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Dredging Operations Technical Support Program

Environmental Evaluation and Management of Dredged Material for Beneficial Use

A Regional Beneficial Use Testing Manual for the Great Lakes

Karen G. Keil, Trudy J. Estes, Joseph P. Kreitinger, Guilherme R. Lotufo, Richard A. Price, Burton C. Suedel, Michael W. Habberfield, Bryan A. Hinterberger, Andrew M. Lenox, Scott W. Pickard, Martin P. Wargo, Jason M. Miller, Jennifer A. Miller, and Paul R. Schroeder

August 2022

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Final report

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Abstract

The *Environmental Evaluation and Management of Dredged Material for Beneficial Use: A Regional Beneficial Use Testing Manual for the Great Lakes* (a.k.a. *Great Lakes Beneficial Use Testing Manual*) is a resource document providing technical guidance for evaluating the suitability of dredged sediment for beneficial use in aquatic and terrestrial environments in the Great Lakes region. The procedures in this manual are based on the Environmental Laboratory extensive research, working with US Army Corps of Engineers (USACE) Great Lakes districts, state resource agencies, and local stakeholders seeking to develop dredged material beneficial use alternatives consistent with regional needs and goals. This manual is the first guidance document developed by USACE for evaluating the environmental suitability of dredged material specifically for beneficial use placements. It provides a tiered framework for evaluating the environmental suitability of aquatic and upland beneficial uses consistent with the Inland Testing Manual and the Upland Testing Manual. This manual is intended to serve as a regional platform to increase collaborative problem-solving and endorse a common understanding of the scientific and institutional practices for evaluating dredged material for any beneficial use. Dredged sediment may be managed as a valuable resource, with great potential to create economic, environmental, and social benefits.

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Contents

Figures and Tables

Figures

Tables

Preface

This study was conducted for the US Army Corps of Engineers Buffalo, Chicago, and Detroit (Great Lakes) Districts, under the Dredging Operations and Technical Support Program, Funding Account Code U4355208; AMSCO Code 086000.

The work was performed by the Environmental Engineering (EPE) and Environmental Risk Assessment (EPR) Branches of the Environmental Processes and Engineering Division (EPED), US Army Engineer Research and Development Center, Environmental Laboratory (ERDC-EL). At the time of publication of this report, Dr. Michael A. Rowland was chief of EPE; Mr. James H. Lindsay was chief of EPR; Mr. Warren P. Lorentz was chief of EPED; and Dr. Jen Seiter-Moser was the technical director for Civil Works Environmental Engineering and Sciences. The deputy director of ERDC-EL was Dr. Brandon Lafferty, and the director was Dr. Edmond J. Russo.

The Great Lakes Dredging Team has served as a forum for input from federal, state, and stakeholder agencies to this manual. Several rounds of feedback were considered, discussed, and incorporated over a stakeholder review and revisions period that occurred between 2016 and 2020. Key feedback was received from the US Environmental Protection Agency Regions 2 and 5, and its Office of Land and Emergency Management, US Geological Survey Columbia Environmental Research Center, Great Lakes Commission, Illinois Environmental Protection Agency, Indiana Department of Environmental Management, Michigan Environment Great Lakes and Energy, Minnesota Pollution Control Agency, New York State Department of Environmental Conservation, Ohio Environmental Protection Agency, Ohio Department of Natural Resources, Pennsylvania Department of Environmental Protection, Wisconsin Department of Natural Resources, along with other municipal, academic, and private entities.

A full list of acknowledged contributors to the manual is presented following this preface.

This manual is dedicated to Mr. Tony Friona, who was an early proponent of the beneficial use of dredged material to support remedial and restoration efforts across the Great Lakes. Officially, Tony served as the

USACE Regional Working Group co-lead for the Great Lakes Restoration Initiative. Unofficially, he was so much more: a visionary colleague and friend who, with his contagious enthusiasm, inspired us to work together to better our region. His light will continue to shine on in the work that we do and in the relationships we form along the way.

The commander of ERDC was COL Christian Patterson, and the director was Dr. David W. Pittman.

Disclaimer

This manual is intended to provide a source of comprehensive regional guidance for testing and evaluating the environmental suitability of dredged sediment for a range of beneficial placement alternatives using risk-based principles to comply with applicable environmental requirements. It is not intended to direct the public but rather to provide a framework and testing methods for environmental scientists, engineers, planners, managers, and regulatory specialists within the US Army Corps of Engineers and other agencies. It is not binding, nor does it regulate or change any authority in determining environmental suitability for the management of dredged material.

This manual does not alter or attempt to replace existing formal federal guidance or guidelines directed at evaluating discharges of dredged sediment into waters of the United States, nor does it infringe on any state authority to determine compliance with applicable water quality standards.

This manual does not, and is not intended to, impose any legally binding requirements on federal agencies, states, or the regulated community.

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The following individuals and agencies contributed to the development of the manual.

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Executive Summary

The *Environmental Evaluation and Management of Dredged Material for Beneficial Use: A Regional Beneficial Use Testing Manual for the Great Lakes* (a.k.a. *Great Lakes Beneficial Use Testing Manual*) is a resource document providing technical guidance for evaluating the suitability of Great Lakes dredged sediment for beneficial use in aquatic and terrestrial environments in the Great Lakes region. It was developed to support opportunities for beneficial use of dredged material by providing a tiered framework for evaluating the environmental suitability of such uses consistent with the Inland Testing Manual (USEPA/USACE 1998a) and the Upland Testing Manual (USACE 2003). Dredged sediment may be managed as a valuable resource, with great potential to create economic, environmental, and social benefits. This *Great Lakes Beneficial Use Testing Manual* was written with resource agencies, regulatory managers, and practitioners in mind.

The manual is intended to serve as a regional platform to increase collaborative problem-solving and endorse a common understanding of the scientific and institutional practices for evaluating dredged material for any beneficial use in aquatic, nearshore, wetland, or upland environments. It also provides characterization procedures for assessing dredged material use in material production processes.

This manual does not interpret or validate regulatory authority granted the US Army Corps of Engineers (USACE), the US Environmental Protection Agency (USEPA), or Great Lakes state resource agencies. The intent is that the *Great Lakes Beneficial Use Testing Manual* offers science-based guidance, yet it does not alter the statutory and regulatory framework for permitting decisions under applicable laws or regulations. The *Great Lakes Beneficial Use Testing Manual* is not intended, nor can it be relied upon, to create rights or obligations enforceable by any party. The *Great Lakes Beneficial Use Testing Manual* does not, and is not intended to, impose legally binding requirements on federal agencies, states, or the regulated communities.

The procedures in the *Great Lakes Beneficial Use Testing Manual* are based on the USACE extensive research and on established procedures for evaluating dredged material management alternatives. It is the first guidance document developed by USACE for evaluating the environmental

suitability of dredged material specifically for beneficial use placements. This manual originated in 2011 from the practical experience of Mr. Richard A. Price (ret.), US Army Engineer Research and Development Center (ERDC), working with USACE Great Lakes districts, state resource agencies, and local stakeholders seeking to develop dredged material beneficial use alternatives consistent with regional needs and goals. The Great Lakes Dredging Team has served as a forum for input from federal, state, and stakeholder agencies to this manual. Several rounds of feedback were considered, discussed, and incorporated over a stakeholder review and revisions period that occurred between 2016 and 2020.

Although the final language in this manual reflects the USACE scientific recommendations, resource managers looking to implement beneficial use of dredged material projects at individual sites can develop quality assurance project plans that provide more site- and project-specific details to their environmental evaluations. These site-specific project plans can also identify the methods for interpreting and evaluating data generated and the resulting decisions to be made. Site-specific plans should reflect the local, project-specific stakeholder partnership priorities or needs addressing dredged material management at that individual site. Development of this regional approach supports rather than precludes the implementation of site-specific plans for making those shared dredged material management decisions.

This document is designed to support environmental scientists, engineers, planners, program managers, project managers, and regulatory review specialists within USACE, other federal agencies, and Great Lakes state agencies. The *Great Lakes Beneficial Use Testing Manual* appendices that present methods or resources will be updated or added to, as practical, when additional research developments and/or regulatory updates are completed and implementation experience is gained. Users are encouraged to obtain the most recent version of the manual, maintained on the USACE ERDC website https://dots.el.erdc.dren.mil/guidance.html.

1 Introduction

The US Army Corps of Engineers (USACE) maintains navigation features across 11,930 mi[1,2] of waterways across the United States, including some 600 mi of channels within the Great Lakes Navigation System[3]. This responsibility requires dredging up to 300 million cy of sediment annually of which 3 million cy are in the Great Lakes, all of which requires methods to evaluate and determine environmentally sound dredged material management alternatives. In addition, other entities dredge a variety of projects across the Great Lakes basin and similarly need to manage dredged sediment. Dredged sediment can be managed in a variety of ways in aquatic and upland environments to provide opportunities for beneficial use. This manual was developed by USACE to provide a regional approach for the technical evaluation of dredged material proposed for beneficial use in the Great Lakes region.

In this manual, the term *dredged material* is intended to be synonymous with the term *dredged sediments*. Although the term *sediments* appropriately describes the material typically dredged from navigation channels, the term *dredged material* may be used more broadly to describe rock, gravel, and various anthropogenic materials including coal ash, sawdust, woody debris, and mine tailings that may be removed during dredging. In some instances, especially under the US Environmental Protection Agency (USEPA) Great Lakes Area of Concern (AOC) program (see Section 1.1.2), these residuals from historical industrial activities have been dredged as part of site remediation and habitat restoration projects. The beneficial use of these materials is sometimes considered, depending on a state's regulatory framework, and may follow the same federal regulatory requirements and risk assessment processes described in this document. Some examples of dredged anthropogenic materials include

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[1] For a full list of the spelled-out forms of the units of measure used in this document, please refer to US Government Publishing Office Style Manual, 31st ed. (Washington, DC: US Government Publishing Office 2016), 248-52, https://www.govinfo.gov/content/pkg/GPO-STYLEMANUAL-2016/pdf/GPO-STYLEMANUAL-2016.pdf.

[2] For a full list of the unit conversions used in this document, please refer to US Government Publishing Office Style Manual, 31st ed. (Washington, DC: US Government Publishing Office 2016), 345-7, https://www.govinfo.gov/content/pkg/GPO-STYLEMANUAL-2016/pdf/GPO-STYLEMANUAL-2016.pdf.

[3] The Great Lakes navigation system is a continuous 27 ft deep-draft waterway that extends from the western end of Lake Superior at Duluth, MN, to the Gulf of St. Lawrence on the Atlantic Ocean; a distance of over 2,400 mi. https://www.lre.usace.army.mil/Missions/Great-Lakes-Navigation/

woody debris located at historical shoreline logging operations and lumber mills (e.g., composting for topsoil amendments; Duluth, MN) and stamp sands from milling copper ore (e.g., road base; Torch Lake, MI).

1.1 Background

1.1.1 Dredged material management

Prior to the passage of federal laws described in Section 2, decisions about the management of dredged sediment were based primarily on ease of operations, cost effectiveness, or local needs that yielded some economic or environmental benefits. Environmental or ecological concerns were oftentimes minimized because the effects of loss of habitat or contamination on fish, wildlife, and humans were not well understood or prioritized. If the dredged sediment was considered physically suitable for any particular need, it was used as such assuming it was cost effective and feasible from an engineering perspective. Many developed areas along Great Lakes coastlines were constructed using dredged material as fill.

Increased concern for the dredging effects of habitat destruction and contamination on fish, wildlife, and humans led to increasingly stringent environmental regulations on dredging in the 1970s. The changes resulted in USACE policy that dredging “shall be accomplished in an efficient, cost-effective, and environmentally acceptable manner to improve and maintain the nation’s waterways to make them suitable for navigation and other purposes consistent with federal laws and regulations” (USACE 1996b). General considerations associated with dredging and dredged material management are outlined in EM 1110-2-5025 (USACE 2015).

Beneficial uses of dredged sediment for habitat restoration have a productive history resulting in more than 2,000 man-made islands, 100 marshes, and nearly 1,000 habitat development projects across the nation. In many areas, islands constructed by the USACE using dredged sediment provide vital habitat for rare, threatened, or endangered species. Circa 1986, it was estimated that 1,000,000 birds nest on dredged material islands each year (Landin 1986). Beach nourishment projects, which provide both habitat and recreational areas, have long been conducted using dredged sediment (NOAA 2000; USACE 1996a).

Recently, USACE has embarked on a single collaborative and cost-effective approach for infrastructure development and environmental management known as Engineering With Nature[4]. Engineering With Nature is the intentional alignment of natural and engineering processes to efficiently and sustainably deliver economic, environmental, and social benefits through collaborative processes. Many beneficial uses of dredged material encompass these principles. Engineering With Nature directly supports the USACE *Sustainable Solutions to America's Water Resources Needs: Civil Works Strategic Plan 2014–2018* and contributes to the achievement of its civil works mission and goals (USACE 2014; Bridges et al. 2021, 2018). One application of the Engineering With Nature concept is strategic placement that leverages nearshore currents to transport the placed sediment to its intended destination and provide the desired benefits (Gailani et al. 2019).

1.1.2 Dredged material management in the Great Lakes

The Great Lakes ports and harbors are maintained by the USACE Great Lakes Districts (Detroit, Chicago, and Buffalo) in a coordinated manner as they operate as a system of interdependent ports. The vast majority of trade in the Great Lakes is within the system. Eight states (Illinois, Indiana, Michigan, Minnesota, New York, Ohio, Pennsylvania, and Wisconsin) border the five Great Lakes (Erie, Huron, Michigan, Ontario, and Superior). Dredged material management decisions require federal, state, and community input. Some of the ports and harbors straddle two different states or involve maintenance activities that can affect more than one state (e.g., Duluth-Superior, Toledo, and Conneaut), which can create challenges for dredged material management decision-making. Therefore, developing a regional approach towards dredged material management is appropriate.

The Great Lakes annual dredging volume is typically 3 million cy—just 1% of the national dredging volume. However, this comparably smaller dredging volume does not result in proportionally smaller costs and/or management needs. Costs to place dredged sediment in the Great Lakes may be twice the national average (on a volumetric unit basis) because of greater handling needs (double and triple handling), a history of more contaminated material requiring confined disposal, higher land costs for confined disposal, higher transportation costs, shorter seasons, and seasonal work limitations due to environmental windows, rough wave climate in outer harbors causing inefficient dredging, smaller harbors and thus smaller dredging volumes per

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[4] See https://ewn.el.erdc.dren.mil/.

project, and a limited pool of dredging contractors. These greater costs require an evaluation of management alternatives that can meet the current dredging demand and also future dredging needs, as affected by climate change (lake level changes and potential increased erosion due to more frequent storm events) and demands for increased cargo capacity for Great Lakes transportation companies to remain competitive nationally and globally. Management alternatives include open-water/nearshore, or upland placement; each offers opportunities to provide additional benefits beyond channel maintenance.

Dredged sediment in federal navigation channels of the Great Lakes has become increasingly cleaner over the last 4 decades as a result of pollution control measures and sediment remediation that has occurred in Great Lakes Areas of Concern (AOCs) under the auspices of the Great Lakes Legacy Act. The 43 Great Lakes AOCs were designated by the International Joint Commission as geographic areas within the Great Lakes basin having environmental degradation (USEPA 2020a). The AOCs are characterized by a variety of beneficial use impairments such as the following:

- Restrictions on fish and wildlife consumption
- Tainting of fish and wildlife flavor
- Degraded fish and wildlife populations
- Fish tumors or other deformities
- Bird or animal deformities or reproductive problems
- Degradation of benthos
- Restrictions on dredging activities
- Eutrophication or undesirable algae
- Restrictions on drinking water consumption or taste and odor problems
- Beach closings
- Degradation of aesthetics
- Added costs to agriculture or industry
- Degradation of phytoplankton and zooplankton populations
- Loss of fish and wildlife habitat.

In 2010, the Great Lakes Restoration Initiative was launched to accelerate removing beneficial use impairments from the AOCs. Beneficial use of dredged sediment has been instrumental in habitat restoration, supporting the removal of beneficial use impairments associated with degradation of habitat in some of these AOCs, especially in Duluth-Superior Harbor.

Between 2015 and 2020, there has been an increase in the number and variety of projects involving the beneficial use of dredged material in Duluth-Superior, many involving restoration of aquatic habitat.

In 2016, the United States Congress passed the Water Infrastructure Improvements for the Nation Act, which included a provision (Section 1122) stipulating that the USACE establish a pilot program to carry out 10 cost-sharing projects for the beneficial use of dredged material. The projects are exempt from federal standard requirements. Under this authorization, the Chicago District began supporting a project to pilot the beneficial use of Waukegan Harbor dredged material for ecosystem restoration on shoreline areas in four Illinois coastal communities. The pilot project would be a proof-of-concept to determine the following:

- The potential for significant benefits to natural and cultural resources to occur.
- Whether similar placement sites could be utilized between the Wisconsin/Illinois state border and the northern city limits of Chicago, assuming environmental compliance documentation is completed.
- The anticipated costs for communities to implement this strategy in the future, particularly as they compare to the cost of trucking in quarried sand for coastal nourishment.
- Whether the implementation process and the final product are satisfactory to local municipalities.
- How long the material can be expected to stay in place before reapplication is required.

Depending on these outcomes, the proposed pilot project has the potential to become a new tool for regional stakeholders who are hoping to expand sustainable and collaborative shoreline management options in the region.

Three other broad programmatic efforts also operate to support beneficial use of dredged material projects in the Great Lakes. Projects involving beneficial use of dredged material to restore habitat have been implemented as a result of Natural Resource Damage Assessment and Restoration efforts in some Great Lakes harbors (https://www.doi.gov/restoration/about). There are also a number of USACE projects related to beneficial use of dredged sediments that illustrate USACE Engineering With Nature (https://ewn.el.erdc.dren.mil/) principles and practices that have broad applicability in Great Lakes waters (e.g., Cat

Island in Green Bay, WI; see GLDT [2016]). In addition, beneficial use of dredged sediment is a component of the USACE efforts supporting regional sediment management (https://rsm.usace.army.mil/). Regional sediment management is a systems approach using best management practices for efficient and effective use of sediments in coastal, estuarine, and inland environments.

Recognizing the benefit of conserving CDF capacity through removal and beneficial use of dredged material and the lack of consistent guidance from state to state, the Great Lakes Commission published guidance entitled *Testing and Evaluation of Dredged Material for Upland Beneficial Uses—A Regional Framework for the Great Lakes* (GLC 2004a). The commission prepared the guidance based on recommendations from the Great Lakes Dredging Team–Great Lakes Beneficial Use Task Force as a framework for merging the regulatory requirements of the Great Lakes states with rules and regulations implemented by the USEPA and USACE.

The Great Lakes Dredging Team recently published *Guide to Policies and Projects Related to Beneficial Use of Dredged Material in the Great Lakes* (GLDT 2016), which may be considered a companion document to this *Great Lakes Beneficial Use Testing Manual*. That guide provides an overview of Great Lakes state and federal policies pertaining to the beneficial use of dredged material, case studies of projects around the basin and the nation, and *lessons learned* from successful projects. In addition, other Great Lakes Dredging Team publications (newsletters, brochures, and posters) showcase regional projects involving beneficial uses of dredged material (https://www.lre.usace.army.mil/Missions/Great-Lakes-Information/Great-Lakes-Dredging-Team/Publications//).

Despite active source control measures and remediation of legacy contaminated sediments, some residual contamination may still remain as the limits of anthropogenic and natural source reductions within the Great Lakes are reached. Given atmospheric deposition and other anthropogenic sources, some materials still contain trace levels of these constituents. This should be taken into consideration when assessing the suitability of dredged material for beneficial uses within the boundaries of the watershed from which it was derived. New or emerging contaminants may be present that should be considered when developing a conceptual site model to frame the evaluations (Section 4.3). Contaminant concentrations in dredged material are best understood in terms of the risks associated with exposure and effects to ecological and/or human receptors, in context with the existing exposures and associated effects within the watershed. Bulk sediment chemistry alone without consideration of chemical bioavailability, and associated toxicity, should not be the determining factor regarding environmental suitability of dredged material for a given placement option (e.g., ITRC [2011b]; Appendix F of this manual).

1.1.3 Development of dredged material evaluation frameworks

USACE and USEPA have jointly developed a series of guidance documents addressing various aspects of dredged sediment evaluation and management. USACE has developed additional guidance documents on this subject. These are described here in the chronological order in which they were originally developed.

