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Ecological Benefits of Contaminated Sediment Remediation

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MANAGEMENT PERSPECTIVE

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ABSTRACT

Contaminated sediment has been identified as a source of ecological impacts in marine and freshwater systems throughout the world and the importance of the contaminated sediment management issue continues to rise in all industrialized countries. In many areas, dredging or removal of sediments contaminated with nutrients, metals, oxygen demanding substances and persistent, toxic, organic chemicals has been employed as a form of environmental remediation. In most situations, however, the documentation of the sediment problem has not been quantitatively coupled to ecological impairments. As a result, stipulating how much sediment needs to be cleaned up, why, and what improvements can be expected over time has been inadequate. In addition, the lack of long-term, post activity research and monitoring for most projects has impeded a better understanding of the ecological significance of sediment contamination. This paper reviews many of the known links and impacts, and examines the results from some case studies.

OLD VERSION

I. Introduction

Sediments contaminated with nutrients, metals, organics and oxygen demanding substances can be found in freshwater and marine systems throughout the world. While some of these contaminants occur in elevated concentrations as a result of natural processes, the presence of many is a result of human activity. Aquatic sediments with elevated levels of contaminants can be found in any low energy area that is the recipient of water associated with urban, industrial or agricultural activity. Such low energy, depositional zones can be found in nearshore embayments and river mouth areas and are also likely to be ecologically significant. These nearshore areas frequently represent the most significant spawning and nursery sites for many species of fish, the nesting and feeding areas for most of the aquatic avian fauna, the areas of highest primary and secondary biological productivity and, the areas of greatest human contact.

Until recently, determining whether or not the sediments are causing detrimental ecological impacts and then quantifying the relationships has been limited to indirect or circumstantial evidence (Burton 1992). Although we now have a number of methods available that assess the quality of sediment, and its interaction with the rest of the aquatic ecosystem, which can be used to estimate ecological risk and even quantify impacts, we cannot accurately measure, or predict ecosystem significance based on an examination of the components alone. We still appear to lack an approach that integrates the physical, chemical and biological components of the ecosystem (Krantzberg et al. 2000).

Sediment removal has been used as a management technique in rivers, lakes and reservoirs to both reduce the health risks from sediment-associated contaminants and to rehabilitate degraded aquatic ecosystems. The technique has been employed in Asia, Europe and North America to address nutrient, metal and persistent organic chemical contamination, with variable success and occasional surprises. In most cases, where some form of sediment remediation has taken place, there has been limited quantification of the sediment-impact linkage prior to taking action and rather limited monitoring afterwards (both temporal and in the ecosystem components examined).

Our purpose is to review the nature and effects of contaminated sediment in aquatic ecosystems, share selected management experiences and the associated ecological response to sediment remediation, and make some recommendations on research and management actions to improve the effectiveness of future remediation projects.

II. Contaminated Sediment in the Aquatic Environment

The accumulation of contaminants in the sediment at levels that are not rapidly lethal may result in long-term, subtle effects to the biota by direct uptake or through the food web. The cycling and bioavailability of sediment-associated contaminants in aquatic systems over both short and long time frames are controlled by physical, chemical, biological, and geological processes.

Physical processes affecting sediment contaminant distribution include mechanical disturbance at the sediment-water interface as a result of bioturbation, advection and diffusion, particle settling, resuspension, and burial. Some examples of significant geological processes affecting contaminant distribution and availability include weathering or mineral degradation, mineralization, leaching, and sedimentation. Chemical processes such as dissolution and precipitation, desorption, and oxidation and reduction can have profound effects, as well as biological processes such as decomposition, biochemical transformation, gas production and consumption, cell wall and membrane exchange/permeability, food web transfer, digestion, methylation, and pellet generation. In addition, there are fundamental differences in the physical, chemical, and biological properties and behavior of organic versus inorganic substances (metals, persistent organics, organo-metals, and nutrients) and this suggests the need for a detailed knowledge of the area and the relative importance of these processes prior to completing an assessment of impact or planning remedial measures to mitigate ecological impairments. Details of the major processes and their effects on contaminant cycling and movement can be found in Forstner and Whittman (1979), Salamons and Forstner (1984), Allan (1986), and Krezovich et al. (1987); however, it is important to explore some of the factors that affect bioavailability and uptake of contaminants, as well as the likely, quantifiable consequences of bioaccumulation.

The rate and mechanism of direct contaminant uptake from sediment by bottom-dwelling organisms can vary considerably among species, and even within species. Factors such as feeding ecology of the organisms, their developmental stage, season, behavior, and history of exposure affect contaminant uptake and body burdens. As well, different routes of uptake (soluble transfers versus contaminated food) can also be expected to affect tissue levels (Russel et al. 1999; Kaag et al. 1998).

Experiments with organochlorine pesticides have yielded conflicting results on the relative significance of diet versus aqueous uptake. Within individual studies, available data on sediment-based bioconcentration factors for various organisms show a wide variation among species for a specific contaminant (Kaag et al. 1998; Roesijadi et al. 1978a; Roesijadi et al. 1978b).

