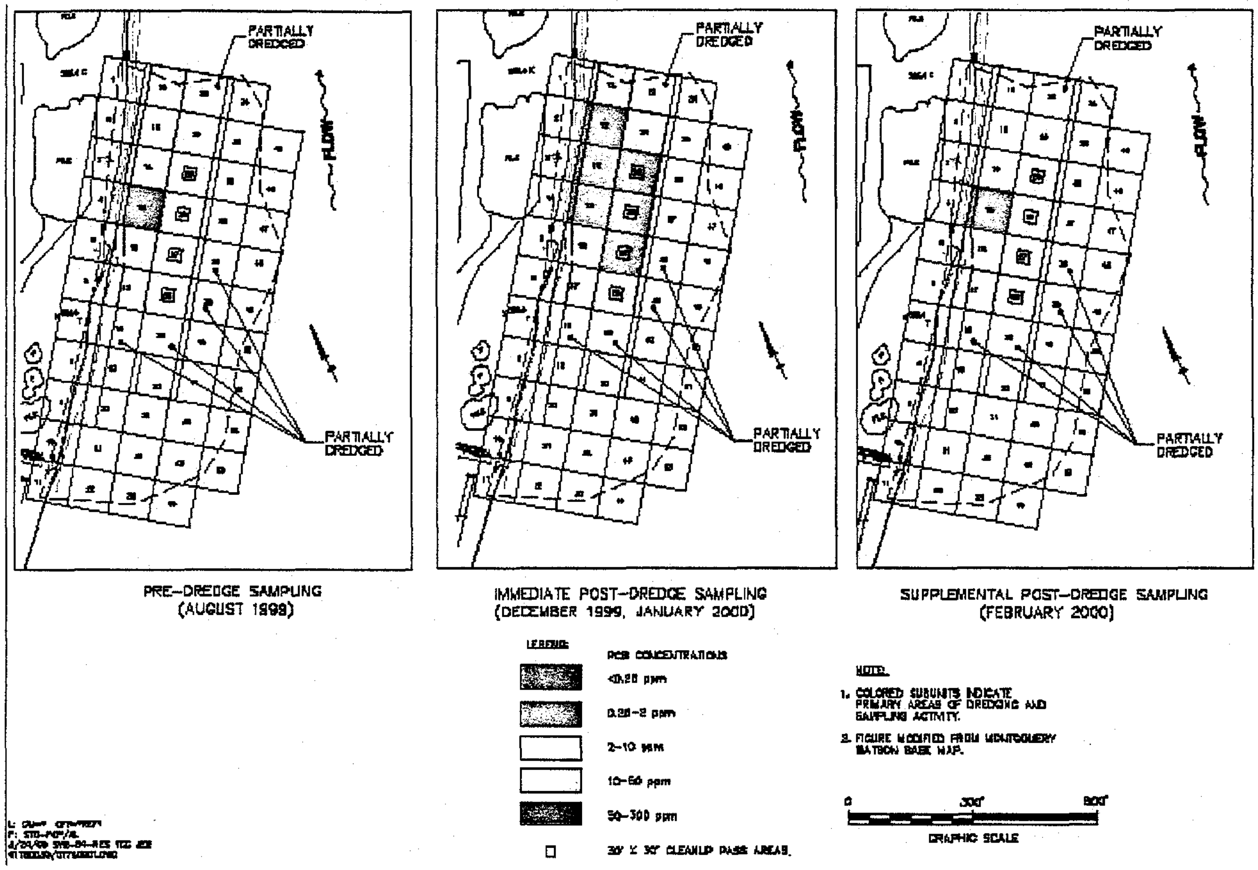
Sediment Management Unit (SMU) 56/57 is a 9-acre area located along the west bank of the Fox River in Green Bay, Wisconsin. Of the 117,000 cy of sediment with PCB concentrations greater than 1 ppm, 80,000 cy were targeted for removal. In August 1999, dredging began and removed approximately 31,500 cy of sediment (mainly from eleven 100-foot by 100-foot subunits) using a hydraulic horizontal auger dredge. The goal of this demonstration project was to understand the implementability, effectiveness and cost of a large-scale sediment removal project. Dredging continued through mid-October 1999, when review of survey information indicated that the dredging process was leaving a very uneven surface on the river bottom. WDNR directed the contractors to stop disturbing new areas and instead redredge areas that had already been disturbed. In December 1999, additional dredging passes were performed on small (30-foot by 30-foot) sections of four subunits designed to remove ridges in the sediment bed left from previous dredging. On average, the additional dredge passes targeted the removal of an additional six inches of sediment.

All of the funds allotted for this demonstration project have been expended with only one-third of the sediment volume removed. The project cost incurred thus far is approximately $9 million, which equates to a unit cost of approximately $317 per cy. However, sediment removal is not yet complete in SMU 56/57.



Pre- and post-dredge PCB data were collected by BBL and Montgomery Watson. Pre-dredge surface PCB concentrations collected in the eleven dredged subunits averaged 3.6 ppm and ranged from 1.7 to 5.9 ppm (BBL, 2000). Two rounds of post-dredging sampling were conducted, the initial round in December 1999/January 2000 immediately following dredging and the second round in February 2000. The average surface PCB concentration in the eleven subunits increased to 75 ppm (range: 0.03 to 280 ppm) based on the December 1999/January 2000 sampling event. A subset of seven of the eleven subunits were sampled during the February 2000 events and the resulting average surface PCB concentration was 43 ppm (range: 16 to 110 ppm).

In those four subunits where an additional "cleanup" pass was performed, pre-dredge surface PCB concentrations were 3.5 ppm (range: 2.7 to 4.7 ppm). In December 1999/January 2000 surface PCB levels decreased slightly to an average of 3.2 ppm (range: 0.03 to 10.8 ppm), while the February 2000 sample results indicated an increase in PCB surface concentration to 26 ppm (range: 16 to 34 ppm) in these four subunits (BBL, 2000).



The pre-dredge surface PCB concentration in those seven subunits that did not receive a cleanup pass was 3.7 ppm (range: 1.7 to 5.9 ppm). Results of the December 1999/January 2000 sampling indicate that average surface PCB concentration in these seven subunits to be 116 ppm (range: 32 to 280 ppm). Only three of these seven subunits were sampled in February 2000 and the resulting average surface PCB concentration was 65 ppm (range: 40 to 110 ppm) (BBL, 2000). Surface sediment concentrations pre-, during- and post-dredging are shown in the above figure.

Dredged sediments were dewatered and disposed (as an in-kind service) at a landfill operated by the Fort James Corporation.

Resuspension Data:
The SMU 56/57 dredge area was enclosed by a silt curtain. PCB levels in the water column were monitored pre-, during- and post-dredging. Generally speaking, PCB concentrations were higher downstream of the removal area than upstream during dredging.

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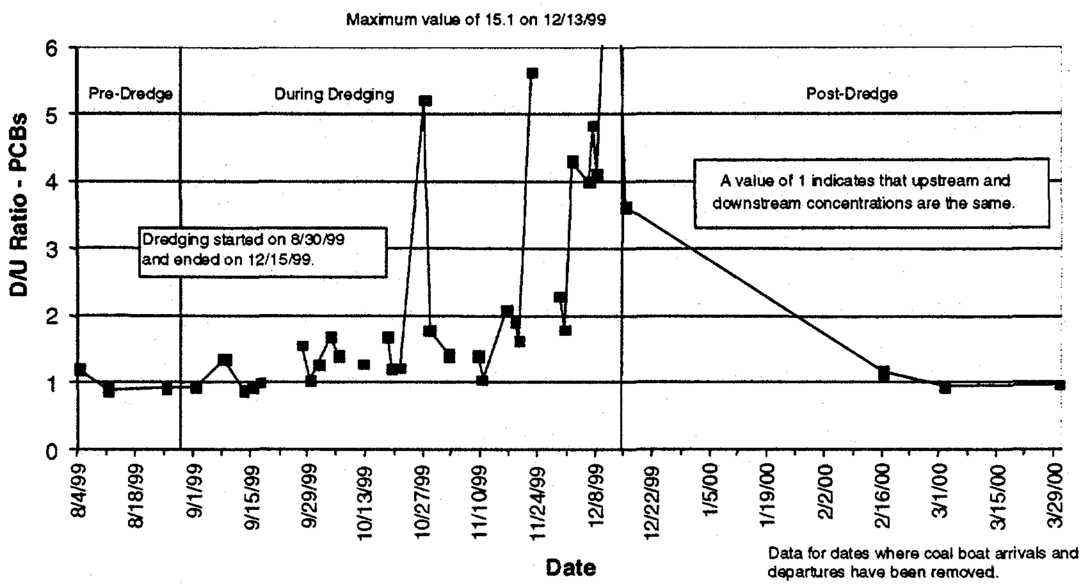
Water Column Data - Ratio of Downstream To Upstream Total PCB Concentration



As shown in the adjacent figure, water column PCB data was analyzed through an evaluation of the downstream to upstream PCB concentration (D/U) ratio. Samples collected during coal boat delivery times were removed to eliminate downstream bias, which may be caused by resuspension due to coal boat travel. The pre-dredging upstream and downstream average PCB concentrations were 53 ng/L and 52 ng/L, respectively (resulting in a D/U ratio of approximately 1.0). The overall during-dredging D/U ratio indicates that, on average, PCB concentration were higher in downstream samples by 2.6 times after removing sampling dates that coincided with coal boat arrivals and departures.

Duwamish Waterway – Seattle, Washington

Sediment Data:

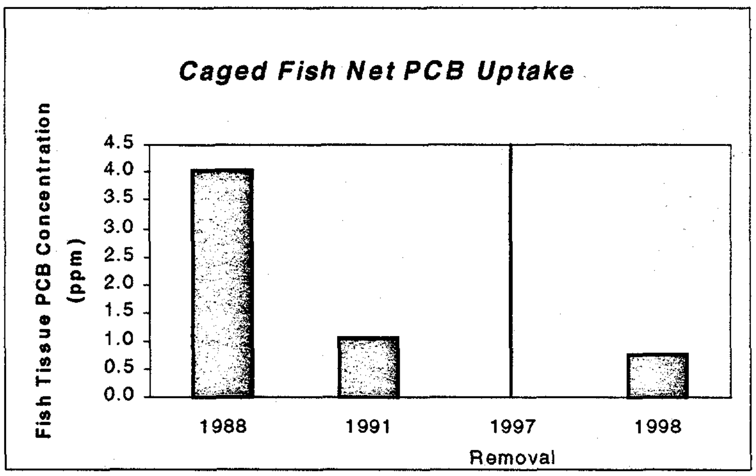
A dredging effort was implemented at Slip 1 of the Duwamish Waterway to cleanup sediment from a 255-gallon PCB spill which occurred on September 12, 1974. Pre-removal PCB concentrations at the spill site were detected in excess of 30,000 ppm (Blazevich, 1977). The first phase of remediation was conducted in October 1974 using divers with hand-held dredges to remove approximately 50 cy of sediment (Willmann, 1976). Post-phase I removal concentrations ranged from 1,200 to 1,900 ppm (Blazevich, 1977). Prior to implementation of Phase II dredging activities in 1976, surficial (top 1 foot) PCB concentrations ranged from non-detect to 42 ppm (average of 4 ppm). Extensive dredging was performed with a PNEUMA pump dredge in an effort to achieve maximum PCB removal near the spill source. After the first dredging pass, sediment PCB concentrations increased to as much as 2,400 ppm. Thus, several passes were employed to achieve maximum removal. According to Willmann (1976), it was originally thought that 4 feet of dredging would be required to sufficiently reduce the concentrations. However, it was found that surface sediment still contained about 200 ppm after 6 feet of material had been removed, so additional dredging to hardpan (a depth of about 10-12 feet) was performed and resulted in residual PCB concentrations of about 10 ppm (Willmann, 1976). Overall, the post-dredge surficial sediment PCB concentrations ranged from 0.2 to 140 ppm (average of 7 ppm), which were higher than the Phase II pre-removal concentrations of non-detect to 42 ppm (average of 4 ppm).

River Raisin – Monroe, Michigan

Sediments were removed from an embayment area of the River Raisin adjacent to a former outfall of the Ford Monroe facility. Approximately 27,000 cy of soft sediment were removed from the embayment between April and October 1997 using a mechanical clamshell operation. A silt containment system was also used at the work area perimeter [Metcalf & Eddy (M&E), 1998].

Sediment Data:
Pre-removal surface concentrations ranged from 11 to 28,000 ppm (average of 4,130 ppm) and subsurface concentrations ranged from 0.78 to 29,000 ppm (average of 6,510 ppm) (M&E, 1993). The cleanup goal for this site was removal of PCBs >10 ppm. Despite removal efforts, potential exposure and risk may not have been reduced because, according to M&E (1998), "confirmatory sample collection activities in many dredge-cells were revealing that sediment remained, even though prior dredging to refusal had occurred." Post-removal PCB levels ranged from 0.54 to 20 ppm (arithmetic average of 9.7 ppm), where only four of the 14 data points were usable for the post-dredging calculation. The other seven had immunoassay results >50 ppm and were redredged; however no sediment reportedly remained from which to obtain a final confirmatory samples. Two of the suspected sources of sediment were "a 0-0.5 foot layer of sediment deposited following resuspension during dredging" and "sloughing of sediment outside of the SRA (sediment removal area) into the SRA along the base of the silt curtain" (M&E, 1998). Cells not meeting the 10 ppm cleanup goal in surficial sediments were redredged until PCBs concentrations were less than 10 ppm in the cells.

Fish Data:
As shown on the figure at right, the Michigan Department of Environmental Quality (MDEQ) performed pre-removal caged fish studies at the mouth of the River Raisin in 1988 and 1991 (remediation occurred in 1997). The total PCB concentration was 4.06 ppm in 1988 and 1.07 ppm in 1991 (MDEQ, 1998). In comparison, the PCB concentration after removal in 1998 was approximately 0.77 ppm. The 1991 concentration was about 25% of the 1988 concentration (a decrease of about 1 ppm/year), and the 1998 concentration was about 72% of the 1991 concentration (a decrease of about 0.04 ppm/year), thus indicating that natural recovery was taking place prior to removal activities and that removal activities did not have a marked effect in reducing the post-removal caged fish concentrations.



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Manistique River and Harbor – Manistique, Michigan

At the Manistique River and Harbor site in Michigan, dredging has been performed in three areas (the North Bay, an area in the River, and the Harbor) to remove PCB sediments. Dredging at the site has been performed using a combination of diver-assisted and hydraulic cutterhead dredging. EPA's goal is to achieve a PCB concentration of 10 ppm at all depths in sediments.

Through the end of 1999, according to the USEPA, a total of less than 100,000 cy of sediment has been dredged and 41,800 tons of dewatered sediments have been shipped to off-site landfills for disposal. The table below summarizes the volumes removed by year.

Year	Volume Removed (cy)	Tons Disposed
1995	10,000 (2)	1,200 (2)
1996	12,500 (2)	2,100 (2)
1997	62,000 (3)	12,000 (3)
1998	31,200 (4)	12,600 (4)
1999	25,000 (5)	13,900 (5)
TOTAL	97,000	41,800

Notes:
1. The volumes are based upon USEPA Pollution Reports; volume to date modified by EPA in 1999 to 72,000 cy through 1998.
2. (2) indicates quantities removed from Area B, POLREP #15 and #20
3. (3) indicates quantities removed from Areas C and D, POLREP #40
4. (4) indicates quantities removed from Area D, POLREP #56
5. (5) indicates quantities removed from Areas B and D, POLREP #70

As of November 1999, the cost for the project is over $35 million. The original budget in 1995 was $15 million.

