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Options for the Remediation of Contaminated Sediments in the Great Lakes

Great Lakes Water Quality Board. Sediment Subcommittee. Remedial Options Work Group

International Joint Commission. Great Lakes Regional Office

Report to the
Great Lakes Water Quality Board

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by the

Sediment Subcommittee
and its
Remedial Options Work Group

International Joint Commission
Great Lakes Regional Office
Windsor, Ontario

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DISCLAIMER

This report to the Water Quality Board was carried out as part of the activities of the Sediment Subcommittee and its Remedial Options Work Group. While the Board supported this work, the specific conclusions and/or recommendations do not necessarily represent the views of the International Joint Commission, the Water Quality Board or its committees.

PREFACE

This report reviews and evaluates technologies for the remediation of contaminated sediments. It is not intended to be a cookbook which tells the reader what must be done for each area being considered. It does, however, critically evaluate each technology and provide guidance on choosing a particular option. Although precise costs could not be ascribed to each technology, dollar ranges based where possible on actual costs are provided. In addition, the factors or parameter affecting the cost of a particular technology are discussed.

This report is one of two documents which present guidance on contaminated sediment issues. The companion report, prepared by the Assessment Work Group, examines the methods for assessing the nature, severity and extent of impairment due to contaminated sediments. This information is essential prior to examining and choosing remedial options.

An earlier draft of this report was presented to the Remedial Action Plan Coordinators in Toledo, Ohio in November 1987.

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1.0 SUMMARY AND RECOMMENDATIONS

SUMMARY

This report reviews and recommends options for the remediation of contaminated sediments in the Great Lakes basin.

Technologies were evaluated based on environmental and regulatory criteria, long- and short-term management requirements and the present feasibility of the technique. These criteria, combined with other factors including cost, social/political considerations, and various site specific characteristics including the type and severity of contamination and physiography, will affect the choice and implementation of a particular option.

Three broad categories of options for dealing with contaminated sediments are: leave them in-place, remediate *in situ* or remove them by dredging. At present, there are no technologies which have been used within the Great Lakes basin to remediate *in situ*, large volumes of contaminated sediments. Therefore, the only currently available options are to dredge contaminated sediments or leave them in-place.

While considerable experience and the technology to dredge large volumes of sediment are available, disposal of this material remains the primary problem. At present, only shore-based confined disposal facilities (CDFs) and upland landfill disposal of the entire volume of dredged material appear to be possible for large-scale operations. However, the increasing lack of available and suitable sites to build CDFs and locate safe landfills presents a major dilemma, which demonstrates the urgent need for the development of alternative techniques.

Although some concentration (separation), inactivation and destruction techniques are operational elsewhere, they remain largely at the laboratory (bench) or pilot scale stages in the Great Lakes. This report examines many different techniques which might be applicable for the remediation of contaminated sediments in the Great Lakes Areas of Concern.

RECOMMENDATIONS

1. Technologies for remediating contaminated sediments, which are already being employed outside the Great Lakes basin, require immediate evaluation for their application in the Great Lakes.

2. Technologies, at or near the pilot scale stage, need to be comparatively tested in one or more field demonstration projects.

3. Promising technologies, currently under investigation in the laboratory, need greater encouragement and financial support.

4. Further research on in situ treatment techniques, in particular, and techniques for the inactivation and destruction of contaminants in general, is necessary.

5. Current dredging and disposal practices require continued review and refinement.

2.0 INTRODUCTION

Contaminated sediments represent a global environmental issue of major ecological and economic proportions. As the repository for many anthropogenic and natural substances, they present resource managers with problems of both current and historic origins. Armed with a limited understanding of the ecological hazards, humans are now faced with developing technology to ameliorate conditions and at the same time minimize the long- and short-term ecological impacts of these corrective actions. The situation in the Great Lakes basin differs from elsewhere in the world only in our limited experience with newer technologies.

As recently as 1987, the Great Lakes Water Quality Board identified in their report to the International Joint Commission that 41 of the 42 Areas of Concern in the Great Lakes basin have contaminated sediments. In order to address the management of contaminated sediment in the Great Lakes, the Water Quality Board brought together a group of experts from the basin to form the Sediment Subcommittee. The subcommittee was asked to review existing protocols for the assessment of contaminated sediments and existing technologies for the remediation of identified problems. It was also asked to recommend a common approach for both assessment and remediation (Appendix III).

The subcommittee has responded to the needs to improve the evaluation of contaminated sediment associated problems and to begin remediation by identifying, evaluating and disseminating information on methods of assessing contaminated sediment impacts and remedial methods in two reports.

The report, Procedures for the Assessment of Contaminated Sediment Problems in the Great Lakes (IJC 1988), provided a means of examining the impacts of contaminated sediment. This report provides information on how remediation of contaminated sediments should proceed once they are implicated as contributors to problem(s) in an Area of Concern. This is a very broad question encompassing social, economic, technical and other issues; however, this report is limited to the technical aspects of remedial options - the set of tools which are available to remediate a specific site. The information is organized to instruct the reader on the nature and attributes of various techniques for managing contaminated sediments. The report does not provide a direct means of prescribing a remedial program for a specific Area of Concern. Instead, a site-specific feasibility analysis, which includes the above information, is necessary. Some guidance on option application is provided in the text and is presented diagramatically in Figure 1.

Features specific to an Area of Concern will influence the feasibility and desirability of each remedial option. The distribution and volumes of contaminated sediment, types and concentrations of contaminants, availability of land for upland disposal, availability of certain types of dredges, and water depths are among some of those features. These need to be considered in conjunction with the attributes of the various remedial options in the technical feasibility analysis. The technical and economic implications of

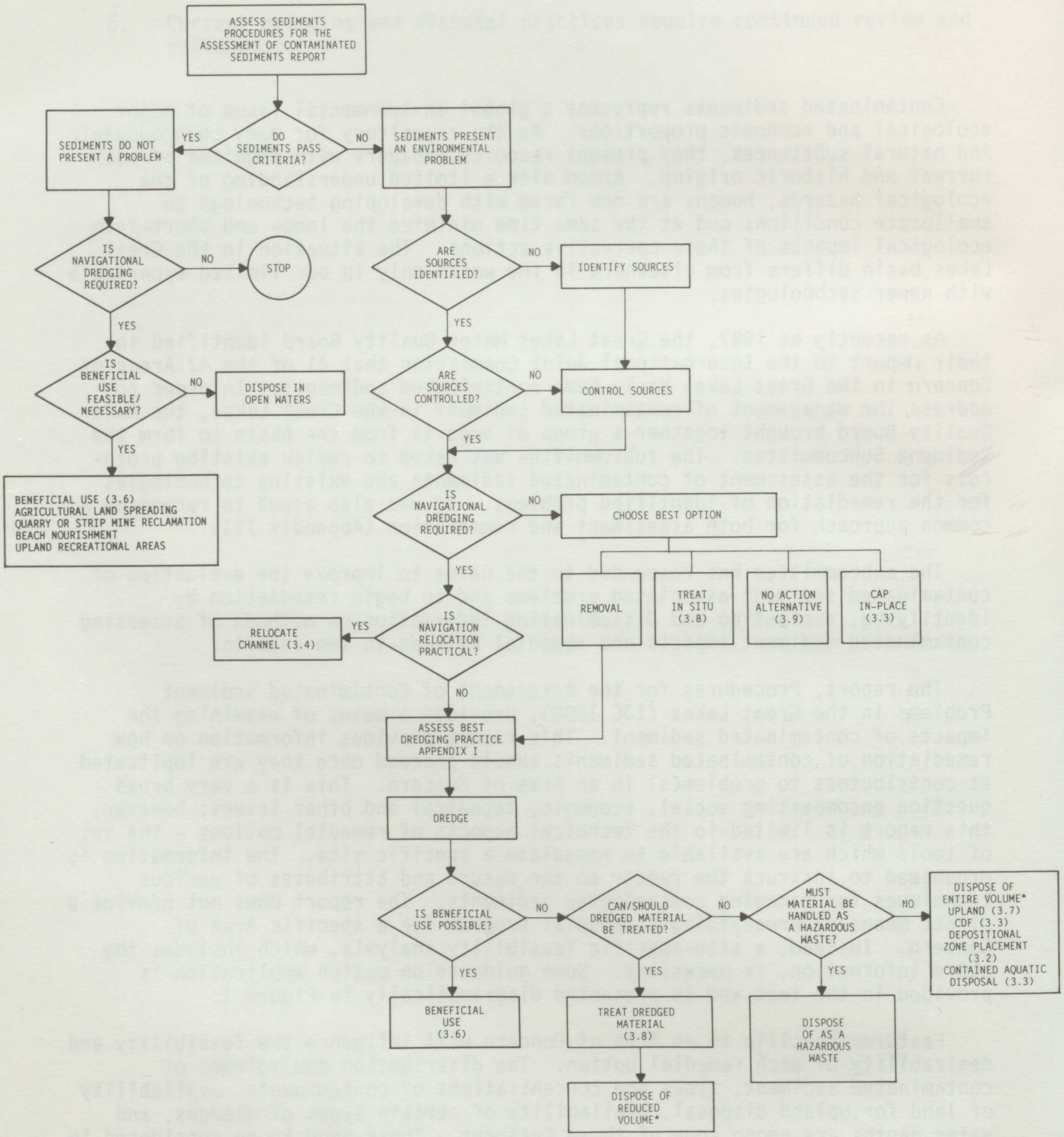

FIGURE 1: SEDIMENT REMEDIAL OPTIONS

regulations and laws pertaining to the maintenance of waterways, management of solid and hazardous waste and environmental protection, which vary among jurisdictions, also need to be considered. The analysis may show that a combination of options are necessary and that further information may be required before remediation can take place.

The remedial options described in this document are based upon available information and data collected through different jurisdictions (i.e. states and provinces) which have an interest in polluted sediments and their remediation. A description of each option is provided, followed by an evaluation of its present feasibility and the criteria which are felt to affect it. Considerable experience and related information exist for some procedures, such as dredging and the disposal of contaminated spoils; however, considerably less knowledge exists on the use of other procedures, which appear feasible on a laboratory scale but require further field evaluation or pilot scale testing. Some of these latter options include the decontamination of polluted sediments after dredging, and isolating the sediment in place. Some options are not suitable for sediments containing high concentrations of contaminants, but are suitable for sediments which are relatively clean and have to be removed for navigational purposes.

Considerable additional support is needed for the development of applicable technology. Until that technology is developed for full-scale operation, only one method for intervention is available in the Great Lakes basin; therefore, only two options exist. Contaminated sediments must either be left alone or removed by dredging.

The disposal of the entire dredged volume of sediments from contaminated areas presents further problems. CDFs have been constructed for the disposal of navigationally dredged spoils and cannot be legally used for the disposal of other materials. In addition, it is unlikely that the present CDFs can either accommodate the large volumes of sediment which would be generated from remediation dredging or are suitably located to cost-effectively dispose of contaminated sediments from many of the Areas of Concern. It is also questionable whether the construction of the additional CDF capacity, even using the most environmentally sound construction techniques, will be acceptable in the context of the ecosystem rehabilitation/protection management approach. Furthermore, the only other disposal option for contaminated material is an upland landfill. This disposal option is even more volume limited and has much higher costs. In addition, landfilling may lead to both short- and long-term environmental/ecological problems.

The contaminated sediments can either be placed on a nonerosional bottom or in a lake or ocean pit. In capping the polluted sediments, the contaminants are isolated from the effects of the biophysical environment which could act to release pollutants from the sediment to biota or the water column.

The use of clean sediments to cover contaminated sediments or dumped dredged material has been considered seriously in recent years. The results of the experimental work in deep water placement, followed by covering with clean material at Port Credit, Ontario are inconclusive and it is doubtful that this procedure can be adopted as a special case technique based on current experience.

Feasibility

Technical factors need to be considered when deep water disposal is being considered. Feasibility will be affected by the volume of material to be disposed of, as well as its physical and chemical character. The availability of a suitable disposal site is also an important factor.

Preliminary assessment requirements in the vicinity of the dump site include the physical and chemical analysis of the local sediment, and an evaluation of the hydrodynamic regime of the area. The dumping of sediment may result in elevated concentrations of suspended solids in the water column at various depths, which could be harmful to fish. If the material is discharged very quickly, or if the material consists of large particles, limited dilution and dispersion occurs. A sediment dispersion model should be used to predict the fate of the dumped material. The characteristics of the receiving environment (turbidity, temperature, direction and speed of currents) and the particle size of the dredged sediment will affect the size of the dispersion plume after disposal of the sediment. From a habitat protection viewpoint it is preferable that the bottom sediment and the disposal material be compatible.

Success in placing caps and keeping them in place depends primarily on the positioning of the discharge (of cap material), the geometry of the disposal mound, the physical nature of the materials, the ratio of volume of cap material to contaminated material, the physical topography at the dump site and the hydrologic regime. It is important to ensure that all contaminated sediment is covered by clean material. This can be checked by taking core samples, or by monitoring with an underwater camera.

In addition to specific environmental factors, the assessment of disposal site suitability will be based on accessibility and cost.

Environmental/Regulatory Criteria

In Canada, the Ontario Ministry of the Environment has a set of guidelines which address the disposal of sediment in open water. If the chemical analyses of the parameters exceed the guidelines the material is considered unsuitable for open water disposal. Deep water disposal and capping operations, however, are not addressed in these guidelines. The physical placement of material in open water is subject to Federal Regulations under the Navigable Water Protection Act.

In the United States, the guidelines and criteria which apply to this contaminated sediment management option are those promulgated relative to Secton 404 of the Clean Water Act which regulate the placement of dredged or fill materials into navigable waters. This regulatory program is administered by the U.S. Army Corps of Engineers (who have coordination responsibilities for the respective states), the U.S. Environmental Protection Agency, and the U.S. Fish and Wildlife Service. Public review is required by the National Environmental Policy Act (NEPA).

Individual states have various policies ranging from prohibition of open water disposal to a case-by-case consideration based on contaminant levels and deep water placement requirements.

Long-Term and Short-Term Management

Short-term management considerations consist mainly of dump site monitoring, directed at verification of the initial predictions regarding the fate of the disposed and/or capped material. Both sediment and benthos sampling should be carried out at the dump site. Sediment sampling should address the verification of the model used for sediment dispersion, prediction, and defining the size or extent of the dump site area. The primary objective of the monitoring program should not only be to evaluate the existing condition of the environment, but also should be designed to permit an evaluation of temporal changes at the site. Further verification of the above could be undertaken by in situ biological sampling and possibly laboratory bioassay studies of dump site sediments to evaluate effects. It should be noted that the high cost of this option is associated not only with dumping, but also with the monitoring.

3.3 SUBAQUEOUS PLACEMENT AND CAPPING (Contained Aquatic Disposal)

Description of Option

Contaminants found in river or harbor sediments may be effectively isolated by subaquatic burying. This option involves the capping (covering) of contaminated sediments with cleaner, less contaminated sediments. Although it is technically feasible to cap highly contaminated sediments in-place, at their original location, conflicting uses such as navigation and the cost of relocating that use may dictate that contaminated sediments be moved from their original site of deposition. Contaminated sediments can be collected from various and disparate sites, placed in a smaller area, and subsequently capped (or buried) in order to achieve isolation. Dredging and dredged material disposal techniques are used to accomplish these tasks. The term Contained Aquatic Disposal (CAD) has been coined in U.S. Army Corps of Engineers publications to describe this option.

It is also technically feasible to cap contaminated sediments in place at their original location. The following discussion may be applied in either case, in regard to the material to be capped, the capping material and the placement of the capping material.

When relocating contaminated sediments it is normally considered desirable to minimize the physical size of the placement through precise deposition. Deposition is controlled through careful selection and operation of the dredging equipment. Precise placement of the dredged materials is complicated, however, by the quantities of materials involved, their density (percent solids), and the difficulty of positioning equipment. Deposition is further complicated by the lack of direct visual contact with the bottom. Precise placement of materials can be accomplished with careful control of a variety of operational factors, including good navigational control of the depositing ship or barge and the maintenance of a relatively consolidated material mass through mechanical dredging or the fitting of low velocity diffusers on the discharge ends of hydraulic pipelines. The most effective means of controlling subaquatic placement can be achieved by the preliminary preparation of the disposal site through excavation of an underwater 'hole,' into which the contaminated materials are placed and subsequently covered.

A predisposal site can be dredged with either mechanical or hydraulic equipment. The hole which is formed should be defined sufficiently to allow identification of its specific boundaries for subsequent disposal of contaminated materials. The material from this excavation should be relatively clean and it can be disposed of through unconfined means, or may be stockpiled for subsequent covering of the hole after contaminated materials have been placed in it. Predisposal excavation can be considered a method to create a superior disposal site and one which generates cleaner material for covering the contaminated material. If the surrounding sediments are unsuitable, then a cover layer of cleaner sediments from elsewhere or other earthen material must be available for placement over the contaminated material.

The sediments which are to be capped should be relatively dense and consolidated in order to support the weight of the capping material. If the materials which are to serve as a cap are denser than the materials to be covered, the capping materials are liable to sink through the contaminated sediments, leaving them uncovered. Gunnison et al. (1987) reported that "attempts to cap sediments having densities (percent solids) below 40% are presently interpreted to mean that clamshell dredging, rather than hydraulic dredging, gives the better substrate of contaminated dredged material for a capping operation. However, this recommendation does not necessarily mean that all clamshell-dredged contaminated sediments are suitable for capping. On occasion, some modifications may be required to increase the density of the contaminated dredged material, decrease the density of the cap material, or otherwise prevent the capping material from sinking into the underlying contaminated dredged sediment."

Although either mechanical or hydraulic methods may be used to place contaminated sediments into the underwater hole, each case should be evaluated based on sediment and capping material characteristics and disposal site considerations, to determine the most appropriate type of equipment to use. While mechanical dredging and placement can result in the deposition of a highly consolidated mass of materials, there is a certain amount of sediment resuspension into the overlying water column (albeit transient) as the materials fall through the water column. Direct placement of the contaminated materials, at a specifically defined disposal site, can also be accomplished through pipelines which are outfitted with diffuser discharge heads to provide for minimum discharge velocities and, therefore, rapid settling of the discharged solids and their associated contaminants. This is the preferred method.

The cover must provide a physical barrier to isolate the contaminated sediments from contact with the biota in the overlying aquatic environment. Thus, it must be of sufficient thickness to prevent chemical diffusion and mechanical breaching of the cover. Mechanical breaching can be caused by wave scour and the burrowing of aquatic organisms such as clams and worms. Gunnison et al. (1987) have described how laboratory testing of three parameters in bench type tests can be used to determine the minimum cap thickness necessary to provide for chemical isolation of contaminated sediments. Depletion of dissolved oxygen, the release of ammonium-nitrogen and occasionally the release of orthophosphate-phosphorus were found to be effective predicters to determine the minimum thickness of capping materials needed to provide chemical isolation of contaminated sediments.