Accumulation of both organic and metal contaminants can be passive due to adsorption onto the organism, or it can be an active process driven through respiration. "Case-dwelling" species of benthic invertebrates have been thought less susceptible to contaminants than "free-living" organisms since the bioconcentration factors (BCFs) have been found to be quite different for metals like copper and zinc. Similar differences have been found for oligochaete and amphipod tissue concentrations for PCBs and hexachlorobenzene.

Sediment type can profoundly influence the bioavailability of sediment-sorbed chemicals. Many researchers have reported an inverse relationship between chemical availability and sediment organic carbon content (Elder et al. 1996; Augenfield and Anderson 1982). There also appears to be a smaller, not as well defined relationship between sediment particle size and chemical availability. In fine-grained sediment, this is most likely due to the increased surface area available for adsorption and the reduced volume of interstitial (Adams et al. 1985). Chemicals sorbed to suspensions of organic particles (both living, such as plankton, and non-living) may constitute sources of exposure for filter-feeding organisms and may be important in deposition. This pathway may be significant, as these organisms have been shown to accelerate the sedimentation processes by efficiently removing and depositing particles contained in the water column (Chen et al. 1999).

Several water quality conditions influence bioaccumulation of contaminants: temperature, pH, redox, water hardness, and physical disturbance. In addition, metals in mixtures may also compete for binding sites on organic molecules, resulting in antagonistic effects (e.g., cadmium and zinc, silver and copper).

The biological community itself can strongly influence the physical-chemical environment in the sediment, and in turn, affect the bioavailability of contaminants. For example: primary productivity influences the pH, which can influence metal chemistry; sulphate reduction by bacteria facilitates sulphide formation; the reduction of oxygen by organisms and their activities to anoxia affects redox conditions, and with it, metal redox conversion; the production of organic matter that may complex with contaminants; bioturbation influences sediment-water exchange processes and redox conditions; and methylation of some metals such as mercury.

Water-based BCFs indicate that benthic invertebrates generally accumulate to higher concentrations than do fish. This may be attributed to the greater degree of exposure of the benthic invertebrates at the sediment-water interface than fish. Biomagnification occurs when contaminant concentrations increase with successive steps in the trophic structure. However, well defined trophic levels may not exist in the aquatic ecosystem under examination, especially ones experiencing (or that have experienced) anthropogenically generated loadings of various contaminants (Kay 1984; Russel et al. 1999). In addition, individual species may occupy more than one trophic level during the life cycle. These factors not only complicate process and exposure understanding, they also complicate monitoring program designs necessary to document improvement after remediation has taken place. However, "it is no longer sufficient to know only whether chemicals accumulate because bioaccumulation itself is not an effect but a process. Regulatory managers must know whether the accumulation of chemicals is associated with or responsible for adverse affects on the aquatic ecosystem and human health" (US EPA 2000).

It was previously assumed that chemicals sequestered within sediment were unavailable to biota, and therefore posed little threat to aquatic ecosystems. Although, this is clearly incorrect, the presence of a contaminant (nutrient, metal or organic) in the sediment does not provide *a* *priori* evidence of ecological effect. In addition, a detailed understanding of the relevant processes and a quantification of the associated impacts is critical prior to developing a management plan.

III. Aquatic Ecosystem Effects of Contaminated Sediment

While laboratory and field studies are not overwhelming in number, both the risk and the actual impairment to organisms, including humans, have been conclusively established (Geisy and Hoke 1989; Burton 1992; Ingersoll et al. 1997). Biota exposed to contaminated sediment may exhibit increased mortality, reduced growth and fecundity, or morphological anomalies. Studies have also shown that contaminated sediment can be responsible for mutagenic and other genotoxic impairments (Lower et al. 1985; West et al. 1986). These effects are not restricted to benthic organisms - plankton, fish, and humans are also affected both from direct contact and through the food chain.

Metals, in their inorganic forms, do not appear to biomagnify appreciably in aquatic ecosystems; however, methylated forms of metals, like mercury, do biomagnify. But, the factors controlling the transfer of mercury from the sediment, especially monomethylmercury (the most bioaccumulative form of mercury) to aquatic organisms is poorly understood (Mason and Lawrence 1999). Most persistent toxic organics demonstrate biomagnification to lesser or greater degrees; however, it appears that biomagnification is not as dramatic within aquatic food chains as terrestrial ones. Also, it appears that where the phenomenon does occur, the biomagnification factors between the lowest and highest trophic levels are usually less than one order of magnitude

(Kay 1984). Recent investigations confirm that there is no simple relationship between contaminant concentrations in the sediment and bioavailability; however, observed toxic effects are related to the internal concentrations of certain chemicals (Kaag et al. 1998).