Initially, EPA expected the dredging to be completed by the end of 1997. Currently, EPA estimates that dredging will be completed by the end of 2000.

Sediment Data:
North Bay (Area B)

Pre-removal surficial sediment PCB concentrations in the North Bay ranged from non-detect to 62 ppm (average of 8.8 ppm) using data collected in 1995.

The EPA originally dredged the North Bay in 1995 and 1996. These activities were initially performed using diver-assisted dredging to remove sediment along with a layer of wood chips. Subsequent removal was then accomplished using a horizontal auger cutterhead dredge. In September 1996, the EPA declared that dredging operations were completed in the North Bay (Nied, 1996a). Post-dredging sampling of the North Bay by EPA in the fall of 1996 revealed that sediment with PCB concentrations greater than 10 ppm remained. In response, the EPA placed washed gravel in the North Bay in October 1996 to "improve the river bottom in this area as habitat for aquatic species as well as enhance containment of the contaminated residuals which could not be cost effectively recovered from beneath the debris layer during dredging" (Nied, 1996b).

In October 1998, BBL collected five sediment cores in the North Bay to confirm whether EPA had reached the 10 ppm PCB cleanup level. PCB concentrations in surficial (0-3 inches) sediment samples ranged from 1.3 to 1,300 ppm, with two of the five detections being greater than 10 ppm, and an overall arithmetic average of 270 ppm. Some of the subsurface intervals sampled also had PCB concentrations greater than 10 ppm. In April 1999, prior to dredging, EPA collected five cores in the North Bay. PCB concentrations in the surficial samples (0- to 1-foot) ranged from 16 to 116 ppm, and averaged 48 ppm. Based on the results of these sampling efforts, EPA decided to conduct additional dredging in the North Bay, which was conducted in May and June 1999.

After the additional dredging had ceased for the season in 1999, BBL collected nine sediment core samples from the North Bay. In the surficial interval (0-3 inches), PCB concentrations ranged from 0.25 to 15 ppm. One sample had a PCB concentration greater than 10 ppm. Six out of 13 subsurface (deeper than 3 inches) samples had PCB concentrations greater than 10 ppm, with a maximum PCB concentration of 620 ppm.

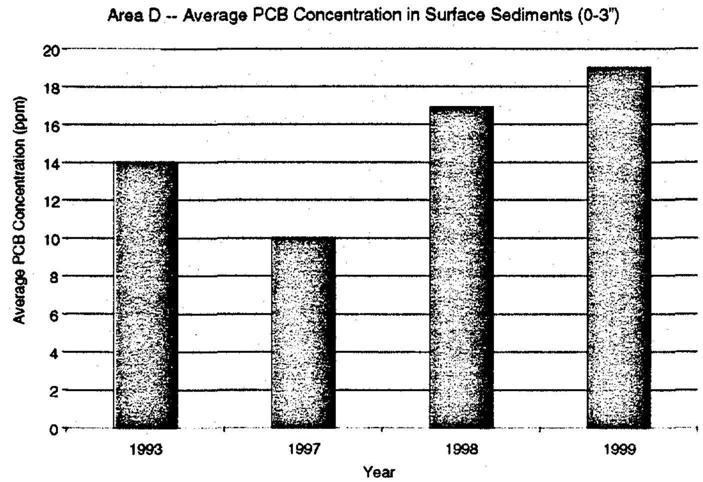
<u>River Area (Area C)</u>

In 1993, an interim geomembrane cap was installed as a temporary measure near an outfall. In 1997, the temporary cap was removed and the sediment was dredged. Sediment PCB concentrations were determined using immunoassay tests to assess whether the clean up goal of 10 ppm was reached. The data document that sediment PCB concentrations remained above 10 ppm. In fact over 20 percent of the samples showed that sediment above 50 ppm was left behind.

<u>Harbor (Area D)</u>

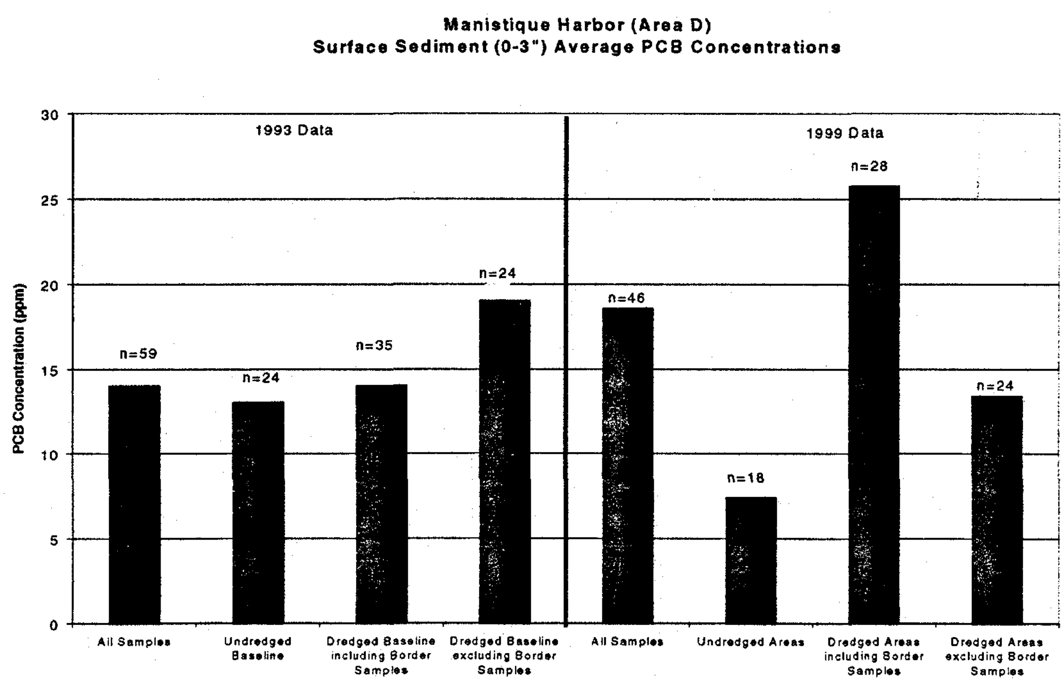
Pre-removal surficial sediment PCB concentrations in the Harbor ranged from non-detect to 340 ppm (average of 14 ppm) using data collected during the Engineering Evaluation/Cost Analysis (EE/CA).

After EPA completed its dredging activities in 1997, 1998, and 1999, BBL collected between 24 and 46 core samples within the harbor. In all years, the samples were distributed throughout the harbor area without bias toward dredged or undredged areas. The average surface sediment PCB data is summarized in the graph below.



In addition, data from 1993 was compared to data from 1999 to determine whether there was any difference between areas which were dredged and those which were not dredged. The delineation of areas dredged (as provided by EPA) was overlaid with the sampling locations in 1993 and 1999 to categorize locations as either within or outside dredged areas.

Given potential mapping inaccuracies, it is possible that some sample locations may be interpretable either way (hereinafter called border samples). Using best judgement, the border samples would be considered within the dredged areas. However, for completeness, both scenarios have the average surface sediment concentrations plotted below.



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The figure shows that while the average PCB concentrations in undredged areas in 1999 roughly two-fold lower than in 1993, this was not the case in dredged areas. The apparent decline in undredged areas may be evidence of natural recovery.

In addition to sampling by BBL, EPA conducted pre-dredging surveys of the Harbor in 1998 and 1999. In 1998, EPA collected 112 samples in the Harbor, and PCB concentrations ranged from non-detect to 1,250 ppm and averaged 16 ppm. In 1999, EPA collected 124 cores in the Harbor. PCB concentrations in the surficial (0- to 1-foot) sediments ranged from non-detect to 1,096 ppm and averaged 30 ppm. The average concentration both years was greater than 10 ppm and increased from 1998 to 1999, generally consistent with BBL data.

EPA continues to have difficulties achieving the 10 ppm cleanup goal in the Harbor. At the end of the 1999 dredging season, EPA collected sediment samples in the Harbor which showed an average PCB concentration greater than 10 ppm. In the 151 grab samples collected by EPA, PCB concentrations ranged from non-detect to 340 ppm and averaged 20 ppm (compared to 19 ppm average using BBL data).

Water Data:
PCB data are available for surface water samples from the Manistique River and Harbor Site from the early 1980s to 1998. In the early 1980s, Marti and Armstrong (1990) collected five surface water samples from the mouth of the River, and in April-May 1994, EPA collected three surface water samples at the site as part of the Lake Michigan Mass Balance Study. These sample results are presented below:

Water Column Total PCB Concentrations (ppb)				
Sampling Period	**Range**	**Mean**	**No. of Samples**	**Reference**
Early 1980s	0.007 - 0.043	0.024 ± 0.015	5	Marti and Armstrong,1990
April/May 1994	0.0002 - 0.0021	0.0009	3	EPA; LMMB Study
1995	ND – 0.49	0.10	102	EPA
1996	ND – 3.5	0.62	23	EPA
1997	ND – 0.81	0.26	10	EPA
1998	ND – 0.14	0.081	17	EPA

The average total water column PCB concentrations in 1994 were an order of magnitude lower than the early 1980s data. Considering EPA's surface water PCB data for 1995 through 1998 (during dredging), the mean PCB concentration was 0.19 ppb (range of 0.042 to 3.5 ppb), an order-of-magnitude or more higher than the pre-remediation concentrations. The annual means are as reported in the table above. Of all the years with water column data, the during-dredging periods show the highest mean PCB detections.

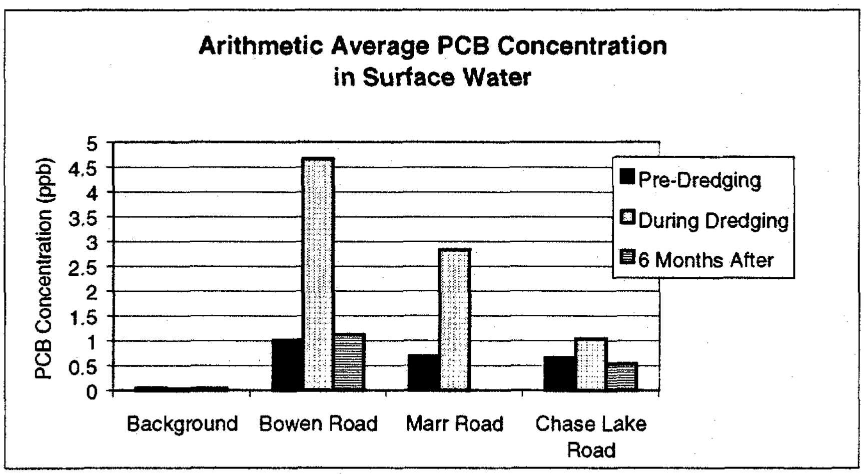
Silt containment has been used during dredging of all three areas. In the North Bay, silt containment included plastic sheeting with wooden shoring at the mouth of the Upper Bay and silt barrier (filter fabric). In the River Area, silt containment included silt barrier constructed from surplus wet felt from a nearby paper mill. In the Harbor, a silt barrier was used for containment.

In 1998, BBL performed sediment trap sampling in Manistique Harbor. The results were generally low; however three of the higher detections observed (9.5, 42, and 84 ppm) suggest resuspension of bottom sediments that may have been due to dredging related activity, including dredged sediment transport by barges to and from the work area. Since no pre-dredging data is available, comparisons with preremoval conditions are not possible.

South Branch of the Shiawassee River – Howell, Michigan

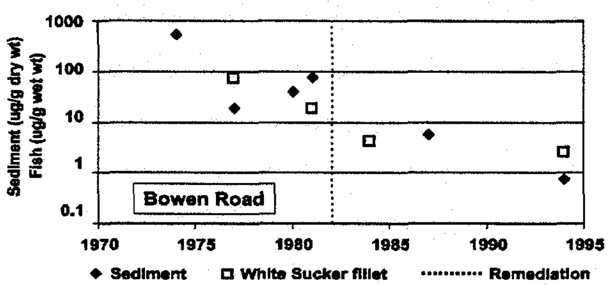
In 1982, a backhoe was used to remove PCB-containing sediment from around a factory discharge, and a dragline was used to remove PCB–containing sediments near Bowen Road, 1.2 miles downstream from the plant site. Small pockets of oily sediments also were vacuumed from this stretch. As discussed by Malcolm Pirnie, "although intended to clean up a total of eight miles of the river, the remediation project stopped at the end of 1982 with only 1.5 miles of river remediated. Cost overruns and the presence of contamination extending farther than initially anticipated were identified as reasons for the incomplete removal action" (Malcolm Pirnie, 1995). No post-removal verification sampling was performed to determine if the 10 ppm cleanup goal was achieved. Only visual and olfactory observations were used to determine the extent of dredging [Environmental Research Group (ERG), 1982].

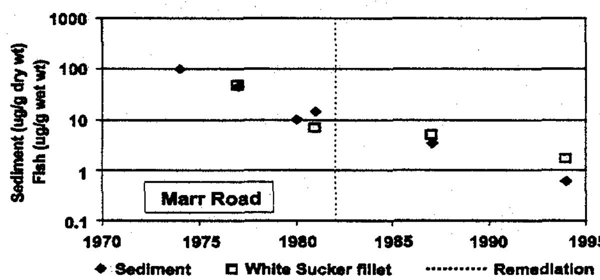
Water Data:
Rice et al. (1984) investigated changes in PCB concentrations in surface water before, during, and after dredging. The results are summarized in the figure below. The two downstream locations show increases in PCB concentrations during dredging; however, the samples collected six months later do not show a significant decrease in PCB concentration when compared to the pre-dredge concen-trations. In fact, it was recognized that "dredging of sediments is likely to cause temporary resuspension of contaminants into the water column which can cause a temporary increase in tissue contaminant concentrations of aquatic biota. Dredging also removed indigenous benthic fauna, which can take years to reestablish" (Malcolm Pirnie, 1995).



Sediment and Fish Data:
The set of graphs presented below show total PCB concentrations in sediment and white sucker fillet samples from the Shiawassee River. Twenty years of data indicate that PCB levels in fish and sediment were undergoing a decline prior to and after the 1982 remediation, which limits the

ability to differentiate the effects of remediation versus other processes such as natural attenuation or source control. Note that data are plotted on a log scale.





To assess the effectiveness of the cleanup, the University of Michigan (UM) performed caged fish and clam studies in the Shiawassee River on behalf of MDEQ (formerly Michigan Department of Natural Resources) before, during, and after the 1982 dredging effort (Rice and White, 1987). At all locations downstream from the plant site and in the area of removal, the UM study indicated an increase in the bioavailability of PCBs following dredging (Rice et al., 1984). For example, at the Bowen Road location (1.2 miles downstream of the source), the PCB levels in caged fathead minnows increased from 64.5 ppm (before removal) to 87.95 ppm dry weight after dredging. PCB concentrations in caged clams collected approximately ¼-mile downstream from the plant site ranged from 13.82 ppm before dredging to 18.30 ppm after dredging, and averaged 59.1 ppm during dredging (Malcolm Pirnie, 1995; Rice et al., 1984), indicating that dredging actually increased exposure rather than decrease it as intended.