In 1987, USACE published two engineering manuals (*Beneficial Use of Dredged Material*, and *Confined Disposal of Dredged Material*), which have since been subsumed into and superseded by EM 1110-2-5025, *Dredging and Dredged Material Management* (USACE 2015). The original discussion of beneficial uses of dredged material in the 1987 publication focused on suitability as a function of the physical attributes of the dredged sediment but did acknowledge that the chemical and biological attributes of sediment must also be considered to determine if placement options are environmentally suitable (USACE 1987). The updates provided on this topic in the 2015 engineering manual do not fully consider the holistic risk-based approach toward determining environmental suitability of dredged material for various placement options that are outlined in this manual (as further described in the final paragraph of this sub-section).

The first joint USACE/USEPA guidance document in this series includes a document entitled *Evaluating Environmental Effects of Dredged Material Management Alternatives — A Technical Framework* (*Technical Framework;* originally drafted in 1992 and revised in 2004) (USEPA/USACE 2004). The *Technical Framework* provided guidance for evaluating and selecting alternatives for the full range of management options: in-water placement, confined disposal facility (CDF) placement and beneficial use applications. The risk-based approach presented in Section 4 of this manual is generally consistent with that framework but updates and expands upon some of the evaluations outlined therein.

In 1998, USACE and USEPA jointly published two documents specifically focused on aquatic placements of dredged sediment, which historically have involved open water placement. The *Evaluation of Dredged Material Proposed for Discharge in Waters of the US — Testing Manual,* (commonly referred to as the *Inland Testing Manual* [ITM] (USEPA /USACE [1998a]) and *Great Lakes Dredged Material Testing and Evaluation Manual* (commonly referred to as the *Great Lakes Testing Manual*; USEPA/USACE 1998b) provide the driving regulatory guidance for evaluating contaminant-related effects. These two manuals are directed at the "contaminant determination" of the Clean Water Act (CWA) Section 404(b)(1) Guidelines (40 Code of Federal Regulations [CFR] 230) and are used as the basis for evaluations presented in Section 5 of this manual.

Generally, state regulatory agencies (rather than USEPA or USACE) are the main entity responsible for evaluating environmental compliance for upland beneficial uses (not including those evaluated under the CWA for confined disposal of dredged sediment in the aquatic environment or as return water) under Resource Conservation and Recovery Act (RCRA) authority for regulating the reuse of solid waste for beneficial purposes. Given the flux in development of technical guidance and establishment of standards in many states, other USACE guidance can be used to support evaluations of environmental suitability for the upland placement of dredged material. The *Evaluation of Dredged Material Proposed for Disposal at Inland, Nearshore, or Upland Confined Disposal Facilities—Testing Manual*, commonly called the Upland Testing Manual (UTM) (USACE 2003), provides testing guidance to evaluate potential risks associated with contaminant migration pathways, including groundwater, surface water, volatilization, and plant and animal bioaccumulation. The testing methods provided in the UTM are useful in assessing potential impacts from these exposure pathways for unconfined upland placement, including for beneficial use, and are placed into context in this manual (Section 6).

Two more recent joint USACE and USEPA manuals were developed to specifically address the beneficial use of dredged material but focused on aspects other than evaluations for determining environmental suitability of the material. The *Identifying, Planning and Financing Beneficial Use Projects Using Dredged Material – Beneficial Use Planning Manual* was created to assist in identifying project partners, planning strategies, financing alternatives, and public input to facilitate beneficial use projects (USEPA/USACE 2007a). A companion document, *The Role of the Federal Standard in the Beneficial Use of Dredged Material from US Army Corps of Engineers New and Maintenance Dredging Projects* (USEPA/USACE 2007b) provides guidance on the role of the federal standard in implementing beneficial use of dredged material. The federal standard is the dredged sediment disposal alternative or alternatives identified by USACE that represent the least costly alternative consistent with sound engineering practices and that meet the environmental standards established by the CWA Section 404(b)(1) evaluation process. The federal standard may be considered a *base plan* that defines the costs associated with the navigational purpose of the project and has a cost-sharing structure specific to the navigation project (dependent upon, for example, whether it is a new navigation project or operation and maintenance of an existing project). The federal standard does not necessarily define the disposal or placement option for a project, which may include beneficial uses with an incremental cost-sharing structure dependent on the project type. While these two manuals do not provide guidance specifically addressing environmental suitability of the dredged material for a given beneficial use placement option, they provide relevant context. The role of the federal standard in beneficial use projects may be evolving since 2007. Section 125 of the Water Resources Development Act of 2020 requires the consideration of the suitability of dredged material for beneficial uses and the economic and environmental benefits, efficiencies and impacts of beneficial uses. Additionally, Section 125 specifies that the economic benefits and efficiencies from beneficial use be included in the determination of the federal standard.

A dredged material management plan (DMMP) is a USACE planning document developed for its civil works water resource projects for navigation (USACE 2000). It is used to assess and recommend an alternative(s) within a federal harbor that would provide for dredged sediment management over at least a 20 yr period. A DMMP evaluates the engineering, economic, and environmental effects of various dredged sediment management alternatives and also includes an environmental assessment or environmental impact statement in compliance with the National Environmental Policy Act (NEPA) (see Section 2). One or more beneficial use alternatives could be considered within a particular harbor and recommended by USACE, assuming there is applicable authority to do so. Within a DMMP, environmental projects that are consistent with the federal standard, have significant stakeholder interest and/or a willing non-federal cost share partner (if required), and that provide greater monetary and/or ecosystem benefits may be given preference over those that do not.

1.2 Objective

The purpose of this manual is to provide a comprehensive source of regional guidance for evaluating the environmental suitability of dredged material for a range of beneficial placement alternatives using risk-based principles to comply with applicable environmental requirements. The evaluation framework and testing approaches are designed to determine suitability of dredged material for beneficial use options in upland and aquatic settings.

1.3 Approach

This *Great Lakes Beneficial Use Testing Manual* expands upon the environmental evaluation protocols outlined in previous guidance documents by proposing a holistic, risk-based approach that considers environmental suitability based on potentially impacted environments (aquatic or upland), jurisdictional authorities (federal or state), receptors at risk (human or ecological), and pathways of exposure (water, sediment, soil, air, biota, etc.). The *Great Lakes Beneficial Use Testing Manual* draws from the existing testing manuals to the extent possible for beneficial use assessments to avoid unnecessary additional testing or duplication of testing. In this manual, environmental suitability is defined as meeting current ecological and human health protection requirements at both the federal and state levels, based on chemical and biological assessments, and as meeting physical requirements for the beneficial use proposed. The *Great Lakes Beneficial Use Testing Manual* incorporates testing and evaluation guidance from the USACE, USEPA, and state resource/regulatory agencies to provide descriptions of placement options and restrictions. This will ensure that managers of beneficial use projects better understand dredged material suitability.

1.4 Scope

This manual is applicable to sediments dredged from any new or existing projects in the Great Lakes, including but not limited to all navigation channels, harbors and ports operated and maintained and/or permitted by

USACE in Lakes Superior, Michigan, Huron, Erie, and Ontario, including tributary watersheds. This manual is not regulatory in nature. This manual is designed to provide guidance for evaluating dredged material removed from any Great Lake by USACE permit to determine lawful placement for beneficial use.

1.4.1 What this manual is intended to address

This manual offers guidance on beneficial use applications of dredged material. This manual focuses primarily on contaminant-related impacts of beneficially used dredged material and covers the following:

- Evaluation methods of dredged sediment characteristics and quality to determine management options
- Aquatic placements
- Upland placements
- Suitability classification for dredged material management decisions
- Testing guidance to characterize physical, chemical, and beneficial use performance attributes
- Guidance for interpreting characterization data and relevance to existing regulatory structure
- Treatment and management options to reduce uncertainty associated with potential adverse impacts.

1.4.2 What this manual is not intended to address

This manual does not address the following:

- Economic considerations for beneficial use projects
- Determination of the federal standard
- Impacts at the dredging site associated with the dredging activity itself
- Physical impacts related to construction activities on placement sites
- The potential introduction of invasive species via beneficial use of dredged material, or measures to mitigate the same
- Microbiological impacts from dredged material placement
- Potential impacts from natural mineral deposits
- Mechanisms to rank proposed beneficial use options
- Climate change impacts
- Non-persistent contaminant-related impacts to fish and wildlife resources related to aquatic placement actions
- Actual opportunities for beneficial use

- Legal implications of any specific project or proposed beneficial use
- Funding, real estate, or other project partnership details needed to implement any specific beneficial use project.

1.4.3 Content

This manual comprises eight sections and six appendices.

- Section 1 presents background information on dredged material management approaches and existing guidance that supports evaluations of dredged material for beneficial uses.
- Section 2 provides an overview of environmental statutes and regulations that should be considered when determining the suitability of dredged material for beneficial uses.
- Section 3 describes the broad categories for dredged material placement options. These categories focus the evaluations that should be performed to determine the environmental suitability of the material to be placed in that environment (aquatic or upland). Examples of beneficial use placements from within the Great Lakes basin are provided.
- Section 4 provides an overall approach to the dredged material evaluations by explaining the concepts of a risk-based approach in which the dredged material evaluations are performed. It includes considerations for developing an appropriate sampling and analysis plan to perform these evaluations and offers some advice on interpreting the testing results within the risk-based approach.
- Section 5 provides evaluations specific to aquatic beneficial use placement options.
- Section 6 provides evaluations specific to upland beneficial use placement options.
- Section 7 presents risk management approaches and options that can be used to reduce potential risks associated with exposure to sediment constituents, which may be part of an adaptive strategy to manage uncertainties and/or potential risks associated with beneficial use placements, especially those which may be innovative.
- Section 8 presents a summary and recommendations.
- Appendix A provides sources of statistically established background concentrations around the Great Lakes region.
- Appendix B presents a listing of Great Lakes state guidance and regulations that may be used for upland beneficial use evaluations (Appendix B1) and aquatic beneficial use evaluations (Appendix B2).

- Appendix C contains a draft ERDC technical note with ecological biota screening levels that support the upland beneficial use evaluations.
- Appendix D provides treatment options for impaired sediments.
- Appendix E describes practical considerations for dredged material management.
- Appendix F provides information on interpreting laboratory bioaccumulation test results on dredged sediment proposed for open-water placement.

2 Statutory and Regulatory Overview

There are seven primary or commonly used federal laws (or statutes) that establish statutory authority over decisions about placement, management, and beneficial use of dredged material in the Great Lakes region. This section does not constitute an exhaustive list of all the federal statutes that may apply to a specific beneficial use proposal. An overview of state-specific regulatory programs (and associated guidance) affecting dredged material management is provided in Appendix B.

- Clean Water Act (CWA)
- National Environmental Policy Act (NEPA)
- Endangered Species Act (ESA)
- Resource Conservation and Recovery Act (RCRA)
- Toxic Substances Control Act (TSCA)
- Coastal Zone Management Act (CZMA)
- National Historic Preservation Act (NHPA)

At a minimum, compliance with NEPA is necessary for every federal recommendation involving the beneficial use of dredged sediment. With some limited exceptions, all federal agencies must comply with NEPA before they make decisions about federal actions that could have an effect on the human environment. Thus, NEPA applies to a very wide range of federal actions and decisions that include federal construction projects, plans to manage and develop federally owned lands or water bodies, and federal approvals of non-federal activities such as grant proposals, licenses, and permits. Compliance with NEPA can involve development of an environmental assessment or environmental impact statement (described in Section 2.1.2).

Such federal recommendations must also follow other federal laws, depending on location and circumstance (e.g., CWA for aquatic placement but not necessarily for upland placement). Additional examples include the presence of federally threatened, endangered, or candidate species, or their designated critical habitats, within a proposed project area that would require compliance actions pursuant to the ESA. Additionally, the presence of cultural resources (e.g., architectural and archaeological) that are listed, or eligible for listing, on the National Register of Historic Properties within a project's area of potential effect would require compliance actions pursuant to the NHPA (36 CFR Part 800). While some laws would apply only to a proposed federal action (e.g., NEPA), other laws could apply to actions proposed at the state or local levels. It is not likely that a TSCA review is required for any dredged sediment being considered for beneficial use in the Great Lakes since the threshold for polychlorinated biphenyl (PCB) regulated under TSCA is 50 mg/kg. Material with high PCB concentrations is more appropriately confined for disposal. Regarding the CWA, any proposed placement of dredged sediment at a specified site in a water of the United States for aquatic beneficial use is regulated by USACE according to Section 404(b)(1) Guidelines. The state has a role in this process whereby it issues, denies, or waives Section 401 water quality certification to confirm whether the proposed placement complies with applicable state water quality standards (WQS).

In addition to the federal authorities above, promulgated state laws and regulations may also factor in the determination of a dredged material's suitability for upland beneficial use applications. The laws can vary significantly from state to state and generally do not specifically address dredged material. Most soil or material regulations have been implemented to address issues with solid wastes (e.g., RCRA) or human health risks associated with industrial sites, spills, and brownfields. Current state laws and policies used to evaluate the suitability of dredged material have been subject to extensive revision and interpretation, making compliance with applicable authorities and other regulatory considerations challenging. Additional challenges may exist for beneficial use projects that move dredged sediment between sub-basins or across jurisdictional boundaries, or that create the potential for contaminant or invasive species migration between sub-basins or jurisdictional boundaries. Such instances require coordination with additional federal and state resource agencies that could include, for example, the need for multiple CWA Section 401 water quality certifications, Tribal coordination, and possibly consultation with multiple offices of the same agency (e.g., from two states). While these issues require case-by-case evaluation, a common and consistent approach between Great Lakes states would be helpful for regional conformity. Individual state environmental regulations and guidance for upland beneficial use placement are included in Appendix B.

2.1 Regulatory considerations for aquatic placement

Aquatic placement is the application of dredged sediment in a Water of the United States at a specified site that may be proposed at various water depths (e.g., in nearshore shallow water or offshore deep water). It may also include the placement of dredged sediment into federal jurisdictional wetlands for the creation or enhancement of such aquatic systems where deemed appropriate.

2.1.1 The Clean Water Act (CWA)

As explained in Section 1.2, the ITM (USEPA/USACE, 1998a) provides the driving regulatory guidance for evaluating contaminant-related effects of placement of dredged or fill materials into waters of the United States for *any* reason, including aquatic beneficial use. The ITM and the companion *Great Lakes Testing Manual* (GLTM) specifically address the "contaminant determination" portion of the larger CWA Section 404(b)(1) Evaluation [described in 40 CFR 230.11(d) and requiring an evaluation of all aquatic impacts (e.g., impacts to fisheries, benthos, water quality). Following a determination that the dredged sediment has physical characteristics that might benefit some type of aquatic use, it must then comply with the requirements of the CWA to be considered *suitable* for aquatic beneficial use at a particular location. Section 404 of the CWA requires permits to be issued by USACE (or by the state in the cases of Michigan, except for USACE civil works projects) for any discharge of dredged or fill material into waters of the United States. This includes any part of the Great Lakes and their tributary systems and connecting wetlands that are within Section 404 jurisdiction. Jurisdictional activities may also cover direct discharges of effluent and runoff from dredged sediment (i.e., *return water*) into jurisdictional waters. In the case of a Section 404 discharge, the CWA also requires the issuance, denial, or waiver of Section 401 water quality certification from the applicable state(s). The Section 401 water quality certification process is intended to ensure that the proposed discharge complies with all federally-approved and applicable state WQS.

2.1.2 The National Environmental Policy Act (NEPA) and Endangered Species Act (ESA)

The NEPA applies to a wide range of federal actions. This and several other federal statutes require that federal agencies consider the environmental impacts of their proposed action(s) on the human environment by considering and evaluating reasonable alternatives, including "no-action." Specifically, NEPA requires federal agencies to consider environmental effects on various public interest factors that include, among others, impacts on social, cultural and economic resources as well as natural resources. Compliance with NEPA is applicable to a wide range of federal actions that include federally-funded construction projects, plans to manage and develop federally owned lands or water bodies, and federal approvals of non-federal activities (e.g., grant proposals, licenses, and permits). An environmental assessment is a NEPA document that a federal agency prepares to document its determination as to whether a proposed federal action is likely to have a significant effect on the quality of the human environment. An environmental impact statement is a more detailed NEPA document that a federal agency prepares for the same purpose, but it is for federal actions that are judged to have a *significant* effect on the quality of the human environment. Each federal agency has adopted its own NEPA implementing regulations and/or procedures. Where more than one federal agency is involved in a proposed project or decision requiring NEPA compliance, however, only one of the agencies is determined to be the lead federal agency.

With respect to the ESA, compliance is dependent on whether federally threatened, endangered, or candidate species, or their designated critical habitats, are present within a proposed project's potential area of direct and indirect effects. Information pertaining to the presence or absence of such species or habitat is available at the Information for Planning & Consultation website maintained by the US Fish and Wildlife Service. If any threatened, endangered or candidate species or their critical habitat are present or potentially present, coordination with the local US Fish and Wildlife Service field office is recommended to ensure that any relevant concerns in this regard are considered and addressed during a project's planning process. Compliance with the ESA is required whether a beneficial use proposal is to be implemented at the federal, state, or local levels.

2.1.3 Coastal Zone Management Act (CZMA)

The Coastal Zone Management Act (CZMA) established the Coastal Zone Management Program (CZMP), which ensures that any federal actions that affect land or water uses or the natural resources of a state's or territory's coastal zone must be consistent, to the maximum extent practicable, with the established policies of that state's or territory's federally approved CZMP. All Great Lakes states have established a Coastal Management Program. Within each state, the Coastal Management Program is a single coordinated program integrating an array of state laws and policies that have some bearing on coastal resources. The CZMA gives states broad flexibility in establishing their coastal programs, but all state programs must aim to reduce erosion and coastal hazards, preserve maritime and cultural heritage, support coastal dependent uses, create and enhance public access, balance coastal community development, and protect and restore coastal habitats, including wetlands. While this broad flexibility can be used as a mechanism to promote beneficial use of dredged sediment, it can also be used to oppose federal dredged sediment management decisions if they are believed to be inconsistent with a state's Coastal Management Program.

2.2 Regulatory considerations for upland placement

Upland placement is the application of dredged sediment in an upland setting (i.e., above the elevation of the ordinary high water mark in a water of the United States, including wetlands).

In the past few years, two Great Lakes states (New York and Ohio) have developed regulatory programs specifically for upland beneficial use of dredged material. An overview of these state-specific regulatory programs is provided in Appendix B.

In the absence of state-specific requirements specific to dredged material, the following regulations should be considered for determining the suitability of dredged material for upland beneficial use (state-specific considerations under RCRA are mentioned in Section 2.2.4).

2.2.1 The CWA

Alternatives to aquatic placement may include placement into a CDF or other location, including unconfined upland placement. Typically, this type of placement involves dewatering of the dredged sediment, which can then result in a discharge of the associated effluent (return water) into a Water of the United States. Such discharges are regulated under Sections 404 and 401 of the CWA. Although not a formal guidance document under the CWA, the UTM (USACE 2003), describes the authority of the CWA and the manner in which it applies to the placement of dredged sediment in upland environments where the discharge of dredged or fill material (or discharge of associated effluent) may potentially impacts waters of the United States (USACE 2003). In such cases, the ITM/GLTM provide the formal CWA guidance for evaluating contaminant-related impacts associated with discharge of the effluent. However, unconfined or other upland placement options may also trigger different regulatory authorities, such as construction stormwater/erosion control requirements administered by the states under Section 402 of the CWA, or groundwater protection requirements. Proposals exist for removing dredged sediment from CDFs and using it to construct wetlands for water runoff, water treatment, or wetland restoration or reclamation purposes in mining areas. Such proposals would require careful review of the jurisdictional authorities governing such use and would need to be evaluated accordingly.

2.2.2 The NEPA and ESA

For upland placement scenarios, consideration of NEPA and the ESA would use an approach similar to those outlined for aquatic placement (Section 2.1.2).

2.2.3 The CZMA

The CZMA may also apply to placement of dredged material in an upland environment because designated coastal zone management areas *may* extend up to several miles inland of a lake, shoreline, or major tributary. If the placement site is within the designated CZMP boundary, then the project may need approval to comply with the applicable management policies of the jurisdictional state.