Nuisance algal growth and nutrient relationships in lakes are well documented, with phosphorus being cited as the limiting nutrient in freshwater systems. Some phosphorus is released from the bottom sediment during spring and fall lake circulation in dimictic lakes. In shallow, polymictic lakes, sedimentary phosphorus release may be more frequent, creating greater nuisance problems with the infusion of nutrients to overlying water, especially during summer recreational periods. This influx of nutrients usually results in abundant, undesirable phytoplankton growth, reducing water transparency, increasing color, and in severe cases, seriously depleting dissolved oxygen and potentially leading to fish kills. In order to prevent this stored release, the bottom sediments need to either be removed (dredged) or isolated from the water column (capped).

Nau-Ritter and Wurster (1983) demonstrated that PCBs desorbed from chlorite and illite particles inhibited photosynthesis and reduced the chlorophyll - *a* content of natural phytoplankton assemblages. In a similar study, Powers et al. (1982) found that PCBs desorbed from particles caused reduced algal growth as well as reduced chlorophyll production. The time course for desorption and bioaccumulation appears to be quite rapid, with effects being documented within hours after exposure (Harding and Phillips 1978). The rapid transfer of PCBs and other xenobiotic chemicals from particulate material to phytoplankton has significant ramifications because it provides a mechanism for contaminants to be readily introduced to the base of the food web.

The detrimental effects of contaminated sediment on benthic and pelagic invertebrate organisms have been demonstrated in several laboratory studies. Prater and Anderson (1977a and b), Hoke and Prater (1980) and Malueg et al. (1983) have shown that sediment taken from a variety of lentic and lotic ecosystems was lethal to invertebrates during short-term bioassays. Tagatz et al. (1985) exposed macrobenthic communities to sediment-bound and water-borne chlorinated organics, and found similar reductions in diversity to both exposures. Chapman and Fink (1984) measured the lethal and sublethal effects of contaminated whole sediment and sediment elutriates on the life cycle of a marine polychaete, and found that both sources were capable of producing abnormalities, mortalities, and reduced-derived benzo[a]pyrene has been shown to result in the formation of potentially mutagenic and carcinogenic metabolites in depositional feeding amphipods (Reichert et al. 1985). Other sublethal effects may be more subtle; for example, infaunal polychaetes, bivalves, and amphipods have been shown to exhibit impaired burrowing behavior when placed in pesticide-contaminated sediment (Gannon and Beeton 1971; Mohlenberg and Kiorboe 1983). Some observations have linked contaminants in sediment with alterations in genetic structure or aberrations in genetic expression. Warwick (1980) observed deformities in chironomid larvae mouthparts, which he attributed to contaminants. Wiederholm (1984) showed similar deformities in chironomid mouthparts ranging from occurrence rates of less than 1% at unpolluted sites (background) to 5-25% at highly polluted sites in Sweden. Milbrink (1983) has shown setal deformities in oligochaetes exposed to high sediment mercury levels.

Fish populations may also be impacted by chemicals derived from contaminated sediment. Laboratory studies have shown that fathead minnows held in the presence of contaminated natural sediment may suffer significant mortalities (Prater and Anderson 1977 a and b; Hoke and Prater 1980). Morphological anomalies have also been traced to contaminated sediment associations with fish. Malins et al. (1984) found consistent correlations between the occurrence of hepatic neoplasms in bottom-dwelling fish and concentrations of polynuclear aromatic hydrocarbons in sediment from Puget Sound, Washington. In addition, Harder et al. (1983) have demonstrated that sediment-degraded toxaphene was more toxic to the white mullet than to the non-degraded form. These studies illustrate the potential importance of sediment to the health and survival of pelagic and demersal fish species, but do not necessarily indicate a cause and effect relationship.

While we can expect that fish will be exposed to chemicals that desorb from sediment and suspended particles, the relative contributions of these pathways to any observable biological effects are not obvious. Instead, laboratory bioassays and bioconcentration studies are often required as conclusive supporting evidence. The Elizabeth River, a subestuary of the Chesapeake Bay, is heavily contaminated with a variety of pollutants, particularly PAHs. The frequency and intensity of neoplasms, cataracts, enzyme induction, fin rot, and other lesions observed in fish populations have been correlated with the extent of sediment contamination. In addition, bioaccumulation of these same compounds in fish and resident crabs was also observed. However, essential laboratory studies were not conducted to establish contaminants in sediment as the cause of the observed impairments (USEPA 1998).

There have been few examples of direct impacts of contaminated sediment on wildlife or humans. Some recent studies have established these direct links with ducks and tree swallows (Hoffman et al. 2000; Secord et al. 1999). For the most part, the relationship is largely inferential. Bishop et al. (Bishop et al. 1995; Bishop et al. 1999) found good correlations between a variety of chlorinated hydrocarbons in the sediment and concentrations in bird eggs. They felt this relationship indicated that the female contaminant body burden was obtained locally, just prior to egg laying. Other studies by Bishop et al. (1991) indicated a link between exposure of snapping turtle (*Chelydra s. serpentina)* eggs to contaminants (including sediment exposure) and developmental success (Bishop et al. 1998). Other investigations of environmentally occurring persistent organics have shown bioaccumulation and a range of effects in the mudpuppy (*Necturus maculosus*) (Bonin et al. 1995; Gendron et al. 1997). In the case of humans (*Homo sapiens*) there is only anecdotal evidence from cases like Monguagon Creek, a small tributary to the Detroit River, where incidental human contact with the sediment resulted in a skin rash. For the most part, assessments of sediment-associated contaminant impacts on the health of vertebrates (beyond fish) are inferential. This approach is known as risk assessment, and it involves hazard identification, toxicity assessment, exposure assessment, and risk characterization (NAS 1983).