Ruck Pond – Cedarburg, Wisconsin

Ruck Pond is one of a series of mill ponds created on Cedar Creek, just upstream of the low-head Ruck Pond Dam. In 1994, an impounded 1,000-foot section of the Creek (Ruck Pond) was drained after a temporary dam was installed on the upstream end and flow was bypassed through siphon piping. The project goal was to remove all soft sediment (contaminated with PCBs) down to bedrock, to the extent practicable.

Sediment Data:
A total of 7,730 cy of sediment was removed by dry excavation and disposed of at commercial landfills. After removal efforts were completed, clean materials used for access to the pond were spread along portions of the pond bottom. Although not intended for capping, these materials inevitably provided some containment of the residual sediment, and likely would have reduced (via burial) the relatively high PCB concentrations remaining at the sediment surface that the dredge equipment could not effectively remove (Praeger, Messur, and DiFiore, 1996).

The maximum PCB concentration measured within the sediments was approximately 150,000 ppm, with an average concentration of 474 ppm (EPA, 1999b). However, sixty soft-sediment surface samples collected from the top 0.5 to 2 feet just before remediation exhibited PCB concentrations ranging from non-detectable to 2,500 ppm (arithmetic average 76 ppm). Despite five months of intensive removal efforts (e.g., use of squeegees attached to a bulldozer blade and vacuum truck), some residual sediment was left on the bedrock surface of the creek bed (Baird and Associates, 1997). Even though 96% of the PCB mass was removed, 7 post-remediation

11.1439

surficial sediment samples exhibited PCB concentrations ranging from 8.3 to 280 ppm (arithmetic average 81 ppm) (Baird and Associates, 1997).

Fish Data:
The WDNR measured whole-body PCB congener concentrations in caged fathead minnows at three locations before and after the sediment removal operation (Amrhein, 1997). Three cages were placed at each of three stations: a site in Cedar Creek upstream of Ruck Pond called Cedarburg Pond, a site within the downstream end of Ruck Pond, and a site downstream of the Ruck Pond Dam, located just upstream of Columbia Dam.

In July 1994, just before the start of removal, PCBs were measured in caged fathead minnows at the three stations. The average PCB concentrations were 0.12 ppm upstream, 24 ppm at the Ruck Pond station, and 12 ppm at the downstream station (7.1, 1,700, and 630 mg/kg lipid normalized PCB, respectively). The average PCB concentrations measured in caged fish in August and September 1995, about one year after remediation, were 0.09 ppm upstream, 4.2 ppm within the pond, and 11 ppm downstream (2.2, 170, and 360 mg/kg lipid normalized PCB, respectively). These PCB levels in the caged fish collected in Ruck Pond would, at face value appear to have declined 75 to 85%[1] on a wet-weight basis and approximately 90% on a lipid basis after remediation. However, caged fish PCB concentrations at the upstream "background" location also declined 25% wet weight and 70% on a lipid basis one year after remediation, and caged fish concentrations downstream of Ruck Pond declined 10% wet weight and 40% on a lipid basis. The declines upstream of Ruck Pond would indicate that other factors, such as natural recovery processes, or metabolism/feeding differences were occurring.

The other more important issue is that construction activities were taking place in the pond (e.g., siphon installation, work boat traffic, etc.) during the pre-remediation sampling. In fact, all three cages in the pond were displaced from their original locations with one cage unrecovered. This all indicates that the pre-remediation cages in Ruck Pond should not be considered representative of pre-remedial conditions.

Waukegan Harbor – Waukegan, Illinois

Waukegan Harbor is approximately 37 acres in size and is located on Lake Michigan approximately 25 miles north of Chicago, Illinois. Remediation areas in the harbor included boat Slip #3 and the 10-acre Upper Harbor. For the Upper Harbor, EPA concluded that, based on modeling, residual sediment PCB concentrations of between 100 ppm and 10 ppm would result in a negligible PCB influx to Lake Michigan. Based on this, EPA set a 50 ppm PCB cleanup level for the Upper Harbor and calculated that 96% of the PCB mass would be removed from the Upper Harbor if the 50 ppm goal was met (EPA, 1984; 1989).

The original goal of the Record of Decision (ROD) was elimination of PCB flux to Lake Michigan (restoration of the harbor fishery was not a specific objective). Regarding the effectiveness of sediment removal, EPA stated in the ROD's Responsiveness Summary that, "Remedial alternatives based on a sediment cleanup level below 50 ppm raise technical and cost-effectiveness concerns. EPA had to consider the technical limitations inherent in the available

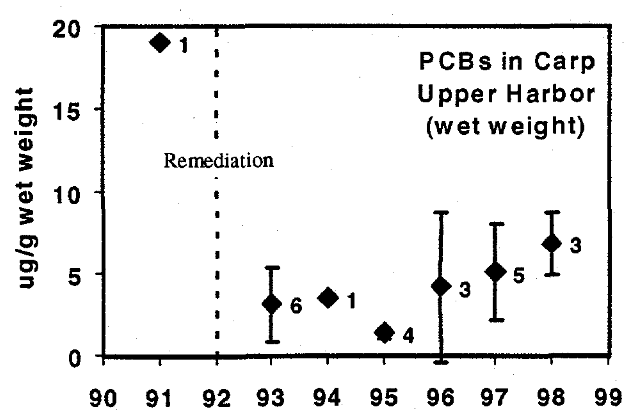
[1] Two exposure periods occurred in Ruck Pond, 29 and 37 days. Average PCB levels were greater in the longer exposure, indicating that the fish were not at steady state with respect to their exposure sources. Therefore, pre-and post-remediation comparisons were carried out independently for each exposure period. The range of values given reflects the two comparisons.

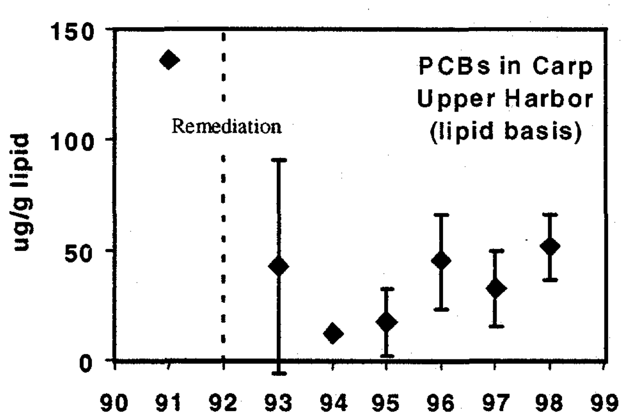
dredging technology. Any dredging technique would involve some resuspension of sediment into the water column, and resettling back into the sediment. It may be difficult to assure that lower sediment levels could be achieved given the technological limitations....As further explained, implementation of the proposed remedy essentially eliminates PCB influx to the Lake from the site."

In late 1991 and early 1992, a total of 6,300 cy of sediment with PCB concentrations greater than 500 ppm were hydraulically dredged from Slip #3, and 32,000 cy were hydraulically dredged from the Upper Harbor. Slip #3 was abandoned and prepared as a permanent containment cell. The 6,300 cy were treated by thermal desorption to remove PCBs and then placed in the cell. The 32,000 cy from the Upper Harbor were pumped from the dredge directly to the cell, and then the cell was capped. The dredging of sediments (primarily organic silts) in 10 acres of the Upper Harbor was completed to a designated depth and to a designated sediment layer such as clay till or sand. Characterization data had shown the underlying clay till and sand layers were only slightly contaminated with PCBs. Sampling was performed during dredging to determine sediment consistency (i.e., to determine if the clay or sand layer had been reached), but not to measure residual PCB concentrations (Canonie Environmental, 1996).

Sediment Data:
No formal post-removal monitoring program was implemented following completion of the dredging, but in April 1996 (over four years after dredging was completed) Illinois EPA reported the results of "... Harbor sediment samples collected to document the effectiveness of dredging." Thirty surface sediment samples (3-inch depth) were collected from 29 locations. Eleven of the samples were archived in a freezer and not analyzed, and two sample bottles were broken in transit. Results for the other 17 samples (one duplicate) showed PCB concentrations ranging from 3 ppm to 9 ppm. Six of the 17 samples were from within the 10 acres of Harbor that were dredged and had PCB concentrations of 5 ppm to 8 ppm.

Fish Data:
Pre-remediation fish data from Waukegan Harbor are extremely limited. For example, only one carp composite sample consisting of two fish and one alewife composite sample consisting of five fish were collected and analyzed in 1991 by the EPA. EPA also concluded that the 1991 alewife data (as well as additional carp data from 1983) should not be used to assess temporal trends because of technical problems associated with the data. Post-remediation data include several fish species collected in the Upper Harbor and in Lake Michigan in the vicinity of the Waukegan Harbor between 1992 and 1998.





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The above figure provides average total PCB concentrations in carp collected from the Upper Harbor (with range representing 2 standard errors). While these graphs seem to indicate that PCB levels were lower in 1993 (compared to 1991), they also indicate a general increasing trend since dredging. The lack of adequate pre-remediation data and the fact that fish tissue concentrations have generally been rising since 1994 indicate the presence of other factors that limit the ability to differentiate the effects of various remedial activities (removal and/or containment) in the harbor. In addition, such a significant drop in PCBs from 1991 is inconsistent with expected trends in tissue PCB levels due to rate of natural depuration of PCBs by fish.

New Bedford Harbor – New Bedford, Massachusetts

In 1976, the EPA detected high concentrations of PCBs in marine sediments over a widespread area of New Bedford Harbor (i.e., PCB concentrations up to 250,000 ppm were reported in 1982). From May 1988 to February 1989, the United States Army Corps of Engineers (USACE) performed a full-scale dredging pilot study at the site to assess the performance of dredge equipment, the suitability for the removal of contaminated sediments, and the recommended procedure for operation (USACE, 1990). Three hydraulic dredges were evaluated: hydraulic cutterhead, horizontal auger (mudcat), and matchbox. The study used two small shallow (water depth less than 5 feet) dredging areas, and approximately 10,000 cy of sediments were removed (USACE, 1990).

Sediment Data:
Prior to removal, both test areas contained higher concentrations in the surface (top 6-inch) sediments (i.e., average of 226 ppm in Area 1 and 385 ppm in Area 2) compared to subsurface concentrations, which were one to three orders of magnitude lower. Post-removal average residual sediment (top 3-inches) concentrations for each of the dredges tested were as follows:

- cutterhead (Area 1): 80 ppm;
- horizontal auger (Area 1): 66.4 ppm;
- cutterhead (Area 2): 8.6 ppm; and
- matchbox (Area 2): 5.4 ppm.

Note that a theoretical versus actual residual PCB concentration evaluation also was performed, which showed that actual post-removal concentrations were much higher than those theoretically predicted.

Following performance of the Pilot Study, the remediation for the New Bedford site was split into two operable units. EPA issued a ROD for the first operable unit (hot-spot areas, those areas with greater than 4,000 ppm PCBs) in April 1990. The 1990 ROD called for dredging of approximately 10,000 cy of sediment with PCB concentrations greater than 4,000 ppm, dewatering (with effluent treatment), incineration of dewatered sediment, and stabilization of the incineration remains (EPA, 1990a). The dredging portion of this phase was initiated in April 1994 and was completed in September 1995. Over the 1994-1995 construction period, a total of about 14,000 cy were dredged and placed in a confined disposal facility (CDF) nearby, pending determination of final treatment and/or disposal. Pre-dredging surficial sediment samples (upper 2 feet) had PCB concentrations ranging from 4,000 to 200,000 ppm, with an arithmetic average of 25,000 ppm (EPA, 1999a). Initial post-dredging sampling showed up to 3,600 ppm PCBs remained after dredging (personal communication with P. L'Hreaux of USACE, 1996). After the completion of the project, it was estimated by Ebasco Services and the EPA, that only about 45% of the PCBs in the Harbor had been removed by dredging (EPA, 1997).

Water Data:
Water-column monitoring was performed during the hot spot removal initiated in 1994 to assess and limit the amount of cumulative transport of PCBs to the lower harbor. For the entire removal operation, EPA calculated that a mass of approximately 57 kg (24% of the maximum allowable cumulative transport) was transported into the lower harbor (EPA, 1997).

Air Data:
During dredging operations, ambient air PCB concentrations were monitored at 16 monitoring locations to characterize impacts from dredging operations. If the airborne PCB concentrations exceeded predetermined action levels (i.e., 0.05, 0.5 or 1 ug/m^3), then modifications or additions of engineering controls were implemented to dredging operations, with respect to severity. Of 4,041 total samples collected over the course of remedial actions, 1,063 (26 %) exceeded the 0.05 ug/m^3 action level, 49 (1 %) exceeded the 0.5 ug/m^3 action level, and 10 (0.25 %) exceeded the 1 ug/m^3 action level. Due to the exceedences, operational changes were implemented to minimize airborne PCB levels, leading EPA to conclude that "control of airborne PCB emissions did contribute to a slower rate of dredging and thus a longer project duration" during the hot spot removal operation (EPA, 1997).

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Appendix C

Effectiveness of Sediment Removal: An Evaluation of EPA Region 5 Claims Regarding Twelve Contaminated Sediment Removal Projects

EFFECTIVENESS OF SEDIMENT REMOVAL:

An Evaluation of EPA Region 5 Claims Regarding Twelve Contaminated Sediment Removal Projects

Submitted to the:

National Academy of Sciences
Committee on the Remediation of PCB-Contaminated Sediments

Submitted by:

The Fox River Group
as a supplement to the presentation by:
J. Paul Doody, P.E.
Blasland, Bouck & Lee, Inc.

September 27, 1999
Green Bay, Wisconsin

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SECTION 1 – INTRODUCTION

Representatives from Region 5 of the U.S. Environmental Protection Agency (EPA) have published articles and made a series of public presentations promoting the "success" of 12 contaminated sediment removal projects.[1] A close examination of the conclusions drawn by EPA Region 5 raises serious concerns about both the accuracy of the claims and the absence of adequate supporting data to substantiate the claims. For example, in one case broad conclusions are drawn from a single pre-dredging data point; in other cases conclusions are made without regard to sampling location, time, age of fish, length of exposure, or a variety of other parameters; and in still other cases conclusions are advanced by choosing some data points and not others. Despite these weaknesses, EPA presents its findings as conclusive without properly qualifying those conclusions based on known uncertainties and limitations of the underlying data.

EPA cites the 12 projects listed in Table 1 as proof that sediment removal is effective in all cases. If anything, however, these projects prove that remedies can be evaluated only on a site-specific basis. For example, can the Shiawassee River project (removal of just 1,805 cubic yards over 15 years ago) or Ruck Pond (a dry excavation while Cedar Creek was diverted through pipes) really be cited as relevant precedents for selecting appropriate remedies for large and complex river systems? Does mass removal make sense as a general rule when each of the projects cited by EPA demonstrates that contaminants are always left behind to one degree or another after dredging? The standard after all is risk reduction – not mass removal – as reflected in CERCLA, 42 U.S.C. 9605(a)(8)(A), and EPA guidance documents. EPA's Contaminated Sediment Management Strategy (EPA, 1998) requires that EPA "consider a range of risk management alternatives" to reduce risk, including source control, natural attenuation, containment, and removal alternatives.