In most cases, organic contaminants found in sediments are much less mobile than ammonium or dissolved oxygen. Thus, a cap thickness that is effective for these inorganic constituents will also be effective for organic contaminants, which are normally strongly bound to the fine grained particles and the oils and greases common to highly contaminated sediments. Organic contaminants which are more strongly bound to the sediments than these inorganic indicators include polynuclear aromatic hydrocarbons (PAHs), petroleum hydrocarbons, and polychlorinated biphenyls (PCBs). In tests on a limited number of sediment samples at the U.S. Army Corps of Engineers' Waterways Experiment Station, the ability to successfully cap contaminated sediments was found to be dependent on the relative densities of the contaminated sediments and the materials to be used as their caps. Fine grained sediments have been found to be a more effective capping material than coarse grained, sandy material.

Minimum cap thickness needed to prevent physical disturbance to buried contaminated sediments should normally be a function of the maximum burrowing depth by benthic organisms found in the region and erosive forces due to currents and turbulence. The depth of biological penetration can be determined through benthic community investigations or from the first hand knowledge of aquatic biologists regarding the habits of the local benthic communities. Erosive forces are a function of wave height and water depth, and the currents generated can be measured with current meters.

Gunnison et al. (1987) described the minimum cap thickness needed to achieve total isolation of the underlying contaminants as being equal to the sum of the individual cap thicknesses which would each be needed to achieve both physical and chemical isolation. It is necessary to sum these two values in order to preclude burrowing organisms from penetrating the zone of chemical diffusion in the cap.

In order to successfully cover and contain contaminated sediments, many operational factors must be coordinated. These factors, as listed by Truitt (1987a), are identified in Table 1.

Feasibility

Subaqueous capping has been conducted using mechanical dredging techniques in Long Island Sound and the New York Bight, New York, and the Duwamish Waterway in the state of Washington. These cases have shown that capping is technically feasible and that the caps are stable under normal tidal, wave

Long-Term and Short-Term Management

Laboratory and field verification studies have demonstrated that capping of contaminated sediments can be effective in short, medium and long time frames for preventing the movement of contaminants into the water column and biota (Brannon et al. 1986).

Close short-term monitoring, hydrographic survey, is required to assure that the contaminated sediments are placed in the proper location and that the subsequent capping completely covers the contaminated sediments to the minimum capping thickness required. Long-term monitoring, hydrographic survey, is required to assure that the capping material remains in place. Additional post-remediation monitoring is needed to assure that contaminated sediments have been effectively isolated from the water column.

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3.4 NAVIGATION RELOCATION

Description of Option

The alteration or relocation of navigational channels and docking facilities in urban-industrial harbours can be practiced, on a limited scale, as a means of dealing with contaminated sediments. This practice may involve the partial closure of docking slips through construction of a berm or sheet pile wall, followed by the removal of contaminated sediment from the open portion and placement of the material in the newly created containment area. The subsequent use of the newly created land will, in part, depend on the nature of contaminants present in the sediment/fill. It should also be noted that the resulting docking space will also be reduced in capacity and range of use. This type of procedure cannot be considered in isolation and will usually be undertaken as part of an overall strategy for waterfront development or redevelopment.

The procedure described below was used in Toronto Harbour in order to seal off a docking slip from the harbour:

1. Placement of a concrete rubble berm across the slip area displacing sediment and sealing the inside slope with fill to prevent escape of sediment during future filling (ensure that the berm is constructed and stabilized before going to Step 2);

2. Construction of an anchored soldier pile and precast concrete plank retaining wall along the outer limit beyond the rubble beam. Backfill with granular material;

3. Fill the remainder of slip, which has been contained by a rubble berm and a retaining wall, with concrete rubble having high void ratio to minimize displacement of sediment.

This procedure received endorsement from the jurisdictions. Monitoring of water and sediment quality occurred before, during and after the project.

<u>Feasibility</u>

This method is an interim solution until joint and diffuse pollution sources become reduced or completely eliminated. The major advantage is that contaminated sediments are confined and therefore not disturbed during shipping activities and sufficient time will be available for the assessment of degree of contamination and design of proper treatment.

Sediment resuspension and generally degraded local water quality may result from this activity; therefore, it is advisable to isolate the construction activity from the harbour area as a whole. This can be achieved through placement of a silt curtain and/or berm at the mouth of the slip or surrounding the activity area.

Once the material has been placed in the confined area, followup investigations must take place on the nature of contamination present in the soil. Depending on the use to which the land base may be put, testing may have to be undertaken.

Final cover of the material in the Toronto situation had the following characteristics (representing site-specific criteria which will vary according to the nature of the contaminants present).

1. The material should have a particle size distribution of at least 80% clay with the remainder being sand (ranging from fine to coarse particle sizes);

2. The texture of the material should be sandy clay with a permeability no greater than 1.0×10^{-6} cm/sec and with a bulk density of 0.90 g/mL;

3. The cover material should be placed to a depth of at least two feet and the surface area must be contoured to prevent surface runoff and prevent ponding or infilling of water;

4. Once the area is stabilized and contoured it could be vegetated (depending on the results of phytotoxicity tests).

The time frame for the above activity was six months from project concept and design through implementation and overtopping of the area. Limitations involve geographic extent of the area to be filled as well as a knowledge of physio-chemical aspects of the sediment to be removed. Geographically, this approach can be applied to dock faces and slips (no larger than 20,000 m^2). Since a fair amount of water will be retained in the impounded area, provisions must be made for removal and treatment of the water (pumping, holding tanks, treatment and release) depending on sediment and water quality at the time of containment. This option can be used in a limited manner and in conjunction with other harbour-wide remedial options.

Environmental/Regulatory Criteria

The material must exceed open water dredge spoil disposal criteria in order for one to consider this option. Once the area is impounded and filling is complete, tests will have to be undertaken in order to determine the nature and depth of cover material to be used.

In the United States, Section 404 of the Clean Water Act governs the use of this option, in Ontario, a set of guidelines for lakefilling are being developed.

Long-Term and Short-Term Management

Additional leachate tests, partitioning tests and limited bioassays of the contained material are required in order to identify the need for vegetative cover and the final use of the created land.

This option is a one time, short-term option which can be used to deal with localized contamination. It will require monitoring over a period of time to ensure that no contaminant pathways exist directly to the water body or indirectly through the cover material. If vegetation of the cover area is allowed to occur, some monitoring of plant material will be necessary.

3.5 SHORELINE AND IN-WATER CONFINED DISPOSAL FACILITIES (CDFs)

Description of Option

At present, almost all contaminated dredged material is disposed of either at suitable upland sites or in engineered confined disposal facilities. Isolation of the material is achieved through placement in an area that has been specifically prepared and dedicated as a long-term storage or disposal site. The dedicated storage location is normally prepared by construction of perimeter dikes to withhold the contaminated materials.

Contaminated sediments are first collected and removed from their original site of deposition by dredging (Appendix I provides a discussion on the selection of dredging equipment). These contaminated sediments are then transported and deposited in the confined disposal facility. The specific design of the CDF, the specific type of dredging equipment to be used, the method of transportation, and the operation of the CDF must be tailored to site specific circumstances in order to insure that contaminants of concern are captured, deposited into and retained by the CDF at minimum cost.

The design and construction methodology for a CDF depends on many factors such as physical characteristics of the dredged material, type and level of contaminants present in the sediment, dredging method, design life of the facility and site-specific considerations such as its location, wave climate and availability of construction material. A more detailed discussion of factors to be considered for disposal of dredged material and their interaction is included in Miscellaneous Paper D-85-1, "Management Strategy for Disposal of Dredged Material: Contaminant Testing and Controls," U.S. Army Waterways Experiment Station (Francingues et al. 1985). A typical CDF consists of a diked enclosure with one large cell for disposal of material, and adjoining cells for retention and decantation of turbid supernatant water. In a mechanical dredging project, the material is usually double handled into the facility in a considerable dewatered state, thus the provision of a decant cell is not required.

Some CDFs have been constructed adjacent to existing breakwaters, incorporating the breakwater as a portion of the containment structure. Some have been built adjacent to the existing shore to take advantage of the shoreline to form a portion of the containment boundaries. Others, located offshore and entirely in the lake and without being attached to any other structure, have formed new, man-made islands. Upon completion of filling operations, the deposited contaminated sediments are covered with a layer of clean fill material. The extent and thickness of clean fill is dependent on the type and level of contaminants present in the sediments. Ultimately, a top vegetative cover is provided for stabilization and to minimize erosion.

Advantages of sites located in the water include maintenance of a saturated soil condition in the lower levels, a relatively neutral hydraulic gradient relative to groundwater, greatly reduced land costs, the ability to locate disposal sites near sources of contaminated sediments (thereby minimizing transportation costs), and public concern is frequently minimized as the sites are not adjacent to or near private residential property. Disadvantages include a high cost of construction due to the need to protect outer dike faces from the severe erosive forces of Great Lakes wave action and the need to provide long-term maintenance to insure the structural integrity of the perimeter dikes. Long-term maintenance is required to repair dike damage caused by the constant erosive action of wave forces and periodic dislodging of armor stone by severe storm waves on the outer faces of the structures.

Containment dikes for maintenance dredging CDFs have been constructed with a variety of designs and methods. A preliminary consideration is to locate the containment dikes where the soils have sufficient bearing capacity to support the weight of the dikes. In-lake and near-lake soils in the Great Lakes sometimes contain sediment type material with high percentages of water and poor bearing capacity. A safety factor of at least 1.3 was used in construction of the Pointe Mouille, Michigan CDF dikes. To achieve this level of bearing capacity certain areas were prepared by excavating as much as 12 feet (3.7 m) of the existing soils to reach firm supporting substrate. Dikes for in-water and shoreline sites have typically been constructed as trapezoidal forms of prepared limestone, covered by boulder sized armor stones to protect the interior core from erosive wave action. The tops of these dikes should be high enough to prevent overtopping by all but the most severe and infrequent waves, and they should typically be wide enough to accommodate vehicular traffic for maintenance and inspections. Dikes have also been made of quarried rock with rip rap and armor stone protection on the exposed sides of the perimeter berms. Dikes constructed of quarried rock, including interlocking steel sheet piling, protected by armor stone, have demonstrated greater permeabilities than dikes made of prepared limestone.

Attempts to reduce dike permeability through the use of fabric cloth and plastic liners have had poor success. Tearing and punctures caused by gravel, stones and the use of heavy equipment in dike construction as well as uneven settling and compaction may cause breaching of these liners. Installation of such liners can be further complicated by flow back and forth through the dike walls and working in saturated or underwater environments. However, sand, soil and sediment placement along the interior dike faces has been found to significantly reduce dike permeabilities. Lining interior dike faces with layers of clay can be very effective in reducing dike permeability. Bentonite-cement slurry trench walls may also be effective at reducing permeability.

Discharge conduits and weirs should typically be provided in cases where perimeter dikes have been constructed with low permeability and in case dikes with higher permeability become clogged with finer solids such as sands, silts or clays. Uncontrolled water flow over dike tops will cause erosion damage and ultimately destruction of the containment walls. Discharge conduits and weirs should be provided with oil skimming and flow control to allow for collection of floatables and to increase settling times if needed. For CDFs used in maintenance dredging, the permeability of dike walls typically precludes the necessity of having to use discrete discharge conduits. In cases where dike walls have low permeabilities and CDF pond waters are considered too turbid for discharge, a variety of techniques may be used to increase effluent quality. Polymer addition has been successfully demonstrated as a method of reducing suspended solids in CDF discharge water. Filter cells for effluent polishing have also been used. A cross-section of a shore-based CDF is shown in Figure 2.

Against the overall landscape, CDFs are fairly large, relatively isolated areas with a variety of physical characteristics attractive to various species of fish and wildlife. As accumulating sediments rise above the level of the interior pond water, the sites will become colonized by a wide variety of opportunistic plant and animal species. Over 145 species of birds have been found on Great Lakes CDFs; gulls, terns, herons, egrets, shorebirds and waterfowl are common. Because these sites are relatively isolated and undisturbed by human presence, CDFs are typically colonized as nesting sites. Nesting colonies of gulls, terns and black-crowned night herons have established themselves in the Saginaw and Pointe Mouille CDFs among others. The shallow water and mud flat areas of CDFs can cause waterfowl botulism problems. Labour-intensive responses to discourage waterfowl use has been found to be effective in response to these problems. Fish populations, trapped through original construction and introduced with waters from dredging, are typically present in the interior pond water. These fish have been found to accumulate significant concentrations of organic lipophilic contaminants found in the sediments. These fish populations may be eliminated

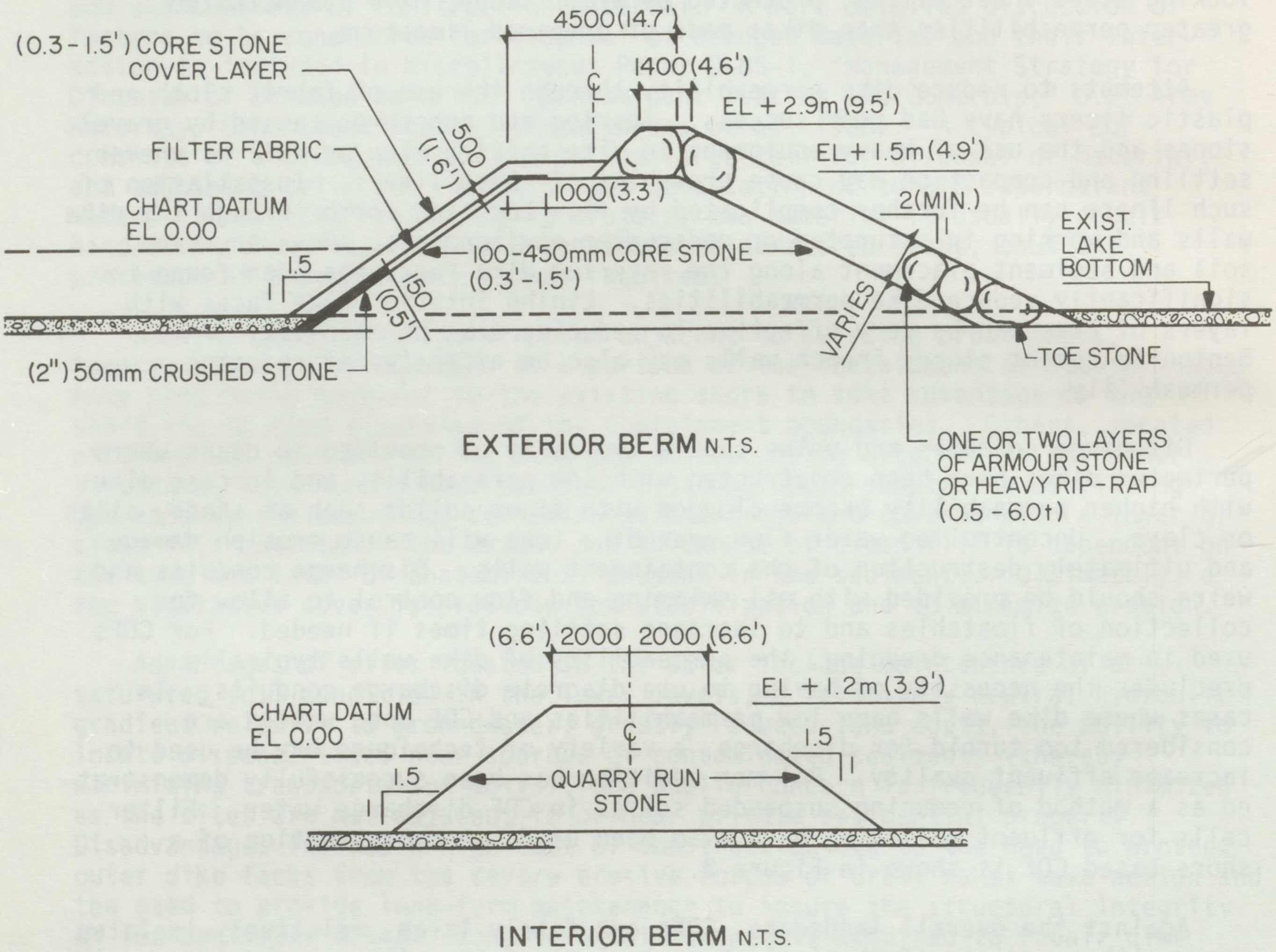


FIGURE 2: TYPICAL BERM CROSS-SECTIONS

periodically through the use of fish poisons such as rotenone. Outside dike faces are also attractive as fish habitat. To date, no problems have been identified regarding contaminant accumulations due to fish using the outside of CDF dikes.

Feasibility

The first CDF to come into use in the Great Lakes was the Grassy Island site for containment of contaminated sediments from the Rouge River, Michigan navigation project in 1960. Use of CDF sites has increased significantly in the Great Lakes since 1970; most sites were constructed between 1972 and 1979. Considerable experience has been gained and improvements made in the design, construction, operation and maintenance of such facilities over this period. Significant improvements also have been made to ensure structural integrity of containment dikes so that minimal loss of contaminants occurs from the CDFs. Monitoring work has shown that CDFs, if properly designed and operated, have succeeded in isolating and preventing polluted sediments from re-entering the lakes. In short, technical and environmental feasibility of CDFs have been well established from experience gained over the years. There are, at present, five long-term sites in Canada and 30 long-term sites in the United States portion of the Great Lakes. The cost of building and operating CDFs are dependent on their size, mode of operation and a host of site-specific factors. The cost of CDFs built in Canadian portions of the Great Lakes have ranged anywhere from \$3.00 to \$10.00 per cubic meter (\$2.30 to \$7.65/yd³) of capacity, which would be comparable to any other mode of disposal. In some instances, the cost of CDFs were more than offset by the value of land created in the process. Costs of construction for the United States sites range from \$.50 to \$15.00 per cubic meter (\$.38 to \$11.47/yd³) capacity. The most typical unit costs range from \$1.00 to \$5.00 per cubic meter (\$.76 to \$3.86/yd³); nearly 60% of the facilities have construction costs within this range. However, in a significant portion of the facilities (about 23%), construction costs exceeded \$10 per cubic meter (\$7.65/yd³) capacity. Costs for siting, engineering, land acquisition, dredging, transportation of contaminated sediments to the site and long-term maintenance need to be added in order to determine the final unit costs for disposal with this alternative.

Environmental/Regulatory Criteria

No single legislation/act is specifically applicable to control of dredging activities including design, construction and operation of CDFs in the Canadian Great Lakes. In the absence of such a dedicated act/regulation, sections of other federal, provincial and local acts and regulations pertaining to adverse impacts on fisheries, terrestrial and water quality, etc., are used to regulate CDFs.