2.2.4 Resource Conservation and Recovery Act (RCRA)

The following text was taken directly from the UTM found at http://el.erdc.usace.army.mil/dots/guidance.html:

> One of the purposes of RCRA is to ensure that generated waste "should be treated, stored, or disposed of so as to minimize the present and future threat to human health and the environment." Since April 1988, with publication of the USACE maintenance dredging and disposal regulations at 33 CFR 335–338, the USACE has asserted that dredged material is not a hazardous waste and

> should not be regulated under RCRA (Federal Register Vol 53, No. 80, April 28, 1988, pages 14903 and 14910). Throughout the 1990's [sic], the USACE made a concerted effort to demonstrate that the CWA/MPRSA [Marine Protection, Research, and Sanctuaries Act] protocols provided a level of environmental protection commensurate with that accorded under RCRA. Based on that demonstrated experience, the USEPA excluded dredged material as a hazardous waste on 30 November 1998, providing the dredged material is regulated undereither the CWA or MPRSA (Federal Register Vol 63, No. 229, November 30, 1998). The effective rule date was 1 June 1999. Specifically, 40 CFR 261.4 of that rule provides that dredged material regulated under "a permit that has been issued under Section 404 of the Federal Water Pollution Control Act (33 USC. 1344) or Section 103 of the Marine Protection, Research, and Sanctuaries Act of 1972 (33 USC. 1413) is not a hazardous waste." The term permit also applies to congressionally authorized Civil Works projects undertaken by the USACE using the CWA or MPRSA regulatory regimes.
>
> The RCRA exclusion for dredged material only applies to activities permitted under either the MPRSA or CWA. Since CDFs would not typically be located in ocean waters, the protocols of the CWA Guidelines are used in this manual. The link between the RCRA rule exclusion and CDFs rests with the CWA Section 404 permit required for the effluent discharges from the CDF. Although that discharge is permitted nationwide at 33 CFR 330.5, the nationwide permit does not authorize the disposal of contaminated dredged material into a CDF where there is potential contaminant release to the environment. (USACE 2003)

While dredged material is exempt from regulation as solid waste, under RCRA that exemption only applies when it is subject to a permit issued under Section 103 of the MPRSA or Section 404 of the CWA (63 FR 65874, 65921: Nov 30, 1998). If dredged material is placed outside the jurisdiction of the CWA, such as for industrial fill in an upland site, state authority under solid waste may apply.

The recovery or expansion of CDF capacity is being sought in some Great Lakes harbors. There are proposals to remove suitable dredged material from existing CDFs for use as fill or topsoil in residential, industrial, or commercial applications. Removal of the sediment to a location where its application would be outside the authority of the CWA could trigger the authority of RCRA.

Application of RCRA would be according to the promulgated laws and policies of each state under either its solid waste program or other authorities for contaminants in soils. These vary significantly by state and will not be discussed in full here. A complete summary by state is provided in Appendix B. Note that when considering the testing needed to ascertain the suitability of dredged material for beneficial use under state regulatory requirements, most standards are based on (1) protecting groundwater, (2) protecting drinking water, or (3) direct human contact. State regulations applied to beneficial use of dredged material may be based on (1) use-specific, (2) general reuse, (3) soil cleanup guidance, or (4) environmental review.

2.2.5 Other possible regulatory considerations for upland and aquatic placement

2.2.5.1 Clean Air Act

Note that the Clean Air Act and/or the Occupational Safety and Health Administration air quality standards may need to be considered in any beneficial use project near human populations, although it is unlikely that concentrations in Great Lakes navigation dredged material would reach the threshold to be regulated under these acts. The potential risks from worker exposure to dredged sediment may be evaluated using the USEPA risk assessment guidance for Superfund framework, which is discussed in Section 6.2.1. It is highly unlikely that any dredged material proposed for beneficial use would contain elevated concentrations of regulated organics. However, even low concentrations near detection limits or below regulated levels may be sensed by humans and can be a concern to owners of property near a beneficial use site. The same can be said of innocuous yet offensive odors often emitted by anaerobic or recently excavated dredged material (e.g., swampy and earthy odors). Another issue may be fugitive dust generated from dry fine-grained dredged material if located near homes or businesses. Particulates are regulated under the Clean Air Act, and in non-attainment areas, a visible dust cloud may create a situation of non-compliance. In addition, dust may contain metals that would represent a non-carcinogenic health risk.

2.2.5.2 Federal Surface Mining Control and Reclamation Act (SMCRA)

The federal Surface Mining Control and Reclamation Act (SMCRA) may also apply for dredged material beneficially used for mine reclamation (see Section 3.3.2). Reclamation plans require approval by state or federal authorities consistent with SMCRA, including leachate tests, and therefore unique considerations may be necessary for the use of dredged material in this way. The SMCRA is regulated by the US Department of Interior – Office of Surface Mining, with transfer to state regulatory authorities under approved programs consistent with SMCRA. Under SMCRA, reclamation actions are tied to intended land uses and must demonstrate capacity to achieve utility on par with what was capable prior to mining. Requirements are performance-based and set minimum levels of environmental protection appropriate to specific climate, geology, geography, and other site conditions.

The specific regulatory guidance, how it is derived, and what it is trying to protect (environmental resource and/or human health) is useful information in gathering the most appropriate data needed for state review. Also, while a dredged material may meet the suitability standards for beneficial use under state law, there may be other regulated exposure or environmental impact concerns not currently addressed by state regulatory structure alone. These are discussed in Section 4, which outlines a risk-based approach to beneficial use dredged material evaluations.

2.2.5.3 National Flood Insurance Program

Within the context of beneficial use, this program primarily applies to the application of dredged sediment within the 100 yr floodplain of a coastline, lake, or stream mapped by the Federal Emergency Management Agency. Unique restrictions and/or mitigation may apply to work within an established floodplain depending on the effect of the dredged sediment on flood levels and flow regimes. Coordination with the Federal Emergency Management Agency and possibly state resource agencies is required in such circumstances.

2.2.5.4 Natural Resources Damage Assessment and Restoration

Under the Comprehensive Environmental Response, Compensation, and Liability Act and the National Oil and Hazardous Substances Pollution

Control Act (USEPA 1990), polluters are liable for the restoration of natural resources damaged by their release of hazardous substances. This restoration occurs via a Natural Resources Damage Assessment and Restoration process, in which the Department of the Interior and the National Oceanic and Atmospheric Administration serve as federal agencies trusted with restoring the natural resource. The Natural Resources Damage Assessment and Restoration process has been used in Great Lakes AOCs (Section 1.3) by incorporating the beneficial use of dredged sediment into habitat restoration projects.

3 Beneficial Use Categories

The term *beneficial use* is applied to a wide range of sediment placement alternatives, including aquatic (in-water, nearshore, and wetland) and upland placement of the sediment as is or as amended with a variety of additives. Because there are so many possible options for beneficial use, it is helpful to group the options into categories. However, there is no single agreed-upon classification system for beneficial use.

Childs (2015) proposed 12 categories for dredged material management, with many of these categories applicable within the Great Lakes (as opposed to a marine environment). The USACE ERDC has a list of beneficial uses on its website (https://budm.el.erdc.dren.mil/beneficialuse.html) with the categories arranged by use or endpoint. USEPA and USACE proposed a list of seven main categories in their 2007 Beneficial Use Planning Manual (EPA842-B-07-001) (USEPA/USACE 2007a). Many more categorization methods are proposed in the literature, with most categories representing the source or end use of the materials.

For this manual, a simplified approach is proposed. Six categories representing upland and aquatic uses are discussed in this section along with the regulatory and statutory processes for each of these categories. Sections 4 through 6 offer specific testing and evaluation guidance.

Beneficial use alternatives should be identified early in dredged material management planning. Stakeholders should be consulted to identify viable options since the testing and evaluation will depend on this initial decision. Early identification of alternatives allows for maximum partnership opportunities, including meeting sustainable use goals, alternative funding, and matching up diverse programs to achieve multiple end goals. For example, trying to use navigational channel sediment for beneficial use such as habitat creation is an attempt to pair two programs with different timelines. The habitat project will require NEPA and a feasibility study, plus regulatory coordination, design, and possibly a project funding agreement. Meanwhile, a *regular* navigational maintenance project may occur on a periodic basis, using an existing contracting vehicle that leaves little time and flexibility. Advance coordination is needed to sync the two programs and match all partners' needs.

3.1 Beneficial use alternatives

Dredged material management is a term used to comprehensively describe the disposition of sediment following dredging and conveyance. Dredged material management can include activities associated with placement, processing and handling the sediment for beneficial use, treatment for reducing environmental risks, screening or physically manipulating the material, adding amendments for enhancing the properties of the material, or other actions associated with the management of the dredged material. Dredged material management also includes confined disposal and the actions needed to support that alternative, although that topic is not discussed further here.

Dredged sediment management alternatives are categorized by combining the terminology of the location of placement with the purpose of placement. Consistent use of these terms will assist tracking of dredged sediment on a national scale and help identify trends and potential areas for research and development to increase use of sediment as a resource. The categories/terms that are used in this document to describe beneficial uses are the following:

1. Aquatic placement
   a. Habitat creation (including wetland, nearshore/littoral, island, reefs, or other *wet* habitat).
   b. Shore protection (including beach nourishment, littoral placement, berm construction. For these projects, habitat creation is not an expected outcome, and the material placement is more for structural purposes).
   c. Capping/remediation (in water at contaminated sediment sites to provide clean material for isolation, stability, and reduction in exposure concentration and contaminant flux).
2. Upland placement
   a. Habitat development (nature preserves, habitat restoration, soil conditioning, and manufactured topsoil used for land restoration or agricultural purposes).
   b. Upland fill for development (residential or commercial sites, brownfield redevelopment, mine-land reclamation, roadbed and embankment construction, cover/cap materials).
   c. Manufactured products (commercial topsoil, bricks, concrete, gunnite as a decorative or structural product that is highly processed).

These categories recognize the type of regulatory framework (in-water versus upland) as well as the type of possible exposures to pollutants (ecological versus human). In most cases, materials may be amended or not, such as in the case of manufactured soils. The amendment may change the exposure pathways and would need to be considered in any testing and regulatory coordination but does not determine the beneficial use category.

3.2 Aquatic placement categories

All proposed discharges (placement) of dredged or fill materials into Waters of the United States at specified sites require compliance with Sections 404 and 401 of the CWA, NEPA, and the CZMA. In determining the environmental suitability for aquatic beneficial use, the CWA 404(b)(1) Guidelines (CFR Title 40 volume 26 part 230) provide the substantive environmental criteria for evaluating CWA regulatory compliance of proposed dredged sediment discharges. For determining contaminant-related impacts associated with such discharges, the ITM (USEPA/USACE 1998a) and the GLTM (USEPA/USACE 1998b) provide the primary testing and evaluation guidance. If the dredged sediment discharge fails to meet the stipulations of the ITM and GLTM, it is determined to not be suitable for aquatic placement without suitable operational or engineering controls (e.g., capping). Such controls must be accompanied by long-term monitoring and maintenance plans and safeguarded by institutional controls. Section 7 of this manual also provides information on potential risk management and performance enhancement options for cases that fail to meet the guidance. Aquatic beneficial use projects can be divided into the three main categories below, consistent with the Engineering With Nature principles (Section 1.1). Each of these categories may require project-specific evaluations to assess not only the potential environmental effects of the beneficial use project implementation but also the suitability of the dredged sediment to meet the project goals and provide the performance required.

3.2.1 Aquatic habitat creation

Aquatic habitat creation involves dredged sediment placed for habitat nourishment or creation, including a range of options related to the water depth or configuration of the placement. Wetlands can be created or restored along shoreline areas, where many coastal wetlands have eroded or been destroyed due to human activities. Benefits include storm protection and/or habitat enhancement, such as bottomland hardwood, swamp, wooded wetland, scrub-shrub, and forested wetland; these habitat areas can support a wide range of native plants, migratory waterfowl, and other highly desired species.

Aquatic berms, fill, and mounds placed at sufficient depth to support submerged vegetation can be stabilized with established submerged aquatic vegetation, either by natural colonization or planting. Similarly, dredged sediment can be placed in the littoral zone to raise bottom elevation to support emergent vegetation. Establishing native vegetated areas increases habitat value for aquatic species, such as fish and benthic organisms, and herbivores, including waterfowl and some mammalian species.

Appropriate placement of dredged sediment can improve ecological functions of fish habitat. Bottom relief created by mounding of dredged sediment may diversify habitat or provide refuge habitat for fish; however, many Great Lakes fish require rocky or coarse-grained areas for spawning. Fine-grained sediment transport can be stabilized by capping with coarse dredged sediment, with the added benefit of providing fish habitat.

A regional example of habitat creation can be found in Duluth-Superior harbor (Section 1.3).

3.2.2 Shore protection

Shore protection involves placement of dredged sediments/sediment on or along the shoreline (coastal and inland), including feeder berms, as an intentional build-up of the land or coast, mainly for erosion protection. This option includes dredged sediment placed directly onto the shore for a beneficial purpose or placed nearshore in the littoral zone with the intent that the majority of dredged sediment will remain within the depth of closure or littoral zone. Sometimes sediment is used a backfill for an area armored with stone or sheetpile, or the entire berm or fill area can be created from sediment alone. In general, preserving the nearshore sediment resources is highly beneficial to shoreline stability and is often a preferred beneficial placement location (USACE IWR 2020).

The influences of waves and seiches keep beach material in continuous motion. Where the prevailing wave direction is at an angle to the beach of less than 90°, some material will move along the beach or foreshore or even offshore in a process called *littoral transport*. This movement is most rapid under storm conditions. If the transported material is not replenished on a continual basis, the beach, and eventually the shoreline, will erode. If lost beach material is not replenished naturally, beach nourishment may be necessary to enhance the beach profile and moderate the wave climate at the shoreline. In addition to the improvement of beaches for coastal protection, improvement may also be required for recreational beaches. Recreational beaches may be improved, or new beaches may be created.

Rivers flowing into the Great Lakes supply a continuous source of sediment, including sand, although in many watersheds, the sediments are dominated by fine-grained silts and clays. Under average flow conditions, much of that sand may not reach the littoral zone, settling instead in dredged navigation channels inland from the lakeshore. Dredging and placement of sandy sediment at strategic locations within the littoral zone can provide the natural separation and transport of sand along beach areas, or if the sediment meets the criteria for beach sand, it can be placed directly on the beach.

The required evaluation to determine the suitability of dredged sediment for beach nourishment is generally based on sand percentage, with a minimum of 80% to 90% of the material meeting the sand classification. If no special contaminant concerns are believed to be present (such as PCBs or dioxins), meeting the sand percentage requirement often exempts the dredged sediment from further testing. However, site-specific conditions, individual state or local regulations, or the potential for human health concerns may result in prohibitions or restrictions on using dredged sediment for beach nourishment.

Dredged sediment may be used to create offshore berms or embankments to modify shoreline wave climate and thus improve beach stability. Berms may also be designed to alter wave direction and modify the rate or direction of local sediment transport. Generally, berms are aligned roughly parallel to the beach, but the optimum alignment at a specific site is determined by the direction of the most destructive wave climate. The formation of berms may provide a particularly attractive use for a wide range of dredged sediment. Because berms are submerged structures, most or all of the structure can usually be created by the bottom discharge of dredged sediment from hopper barges. Berms may gradually erode and be dispersed, but the dispersed material will typically benefit the local coastal regime, either through beach feeding or by increasing foreshore gradients. Modification of the wave climate by berms may also improve recreational opportunities for surfing, swimming, sailing, and other activities. Care must be taken to avoid interference with other users such as fisheries, ports, harbors, outfalls, and intakes.

An example of shoreline protection in the Great Lakes is the Cat Island project in Green Bay. Constructed near the location of a historical island chain that had eroded, this armored backbone is designed to act as a *semi-confined* aquatic placement site. Armor stone blocks waves from the lakeward side of the chain while the placement area on the shoreside of the structures is open to wave action to distribute sediment naturally. The project constructed in 2013 has been very successful as an inexpensive placement site while providing coastal erosion protection. Although designed for coastal protection, the islands also provide bird habitat even early in the life cycle of the project. (Port of Green Bay 2016).

Beach nourishment and littoral zone placement for beach nourishment are also widely practiced in the Great Lakes, including for Waukegan, Burns (Port of Indiana), Michigan City, St. Joseph, and other harbors around Lake Michigan, where an abundance of sand lends itself to this treatment.

3.2.3 Capping/remediation or confined aquatic placement for beneficial use purpose

Capping isolates elevated residual pollutants, especially when it is not physically or economically possible to remove *all* contaminated materials. This can be accomplished via aquatic placement on a targeted footprint, such as a nearshore fill zone or defined contaminated area that requires capping as a remedial action to control exposure pathways.

Aquatic capping has been used to remove Beneficial Use Impairments at several of the AOCs around the Great Lakes. Often this material is placed to provide a lower sediment surface concentration after the bulk of contaminated material has been removed or to cover legacy sediment deposited prior to regulatory controls.

3.3 Upland placement categories

There is no direct permitting process administered under federal authority for use of dredged material in an upland environment. The permitting authority granted to USACE under the CWA does not extend into upland environments unless federal jurisdictional Waters of the United States are directly impacted. USEPA also lacks direct permitting authority for upland beneficial uses; however, under the RCRA, USEPA does provide guidance and has delegated permitting authority to states under their solid waste programs to regulate the reuse of waste materials for beneficial purposes.

While the CWA categorically excludes dredged material from regulation as solid waste when the dredging activities are covered under that program, dredged material managed or used outside the jurisdiction of the USACE CWA authority may be regulated as solid waste under state authority. Many states have adapted this authority in reviewing the upland beneficial use of dredged material and have established permitting processes and standards for various applications. These vary from state to state and continue to evolve since increased consideration of beneficial use has become necessary to ensure future navigation dredging and the utility of the material is recognized. State regulatory processes are discussed in more detail in Section 6 and are provided for each state in Appendix B. Beneficial use projects that may be regulated under different categories of use are described below.

3.3.1 Upland habitat development

Upland habitat development involves dredged material placed for upland habitat, such as upland forest, dune, prairie, or bird habitat, or in the creation of wetland areas outside of the coastal zone. The main concerns from a regulatory perspective are ecological exposures to contaminants. Some ecological receptors may be more sensitive to low levels of contaminants below thresholds established for human health. An evaluation of ecological receptor sensitivity should be performed in conjunction with relevant state and federal agencies (e.g., US Fish and Wildlife Service). Where specific vegetative establishment and/or species support is an objective, specific plant establishment and food web exposure testing may be necessary to evaluate suitability for success and to evaluate potential risks of any dredged material contaminants to flora and fauna not otherwise addressed in soil standards for human exposure risks .

The conversion of former CDFs to nature preserve, such as the Cleveland Lakefront Nature Preserve (formerly CDF Dike 14), is one example of this (Port of Cleveland 2016).

3.3.2 Upland general fill

Upland fill involves dredged material placement for to raise low-lying areas or develop land, including commercial, brownfield redevelopment, agriculture, infrastructure (roads, embankments), and parks and recreation lands. Sediment of suitable physical characteristics can be used as general *satisfactory fill* materials. Dewatering is required prior to use, and in many cases, processing to separate fine- and coarse-grained fractions may also be required

With the general improvement in sediment quality since the promulgation of the CWA in 1972, more and more recently deposited and dredged sediment is suitable for beneficial use. In some cases, suitable materials have been placed into confined facilities due to favorable logistics and economics and available capacity unneeded for unsuitable material. Currently, some of these suitable, confined, and dewatered sediments are being harvested or recovered for beneficial use, and confined disposal facilities are being used as processing and dewatering facility to facilitate upland beneficial use. Dredged material, either from a confined facility or newly dredged, may be used as fill when the physical qualities are superior to soils near the site or where locally sourced clean fill material is in short supply, as in urban areas. In commercial fill sites, peat and clay-type soils are usually removed and replaced with sand or other granular dredged material to improve physical properties needed to meet building requirements. Weak soils may be replaced with sand where tunnels, bridges, fairways, and ports are constructed. Fine-grained soils do not typically have the necessary physical properties for structural fill in most civil works projects; however, green areas or parks may be suitable applications, as would general embankment areas alongside. Some examples of sediment used as fill include the following:

- Surface mine reclamation
- Structural foundations
- Land creation and port development.

Where the quality of existing land is not adequate for a planned use such as where land is exposed to occasional flooding, dredged material can be used to build up the land or to provide a cover layer. Proven methods for land improvement include filling with fine material, such as silts and clays, produced by maintenance dredging. Various dewatering techniques, such as subdividing the placement area to allow filling to a limited depth on a rotational basis, reworking the filled area with low ground-pressure agricultural or earth-moving equipment, and mixing coarse-grained material with the fine-grained upper layer, may be used to condition soil. Soil amendments may be added to increase nutrient content or other qualities such as strength.