Focusing on risk reduction, as opposed to mass removal, may make decisions more challenging and complex, but an appropriate understanding of the factors driving risk in aquatic systems (e.g., the availability of contaminants in the biologically active zone of surface sediments) is necessary to improve the health of our lakes, rivers, and harbors. Dredging may very well have its place in certain circumstances, but from a national policy perspective, the focus has to be on the proper management of sediment to reduce risk. These decisions will have to be made on a case-by-case basis reflecting the unique characteristics of each affected water body and the unique physical conditions influencing current and future exposure potential within each system.

[1] For example, EPA's presentations have included "USEPA Sediment Cleanups: Results and Costs of Dredging Projects," given during a televised public information forum called "The ABCs of PCBs" (hosted by the Appleton, Wisconsin chapter of the League of Women Voters), and a June 7, 1999 presentation to the National Academy of Sciences Committee on the Remediation of PCB-Contaminated Sediments. Portions of the presentation materials and related information have been published by EPA Region 5 staff in an article titled "Long-Term Benefits of Environmental Dredging Outweigh Short-Term Impacts," written by James J. Hahnenberg and appearing in *Engineering News Record* (Hahnenberg, 1999).

Table 1 - Sediment Removal Projects Evaluated by EPA Region 5	
Project Name/Location	**Sediment Removed (cubic yards)**
Dredging Projects	
Black River, OH[1]	60,000
Lake Jarnsjon, Sweden	196,000
Manistique River/Harbor, MI[2]	72,000
River Raisin (Ford Outfall), MI	28,500
St. Lawrence River (GM Massena), NY	13,300
Sheboygan River, WI	3,800
Shiawassee River, MI	1,805
Waukegan Harbor, IL	38,300
Dry Excavation Projects	
Bryant Mill Pond, MI	165,000
Ottawa River Tributary, OH	8,000
Ruck Pond, WI	7,730
Willow Run Creek, MI	450,000

[1] Contaminant of concern is PAHs, not PCBs.

[2] In progress; value is total volume removed through the end of the 1998 construction season, as reported by EPA.

Although there is limited monitoring data for the 12 projects cited by EPA, scientists and engineers from Applied Environmental Management, Inc. (AEM), Blasland, Bouck & Lee, Inc. (BBL), and others undertook an evaluation to: 1) identify and reconstruct how EPA may have reached its findings (primarily the claims of several-fold reductions in fish tissue concentrations as a result of sediment removal) and 2) provide a critical review of EPA's claims using all data available in our files and the "Major Contaminated Sediment Sites Database" (AEM, 1999) for the highlighted projects. As noted in Table 1, eight of the 12 projects involved dredging technology, and four relied upon dry excavation techniques. Eleven of the 12 projects targeted polychlorinated biphenyls (PCBs) for remediation, and one targeted polycyclic aromatic hydrocarbons (PAHs).

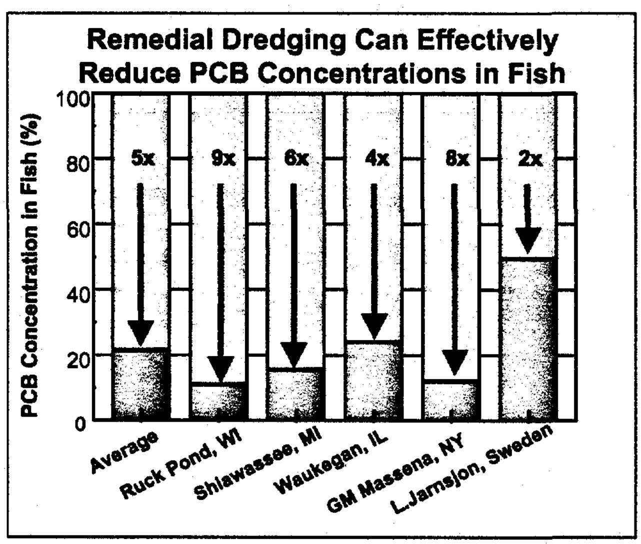

Figure 1 - This is a reproduction of the summary graphic presented by EPA Region 5 during a June 1999 presentation to the NAS Committee on the Remediation of PCB-Contaminated Sediments. The five projects cited formed the basis for EPA's claim that sediment removal resulted in an average 5-fold decrease in fish PCB concentrations.

Section 2 of this paper focuses on EPA's use of fish tissue data as the basis for reaching conclusions regarding the effectiveness of sediment removal, and provides a detailed review of the five case study projects EPA discussed during its June 1999 presentation to the National Academy of Sciences (NAS) Committee on the Remediation of

PCB-Contaminated Sediments (EPA's summary figure is reproduced for reference as Figure 1). In response to that presentation, J. Paul Doody, P.E., a principal engineer at BBL, presented a summary of our evaluation of the five case studies to the NAS committee during its meeting in Green Bay, Wisconsin on September 27, 1999. The five case study projects are the Shiawassee River in Michigan, Lake Jarnsjon in Sweden, Waukegan Harbor in Illinois, the St. Lawrence River in New York, and Ruck Pond in Wisconsin.

Section 3 presents our review of three other broad conclusions made by EPA Region 5 regarding the effectiveness of sediment removal: 1) contaminant mass removal is the primary measure of remedial success, 2) short-term adverse impacts of dredging are minor, and 3) unit costs tend to decrease with increasing scale of sediment removal.

Section 4 presents an overall summary of this paper and our findings.

SECTION 2 – EVALUATION OF EPA REGION 5 CASE STUDY PROJECTS

EPA Region 5's claims of reductions in contaminant concentrations in fish tissue are based on three hydraulic dredging projects (Lake Jarnsjon, Waukegan Harbor, and St. Lawrence River/GM Massena), one mechanical dredging (i.e., wet excavation) project (Shiawassee River), and one mechanical "dry" excavation project after the overlying water column was drained (Ruck Pond). A careful evaluation of the facts for these five case studies provides findings substantially different from those of EPA. This section presents our review of how EPA Region 5 may have reached its conclusions and offers alternative findings and supporting rationale that are apparent from the five projects. We reached three primary conclusions as a result of our evaluation:

- EPA has not demonstrated that the sediment removal actions at the cited projects reduced PCB exposure and risk.
- Reduction of PCB concentrations in fish is a meaningful measure of risk reduction, but the uncertainty associated with limited data availability, data quality concerns, and EPA's selective use of data do not support EPA's conclusions regarding the effects of sediment removal on fish at these sites.
- EPA's analysis does not differentiate the effectiveness of sediment removal from that of several other factors such as source control, containment, capping, or natural attenuation.

Our basis for reaching these conclusions is discussed below within the context of the five case studies highlighted by EPA Region 5.

2.1 – Shiawassee River, Michigan

This Superfund site includes the former Cast Forge Steel Company aluminum die-cast facility and 8 miles of the South Branch Shiawassee River in Howell, Michigan. The South Branch is 15 to 30 feet wide, with a depth of several feet and a floodplain ranging from approximately 100 to 300 feet wide. The river features numerous bars and mud flats, as well as moderate scour areas. Considerable blockage occurs as a result of deadfalls and beaver activities. The waterway is a small river with nominal flow of approximately 15 cubic feet per second (cfs) and spring floods reaching 75 cfs.


The Shiawassee River, looking upstream from Bowen Road, which is approximately 1.2 miles downstream of the Cast Forge Plant – the reach that was remediated in 1982.

The Shiawassee River received discharges of PCBs in hydraulic fluid and wastewater until the 1970s. A Consent Judgment in 1981 led to a removal action in the river with a cleanup goal of 10 parts per million (ppm) PCBs. In 1982, a backhoe was used to remove PCB-containing material from around the discharge area at the plant site and a dragline was used to remove contaminated sediments from an area in the river near Bowen Road, which is about 1.2 miles downstream of the Cast Forge facility. In addition, small pockets of stream sediments exhibiting an oily appearance were vacuumed from this 1.2-mile reach of the river (ERG, 1982). The remedial action resulted in removal of 1,805 cubic yards of sediments, but no sediment samples were collected to verify achievement of the cleanup goal. Removal was stopped at the end of 1982 due to exhaustion of funds and the presence of PCB contamination extending farther downstream than anticipated.

To assess the effectiveness of the cleanup, University of Michigan researchers measured PCB concentration changes in fish and surface water and evaluated the potential for bioaccumulation of PCBs in the river ecosystem (Rice and White, 1987). Caged fish and clam studies were performed in the river before, during, and after remediation. At all locations downstream from the plant site and in the area of removal, the study indicated an increase in the bioavailability of PCBs following remediation. At the Bowen Road sampling location, for example, the concentration of PCBs (dry weight) in caged fathead minnows increased from 64.5 milligrams per kilogram (mg/kg) to 88 mg/kg after remediation. This increase in concentration was cited as a short-term impact in EPA presentations, but the increase points to the likelihood that the residual PCBs remaining at the sediment surface after dredging increased exposure.

EPA's presentation of its evaluation is limited to just one chart comparing 1981 pre-remediation fish data with 1994 post-remediation data. This approach omits important information such as species and age of fish, type of analysis (fillet or whole body), location in the river, whether the reported concentrations were discrete values or averages, and fish tissue data from years other than 1981 and 1994.[2] EPA Region 5 relies on limited fish data collected 13 years apart, which ignores other available data, and attempts to use these selective data to illustrate a long-term 6-fold reduction in fish tissue concentrations resulting from the 1982 removal project. EPA's approach is misleading and greatly oversimplifies the rigorous approach that this kind of data analysis requires.

To provide a more careful evaluation and to fill in the missing information, we consulted two documents prepared for the Michigan Department of Natural Resources (MDNR): a remedial investigation (RI) report for the South Branch (Warzyn, 1992) and a report to develop sediment quality objectives for PCBs (Malcolm Pirnie, 1995). These documents provide a great deal of additional data on sediment and fish tissue PCB concentrations over a period of years. Table 2 provides a summary of that data. Note that the fish tissue data are for white sucker, which was the only species of fish sampled during each sampling event between 1977 and 1994.

[2] In fact, this type of important qualifying information was typically missing from the charts presented by EPA Region 5 for each of the five case studies cited as demonstrating reductions in fish tissue concentrations.

Table 2 - Average PCB Concentrations in Sediments and Caged Fish (white sucker) from the South Branch Shiawassee River				
	Bowen Road (1.2 miles downstream of plant)		**Marr Road** (3.4 miles downstream of plant)	
Year	**Sediment (mg/kg dry wt.)**	**Fish (mg/kg wet wt.)**	**Sediment (mg/kg dry wt.)**	**Fish (mg/kg wet wt.)**
1974	530	--	97	--
1977	18.6	76	44	47
1980	40[1]	--	9.9[1]	--
1981	75[1]	19	14	6.7
1982	*Remediation performed*			
1984	--	4.2	--	--
1987	5.7[1]	--	3.3[1]	5
1994	0.72	2.56	0.59	1.7

[1] Average of duplicate samples. All other entries are average values as reported in Malcolm Pirnie (1995).
Data source: Malcolm Pirnie (1995) Tables 2-1 and 2-2.

The data reveal that at Marr Road, which is 3.4 miles downstream from the plant and about 2 miles downstream of Bowen Road, PCB concentrations in white sucker samples averaged 47 mg/kg in 1977, but declined to 6.7 mg/kg in 1981 *before remediation took place*. In 1987, five years after sediment removal, remediation did not appear to have had much effect in reducing white sucker PCB concentrations beyond rates already under way from other causes – average concentrations decreased from 6.7 mg/kg in 1981 to 5 mg/kg in 1987 (declines continued through 1994 as well). Similar trends are seen in sediment concentrations at both locations. The RI report (Warzyn, 1992) attributes the reductions in white sucker PCB concentration primarily to natural attenuation, although it is important to note that source control measures implemented at the plant in the late 1970s and early 1980s likely contributed to the observed declines.

Between the plant and Bowen Road, the 1.2-mile reach where remediation took place, dredging may have had some impact on reducing white sucker PCB concentrations. The data for the Bowen Road sampling station show that natural recovery processes were reducing PCB concentrations substantially prior to 1982. However, it is possible, but far from certain as EPA would have one believe, that dredging contributed to the reductions in sediment and fish tissue PCB concentrations seen after 1981 at either the Bowen Road or Marr Road locations.

The uncertainty regarding whether any reductions in fish tissue concentrations occurred due to sediment removal is best illustrated by the trends evident on Figure 2. The graphs for both Marr Road and Bowen Road depict trends that are approximated by straight lines (note log scale), and there is no pronounced acceleration in the reduction of fish tissue concentrations related to the remediation event in 1982. The data could just as well be used to support claims of approximately 6-fold reductions at Marr Road and 4-fold declines at Bowen

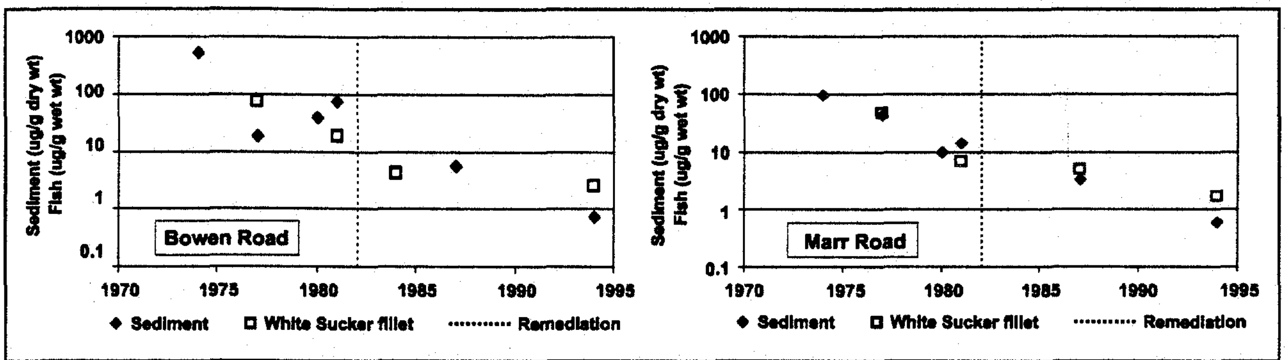

Figure 2 - Total PCB concentrations in white sucker fillet and sediment samples from the Shiawassee River. Twenty years of data indicate that PCB levels in fish and sediment were undergoing a decline prior to and after the 1982 remediation, which limits the ability to differentiate the effects of remediation versus other processes such as natural attenuation or source control. Note that data are plotted on a log scale.

Road between 1977 and 1981 *due to natural attenuation*.[3]

EPA Region 5 is overreaching when it states that the data show a 6-fold decline in fish tissue concentrations due to sediment removal, and EPA apparently compared just two data points, 1981 and 1994, to support its claim. When the entire data set is considered, as we have done here, the data do not support the conclusion that sediment removal at the Shiawassee River – *any more than natural attenuation* – was responsible for reductions in fish tissue concentrations. Moreover, the data provide no basis for any claim regarding the *extent* to which reductions in fish tissue concentrations are attributable to sediment removal.