Ontario Ministry of the Environment regulations require that the following three basic criteria be met in design and construction of CDFs.

(a) Containment dikes be adequate to contain polluted dredged material under force of lateral pressure, seepage and erosion

(b) The quality and quantity of any supernatant/effluents from CDFs draining to a water course meet applicable standards and guidelines

(c) The native subsoils be adequate for containment and isolation of contaminants including protection of groundwater quality

Apart from the above, there are no other specific criteria or regulations dealing with CDFs. Criteria for the maintenance, closure and long-term use are being developed.

In the United States, the guidelines and criteria which apply to this contaminated sediment management option are related to Section 404 of the Clean Water Act, which regulates the placement of dredged or fill materials into navigable waters. This regulatory program is administered by the U.S. Army Corps of Engineers with coordination responsibilities for the respective states, the U.S. Environmental Protection Agency and the U.S. Fish and Wildlife Service. General public review is required by the National Environmental Policy Act (NEPA).

Long-Term and Short-Term Management

Laboratory and field verification studies have demonstrated that containment of contaminated sediments can be effective in the short, medium and long term. Management during the filling stages for CDFs should address concerns regarding the potential for pollutants to permeate through the dike walls. After the CDF is filled and sediment particles fill the voids of the interior dike surfaces, permeability is significantly reduced. While there is still pond water within the dike perimeter, fish populations in the pond water will accumulate and potentially transfer pollutants to colonizing bird populations. Additional attention may need to be directed to preventing outbreaks of waterfowl botulism during specific portions of the year. On a long-term basis, dike walls need to be periodically inspected and repaired to insure their structural integrity. Wave and ice action may cause extreme forces on the face of the dikes and cause periodic damage. In some cases it may be desirable to cap these facilities with cleaner layers of sediment to isolate the contaminants from the colonizing terrestrial community. Grasses or prairie type plant communities may be established to prevent root penetration by tree species if this is deemed necessary. Although no data are available, the presence of burrowing mammals may represent a problem to the contaminant integrity of the facility. However, these sites have the potential to serve as valuable wildlife habitat on a long-term basis due to their potential for isolation and freedom from significant human disturbance.

A major problem with CDFs is the lack of suitable space, in the midst of major ports and harbors, to build such disposal facilities. There is, therefore, an urgent need for research and development to establish alternative disposal techniques which would alleviate this growing problem. The ideal solution, but not yet practical at this stage, is to detoxify and immobilize contaminants present in the sediment before permanent disposal is undertaken.

REFERENCE

Francingues, N.R., M.R. Palarmo, C.R. Lee, and R.K. Peddicord. 1985. Management Strategy for Disposal of Dredged Material: Contaminant Testing and Controls, Miscellaneous Paper D-85-1, U.S. Army Waterways Experiment Station, Vicksburg, Mississippi.

International Joint Commission. 1986. Report of the Dredging Subcommittee to the Great Lakes Water Quality Board: A Forum to Review Confined Disposal Facilities for Dredged Materials in the Great Lakes. Windsor, Ontario. 97 pp.

3.6 BENEFICIAL UTILIZATION OF DREDGED MATERIAL

Introduction

The following discussions evaluate situations where the disposal of dredged sediment can be used to enhance or reclaim upland sites. These options are not suitable for materials classified as hazardous wastes, or in situations where toxicants can be mobilized.

Perhaps one of the best uses of dredged sediments high in organic silt and nutrients is for soil enhancement. The growing capacity of poor soils can be significantly improved by incorporating enriched sediments into their makeup. Moderately contaminated sediments (U.S. EPA guidelines) may be suitable for nonagricultural purposes such as landscaping or restoring barren land that has been severely graded along highways. Agricultural applications may be possible after determining that the contaminants will not be translocated into the crops. In some coastal areas, agricultural lands are protected by dikes. Raising the elevation of this land would decrease the need for dike maintenance. Reclamation of quarries and strip mined areas could also be a beneficial use of dredged sediments.

Upland disposal for recreational purposes has also been considered. In areas lacking topographic relief, the construction of a hill for water slides or sledding could provide a unique local recreational opportunity. This would facilitate the disposal of a large volume of sediment in a relatively small surface area. The major drawback of this use is the logistical problem of drying and stabilizing the material. Sediment erosion during construction could create some problems until adequate vegetative cover is established.

The use of relatively clean and sandy sediments for beach nourishment is a viable disposal option from areas where maintenance dredging is required. Sediments high in silt and nutrients can also be utilized if adequate protection is provided to minimize wave erosion. Unlike CDFs, a lesser degree of protection (i.e. silt fences) could be employed and at a much reduced cost. Re-establishing land forms lost during recent record high water conditions could be a high priority for some communities.

Transport and handling costs represent some of the factors to be considered for any upland use of dredged materials. However, these costs may be more acceptable when weighed against the benefits that can be derived from the use of this potential resource.

3.6.1 Agricultural Land Spreading

Description of Option

Several options have been studied for agricultural utilization of dredged materials:

A. Spread the dredged material as soon as it is dry enough to handle or pump the material directly from the dredge site to a predesigned site. The design of a site for liquid disposal may include perimeter diking and subtiling. Plowing or disking into the existing soils may facilitate stabilization of the site.

B. Mix dried dredgings with additional types of waste products such as water treatment lime sludges or wastewater sludges. This could produce a topsoil material which could be marketed for use in commercial and residential applications.

C. Remove and store the top soil from a farm field, place the dredged material either as a dry or wet material (as in A above) in a uniform layer, allow the site to dry, and then replace the topsoil. Again, the field should be tiled to promote drying.

Feasibility

The above options are being evaluated or used by the Toledo/Lucas County Port Authority in northwest Ohio. Sediment removal in the Maumee River and harbor area is the largest annual maintenance dredging operation in the Great Lakes system. Most of these sediments are deposited from agricultural runoff, and have high potential for reuse in agricultural applications.

Actual use is presently limited to Option B above. The pilot plots using this new topsoil were constructed in the summer of 1986 and raised excellent sod grass using various percentages of dredgings and sewage sludge; however, marketing the finished product for residential and commercial use has not met initial expectations. Efforts to fully utilize this option are continuing.

The logistics required to implement these options are mainly earth moving, mixing and spreading. The main constraint is the material has to be moved and whether the materials require single, double or triple handling. The design may include pumping facilities, dewatering facilities, stockpiling and mixing.

Time constraints for the alternatives outlined in this section depend on the availability of dried material for Item B. For procedures A and C, timing would depend on the availability of land (i.e. after crops are harvested). If the project requires significant engineering (i.e. pumps, force mains, and dikes), it will preclude crop production during the life of the project.

The primary limitation is usually cost. The basic costs used for estimating purposes are in the range of \$.50/cu. yd/mi (\$.41/m^3/km) for pipeline transportation. An additional cost is the pump, mooring station, which is estimated to be about \$500,000 and have a 20-year life. Pipe is estimated at \$185,000/mile (\$115,000/km).

Another limitation can be the presence of toxics in the dredged material. If the dredged materials are to be applied to an agricultural site, the concentration and type of contaminants must be considered. Some contaminants can be translocated from the soil and into the crop to be harvested. The application rate will be governed by the type of contaminants, the concentration, and the type of crop.

Heavy metal contamination to the crop yield is also controlled by the cation exchange capacity and the pH of the soil. Excessive levels of heavy metals at low pH levels and cation exchange rates can cause crop production or food chain problems.

U.S. EPA guidance documents for agricultural application of sludges are shown on the following tables when the soil pH is maintained at 6.5 or greater. These same guidelines should be followed when using contaminated dredged sediments.

SUGGESTED MAXIMUM CUMULATIVE AMOUNT OF SLUDGE METALS
WHICH CAN BE ADDED TO AGRICULTURAL LAND

	Soil Cation Exchange Capacity (milliequivalents/100g)					
	Less than 5		5-15		Over 15	
	Maximum Cumulative Amount of Metal					
Metal	lb/acre	kg/ha	lb/acre	kg/ha	lb/acre	kg/ha
Pb	500	560	1,000	1,121	2,000	2,242
Zn	250	280	500	560	1,000	1,121
Cu	125	140	250	280	500	560
Ni	125	140	250	280	500	560
Cd	(See below for cadmium limits)					

Cadmium

Cadmium is generally the metal that causes the most concern for sludge applications to cropland.

The U.S. EPA interim final guidelines provide for two approaches to controlling land application of sludges containing cadmium. The first approach consists of three requirements: 1) the pH of the sludge and soil mixture is 6.5 or greater at the time of application, except for sludge containing cadmium at concentrations of 2 mg/kg (dry weight) or less; 2) the annual application of cadmium from sludge does not exceed 0.4 lb/acre (0.5 kg/ha) on land used for the production of tobacco, leafy vegetables, or root crops grown for human consumption. For other food chain crops, the annual cadmium application rate should not exceed 0.4 lb/acre or 0.5 kg/ha.

The cumulative application of cadmium from sludges should not exceed the following levels:

Soil Cation Exchange Capacity	Maximum Cumulative Application			
	Background soil pH* Less than 6.5		Background soil pH 6.5 or greater	
(milliequivalents/100g)	lb/acre	kg/ha	lb/acre	kg/ha
Less than 5	4.5	5	4.5	5
5-15	4.5	5	8.9	10
Greater than 15	4.5	5	17.8	20

*If the pH of the sludge and soil mixture is adjusted to and maintained at 6.5 or greater whenever food-chain crops are grown then the levels of the next column are allowed.

The second approach consists of four requirements. These requirements are management oriented rather than providing specific cadmium levels: 1) the only food-chain crop grown is animal feed; 2) the pH of the sludge and soil mixture is 6.5 or greater and is maintained at this level during periods when food-chain crops are grown; 3) there must be a facility operating plan which demonstrates how the animal feed will be distributed to preclude ingestion by humans. Measures required to safeguard against possible health hazards from cadmium entering the food chain should be included in the operating plan; and 4) notify future property owners by a stipulation in land record or property deed that the property has received sludge at high cadmium application rates and that food-chain crops should not be grown.

Other metal specific considerations include:

Nickel

Recommended maximum addition levels of nickel by sludge application to cropland are shown in the previous table. Presently, toxicity of nickel to plants has only been observed in acid soils. If nickel concentrations are excessive then pH adjustment (upward) by lime addition should lessen the chances of plant damage.

Copper

Copper is useful to plants in small quantities but can be hazardous in excessive amounts to grazing animals (especially sheep). Recommended maximum accumulative levels by sludge application to cropland of copper are shown in the previous table.

Zinc

Small amounts of zinc are necessary for crop growth. Zinc toxicity in plants is uncommon. Sheep and cattle are susceptible to zinc toxicity at high zinc concentrations. If pH levels are maintained at or above 6.5, zinc toxicity should not be a problem.

Lead

Maximum recommended cumulative levels of lead by sludge application to cropland are shown in the previous tables. Toxicity of lead to plants is likely only when the soil/sludge mixture pH is less than 5.5. Plant uptake of lead is minimized by increased soil/sludge pH, cation exchange capacity, and available phosphorus.

The above limitations will determine the extent to which the dredged material can be safely applied. These limits will basically dictate whether land application is a viable alternative or whether the land requirements for such applications are a limiting factor.

<u>Environmental/Regulatory Criteria</u>

The above criteria consider only the heavy metals which are not additive. Organic concentrations are not commonly agreed upon or established at this time. Uniquely contaminated dredged materials (i.e. mercury, PCB, pesticides, etc.) will require special determinations on a case-by-case basis by the regulatory agency involved.

Long-Term and Short-Term Management

It is anticipated that the expense of purchasing a manufactured topsoil would make this option a self limiting 'one shot' application for commercial or residential use (Procedure B).

Procedures A and C could handle a thick layer of material in order to raise a field to a predetermined elevation and justify the cost of any permanent or long-term investment such as piping, dikes, etc. In Toledo, for example, it is estimated that a 500 acre (202 hectares) field could be used to store the entire volume of dredged material from the harbor and lake channel for five years if it were placed to a depth of six feet over the entire field.

No long-term management is anticipated for Procedure A, B or C once the material is dry and planted with appropriate cover crops.

3.6.2 Quarry or Stripmine Reclamation

Description of Option

The application of sewage sludges for reclaiming stripmined areas has been used successfully in many instances. Utilizing enriched dredged material in the same manner could be a viable disposal method. The dredged materials could be used as a 'top dressing' on the final grade or the bulk of the material could be used for filling a site. If a strip pit or quarry is selected to be filled, the site would be viewed as a solid waste disposal project.

Feasibility

This option must be viewed as a long-term disposal procedure. The stages in evaluating this option include the following:

A. Location and evaluation of potential sites

B. Transportation costs and feasibility to reach these sites

The material could be placed into an existing quarry as a liquid through pumping or as a solid from dump trucks, rail cars or barges. The engineering problems with standard surface transportation will be loading, unloading, and a staging area for drying the material prior to transporting it.

Costs for the use of quarries or stripmines would include transportation, handling and access to the site (ownership, lease, etc.). The value of the completed site could be a very significant factor in the cost feasibility of this kind of reclamation proposal. It has been estimated that rail transportation of 100-250 miles (161-402 km) would cost approximately $12.50 per cubic yard ($16.35 per cubic meter). A cost reduction might be available if the cars were used as part of a two-way haul; for example, hauling coal from a stripmine to a port facility and dredged material on the return trip to a nearby strip mine reclamation project. The limitations to this option are the availability of sites, the haul distance, and available modes and costs of transportation. The disposal site can be selected or modified to meet the needs of a wide range of contaminants.

A limestone quarry could present the best case for disposal of dredged material with elevated levels of heavy metals. The high pH of the surrounding rock will immobilize the metals and prevent leachate into the surrounding strata. However, the potential for impact on the surrounding groundwater may preclude use of such a facility unless elaborate control measures are taken. These control measures could include: artificial or recompacted clay liners, double liners and monitoring wells.

Utilizing a coal strip pit could present a different problem if it is an acidic environment. These conditions could cause dredged sediments with elevated metals to leach. With pH levels below 6.5 complexed metals can become soluble and could be mobilized. pH adjustment could be a remedy or leachate collection and treatment may be necessary.

If the reclaimed area is to be used for agricultural purposes, procedures described in section 2.6.1 should be considered. If food production is not a final use, less stringent controls could be applied. Based on the relatively low metals concentrations in some dredged materials, 1,000 dry tons/acre could be applied and not exceed the allowable loading rates described in section 2.6.1, (assuming Cd is the limiting metal and 4.0 mg/kg as a typical value).

Long-Term and Short-Term Management

While it is anticipated that either of these options would be able to accommodate a large volume of material, they would essentially be a 'one-shot' application or disposal facility. It may take a fairly long time to place the material within the site; however, once filled, the site would be finished off and a new site would need to be developed.

Long-term management implications might include monitoring wells and/or inspections of the area to insure that erosion does not become a problem.

3.6.3 Beach Nourishment

Description of Option

Shoreline features in many areas have been eroded due to increased wave action caused by higher lake levels. As these landforms are lost, fragile wetlands and marshes, agricultural land and residential properties are left vulnerable to seasonal and storm-related high water and wave action. Many vulnerable landforms could be re-established or protected by barriers partially constructed with dredged material.

One option is to construct a perimeter barrier around the area to be protected and reclaimed. This barrier could be constructed of various materials but would likely be armored or otherwise stabilized on the lake side. The back side could be constructed in a less stable fashion using silt fence or other means to keep the dredged material in place.

Another option would be to design shoreline protection using a sacrificial beach which would be renourished each year by dredging. An alternative would be to construct a nearshore structure with fill capacity.

A third option would involve reconstruction of barrier beaches if relatively clean sediments are available. Many marsh areas are threatened due to barrier beaches being breached by wave action and eventually submerged. The intense flooding and wave action in the marshlands could alter the area landforms and ecosystems. Placing dredged materials where the beaches used to be would provide replacement of eroded material and protection of fragile wetlands.

Feasibility

Because of the shallow nature of the areas involved in all of these options, pumping would probably be required. However, it is important to weigh the costs of these projects against the potential benefits that could be derived, i.e. flood protection, habitat enhancement and restoration of valuable shoreline. In addition, the creation of barriers, beaches or reconstructing eroded landforms will not result in a land cost as would occur in the acquisition of property for upland operations. Therefore, only the cost of dredged material transport to a shoreline reconstruction site is considered here.

The basic costs of pipeline transportation (power, maintenance) are on the order of $0.50/cu yd/mi ($.41/m^3/km). An additional cost is the pumping, mooring station which is estimated to be about $500,000 and have a 20-year life. Pipe is estimated at $185,000/mile ($115,000/km).

Environmental/Regulatory Criteria

These options could be utilized for protecting shoreline areas and would provide a beneficial use as opposed to open lake disposal. Low to moderately contaminated sediments can be open lake disposed in some areas. In the United States, the guidelines and criteria which apply to this contaminated sediment management option are those promulgated relative to Section 404 of the Clean Water Act. These guidelines regulate the placement of dredged or fill materials into navigable waters. This regulatory program is administered by the U.S. Army Corps of Engineers who have coordination responsibilities for the respective states, the U.S. Environmental Protection Agency, and the U.S. Fish and Wildlife Service. General public review is required by the National Environmental Policy Act (NEPA).

Guidelines are currently being developed in Ontario, under Regulaton 309 of the Environmental Protection Act, which will address the placement of material in, or abutting, a water body.

Long-Term and Short-Term Management

Those facilities protected by stable or armored dikes will weather the rigors of wind and water for many years without much maintenance. They present a finite amount of disposal space, not unlike a CDF. On the other hand, placing material with temporary or no confinement depends on lake levels, storms, and the establishment of vegetative cover for long-term stability. Erosion from year to year is quite likely, meaning that yearly renewal of eroded areas would be required in order to provide structural stability and inshore protection.

3.6.4 Upland Disposal for Recreation Areas

Description of Option

Upland disposal is an alternate to a conventional CDF. One potential upland use would be to construct a recreational hill on a relatively small amount of surface area, keeping land costs per cubic yard to a minimum while providing an end use. In an area lacking any topographic relief, the finished project could provide some unique recreational opportunities such as waterslides or winter sledding.

For example, a relatively square site of 160 acres (65 hectares) could be used to develop a recreational hill up to 240 feet (73 m) in height while providing disposal capacity of over 17,000,000 cubic yards (13,000,000 m^3). An additional 80 acres (32 hectares) would be needed for dewatering and staging after the dredged material is pumped to the site.

Feasibility

The engineering required is mainly concerned with pumping to the site and drying and moving the material into the final configuration. Also of concern would be the final load bearing capacity of the fill material. On-site earth moving would consist of conveyors and loading. Vegetative cover would be necessary to stabilize the site and prevent erosion and runoff.