USEPA Superfund risk assessments, which are aimed at evaluating and protecting human health, are designed to evaluate current and potential risks to the "reasonably maximally exposed individual" (USEPA 1989). Both cancer and non-cancer health effects for adults and children are evaluated. Data for the evaluation include concentrations of specific chemicals in the sediment, water column, and other media that are relevant to the potential exposure route. These routes of exposure may include: ingestion of contaminated water, inhalation of chemicals that volatilize, dermal contact, and fish consumption. The media-specific chemicals of potential concern are characterized based on their potential to cause either cancer or non-cancer health effects, or both. Once the "hazards" have been identified, the prescribed approach is continued to include toxicity evaluation, exposure assessment, and risk characterization. All of this leads to a potential remedial action, which itself follows a set of prescribed rules.

"Ecological risk assessment (ERA) is the estimation of the likelihood of undesired effects of human actions or natural events and the accompanying risks to nonhuman organisms, populations, and ecosystems" (Suter 1997). The structure of ERA is based on human health risk assessment (HHRA), but it has been modified to accommodate differences between ecological systems and humans. "The principal one is that, unlike HHRA, which begins by identifying the hazard (e.g., the chemical is a carcinogen), ERA begins by dealing with the diversity of entities and responses that may be affected, of interactions and secondary effects that may occur, of scales at which effects may be considered, and of modes of exposure" (Suter 1997). Risk characterization is by weight of evidence. Data from chemical analyses, toxicity tests, biological surveys, and biomarkers are employed to estimate the likelihood that significant effects are occurring, or will occur. The assessment requires that the nature, magnitude, and extent of effects on the designated assessment endpoints be depicted. More recent work has focused on the development of, and the relationship between assessment of measurement endpoints for sediment ecological risk assessments. In addition, scientists active in the field have strongly recommended that a weight-of-evidence approach be used (Ingersoll et al. 1997).

It is apparent that rarely is the relationship between a particular contaminant in the sediment and some observed ecological effect straightforward. Physical, chemical, and biological factors are interactive, antagonistic, and highly dynamic. These things often preclude a precise quantification of the degree of ecological impairment or effect attributable to a contaminant present in the sediment, and therefore, the degree of ecological improvement or benefit that can be achieved through remediation. Precision in quantifying impairment, remediation, and recovery is always improved through a better understanding of both the specifics of ecosystem functioning, as well as the behavior of the chemical(s) of concern in that particular ecosystem. "Technically feasible goals require adequate knowledge about basic processes and reliable methods to effect repairs" (Cairns Jr. 2000). Although a basic understanding of aquatic ecosystem function and chemical fate is generally available, it is evident that systems appear to be sufficiently unique and our understanding sufficiently lacking that an adaptive management approach to the mitigation of contaminated sediment is the prudent course to follow. This requires a much tighter coupling of research, monitoring, and management in every case to develop quantifiable realistic goals and measures of success to achieve them.

IV. Ecological Response to Sediment Removal

Although other sediment remediation techniques have been employed, such as capping and *in situ* treatment, sediment removal or dredging has been used longer and more extensively, not only for navigational dredging purposes, but also for environmental mitigation. Sediment removal has been used as a management technique in lakes as a means of deepening a lake to improve its recreational potential, to remove toxic substances from the system, to reduce nuisance aquatic macrophyte growth, and to prevent or reduce the internal nutrient cycling which may represent a significant fraction of the total nutrient loading (Larsen et al. 1975). Below are some examples of the removal of sediment contaminated by a nutrient (phosphorus), a metal (mercury), and a persistent toxic organic compounds (PCBs and PAHs) from lakes, rivers, and embayments.

A. Nutrients

Lake Trummen, Sweden, is one of the most thoroughly documented dredging projects in the world. An evaluation of the effectiveness of the dredging, whose main purpose was to reduce internal nutrient cycling and enrichment through sediment removal, took place over a twenty year plus time frame.

Lake Trummen, with a surface area of approximately 1 km^2, a drainage basin of some 12 km^2, and a mean depth of 2 m, was originally oligotrophic; however, it became hypertrophic after receiving both municipal and industrial discharges over a long period of time. In order to rectify the problems, both municipal and industrial waste effluents were curtailed in the late 1950s; however, the lake did not recover. In the late 1960s, extensive research was undertaken, resulting in the removal of some 400,000 m^3 of surface sediment (the top meter, in two 50 cm dredgings) from the main basin in 1970 and 1971.