2.2 – Lake Jarnsjon, Sweden

Lake Jarnsjon is a 62-acre lake located 72 miles upstream of the mouth of the Eman River in Sweden. PCBs were discharged to the lake from a paper mill that had used recycled paper as raw material. In 1991, 12 core samples from the top 40 cm of lake sediment had PCB concentrations ranging from 0.4 mg/kg to 30.7 mg/kg. Sediment, biota, and water column measurements in the late 1980s and early 1990s indicated elevated PCB levels in fish and an average annual loss of 12 to 15 pounds of PCBs from the lake to the downstream river.

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[3] A similar conclusion was drawn by MDNR's consultant, Warzyn (1992), who stated that "the 1982 remediation in the reach of the River upstream of Marr Road did not substantially affect the PCB concentration of the edible portion of white suckers." "It appears that the remediation had an effect on PCB concentrations in white suckers near Bowen Road. It was also apparent that the natural spreading of PCBs by sediment transport between 1974 and 1981 substantially decreased the concentration of PCBs in fish from both locations (Marr and Bowen Roads). Without remediation, PCB concentrations were slowly dropping over time in fish at Bowen Road."

Lake Jarnsjon in Sweden during dredging of the enclosed eastern part of the lake. 196,000 cubic yards of sediment were removed in 1993 and 1994. Photo: T. Svahn.

In response to these findings, the entire lake bottom was dredged in 1993-94 to remove PCBs to a target concentration of 0.5 ppm or less (196,000 cubic yards of sediment were removed). Removal depths ranged from 40 cm (1.3 feet) to 160 cm (5.25 feet). Sediment was disposed of in a nearby dedicated landfill. Based on pre- and post-remediation sediment samples, an estimated 97% of the PCBs were removed. Sixty-two post-dredging surface sediment samples collected from across the lake exhibited PCB concentrations ranging from 0.01 mg/kg to 2.4 mg/kg (most were five-part composites collected from depths of 0 to 20 cm).

Table 3 and Figure 3 present summaries of PCB data for several locations on the Eman River, including Lake Jarnsjon. Two years after remediation ended in 1994, average PCB concentrations in Lake Jarnsjon surface water had decreased to 2.7 nanograms per liter (ng/L) in 1996 from 8.6 ng/L in 1991. Similarly, average PCB concentrations in year-old perch from the lake, fish that would have hatched in the summer after remediation, declined from 36 mg/kg lipid in 1991 to 16 mg/kg lipid in 1996, which is apparently the 2-fold reduction claimed by EPA Region 5. However, measurements taken downstream and at upstream reference stations showed that PCB levels in water and fish were already declining throughout the 1990s.

Table 3 - Average PCB Concentrations in Fish and Surface Water from the Eman River, Sweden

Approx. River Kilometer	One-Year-Old Perch (mg/kg lipid)			Surface Water (ng/L)		
	Station	1991	1996	Station	1991	1996
-35	1	1.4	0.9	2	0.7	0.2
-10	3	9.1	6.1	4	1.2	0.9
0 (Lake Jarnsjon)	5	36	16	5	8.6	2.7
+20	--	--	--	6	5.1	2.3
+80	7	6.7	5.2	8	1.3	1.1

Data sources: geometric means reported by Bremle et al. (1995) and Bremle and Larsson (1998).

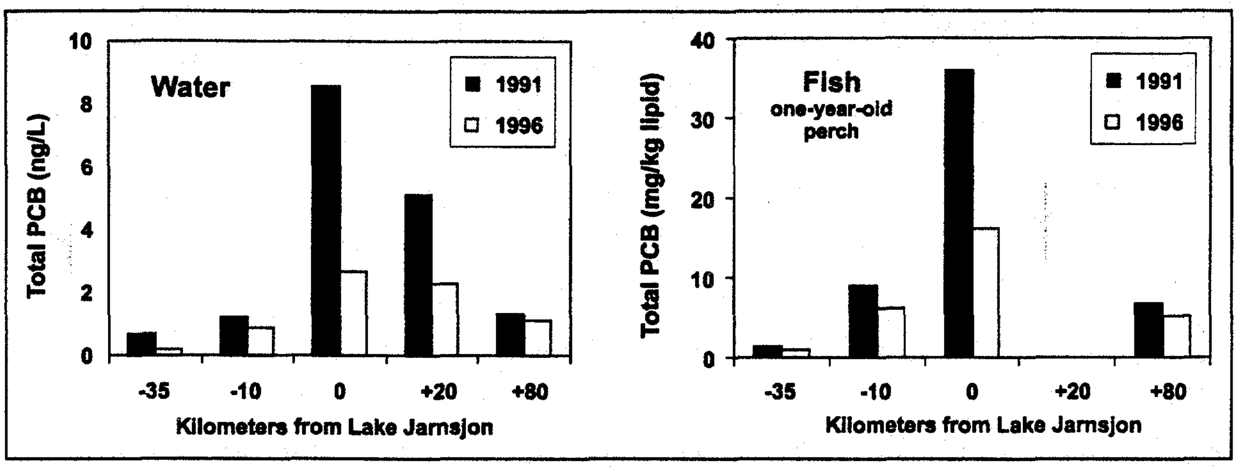
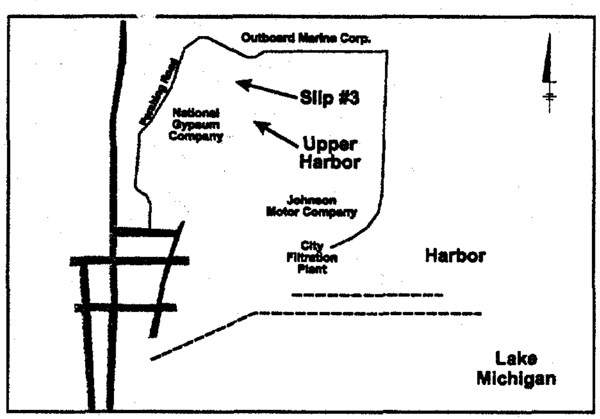


Figure 3 - Total PCB concentrations in fish (one-year-old perch) and water from the Eman River, comparing 1991 pre-remediation levels with 1996 post-remediation levels. Spatial trends are also apparent and indicate that while PCB concentrations decreased by approximately 50% in Lake Jarnsjon, upstream and downstream concentrations were also on the decline likely due to ongoing system-wide natural recovery processes. Finally, it is apparent that even after dredging the entire bottom of Lake Jarnsjon, lake sediments remained a dominant source of PCBs to fish and the water column.

Despite large-scale dredging, PCB levels in fish and surface water after remediation remained greatest in the lake as compared to both upstream and downstream locations. This means that the sediments of the lake remained an important source of PCBs to fish despite dredging an estimated 97% of PCB mass from the entire lake bottom. Taken together, these data indicate that the decline measured at Lake Jarnsjon, and the Eman river as a whole, is at least partly due to system-wide natural recovery processes operating both before and after remediation (Bremle and Larsson, 1998). These observations limit the ability to differentiate the effects that dredging may have had versus the apparent natural recovery processes operating within the system, and call into question the basis of EPA Region 5 claims about the project.

2.3 – Waukegan Harbor, Illinois

Waukegan Harbor is approximately 37 acres in size and is located on Lake Michigan approximately 25 miles north of Chicago, Illinois. Areas targeted for remediation in the harbor included boat Slip #3 and the 10-acre Upper Harbor (see map). For the Upper Harbor, EPA concluded that, based on modeling, residual sediment PCB concentrations of between 100 ppm and 10 ppm would result in a negligible PCB influx to Lake Michigan. Based on this, EPA set a 50 ppm PCB cleanup level for the Upper Harbor and calculated that



Waukegan Harbor is located on Lake Michigan north of Chicago, Illinois. In 1991 and 1992, the Upper Harbor and Slip #3 were remediated.

96% of the PCB mass would be removed from the Upper Harbor if the 50 ppm goal was met (EPA, 1984; 1989).

The original goal of the Record of Decision (ROD) was elimination of PCB flux to Lake Michigan (restoration of the harbor fishery was not a specific objective). Regarding the effectiveness of sediment removal, EPA stated in the ROD's Responsiveness Summary that, "Remedial alternatives based on a sediment cleanup level below 50 ppm raise technical and cost-effectiveness concerns. EPA had to consider the technical limitations inherent in the available dredging technology. Any dredging technique would involve some resuspension of sediment into the water column, and resettling back into the sediment. It may be difficult to assure that lower sediment levels could be achieved given the technological limitations...As further explained, implementation of the proposed remedy essentially eliminates PCB influx to the Lake from the site."

In late 1991 and early 1992, a total of 6,300 cubic yards of sediment with PCB concentrations greater than 500 ppm were hydraulically dredged from Slip #3, and 32,000 cubic yards were hydraulically dredged from the Upper Harbor. Slip #3 was abandoned and prepared as a permanent containment cell. The 6,300 cubic yards were treated by thermal desorption to remove PCBs and then placed in the cell. The 32,000 cubic yards from the Upper Harbor were pumped from the dredge directly to the cell, and then the cell was capped. The dredging of sediments (primarily organic silts) in 10 acres of the Upper Harbor was completed to a designated depth and to a designated sediment layer such as clay till or sand. Characterization data had shown the underlying clay till and sand layers were only slightly contaminated with PCBs. Sampling was performed during dredging to determine sediment consistency (i.e., to determine if the clay or sand layer had been reached), but not to measure residual PCB concentrations (Canonie Environmental, 1996).

No formal post-removal monitoring program was implemented following completion of the dredging, but in April 1996 (over four years after dredging was completed) Illinois EPA reported the results of "...Harbor sediment samples collected to document the effectiveness of dredging." Thirty surface sediment samples (3-inch depth) were collected from 29 locations. Eleven of the samples were archived in a freezer and unanalyzed, and two sample bottles were broken in transit. Results for the other 17 samples (one duplicate) showed PCB concentrations ranging from 3 mg/kg to 9 mg/kg.[4] Six of the 17 samples were from within the 10 acres of harbor that were dredged and had PCB concentrations of 5 mg/kg to 8 mg/kg. However, these 1996 sediment data are of limited value because no information is presented on physical characteristics of the

[4] The 17 samples were also analyzed for other parameters. The report (Lesnak, 1997) states that all sediment samples contained arsenic (11 to 120 mg/kg), copper (46 to 228 mg/kg), and lead (45 to 188 mg/kg) at levels that classify them as "heavily polluted" based on the guidelines for pollution classification of Great Lakes harbor sediments. Metals, however, were not a consideration in the 1984 ROD or the 1989 ROD Amendment.

samples, and no attempt was made by EPA to compare these results with historical results from the same sample stations. The Illinois EPA assessment report does not attempt to draw conclusions as to the meaning of these results or the success or failure of the remedial dredging, nor does it define any follow-up sampling or other actions (Lesnak, 1997).

EPA and Illinois EPA generated a great deal of publicity regarding the declines in Waukegan Harbor fish tissue PCB concentrations and subsequent easing of the fish consumption advisory, attributing these results to the beneficial effects of harbor dredging. However, the basis for such broad claims is unclear. For example, pre-remediation fish data from Waukegan Harbor are extremely limited. One carp composite sample consisting of two fish and one alewife composite sample consisting of five fish were collected and analyzed in 1991 by the EPA. EPA has indicated that the 1991 alewife data (as well as additional carp data from 1983) should not be used to assess temporal trends because of technical problems associated with the data. Consistent with this, EPA Region 5 did not use the alewife data to assess temporal patterns, but did rely on the single carp sample. Post-remediation data include several fish collected in the Upper Harbor (Station QZ001) and in Lake Michigan in the vicinity of Waukegan Harbor (Station QZB02) between 1992 and 1998. We evaluated the data collected through 1998 to explore temporal trends after remediation. Based upon uncertainty associated with the 1991 alewife value, only the carp data were used for analysis of temporal trends.

As shown in Figure 4, total PCB levels in carp declined from 136 mg/kg lipid in 1991 (based on the single carp

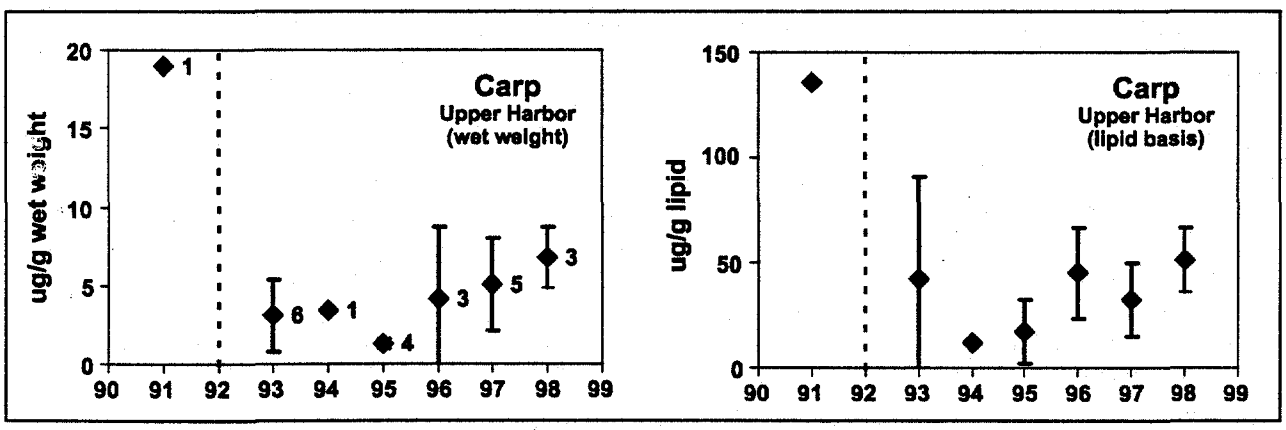


Figure 4 - Average total PCB concentrations in carp collected from the Upper Harbor of Waukegan Harbor. A single carp sample in 1991 apparently forms the basis for EPA characterization of the effects of dredging on fish PCB levels in the Upper Harbor. While these graphs indicate that PCB levels were lower in 1993, the lack of adequate pre-remediation data and the fact that fish tissue concentrations have generally been rising since 1994 indicate the presence of other factors that limit the ability to differentiate the effects of various remedial activities in the harbor. Note that data markers indicate mean values with error bars indicating +/- two standard errors. Numbers next to the mean indicate number of samples.

sample) to an average of 36 mg/kg lipid for the period from 1993 to 1998. Note that the post-dredging data included one value greater than the 1991 value (156 mg/kg lipid, collected in August 1993). The wet-weight-based fillet concentrations showed a similar pattern, namely, an apparent decline from 19 mg/kg to an average of 3.9 mg/kg. These declines apparently form the basis of EPA's claim about a 4-fold decrease in fish tissue concentrations. However, there are several features of these data that raise questions as to EPA Region 5's conclusion that dredging caused these decreases in Waukegan Harbor fish PCB concentrations, including:

- The actual extent of the decline in fish PCB levels is not clear because only one PCB measurement was obtained to establish 1991 pre-dredging levels in carp, and this single value was within the range of the concentrations measured after dredging.

- Isolation of Slip #3 by containment likely contributed significantly to decreased exposure, and therefore decreased fish tissue PCB concentrations. The observed impacts on fish concentrations were undoubtedly influenced by the isolation of Slip #3, the most contaminated part of the harbor, as a containment cell. For example, based on the average sediment PCB concentrations measured in Slip #3 and the other areas of the harbor in 1977-78 and 1985-86, containment and isolation of the slip alone equates to a 65% to 75% reduction in the area-weighted average sediment PCB concentration in the harbor. It is therefore difficult to distinguish between the relative contributions of Slip #3 containment and Upper Harbor dredging, or other factors, in judging the overall declines in fish data.