Environmental/Regulatory Criteria

It is anticipated that with proper erosion and runoff control, any degree of contamination could be accommodated with the exception of that which would be classified as a hazardous waste.

Long-Term and Short-Term Management

This option could have a very long life in a rather limited space. Short-term management could be more of a problem than long-term management. The short-term problems include controlling water and wind erosion during construction and contaminant leaching. Once the site is stabilized and vegetative cover is established, a monitoring plan should be part of the long-term management of the site. This plan should include surface and groundwater sampling to ensure contaminants are not leaching from the hill.

3.7 Upland Confined Disposal

Description of Option

The landfilling of contaminated sediments has three stages:

1. dredging or other removal process;
2. transportation to a landfill; and
3. disposal in the landfill.

The dredging requirements of contaminated sediments destined for landfilling need not be different from those that may be associated with other sediment management programs. However, sediments to be landfilled may have significant dewatering requirements in order to reduce the quantity of material to be landfilled or, in some cases, to permit the sediments to be classified as a 'solid' for landfilling purposes. As described below, the landfilling of material classified for these purposes as 'liquid' is not allowed in either Ontario or the United States.

Transportation of sediments to a landfill requires compliance with applicable provincial, state and federal regulations. Landfilling must only occur in appropriately licensed facilities. The specific transportation and disposal requirements for contaminated sediments will vary according to the characteristics of the waste as determined for waste management purposes in a particular jurisdiction. These are described in the next section.

Feasibility

Landfilling of wastes is an established practice in many jurisdictions and is continually being improved through new standards and procedures. Although there are no technical or regulatory provisions prohibiting the disposal of contaminated sediments in landfills, there are three major reasons why the routine landfilling of contaminated sediments may be either impractical or undesirable from a public policy perspective.

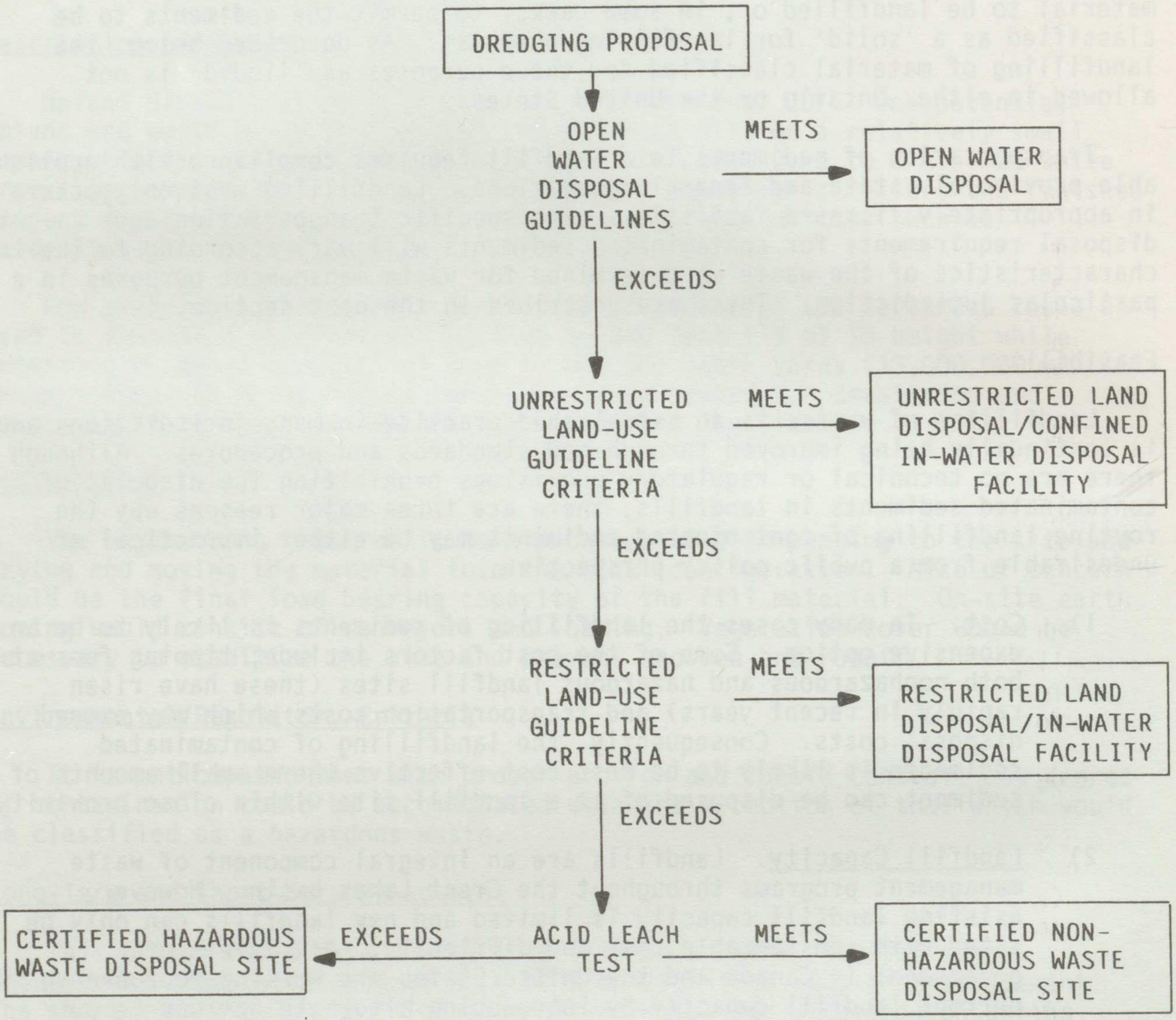
1) Cost. In many cases the landfilling of sediments is likely to be an expensive option. Some of the cost factors include; tipping fees at both nonhazardous and hazardous landfill sites (these have risen rapidly in recent years) and transportation costs which may exceed disposal costs. Consequently, the landfilling of contaminated sediments is likely to be most cost-effective where small amounts of sediment can be disposed of at a landfill site within close proximity.

2) Landfill Capacity. Landfills are an integral component of waste management programs throughout the Great Lakes basin. However, existing landfill capacity is limited and new landfills can only be sited with considerable cost and difficulty. All levels of government in Canada and the United States are working to conserve current landfill capacity by introducing alternate options (e.g. recycling) for many types of solid waste. Therefore, while there may be contaminated sediments for which landfilling is a viable management option, landfilling of these sediments on a routine basis would be counter to these efforts if other acceptable management options were available.

3) Selective Use. In some cases it may be possible to use contaminated sediments in a selective fashion. As Figure 3 indicates, and as discussed with reference to other management options, there are a range of uses for contaminated sediments. Instead of occupying a large volume of a landfill site, the disposal of this material can be phased over time, e.g. in a landfilling context, it may be possible to use sediments as daily cover material in place of topsoil that may otherwise be required.

FIGURE 3

DREDGED MATERIAL
CLASSIFICATION PROCESS



Source: Guidelines for the Management of Dredged Material in Ontario, Ontario Ministry of the Environment, 1987, Unpublished.

Environmental/Regulatory Criteria

All levels of government may have jurisdiction over different aspects of the landfilling of contaminated sediments or other wastes.

Canada

Primary responsibility for the oversight of landfilling rests with the provinces. In Ontario, the provincial Environmental Protection Act (EP Act) provides the regulatory framework for waste management in this province. In addition, Guidelines For The Management Of Dredged Material in Ontario has been developed to provide guidance on when dredged material that are contaminated should be considered 'waste' for landfilling purposes.

The landfilling options for contaminated sediments are a function of contaminant levels measured on a bulk and leachate basis, and on whether the material is classified as a solid or a liquid.

The classification process proposed in the guidelines to define when contaminated sediments should be considered as waste for landfilling purposes is given in Figure 3. As indicated elsewhere in this report, other management options are available for dredged materials provided that contaminant levels meet the appropriate guideline. However, if dredged sediment contaminant levels exceed the Restricted Land Use/Confined Disposal bulk analysis criteria given in Table 2, the sediments will generally be considered a 'waste' and disposal of these materials in a landfill may be appropriate, subject to the classification of the materials as nonliquid according to Regulation 309 of this EP Act (see below). Ministry staff are using these guidelines to assist in making decisions regarding the management of dredged material.

In addition, dredged sediments that are contaminated must be classified according to their leachate characteristics following the testing protocol given in Ontario Regulation 309. Where any of the parameters are present in the sediment leachate at 100 times or more the values given in Table 3, the material will be classified as a hazardous waste. Hazardous wastes must be disposed of at an approved hazardous waste disposal site. Ontario Regulation 309 also provides a classification protocol for establishing whether a waste is a liquid or a solid.

Nonhazardous, solid waste may be disposed of in a municipal landfill site. Hazardous solid waste must be disposed of at an approved hazardous waste landfill site; the facility owned and operated by Tricil near Sarnia is the only such facility in Ontario. All sediment wastes that are landfilled should be registered with the Ontario Ministry of the Environment in accordance with the procedure detailed in Section 15 of Ontario Regulation 309. Hazardous wastes should also be manifested in accordance with this regulation and all wastes must be transported by an approved hauler. Dredged sediments classified as liquid wastes cannot be landfilled in Ontario until they have been dewatered to the point that they are not classified as liquid waste according to Regulation 309.

TABLE 2

ONTARIO DREDGED MATERIAL CLASSIFICATION CRITERIA:
RESTRICTED LAND USE/CONFINED IN-WATER DISPOSAL FACILITY

PARAMETER	OPEN WATER DISPOSAL	UNRESTRICTED LAND USE	RESTRICTED LAND USE
<u>1A</u>			
Cadmium	1.0	1.6	4.0
Lead	50.0	60.0	500.0
Mercury	0.3	0.5	0.5
PCBs	0.05	2.0	2.0
<u>1B</u>			
Loss on Ignition	6.0		
Oil and Grease	1,500.0		
Total Phosphorus	1,000.0		
Total Kjeldahl Nitrogen	2,000.0		
Ammonia	100.0		
Grain Size			
Visual Description			
<u>2</u>			
Arsenic	8.0	14.0	20.0
Copper	25.0	100.0	100.0
Zinc	100.0	220.0	500.0
Chromium	25.0	120.0	120.0
Iron	10,000.0	350,000.0	350,000.0
Nickel	25.0	32.0	60.0
Cobalt	50.0	20.0	25.0
Silver	0.5		
Cyanide	0.1		
Molybdenum		4.0	4.0
Selenium		1.6	2.0

All units ppm unless otherwise specified.

*Additional parameters may be requested by the reviewer because of known dischargers.

TABLE 3

ONTARIO LEACHATE QUALITY CRITERIA (DRINKING WATER GUIDELINES X 100)

CONTAMINANT	CONCENTRATION (milligrams per litre)
2,4,5-TP/Silvex/2-(2,4,5-Trichloro-phenoxy) propionic acid	1.0
2,4-D	10.0
Aldrin + Dieldrin	0.07
Arsenic	5.0
Barium	100.0
Boron	500.0
Cadmium	0.5
Carbaryl/1-Naphthyl-N-methyl carbamate/Sevin	7.0
Chlordane	0.7
Chromium	5.0
Cyanide (free)	20.0
DDT	3.0
Diazinon/Phospordithioic acid, 0,0-diethyl 0-(2-isopropyl-6-nethyl-4-pyrimidinyl) ester	0.02
Endrin	0.02
Fluoride	240.0
Heptachlor + Heptachlor epoxide	0.3
Lead	5.0
Lindane	0.4
Mercury	0.1
Methoxychlor/1,1,1-Trichloro-2,2-bis(p-methoxyphenyl)ethane	10.0
Methyl Parathion	0.7
Nitrate + Nitrite	1000.0
Nitrilotriacetic acid	5.0
Nitrite	100.0
PCBs	0.3
Parathion	3.5
Selenium	1.0
Silver	5.0
Toxaphene	0.5
Trihalomethanes	35.0
Uranium	2.0

Note: The presence of any of these contaminants in dredged material leachate at or greater than the values given will result in the classification of the material as hazardous waste.

Source: Ontario Regulation 309, Schedule 4.

United States

There are many similarities between Ontario and the United States in the landfilling of waste, including contaminated sediments. These include the need to classify the sediments as solid or liquid and hazardous or nonhazardous waste, consideration of manifest listing for transportation and landfill approval requirements.

The principal legislation governing waste management in the United States is the Resource Conservation And Recovery Act (RCRA). This legislation sets out the waste management procedures and standards that must be met on a national basis. However, individual states may implement waste management programs that complement the RCRA program or which replace it if the U.S. Environmental Protection Agency (EPA) deems that the state program is at least equivalent to the federal program. The landfilling of sediments in the states is therefore subject to the minimum requirements of the federal legislation, but requirements may vary according to individual state programs.

RCRA provides for the classification of hazardous waste, the definition of solid and liquid waste and requirements for the permitting of hazardous and nonhazardous waste landfills. Sediments may be classified as 'hazardous' according to their leachate characteristics as defined by the Extraction Procedure (EP) Toxicity Test. Sediments classified as nonhazardous waste may be disposed of in landfills approved under Subtitle D of RCRA; sediments classified as hazardous must be disposed of in landfills approved under Subtitle C of RCRA. Hazardous wastes must be registered with the U.S. EPA prior to transportation or disposal. Manifesting of these wastes is also required and transportation must be undertaken by an approved hauler.

Liquid wastes, as defined by the Paint Filter Liquid Test specified in RCRA, may not be landfilled in the United States. Contaminated sediments classified as liquid waste by this test must be solidified by dewatering, or some other means, if they are to be landfilled.

In both Canada and the United States, sediments may contain contaminants of concern for which hazardous waste classification criteria have not been developed. Where landfilling of these sediments is proposed, regulatory authorities in both countries may require testing to establish the nature and extent of these contaminants. Landfilling of these sediments may only be permitted in hazardous waste disposal facilities.

International

Wastes generated in Canada may be disposed of in the United States and vice versa. In order to expedite the transfer of hazardous wastes across the international boundary and the proper management of these wastes in the receiving county, a prenotification program has been established between the two countries. This requires the waste generator to notify the federal authority in that country (either Environment Canada or the U.S. EPA) of both the intent to export the waste and of the transporter and receiver of the waste in the receiving country. The waste cannot be shipped until the receiving country approves acceptance of the waste. This process may take up the 60 days.

Landfill Site Ownership

While federal, provincial and state agencies are responsible for regulating waste management, these agencies do not own or operate waste management facilities. Landfills may be owned and operated by the private sector or, in the case of nonhazardous waste landfills, by municipalities.

3.8 TREATMENT

Introduction

Dredging of sediments has been carried out in the Great Lakes mainly to maintain navigation channels; however, dredging of contaminated sediments is now being considered as one remedial option available for the rehabilitation of the aquatic ecosystem. Before 1970 dredged material from the Great Lakes harbours and waterways was disposed of by open water dumping (the most economical method) at designated locations. Since the early 1970s, this practice has only been permitted when an environmental assessment indicates that no significant adverse effect on the lake ecosystem will likely result. Alternative methods to open water dumping are to be used for the disposal of contaminated sediments, for example, upland and unconfined land disposal, confined disposal at the nearshore zone, beach nourishment or reuse of the dredged material.

Contaminated dredged material is disposed often in confined areas along the shoreline. The confined disposal facility (CDF) must be properly constructed so as to prevent the movement of contaminants from the sediment into the environment. Shoreline CDFs occupy economically valuable portions of the nearshore and compete or interfere with other uses. Utilization of upland CDFs involves the transport of dredged sediment by trucks, which is considerably more expensive and may not be environmentally desirable. In some cases, disposal in CDFs may not be possible due to high concentrations of contaminants in the sediments.

It has become more difficult recently to locate acceptable areas for CDFs, particularly in highly populated countries such as those in western Europe. For example, the amount of annually dredged sediment from the Rotterdam Harbour (The Netherlands) is about 20 million m^3 (26 million cubic yards) of which about 10% is severely contaminated. Confined disposal sites are either not available or their size is so limited that the volume of contaminated sediment must be reduced. Consequently, the volume of contaminated sediments must be reduced before transport to a disposal site.

A list of locations throughout the Netherlands, where the dredging of contaminated sediments will be necessary for remedial purposes, is presently being assembled.

Remedial options for the volume reduction and treatment of contaminated sediments have to be environmentally safe and economically feasible. Different methods have been proposed and developed to isolate, treat *in situ* or to remove and treat contaminated sediments. The removal of contaminated sediment by dredging and subsequent decontamination has received more attention recently and several isolation and *in situ* treatment methods also have been described.

3.8.1 Isolation and In Situ Treatment

Chemical and Biological Treatment

This approach usually involves the reduction or leaching of contaminants from sediments. The rates of transfer of many contaminants are affected by the dissolved oxygen levels and pH of the overlying water, as well as the depth of burial. All of these factors can be, at least partially, controlled over a small area.

The solubility of metals decreases with increasing pH which can be maintained by the addition of calcium carbonate or lime. Experiments using a lime application in Hamilton Harbour are being conducted by the National Water Research Institute to investigate the efficiency of this treatment on the immobilization of metals and nutrients in water and sediments. In addition, recent studies have reported on the coprecipitation of phosphorus with calcium carbonate and aluminum sulphate to reduce eutrophication (Murphy et al. 1985, Kennedy and Cooke, 1982).

The major advantage of in situ chemical and biological treatment is that these methods eliminate the need for removal of the contaminated sediments. However, all of these treatments have the potential for undesirable secondary impacts such as chemicals used for the treatment or their toxic degradation products. At present, such treatment is considered suitable for areas where contaminated sediments can be contained during the treatment such as in streams where the water flow can be diverted during the treatment. A summary of some of the more recent treatment methods being investigated was published by the U.S. Environmental Protection Agency (U.S. EPA 1985).

One biological treatment previously attempted uses the *Desulfovibrio* bacteria, in a sulphate medium combined with a pH control to convert metals into insoluble sulfides (Acres 1972).

Reducing the sources of phosphorus to control accelerated eutrophication rates has been one of the primary goals of the IJC and the jurisdictions. Controlling the phosphorus loadings from point and nonpoint sources is the first step in reducing nutrient enrichment and has been achieved in many areas. However, continued phosphorus releases from contaminated sediments has precluded reduction of eutrophication rates in some areas. In these cases it may be necessary to dredge contaminated sediments or treat in situ.

In situ treatments utilizing aluminum sulfate have been used successfully in many lakes to control internal phosphorus releases (Cooke and Kennedy, 1981; Kennedy and Cooke, 1982; U.S. EPA 1979). Aluminum sulfate, when added to water with carbonate alkalinity, will form aluminum hydroxide which will irreversibly bind with inorganic phosphorus in a natural lake system. The precipitated flocculent will settle and consolidate with bottom sediments, where phosphorus in the sediments can also be chemically found. This treatment can stop the seasonal phosphorus cycle and prevent further releases into the water column.