Bengtsson et al. (1975) indicated that post-dredging water column concentrations of phosphorus and nitrogen decreased drastically and that the role of the sediment in recycling nutrients was minimized. Phytoplankton diversity increased substantially, while at the same time their productivity was significantly reduced. The size distribution of phytoplankton also shifted to much smaller cells, and water column transparency more than tripled. The troublesome blue-green algal biomass was drastically reduced, with some nuisance species disappearing altogether (Cronberg 1975). Conditions in the lake had improved to such a degree by the mid 1970s that an additional research and management program was undertaken on the fish community. From the late 1960s throughout the 1980s, an extensive monitoring program was maintained. By the mid

1980s, this program not only documented a deterioration in water quality, but also the ecological response to the change; and it also helped to ascertain that the changes were due to increased nutrient inputs from the atmosphere and the surrounding drainage basin.

Similar sediment removal projects have been conducted in other areas: Vajgar pond in the Czech Republic, Lake Herman in South Dakota, and Lake Trehorningen in Sweden, just to name a few. The latter named project is of particular note, because although there were significant decreases in the water column concentrations of phosphorus, it remained too high to be algal growth limiting. As a result, algal biomass remained the same as before the dredging was undertaken. This illustrates the importance of having a good understanding and quantification of ecological processes prior to undertaking a remediation project. In addition, Peterson (Peterson 1982) notes that through the early 1980s there was little evidence to support the effectiveness of sediment removal as a mechanism of ecological remediation. This lack of supporting research and monitoring data continues to be an obstacle to establishing the effectiveness of sediment cleanups.

B. Metals

Minamata Bay, located in southwestern Japan, is the site of one of the more notorious cases of metal pollution in the environment, and its subsequent impacts on human health. A chemical factory released mercury contaminated effluent into the Bay from 1932 to 1968. In addition to contaminating the water and sediment, methylated mercury accumulated in fish and shellfish. This resulted in toxic central nervous system disease among the individuals who ate these fisheries products over long periods of time. In 1973, the Provisional Standard for Removal of Mercury Contaminated Bottom Sediment was established by the Japanese Environmental Agency. Under this criterion, it was estimated that some 1,500,000 m^3 of sediment would need to be removed from an area of 2,000,000 m^2. Dredging and disposal commenced in 1977 along with an environmental monitoring program to ensure that the activities were not further contaminating the environment. Monitoring included measuring turbidity and other water quality variables, as well as tissue analysis of natural and caged fish for mercury residues. Dredging was completed in 1987, and by 1988 the sampling surveys provided satisfactory evidence that the goals had been achieved. Results of the ongoing monitoring showed that no further deterioration of water quality or increase in fish tissue concentration was occurring. By March of 1990, the confined disposal facility received its final clean cover. The total cost for the project was approximately $40-$42 million U.S.

Post-project monitoring provided clear evidence of a reduction in surficial sediment concentrations of mercury to a maximum of 8.75 mg/kg and an average concentration of below 5 mg/kg (national criterion is 25 mg/kg) (Ishikawa and Ikegaki 1980; Nakayama et al. 1996; Urabe 1993; Hosokawa 1993; Kudo et al. 1998). Mercury levels in fish in the bay rose to their maximum between 1978 and 1981, after the primary source had been cut off and some dredging had begun. Tissue concentrations declined slightly as dredging continued; however, they did fluctuate considerably. Fish tissue levels did finally decline below the target levels of 0.4 mg/kg in 1994, some four years after all dredging activity had ceased (Nakayama et al. 1996). These results demonstrate that mercury in the sediment continued to contaminate the fish and that removal or elimination of that exposure was essential for ecological recovery to occur. It also demonstrates that some impact (increased availability and increased fish tissue concentrations) could be associated with the dredging activity, and that a significant lag time from the cessation of remediation activity was necessary for the target body burdens to be achieved.

C. Persistent Toxic Organic Substances

PCB contaminated sediment remediation in Waukegan Harbor

Waukegan Harbor is situated in Lake County, Illinois on the western shore of Lake Michigan. Constructed by filling a natural inlet and portions of adjacent wetlands, Waukegan Harbor has water depths varying from 4.0 to 6.5 m. The harbor sediment is composed of soft organic silt (muck) which lies over medium, dense, fine-to-coarse sand.

Although substantial recreational use occurs in the area around the harbor, land use in the Waukegan Harbor area is primarily industrial. Of the major facilities present, the Outboard Marine Corporation (OMC) was identified as the primary source of PCB contamination in harbor sediment. U.S. EPA investigations in 1976 revealed high levels of PCBs in Waukegan Harbor sediment and in soil close to OMC outfalls. Concurrently, high levels of PCBs (above the U.S. Food and Drug Administration action levels of 2.0 mg/kg PCB) were also found in resident fish species. As a result, in 1981, the U.S. EPA formally recommended that no fish from Waukegan Harbor be consumed.