- The observed decline is inconsistent with the dynamics of the bioaccumulation process. The decline in wet-weight PCB concentration claimed between 1991 and 1993 implies a PCB half-life of approximately nine months within the carp body. We developed a basic bioaccumulation model for carp with weight and lipid fraction similar to those samples in the harbor (approximately 5 kg with a lipid fraction of 13%).[5] This model is considered realistic in that it computes a biota/sediment

[5] Bioaccumulation models provide a means by which the bioenergetic and toxicokinetic mechanisms controlling PCB uptake and loss rates can be explored in an integrated, quantitative fashion, subject to the constraints of mass balance and the requirement to match contaminant concentrations measured in the field. Metabolism has in general been found to be insignificant in models of total PCB bioaccumulation (Gobas et al., 1995; Morrison et al., 1997; Connolly, 1991; Connolly et al., 1992). Thus, PCB elimination is slow and metabolism is probably not an important loss mechanism. The carp model included elimination across the gill and growth dilution as the two mechanisms causing PCB concentrations to decline in the fish.

For metabolism to be important in the field, the rate must be significant relative to the other known mechanisms by which PCB concentrations are reduced in fish: elimination by diffusion across the gill surface, and growth. The depuration of PCBs by fish subject to chronic exposure is often very slow, much slower than observed in short-term experiments (de Boer et al., 1994; Lieb et al., 1974; O'Connor and Pizza, 1987; Sijm et al., 1992). Half-lives on the order of years have been measured (de Boer et al., 1994).

The lipid fractions reported are apparently for fillets, but this has not been confirmed and fish aging data are not available for this data set. They were used to represent whole-body lipid contents in the model. In general, whole-body lipid

(continued...)

accumulation factor of 8 grams of organic carbon per gram of lipid, which is within the range of values measured in, for example, Green Bay and Lake Orono (HydroQual, 1995). The calculated depuration half-life in the model is 6.5 years, as depicted on Figure 5. The predicted decline in fish PCB levels following removal of all exposure sources is much slower than the rate apparent in the empirical data. Thus, the observed rate of decline is not consistent with the principles of toxicokinetics and bioenergetics, meaning either the single 1991 value is inaccurate or non-representative, or that the fish sampled after remediation did not accumulate PCBs from the same exposure sources as the single fish sampled in 1991.

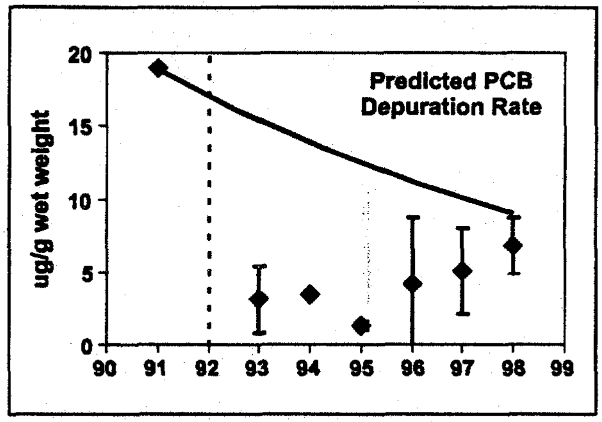


Figure 5 - Average total PCB concentrations in carp collected from the Upper Harbor of Waukegan Harbor. The trend line added to this graph is output from a bioaccumulation/depuration model for carp, assuming all exposure sources have been removed. The predicted half-life for depuration of PCBs is 6.5 years, which is inconsistent with empirical data and indicates that the carp sample in 1991 is not representative and/or the fish sampled after remediation may not have accumulated PCBs from the same sources as the carp sampled in 1991.

- The temporal trends in PCB concentrations in harbor fish are inconsistent with the removal of the local exposure source, meaning other factors must be playing a role in determining fish tissue concentrations. PCB levels in fish are expected to decline monotonically following the removal of the primary exposure source, but as shown in Figures 4 and 5, PCB levels in carp collected in the harbor show an increasing trend. Increases in PCB levels after 1993 were observed in other species as well (e.g., lake trout; see Figure 6). The reasons for the observed increases are not known, but they suggest that there are factors other than containment or harbor dredging controlling PCB levels in the fish of Waukegan Harbor.

It is unlikely that the decline in lake trout PCB levels from 1991 to 1992 was due solely to remediation activities in Waukegan Harbor. First, such a dramatic and rapid decrease could only have occurred if the sediments of Waukegan Harbor provided nearly all of the PCBs to the pelagic food web of the lake trout in Lake Michigan at station QZB02 (outside the harbor). This seems improbable, based on the observation that the lake trout at station QZB02 sampled in the late 1980s and in the mid-1990s appear to have total PCB concentrations that are similar to average levels measured elsewhere in Lake

(...continued)
contents are greater than fillet values. Increasing the whole-body lipid contents would result in a greater half-life, and therefore would show an even slower depuration rate.

Michigan, on the order of 1 mg/kg to 5 mg/kg wet weight in skin-on fillets (Stow et al., 1995).

In addition, Figure 6 shows that a similar temporary decline was observed in lake trout from station QZB02 in 1984-85. The reasons for the declines in 1984-85 and 1991-92 are not known, but it is likely that effective removal of a major exposure source would result in a permanent decline, not a temporary one. Thus, the observation of a decline in lake trout PCB levels in 1991-92, at the same time as the removal action in Waukegan Harbor, may have been fortuitous. The observation of similar declines and subsequent rises within and outside of the harbor suggest that regional processes not related to the sediments of the harbor may have significant impacts on PCB levels in fish collected within the harbor.[6]

In conclusion, the impacts of dredging on PCB levels in Waukegan Harbor fish cannot be quantified for several reasons, including: 1) the usable pre-dredging fish data are limited to one carp sample (with a PCB

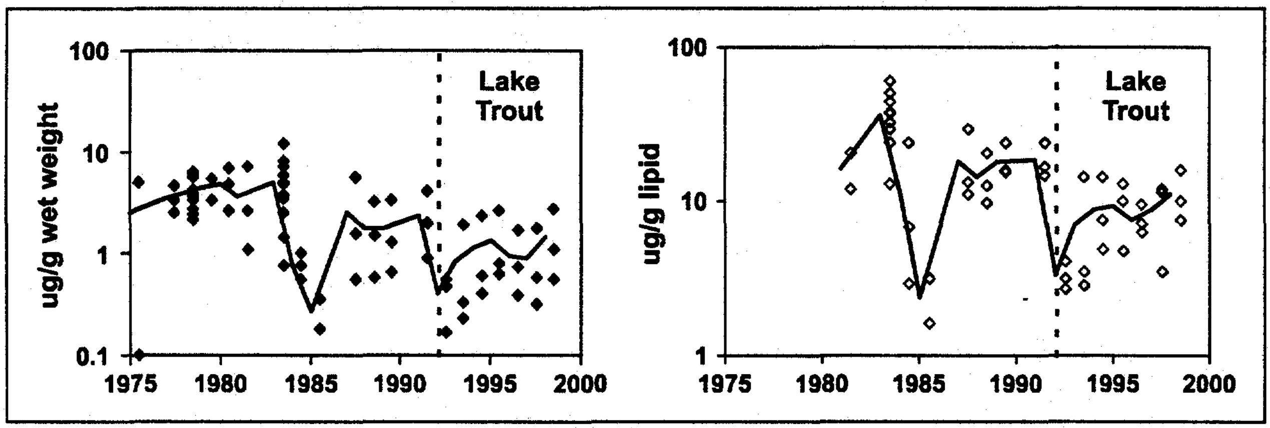


Figure 6 - Total PCB concentrations in lake trout collected from Lake Michigan in the vicinity of Waukegan Harbor (Station QZB02). The data (plotted as symbols) and annualized averages (plotted as lines) indicate a great deal of variability during each sample year and through time. The remediation of the harbor in 1992 cannot account for the declines observed in fish collected near the harbor in 1984-85 and 1991-92, or the recurring increases that are apparent over the past 25 years. The vertical dashed line denotes when remediation occurred.

concentration that lies within the range of the post-dredging measurements), 2) the containment and isolation of Slip #3 most likely contributed significantly to the decline in PCB exposure and fish tissue PCB concentrations, 3) the observed rate of decline is much faster than expected based upon predicted rates of fish

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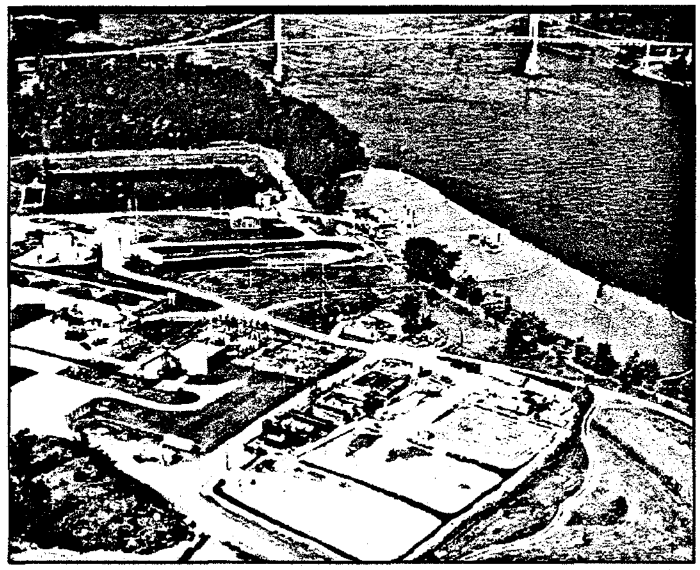
[6] For example, one member of the NAS Committee asked Mr. Doody during his presentation about the potential influence of zebra mussels that are now widespread in the Great Lakes basin. Zebra mussels filter large quantities of particles and deposit much of that material on the sediment surface in the form of feces and pseudo-feces. Hydrophobic contaminants associated with those particles are thereby transported from the water column to the sediment bed. This can result in decreased availability of such contaminants to strictly pelagic food webs, or to increased availability to food webs associated with benthic invertebrates. Zebra mussels may be playing a part in recent PCB dynamics in Lake Michigan, but their relationship to the trends observed in the Waukegan Harbor vicinity is not clear.

depuration, and 4) the temporal trends in harbor and Lake Michigan fish do not indicate steady declines in PCB concentrations as would be expected after the removal of a primary local exposure source.

2.4 – St. Lawrence River/GM Massena, New York

In November 1995, along the shore of the St. Lawrence River in northern New York State, 13,300 cubic yards of PCB-containing sediments were removed from an 11-acre nearshore site adjacent to the General Motors facility in Massena, New York. Extensive verification sampling of six dredged sub-areas demonstrated that PCB levels in none of the sub-areas within the removal area met the 1 ppm PCB cleanup level, even after a significant number of repeated passes of the hydraulic dredge. Average surface sediment PCB levels left in the six sub-areas ranged from 3 mg/kg to 27 mg/kg PCBs. The 1.72-acre sub-area having the 27 mg/kg average was subsequently capped.



An 11-acre nearshore area along the St. Lawrence River near Massena in northern New York State was dredged in 1995. 13,300 cubic yards of PCB-containing sediment were removed, and residuals in a 1.7-acre area were capped after dredging operations were complete.

Post-remediation monitoring is being performed in accordance with a St. Lawrence River Monitoring and Maintenance Plan, prepared in 1996 upon completion of remediation (BBLES, 1996). One impediment to implementing the monitoring plan, and thus adding uncertainty to the interpretation of associated data, is the fact that a targeted cove with elevated PCB levels adjacent to the remediated area was not remediated due to property access restrictions (which still exist).

According to the monitoring plan, fish monitoring efforts include annual collections of juvenile spottail shiners, a resident minnow species common to the St. Lawrence River. Data describing whole-body PCB concentrations (and lipid content) in spottails are being used to monitor the effects that sediment remediation activities may have on PCB concentrations in nearby populations of St. Lawrence River aquatic biota. The monitoring objective is to provide a measure of the effectiveness of the dredging and sub-area capping in reducing the bioavailability of sediment-based PCBs to resident aquatic biota of the St. Lawrence River and to provide a baseline for future remedial actions in the cove. Annual sampling efforts include the collection of seven whole-body composite samples from each of two sample locations, the nearshore remediation area and the cove, for a maximum total of 14 samples. Sampling began during the fall of 1997. However, due to access restrictions, spottail shiners still could not be collected from the cove.

Two annual monitoring reports have been issued (BBLES, 1998; 1999), and include spottail shiner wet-weight and lipid-normalized PCB data for the remediation area. According to the two reports, PCB concentrations in spottail shiners collected in 1998 appear slightly higher than those collected in 1997, with an arithmetic mean of seven composite whole-body samples exhibiting 3.6 mg/kg PCBs in 1998 versus 1.2 mg/kg in 1997. However, PCB concentrations remain much lower than data from 1988 and 1989 reported by the Ontario Ministry of Environment (OME) and New York State Department of Environmental Conservation (NYSDEC), but similar to 1990-91 and 1994 data (see Figure 7). Direct comparison of pre-remediation fish data with post-remediation data is complicated by uncertainties about collection locations for the pre-remediation fish.

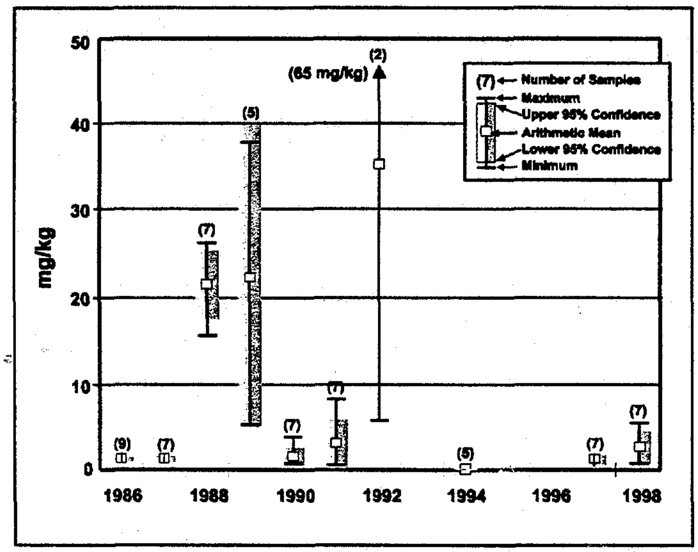


Figure 7 - Total PCB concentrations in spottail shiner whole-body composite samples collected from the GM Massena site on the St. Lawrence River. PCB levels may have decreased since the late 1980s, but the pre-remediation data are limited by factors such as variability (especially the 1988-89 and 1992 data relative to all other years) and the fact that pre-remediation sampling locations cannot be identified in order to make reliable comparisons.

According to BBLES (1999), OME and NYSDEC have indicated that it is not possible to verify the locations where specific pre-remediation fish were collected.