The suitability of dredged material for the purposes above is dependent on the dredged material's physical characteristics and the fill setting. Upland use of dredged material may be prohibited or restricted based on guidance promulgated under state laws regarding contaminant concentrations. Generally, higher concentrations of contaminants are allowed for fill applications than for topsoil, particularly when used for industrial/commercial applications, except where impacts to groundwater may be a concern. A description of the testing protocols for evaluating fill suitability are detailed in Sections 4 and 6.

Dredged material may be suitable for mineland reclamation if the characteristics of the material match the land use goals of the mined site, which may include preventing acid mine drainage, revegetation, agriculture, and/or wildlife habitat. Moreover, site- and state-specific issues may apply, as reclamation plans require approval by state or federal authorities consistent with the Surface Mining Control and Reclamation Act (see also Section 2.2.5). Although these considerations must be taken into account, dredged material otherwise deemed suitable for upland use will often provide additional human health and environmental benefits by allaying existing risks associated with mine spoils. This has been the case for other types of impacted materials beneficially used for mineland reclamation (e.g., Vories 2003).

An example of sediment mining from a CDF with beneficial use of the recovered material can be found in the Erie Pier Sediment Management Facility in Duluth. The Erie Pier facility receives a mixture of sand and fine-grained sediment. The material is sorted using a washing process during offloading. The sand is stockpiled, tested, and has been used for mine reclamation and as general fill (Duluth-Superior Metropolitan Interstate Council 2007).

3.3.3 Manufactured soils

Dredged material has been used successfully as topsoil replacement or enhancement at many project locations. The fine-grained dredged materials in many Great Lakes watersheds have been shown to reflect the soil qualities indigenous of the watershed itself. While most dredged material can be used directly to support plant growth in a residential setting, it is often advantageous to enhance its quality with amendments (such as composted vegetation) to improve the soil's physical and chemical properties. Demonstration projects in the Great Lakes have shown added materials, such as yard wastes, can enhance dredged material into a more versatile soil product similar to commercially available bagged soil. Extensive testing at the ERDC has shown blends of dredged material and waste materials can produce a topsoil product that can outperform many bagged soil products. Dredged material used in processed topsoil may be evaluated differently than dredged material applied directly for topsoil use. Topsoil for residential use must meet specific standards established by each jurisdictional state permitting authority. In most cases, promulgated standards allowing the use of dredged material for residential topsoil do not restrict such use from including producing edible crops, such as those grown in residential gardens. However, the potential bioaccumulation of metals, common in most fine-grained dredged materials in the Great Lakes, is a likely outcome that should be carefully considered. Most state standards establishing contaminant limits in reuse of waste materials, including dredged material, are typically based on human health risks associated with contaminant exposure through consuming groundwater or by direct soil contact. Exposure through consuming homegrown produce exposed to dredged material contaminants is not an exposure route for which standards have been developed in most states (one exception being New York State). In some cases, use of mixed materials (vegetative waste plus sediment) may be regulated as a composting action and must comply with the procedures and requirements of the jurisdictional state (http://www.recycle.cc/compostregs.htm). Therefore, suitability of any dredged material including any additives for producing edible crops may require site-specific evaluation to fully understand the potential risks. Note that manufactured soils may be used for habitat purposes or for human land-use purposes. Although this sub-section is placed in the "human uses" category, the ecological considerations from the preceding section may also need to be considered.

Outside of Chicago, more than 100,000 tons of dredged sediments were placed as topsoil on a former steel mill site, which was subsequently redeveloped into park land. Peoria Lake sediment was hauled in 68 barges to the placement location where it was dried in place. Sediment was placed with no amendment and was shown to be more than adequate for use as topsoil. In this case the dredged materials originated from the erosion of farm areas. Information on the site and many other are available on-line (https://www.ideals.illinois.edu/handle/2142/99159). The Illinois Department of Natural Resources operates a "Mud to Parks" program that has successfully used dredged material for several projects. Information on the program is available here: https://www2.illinois.gov/dnr/conservation/m2p/Pages/default.aspx.

Sediment can also be used as the base for manufactured soils and especially manufactured topsoils. A commercial soil dealer cooperates with the St. Paul Port Authority to use USACE dredged sand to produce >100,000 cy topsoil annually. A similar operation was conducted on a smaller municipal scale in Grand Haven, MI, where sediment was mixed with composted leaf waste by the city, which then provided the sediment to local residents. (See https://cdn2.cloud1.cemah.net/wp-content/uploads/sites/38/2016/12/GrandHavencasestudy.pdf.) In general, these manufactured materials are used for commercial landscaping, home landscaping or other *human uses*, although a similar amendment process could be used to create topsoil for habitat restoration. This alternative should be effective in Great Lakes cities with new development and construction.

3.4 Processed products

Dredged material may be used as material to be processed into another product such as sand or gravel for construction materials (concrete, asphalt, bricks, etc.) or landscaping products (blocks, paving stones). Depending on the sediment type and processing requirements, dredged material may be used as concrete aggregates (sand and gravel); backfill material or when producing bituminous mixtures and mortar (sand); raw material for brick manufacturing (clay with less than 30% sand); ceramics, such as tile (clay); pellets for insulation or lightweight backfill or aggregate (clay); raw material for producing riprap or blocks to protect against erosion for dikes and slopes (rock, mixture); and as a raw material for producing compressed blocks for security walls at military installations and for gated communities and home subdivisions. Dredged material can be blended with recycled materials, such as glass, gypsum, plastic bottles, and automobile interiors, to manufacture statues, figures, garden benches, stepping patio pavers, plant vases, artificial rocks, and water fountains. These products can be used to landscape gardens, backyards, swimming pool environments, areas around monument stones, miniature golf courses, highway rest areas, tourist welcoming centers, zoos, and theme parks (GLDT 2016).

The testing and evaluation for suitability of dredged material for these uses may require testing for physical and chemical properties to determine if the needs for final product formulation can be met. However, other tests necessary to ensure the final product meets required product specifications and environmental/human health-risk standards associated with the product's use are generally the responsibility of the product manufacturer. It is the USACE responsibility to determine the intended use of the dredged material and the final product and to disclose the physical and chemical characteristics normally required to conduct dredging and placement operations. Other testing requirements would be determined on a case-by-case basis.

4 Principles for Beneficial Use of Dredged Material Evaluations

When choosing the preferred alternative for managing dredged material, cost, benefits, engineering feasibility, and environmental concerns must all be considered. Costs include planning and regulatory coordination, contracting, equipment mobilization/demobilization, dredging operations, material transport, placement and finishing operations, monitoring, and maintenance after placement. The engineering feasibility requires information such as the physical characteristics of the dredged sediment, constraints based on property ownership, site infrastructure, usage of areas adjacent to the placement site, dredge equipment availability, transportation costs, access, and other site-related factors. Environmental concerns for a project are related to both short- and long-term risks associated with sediment and water quality using a general risk-based approach as presented in the following flowchart: "Making Sediment Beneficial Use Decisions: A Risk-Based Approach" (page 38). The USACE Engineering With Nature initiative described in Section 1.1 attempts to integrate all of these factors (economic, environmental, and social concerns or benefits) when making dredged material management decisions.

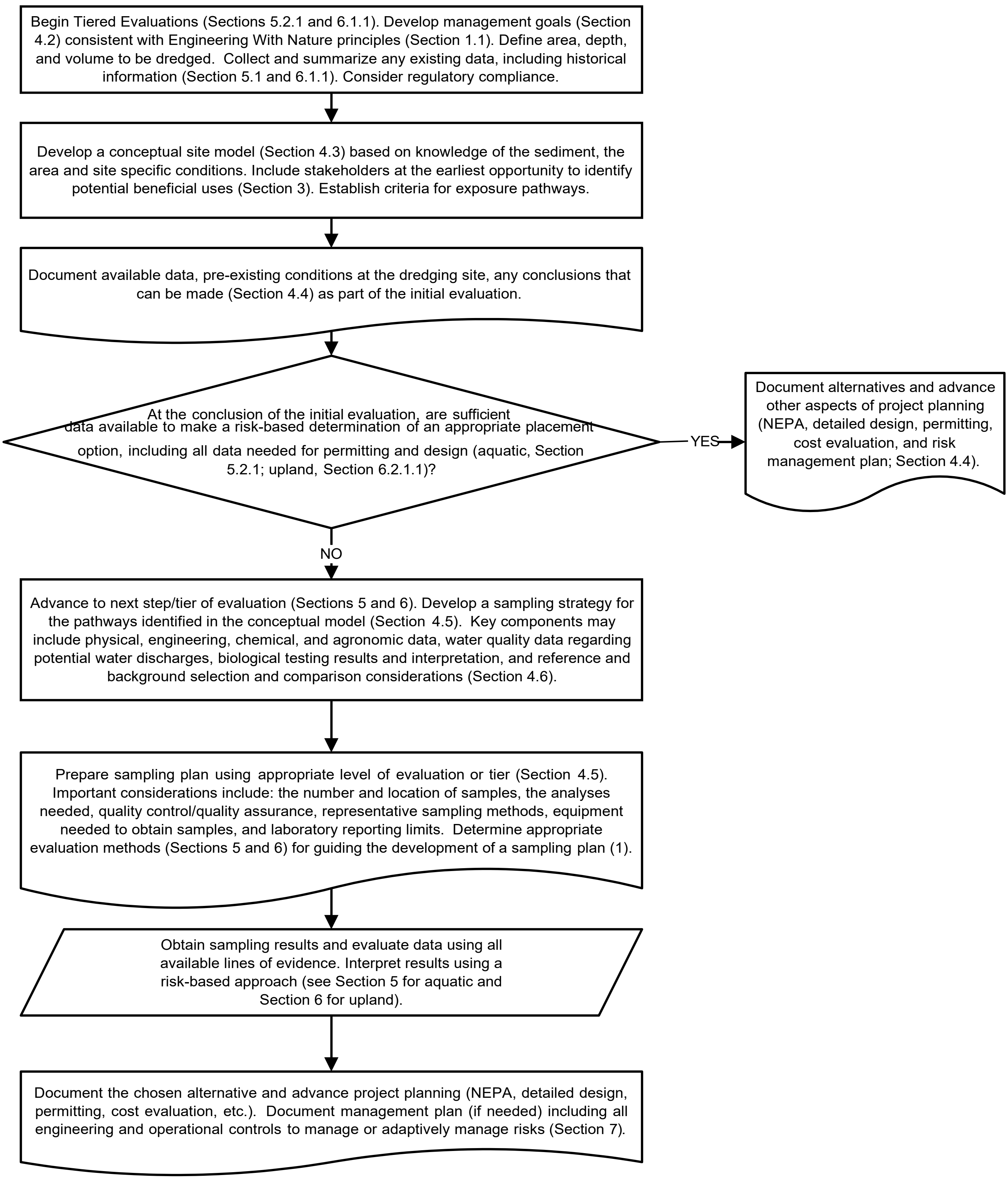
Historically, three general categories for dredged material management have been evaluated under the framework for determining environmentally suitable alternatives: (1) open-water placement; (2) confined disposal (e.g., CDF placement); and (3) beneficial use (USEPA/USACE 2004). For open-water placement of dredged material at a specified site, the primary environmental standard is compliance with CWA Section 404(b)(1) Guidelines, which is determined by USACE. Formal guidance specific to evaluating contaminant-related impacts directed at this proposed discharge of dredged sediment (including any return water from a CDF or upland site) in aquatic environments is prescribed in the ITM (USEPA/USACE 1998a) and the GLTM (USEPA/USACE 1998b) (see Section 5). This discharge must also comply with an array of other relevant and state federal laws and requirements (e.g., applicable state WQS, coastal management program policies). For dredged sediment placed in island, nearshore, or upland CDFs, the evaluation of environmental suitability follows the guidance provided by the UTM (USACE 2003).

Environmental suitability of beneficial use alternatives may be determined by any or all of the guidance summarized in Section 2 in coordination with stakeholders. Additional evaluations not described in any of the preceding guidance may be necessary to characterize the risk and benefits for a proposed management alternative. This document incorporates existing guidelines and guidance developed to comply with CWA and NEPA requirements and includes additional federal guidance necessary to comply with other laws, regulations, standards, criteria, and policies that may apply to certain beneficial use alternatives. The three testing manuals (ITM, GLTM, and UTM) mentioned previously describe the tiered risk-based process used to assess environmental suitability for either aquatic placement or confined (upland or nearshore) placement. The tiered approaches and testing methods described in these manuals are used to assess environmental suitability for beneficial use as well. However, existing guidelines and guidance have not contemplated the many possible beneficial uses for dredged material. The following provides a framework for characterizing the risks associated with beneficial use management plans that are not explicitly considered by existing guidelines and guidance. A crosswalk between the risk-based approach described in this section and the specific tiers applied to the assessment of environmental suitability for aquatic or upland placement (further described in Sections 5 and 6, respectively) is provided in Table 4-1.

Table 4-1. Crosswalk between aquatic and upland placement evaluation tiers and relevant risk-based processes.

<table>
<tr><th rowspan="2">Tier</th><th rowspan="2">Risk-Based Process</th><th colspan="2">Aquatic Pathways</th><th colspan="2">Upland Pathways</th></tr>
<tr><th>Water Column</th><th>Benthic Exposure</th><th>Human Health</th><th>Environmental Health</th></tr>
<tr><td>Tier I</td><td>Development of project goals and conceptual site model to focus pathways being evaluated</td><td colspan="2">Comparison to placement/reference site sediment concentrations</td><td colspan="2">Comparison to placement/reference site soil concentrations</td></tr>
<tr><td>Tier II</td><td>Reliance on chemical analysis of samples, and modeling</td><td>Elutriate chemistry and dispersion/dilution modeling</td><td>Theoretical bioaccumulation potential</td><td>Comparison to generic soil screening levels</td><td>Modeling and/or further chemical analysis</td></tr>
<tr><td>Tier III</td><td>Incorporation of laboratory bioassays and/or additional site-specific exposure assumptions</td><td>Elutriate toxicity tests</td><td>Sediment toxicity tests and bioaccumulation tests</td><td>Site-specific risk-based screening levels and/or modeling or extractions</td><td>Soil toxicity tests
Bioaccumulation tests
Plant growth and uptake tests</td></tr>
<tr><td>Tier IV</td><td>Site-specific evaluations</td><td colspan="2">Site-specific sampling, analysis, and/or evaluations</td><td colspan="2">Site-specific sampling, analysis, and/or evaluations</td></tr>
</table>

Making Sediment Beneficial Use Decisions: A Risk-Based Approach



(1) See Section 8 for list of appropriate guidance and other references. All sediment and water sampling should follow appropriate current technical guidance to ensure completeness and representativeness.

4.1 Key concepts for beneficial use of dredged material evaluations

4.1.1 Project goals and objectives

Clearly stated, consensus-based environmental, social, and economic goals effectively guide project decisions to include planning, design, and permitting that ultimately lead to successful beneficial use projects. Project goals often identified in the Great Lakes include beach nourishment, habitat restoration for fish or shorebirds, wetland creation, storm protection, urban park development, port, landfill covers, or brownfields and contaminated sites remediation (discussed in Section 3). The project goals should lead to quantifiable objectives that allow for the comparison of alternatives and an engineering design that maximizes project benefits for the least cost or given budget. Further discussion of project goals is provided in Section 4.2.

4.1.2 Assess project risks using conceptual site models (CSMs)

Conceptual site models (CSMs) are recommended because they support holistic evaluation and communication of project benefits and risks. Note that different alternatives may require different conceptual site models due to the differences in organisms at risk and the prevalent exposure pathways (see Section 4.3). The development of a CSM is the initial step necessary to determine how dredged material should be evaluated and the information that may be required for the evaluation.

Risk characterization is typically conducted using both an exposure assessment and toxicity assessment. The exposure assessment must consider both spatial and temporal factors contributing to exposure. Simply stated, the contaminant or stressor at a level of concern needs to be collocated with receptors of interest for a duration of concern. The toxicity assessment determines the occurrence of lethal to sub-lethal effects for sensitive organisms at a range of exposures (concentrations and durations) to assess the potential for adverse impacts when considering actual exposures by the project.

The presence of complete exposure pathways (i.e., a receptor is expected to be exposed) will dictate which testing protocols are appropriate to use since the tests described further in Sections 5 and 6 need only be applied when a complete exposure pathway exists. In some cases, an initial evaluation based on physical and chemical characterization (described in

Section 5.1 for aquatic placement and Section 6.1.1 for upland placement) may be all that is needed to document that dredged material is suitable for beneficial uses. Such a risk-based approach—incorporating conceptual site models, exposure pathway analysis, and a tiered evaluation framework—is consistent with recent USEPA general guidance on beneficial use evaluations (USEPA 2016a, 2016b). The characterization of ecological and human health risk using conceptual site models is generally similar to what is commonly performed for USEPA Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) projects; however, the guidance and options for risk characterization under the CWA provides significantly more flexibility than what is prescribed under the USEPA Superfund program.

4.1.3 Measurement endpoints vs. risk assessment endpoints

Standard toxicity and bioaccumulation tests are typically conducted with cultured fish or macroinvertebrates (aquatic or terrestrial) in beakers or aquaria under controlled laboratory conditions. Survival, growth, or reproduction of test organisms can be readily measured in the laboratory and often serve as measurement endpoints for the predicted effects. In contrast, risk assessment endpoints are elements of the environment or ecosystem about which risk-based decisions are made; these are often stated as a desired population density of a species or the diversity of a community. Thus, interpreting results from laboratory tests typically requires extrapolation from laboratory test species to the biota relevant to the site, from laboratory test conditions to environmental conditions in the field, and from lower to higher levels of biological organization (e.g., individual organism level to a population or community level) (USEPA 2016d). Careful consideration of the measurement endpoint (i.e., laboratory test with a measured effect) needs to occur to ensure relevance to the desired risk assessment endpoint. In some cases, it is difficult or impossible to mimic the biogeochemistry of a contaminant found in the field using standard laboratory toxicity and bioaccumulation tests, and caution should be applied in extrapolating laboratory measurement endpoints to risk assessment endpoints. In other cases, it is difficult to extrapolate small, measured changes in a test species growth or reproduction under laboratory conditions to a relevant risk assessment endpoint such as the population density of a species or diversity in a community.

4.1.4 Use multiple lines of evidence for decision-making

Multiple lines of evidence are often used to assess whether exposure to toxic chemicals or other stressors (such as turbidity) associated with dredged sediment placement may result in adverse effects to human health or biota (Bates et al. 2018). Because there is no single test to evaluate all effects of a proposed placement (40 Code of Federal Regulations [CFR] 230.61), lines of evidence can be used during the evaluation process. Lines of evidence are broad categories of information that may include prior evaluations; site conditions; physical, chemical, and/or biological information; presence and numbers of a species; and proposed project actions. These lines of evidence are incorporated into an overall weight-of-evidence approach that informs decision-making on dredged material management options (USEPA 2016e).

4.1.5 Spatial and temporal scales when assessing risk

An exposure assessment estimates the magnitude, frequency, and duration of exposure for an organism and therefore must consider the relevant spatial and temporal scales of exposure. Spatial scale considerations include the size of the affected beneficial use site relative to the home range or foraging range size of a receptor, sensitive sub-populations or metapopulations, migratory and life-history spatial patterns, and plasticity in habitat selection or temporary avoidance. Temporal scale considerations fundamentally assess the timing, duration, and continuity of exposure to a constituent of potential concern (COPC) or stressor, and any anticipated adverse effects from such stressors. The timing, duration and continuity elements are placed within the context of the temporal scales associated with the natural processes for the receptors at the site. Additional temporal considerations include repetitive exposures and the rate at which COPCs may be accumulated in and depurated from tissues.

As an example of assessing interacting spatial and temporal scales, turbidity plumes generated during hydraulic placement of dredged sediment may be short term and of limited spatial scale. The limited duration may be commensurate with the natural variation of turbidity in a waterway and may therefore have no long-term consequences to the risk assessment endpoint of interest. The limited spatial scale of temporary increases in turbidity may be considered acceptable if it occurs within the boundaries of the construction project and sensitive receptors (or life stages) are unlikely to be present. In summary, population, community, and ecosystem functions operate on various spatiotemporal scales, resulting in assessment endpoints with scales that may or may not align with exposure scenarios.