Remedial efforts in the harbor began in 1990, with harbor dredging conducted in 1992. Approximately 24,500 m^3 of PCB contaminated sediment was removed from the harbor using a hydraulic dredge. Approximately 2,000 m^3 of PCB contaminated sediment in excess of 500 mg/kg PCBs was removed from a "hot spot" that accounts for the majority of the PCBs on the site, and thermally extracted onsite to at least 97%. Soils in excess of 10,000 mg/kg of PCBs were also excavated and treated onsite by thermal extraction (Hartig and Zarull 1991). In all, 11,521,400 kg of material were treated, and 132,500 liters of PCBs were extracted and taken offsite for destruction, with a total cost of $20-25 million. No soils or sediment that exceeded 50 mg/kg PCBs remained onsite, except those within specially constructed containment cells.

Fish contaminant monitoring, conducted after the dredging in 1992, shows a substantial decrease for PCB concentrations in carp fillets. Figure 1 presents trend data for PCBs in Waukegan Harbor carp fillets. PCB levels in 1993 fish suggest that dredging did not cause significant PCB resuspension. Contaminant levels in 1993 fish averaged 5 fold lower than those tested in previous years up through 1991. Contaminant levels from 1993-1995 appeared to remain at these lower levels, but there is a suggestion of an apparent increase for the period 1996-1998. There is no statistically significant difference between the 1983 and 1998 levels of PCBs in carp (based on a two sample t-test).

As a result of the dramatic decline of PCBs in several fish species between the late 1970s and 1990s, the posted Waukegan Harbor fish advisories were removed, although fish advisories still exist for carp and other fish throughout Lake Michigan. The Illinois Lake Michigan Lakewide Advisory is protective of human health, as PCB concentrations in Waukegan Harbor fish are considered similar to those found elsewhere in Lake Michigan.

PAH contaminated sediment remediation in the main stem, Black River

The Black River enters the south shore of Lake Erie at Lorain Harbor, in north-central Ohio. The Black River drainage basin is dominated by agricultural and rural land uses (89%). Residential, commercial, and recreational uses make up the remaining 11%, and are concentrated in the lower regions of the river. The area has 45 permitted dischargers - 26 industrial and 19 municipal. The only industrial discharger that is considered to be "major" (discharging >1 million gallons/day) by the U.S. EPA is USS/KOBE Steel, located in the lower portion of the river. Until 1982, USS operated a coking facility, which is considered to have been the major source of PAH and metal contamination within the area.

A 1985 Consent Decree (U.S. District Court - Northern District of Ohio 1985) mandated USS/KOBE Steel Company to remove 38,000 m^3 of PAH contaminated sediment from the main stem of the Black River. The goal of the sediment remediation project was to remove PAH contaminated sediment in order to eliminate liver tumors in resident brown bullhead populations.

Tests from 1980 confirmed the presence of elevated levels of cadmium, copper, lead, zinc, cyanide, phenols, PAHs, oils, and grease in sediment adjacent to the former USS steel coke plant outfall. PAH concentrations in this area totaled 1,096 mg/kg (Baumann et al. 1982). Tests also confirmed the presence of low levels of pesticides (DDT and its metabolites) in both the main stem and the harbor regions. This sediment exceeded U.S. EPA's Heavily Polluted Classification for Great Lakes harbor sediment. As a result, all main stem and harbor sediment dredged during U.S. Army Corps of Engineers maintenance operations required disposal in a confined disposal facility.

High sediment PAH levels corresponded to a high frequency of liver tumors in resident populations of brown bullheads. Although sediment PAH levels had declined since the USS's coking facility was shut down, levels were still of concern.

Sediment remediation occurred upstream of the federal navigational channel in the vicinity of the coke plant outfall. Dredging of the sediment began in 1989. A total of 38,000 m^3 of sediment were removed during the operation. This action was completed in December 1990 and cost approximately $1.5 million for the dredging and containment of the sediment.

The primary cleanup target was the removal of sediment in the area of the former USS coke plant to "hard bottom", or the underlaying shale bedrock. No quantitative environmental targets or endpoints were established, although post-dredging sampling was required to test for remaining areas of elevated PAH concentrations. Prior to dredging, PAH concentrations ranged from 8.8-52.0 mg/kg within Black River sediment. As a result of dredging, PAH concentrations in sediment declined (Table 1).

PAH levels in brown bullheads, which had been monitored since the early 1980s (Baumann et al. 1982; Baumann and Harshbarger 1995; 1998), suggest some very interesting relationships between liver neoplasms and the dredging of sediment. Figure 2 illustrates the prevalence of hepatic tissue conditions (cancer, non-cancer neoplasm, altered hepatocytes, normal) found in fish of age 3 in 1982 (during coke plant operations), 1987 (after coke plant closing, prior to dredging), 1992 (exposed to dredging as age 1), 1993 (exposed to dredging as young of year), and 1994 (hatching after dredging was completed).

The incidence of liver cancer in bullheads of age 3 decreased between 1982 and 1987, corresponding with decreased PAH loadings following the coke plant closure in 1982. There is general consensus that the increase in liver cancer found in the 1992 and 1993 surveys is a result of PAH redistribution which occurred during the 1990 dredging efforts. No instance of liver cancer was found in 1994 samples of age 3 brown bullheads. Further, the percent of normal liver tissues increased from 34% to 85% between 1993 and 1994. This elimination of liver tumors and the increase in the percentage of normal tissues in the resident brown bullhead populations as a result of sediment remediation provides substantial evidence of the efficacy of the remedial strategy.