Isolation of Contaminated Sediments

Capping with active cover materials such as limestone, gypsum or ferric sulfate is used to detoxify the contaminated sediments. Oyster shells cover has been tested in Japan (Kikegawa 1981). If active cover materials are to be used, they must remain in place for sufficient time to react with the contaminants. However, further research is needed on application rates, secondary effects and the effectiveness of active cover material.

Application of activated carbon can reduce the quantity of contaminants leaching from the sediment into the water column. Mackenthun et al. (1979) suggested the establishment of a three-phase equilibrium in sediments treated with activated carbon: higher concentrations of contaminants being absorbed to carbon, a lower concentration absorbed to sediment particles and the lowest concentration occurring in the water. However, the use of activated carbon has been tested only in laboratory scale studies.

3.8.2 Decontamination after Dredging

Two objectives for the decontamination of sediments are:

1. to minimize the content of contaminants to obtain material which may be re-used, and

2. to minimize the volume of sediments which have to be disposed.

Investigations on the treatment of contaminated sediments began in the 1980s with an inventory of techniques to achieve the first objective. Present investigations involve testing and upgrading selected techniques. Some of the techniques have been applied on large-scale projects.

Case studies carried out in the United States involving techniques for the separation of dredged material were described recently by U.S. EPA (1985). These techniques include gravity settling (with and without polymer additions) in impoundments, tanks and barges; belt filtration; and fixation using chemicals to remove water. Some sediments were sorted by grain-size with screens. Disposal methods were dictated by contaminant levels in the separated sediment fractions. Costs were lowered by reducing the volume of the highly contaminated sediments (which require disposal at a permitted hazardous waste facility).

The development of technology for cleaning contaminated dredged sediments has been initiated recently in The Netherlands and West Germany. Results from some research in The Netherlands are already available. For example, one of the treatments of contaminated sediments involves the separation of more contaminated fine-grained sediment particles from the relatively uncontaminated coarse particles by hydrocyclones. Contaminated fine-grained sediment is further de-watered (to reduce its volume) and subjected to different decontamination treatments. No universal method is available for treating all types of contamination. There are two broad categories of treatment methods: one for removing heavy metals (acid leaching, ion exchange, magnetic separation, electrochemical methods, biological and ligand leaching), and one for removing organic contaminants (biodegradation, solvent extraction, thermal treatment, steam stripping and washing). Laboratory tested treatments showed up to 95% removal of some contaminants from the sediments (van Luin and Rulkens, 1987). The development of technology for contaminated dredged material remediation was described by van Veen and de Waaij (1987).

At present, more research and field verification programs are being conducted in The Netherlands on the treatment of contaminated sediments. A proposal entitled "Polluted sediments - comparison and development of procedures/actions" was prepared recently for a joint implementation by the Federal Republic of Germany, Belgium and The Netherlands. Participating European countries noted that sediment contamination is a worldwide problem, and that cooperation with jurisdictions on other continents could accelerate the development of a solution.

Feasibility

Any technique available for dredging of contaminated sediments may be used to dredge the sediment prior to the dewatering and decontamination treatments. The initial high cost of equipment used for the dewatering and separation of sediment particles (for example classifiers, hydrocyclones, large impoundments or tanks) has to be considered.

The method was designed for a treatment of highly contaminated sediments unsuitable for conventional disposal, in particular for sediments which can be separated into less contaminated, coarse particle size fraction (sand), and more contaminated fine grained material (silt and clay). Initially, large volumes of the dredged contaminated sediment can be reduced by the dewatering and separation of less contaminated coarse material from that which requires further special decontamination or disposal into designated facilities.

Environmental/Regulatory Criteria

Many laboratory methods for the isolation and in situ treatment of contaminated sediments, and for decontamination of dredged sediments, are presently being tested and verified in the field (Triangle 1986). Further research is required to better define the costs and evaluate the environmental effects of these techniques.

Long-Term and Short-Term Management

Little information on the effects of different in situ contaminated sediment treatments will command intensive monitoring activities at the treated area. These should include short- and long-term measurement of resuspension of the treated sediment/cover; recolonization of the treated area by benthic organisms and uptake of contaminants from treated sediments; release of contaminants from sediments by measuring water quality at the sediment-water interface; effects of treatment on the concentration of dissolved oxygen, sediment pH and redox potential; and gas generation within treated sediments.

It appears that the isolation, in situ treatment, dewatering and decontamination will be more expensive than any other remedial option. For example, the estimated cost given for dredging/dewatering/fixing and disposal of about 8,400 m³ (11,000 cubic yards) of PCB contaminated material from Waukegan Harbor, Illinois, ranged from $7.5 to $10.5 million (U.S. EPA 1985).

3.8.3 Solidification

A number of processes and products are currently being evaluated as a means of isolating and stabilizing contaminated sediments. The term solidification describes the elimination of free water from a semi-solid by addition of a setting agent such as portland cement, lime, flyash, kiln dust, and slag. Final products with a variety of properties are created by using co-additives such as soluble silicates and bentonite. The application of solidification technology can result in the physical stabilization of the contaminated sediments, the chemical immobilization of the contaminants, or both. Physical stabilization refers to processes aimed at optimizing certain engineering properties such as bearing capacity. Chemical stabilization refers to the minimization of contaminant leachability and solubility. Most solidification processes provide physical stabilization; however, attaining a chemically stable product warrants examination of the interactions among contaminants, sediment, and setting agent. Both in situ treatment and dredged-sediment treatment employing solidification technologies have been reviewed.

In Situ Treatment

In situ applications of solidification processes, using contaminated sediments as the aggregate in cement, are limited. Appropriate proportions of setting agent, water, and sediment and proper mixing conditions, which are difficult to ensure under water, are necessary. In situ solidification processes have been implemented in Japan (Otsuki and Shima, 1982); however, the emphasis of this application was on physical stabilization and contaminant immobilization was not directly addressed. Two potentially applicable in situ technologies are grout injection and the use of concrete filled mats (Armorform™).

Grout Injection

In situ solidification of bottom sediments via grout injection can be accomplished by two methods. The first method involves the injection of a grout (a mixture of portland cement, polymers and proprietary co-additives) into prebored, closely-spaced holes in the sediment bed. In the second method, the grout is injected into the top sediment layer, while the layer is mixed with a rotary tiller. Grout injection is believed effective when sediment contaminants are of low-organic content (Ebasco 1986a). This option was considered as a portion of the remedial activities slated for a cadmium-contaminated area (Marathon Battery Site) in the lower Hudson River in New York. Sediment metal concentrations peak at 17% for cadmium, 0.6% for cobalt and 15.6% for nickel (Ebasco 1986a) with cadmium water levels as high as 0.27 mg/L (0.27 ppm).

Initial screening of the grout injection process conducted as part of a feasibility study of the Marathon Battery site resulted in its rejection based upon probable quality control problems and the lack of data documenting the long-term stability of the resulting solidified material in an aqueous environment (Ebasco 1986a).

Armorform™ Articulating Block Mats

Armorform™ articulating block mats, commercially manufactured products evaluated for use at the Marathon Battery Site, are mats of permeable, double-layered, polypropylene or nylon panels which are filled with mortar. The mats are placed on the bottom and pumped full with a clean mortar. The fabric allows excess water to escape while solids are retained. This speeds hardening and results in the formation of a durable concrete structure. Use of Armorform™ is actually a containment process as opposed to a solidification technology. The process has been widely applied to civil engineering projects near and under water but as yet has not been applied to the containment of contaminated sediments. On initial screening, Armorform™ was considered a feasible option to be implemented in conjunction with layers of filter fabric and clean sediment at the Marathon Battery site. In the preliminary design, the filter fabric would be placed on the contaminated sediment, the Armorform™ mats on top of the filter fabric, and clean sediment would be used as a top cover. Further evaluation on performance, reliability, implementability, safety and level of remediation achievable resulted in an overall low rating and elimination of the option. Disadvantages of this technique include:

1. scouring of the clean sediment at tidal flow exposing the Armorform™ mat;
2. undefinable useful cap life;
3. tearing of the filter fabric during placement;
4. cap penetration by aquatic biota; and,
5. installation time frame in East Foundry Cove of approximately 17 months (one of the longest implementation periods of the alternatives considered).

Solidification Following Sediment Removal

The bulk of solidification technologies currently under evaluation require the removal, and in some instances dewatering, of contaminated sediments. The primary processes, as distinguished by setting agent, include cement and pozzolan matrices and the Sil-B method.

Cement and Pozzolan Matrices

Solidification processes utilizing portland cement, pozzolan (finely divided siliceous or siliceous and aluminous materials) such as slag, cement kiln dust and flyash as setting agents show potential as both chemical and physical stabilizers. These processes are being studied on a laboratory scale and in some instances implemented on a case-by-case basis. Several are commercially available in the United States. The formation of a solidified product is achieved during a hydration reaction in which free water from the dredged sediments is bound to the setting agent. The physical and chemical stability of the resulting product are functions of the sediment characteristics, type of setting agent and co-additives used.

Unpatented processes using combinations of setting agents and co-additives such as a) portland cement; b) portland cement with flyash; c) portland cement with flyash and sodium silicate; d) Firmex (a neutral, non-toxic form of silica used as an additive); e) portland cement with Firmex™; f) portland cement with West-P (a proprietary polymer); g) Firmex™ with West-P™; and h) flyash with lime were evaluated on a laboratory scale at the U.S. Army Corps of Engineers Waterways Experiment Station in the assessment of methods for the treatment of the PCB-contaminated sediments from Indiana Harbor. The analysis conducted by the U.S. Army Corps of Engineers involved physical and chemical testing. The two key tests were unconfined strength (UCS) tests used as a measure of physical stability (i.e. bearing capacity) and specifically designed serial, graded batch leach tests followed by the development of contaminant desorption isotherms as a measure of chemical stability. The Extraction Procedure (EP) was not performed. The design intent of the EP is the simulation of leaching of a waste disposed of in a sanitary landfill. The method defines wastes which are hazardous and those which possess the characteristic of EP toxicity as any waste whose EP extract contains any of the specified substances at or above the levels established in 40 CFR 261.24 (U.S. EPA 1986).

The results of 28-day UCS tests ranged from 48.5 psi to 682 psi (UCS of 10.4 psi is generally required for landfill; the equivalent of a 'medium clay'). Leach test results show cadmium and zinc to be completely immobilized by five of the eight processes; specifically, portland cement with flyash, portland cement with Firmex™, Firmex™, Firmex™ with West-P™ and portland cement with West-P™. Lime and flyash mixtures were shown to enhance zinc, arsenic, chromium and lead desorption indicating a need for contaminant-specific process selection. No significant change in the sediment's sorptive capacity for organic carbon was detected. The leachability of specific organic compounds was not assessed, leaving questions regarding PCB immobilization unanswered. Firmex™ and West-P™ and Firmex™ alone were found to be the most capable setting agents in achieving metal immobilization.

Bench-scale work was performed also by Chemfix Technologies and by Associated Chemical and Environment Services (ACES) to assess the feasibility of using a cement-based or pozzolanic solidification process as a component in the remediation plan for the Marathon Battery Site (Ebasco, August 1986b).

Although the Chemfix™ process is patented, different mixtures of setting altered to optimize both the physical and chemical agent and waste are tested and stabilization of the waste to be treated. In the case of cadmium-contaminated sediments from Foundry Cove, Chemfix Technologies tested 1) sodium silicate and portland cement; 2) sodium silicate and cement kiln dust; and 3) sodium silicate, portland cement and catalyst. The products were subjected to EP toxicity testing for metals and 48-hour UCS tests. UCS for mixtures 1, 2 and 3 were 34.7 psi (239.2 kPa), 20.8 psi (143.4 kPa) and 17.4 psi (120 kPa). Only the sodium silicate and portland cement mixture passed the EP toxicity testing with a cadmium concentration in the extract of 0.709 mg/L or 0.709 ppm (the EP toxicity maximum is 1 mg/L or 1 ppm). Cobalt and nickel are not standard EP toxicity parameters and were not measured. ACES conducted bench-scale studies with three mixtures composed of differing weight percentages of waste, pozzolan and lime. Forty-eight-hour UCS test results range from 7-19 psi (48.3-131 kPa). Cobalt and nickel were included in the EP toxicity testing. Two of the three mixtures were found to have cadmium, cobalt and nickel levels less than 1.0 mg/L or 1 ppm.

Solidification, specifically the Chemfix™ process, has been chosen in conjunction with hydraulic dredging and off site disposal as the remedial action to be implemented at East Foundry Cove Marsh (34 acres or 14 hectares) and East Foundry Cove (14 acres or 5.7 hectares). Both of these areas are portions of the Marathon Battery site in the lower Hudson River, New York. Implementation would include: hydraulic dredging, dewatering, thorough agitation and mixing, continuous pumping through Chemfix™ treatment units, extruding the treated waste to a solidification area and transfer of the solidified sediment to a disposal site.

Sil-B Treatment

The Silica Bonding (Sil-B) method is not based on a hydration reaction as in the formation of a cementitious matrix. Instead, sodium silicate is neutralized via acid addition and forms silicic acid liquid, an aqueous solution of low viscosity and good sediment-mixing properties. The silicic acid liquid contains very reactive silanol radicals (Si-OH) which polymerized by way of a dehydration reaction forming a stable gel. The gel, formed in four minutes at a pH of seven, cannot be dissolved by hydration (Oshita 1981). Small-scale testing of this technique was conducted in the River Waka in Uchikawa, Japan (Tabuse 1981). Water quality problems of the river are attributed to raw sewage discharges and the presence of heavy metals in bottom sediments.

The small field application of the Sil-B process involved placement of a bottomless-box frame in the river bottom such that river water trapped within the frame could be pumped out, followed by addition of Sil-B agent to exposed bottom sediment, and grab-bucket mixing of sediments with Sil-B agent. The 5 m^3 (6.5 $yards^3$) of sediment coagulated in 15 minutes.

The thrust of Japanese implementation of solidification technology is to remove and 'improve' (physically) bottom sediments for use as fill in creating new land. Applications of solidification techniques in Japan, are quite common place (Tabuse 1981; Kita and Kubo, 1983; Nakamura 1983; Otsuki and Shima, 1983). However, analyses do not sufficiently address the extent of contamination prior to treatment or the chemical stability of treated sediments.

Mixing Considerations

The mixing of the setting agent with dredged materials can be accomplished by putting the sediment into a mixing plant or by mixing in situ by machine. In situ mixing accomplished via conventional construction machinery (i.e. a backhoe, clamshell or dragline) is potentially best suited for mixing dewatered sediments with large quantities of low-reacting setting agents. The mixing efficiency of this method has not been accurately assessed at field-scale. Special equipment could be used to inject the setting agents into contaminated materials and mix the two while travelling around the perimeter of the mixing area. Once the first batch of treated materials is capable of supporting the injection carrier, the carrier is positioned on top of the solidified materials and a second pass is initiated (U.S. Army Corps 1987). Plant mixing is justified when large amounts of dredged sediments require treatment. The plant would consist of mixing and treatment units.

Feasibility of Solidification Processes

Although solidification processes appear promising in their ability to chemically and physically stabilize contaminated bottom sediments, further evaluation is required prior to full-scale implementation. The potential for chemicals to interfere with the hardening process has been established (Jones et al. 1985) but the mechanism of interference is poorly understood. Until such data are available any feasibility study exploring the application of solidification technologies at a specific site will require laboratory testing of a variety of setting agents and their interactions with the waste material. In addition, there is a lack of sufficient field-related data necessary for the implementation of full-scale application; therefore, field testing examining mixing efficiency and long-term stability of the treated material should be conducted.

Costs

Actual project cost data is limited and affected by the selected implementation strategy and costs related to additives, and volume requirements (U.S. Army Corps 1987). A trade-off can be made between cost and the extent of physical stability of the final product. For example, use of portland cement results in a highly physically stable product, whereas, less expensive agents produce a solidification product with much lower physical stability as measured by UCS. A similar trade-off can be made on the basis of setting time: use of high cost additives result in a significantly more rapid set.

Some site-specific cost estimates are available. A cost of $30-$50 per ton ($27-$45 per tonne) was estimated for industrial waste application (Cullinance 1985). Bench studies conducted by ACES and ChemFix with contaminated sediments from Foundry Cove resulted in cost estimates for the treatment of 64,500 yd³ (49,312 m³) of $50-$75/yd³ ($65-$98/m³) and $45-$50/yd³ ($59-$65/m³), respectively. These estimates do not include removal or disposal costs.

Environmental/Regulatory Criteria

The application of solidification technologies is evaluated on a case-specific basis focusing on the geographic characteristics of the size, the type of contamination and bench-scale results involving representative samples of the waste materials.

Chemfix™ is considered a proven technology for the treatment of specific hazardous waste streams, and in two instances the U.S. Environmental Protection Agency has granted waste-specific "delisting" (reclassification of a listed hazardous waste as nonhazardous) of Chemfix™-treated materials. (Federal Register, November 1986; Federal Register, September 1985).

Rigid generalized assessment criteria describing "how clean is clean" do not exist. This issue must be addressed for each case as the extent of the cleanup must be balanced by economic feasibility. Additionally, tolerance levels for specific contaminations must be weighed against public health, public safety and potential for worker-resident exposure.

Long-Term and Short-Term Management

The versatility of solidification technology results in the production of treated materials with a wide range a characteristics and many options for use or disposal of the treated materials exist. Choices may be made based on data generated via site-specific and waste-specific scale up and stability testing as to the need for long- or short-term management. Solidification technologies could be implemented to deal with a "hot spot," again after sufficient testing and regulatory approval, or in increments to remediate an area over a period of time. Post-treatment monitoring will depend on leachability test results and the chosen disposal site.

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3.9 NO ACTION ALTERNATIVE

Description of Option

The no action alternative is a real option, rather than a default condition stemming from an inability to implement other, necessary remedial activities. This is true, only if the essential information on sediment quality, quantity, dynamics and impacts has been collected. This information must also be evaluated in conjunction with other factors. The additional factors requiring evaluation include: use goals, restoration time frame, public and ecosystem health and safety, as well as the long-term stability of natural and anthropogenic conditions within the drainage basin. This option, more than any of the others presented in this report, is contingent on the elimination of all active sources.

This option relies on natural processes such as the input of uncontaminated sediments from the drainage basin and their integration with in-place contaminated material through dispersion, mixing, burial and biological reworking.

Many of these processes can be enhanced or at least maintained at their present levels through both direct and indirect means. Some direct actions include the periodic use of a flocculent in the water column or a moveable silt curtain to enhance sediment deposition. Some indirect actions include the management of both point and nonpoint sources of solids, to maintain rather than reduce their present input levels. This is counter to the current practice of reducing solids inputs from both point and nonpoint sources.