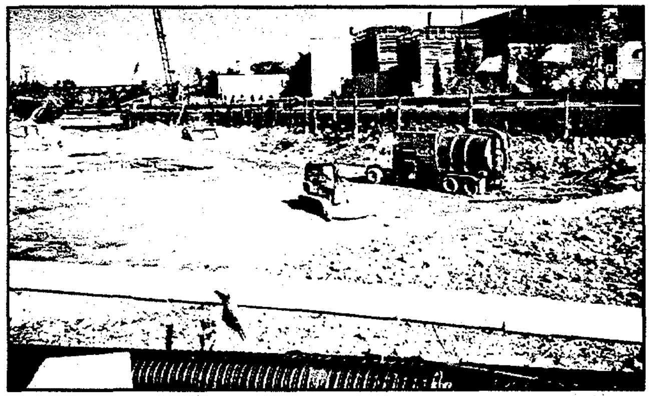
The monitoring reports describe an anomaly to the apparent general downward trend since the late 1980s: two spottail shiner samples collected by NYSDEC in 1992. The wide difference in concentrations for these two samples (total PCB concentrations of 5.7 mg/kg and 65 mg/kg) is difficult to explain. Similar variability, although not as great, is also evident in the data collected by the OME in 1989. The variability of the data may be due to several factors, including differences in sampling locations, fish lengths and sizes, fish lipid content, or species mobility. Regardless, the variability of the data precludes a more detailed evaluation and interpretation of the overall spottail shiner data. As such, the monitoring reports conclude that the significance of the 1997 and 1998 PCB data, and any apparent trends, will need to be more thoroughly evaluated following the collection of additional data over the next three years.

We have been unable to reconstruct how EPA Region 5 has used the St. Lawrence River data to calculate an

8-fold reduction in post-remediation fish concentrations, especially when fish data for five of the eight pre-remediation sampling events show PCB concentrations at levels similar to post-remediation levels. Although fish levels may seem to be on a downward trend, the question of how and where the pre-remediation fish were exposed (i.e., within the 11-acre site, the cove, or the very large St. Lawrence River channel?) precludes a complete and direct comparison, and therefore limits the certainty of any associated conclusions. Clearly, the need for post-dredging capping of a portion of the removal area also makes it difficult to differentiate the effects of dredging versus these other factors.

2.5 – Ruck Pond, Wisconsin

Ruck Pond is one of a series of mill ponds created on Cedar Creek, just upstream of the low-head Ruck Pond Dam in the town of Cedarburg, Wisconsin, north of Milwaukee. In 1994, an impounded 1,000-foot section of the creek (Ruck Pond) was drained after a temporary dam was installed on the upstream end and flow was bypassed through siphon piping. The project goal was to remove all soft sediment (contaminated with PCBs) down to bedrock, to the extent practicable. The 60 soft-sediment samples that were collected from depths of 6 to 24 inches just before remediation exhibited PCB concentrations ranging from non-detectable to 2,500 mg/kg (average 76 mg/kg).



Ruck Pond on Cedar Creek in Wisconsin was remediated in 1994 using dry excavation techniques after the stream flow was diverted and the pond drained. 7,730 cubic yards of sediment were removed.

A total volume of 7,730 cubic yards of sediment was removed by dry excavation in 1994 and disposed of at commercial landfills. Despite intensive and painstaking removal efforts over a five-month period, some residual sediment was left on the creek bed. Seven samples of the residual sediment exhibited PCB concentrations ranging from 8.3 mg/kg to 280 mg/kg (average 84 mg/kg). As part of pond restoration efforts, clean materials used for access to the pond were spread along portions of the pond bottom. Although not intended for capping, these materials inevitably provided some containment of the residual sediment, and likely would have reduced (via burial) the relatively high PCB concentrations remaining at the sediment surface that the dredge equipment could not effectively remove.

The Wisconsin Department of Natural Resources (WDNR) measured whole-body PCB congener concentrations in caged fathead minnows at three locations before and after the sediment removal operation (Amrhein, 1997). Three cages were placed at each of three stations: a site in Cedar Creek upstream of Ruck Pond called Cedarburg Pond, a site within the downstream end of Ruck Pond, and a site downstream of the Ruck Pond Dam, located just upstream of Columbia Dam.

In July 1994, just before the start of removal, PCBs were measured in caged fathead minnows at the three stations. The average PCB concentrations were 0.12 mg/kg upstream, 24 mg/kg at the Ruck Pond station, and 12 mg/kg at the downstream station (7.1, 1,700, and 630 mg/kg lipid, respectively). The average PCB concentrations measured in caged fish in August and September 1995, about one year after remediation, were 0.09 mg/kg upstream, 4.2 mg/kg within the pond, and 11 mg/kg downstream (2.2, 170, and 360 mg/kg lipid, respectively). These PCB levels in the caged fish collected in Ruck Pond appear to have declined 75 to 85%[7] on a wet-weight basis and approximately 90% on a lipid basis after remediation. It is apparently on this basis that EPA Region 5 concluded that sediment removal in Ruck Pond resulted in an 9-fold reduction in fish PCB concentrations. However, caged fish PCB concentrations at the upstream "background" location also declined 25% wet weight and 70% on a lipid basis one year after remediation, and caged fish concentrations downstream of Ruck Pond declined 10% wet weight and 40% on a lipid basis. These declines outside of Ruck Pond indicate that system-wide natural recovery processes may be occurring.

Two years later, samples of resident fish were collected in 1997 by the WDNR and analyzed for PCBs. Fish were collected from two stations: within Ruck Pond and a downstream location. Average total PCB concentrations measured in fillets of four species of resident fish still exceeded the U.S. Food and Drug Administration (FDA) 2 mg/kg tolerance level and ranged from 0.35 mg/kg to 3.1 mg/kg at the station within Ruck Pond, and 1.7 mg/kg to 13.8 mg/kg at the station downstream of Ruck Pond. Fish species included carp, pike, rock bass, and white sucker. We are attempting to obtain lipid values and additional pre-remediation fish data in order to develop a full temporal and spatial comparison.

The reasons for the differences in fish tissue concentrations between the upstream and downstream stations and the Ruck Pond station are unclear. James Amrhein (1997) of the WDNR has indicated that the smaller decline at the Columbia Dam station may be an artifact of cage location. It is also possible that the PCB levels measured at the most downstream station are a more realistic reflection of post-remediation exposure levels than the Ruck Pond station. However, difficulties in implementing the caged fish program may have been a factor.

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[7] Two exposure periods occurred in Ruck Pond, 29 and 37 days. Average PCB levels were greater in the longer exposure, indicating that the fish were not at steady state with respect to their exposure sources. Therefore, pre-and post-remediation comparisons were carried out independently for each exposure period. The range of values given reflects the two comparisons.

For example:

- Pre-remediation cages in Ruck Pond were deployed during the time that pre-removal in-water construction preparations and disturbances were occurring (e.g., work boat traffic, installation of the dam and siphon).

- One of the pre-remediation cages in Ruck Pond was lost; two others were displaced about 100 feet and were not found for removal until 29 and 37 days after placement (rather than the targeted 28 days).

- Pre- and post-exposure periods were in different months (June vs. August) with different water temperatures likely.

In conclusion, the great majority of soft sediment was removed from Ruck Pond; however, elevated PCB levels up to 280 mg/kg remained in residual sediment after remediation. PCB levels in caged fish placed in Ruck Pond one year after remediation exhibited significant declines compared with pre-remediation caged fish. However, at the same time, upstream (background) and downstream caged fish also exhibited substantial declines. The presence of residual PCBs, the disturbance of the pre-remediation cages, and the observation of a decline in fish levels upstream of Ruck Pond, all add considerable uncertainty to EPA's conclusions and attempts to isolate and quantify the effectiveness of dry excavation sediment removal on fish PCB levels. In addition, the pond restoration materials provided some containment of the residual PCBs, thereby further limiting the ability to demonstrate the effectiveness of sediment removal versus other factors.

2.6 – Summary of Case Study Evaluation

The impacts of sediment removal by excavation or dredging are influenced by several site-specific factors, including the presence of pre-existing system-wide natural recovery processes, the potential for resuspension of sediments during remediation, the presence of residual PCBs that can recontaminate the sediment surface after remediation, and modification or destruction of fish habitat as a result of remedial action. Thus, the impacts of sediment removal are likely to vary among sites, and a robust understanding of these impacts should be based on adequate data from many sites. Therefore, the analysis of results from several sediment remediation projects is relevant and critical. The focus on fish tissue PCB concentrations also is reasonable, since risk reduction should be the focus of all remedial activities, and fish ingestion is typically a primary exposure pathway driving both ecological and human health risks.

However, because EPA has not addressed or accounted for each of these factors in its analyses of the five case study projects (or any of the 12 projects cited overall), EPA cannot support its conclusions regarding the impacts of sediment removal actions on declines in fish tissue PCB levels. This is because the effects of sediment removal at such sites cannot be separated from other recovery processes or remedial actions, including natural attenuation, source control, or containment. At all five sites there is evidence of system-wide changes in biota PCB levels and other factors that make it very difficult to demonstrate sediment removal as the only factor that has led to declines in fish tissue concentrations.

Collection of several years of high-quality and comparable data before and after remediation is critically important to developing a technically sound assessment of the effectiveness of sediment removal in reducing PCB levels in fish, and the associated reductions in PCB bioavailability, exposure, and risk. An adequate sampling program, database, and evaluation methodology should include the ability to: 1) distinguish removal impacts from the effects of other processes such as the natural attenuation, transport, or containment of PCBs, 2) reduce the uncertainties inherent in field sampling of biota, and 3) account for the long biological half-lives of strongly hydrophobic chemicals such as PCBs that can delay the response of fish tissue levels to changes in their degree of exposure. These important pre-condition data are simply not in place for the sediment remediation projects cited by EPA. At the Waukegan Harbor site, for instance, the pre-remediation fish tissue data consisted of one PCB measurement and, at the Ruck Pond site, the pre-remediation study included fish cages that were disturbed and one that was lost completely. The uncertainties associated with these types of monitoring datasets limit their utility for quantifying and therefore demonstrating the impacts of dredging on fish contaminant levels.

The mixed results observed for all five of the case study projects cited by EPA indicate that an emphasis on mass removal efficiency alone as an objective for management of contaminated sediment cannot be relied upon as a measure of the effectiveness of sediment removal in reducing contaminant bioavailability and exposure, and therefore potential risks associated with residual contaminant levels in post-remediation sediments and fish. Evaluations of risk reduction, when based on adequate data and methodology, represent a more technically sound measure of remedial effectiveness than removal efficiency. Thus far, the pre- and post-remediation monitoring programs and EPA's subsequent data analyses have not achieved these basic requirements in order to substantiate its numerous claims regarding the effectiveness of sediment removal.

SECTION 3 – EVALUATION OF OTHER EPA REGION 5 CLAIMS

This section critiques EPA Region 5's three other major assertions regarding the effectiveness of sediment removal, based on the 12 projects cited by EPA (listed in Table 1). These other assertions are:

- Contaminant mass removal is the primary measure of remedial success;
- Short-term adverse effects of dredging are minor; and
- Unit costs tend to decrease with the increasing scale of sediment removal.

3.1 – EPA Claim Regarding Mass Removal as a Measure of Dredging Success

A remedy designed solely to remove a large percentage of the contaminant mass may not lead to reductions in exposure and risk because risk in aquatic systems is driven by the *position* of contaminant mass, not just the *presence* of that mass. This means that contaminants in the biologically active zone of surficial sediments are potentially available for exposure to the benthic and pelagic food webs, but contaminants positioned well below the sediment surface (i.e., buried) do not pose risks because they are not available to various receptors. Nevertheless, in its evaluations EPA Region 5 judged remedial success based on the amount of mass removed without regard to where in the sediment profile the mass was located, whether stated concentration-based cleanup goals were achieved, or whether exposure potential and risk were reduced.

Regarding attainment of stated cleanup goals, EPA Region 5 has not demonstrated that low sediment cleanup levels have been achieved throughout the remedial target area at any of the eight dredging projects cited by Region 5. For one project, the cleanup level was not attainable in any sector of the target area (St. Lawrence River/GM Massena). At three sites, cleanup levels were not achieved in several areas targeted (River Raisin/Ford Outfall, Manistique Harbor, and Lake Jarnsjon). For three projects, the residual contaminant level is unknown because verification sampling and analyses were not performed (Shiawassee River, Waukegan Harbor, and Black River). For one project, no sediment target was set, but PCB levels as high as 295 mg/kg remained after dredging (Sheboygan River).

Six of the 12 projects cited in Table 1 were used by Region 5 to claim that 98% or more PCB mass removal was achieved. However, four were relatively small-scale hot spot removal projects (River Raisin/Ford Outfall, St. Lawrence River/GM Massena, Ottawa River, and Sheboygan River), and two were projects involving removal across the entire bottom of three ponds and a lake (Willow Run Creek and Lake Jarnsjon, respectively). Even if EPA's mass removal claims were relevant to risk reduction, the claim of an average PCB mass removal of 98% or greater is misleading from at least two other standpoints, namely:

- EPA's mass removal calculations are confined only to the targeted area. In the case of hot spot removal projects, there is no recognition that PCB mass present in the water body outside of the targeted area may be considerable and equally as bioavailable as the PCB mass in the targeted area. For example, accounting for the presence of PCB mass in an extended river or stream outside of the target area would add greatly to the pre-dredging mass value and would typically make the calculated percentage of mass removal from a hot spot a much lower and less impressive value.

- Calculating mass removal strictly from a hot spot produces high removal percentages that appear to make dredging highly efficient. For rivers, streams, or other water bodies with diffuse and widespread contamination, and few or no targetable hot spots (e.g., the Fox River), the ability to remove a high percentage of overall PCB mass with a small dredging project may not exist.

Claiming "success" through PCB mass removal calculations ignores the actual project goals and objectives set out in decision documents before the remediation. For example, in Table 4 we have summarized the primary goals, the sediment remedial target, and the outcome for the eight largest projects evaluated by EPA Region 5, including some of the projects mentioned above.[8] Mass removal is not a stated objective of the remediation effort in any of the eight projects, and achievement of the primary goal or significant risk reduction has not been confirmed for any of these projects.

In summary, contaminant mass removal is an easily defined and calculated result that, at face value, may seem sensible and beneficial. However, mass removal may produce little observable long-term benefit or risk reduction, may result in more harm to the environment than benefit, and as a result, may be an inefficient and even counter-productive method to reduce risk from exposure to contaminated sediments.

[8] The four smallest projects (less than 10,000 cubic yards removed) were omitted from Table 4 because of their small size and interim or pilot status. Further, the smallest of the four projects (Shiawassee River, removal of 1,805 cubic yards) was implemented 17 years ago, before the site was listed on the National Priorities List and at a time when such projects were less likely to be approached with scientific rigor. Nonetheless, for the two (of these four) small projects cited by EPA Region 5 as attaining 98% mass removal (Ottawa River unnamed tributary and Sheboygan River pilot project), mass removal was not set out as an objective. For the unnamed tributary, the objectives were to reduce the potential for PCB movement and to minimize the potential for human and wildlife exposure. For the Sheboygan River pilot project, the objectives were to test dredging and armoring technologies and to remove sediments with greater than 686 mg/kg PCBs, based on dermal exposure risk (AEM, 1999).