Summary and Conclusions

Contaminated sediment has been identified as a source of ecological impacts in marine and freshwater systems throughout the world and the importance of the contaminated sediment management issue continues to rise in all industrialized countries. In many areas, dredging or removal of sediments contaminated with nutrients, metals, oxygen demanding substances and persistent, toxic, organic chemicals has been employed as a form of environmental remediation. In most situations, however, the documentation of the sediment problem has not been quantitatively coupled to ecological impairments. In addition, the lack of long-term, post activity research and monitoring for most projects has impeded a better understanding of the ecological significance of sediment contamination.

Establishing quantitatively the ecological significance of sediment-associated contamination in any area is a difficult time- and resource-consuming exercise. It is, however, absolutely essential that it be done. It will likely be used as the justification for remedial and rehabilitative action(s), and also as the rationale for proposing when intervention is necessary in one place but not another. Bounding the degree of ecological impact (at least semi-quantitatively) provides for realistic expectations for improvement if sediment remediation is to be pursued. It should also provide essential information on linkages that could be used in rehabilitating other ecosystem components such as fish or wildlife habitat.

The lack of information coupling contaminated sediment to specific ecological impairments has, in many instances, precluded a clear estimate of how much sediment requires action to be taken, why, and what improvements can be expected to existing impairment(s) over time. Also, it has likely resulted in either a delay in remedial action or abandonment of the option altogether.

A clear understanding of ecological links not only provides adequate justification for a cleanup program, it also represents a principle consideration in the adoption of non-intervention alternative strategies. In developing this understanding, it is important not only to know the existing degree of ecological impairment associated with sediment contaminants, but also the circumstances under which those relationships and impacts might change (i.e., contaminants become more available and more detrimental).

Since contaminated sediment remediation often costs millions of dollars per area, adequate assessment, prediction, and monitoring of recovery would seem obvious. However, experience has shown that this is not always the case, particularly for prediction and monitoring of ecological recovery. This would never happen in business world and shouldn't in the environmental management field.

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Table 1. PAH concentrations (mg/kg) in Black River sediment in 1980 (during coke plant operations), 1984 (coking facility closed, pre-dredging), and 1992 (post-dredging)

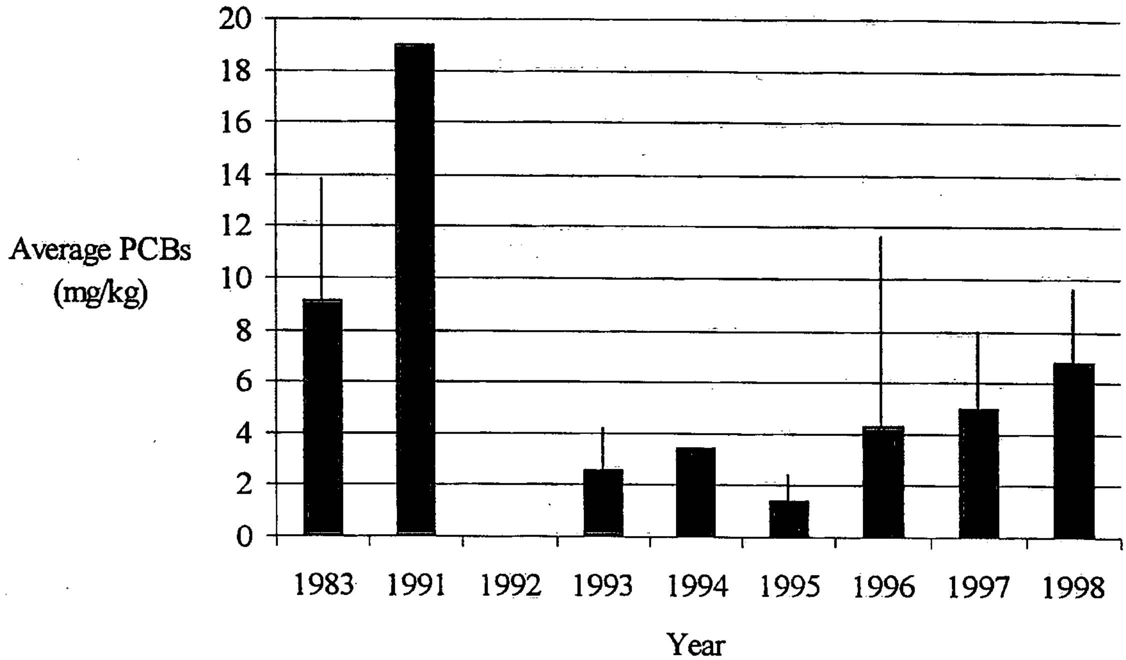
PAH compound	1980	1984	1992
Phenanthrene	390.0	52.0	2.6
Fluoranthrene	220.0	33.0	3.7
Benzo(a)anthracene	51.0	11.0	1.6
Benzo(a)pyrene	43.0	8.8	1.7