4.1.6 Relative vs. absolute risk

Absolute risk can be considered the probability of a particular outcome occurring whereas *relative* risk is the absolute risk compared to the pre-existing level of risk. All management actions include some risk, including the no-action alternative. The regulations included in Subpart H of 40 CFR 230 allow restoring or creating new habitat to "produce a new or modified environmental state of higher ecological value by displacement of some or all of the existing environmental characteristics" (40 CFR 230.75 [d]). By definition, both the risks and benefits associated with a project must be compared, and decisions regarding the suitability of using sediment for a beneficial use should be made by considering relative risks in addition to absolute risks. One of the key components to evaluating the relative (i.e., comparative) risk of a beneficial use project is the determination of how contaminant concentrations might change and whether exposure is reduced.

PCB bioaccumulation can be used as an illustrative example of how relative risk may be assessed. The use of dredged sediment with low levels of PCBs for a habitat restoration project may result in a net environmental benefit when the bioavailable concentration of PCBs in the dredged sediment is lower than the preexisting bioavailable concentration of PCBs in sediments at the habitat restoration site. In this case, the overall risk of PCB bioaccumulation and subsequent impacts to fish and wildlife is predicted to be lowered at the placement site because of dredged sediment placement. If benthic bioaccumulation presents a meaningful pathway at the placement site, the placement of dredged material over contaminated sediments may serve to reduce PCB bioaccumulation in the aquatic food web.

4.1.7 Understanding the relationship between predicting impact of a project and uncertainty

It is important to differentiate between the estimation of potential impacts to human health or ecosystems and the uncertainty in the estimate of those impacts. The estimated potential for an impact can be considered the risk (i.e., the probability of a given impact occurring). The uncertainty is the confidence associated with that estimate of probability. The decision to evaluate the merits of a beneficial use project relies upon both the magnitude of the potential impacts (positive or negative) and the uncertainty associated with those predictions. The magnitude includes the significance of the outcome for which that risk was estimated (e.g., a 1 in 10 risk of a fish population collapse is considerably more significant than a 1 in 10 risk of an increase in fish tumors or lesions). Uncertainty describes the relative level of confidence in such estimates, integrating data that indicate such an outcome is less probable with those indicating it is more probable to arrive at the point estimate of risk. In some situations, all data and multiple lines of evidence tend to agree, and the uncertainty is low; in other situations, data may indicate differing possible outcomes, and the uncertainty is high. Uncertainty also results from an incomplete understanding of the processes that might produce the predicted outcome and how such an outcome corresponds to the context of the decision endpoints for the project. Estimating uncertainty can be done quantitatively or qualitatively.

The information used to make predictions of negative or positive impacts is always imperfect, and decisions made based on little or imprecise information—which produces greater uncertainty—must be accompanied by greater risk tolerance. For any particular project, the upper and lower uncertainty bounds for potential negative impacts may have a large range (e.g., estimating contaminant concentrations associated with turbidity during storm events), but the overall impact can be low if the actual exposure is low and temporary. The converse is also true where uncertainty is low (e.g., 100% lethality to macroinvertebrates in laboratory tests) and the impact is high (e.g., the benthic macroinvertebrate community being impacted extends over a large area or provides a critical food source for a species of concern). In the text that follows, the authors have used the terms *impact* and *uncertainty* to differentiate between these types of characterizations.

4.1.8 Weighing benefits against uncertainties

Clear and obvious benefits may outweigh short-term negative impacts or minor long-term negative impacts that have a highly uncertain probability. In contrast, a clear and obvious potential impact to human health will certainly outweigh any environmental or economic benefits. In addition to the uncertainty about the occurrence of small negative impacts, there is often uncertainty in how well the selected measurement endpoints represent the target risk endpoint (as described previously). The potential benefits would be considered in the NEPA evaluation required for the project. (The application of NEPA to aquatic and upland dredged material management is explained in Sections 2.1.2 and 2.2.2, respectively.) Weighing the importance of potential benefits vs. potential negative impacts and evaluating the uncertainties are stakeholder-driven processes.

4.1.9 Engineering and operational controls to manage risk

When the risk evaluation concludes that unacceptable adverse effects are likely, engineering and site management controls can often be used to mitigate the potential risk (see Section 7). Management of adverse effects can also occur during construction (Section 7.1.2). Physical and numerical modeling are sometimes performed to assess changes in exposure that result from various operational and engineering controls. If the resulting exposure of target organisms to contaminants or stressors is reduced, the potential for adverse effects may no longer be unacceptable. Sensitivity analysis and management of adverse impacts is consistent with USEPA guidance on beneficial use evaluations (USEPA 2016a, 2016b) and is discussed in Section 7 of this document. In addition, previous projects, professional knowledge, engagement with subject matter experts from other agencies, comparison to nearby reference areas, and pilot projects can be used to develop lines of evidence to assess risk reduction using various risk management alternatives.

4.1.10 Pilot projects and adaptive management

When an unacceptable level of uncertainty is associated with a project, implementing a pilot project or adaptive management process can be useful to reduce this uncertainty (Sections 7.2 and 7.3). Adaptive management strategies and pilot projects are based on monitoring project performance with the intent for modifying the project execution or design to increase benefits and reduce risks. Pilot projects and adaptive management through monitoring allows for small-scale implementation to “learn-by-doing,” while preparing for “corrective actions if adverse impacts occur” (40 CFR 230.75 [d]) (see Section 7.5).

4.2 Development of project goals for beneficial use of dredged material

Project goals are simple statements reflecting the benefits expected from the dredged material placement at a specific location. For example, the project goals will specify the benefits desired such as beach nourishment, habitat restoration, shoreline protection, port construction, or brownfield site remediation. These goals can then be developed into quantitative spatially and temporally explicit objectives. Project goals should reflect the current understanding of the project's technical, engineering, financial and social constraints as well as the expected benefits (e.g., increased recreation, positive externalities). The project goals and objectives are then used to guide any subsequent short- or long-term risk management considerations discussed in Section 7.

These dredged material beneficial use goals and objectives are akin to the ecological risk management goals and problem statement developed at the onset of the USEPA Ecological Risk Assessment paradigm (USEPA 1998b), which are general statements about the desired condition of ecological values of concern at the project site. Those management goals become the cornerstone of subsequent phases of the ecological risk assessment. Similarly, for dredged material beneficial use projects, the project goals and quantitative objectives inform the risk assessment and subsequent risk management options of any potential ecological or human health impacts from the use of dredged material.

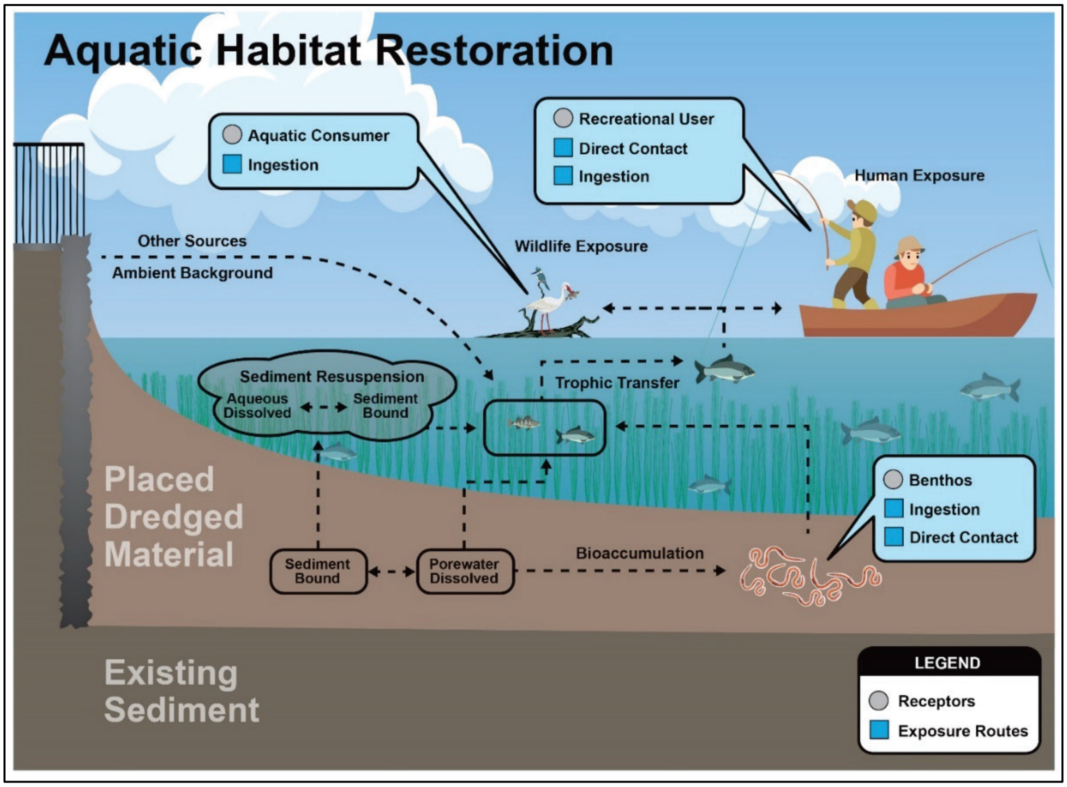
4.3 Development of a CSM

An analysis of the risks and management alternatives for assessing beneficial use begins with constructing a CSM. The CSM describes in writing and/or illustrates visually the exposure pathways and target organisms (receptors) that are of concern. Specifically, the CSM captures the mechanisms by which receptors may be exposed to contaminants or other stressors associated with the dredged sediment. By capturing how this happens, the CSM forms the basis for identifying potentially complete exposure pathways. CSMs are well established in dredging programs and include the exposure pathways for chemical and physical stressors of potential concern (USEPA 1998b; Bridges et al. 2005). Four illustrative CSMs are developed below for typical beneficial use projects in the Great Lakes, which include aquatic habitat restoration, creation of an upland nature preserve (new natural areas), agricultural use, and brownfield site restoration. These CSMs identify the location of dredged material placement, the primary exposure pathways, and typical receptors of concern. A site-specific CSM can be much more complex, with different ecological receptors and exposure pathways that are identified when scoping the project. Although the example CSMs provided are graphical, often CSMs are presented in tabular or other simplified format (Fischenich 2008; USACE 2012).

4.3.1 Aquatic placement

Figure 4-1 provides an illustrative example for the primary exposure pathways and the typical ecological and human receptors that are present during placement of dredged material in an aquatic environment.

Figure 4-1. Generalized conceptual site model for dredging operations at beneficial use aquatic placement sites.



For bottom sediment-dwelling (benthic) macroinvertebrates, the primary risk is toxicity resulting from feeding on and contact with contaminated sediment, sediment porewater, and overlying water. Water column-inhabiting (pelagic) organisms can be exposed through transport of dissolved contaminants from the sediment into the water column and the release of dissolved contaminants into the water column from resuspended sediment. Piscivorous (fish-eating) wildlife and humans can then be exposed to contaminants that have bioaccumulated in aquatic prey species. Finally, people may be exposed by coming into direct contact with the sediment during recreational exposures, although the direct contact pathway likely results in limited exposure.

Note that habitat restoration projects are often located at sites where fine-grained sediments naturally accumulate. Contaminants are often associated with these fine-gained sediments and thus can impact sediment quality over time. In addition, contaminants may be present in the water column arising from upstream sources, stormwater, or combined sewer outfalls, and/or sewage treatment discharges. For bioaccumulative compounds such as PCBs and mercury, the contaminants measured in biota may result from on-going sources to the water column, including atmospheric deposition. The increase in risk resulting from continuing sources can be significant and should be included in the CSM when assessing the relative risk from beneficial use of dredged material. A more detailed discussion on background and ambient contaminant concentrations is provided in Section 4.5.2. Finally, note that bioaccumulation of a COPC in receptors at the placement site occurs not only from benthic prey but also from epibenthic and pelagic prey and the water column and that bioaccumulation in the aquatic environment is influenced by numerous factors in addition to concentrations in prey.

Section 5 provides an overview of the approach used by the USACE to evaluate the suitability of dredged sediment for aquatic beneficial use placement, drawing upon the existing guidance in the ITM/GLTM to address the potentially complete exposure pathways presented in Figure 4-1.

4.3.2 Upland conceptual site models for nature preserves and agricultural sites

Figures 4-2 and 4-3 provide CSMs illustrating the primary exposure pathways and the typical ecological and human receptors that are present during placement of dredged material upland for the creation of new natural areas and enhancement of agricultural sites, respectively.

Figure 4-2. Generalized conceptual site model for dredging operations at beneficial use nature preserves sites.

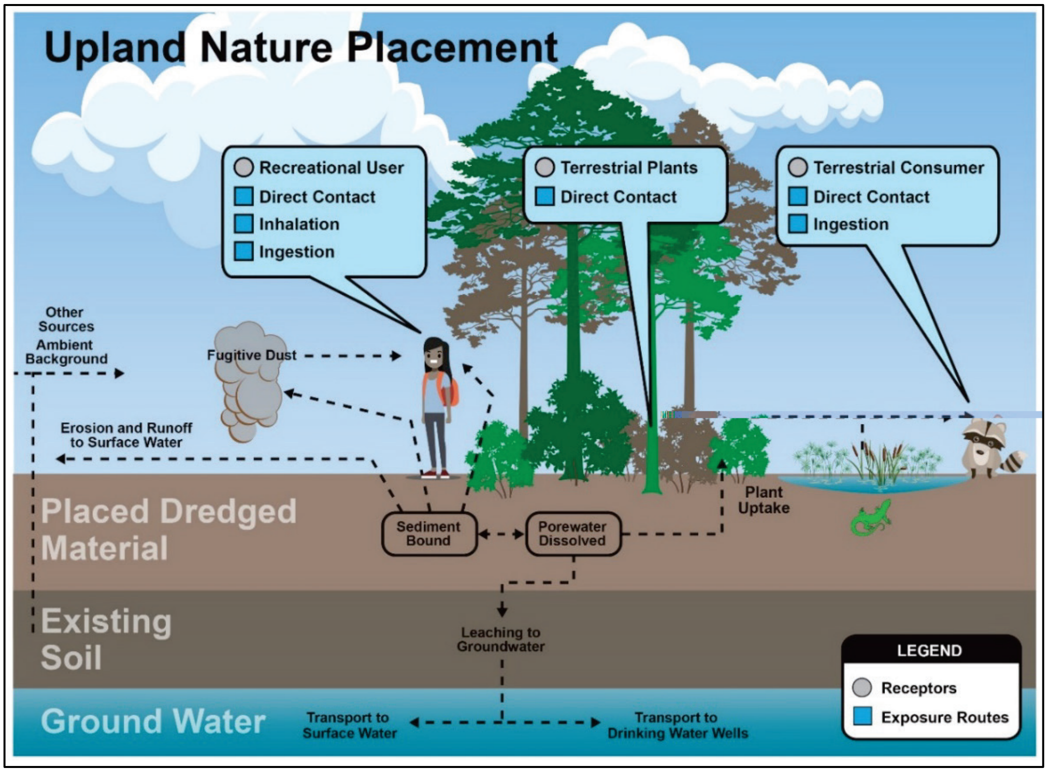
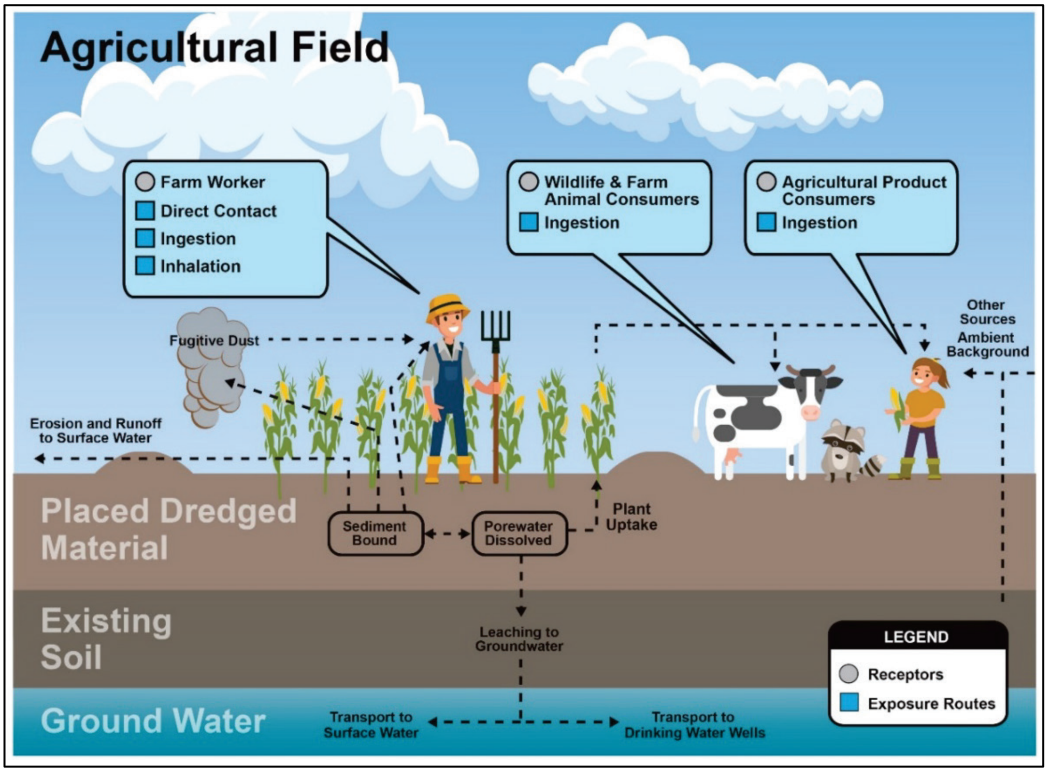


Figure 4-3. Generalized conceptual site model for dredging operations at beneficial use agricultural field sites.



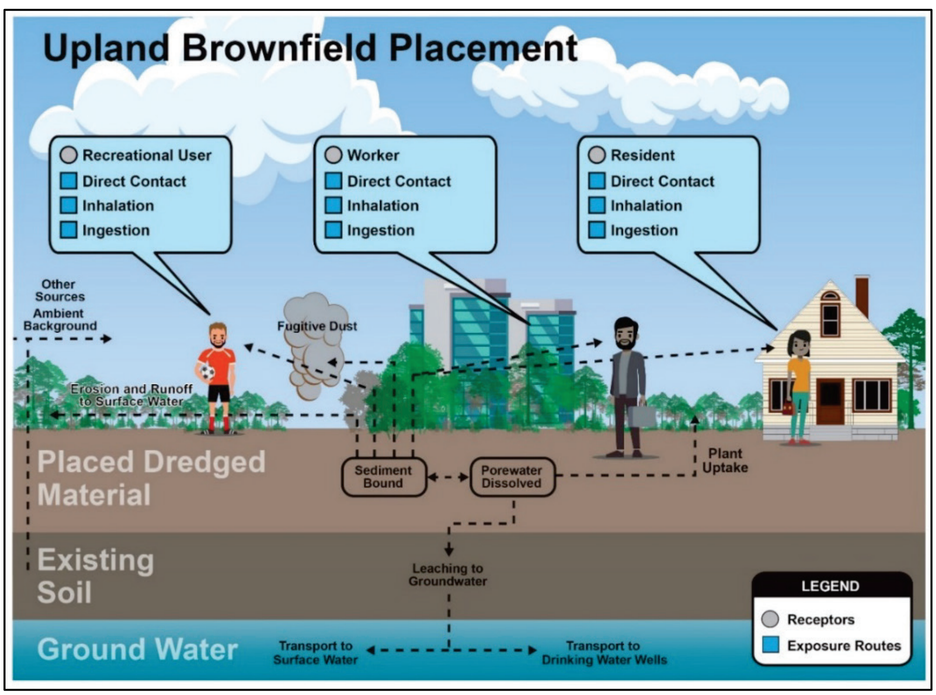
The typical exposure pathways for restored natural areas and agricultural sites are similar. Plants may take up contaminants from the dredged material, with the subsequent potential for exposure of farm animals and wildlife through consumption of crops and other vegetation. Exposure of farm workers at agricultural sites and recreational users at nature preserves (new natural areas) occurs primarily through dermal contact with contaminated soils and inhalation of fugitive dust. The potential for exposure via ingestion by wildlife or consumers of agricultural products needs to be considered. The potential for offsite migration of contaminants into groundwater and surface water (e.g., via leaching, erosion, and/or run-off) will need to be reviewed for upland placement of dredged material to confirm that additional ecological and human receptors are not exposed at unacceptable levels.