Feasibility

Since no actions are required, the option has no technical barriers associated with it. The stringent information requirements, however, present some very real technical challenges as well as financial demands. Sufficient information must be collected to stipulate, with considerable certainty, the length of time required for recovery. This information should then be compared with the restoration time frame for the entire Area of Concern to see if they are compatible.

In addition, the feasibility of this option is dependent on the long-term stability of natural and anthropogenically induced processes throughout the drainage basin. If the delivery rates of solids, the flow regime of sources, or the physical factors affecting in situ sediment dynamics are likely to change over the projected recovery interval, then this option may be less attractive or even infeasible. In addition, the area should not be subject to periodic dredging for navigational purposes, which would likely reintroduce contaminants to the surface sediments.

Environmental Regulatory Criteria

A primary consideration for this approach is the present nature, severity and degree of biological impact which can be related to the existing contamination of the sediments. In cases where ecosystem or human health are already felt to be significantly impacted, waiting for ten or 20 years for the problems to be rectified may be unacceptable. The study of sediment dynamics in the Area of Concern may show that contaminants are being transported to the adjacent lake (e.g. bedload). Therefore, the sediments represent a continuous and possibly significant, long-term source of contaminants to the lake and should likely be dealt with at their present source.

There are a number of federal, state and provincial acts, regulations and pieces of legislation already in existence which appear to extend some jurisdiction over in-place pollutants. In practice, however, existing agency policies do not presently extend to contaminated sediments unless they are to be removed and subsequently require relocation or disposal.

Long-Term and Short-Term Management

The "implementation" of this option implies that a rather long period of time will expire prior to recovery or restoration of the area to acceptable levels. During this time, two major management requirements are the regular monitoring of conditions to check that the predicted course of events are being realized, and the control of inputs of uncontaminated solids at the required level.

In the short-term, active sources must be eliminated for this to be considered a viable option.

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4.0 OPTION SUMMARY

TABLE 4

OPTION SUMMARY

FEASIBILITY CRITERIA	OPTIONS			
	DEPOSITIONAL ZONE PLACEMENT	CONTINUED AQUATIC DISPOSAL AND CAPPING IN-PLACE	HYDRAULIC CONTROL/ NAVIGATIONAL RELOCATION	CDF
Technological	Minimum depth 30 m or 3x maximum wave height.	Predisposal excavation recommended. Capping stable under normal conditions.	Not often used.	Most commonly used technique.
Environmental Acceptability	Must stay in place.	Field verification proved effectiveness of preventing the movement of contaminants into water and biota.	Minimizes problem due to navigation.	Filling of wetlands or shallow areas becoming less acceptable.
Costs ($)	0.26/yd³/mile. (transportation-U.S.) 1.25/m³/km - (Can.)	11-27/m³ (total cost) 14-35/yd³		4/yd³ (U.S.) 2-15/m³ (Can.)
Comments	Compatibility with substrate. Follow IJC site selection criteria. Additional testing and validation needed due to limited experience in the Great Lakes.	Technique offers promise. More assessment required if technique attempted on a case-by-case basis.	Interim measure until controls placed on sources and long-term solution identified.	Considerable data and experience on use of CDFs from engineering perspective. Becoming less acceptable from environmental socio-economic perspective.

Table 4 - cont'd.

FEASIBILITY CRITERIA	OPTIONS AGRICULTURAL	QUARRIES/ STRIP MINES RECLAMATION	BEACH NOURISHMENT	UPLAND FILL FOR RECREATION
Technological	Demonstration on Project Toledo Port Authority. Filling of low lying agri-cultural lands. No technological barriers.	Proximity of site is the major constraint.	Material must be con-tained in a cost-effective manner. Transport to site may pose problems.	Most efficient use of space because the site can go as high as necessary. (e.g. ski slopes)
Environmental Acceptability	Represents reuse and farmland improvement. Needs to be dried out prior to use.	Groundwater impacts must be carefully considered.	Good use for materials of high nutrient content, unsuitable for open water disposal. Material should be compatible with area of placement.	Seasonal uses would be possible during filling process.
Costs ($)	0.50/yd^3/mile or 0.41/m^3/km (pumping) 185,000/mile or 115,000/km (pipe) 0.45/yd^3/mile or 0.31/m^3/km (transport)	12.50/yd^3/200 miles or 13.67/m^3/322 km (transportation)	0.50/yd^3/mile or 0.41/m^3/km (pumping) 185,000/mile or 115,000/km (pipe)	0.50/yd^3/mile (pumping-U.S.) 185,000/mile (pipe-U.S.) 10-20/m^3 (pumping within 2 km - Can.)

Table 4 - cont'd.

FEASIBILITY CRITERIA	OPTIONS			
	AGRICULTURAL	QUARRIES/ STRIP MINES RECLAMATION	BEACH NOURISHMENT	UPLAND FILL FOR RECREATION
Comments	Must meet all regulatory requirements.	Recommended where suitable sites are available.	Material must match beach substrates or be similar in nature (nonerodible). Recommended for material with limited quantities of nutrients.	Geographic concentrations (availability of land and transportation to site).

Table 4 - cont'd.

FEASIBILITY CRITERIA	OPTIONS			
	SECURE LANDFILL	TREATMENT	SOLIDIFICATION	NO ACTION ALTERNATIVE
Technological	Applies to material designated as hazardous. Appropriate regulatory criteria applies.	Any technique available for dredging contaminated sediments can be used. Dewatering and decontamination requires costly equipment. Limited information available.	Technique is proven with industrial/municipal waste. No full-scale applications on sediments. The necessary equipment is readily available.	Limited by resuspension activity encountered, rate of accumulation and quality of new material entering system.
Environmental Acceptability	Material must have been dewatered and less contaminated sediment subjected to another option.	Suitable for highly contaminated material which is unsuitable for any conventional disposal; reduces the volume of sediment prior to decontamination treatment.	Long-term stability of solidification products undetermined.	Length of time to alleviate sediment related problem may be too long. Since sediments are not disturbed, effects of contaminants present are not increased.
Costs ($)	Case-by-case.	Estimated cost for dredging, dewatering, fixing and disposal of about 8,400 m³ of PCB contaminated material ranged from U.S. $7.5 to 10.5 million.	In situ containment The cost for containment using Armorform™ range from $3.75 to $6.75 per ft.² for an area of 50,000 ft². ($40-$72 per m² for an area of 4,600 m²) Treatment of dredged materials Estimates for treatment of 64,500 yd³ (49,317 m³) of sediment from Foundry Cove, N.Y. range from $45 to $75 per cubic yard ($59 to $98/m³).	No expenditures required.

Table 4 - cont'd.

FEASIBILITY CRITERIA	OPTIONS			
	SECURE LANDFILL	TREATMENT	SOLIDIFICATION	NO ACTION ALTERNATIVE
Comments	Depends on regulations of jurisdictions.	Need for technology development and testing on pilot basis. Technology transfer from western Europe possible.		Control of resuspension and transport as well as enhanced accumulation may be necessary. Length of time for sediment remediation may be too long for use restoration goals.

5.0 FUTURE OPTIONS

The options described in this chapter are presently in the research and development stage. They include both in situ and in vitro methods of handling and decontamination. These techniques are for all practical purposes unavailable as remedial measures in the Great Lakes. While considerable information is available on process in some cases, limited information is available on the required engineering and environmental impacts of these treatments.

1. SUPERCRITICAL WATER OXIDATION

This method involves low temperature oxidation which uses the temperature and pressure of supercritical water to break down hazardous organics to carbon dioxide, water and other simple, less harmful molecules. Supercritical water results in inverse solubility characteristics (i.e. insoluble organics become soluble and soluble salts become almost insoluble). The density maintained during the process is low enough and temperature high enough that water becomes a good solvent for organic substances.

Modar Supercritical Process

This process can be used on PCB-contaminated sediments containing 20-40% solids. The slurry is pressurized to supercritical pressure (>22.09 MPa or 3,204 psi) and then heated (>374°C or 705°F). The preheated slurry, pressurized oxygen, organic fuel and oxidizer are brought together at the oxidizer and organics are oxidized in a controlled rapid reaction. Inorganics and sediments are removed from the oxidizer effluent in a salt and sediment separator, and chlorine atoms are liberated as chlorine ions.

Efficiency:

The estimated efficiency is greater than 99.9995%, with an estimated residual of greater than 0.1 ppb.

Development stage:

This technique is presently undergoing bench scale testing.

Field tests have been conducted with PCB liquids, and a pilot test unit exists. This unit has an organic material flow capacity of 190 litres (50 U.S. gallons)/day. The process has been used to destroy PCBs in oils and decontaminated dioxin-tainted soil.

Feasibility and Limitations:

The use of a closed system allows control of waste materials at all times.

Large solids and particles must be creened out by pulverizing and hydro-seiving. It is adaptable to a wide range of feed mixtures and scales of operation. If the sediments being treated do not contain sufficient organics, a supplementary fuel is required to maintain the reaction temperature.

At present a 570m³ (746 yd³)/day capacity unit is being developed and should be available in 21.5 months.

Estimated Cost:

The total costs are estimated to range from \$250-\$733/m³ (\$191-\$560/yd³), which includes: dredging (\$20/m³ or \$15/yd³), transportation (\$13-\$126/m³ or \$10-\$96/yd³), treatment (\$184-\$554/m³ or \$140-\$423/yd³), and redeposition (\$33/m³ or \$25/yd³).

2. NUCLEOPHILIC SUBSTITUTION

This process involves the chemical dehalogenation of PCBs under mild conditions using the electron donating principle. Two potential applications of the process are seen: 1) the treatment of sediments themselves; and 2) the treatment of concentrated PCBs from extraction process. The former is the Terraclean process.

KPEG TERRACLEAN C1 PROCESS

Potassium hydroxide (KOH) is reacted with polyethylene glycol (PEG) to form an alkoxide. The alkoxide reacts with one of the chlorine atoms on a chlorinated molecule to produce an ether and potassium chloride. The addition of an RO-group enhances the solubility of the molecule and makes it less toxic. The reaction continues until several chlorine atoms are removed from the PCB molecule.

The reagent consists of a mixture of PEG, potassium hydroxide and dimethyl sulfoxide (DMSO). DMSO is used as a phase transfer catalyst to promote extraction of PCB from the sediment. Reagent formulation varies according to specific soil and contaminant combinations. Water from sediment is volatilized and used later. The soil is then washed with water two to three times to remove residual reagent and dechlorinated byproducts, with an overall reagent recovery of greater than 99%. The preferred reaction temperature is 150°C, with a reaction time of 30 to 120 minutes.

Efficiency:

This results in an estimated efficiency of greater than 98% and an estimated residual of less than 1 ppm. The efficiency varies with reaction temperature and reagent formulation.

Development Stage:

Pilot tests are presently being conducted and the application of alkali PEG (APEG) reagent to decontaminate PCB contaminated soils is under investigation at U.S. EPA.

Feasibility and Limitations:

The nucleophilic substitution process yields substituted biphenyls rather than ultimate products of decomposition, and considerable analytical chemistry is needed to get a complete byproduct picture. In all of the tests to date, PCBs were done as a preliminary to dioxin studies; therefore, reagent was formulated for dioxin only. High carbon soils create the slowest rate of reaction and fine soils take longer to drain. The rate of reaction, for different PCB isomers, has been found to be inversely proportional to their biodegradability.

If the temperature is less than 100°C (212°F), the water content in the soil affects the rate of reaction (at 150°C or 302°F water rapidly volatilizes and therefore is of no concern). The reaction rate increases with a temperature up to 200°C or 392°F, but degradation and volitization of reagents becomes a problem. Sediment containing 40% water would take 4,818,000 kJ of energy/m³.

The material can be handled in several types of mobile trailer mounted units. Since the process is closed, no pollution control equipment is required. There is, however, a fine buildup on reactor filter screens which results in a slow draining time, but this should be prevented by the constant mixing in the washing phase. At present, it is not known if there are any residual, totally dechlorinated PCBs remaining in the soil; it is probable that they were all washed away. There is a definite need for various bioassessment tests for bioaccumulation, mutagenicity and toxicity to be conducted.

At present, a unit which treats 45.4 kg/batch (100 lb) is under development and should be available in approximately 19.5 months.

Estimated Cost:

Since exact costs are dependent on contaminant concentrations, sediment, water content, etc., only part of the process can really be accounted for. Actual costs will include capital, reagent, energy, maintenance, labor water disposal and about a 50% profit/contingency.

Process operating costs are approximately $160.37 for a four- hour cycle and $191.16 for a six- hour cycle. A range for the entire cost is about $211 to $378/m³ ($161-$300/yd³) which includes treatment, dredging, transport and redeposition.

3. <u>RADIENT ENERGY ULTRAVIOLET LIGHT</u>

This process involves the use of a UV light and a reducing environment to create a photochemical reaction that destroys PCBs, leaving biphenyls and sodium chloride. The use of various solvents such as water, alcohol, and hydrocarbons produce a difference in efficiency and mechanism.

LARC PROCESS (Light activated reduction of chemicals)

The extraction solvent used is isopropanol which dissolves PCBs readily and is relatively inexpensive. The isopropanol is mixed with sediment containing 25% water. The liquid is then decanted, and the process is repeated. Sodium hydroxide pellets are then added to the PCB extract to form 2% solutions. The solution is placed in a reactor, hydrogen gas is added and the UV light is turned on. The reaction time is from 1.5 to 2.0 hours. The LARC testing to date has used a single lamp because of the small amount of solution used.

Efficiency:

The estimated efficiency of this process is greater than 90%, with an estimated residual of from 38-50 ppb. A five stage extraction would be necessary to reduce concentrations to background levels.

Development Stage:

Lab tests have shown that PCBs can be extracted from wet and dry soils using a solvent such as isopropanol and that the extracted PCBs can be degraded by LARC.

Feasibility and Limitations:

Oxygenated derivatives chlorinated dibenzofurans and chlorinated dioxins, have not been observed. The concentration of chlorinated organics entering LARC can be tightly controlled, even though the concentration in sediment varies dramatically. The process provides flexibility and degradation proceeds rapidly. A larger reactor with greater light density could increase degradation rate by 2 to 2.5 times.

Estimated Cost:

Based on an average concentration in soil of 1,500 ppm, and including daily operation, labor, analytical, travel, profit, the total cost should be from \$233 to \$336/m³ (\$178 to \$257/yd³). This includes treatment, dredging, transport and redeposition, with the treatment cost being approximately \$157/m³ (\$120/yd³) of soil.

4. PYROLYSIS

This process involves the thermal rupture of molecular, chemical bonds which destroys it without oxidation.

ADVANCED ELECTRIC REACTOR (J.M. Huber Corporation)

Feed material (contaminated sediment) is pretreated to less than 3% moisture content and 35 mesh particle size. A high temperature fluid wall reactor heats organic compounds rapidly using intense thermal radiation near infrared. The reactants (nonliquid PCBs) are surrounded by a gaseous blanket of nitrogen. Carbon electrodes are used to heat the reactor core, and heat transfer is accomplished by radiative coupling from the core to the feed material.

The principle products are hydrogen, chlorine, hydrochloric acid, elemental carbon and a granular free-flowing solid derived waste. Gas and solids pass through two treatment zones. Gas then moves through a bag house and activated charcoal beds and is then released into the atmosphere. The solids are collected in a sealed bin. The process is run at 2,000°C (3,632°F) to 2,300°C (4,172°F) with a feed rate of 7 kg (15.4 lb)/min to 7.2 kg (15.9 lb)/min . Feed masses range from 1,580 kg (3,483 lb) to 1,927 kg (4,248 lb).

Efficiency:

The estimated efficiency of the process exceeds 99.9999% with an estimated residual of less than 1 ppb.

Development Stage:

A pilot test with a four trial burns has been conducted over three days, using Arclor 1260.

Feasibility and Limitations:

The byproducts, which include treated sand, baghouse filter catch, activated charcoal and scrubber liquid, are all nonhazardous wastes. The process requires approximately 1,100 kW of electricity per ton of soil, while the operation of pretreatment equipment and other auxillary items uses an additional 200 kW/ton.

Estimated Cost:

Huber estimates that the cost for a large volume cleanup would be approximately $763/m³ ($583/yd³). Costs are likely to range from $839 to $942/m³ ($639 to $720/yd³) for treatment, dredging, transport and redeposition.

EXTRACTION

This technique involves the leaching of soils with organic solvents and is often a prerequisite for other chemical processes. Suitable extraction solvents such as kerosene, methanol, ethynol, isopropanol, furfural, dimethyl formamide (DMF), dimethyl sulfoxide, ethylene diamine, and freon mixtures can be used. Supercritical fluids such as carbon dioxide have also been considered.

ACUREX SOLVENT WASH PROCESS

The solvent is chosen by comparison of absorption isotherms and PCB diffusion rates into the following fluids: pure hexane, pure FC-113, a proprietory solvent blend, and an FC-113/hexane blend. Each wash liquid has a PCB content of approximately half the previous wash and this value is independent of soil type and its PCB concentration.

Efficiency:

Treatment is continued until a residual level of less than 2 ppm is reached.

Development Stage:

This method is present in the pilot test stage with field tests planned.

Feasibility and Limitations:

The major advantage of this technique is that it tolerates up to 40% water in PCB extraction step.

Estimated Cost:

Basic treatment costs range from \$130 to \$390/m³ (\$99-\$298/yd³), which included with the costs of dredging, transport and redeposition result in final estimated costs of \$196 to \$569/m³ (\$150-\$435/yd³).

O.H. MATERIALS EXTRACTION PROCESS

The soil is predried to less than 5% moisture and slurried with methanol. It is then separated and redried. The solvent is cleaned up for resale using activated carbon, subjected to incineration (other methods can be used). The soil is then subjected to light land farming for degradation of any residual methanol, while wastewaters are treated in a holding pond.

Efficiency:

The estimated efficiency of the process is around 97%, with an estimated residual of less than 25 ppm. Additional extractions should improve the process efficiency.

Development Stage:

Field tests are currently underway.

Estimated Cost:

Based on conceptual process and experience so far, a cost of \$100 to \$514/m³ (\$76-\$393/yd³) is expected (this includes dredging and transport). Methanol costs approximately \$0.18/L (\$0.05/U.S. gallon) and can be resold at \$0.06/L (\$0.02/U.S. gallon).

SOILEX SOLVENT EXTRACTION

Kerosene and water are used as the solvent, since water helps break up the soil particles. The process operates best at 42 to 45 percent water by volume. A water to soil ratio of three to five is best, with a kerosene to soil ratio of three to five. Kerosene retention in the soil has been found to be about 25 percent by volume. Each batch takes about three days.

Efficiency:

The estimated efficiency is around 95% for the three stage process, with an estimated residual of between six to nine ppm. Lower values can be obtained using additional stages.

Development stage:

This technique is presently in the pilot stage.