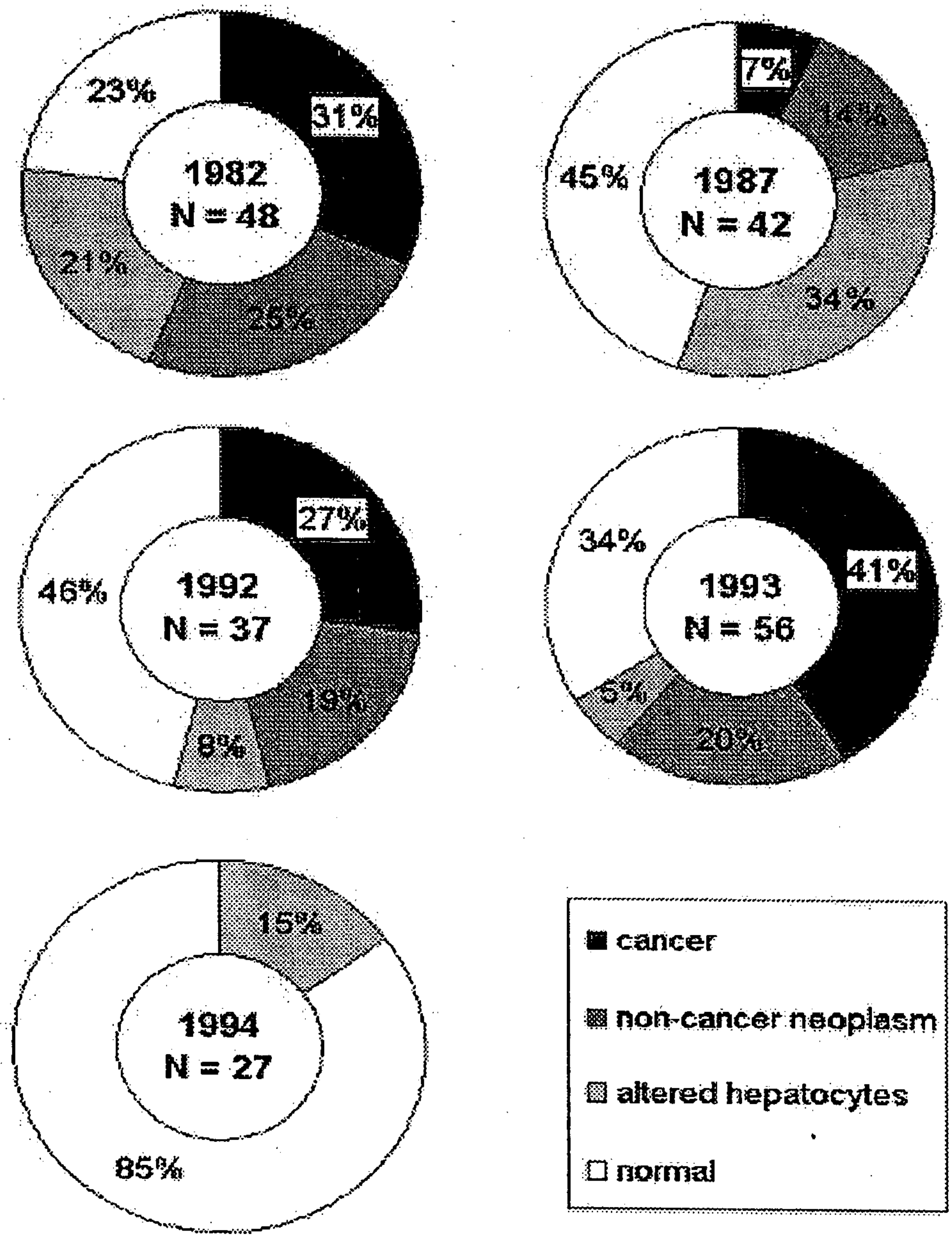
(USS coking facility closed down in 1982, dredging occurred from 1989-1990)

FIGURE LEGENDS

Fig. 1. Average PCB Levels, with 95% Confidence Intervals, in Waukegan Harbor Carp Fillets (U.S. EPA and Illinois EPA, unpublished data) [1991, one sample only; 1992 dredging occurred, no sampling].

Fig.2. Percentage of Age 3 Brown Bullheads from the Black River Having Various Liver Lesions (adapted from Baumann and Harshbarger 1998) [1987 Post Coking Plant Closure; 1990 Dredging Took Place].


Average PCBs (mg/kg)
20
18
16
14
12
10
8
6
4
2
0
1983
1991
1992
1993
1994
1995
1996
1997
1998
Year


1982
N = 48
31%
25%
21%
23%
1987
N = 42
7%
14%
34%
45%
1992
N = 37
27%
19%
8%
46%
1993
N = 56
41%
20%
5%
34%
1994
N = 27
15%
85%
cancer
non-cancer neoplasm
altered hepatocytes
normal

Environment Canada Environnement Canada

Canada

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