Section 6 provides guidance on evaluating each of these potentially complete exposure pathways.

4.3.3 Upland Conceptual Site Model for Brownfield Restoration sites

Figure 4-4 provides a CSM illustrating the primary exposure pathways and typical human receptors relevant to the placement of dredged material at urban brownfield restoration sites.

Figure 4-4. Generalized conceptual site model for dredging operations at beneficial use upland placement commercial, residential, or recreational field sites.



The potential for humans to be exposed to contaminants at a brownfield site depends on the end use of the property and may include industrial, commercial, residential, or recreational end use. In addition, construction workers involved with excavation and installation of subsurface infrastructure may be exposed to contaminants in dredged material placed at the site prior to construction (not shown in Figure 4-4). The primary exposure pathways for humans typically includes inhalation of fugitive dust, ingestion, and dermal contact with soils. Offsite transport to surface water through erosion and leaching to groundwater should also be considered when placing dredged material in upland environments. The major sources of drinking water for most communities along the Great Lakes are the lakes themselves. However, for situations where groundwater is used as a drinking water source, a direct exposure pathway to contaminants that can leach into groundwater should be identified and considered.

4.3.4 Environmental setting and general characteristics of the placement site

The general environmental setting of the beneficial use placement site should be described from the perspective of factors that might influence the potential for contaminants to be mobile resulting in complete exposure pathways to ecological receptors and humans. Such factors, for example, may include the following:

- Type of site: aquatic, wetland, agricultural, industrial, or urban
- Size and circulation of adjacent water body
- Groundwater resources underlying the site and their use
- History of site use
- Land use in the watershed and local area surrounding the project site
- Identification of ambient contamination or stressor levels
- Review of potential continuing sources of contamination that will impact the project benefits and assessment of risk
- Regulatory environmental, zoning, and local controls that may affect future land use or development.

4.3.5 Placement operations and dredged material characteristics

The general characteristics of the sediment being beneficially used should be described from the perspective of factors that might indicate the presence, type, and mobility of contaminants in the material. Characterization of the placement sediments or soils, as well as suitable reference material, would also be appropriate to understand relative risks (see Sections 4.1 and 4.5.2). In addition, the anticipated construction operations and alternative engineering approaches that may impact the potential for exposure should be considered. Such factors may include but are not limited to (as depicted in the CSM) the following:

- Project dredging history
- Volume of material to be dredged
- Method of dredging and placement
- Dredging schedule
- Expected homogeneity of the material that will be dredged
- Known spills or discharges in the area
- Physical characteristics of the material (grain size distribution, water content, plasticity indexes, etc.)
- Ambient turbidity and sediment contaminant levels at the placement site
- Unique characteristics (e.g., potential for emerging contaminants such as microcystin toxins; McQueen et al. [2020a]).

4.3.6 Constituents of Potential Concern (COPCs)

COPCs are the contaminants present in the dredged material being evaluated that have a potential to adversely affect human health and aquatic or terrestrial biota. General COPC concepts are presented here, and COPCs are discussed in detail specific to the contaminant pathway in Sections 5 and 6. The COPCs are site and dredged-material specific. Different COPCs will be associated with different exposure pathways based on the physical/chemical properties of the contaminant and geochemical properties of the site. COPCs to be evaluated are identified on a case-specific basis in the Tier I evaluation for each pathway. If little information is available, the evaluation may enter Tier I with a default COPC list (such as heavy metals, polycyclic aromatic hydrocarbons [PAHs], PCBs, and pesticides). However, through the Tier I process, the default list of COPCs should be replaced by a list of identified COPCs specific to the dredged material, the proposed placement area, and pathways being investigated. It is important that all constituents analyzed in the dredged sediment be initially reviewed to determine whether they should be identified as a COPCs.

4.3.7 Identifying relevant pathways for exposure

When a potential beneficial use site or placement scenario is first identified, the potential migration pathways for COPCs should be evaluated with respect to receptor exposure with a CSM before proceeding further in the tiered testing process. The purpose of this preliminary review is to identify the potential exposure pathways and receptors that warrant evaluation and to eliminate receptors and pathways that clearly do not warrant evaluation. For example, if the beneficial use project specifies that dredged material will be covered with impervious materials (e.g., the material is to be used for road construction), then runoff, volatilization, and direct uptake pathways would not warrant evaluation for that project.

4.3.8 Receptors of concern

Receptors of concern are the populations of animals, humans, and plants that have the potential to be adversely affected by COPCs or other stressors. These are identified through the problem formulation portion of the ecological risk assessment process (USEPA 1989, 1998). Receptors of concern may be different for different beneficial uses, exposure pathways, and COPCs. They should be selected based on relevance to ecosystem and human health, susceptibility to known or potential stressors, and relevance to management goals (USEPA 1998b). The potential exposures and receptors of concern at the placement site determine the tests that will be conducted. In some cases, receptors of concern are evaluated indirectly, such as when water quality is evaluated by measuring COPC concentrations and comparing these to standards. In other cases, it may not be easy to directly evaluate receptors of concern. For example, the receptor of concern may be a local population of edible fish. It is often not possible to directly evaluate potential effects on the population present at a project site, and it may not even be possible or practical to test individual fish of the species of interest. Such cases are common and are addressed with tests of surrogate species from which effects on the population of interest are inferred (USACE 2003).

Note that although identified as potential ecological receptors, especially in the context of habitat restoration applications, aquatic plants, and phytoplankton are not explicitly evaluated as a receptor of concern. Instead, the aquatic placement evaluations focus on impacts to the water column and also impacts to the benthos, including benthic toxicity and bioaccumulation. The use of benthic invertebrates as keystone aquatic receptors of concern and an indicator of sediment quality has long been recognized. The USEPA *Guidance Manual to Support the Assessment of Contaminated Sediments in Freshwater Ecosystems* (USEPA 2002b) refers to the earlier International Joint Commission *Framework for Developing Indicators of Ecosystem Health for the Great Lakes Region* (IJC 1991) as providing the rationale behind the selection of benthic invertebrates (as representative sediment-dwelling organisms) as an indicator of ecosystem health. More recently, the Interstate Technology and Regulatory Council (ITRC), in publishing its guidance entitled *Incorporating Bioavailability Considerations into the Evaluation of Contaminated Sediment Sites,* provides this supporting explanation of use of benthic invertebrates as an environmental indicator of biological integrity and to identify impaired conditions (ITRC 2011b):

> Benthic invertebrates are relatively sedentary organisms that inhabit or depend on the sediment environment to sustain life functions. Because they largely live on (epibenthic) or in (infaunal) the sediment, they are sensitive to both short- and long-term changes in sediment and water quality. Benthic invertebrates are frequently used as environmental indicators of biological integrity because they are found in most aquatic habitats; are of a size permitting ease of collection; reflect water quality conditions or sustainability of ecosystem components; are consumed by a wide range of wildlife species, including fish, amphibians, reptiles, birds, and mammals; and can be used to identify impaired conditions. (USEPA 1989)

The ITRC (ITRC 2011b) goes on to provide the following guidance specific to aquatic plants:

> Determination of direct plant toxicity from plant tissue concentration measurements is generally not a factor in ecological risk assessment and management. More often, measured plant tissue concentrations are intended to be used in the food chain exposure assessment for humans and wildlife.
>
> Plants serve as primary producers in ecosystems. At its most extreme, plant toxicity can result in loss of this function (e.g., unvegetated areas). Secondary effects may include erosion,

> habitat loss, or food loss for other trophic levels. However, because of their sessile nature (with the exception of aquatic algae), plants have evolved unique chemical exclusion (e.g., at the root zone) and compensatory (e.g., metals chelation) mechanisms that allow them to control chemical bioavailability and to survive in environments that could be toxic to other types of life.... (ITRC 2011b)

If there is a site-specific reason to specifically evaluate the effects of dredged sediment on aquatic plants, guidance from the UTM discussed in Section 6.2.3 of this manual for upland placement scenarios, which also may apply in wetland settings, may be useful.

4.4 Documentation of initial evaluation and/or exclusions

It is recommended that a formal memorandum for record or short report be produced at every completed tier of the risk assessment process. Although such documentation can require a significant investment of time, these brief summaries can be invaluable to record the basis for decision-making, to ensure references are not lost, and to provide a communication platform for a potentially large and diverse team. The documentation need not be long but should strive for completeness and should focus on the information used and the decisions reached based on that information. Based on practical experience, consistently documenting discussions, data, and decisions will save time over the course of a large project. Such documentation will facilitate future communication and timely resolution when technical critiques reemerge, following perceived resolution. This documentation can be used to support project documentation and assessments required under the NEPA.

Note that the initial evaluation may be limited to determining the applicability of exclusions, as specified in 40 CFR Part 230.60 and explained in the guidance manuals regarding aquatic placement of dredged material (USEPA/USACE 1998a,b; USACE 2015).

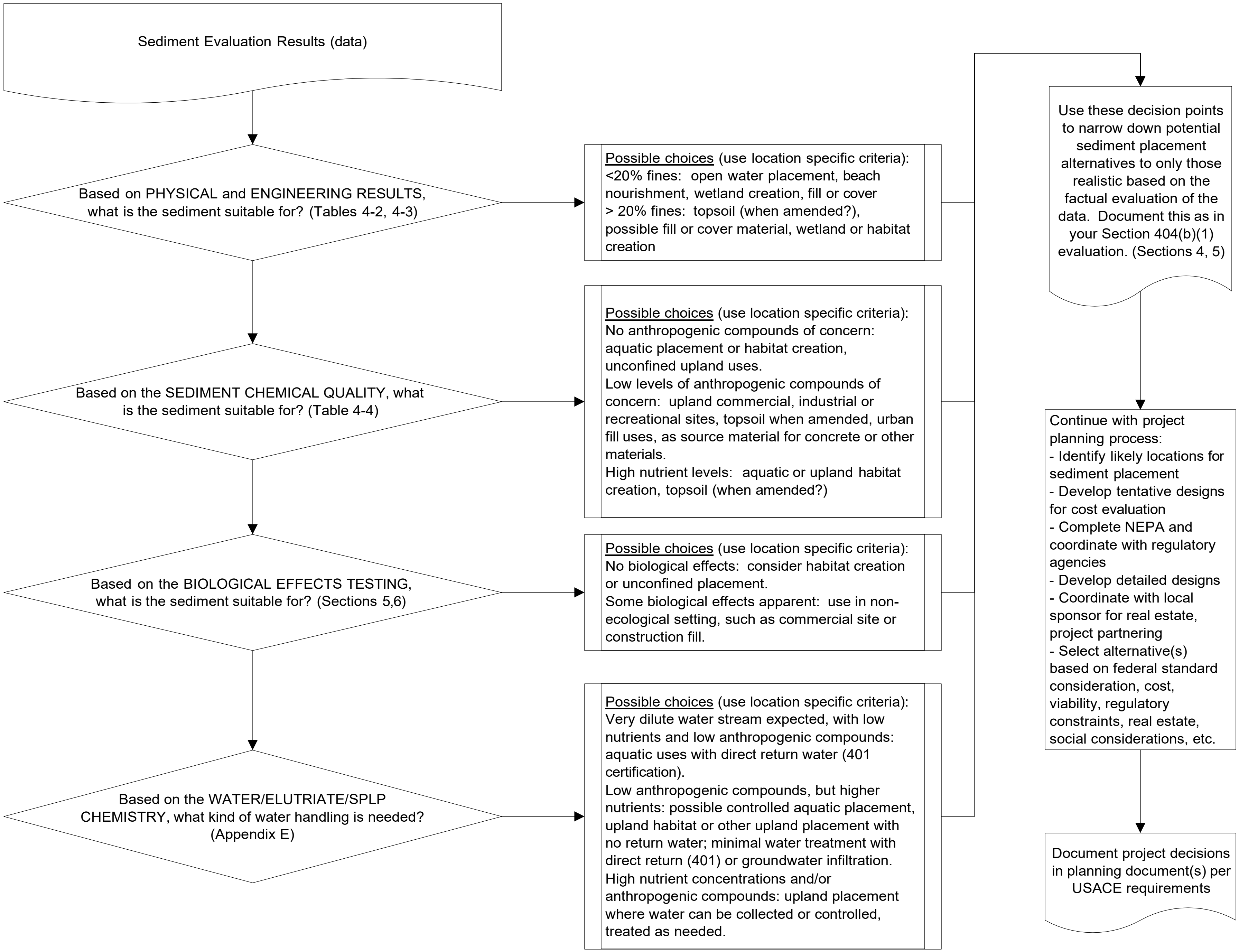
Material may be excluded from further testing prior to aquatic placement if there is reasonable assurance that it is not a carrier of contaminants. Dredged material in the Great Lakes is most likely to be free from chemical, biological, or other pollutants where it is composed primarily of sand, gravel, and other inert materials (USEPA/USACE 199b). For this reason, sandy material has historically been more readily used for aquatic beneficial uses in the Great Lakes. Contaminants and organic matter tend to adhere to finer-grained sediments. Alternatively, if the sediments are from locations far removed from sources of contamination, which may be identified by performing an evaluation similar to a Phase I environmental site assessment (ASTM 2005), extensive testing may not be needed. This determination should be documented.

4.5 Sampling strategy for beneficial use evaluation

When planning a sampling strategy, the existing physical, biological, and chemical characterization for the dredged material should be reviewed. Characterization must include all sediment types to be encountered. These data may exist to support operations and maintenance of the navigation channel, a DMMP, and/or a specific beneficial evaluation (e.g., Section 204). For many Great Lakes dredging projects, a DMMP is established based on physical and chemical characteristics of the dredged material and previous evaluations of management alternatives (see Appendix E of USACE 2000). The DMMP process is a separate planning process that is not discussed in detail here; however, the general framework provided in this manual is directly applicable to the determinations needed for developing beneficial use alternatives and should be incorporated into the beneficial use planning process as appropriate. Guidance on developing sampling strategies can be found in several existing documents (USEPA 2016a; USEPA/USACE 1998b) and Appendix B-2 of this document (Minnesota Guidance on Aquatic Placement of Dredged Sediments for Ecosystem Restoration).

The following flowchart, "General Overview of Evaluations Needed and Some Possible Outcomes Based on Testing Results," provides a general overview of the types of sampling, analyses, and evaluations needed, and some possible dredged material management options based on the testing results. A discussion of physical and chemical sampling, as well as considerations for sampling reference or background locations, is provided below. Further discussion on biological sampling and analyses is provided in Sections 5 (aquatic placement) and 6 (upland placement) as biological sampling and analyses will depend on the environment in which the dredged material is being placed.

General Overview of Evaluations Needed and Some Possible Outcomes Based on Testing Results



4.5.1 Physical and chemical testing for performance suitability

Determining the physical characteristics of the dredged material provides the initial baseline assessment to evaluate whether dredged material is suitable for a desired beneficial use (Section 3). For navigation projects where past dredged material management has been based on a single interpretation of sediment quality (such as a single dredged material management unit for the entire project length for placement in a CDF), separating the dredging area into more than one management unit can be useful. Beneficial use alternatives can then be assessed based on the suitability of material present in different management units (Kreitinger et al. 2011). Examples of this approach have been developed in Duluth, Cleveland, and Buffalo Harbors. For example, in Duluth Harbor, dredged material management units based on the vertical distribution of finer-grained vs. coarser-grained sediments were used to create the sampling plan where decision units and dredging operations were vertically distributed. In Cleveland and Buffalo Harbors, the upper reach of these navigation channels has higher sand content and comprises a significant volume of the annual dredging required.

Chemical and physical characterization of sediments in project navigation channels every 5 yr is often considered sufficient unless extreme storms or spills of toxic materials to the waterway provide reason to believe that the sediment's physical and/or chemical characteristics have significantly changed. However, additional sediment characterization may be needed if the previous sampling strategy was designed to address a single management alternative, such as CDF placement, and a new beneficial use is proposed requiring new or additional information.

4.5.1.1 Physical and engineering testing

Dredged sediment physical characteristics are a key component in determining the most compatible beneficial use. Table 4-2 describes the types of physical tests (describing dredged material index properties) to determine the composition of dredged material and engineering tests needed to determine suitability for various engineering applications, such as structural fill, dike construction, and engineered caps. Table 4-3 provides examples of different beneficial use options that are compatible with various physical characteristics of dredged material (i.e., grain size).

Table 4-2. Appropriate characterization tests for determining physical and engineering properties of dredged material and evaluating suitability for beneficial uses. Updated from Winfield and Lee (1999). http://el.erdc.usace.army.mil/elpubs/pdf/doerc2.pdf.

Physical Analysis (Index Properties)	Source
1. Grain Size	ASTM D422
2. Particle Shape/Texture	ASTM D2488, D4791, D3398
3. Water Content	ASTM D2216
4. Atterberg Limits (Plasticity)	ASTM D4318
5. Organic Content	ASTM D2974
7. Classification	ASTM D2487
Engineering Properties	**Source**
8. Compaction Tests	
Proctors	
Standard Compaction Test	ASTM D698
Modified Compaction Test	ASTM D1557
15 Blow Compaction Test	ASTM D698
California Bearing Ratio	ASTM D1883
9. Consolidation Tests	ASTM D2435
10. Shear Strength	
Unconfined Compression	ASTM D2166
UU (unconsolidated, undrained)	ASTM D2850 ASTM D4767
CU (consolidated, undrained)	ASTM D2434, D5084
10. Permeability	

Note: ASTM = American Society for Testing and Materials

Table 4-3. Beneficial uses most compatible with dredged material of a given composition.

Rock	***Reef Restoration and Creation*** ***Shoreline Protection (offshore berms only)*** ***Bank Stabilization*** Aquaculture ***Construction/Industrial Development***
Sand and Gravel	***Habitat Restoration and Development*** ***Beach Nourishment*** ***Parks and Recreation*** Agriculture, Horticulture, and Aquaculture Strip-Mine Reclamation/Solid Waste Management ***Construction/Industrial Development***
Consolidated Clay	***Habitat Restoration and Development*** Parks and Recreation Construction/Industrial Development
Silt/Soft Clay	***Habitat Restoration and Development*** Parks and Recreation ***Agriculture, Forestry, Horticulture, and Aquaculture*** Construction/Industrial Development (topsoil)
Mixture (rock/sand/ gravel/silt/soft clay)	Habitat Restoration and Development Beach Nourishment (offshore berms only) Parks and Recreation Agriculture, Forestry, Horticulture, and Aquaculture Strip-Mine Reclamation/Solid Waste Management Construction/Industrial Development

*Uses in bold italics text are the most suitable uses for the corresponding material type.
Adapted from USEPA/USACE (2007a) Table 2.1.

4.5.1.2 Chemical and agronomic testing

Many beneficial use alternatives require dredged material to support plant growth to provide various types of habitat in subaquatic, aquatic, wetland, and upland environments. Several soil chemical characteristics are important in determining suitability to support plant growth, including pH, cation exchange capacity, macro and micronutrients, and others. Tests that are often specified for evaluating the suitability of dredged materials for various beneficial uses are shown in Table 4-4.

Table 4-4. Common characterization tests for chemical properties of dredged material to determine suitability for beneficial uses.

	Analysis	Method/Source[1]
	Chemical Parameters	
1.	pH	USEPA 9045D; ASTM D4972; SM 4500-H^+
2.	Calcium carbonate equivalents (hardness)	SM 2340
3.	Cation exchange capacity (CEC)	USEPA 9080/9081
4.	Salinity	SM 2520
5.	Chloride	USEPA 9056A; SM 4500-Cl^-
6.	Sodium	USEPA 6010D/6020B; SM 350-Na
7.	Sodium adsorption ratio (SAR) (porewater)	Soil Survey Laboratory Methods Manual (USDA 2014)
8.	Electrical conductivity (specific conductance) (porewater)	USEPA 9050A; SM 2510
9.	Total organic carbon (TOC)	USEPA 9060A[2]; SM 5310B; ASTM D2974; ASTM D4129
10.	Carbon:Nitrogen ratio	Analyses 9, 11–14 in this table
11.	Total Kjeldahl nitrogen (TKN) (organic + NH_4^+ + NH_3)	USEPA 351.2; EPA 1688; SM 4500-N_{org}
12.	Ammonia-nitrogen (NH_3–N)	USEPA 350.1; SM 4500-NH_3
13.	Nitrate-nitrogen (NO_3^-–N)	USEPA 300.1; USEPA 9056A; SM 4500-NO_3^-
14.	Nitrite-nitrogen (NO_2^-–N)	USEPA 300.1; USEPA 9056A; SM 4500-NO_2^-
15.	Cyanide (CN^-)	USEPA 9012B (with 9013A); SM 4500-CN^-
16.	Total phosphorus	USEPA 365.1; SM 4500-P.B
17.	Orthophosphorus (PO_4^{3-}–P)	USEPA 365.1; USEPA 9056A; SM 4500-P.F
18.	Sulfide (S^{2-})	USEPA 9034; ASTM D4658M; SM 4500-S^{2-}
19.	Acid-volatile sulfide/simultaneously extracted metals	USEPA 821/R-91-100 (Allen et al. 1991)
20.	Sulfate (SO_4^{2-})	USEPA 9056A; ASTM D4327; SM 4500-SO_4^{2-}
21.	Diethylenetriamine–pentaacetic acid (DPTA) metals	Lee et al. (1983); UTM
22.	Total metals	USEPA 6010D/6020B; USEPA 7471B or 7474 for mercury
23.	Pesticides (chlorinated)	USEPA 8081B

Table 4-4. Continued.