Table 4 - Primary Goal versus Outcome for the Eight Largest Sediment Remediation Projects Cited by EPA Region 5

Primary Goal	Basis for Goal	Cleanup Goal	Achievement of Cleanup Goal	Achievement of Primary Goal
Willow Run Creek, Michigan - 450,000 cubic yards removed by dry excavation				
Eliminate adverse ecological impacts	Ecological assessment based on ingestion modeling, then feasibility and compliance with MI Act 307	Depending on locale, removal to 21 or 1 ppm PCBs below waterline and 21 or 2.3 ppm above waterline	Achieved; based on verification sampling	Unknown; no formal post-remediation monitoring program is planned
Lake Jarnsjon, Sweden - 196,000 cubic yards removed by dredging				
Substantially reduce the transport of PCBs from lake sediments to lake water and downstream system in order to reduce PCB concentrations in biota	Analysis of sediment, water, and biota data collected during late 1980s and early 1990s	Maximum 0.5 ppm PCBs and no more than 25% of the remediated area greater than 0.2 ppm	PCBs in 8 shallow (undefined) surface samples from east end of lake after remediation ranged from 0.7 to 2.4 ppm; PCBs in 54 composite samples from entire lake bed (0-20 cm depth) ranged from 0.01 to 0.85 ppm.	Apparently achieved; PCB levels in water and year-old perch had decreased 2 years after remediation, but PCB levels in water and fish from upstream and downstream reference locations also decreased
Bryant Mill Pond, Michigan (tributary to Kalamazoo River) - 165,000 cubic yards removed by dry excavation				
Time Critical Removal Action to mitigate direct contact exposure and threat to aquatic life/wildlife from ongoing releases	Ecological risk assessment and observation of continuing releases from erosion and sloughing from banks	10 ppm PCBs	Reportedly achieved based on verification sampling, but sampling avoided the top 3 inches of sediment/soil where PCBs could remain bioavailable; no justification of this verification sampling technique given	Reportedly achieved, but may be unknown based on verification sampling that avoided surface materials; Action Memorandum stated "the nature of the removal is, however, expected to minimize the need for post-removal site control, at least in the Bryant Mill Pond area"

Primary Goal	Basis for Goal	Cleanup Goal	Achievement of Cleanup Goal	Achievement of Primary Goal
Manistique River and Harbor, Michigan - 72,000 cubic yards removed by dredging				
Reduce PCB levels in fish to reduce cancer and non-cancer risks to <10-4 and <1 H.I.	Human Health Risk Assessment; risk targets exclude high-end exposure for subsistence and certain recreators	10 ppm PCBs, which was default level based on BSAF to estimate target, then increased to 10 ppm	In progress, 10 ppm level proving difficult to achieve; PCBs up to 3,000 ppm in residual sediment (Dredge Area 9) were left behind over winter after the 1998 construction season	Dredging continues in 1999, so too early to determine after 5 years of dredging; no post-remediation monitoring program yet defined
Black River, Ohio - 60,000 cubic yards removed by dredging				
Remove all PAH and metal contaminated sediments	Clean Air Act Consent Decree	Depth horizon (removal down to "hard bottom" or bedrock)	Depth horizon achieved, but no analytical verification	Depth horizon achieved, but no analytical verification
Waukegan Harbor, Illinois - 38,300 cubic yards removed by dredging				
Eliminate PCB flux from the harbor into Lake Michigan	Hydrodynamic modeling	50 ppm PCBs in the harbor; 500 ppm PCBs in Slip #3	Unknown; no analytical verification; dredging advanced to a pre-defined depth (reportedly to the underlying uncontaminated sand layer)	Unknown; limited analysis of surface samples from the harbor over 4 years after dredging showed PCBs from 3 to 9 ppm
River Raisin (Ford Outfall), Michigan - 28,500 cubic yards removed by dredging				
Reduce PCB levels in fish	Risk analysis by EPA	10 ppm PCBs (after removal down to native clay layer)	Partially achieved; removal to refusal achieved; 3 verification samples had 12-20 ppm PCBs, 4 had 0.5-7 ppm, 7 had insufficient media	Unknown; no formal post-remediation program, but 2 sediment cores had 60-110 ppm PCBs; MDEQ fish data not available
St. Lawrence River (GM Massena), New York - 13,300 cubic yards removed by dredging				
Reduce PCB levels in fish	Human Health Risk Assessment	1 ppm PCBs	Not achieved; average residual PCBs in 6 quadrants ranged from 3-27 ppm with a maximum concentration of 90 ppm	Two annual post-remediation fish monitoring events complete; no discernible trends between years

3.2 – EPA Claim Regarding the Short-Term Impacts of Dredging

EPA Region 5 makes the unfounded claim that PCB losses during dredging are much less than the annual PCB losses from natural erosion. As discussed below, this claim is illogic because it is based on an inappropriate comparison (i.e., comparing losses from discrete removal areas to losses from entire systems) and ignores the fact that PCB mass is not directly related to risk reduction. Even if one were to ignore these flaws, EPA does not present data to support its conclusion.

First, comparing average annual erosional losses from an entire contaminated sediment site to losses from the surface area of a *particular* dredging removal area makes no sense because it is an "apples to oranges" comparison. For example, the Deposit N dredging project on the Fox River in Wisconsin, which is targeting just 13,000 cubic yards of sediment (out of the estimated 11 million cubic yards of contaminated sediment in the lower 39 miles of that river), will likely result in losses to the water column that are much less than annual erosional losses from the entire 39 miles of the river. However, this comparison says nothing whatsoever about what the losses to the water column might be if one were to dredge all (or a significant part of) 11 million cubic yards of contaminated sediments. Thus, it is misleading to compare the mass of PCB transport resulting from annual erosional losses with the mass of PCB lost to the water column from dredging.

Second, as noted previously, the mass of PCBs transported by erosional (or other) events is not as important to risk reduction as the presence and concentration of PCBs in the biologically active zone of surficial sediments. For example, PCB discharges to the Fox River were virtually eliminated in the 1970s, which has allowed over two decades of natural recovery to bury these historical PCB deposits under progressively cleaner layers of fresh sediment from the watershed. This has led to conditions today where surface sediments have low PCB concentrations (most average about 2 mg/kg, which is already lower than EPA cleanup goals at many other sites), and over 85% of PCB mass is buried below one foot or more of cleaner sediment in very depositional areas that are not susceptible to scour at that depth. Therefore, if erosion results in transport and redeposition of these relatively clean surficial sediments, the sediment surface will not become more contaminated over time. Instead, transported sediments mix with clean solids coming in from the watershed so that the mixture that is redeposited will be progressively cleaner over time. The net effect is that PCBs in the surface bioavailable zone will become less available for exposure or transport. On the other hand, if the sediments that are mobilized by dredging come from the more contaminated deep sediment layers, the material transported downstream may, upon redeposition, cause increased exposure because the surficial layer has become more contaminated than pre-dredging conditions.

Third, even if one were to ignore the facts that comparing annual erosional losses from an entire contaminated sediment area with losses from a particular dredging project is irrelevant, and that any such comparisons on the basis of mass are misleading, the data that EPA Region 5 cite do not support EPA conclusions. Region 5 used two sites for its comparisons of annual erosional losses to dredging project losses – Manistique River/Harbor and the Fox River.

In the case of the Manistique River and Harbor, EPA used analyses of PCBs in the water column downstream of the silt-curtained dredging areas, then calculated the equivalent PCB load and compared this loading with a prorated (and previously calculated) annual PCB discharge from natural erosion. Since the surface water concentrations measured during dredging were often low or not detectable, the results at first glance appear quite favorable (however, note that although water-column PCB concentrations were low, levels were still higher than pre-dredging values). In the Fox River case, EPA compared previously calculated annual PCB discharges from natural erosion in the river with the estimated loss from a hypothetical sediment dredging project. The estimated loss was set at 2% of the removed sediment mass, an unverified resuspension loss rate from hydraulic dredges based on "engineering judgment." Again, the comparison appears at first glance favorable – PCB losses during hydraulic dredging for a hypothetical Fox River project are predicted as a factor of 2.5 less than those from annual erosion. However, these comparisons need to be evaluated in light of the following points regarding resuspension losses:

- The idea for this type of a sediment resuspension analysis likely originates with the Interagency Review Team Report for the Manistique River (April 1995) in which the team concluded that: 1) "The adverse effects of implementing dredging (the additional 900 pounds of PCBs released to the harbor) are equivalent to 9 years of PCB loading at the current rate; the review team considered this an acceptable tradeoff . . ." and 2) "Even at a 2% release rate, a 280 pound PCB loss during dredging is only equivalent to a 2 to 3 year loss of PCB under existing conditions." This finding is flawed from several standpoints, namely, it is hypothetical, the loss rates and resuspension rates are unsubstantiated, and the above "adverse effects of implementing dredging" assumed two years of dredging and not the actual five or more years of dredging being implemented at the Manistique site.

- Sediment resuspension is a complicated issue and is influenced by numerous variables. We have determined that data collected to date from *all* small- and full-scale dredging projects are sparse and not sufficient for quantifying resuspension rates. Other important unresolved issues regarding resuspension include the fact that: 1) a portion of the resuspended contaminants falls back onto the dredged surface, making attainment of a low cleanup level extremely difficult, particularly if deep sediments containing higher levels of contaminants are resuspended and redeposited on the surface, and 2) "resuspension plumes" tend to stay close to the bottom as they move away from the dredge, in which case, downstream *surface* water samples may not detect the bulk of resuspended material.

- For multi-year projects with winter shutdowns, the resuspended material that settles onto, and is left on, the surface tends to be loose and unconsolidated and more susceptible to mobilization and downstream transport for months between construction seasons. For example, so-called "short-term" impacts at Manistique Harbor include EPA's leaving sediment PCB concentrations of up to 3,000 mg/kg over the five winter months between construction seasons, as happened at the end of the 1998 season. After years of these "short-term" impacts, they begin to evolve into long-term concerns and opportunities for increased exposure and downstream transport. In short, even though the mass of resuspended material might be relatively small in absolute terms, it may contribute significantly to the risk associated with biological uptake.

3.3 – EPA Claim Regarding Dredging Unit Costs and Economies of Scale

EPA Region 5 concludes that unit costs for sediment remediation decrease as removed sediment volume increases and that very large removal projects will yield much lower unit costs than have been realized on sites to date. This conclusion is not consistent with what is known about the primary determinants of dredging project costs, and is not supported by the cost figures for the projects highlighted by EPA.

The two primary determinants of cost for remedial dredging projects are dredge production rate and disposal cost. Dredge production rate depends on unique site conditions such as access, water depth, and debris; the targeted depth or cleanup level; limitations in land-based water management facilities; and whether verification sampling is performed during dredging. Disposal cost depends on type of contaminant, type of disposal facility (on-site, dedicated nearby, or commercial), and distance of the disposal facility from the site. To a large extent, these variables are not volume-dependent. Economy-of-scale advantages, such as longer use of temporary support facilities and water treatment facilities and possible slightly lower unit disposal costs for large volumes, are small in comparison. As a result, large projects will still be extremely costly.

In an article by EPA Region 5 titled "Dredging: Long-Term Benefits Outweigh Short-Term Impacts" (Pastor, 1999), EPA states that, "Although removing greater volumes increases total costs, economies of scale on larger projects also give you lower unit costs. In other words, as projects increase in size, the cost of removal and treatment and/or disposal per cubic yard of contaminated sediment goes down." To evaluate EPA's claim, we compared total unit cost versus volume of sediment removed for 40 completed projects in the United States:

20 remedial dredging projects and 20 dry excavation projects.[9] Although the smallest projects (e.g., pilot-scale removals and others less than 10,000 cubic yards) tended to have high unit costs, no clear trends in economy of scale were discernible as unit costs ranged widely from about $50 to $1,500 per cubic yard with no apparent relationship to sediment volume removed. Therefore, it is unclear how and on what basis EPA arrived at its definitive claim regarding the existence of economies of scale.

[9] This evaluation is not necessarily a definitive test of EPA's claim because no "very large" removal project has been implemented, and the projects represent a wide variety of site conditions, remedial goals, and disposal methods that are not necessarily directly comparable. Nonetheless, site data were evaluated for apparent trends in economy of scale.

SECTION 4 – SUMMARY OF OVERALL EVALUATION

The purpose of this paper was to review how EPA Region 5 reached its stated conclusions regarding the effectiveness of sediment removal based on the data from the 12 sites listed in Table 1, and to present our own findings and supporting rationale. Our primary conclusions include: 1) EPA has not demonstrated that the sediment removal actions at the 12 cited projects reduced PCB exposure and risk, 2) reduction of PCB concentrations in fish is a meaningful measure of risk reduction, but the uncertainty associated with limited data availability, data quality concerns, and EPA's selective use of data do not support EPA's conclusions regarding the effects of sediment removal on fish at these sites, and 3) EPA's analysis of the 12 sites cannot differentiate the effectiveness of sediment removal from that of several other factors such as source control, containment, capping, or natural attenuation. We also note:

- In many instances, the factual basis for EPA's claims and conclusions is not apparent. References are not cited and backup data are not provided. Further, the available data are used selectively by EPA, and the impacts of mechanisms other than sediment removal are not adequately recognized or accounted for in EPA analyses.

- EPA neither defines the original remediation goal for each project nor fully reports results relative to whether risks were reduced and other remedial goals were achieved. Instead, EPA measures "success" by the degree of mass removal or concentration reduction without regard to risk-based benefits to be achieved. Even on projects with high contaminant mass removal efficiency, residual surface sediment concentrations in the remediated area often exceed stated cleanup goals and remain available for transport or uptake into food webs, which does not serve to reduce risk.

- Contaminant mass removal is an easily defined and calculated result that, at face value, may seem sensible and beneficial. However, mass removal may in fact produce little observable long-term benefit or risk reduction, may result in an overall net harm to valuable habitat and the environment and, as a result, may be an inefficient and even counter-productive expenditure of dollars and resources.

- EPA's data collection and analysis methods for the 12 projects are flawed. In most cases the pre-remediation fish data are sparse, and monitoring was not planned or documented with the foresight or intent of comparison with post-removal data, making EPA's stated conclusions difficult to support. Our detailed review (in Section 2) of the five case study projects evaluated by EPA demonstrates how the limitations in Region 5's data and methodology make it difficult to determine what, if any, beneficial or other effects on fish can be attributed to sediment removal rather than other observed factors such as natural attenuation.

- EPA's claim that contaminant losses due to sediment resuspension during dredging are temporary and produce only minor short-term impacts is suspect. The claim ignores that fact that contaminant mass, whether in-situ or transported, is not directly related to risk reduction. Rather, contaminated sediment resuspension and redeposition caused by dredging can lead to unacceptable increases in risk as contaminants are made available for transport or biological exposure.

- EPA cannot substantiate its claim that unit costs for sediment remediation decrease as volume of sediment removed increases. In contrast, we have concluded from evaluations of actual cost data that the two primary determinants of cost for remedial dredging projects are dredge production rate and disposal cost, neither of which is very volume-dependent.

- Finally, removal of sediment by dredging or dry excavation is not a cure-all for managing contaminated sediment. On future projects, it is recommended that EPA:
  - Seriously consider the limitations and potential negative impacts associated with sediment removal as a remedy, including an evaluation of overall environmental and social costs and benefits;
  - Not ascribe benefits to sediment removal based on limited or inappropriate data;
  - Provide for sufficient pre-and post-remediation data and analysis to demonstrate benefit. The approach used by EPA for justifying sediment removal at the 12 project sites evaluated here is inconclusive and not technically sound; and
  - Not pursue large dredging projects until the risk-reduction benefits of sediment removal have been adequately demonstrated.

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