Feasibility and Limitations:

The process accepts wet sludge and only simple stir tanks are required for extraction. Each batch takes three days for treatment. Further tests are necessary to develop more efficient separation techniques. The process also generates RCRA waste (concentrated PCBs).

Estimated Cost:

A rough estimate of cost, based on increasing the number of extraction stages to achieve desired background levels, is from \$856 to \$913/m³ (\$654 to \$698/yd³), which includes dredging, transportation, treatment and redeposition.

NEW YORK UNIVERSITY LOW ENERGY EXTRACTION PROCESS

This process uses a low energy technology for the extraction of PCBs from soils and sediments, and concentrating the extract in a manner suitable for chemical destruction. It involves the separation of sediment into solid and liquid, and then leaching the separated sediment with a hydrophylic solvent such as acetone, leaving a cleaned sediment. PCB extracts are then transferred to a hydrophobic solvent such as kerosene where they are concentrated. Finally, the acetone is then steam stripped from the sediment.

Feasibility and Limitations:

The proper selection of equipment is absolutely essential. The basic principles have been confirmed experimentally, using PCB-contaminated sediments from Waukegan Harbor, Illinois. There remains the need for field testing.

The present capacity is approximately 119 m³ (156 yd³)/day , which could be extrapolated to 570 m³ (746 yd³)/day. The process is estimated to be functional at this level in about 25 months.

Estimated Cost:

The treatment cost is estimated to be \$31.57/m³ (\$24/yd³). The total cost is estimated to be \$56.67/m³ (43.33/yd³), if incineration of the waste stream is included.

PROPANE EXTRACTION PROCESS, CRITICAL FLUID SYSTEM (CFS)

This technique uses propane, at ambient temperatures and 1378 kPa (200 psi) pressure, to extract PCBs with other oily organics from a water slurry of sediment. The treated slurry is discharged, after separation from the liquid propane which contains dissolved contaminants. The propane solution is fed to a separator where the solvent is removed by vaporization and then recycled. The contaminants are drawn off as a concentrate for final treatment.

Feasibility and Limitations:

The process has been tested for PCB containing refinery sludge; however, there is a need for further work to identify the optimum ratio of propane to sediment. A number of extractions must be carried out in order to get the prescribed level of decontamination.

At present, the process has an operating capacity of 114 litres (30 U.S. gallons)/minute, and should be available in 25 months.

Estimated Cost:

Costs are estimated to range from \$155 to \$266/m³ (\$118 to \$203/yd³).

BEST EXTRACTION SLUDGE TREATMENT (BEST) PROCESS

Sediments, with free water, are mixed with triethylamine (TEA) in a mixer using the settling characteristics of the sediment. The ratio of TEA to the sediment feed must be high enough for all the bound and unbound water in the solid matter to be completely mixable in a single phase, and may range from one to seven parts by weight of TEA to one part by weight of water. The mixture is then mechanically separated by centrifuging and the solids are dried to remove any residual TEA. The liquid, containing TEA, oil, contaminants and water, is heated to 60°C (140°F) or higher. The liquid forms an oil solvent phase and a water phase (the former contains most of the contaminants). The two phases are then separated by decanting. The TEA is recovered from the oil/solvent by flash evaporation and countercurrent steam stripping. Recovered TEA is chilled and then recycled.

Efficiency:

The first extraction removes approximately 79.3%, based on the total sample. Subsequent extractions remove up to 98.7% of the initial amount, and from then on, there is little improvement.

Feasibility and Limitations:

The technique should be available in 14 to 19 months.

Estimated Cost:

It is estimated that a system designed to treat an average of 520 m³ (680 yd³)/day, with a peak capacity of 675 m³ (883 yd³)/day, should cost \$133.30/m³ (\$102/yd³) to operate.

VITRIFICATION - BATTELLE IN SITU VITRIFICATION PROCESS (ISV)

The process of in situ vitrification involves the insertion of electrodes vertically in the dredged material, in a square array. A mixture of graphite and glass is then placed in 5 cm by 5 cm (2 in. by 2 in.) trenches on the surface of the sediment between the electrodes to form a conductive path. An electrical current is passed between the electrodes, creating temperatures high enough to melt the soil. The graphite is consumed by oxidation, as the matter zone grows downward, incorporating the soil contaminants and producing a vitreous mass, while convective currents within the melt distribute the wastes evenly. Gasses generated from the water mass are collected by a hood over the area and treated. When the power is turned off, the molten volume begins to cool, producing a block of glass and crystalline material resembling natural obsidian or basalt.

Efficiency:

The estimated efficiency of the process is 99.9%, with virtually no residual in the vitrified block or outside the treatment zone.

Development Stage:

This technique is presently undergoing pilot testing with soil.

Feasibility and Limitations:

The sediments must first be dredged; however, the vitrified mass produced has shown no detectable residual level of PCBs, and no PCBs were detected in the majority of the surrounding soil. The product of the process (solid glass and crystalline block) may be more costly to deposit, with fewer options, than ordinary sediment material. In addition, to conserve energy, sediments high in moisture should be predried. The nature and extent of emissions from melt vary from one type of sediment to another. This technique should be available for use in about 19-24 months.

Estimated Cost:

Treatment costs appear to range from \$293 to \$332/m³ (\$224 to \$254/yd³), with capital costs running around \$47.04/m³ (\$36/yd³). Soil moisture increases costs.

7. MICROORGANISMS AND ENZYMES

These processes employ microbes, either bacteria or fungi which may be oligate or facultative aerobes or anaerobes. Total degradation is termed mineralization. The use of enzymes has great conceptual potential, but has not progressed very far.

INDIGENEOUS AND CONVENTIONAL CHEMICAL MUTANTS

This technique employes naturally occuring organisms found in diverse PCB-containing soils and sediments. Tests have been conducted for PCB degradation, using an assay mixture containing five types of cogeners selected to provide different resistances to enzyme attack. Results showed that

Alcaligenes eutrophus H850 and Psuedomonas putida LB400 were able to degrade 13 test cogeners from 80 to 100%. Fungi have been found to be less effective than bacteria. It has also been found that degradative pathways and cogener preferences differ among the microorganisms.

ENZYME MECHANISMS

In most cases bacteria can degrade one ring of a chlorinated biphenyl but are unable to degrade the resulting chlorinated benzoates. Mixed cultures of bacteria can mineralize PCBs with four of fewer chlorines per molecule; however, more heavily substituted PCBs resist degradation.

BIO-CLEAN PROCESS

This process involves the use of Anthrobacteria and/or other naturally occuring microbes to destroy PCBs (and related organics) under aerobic conditions. Contaminated sediment is put into a digester, in slurry form, with solids up to 33% by weight. A reagent is added to enhance contaminant extraction and heated to 82° C where it is held for an hour with agitation, to extract and solubilize contaminants, and to sterilize the sediments. After extraction, the slurry is cooled to 30° C, neutralized as needed, and innoculated with a suitable microorganism to initiate decontamination. Supplementary nutrients (ammonia nitrogen, phosphorus, and potassium) may be required with some sediments. The slurry is kept this way for 48 to 72 hours until degradation occurs. The treated sediment is transferred to a dewatering pit where solids and liquids are separated for redeposition. Therefore, this is a two-step process: 1) extraction, sterilization, and solubilizing the contaminants using high pH and temperature; and 2) bacterial destruction of contaminant.

The daily capacity of the process is about 30.6m³ (40 yd³)/day of soil, using the batch formula, and the whole process takes about three days. The degradation products are carbon monixide, sodium chloride, and bacterial cells.

Efficiency:

In lab tests, selected PCB cogeners in soil have been reduced to 10 ppb in 48 to 72 hours.

Development Stage:

This technique is presently in the lab; however, pilot scale evaluation facilities are available. The process requires further testing and the use of some additional strains of microorganisms.

Feasibility and Limitations:

The organisms used are naturally occuring, and pretreatment is not required to dewater sediments. The process does not appear to generate RCRA waste or emissions; however, very high PCB concentrations (i.e. >300 ppm) inhibit degradation process.

The technology should be available in about 19 months, with a capacity to treat about 650 m³ (850 yd³)/day of sediment.

Estimated Cost:

The Estimated treatment cost is around $156/m³ ($119/yd³).

SYBRON BI-CHEM 1006 PB/HUDSON RIVER ISOLATES PROCESS

This process is still in laboratory stage, with only limited information available.

COMPOSTING

Aerobic and anaerobic composting have been studied in the laboratory. For aerobic composts warm air is drawn through a sodium hydroxide trap to remove any carbon monoxide and then through a water trap to humidify the air and remove any caustic material. The atmosphere above the compost is drawn through sulfuric acid, sodium hydroxide and activated carbon traps to remove any gaseous biodegradation products. For anaerobic composts, an initial sodium hydroxide scrubbing trap has been eliminated; instead, material is initially flushed with nitrogen to remove any oxygen and then flushed every three to five days. The compost used had a 60% moisture content, and the process was operated at a temperature of 55°C (131°F).

Efficiency:

The aerobic process has an estimated efficiency of 62%, over four weeks, with an estimated residual of 504 to 688 ppm. The anaerobic process has an estimated efficiency of from 27% to 47%, over four weeks, with an estimated residual of 825 to 1120 ppm. Results depend strongly on the kind of microorganisms which are present.

Feasibility and Limitations:

This method requires considerable work site space and monitoring. The results are uncertain because of a lack of control of weather and other conditions.

Estimated Cost:

At present, there is not enough data to estimate costs.

RECOMMENDATIONS

It is recommended that the development of techniques for remediating contaminated sediments should have a high priority in future research, since the treatment of contaminated sediments will figure prominently in most Areas of Concern RAPs.

Estimated Cost:

The estimated treatment cost is around $165/m3 ($125/yd3).

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COMPOSTING

Aerobic and anaerobic composting have been tested in the laboratory. For aerobic composting warm air is drawn through a sodium hydroxide trap to remove any carbon monoxide and then through a water trap to humidify the air and remove any caustic material. The atmosphere above the compost is drawn through sulfuric acid, sodium hydroxide and activated carbon traps to remove any gaseous biodegradation products. For anaerobic composts, an initial sodium hydroxide scrubbing trap has been eliminated; instead, material is initially flushed with nitrogen to remove any oxygen and then flushed every three to five days. The compost used had a 60% moisture content and the process was operated at a temperature of 55°C (131°F).

Efficiency:

The aerobic process has an estimated efficiency of 62%, over four weeks with an estimated residual of 304 to 488 ppm. The anaerobic process has an estimated efficiency of from 27% to 47%, over four weeks, with an estimated residual of 825 to 1120 ppm. Results depend strongly on the kind of microorganisms which are present.

Feasibility and Limitations:

This method requires considerable work site space and monitoring. The results are uncertain because of a lack of control of weather and other conditions.

Estimated Cost:

At present, there is not enough data to estimate costs.

RECOMMENDATIONS

It is recommended that the development of technology for remediation of contaminated sediments should have a high priority in future research, since the treatment of contaminated sediments will figure prominently in most Areas of Concern RAPs.

APPENDIX I

SELECTION OF DREDGING EQUIPMENT

SELECTING DREDGING EQUIPMENT TO MINIMIZE SEDIMENT RESUSPENSION

The successful containment or treatment of contaminated sediments is dependent on efficient and effective removal and transportation of the contaminated portion of the sediments. Typically, those contaminants which are of significant concern have hydrophobic, lipophilic properties. These properties cause the contaminants to be unevenly distributed among the sediment particles. Typically, the most heavily contaminated fraction of the sediment tends to be those fine grained particles which are of organic origin. The organic compounds which tend to be bioaccumulating have a natural affinity for contaminants such as oils and greases. A strategy for removing contaminated sediments should be designed to capture and retain the fine grained and organic fractions.

When a plan to deal with contaminated sediments is developed it is important to keep in mind that the scale of the operation is of a size normally considered to be industrial in magnitude. The equipment and machinery needed to pick up, move and place the contaminated sediments must by necessity be of industrial scale. Basically, removal, handling and disposal of contaminated sediments is an earthmoving type of operation, albeit one that is done underwater. While there is a variety of equipment for undertaking such a task, it should be kept in mind that the types of available equipment is somewhat limited. It is important to understand the operating characteristics and limitations of each of the various types of equipment which are available for underwater excavations.

When choosing a removal and relocation strategy, a significant consideration should be the propensity for sediment resuspension which may be caused during the overall effort. The primary element in this consideration is the amount of sediment resuspension caused by the dredging operation. For the purpose of this discussion, sediment resuspension caused by dredging will be defined as those sediment particles resuspended into the water column during the dredging operation that do not rapidly settle out of the water column. This includes any resuspension caused by barge or hopper overflow, spillage, leakage, spud movement, or other contributors directly related to the dredging operation. Contributions of sediment from the prop wash by tenders, barge movement, or other operations not directly involved in the dredging operation are not considered. The method of disposal was not considered in evaluating the sediment resuspension or in the rating of various dredge types.

The purpose of this discussion is to compare the operating characteristics of various dredge types for the purposes of evaluating their propensity for resuspension of sediments. The organic, fine grained particles are those which tend to move into resuspension easiest, remain in resuspension the longest, and therefore be dispersed the greatest distances. The remainder of this discussion is taken directly from the Environmental Effects of Dredging Program Technical Note EEDP-09-1, December 1986, by Donald F. Hayes of the U.S. Army Corps of Engineers Waterways Experiment Station.

Conventional Dredges

Conventional dredges include unmodified types commonly used in the United States such as hydraulic dredges (e.g. the bucket or clamshell dredge). Operational parameters that affect sediment resuspension are discussed, and control measures that may reduce resuspension are presented.

Cutterhead Dredges

The popular high-production cutterhead dredge may not seem a very likely candidate for efficient removal of contaminated sediment because of the high-energy cutting and sweeping actions associated with its operation. However, field studies conducted in the James River near Norfolk, Va. (Raymond 1984) and in the Savannah River near Savannah, Ga. (Hayes et al. 1984) indicated that the cutterhead dredge is capable of removing sediment with relatively small amounts of resuspension extending beyond the immediate vicinity of the dredge as compared to other conventional dredge types. Figure 1 gives an indication of typical suspended solids concentrations in a turbidity plume generated by a cutterhead dredging operation.

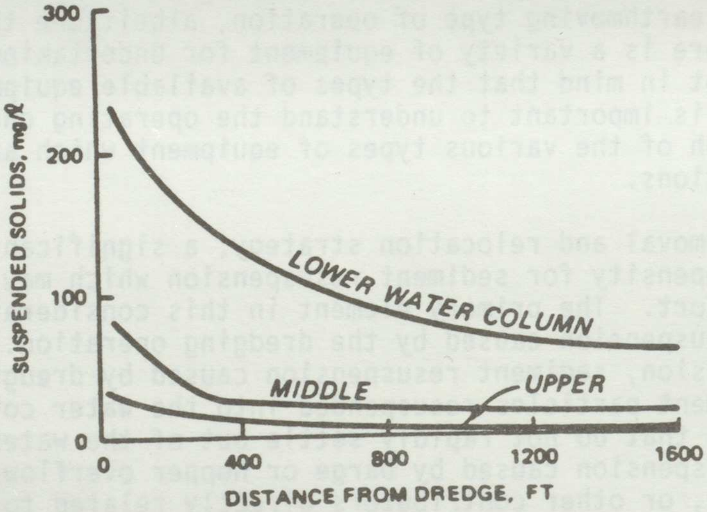

Figure 1. Resuspended sediment levels from cutterhead dredge operations in the Savannah River.

Research under the Improvement of Operations and Maintenance Techniques Program (IOMT) has shown that sediment resuspension by a cutterhead dredge can be reduced by proper selection of the cutter rotation speed, ladder swing speeds, and depth of cut. This does not suggest that restrictions should be placed on these parameters to minimize resuspension. In fact, data presented by Hayes et al. (1984) suggest that the optimum selection of these parameters for minimum resuspension generally corresponds to the selection for achieving highest production. So by properly optimizing production, as every dredge operator attempts to do, minimum resuspension will usually occur. The primary exception to this is the practice of undercutting to remove large bands of material (i.e. material thickness of 10 ft. or greater). This technique involves cutting the bank at near the project depth and allowing the large volume of overlying bank material to collapse into the cutterhead. Overload of suction capacity of the inlet pipe may occur, causing excess sediment particles to be resuspended rather than carried through the pipe. For this reason, excessive submergence of the cutterhead below the sediment line should be avoided.

Dustpan Dredge

The dustpan dredge is a hydraulic suction dredge that uses a widely flared dredgehead along which water jets are mounted. The jets loosen and agitate sediment particles, which are then captured in the dustpan dredgehead and the dredge moves forward. This type of dredge works best in free-flowing granular material and is not generally used to dredge fine-grained (clay) sediment. However, in 1982, an experiment was conducted using a modified dustpan head (without water jets) to dredge fine-grained sediment in the James River. A modified dustpan head and a conventional cutterhead were operated in the same reach of the river for comparison. It was hoped that the modified dustpan head using suction only could excavate thin layers of contaminated clay sediment with less resuspension than a cutterhead. Unfortunately, the dustpan head experienced repeated clogging and produced at least as much resuspension as the cutterhead operating in the same material (Raymond 1984).

Hopper Dredges

Hopper dredges typically remove sediment by dragging a large flat draghead and using hydraulic suction to remove the disturbed material. Because of the location of the drag arm beneath the dredge, it is difficult to measure the resuspension near the draghead; however, data presented by Hayes et al. (1984) indicated that the resuspension without overflow may actually be less than for a cutterhead dredge.

A hopper dredge can continue to operate beyond the initial filling of the hoppers and discharge overflow from the hoppers into surrounding waters, resulting in a large increase in the turbidity plume. The differences between the turbidity plume generated by overflow and nonoverflow operations are shown in Figure 2. This suggests that some restrictions on overflow may be necessary if a hopper dredge is used for removing contaminated sediment.

Bucket Dredges

Clamshell dredges, the most common type of bucket dredge, are typically used in areas where hydraulic dredges cannot work because of the proximity of piers, docks, etc. or where the disposal area is too far from the dredge site for it to be feasible for a cutterhead dredge to pump the dredged material. Resuspension from operation of open bucket clamshell dredges is typically higher than that from most cutterhead dredges. This resuspension is generally due to the dynamic impact of the bucket on the channel bottom, the spillage and leakage from the filled bucket, and the washing action of the empty bucket falling through the water column. Resuspension levels of the dredging operation are even higher when the scow is allowed to overflow.