	Analysis	Method/Source[1]
	Chemical Parameters	
24.	Polycyclic aromatic hydrocarbons (PAHs)	USEPA 8270E
25.	Polychlorinated biphenyls (PCBs) Aroclor mixtures 209 individual congeners	 USEPA 8082A[3] USEPA 1668B
26.	Dioxins/furans	USEPA 8290A; USEPA 1613B
	Leaching/Runoff	
27.	Toxicity characteristic leaching procedure (TCLP)	USEPA 1311
28.	Synthetic precipitation leaching procedure (SPLP)	USEPA 1312
29.	Leachate Environmental Assessment Framework (LEAF)	USEPA 1313/1314/1315/1316 (USEPA 2017)
30.	Simplified laboratory runoff procedure (SLRP)	Price and Skogerboe (2000); UTM
31.	Rainfall simulator/lysimeter system	UTM

Notes:
[1] ASTM = American Society for Testing and Materials
USEPA = Environmental Protection Agency (https://www.epa.gov/hw-sw846)
SM = Standard Method (https://www.standardmethods.org/) (apply to aqueous matrices [e.g. water or elutriate])
UTM = Upland Testing Manual (USACE [2003])
[2] Method requires modification for use with sediments
[3] USEPA method 8082A also provides results for 19 individual PCB congeners.

4.5.1.3 Chemical and biological testing for environmental suitability

The chemical tests for the presence of contaminants in dredged material are specific to the source and use of dredged material and are often specified by state regulations. The typical COPCs found in harbor sediments of the Great Lakes include heavy metals, PAHs, PCBs, and polychlorinated dibenzodioxins and furans (dioxins). Common test methods used are shown in Table 4-4. Additional information on chemical and biological testing is provided in Sections 5 and 6.

4.5.2 Background, ambient, reference, and control samples

Natural background, ambient, and reference samples constitute various estimates of the baseline concentration of contaminants present in various media (i.e., surface water, groundwater, soil, sediment, or biota) at the beneficial use site or its vicinity. These samples are used to evaluate the net change and potential for adverse effects resulting from use of dredged material for beneficial uses. The appropriate baseline concentration is compared to the concentration in the dredged sediment or to modeled predictions of the expected concentration. This is especially important when a risk-based concentration or criteria is lower than typical background (or ambient) conditions. Thus, a key component to evaluating risk is understanding the ambient concentration of contaminants at the placement site and knowing the potential for other on-going sources of contamination that may affect the potential for increased risk that may result from a beneficial use project.

To ensure that an appropriate approach is used, information gathered during the development of the beneficial use project as well as data from other studies should be used. A well-designed sampling plan is essential to the collection, preservation, and storage of samples so that potential toxicity and bioaccumulation can be accurately assessed. Implementing such a plan is equally essential for dredged material and reference samples (USACE 2003). Guidance on performing sampling and analysis to characterize distributions of constituents at a placement and reference site is provided in existing dredged material testing manuals (e.g., UTM/ITM/GLTM), by the USEPA (2002a), and also by some of the Great Lakes States (IEPA 2013; IDEM 2012; Ohio EPA 2009).

The terminology used to describe different baseline conditions and samples types is often confusing. For the purpose of discussion, the following terms are defined:

- Natural Background: The concentration of a naturally occurring chemical substance derived/originating from natural processes in the environment as close as possible to natural conditions.
- Ambient Background: The concentration of chemical substances in the environment that are representative of the area surrounding the site not attributable to a single identifiable source. These are typically from historic activities with widespread diffuse impacts (e.g., aerial deposition of PAHs or dioxins from motor vehicles and other combustion sources, and PCBs from urbanized areas).
- Reference Sample: Samples (sediment, soil, water, or biota) collected either from the beneficial use site or an alternate location representative of ambient conditions for the beneficial use site. Reference samples are often collected at an alternate location when the

beneficial use site is believed to be contaminated from a defined incident, activity, or source of pollution or when remedial action is being considered for risk reduction.

- Control Sample: Samples of the appropriate media used for quality control during chemistry, toxicity, or bioaccumulation laboratory tests. These samples are unrelated to the project site and characterization of risk. They are used only for determining whether the testing procedures meet the quality assurance and control requirements of the method.

4.5.2.1 Background and ambient environmental data

Some contaminants, such as heavy metals (e.g., arsenic, cadmium, chromium, copper, lead, mercury, silver, nickel, and zinc), are naturally present in environmental media (soils, sediments, water, and biota) and therefore may not arise from an anthropogenic source. The natural concentration of metals in soils, sediments, and plants varies, in part, with the regional bedrock and the geology of the parent material from which the soil or sediment is formed (Dragun 1991). In addition, contaminants with an anthropogenic source are also ubiquitous in urban soils and harbor sediments. PAHs, PCBs, and polychlorinated dibenzodioxins and furans have been detected at low levels in nearly all urban soils and harbor and lake sediments in the Great Lakes.

In some Great Lake states, regional background values for contaminants present in soils and aquatic sediment have been published (see Appendix A of this manual for a listing). These background data vary in the environments that they represent with some studies focused on providing unbiased estimates of soil quality specific to urban (New York, Chicago) or rural soils (New York, Illinois) (Azzolina 2016; Tetra Tech 2003; NYSDEC 2005; EPRI 2004).

Other studies have focused on ambient conditions for aquatic sediments within specific Great Lake harbors. For example, a considerable database has been developed for Duluth/Superior Harbor to support habitat restoration and remediation projects under the USEPA Great Lakes Restoration Initiative. These data have been useful in supporting the assessment of ambient contamination levels for determining the suitability of dredged sediment for habitat restoration projects (LimnoTech 2016; ERDC 2017). Caution should be applied when identifying studies for beneficial use projects within urban or industrial harbors to ensure that the data are truly representative of the urban or industrial ambient conditions. For example, the use of data from biased sampling programs where soils or sediment samples have been collected from areas believed not to be impacted by urban runoff or historical industrial activities may not be truly reflective of ambient conditions in the vicinity of the placement site.

4.5.2.2 Reference samples

Selecting appropriate reference samples is an integral component in identifying COPCs, analytically comparing to COPCs in the test material, and evaluating the relative risk from COPCs in laboratory toxicity and bioaccumulation tests. A reference soil is used in terrestrial evaluations, and reference and placement site sediment is used in wetland and aquatic evaluations. In general, reference soil or sediment samples are obtained at or in the vicinity of the beneficial use site. In some cases, it may be appropriate for one reference site to serve more than one beneficial use project site. In other cases, multiple locations may be used to collect reference samples for a single beneficial use site. This latter case could occur, for example, when the dredged material has a wide range of grain sizes, organic carbon content, or when management needs suggest that placement of different dredged materials at different locations within the beneficial use site is desirable.

Reference soil or sediment samples are generally collected from both the beneficial use site and an alternative ambient or reference site when the placement site is considered potentially contaminated. The alternative reference samples should represent the expected variability of the beneficial use site. When the beneficial use of dredged material is being considered as part of the remedial plan to reduce risk, reference samples are often collected from both the contaminated area to be remediated (in order gauge remedy effectiveness) and from another area(s) that is believed to represent the ambient level of risk.

A reference soil or sediment is not expected to be free of contaminants but depending on the beneficial use project goals, should reflect conditions considered acceptable for the protection of ecological/human health. In addition to this essential characteristic, the physical characteristics of reference soil or sediment should be sufficiently similar to the dredged material so that there is no discernible effect on the response being measured in toxicity tests using plants or animals.

The importance of thoughtful selection of the reference sampling approach cannot be overemphasized; it should be recognized that concentrations within reference sediment are variable.

4.5.2.3 Laboratory control samples

Laboratory control samples are used for quality control during chemistry, toxicity, or bioaccumulation laboratory tests to assess the precision and accuracy of the test. These samples are unrelated to the project site and are only used for determining whether the testing procedures meet the quality assurance and control requirements of the method. The testing laboratory is responsible for collecting, storing, and analyzing control samples. In some cases, control samples are obtained from the National Institute of Standards and Technology, which are described as standard reference materials for chemical analyses. Appropriate control sediments for biological testing are typically collected from a reference location representing clean but otherwise physically comparable natural sediment.

5 Aquatic Beneficial Use Placement Evaluation Methods

5.1 Introduction

Guidance in this section for the evaluation of aquatic beneficial use placement of dredged sediment follows current formal joint USEPA/USACE guidance prescribed in the national ITM (USEPA/USACE 1998a) and regional GLTM (USEPA/USACE 1998b). Pending any revision to these manuals, it is suggested that the user of this manual consult the most up-to-date form of joint USEPA/USACE guidance for the testing and evaluation of dredged sediments.

Section 404 of the CWA required USEPA, in conjunction with the USACE, to promulgate guidelines for the discharge of dredged or fill material to ensure that such proposed discharges in Waters of the United States at a specified site would not result in unacceptable adverse impacts. These guidelines, termed the Section 404(b)(1) Guidelines specified in the CFR Title 40, volume 26, part 230 (40 CFR 230), provide the substantive environmental criteria for evaluating regulatory compliance of proposed dredged sediment discharges with the CWA. The ITM and GLTM provide the primary testing and evaluation guidance for determining contaminant-related impacts evaluated under CWA Section 404(b)(1) Guidelines. These manuals are specifically directed at making a "contaminant determination" per 40 CFR 230.11(d). The guidance in these manuals is appropriate to follow in making risk-based decisions regarding suitability of dredged sediment for beneficial uses in an aquatic environment.

5.2 Tiered approach to aquatic testing and evaluation

Formal guidance in the ITM and GLTM follows a tiered approach to testing and evaluation of dredged sediments that utilizes data on placement site/reference sediment as a primary point of comparison in most cases. There are four consecutive tiers. COPCs are initially identified in Tier I. COPCs are either eliminated from further concern in subsequent tiers (II, III, or IV) or carried forth in the subsequent tier for further testing and/or evaluation. When discharge of the dredged sediment at the specified placement site is predicted to result in no unacceptable adverse effects on the aquatic ecosystem with respect to that COPC, the COPC is eliminated. If a COPC cannot be eliminated through further testing and evaluation, discharge of the dredged sediment at the specified placement site is determined to result in unacceptable adverse effects on the aquatic ecosystem with respect to that COPC. This process is further detailed below.

5.2.1 Tier I – Initial evaluation of potential ecological and human health impacts, and determination of exclusions

Tier I evaluations provide for a factual determination of potential ecological and human impacts based on existing information. Oftentimes this is based on a simple comparison of recent bulk sediment chemistry of dredged sediment to placement site/reference sediment chemistry. If a constituent's concentration is greater in dredged sediment than in placement site/reference sediment such that it could potentially be of toxicological concern based on comparisons to sediment quality criteria, the constituent may be identified as a COPC. In such cases, the dredged sediment is subjected to further testing and evaluation. If no COPCs are identified, no further testing and evaluation are needed provided that existing information indicates compliance with applicable state WQS. A factual determination can also be made at this tier using a multitude of existing sediment-related data and information (e.g., previous bulk chemistry, bioassay and elutriate data) collected or obtained over a period of years among various sampling events. This information and data can be used in the absence of recent data, or to complement recently generated data.

In Tier I, any further exclusions from testing should also be documented, as indicated in Section 4.4. The "contaminant determination" portion of the CWA Guidelines define exclusions from testing in 40 CFR Sections 230.60 (a), (b), (c), and (d). These are outlined in the ITM Section 4.0 and Section 3.4 of the GLTM (USEPA/USACE 1998 a,b). Sections 230.60 (a) and (b) state that if an evaluation of the extraction (dredging) site indicates that the dredged material is not a "carrier of contaminants," the determination of the presence or effects of contaminants can be made without testing. The Guidelines further states that "Dredged or fill material is most likely to be free from chemical, biological, or other pollutants where it is composed primarily of sand, gravel and other inert materials." Section 230.60 (c) states that testing will not be required "where the discharge site is adjacent to the excavation site and subject to the same sources of contaminants, and materials at the two sites are substantially similar." This exclusion applies even if the dredged material is a carrier of contaminants providing that "dissolved materials and suspended particulates can be controlled to prevent carrying pollutants to less contaminated areas." Finally, Section 230.60 (d) states that testing may not be necessary with material likely to be a carrier of contaminants if constraints acceptable to the USACE District Engineer and USEPA Regional Administrator are available to "reduce contamination to acceptable levels within the disposal site and to prevent contaminants from being transported beyond the boundaries of the disposal site.”

5.2.2 Tier II – Screening potential ecological impacts

Ecological impact screening provides an evaluation of benthic and water column impacts using bulk sediment physical and chemical testing, screening, and modeling. Elements included in this tier are described in the following.

Predicting the benthic bioaccumulation of non-polar organic constituents from sediment—The theoretical bioaccumulation potential (TBP) model estimates the bioaccumulation of non-polar organic constituents from sediment by benthic macroinvertebrates (McFarland 1984; McFarland and Clarke 1987). Model variables include bulk sediment concentration, total organic carbon content, biota-sediment accumulation factors (BSAF) and lipid content. Application of the TBP model is relevant only if a non-polar organic constituent is identified as a COPC in Tier I. While the TBP model predicts bioaccumulation in benthic macroinvertebrates from sediment only, it is initially used to screen for bioaccumulation in benthic macroinvertebrates which can then serve as a contaminant pathway to predatory fish. The TBP algorithm is not recommended to be used to predict bioaccumulation of sediment-associated non-polar organic constituents in fish since it does not address spatial exposure, and it does not address various other pathways or consider the variety of other factors that affect bioaccumulation in fish (further discussed in Tier III). In addition, the model may not be predictive if total organic carbon content in the sediment is less than 0.2% to 0.5% (e.g., McFarland et al. 1996; Burgess et al. 2012). The overall drivers of the TBP model tend to be the constituent concentration and BSAF variables. If site- and constituent-specific BSAFs are used, the model will be comparably more predictive than when generic (e.g., default or theoretical) or non-site- or non-constituent-specific BSAFs (e.g., those provided in the database maintained by ERDC (ERDC 2020) are used.

TBP model output is interpreted by comparing the predictions for the dredged sediment and placement site/reference sediment. If non-polar organic constituent residues in macroinvertebrates associated with the dredged sediment are predicted to be lower than those associated with the placement site/reference sediment, no further testing and evaluation are needed, and the constituent is eliminated as a COPC. If a particular non-polar organic constituent residue in the macroinvertebrate associated with the dredged sediment is predicted to be greater than that associated with the placement site/reference sediment, then that constituent continues to be identified as a COPC. In such cases, the dredged sediment is subjected to further testing and evaluation with respect to the constituent.

Evaluating compliance with applicable state WQSs—Section 401 of the CWA specifies that all projects requiring a federal permit, license, or involving federal funding that also propose a discharge of dredged or fill material into Waters of the United States, authorized pursuant to Section 404 of the CWA, also receive Section 401 Water Quality Certification, or a waiver of this certification, from the state. This certification confirms that the discharge complies with applicable state WQS. Each state has its own WQS. WQS for a water body are comprised of three core components: designated uses, water quality criteria and antidegradation requirements (USEPA 2020c).

The CWA Section 404(b)(1) Guidelines at 40 CFR 230.10(b) state in part that "No discharge of dredged or fill material shall be permitted if it: (1) Causes or contributes, after consideration of disposal site dilution and dispersion, to violations of any applicable State Water Quality Standard." This applies at the edge of a designated mixing zone or disposal site. Therefore, evaluating potential impacts to the water column must determine that after dissolved contaminants from the dredged sediment mix and disperse, contaminant concentrations will remain below applicable state WQS for aquatic life protection. Contaminant concentrations below applicable state WQS are considered protective of aquatic life and ecosystem health. For many contaminants, numeric water quality criteria are derived for both acute and chronic effects. Determining which type of criteria is applicable should be a function of exposure time to the dissolved contaminants in dredged sediment effluent in the water column. For aquatic beneficial use projects, exposure time can be either acute (e.g., episodic scow discharges, mechanical off-loading, discontinuous hydraulic discharges) or chronic (e.g., continuous hydraulic discharges). USEPA (1991) defines chronic as a stimulus that continues for a relatively long period of time, often one-tenth or more of the life span for a species of interest, and the term is, henceforth, considered a relative one depending on the life span of the organism. Organisms are assumed to tolerate exposure to concentrations at or below acute criteria for a short periods of time with little to no impact and can be exposed indefinitely at or below chronic criteria. (USEPA 1991).

Evaluating compliance with applicable state WQS can follow one of two approaches. The first approach employs desktop, equilibrium partitioning-based screening that utilizes bulk contaminant concentrations and other information to conservatively estimate dissolved releases into the water column from a proposed discharge of dredged sediment. The second approach employs the use of standard elutriate test data to predict dissolved contaminant releases into the water column from a proposed discharge of dredged sediment. The results of both approaches are used to evaluate whether the discharge would meet or exceed relevant numeric water quality criteria. If the predicted concentration of any constituent in the dredged sediment exceeds any of the water quality criteria, dilution, and dispersion of the discharge within the water column are considered to evaluate compliance at the edge of the mixing zone.

Other elements—Tier II can also employ other screening approaches that predict the toxicity of a COPC identified in Tier I to benthic organisms. These approaches often assume that the fraction of sediment-associated contaminants responsible for eliciting toxicity is the bioavailable form (Burgess et al. 2013). Examples include equilibrium partitioning modeling for nonionic organic chemicals (USEPA 2012b), along with acid volatile sulfide and simultaneously extracted metal testing and modeling (USEPA 2005a). If the modeling predicts that the concentration is below relevant effects thresholds for benthic organisms, no further testing or evaluation is required, and the constituent can be eliminated as a COPC. If the predicted concentration exceeds relevant effects thresholds for benthic organisms, then the constituent would remain as a COPC. In such cases, the dredged sediment is subjected to further testing and evaluation with respect to the COPC.

5.2.3 Tier III – Testing potential ecological impacts

Ecological testing examines water column and benthic impacts using laboratory biological testing (bioassays) to assess the effects of an identified COPC associated with dredged sediment. Most biological measurement endpoints for the prescribed bioassays require a comparison of dredged material to placement site/reference sediment. This is consistent with the basic application, meaning, and intent of CWA Section 404(b)(1) Guidelines, which contain testing and evaluation procedures that require "comparing sediment at the dredging site with sediment at the disposal site," in terms of chemical, physical, and biological characteristics (40 CFR 230.61[c]). The guidelines require ambient conditions be considered during the CWA Section 404 permitting process (40 CFR 230.11[e] and [f]). Placement site/reference sediment and water reflects the conditions at the placement site, and the results of benthic bioassays utilizing placement site/reference sediment provide a point of comparison against which potential effects of dredged sediment placement can be compared.

The bioassays associated with this tier are described in the following.

5.2.3.1 Water column toxicity testing and evaluation

Water column toxicity tests directly determine the potential contaminant-related impacts of a sediment elutriate on organisms in the water column, and the dilution necessary to reduce risk to an acceptable level. Elutriate toxicity testing is not required, but it may be recommended under the CWA for projects in the absence of applicable WQS for COPCs or because of concerns about interactive toxic effects of the contaminant mixture in sediments. Standard laboratory water column toxicity tests for acute effects include 48 hr *Ceriodaphnia dubia* (a daphnid) and 96 hr larval *Pimephales promelas* (fathead minnow), both of which utilize survival as the test measurement endpoint (USEPA 2002c). Depending on the difference in mean survival resulting from exposure to the undiluted (100%) elutriate in comparison to dilution water, the test may warrant exposure to the dilution series. Short-term (i.e., 48 hr) tests with other species that utilize survival as the test measurement endpoint may be considered as an alternative to the 48 hr *C. dubia* and 96 hr larval *P. promelas* tests, but only if they are included in the most recent relevant USEPA or ASTM International guidance documents (e.g., USEPA [2002c] and/or ASTM [1994] and any associated updates) are widely offered as options by commercial laboratories and have been adequately evaluated for decision-making regarding ecological impacts to water column (pelagic) organisms. For example, *Daphnia magna* (another cladoceran like *C. dubia*) is listed as a test species in the GLTM although most recently only *C. dubia* has been used routinely for elutriate toxicity testing.