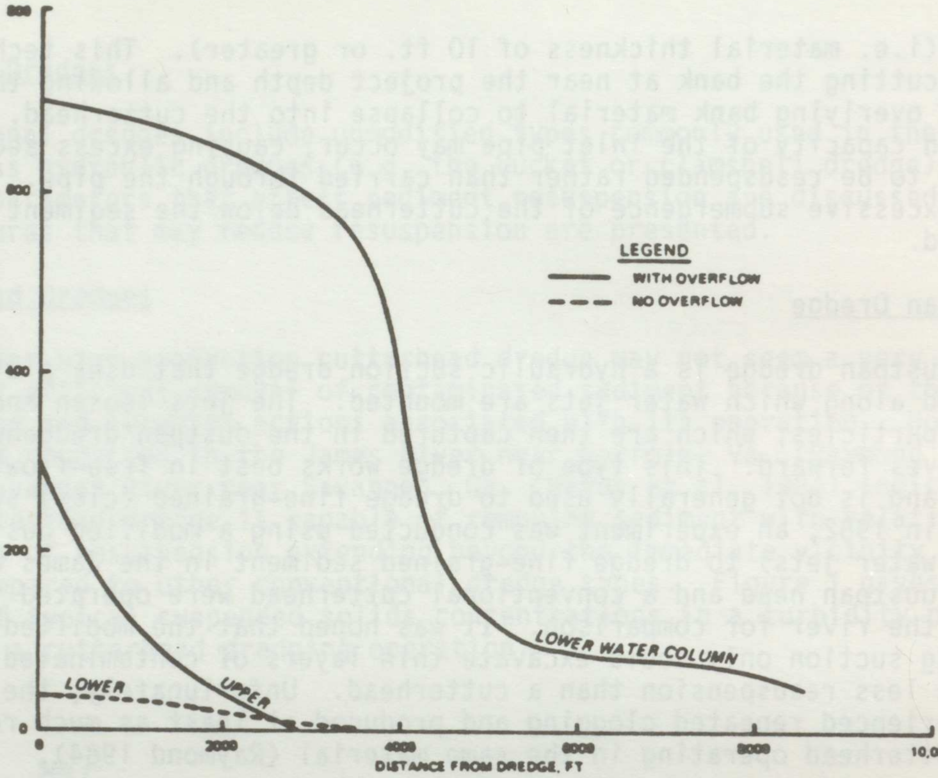

Figure 2. Resuspended sediment levels measured behind the dredge during hopper dredge operations in Grays Harbor with and without overflow.

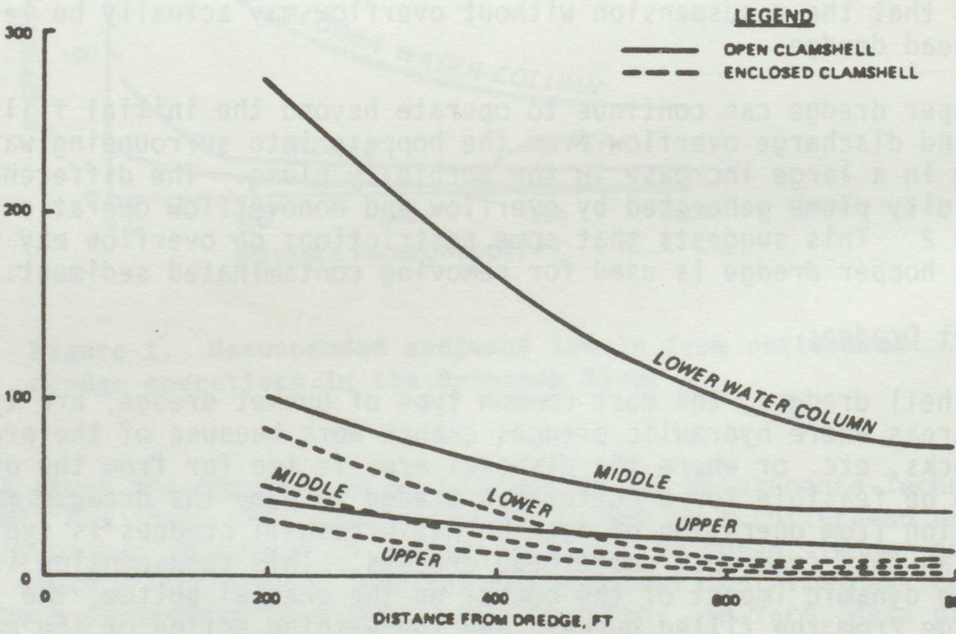

Figure 3. Resuspended sediment levels from open and enclosed clamshell dredge operations in the St. Johns River.

Sediment resuspension from clamshell dredges can be reduced by the use of an enclosed clamshell bucket. This bucket significantly reduces spillage and leakage, which are major contributors to the turbidity plume. Figure 3 shows the benefit of using an enclosed bucket. The operation of the dredge can be modified slightly to reduce sediment resuspension by slowing the raising and lowering of the bucket through the water column. It must be noted that this operational modification reduces the production rate of the dredge.

Special-Purpose Dredges

Special-purpose dredging systems have been developed during the last few years in the United States and overseas to pump dredged material slurry with a high solids content and/or to minimize the resuspension of sediment. Most of these systems are not intended for use on typical maintenance dredging operations; however, they may provide alternative methods for unusual dredging projects such as in removal of chemical hot spots (sediment contaminated by discharge or spill of hazardous material). The special-purpose dredges that appear to have the most potential in limiting resuspension are shown in the following tabulation, which is taken from Herbich and Brahme (in press).

NAME OF DREDGE	REPORTED SUSPENDED SEDIMENT CONCENTRATION*
Pnuema pump	48 mg/L 3 ft. above bottom. 4 mg/L 23 ft. above bottom (16 ft. in front of pump).
Clean-up system	1.1 to 7.0 mg/L above suction. 1.7 to 3.5 mg/L at surface.
Oozer pump	Background level (6 mg/L) 10 ft. from head.
Refresher system	4 to 23 mg/L 10 ft. from head.

*Suspended solids concentrations were adjusted for background concentrations.

SUMMARY

The IOMT research has shown that most conventional dredges can be used to remove sediment with a limited amount of sediment resuspension if they are properly operated and a few precautions are taken or plant modifications are made. This can be accomplished with only a small increase in cost over a normal dredging operation, and typically conventional dredging equipment is readily available. The data show that cutterhead dredges and hopper dredges with no overflow generate less resuspended sediment than mechanical dredges. The following tabulation gives a summary of suspended sediment levels observed during IOMT filed studies. However, in many cases, maneuverability requirements, hydrodynamic conditions, location of the disposal site, and other factors may dictate the type of dredge that must be used; the strategy then must be to minimize the resuspension levels generated by that dredge.

If no conventional dredge is acceptable, a special-purpose dredge may have to be selected. These dredges generally resuspend less material than conventional dredges, but associated costs may be much greater. As in the case of conventional dredges, the selection of a special-purpose dredge will likely be dictated by logistics, economics, and availability.

DREDGE TYPE	SUSPENDED SOLIDS CONCENTRATION, mg/L* WITHIN 100 ft.	WITHIN 200 ft.	WITHIN 400 ft.
Cutterhead	25 - 250	20 - 200	10 - 150
Hopper			
With overflow	250 - 700	250 - 700	250 - 700
Without overflow	25 - 200	25 - 200	25 - 200
Clamshell			
Open bucket	150 - 900	100 - 600	75 - 350
Enclosed bucket	50 - 300	40 - 210	25 - 100

*Suspended solids concentrations were adjusted for background concentration.

FUTURE DEVELOPMENTS

Research is being conducted to identify modifications to conventional dredges that may decrease the sediment resuspension to levels nearer those of special-purpose dredges. An example is the matchbox suctionhead tested by the U.S. Army Engineer District, Chicago. The Dutch-developed matchbox suctionhead entrains sediment into the suction pipe of a hydraulic dredge by using the swinging action to force material into a large funnel-shaped opening on one side of the suctionhead and adding water through the other side. Since the suctionhead is symmetrically designed, it will operate during swings in both directions.

REFERENCES

Hayes, D.F., G.L. Raymond, and T.N. McLellan. 1984. Sediment Resuspension from Dredging Activities. Proceedings of the Conference Dredging '84, American Society of Civil Engineers, Clearwater Beach, Fla.

Herbich, J.B. and S.B. Brahme. In Press. A Literature Review and Technical Evaluation of Sediment Resuspension During Dredging. Technical Report HL-87-, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Raymond, G.L. 1984. Techniques to Reduce the Sediment Resuspension Caused by Dredging. Miscellaneous Paper HL-84-3, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

APPENDIX II

TERMS OF REFERENCE AND MEMBERSHIP

TERMS OF REFERENCE
FOR THE
REMEDIAL OPTIONS WORK GROUP
OF THE
SEDIMENT SUBCOMMITTEE

The Remedial Options Work Group will report to the Sediment Subcommittee which will review its progress, approve its work plan and membership. Upon completion of its terms of reference, the work group will be dissolved by the Sediment Subcommittee.

TERMS OF REFERENCE

1. Review existing technologies for the remediation of ecosystem related impacts due to sediment contaminants.

2. Evaluate the effectiveness and feasibility of existing technologies.

3. Develop a system for evaluating the most applicable technology to be used for remediating identified ecosystem impacts due to sediment contamination in the nearshore areas of the Great Lakes basin.

4. Identify research needs to further test existing technologies or establish new approaches to mitigate sediment contaminant problems.

5. Establish in conjunction with the Assessment Work Group and other committees, work groups or task forces, as necessary, a monitoring program to assess any adverse effects on the ecosystem which may result from moving or otherwise isolating existing contaminated sediments from their present location.

6. Assist the Sediment Subcommittee in conducting workshops, round table discussions or special meetings in order to fulfill its terms of reference.

MEMBERSHIP

Members of the Remedial Options Work Group shall be drawn from the Sediment Subcommittee as well as research, surveillance or management agencies to provide the necessary expertise.

REMEDIAL OPTIONS WORK GROUP

SEDIMENT SUBCOMMITTEE

Mr. Ian Orchard (Chairman)
Environmental Protection Service
Ontario Region
Environment Canada
25 St. Clair Avenue East
Toronto, Ontario M4T 1M2
(416) 973-1089

Mr. Ansar Khan
Environmental Officer
Design and Construction
Public Works Canada
Ontario Region
4900 Yonge Street
Willowdale, Ontario M2N 6A6
(416) 224-4119

Mr. Paul Mudroch
Ocean Dumping Program Officer
Environmental Emergencies & Operational Support Branch
Conservation & Protection
Environment Canada
351 St. Joseph Blvd.
Ottawa, Ontario K1A 0E7
(819) 997-2046

Ms. Alena Mudroch
Lakes Research Branch
National Water Research Institute
Environment Canada
P.O. Box 5050, 867 Lakeshore Road
Burlington, Ontario L7R 4A6
(416) 336-4707

Dr. Mark P. Brown, Chief
Special Projects Section
Division of Water
New York State Department of Environmental Conservation
50 Wolf Road
Albany, New York 12233-3502
(518) 457-7470

Mr. Robert Wysenski
(Replaced R. Manson†)
Division of Water Quality Monitoring and Assessment
Northeast District Office
Ohio Environmental Protection Agency
2110 East Aurora Road
Twinsburg, Ohio 44087
(216) 425-9171

Mr. Roy Deda (Replaced D. Raven)
North Central Division
Corps of Engineers
536 South Clark Street
Chicago, Illinois 60605
(312) 353-7762
FTS: 353-7762

Mr. Robert Dalrymple
Waste Management Branch
Policy Section
Ontario Ministry of the Environment
40 St. Clair Avenue West, 5th Floor
Toronto, Ontario M4W 1M2
(416) 323-5211

Mr. Deo Persaud
Water Resources Branch
Ontario Ministry of the Environment
135 St. Clair Avenue West
Toronto, Ontario M4V 1P5
(416) 323-4926

Secretariat Responsibilities:

Dr. Michael A. Zarull
Great Lakes Regional Office
International Joint Commission
100 Ouellette Avenue, 8th Floor
Windsor, Ontario N9A 6T3
(519) 256-7821 Windsor
(313) 226-2170 Detroit
FTS: 226-2170

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APPENDIX III

TERMS OF REFERENCE AND MEMBERSHIP

APPROVED BY IJC
1986.12.09

TERMS OF REFERENCE
FOR THE
SEDIMENT SUBCOMMITTEE
OF THE
WATER QUALITY PROGRAMS COMMITTEE

BACKGROUND

Sediment can function as a sink for contaminants and also as a source of these contaminants to the water column and the biotic community. The magnitude and the conditions for the transfer of contaminants either way, is largely unknown. Therefore, it is difficult to establish the relative significance of the sediment as a sink or source of contaminants compared with other sources and pathways.

Previously, the focus has been on contaminated sediment in relation to dredging and disposal operations. This was the thrust of the Water Quality Board's Dredging Subcommittee, which largely completed its charge under its terms of reference. The current uncertainty over management and technical options available to address the issue of contaminated sediments affects the final resolution of most of the Areas of Concern identified by the Water Quality Board. In order to address the broader issue of sediment management in the Great Lakes ecosystem, the Water Quality Board has broadened the role of the Dredging Subcommittee to that of a Sediment Subcommittee.

TERMS OF REFERENCE

The Sediment Subcommittee will report to the Water Quality Programs Committee. The primary focus will be on management options for contaminated sediment in areas of concern and on the Critical Pollutants, as identified by the Water Quality Board. The Subcommittee will:

1. Review existing protocols designed to quantify the transfer of contaminants to and from sediments and to establish ecosystem impact. Based on this review, recommend protocols to be used or specific research to be undertaken to define the significance of these in-place pollutants in Areas of Concern.

2. Review existing technologies, including removal and disposal, treatment, capping and others for remediation of in-place pollutant impacts. Based on this review, evaluate the effectiveness, feasibility and costs of existing techniques. Report on the most promising and/or proven technologies for application to Areas of Concern and recommend technologies that should be further evaluated or demonstrated.

3. Maintain a registry where remediation of the contaminated sediments has been attempted or proposed. Use these examples for evaluating or demonstrating mitigative techniques.

4. Periodically review criteria and guidelines for the classification of sediments.

5. Maintain a register of significant dredging projects in the Great Lakes Basin with information on sediment volume and contaminant concentation.

6. Facilitate the exchange of sediment management information including, but not limited to, information relating to: development of sediment management technology; development of sediment evaluation protocols; procedures to characterize and quantify mass transport; and fate and effect of sediment and associated contaminants.

7. Identify research and information needed to remediate problems associated with contaminated sediment and encourage research and demonstration to investigate advances in sediment management technology and the pathways, fate and effects of sediments and sediment contaminants.

8. Develop a work plan for submission to the Water Quality Programs Committee, in accordance with the Board's planning and budget requirements. Review the work plan and revise as needed at least once a year.

9. Prepare periodic reports on the above items and undertake other activities as directed by the Water Quality Board.

MEMBERSHIP

Members of the Sediment Subcommittee shall be drawn from the jurisdictions or other organizations engaged in sediment management and related activities, and shall serve in their personal and professional capacity.

In consultation with the Water Quality Programs Committee, the Subcommittee may establish task forces to address specific sediment-related issues as need be. A chairperson shall be designated by the Water Quality Board for a two-year term.

SEDIMENT SUBCOMMITTEE

of the

WATER QUALITY PROGRAMS COMMITTEE

U.S. MEMBERS

Dr. Michael J. Mac
National Fisheries Center-Great Lakes
U.S. Fish and Wildlife Service
1451 Green Road
Ann Arbor, Michigan 48105
(313) 994-3331, Ext. 245
FTS: 378-1245

Ms. Linda Talbot, WR/2
Water Resources Management Program
Wisconsin Department of Natural Resources
P. O. Box 7921
Madison, Wisconsin 53707
(608) 266-8148

Mr. Anthony G. Kizlauskas
Great Lakes National Program Office
5GL-TUB-10
U.S. Environmental Protection Agency
230 South Dearborn Street
Chicago, Illinois 60604
(312) 353-3576
FTS: 353-3576

Dr. Mark P. Brown, Chief
Special Projects Section
Division of Water
New York State Department of
Environmental Conservation
50 Wolf Road
Albany, New York 12233-3502
(518) 457-7470

Ms. Diana Klemans
Surface Water Quality Division
Michigan Department of Natural Resources
Stevens T. Mason Building
P.O. Box 30028
Lansing, Michigan 48909
(517) 373-2758

8/88

U.S. Members - cont'd.

Mr. Roy Deda (Replaced D. Raven)
Department of the Army
North Central Division
Corps of Engineers
536 South Clark Street
Chicago, Illinois 60605
(312) 353-6373
FTS: 353-6373

Mr. Robert Wysenski
(Replaced R. Manson†)
Division of Water Quality Monitoring
and Assessment
Northeast District Office
Ohio Environmental Protection Agency
2110 East Aurora Road
Twinsburg, Ohio 44087
(216) 425-9171

Dr. Brian J. Eadie
Great Lakes Environmental
Research Lab/NOAA
U.S. Dept. of Commerce
2205 Commonwealth Blvd.
Ann Arbor, Michigan 48105
(313) 668-2281
FTS: 378-2281

CANADIAN MEMBERS

Mr. Deo Persaud (Chairman)
Water Resources Branch
Ontario Ministry of the Environment
135 St. Clair Avenue West
Toronto, Ontario M4V 1P5
(416) 323-4926

Mr. Ian Orchard
Environmental Protection Service
Ontario Region
Environment Canada
25 St. Clair Avenue East
Toronto, Ontario M4T 1M2
(416) 973-1089

SEDIMENT SUBCOMMITTEE - cont'd

Canadian Members - cont'd.

Mr. Ansar Khan
Environmental Officer
Design and Construction
Public Works Canada
Ontario Region
4900 Yonge Street
Willowdale, Ontario M2N 6A6
(416) 224-4119

Dr. Mohi Munawar
Research Scientist
Great Lakes Laboratory for Fisheries & Aquatic Sciences
Department of Fisheries & Oceans
Canada Centre for Inland Waters
P.O. Box 5050, 867 Lakeshore Road
Burlington, Ontario L7R 4A6
(416) 336-4867

Dr. Trefor B. Reynoldson
National Water Research Institute
Environment Canada
Canada Centre for Inland Waters
P.O. Box 5050, 867 Lakeshore Road
Burlington, Ontario L7R 4A6
(416) 336-4692

Dr. John P. Coakley
National Water Research Institute
Environment Canada
Canada Centre for Inland Waters
P.O. Box 5050, 867 Lakeshore Road
Burlington, Ontario L7R 4A6
(416) 336-4881

Secretariat Responsibilities:

Dr. Michael A. Zarull
Great Lakes Regional Office
International Joint Commission
100 Ouellette Avenue, 8th Floor
Windsor, Ontario N9A 6T3
(519) 256-7821 Windsor
(313) 226-2170 Detroit
FTS: 226-2170

International Joint Commission
Commission mixte internationale

100, avenue Ouellette Avenue, Windsor, Ontario N9A 6T3
or/ou P.O. Box 32869, Detroit, Michigan 48232