Water column toxicity test data are interpreted by comparing the mean test species survival in the undiluted elutriate (and dilution treatments, if conducted) to that associated with the dilution water. The initial benchmark in this case is whether the mean survival of the test species exposed to the dredged sediment elutriate is at least 10% reduced compared to mean survival in the dilution water. If survival is less than 10% reduced (relative to survival in the dilution water), there is no acute impact. If more than 10% reduced, then the second benchmark is whether mean survival in the undiluted elutriate is statistically reduced in comparison to the mean survival of the test species exposed to the dilution water. If acute toxicity is predicted through statistically significant differences (e.g., two sample t-test), further evaluation is required to derive water column toxicity standards (Clarke et al. 2002) and model water column mixing. If 50% or greater mortality is observed in the undiluted elutriate, it is logical to test a dilution of the elutriate (e.g., 100%, 50%, 25%, 12.5%, 6.25%) to determine the bounds of toxicity through standard toxicity reference values, such as the lethal median concentration (LC50). Where 50% or greater mortality (or alternative effects) is observed in the standard laboratory water column toxicity tests for acute effects, the toxicity standard as traditionally applied is set to be 1% of the LC_{50} (lethal concentration to 50% of the population) or EC_{50} (effective concentration to 50% of the test organisms) to provide protection from a chronic exposure (Kennedy et al. 2015). Using an application factor of 1% to create the toxicity standard may have limited value, however, because the relationship between concentration and mortality is seldom linear and cannot be used to predict long-term or sublethal effects (Clark et al. 2002). Therefore, the 1% application factor is only used in the absence of other appropriately derived application factors for a particular constituent of concern (COC) for chronic exposures. For example, a 5% to 10% application factor has been identified as protective for ammonia (Kennedy et al. 2015). When observing significant lethality but less than 50% mortality or effects or when evaluating acute or very short non-continuous exposures, the use of a no observable effects concentration or no observable effects level or even a lowest observable effects level may be appropriate benchmarks to serve as the toxicity standard.

Note that while the no observable effects concentration and lowest observable effects concentration were historically used in sediment evaluations and also in other risk assessment applications, it is recognized their generation from statistical hypothesis testing has technical limitations. For example, they are dependent on the use of arbitrarily selected exposure concentrations, and they fail to account for both variability and dose response curve slope. Therefore, a more probabilistic approach is warranted for sound risk management decisions (Chapman et al. 1996; Warne and van Dam 2008; Landis and Chapman 2011; Jager 2012). This approach is more robust if at least five elutriate concentrations are tested. Therefore, approaches such as the USEPA BenchMark Dose, with consideration to the Benchmark Dose Low (Benchmark dose software Ver 2.7 or 3.X [USEPA 2018]), are more appropriate for determining no effects levels. Use of the benchmark dose resolves arbitrarily determined no observable effects concentration values by effectively selecting a low response level (e.g., 5%, 10% response), which may be conceptualized as a point of departure within the dose response curve from the control (i.e., no effects) condition. This strategy is consistent with the USEPA Risk Assessment Forum (USEPA 2012a). The benchmark dose and benchmark dose low may be considered in lieu of the lowest observable effects concentration and no observable effects concentration for the toxicity standard and no effects endpoint, respectively.

After selecting a water column toxicity standard, the mixing of the discharge with the receiving water is modeled, and the concentration of the discharge is predicted as a function of distance from the point of discharge. The predictions are examined to determine if, after mixing, the concentration of the discharge is likely to be below the selected water column toxicity standard at all times outside of the mixing zone and therefore in compliance with CWA regulations.

<u>Note</u>: When a non-persistent substance, such as ammonia, is present at elevated levels that are sufficient alone to cause a toxicological effect in elutriated bioassays, methods to reduce ammonia levels prior to conducting the bioassay to allow toxicological assessment for more COPCs should be used (Melby et al. 2018). The toxicological effects of ammonia are evaluated in Tier II by elutriate testing or modeling of the dissolved form.

5.2.3.2 Benthic toxicity and bioaccumulation testing and evaluation

The prescribed standard whole-sediment toxicity and bioaccumulation tests used to evaluate COPCs in the dredged sediment are briefly described in the following.

Whole sediment toxicity testing: Whole sediment toxicity tests directly determine the potential contaminant-related impacts of the aquatic placement of dredged sediments on benthic invertebrates. Direct effects may occur via contaminant exposure and uptake, which may be followed by a toxic response associated with contaminants. Survival is the main measure used to evaluate the potential toxic effects of COPCs on benthic invertebrates. These tests measure the combined effects of all bioavailable contaminants in a sediment sample that typically include some degree of chemical interaction. Acute (short-term) sediment toxicity tests have been demonstrated to be adequately responsive to legacy contaminants and are the primary choice for evaluating dredged sediment for aquatic beneficial use placement. Testing involving more than one test species is recommended but not required. The use of two or more test species with different life history strategies provides a wider range of varying species' sensitivities and biological endpoint responses.

The standard laboratory solid phase acute toxicity tests include 10-day *Hyalella azteca* (an amphipod) and 10-day *Chironomus dilutus* (formerly *C. tentans*) (a midge fly in its larval form), both of which assess survival as the test measurement endpoint (ASTM International 2020). The *C. dilutus* bioassay can also assess growth as a test measurement endpoint. For both of these bioassays, the survival endpoint is evaluated using a minimal percent difference and statistical comparison of the dredged sediment and placement site/reference sediment. The specific survival benchmarks in comparison to placement site reference are as follows: for *H. azteca*, the mean mortality associated with the dredged sediment is more than 10% greater with the difference being statistically significant, and for *C. dilutus*, the mean mortality associated with the dredged sediment is more than 20% greater with the difference being statistically significant. The *C. dilutus* bioassay growth endpoint is initially evaluated through a comparison to a minimum growth threshold, and then using a minimal percent difference and statistical comparison of the dredged sediment and placement site/reference sediment. Specifically, the growth benchmark is when the mean dry weight of *C. dilutus* exposed to the dredged sediment is (1) less than 0.6 mg per organism; (2) more than 10% less than the mean dry weight of *C. dilutus* exposed to the placement site/reference sediment; and (3) statistically different in comparison to the mean dry weight of *C. dilutus* exposed to the placement site/reference sediment. Finally, it is noted that 10-day tests utilizing other test species to measure survival and growth endpoints may be considered as an alternative to the standard

10-day *H. azteca* and 10-day *C. dilutus* tests. This is predicated on whether or not the tests are included in the most recent relevant USEPA or ASTM International guidance documents, are widely offered as options by commercial laboratories, and have been adequately evaluated for decision-making regarding ecological impacts to the benthos.

As indicated earlier, acute sediment toxicity tests measuring lethality have been demonstrated to be adequately responsive to legacy contaminants and are currently used as the only testing in Tier III for evaluating dredge sediment for open-water placement. With respect to the use of chronic sublethal toxicity tests, the major challenge is the lack of adequate interpretive guidance. Absent such guidance, judgments about the biological and ecological relevance of the chronic endpoints measured must be made before considering use of the data to make decisions regarding whether a discharge of dredged sediment at a specified site would have the potential for unacceptable adverse impacts on benthos. Decisions should not be made based on statistical comparisons to data on open-water placement/reference site sediments alone. Appropriate, ecologically relevant, and scientifically defensible interpretive guidance (e.g., minimum response magnitude, threshold criteria) must be developed before utilizing a toxicity test employing chronic survival and/or sublethal endpoints in the decision-making process for aquatic beneficial use. Other noteworthy issues with the application of chronic sublethal tests include increased variability associated with sublethal endpoints, complexity/difficulty in performing the test, higher rate of failures in meeting performance criteria, higher susceptibility to non-contaminant-related factors (e.g., grain size, ammonia, hydrogen sulfide, indigenous organisms) and increased costs (ASTM International 2020; Kennedy et al. 2009).

Note: The potential influence of sediment non-contaminant factors should be taken into consideration when selecting test species and performing elutriate and whole-sediment toxicity tests. Factors other than sediment-associated contaminants (e.g., grain size, ammonia, hydrogen sulfide, indigenous organisms) can affect biotic responses during exposure to elutriate or whole sediment (Kennedy et al. 2009). Toxicity originating from such factors can be falsely attributed to persistent sediment contaminants and can confound test data leading to inaccurate interpretations that could lead to false-positive conclusions. For example, if ammonia levels exceed the tolerance limits for a test species and ammonia is suspected as a cause of the observed toxicity, then additional ammonia reduction procedures could be employed or an alternate species tolerant of the ammonia levels encountered could be selected for further toxicity testing. Pore water ammonia concentrations above species specific thresholds should be purged prior to testing through application of daily water exchanges. When evidence exists for such factors influencing the outcome of toxicity tests, appropriate toxicity test procedures or methods for conducting toxicity reduction/identification evaluations should be employed (Kreitinger et al. 2017; Melby et al. 2018).

Bioaccumulation testing: The standard 28-day *Lumbriculus variegatus* laboratory bioaccumulation test is utilized to assess the potential bioaccumulation of COPCs from freshwater dredged and placement site/reference sediments. Essentially, mean tissue residues of COPC in test organisms exposed to the dredged sediment are compared to those exposed to placement site/reference sediment. In addition to determining COPC residues, lipid content data on test organism tissues can also be collected for a variety of useful reasons, including providing the option to lipid-normalize non-polar organic constituent data when appropriate.

Before bioaccumulation data are interpreted, any analytical data that are below detection limits or require summation must be converted to an appropriate form for use. The initial benchmark is to determine whether mean COPC concentrations in *L. variegatus* exposed to the dredged sediment are greater than those exposed to the placement site/reference sediment. If they are, the difference is tested for statistically significant differences. If the mean of contaminant tissue concentrations in *L. variegatus* exposed to the dredged sediment is found to be statistically greater, further evaluation is required (USEPA/USACE 1998a) and recommended to make a fully informed risk-based decision (as discussed in Section 4). Various other factors, such as MODs, biomagnification, and bioaccumulation in comparable benthic species in the vicinity of the placement site and in higher trophic levels the aquatic food web, should be considered when determining whether any observed statistically significant difference in laboratory benthic bioaccumulation may be ecologically meaningful (e.g., pose an unacceptable risk) (USEPA/USACE 1998a). Basing dredged sediment management decisions solely on statistically significant differences in laboratory-derived benthic bioaccumulation test tissue residues among dredged sediment and placement site/reference sediment does not consider various variables such as the inherent variability in analytical test data, test organism biology, as well as uncontrolled field conditions and the differences between benthic contaminant exposure in laboratory and field environs. In addition, it is important to recognize that natural and analytical variation in some cases may account for statistically significant differences observed between bioaccumulation tests conducted on dredged sediment and placement site/reference sediment. With respect to evaluating MODs between measured mean COPCs in *L. variegatus* exposed to dredged sediment in comparison to placement site/reference sediment, McQueen et al. (2020b) considers a factor of 2 an appropriate benchmark based on a compilation and assessment of bioaccumulation test data. This finding is consistent with standard laboratory benthic bioaccumulation design prescribed in ASTM International (2019) where it is stated that "the bioaccumulation experiment be designed to detect a two-fold difference between tissue residues in the test and control sediments or the test and reference sediments. A two-fold difference should provide a sufficient signal for ecological and human health concerns in most cases." Appendix F provides further analysis toward interpreting laboratory-derived *L. variegatus* PCB bioaccumulation experiment data, including the application of MODs when encountering statistically significant differences between results on dredged sediment and placement site/reference sediment.

While evaluation of the bioaccumulation endpoint needs to assume dietary exposure given the nature of a discharge of dredged sediment, it is not intended to imply that benthic bioaccumulation from sediment is the sole or primary pathway, or driving factor, influencing bioaccumulation in higher trophic level organisms such as fish. McLeod et al. (2014) (and Appendix F) discuss various factors and processes known to contribute to and affect, to varying degrees, the bioaccumulation of persistent organic contaminants in fish.

5.2.4 Tier IV – Evaluating project-specific ecological exposures and lines of evidence

Dredged sediment proposed for beneficial use in an aquatic setting would generally be anticipated to have a sediment quality similar to and oftentimes better than the placement/reference site sediment. Consequently, the need for Tier 4 is expected to be infrequent. However, there can be instances when information, testing, and evaluation in lower tiers are judged to be insufficient to make a complete factual determination whereby a Tier 4 evaluation should be conducted.

Insufficient information for a determination may include an inability to reach a clear conclusion based on existing data and when evidence is conflicting. This tier may be used for the further testing or evaluation of differences in bioassay results between dredged sediment and placement site/reference sediment that are statistically significant, including when absolute MOD and statistical variances are small or when the results are inconsistent with historical or previous spatial findings. This tier may involve case-specific testing or modeling for toxicity and/or bioaccumulation. This tier can further consider human and ecological health concerns specific to dredged sediment placement configuration and use of the site.

Management actions to control exposure, potential toxicity, or bioaccumulation could be considered in lieu of a Tier IV evaluation. A full discussion of ways to manage and/or mitigate potential risks posed by dredged material placement options is presented in Section 7. Section 6.1.4 also discusses Tier IV analysis and risk management considerations.

6 Upland Beneficial Use Evaluation Methods

Upland placement of dredged material that does not have a direct or point-source discharge of return water to regulated Waters of the United States eliminates the need for a CWA Section 404 approval and Section 401 Water Quality Certification. However, the evaluation of dredged material for upland beneficial use may require evaluation as a solid waste or soil according to state regulations, and the return water may be regulated under Section 402 of the CWA or under groundwater rules (for infiltration). Under jurisdictional state regulations, most upland beneficial use projects will require exemption or water discharge permits before implementation.

All Great Lakes states require testing to determine if soil/dredged material can be classified as “uncontaminated” or “inert” and exempt from further evaluation (e.g., if the evaluation can be limited to Tier I). This may be the case based on no known sources of contamination and low fines content of the sediments. Based on these screening assessments for the presence of contaminants, additional testing and evaluation may be required to determine potential risk and restrictions on end uses per state regulations. Generally, COPC concentrations in dredged material are compared to a set of numerical soil standards designed to protect human and/or ecological exposures through direct contact, and/or through leaching to surface and/or drinking water. Site-specific evaluations and a CSM are needed first to fully understand potential exposure pathways, receptors, and associated COPCs driving any potential risk at a beneficial use site that may be apparent from a simple comparison of dredged material constituent concentrations and numerical soil standards.

The primary environmental standard for establishing the federal standard for USACE civil works projects in the Great Lakes is determining whether a proposed discharge of dredged sediment at a specified site meets CWA Section 404(b)(1) Guidelines. There is no equivalent set of federal guidelines for determining the environmental suitability for upland placement options that fall outside the authority of the CWA. The Great Lakes Commission published a regional framework for evaluating upland beneficial uses (GLC 2004a,b) in response to the lack of guidance for beneficial use, particularly for upland use. It was adapted from the framework described in *Evaluating Environmental Effects of Dredged Material Management Alternatives* (USEPA/USACE 2004) and divided into three phases:

- Screening and preliminary assessment
- Testing and evaluation
- Implementation.

The Great Lakes Commission (GLC 2004a) framework, like the USEPA /USACE (2004) framework, considers testing and evaluation if the preliminary assessment has identified a potential risk from the dredged material for certain beneficial use applications. This manual focuses on identifying suitability for beneficial uses based on physical and chemical characteristics first, so the regulatory issues are resolved before a beneficial use opportunity is identified. Note that beneficial use opportunities can be explored while the environmental suitability is determined for a range of beneficial use options, as is consistent with an Engineering With Nature approach (Section 1.1, Flowchart page 38) (Kreitinger et al. 2011; Bailey et al. 2010) and USACE planning guidance. This section attempts to clarify the process of determining suitability, both as regulated by state authority and as evaluated by existing approaches where regulatory guidance is lacking.

Two Great Lakes states (New York and Ohio) have developed regulatory programs specifically for upland beneficial use of dredged material (Appendix B). Where a state has a regulatory program in place specific to the upland beneficial use of dredged material, state-specific permitting protocols should be followed for determining the suitability of the dredged material for the placement (Appendix B). In the absence of a formal and specific program for regulating upland beneficial use of dredged material, a risk-based framework as developed by the USEPA and described in Section 4 and this section is appropriate to follow for evaluating impacts to human health (USEPA 2016a,b).

6.1 Tiered approach for upland testing and evaluation

The evaluation procedures described in this section for the placement of dredged material in upland beneficial use settings builds on the guidance provided in the UTM (USACE 2003). The UTM uses progressive tiers of evaluation, analogous to the tiered evaluations described in Section 5 for aquatic placement of dredged material for beneficial use. Thus, the general approach described in Section 5 for evaluating compliance of aquatic placement under the CWA—initial screening, tiered testing evaluations, and establishing lines of evidence (LOEs)—is also used to evaluate upland beneficial uses outside the authority of the CWA. A crosswalk between the aquatic and upland tiers (described in Section 5 and this Section, respectively) and associated risk-based processes (described in Section 4) is provided as Table 4-1.

When dredged sediment will be placed in an upland setting outside of the authority of the CWA, the state has the sole authority for permitting upland beneficial use not regulated under CWA Sections 404/401. Soil or solid waste standards established by each state are incorporated here into the tiered process in lieu of CWA standards, but given the generic and/or pathway-specific basis for which many of these standards are adopted, there should be other evaluations to ensure all human/ecological exposure to COPCs are considered.

The tiered approach for characterizing potential impacts to human health and the environment is structured so that each reasonable pathway for exposure and toxicity (identified during development of the site-specific CSM, Section 4.3) is evaluated. The goal of this tiered assessment is to rapidly and efficiently identify the important potential risks to human health and the environment so that a balance can be found between the level of effort required to assess feasibility for a beneficial use project against the value of the benefits to the local and state stakeholders. Sufficient information for evaluating the environmental feasibility of a project will normally be available after a Tier I or Tier II evaluation, often by using regulatory guidance provided by the states. However, a Tier III or Tier IV evaluation will sometimes be necessary when the potential risks are uncertain and the value of the benefits are high. These higher tier evaluations are more costly but may be warranted to provide greater certainty of potential impacts to ecological/human health and the potential need for risk management.

Dredged material that may be placed in an upland setting may result in exposure to ecological and/or human receptors. Upland environmental exposure pathways and associated exposure media (e.g., surface runoff of precipitation, leachate into groundwater, volatilization to the atmosphere, plant and animal bioaccumulation) are listed in Section 4.3.2 and Table 6-1a. The three CSMs presented in Section 4.3 illustrate different potential ecological and human receptors that may be exposed to dredged material when placed in various upland settings: agricultural fields, commercial or residential development areas, parks, and habitat restoration.

Tiers for evaluating exposure pathways are described in Tables 6-1a for ecological receptors and Table 6-1b for human receptors along with evaluation methods applicable for each exposure pathway. The tests identified in Tables 6-1a and 6-1b are presented in the UTM (USACE 2003). The system is structured so that Tier I should be conducted for every pathway evaluated. Generally, Tier I involves identifying existing information and determining if the dredged sediment in question meets the criteria sufficient to be excluded from further evaluation. For purposes of upland beneficial use outside the authority of the CWA, this generally involves either a particle-size criteria suitable for the intended end use and a bulk soil contaminant concentration screening value for listed COPCs that permit the dredged material to be excluded from regulation as solid waste.

Currently, these criteria and the framework for applying solid waste regulations to dredged material vary for each of the Great Lakes states. However, USACE is required, at a minimum, to meet state regulations for upland beneficial use of dredged material if the material is not managed under the regulatory authority of the CWA. USACE, as the permitting authority for dredged material removal and disposal, is also responsible (under NEPA) for assessing the potential environmental effects of the beneficial use of dredged material project on the quality of the human environment and communicating these effects to stakeholders, regardless of the permitting authority under which it is approved. With this in mind, there may be additional evaluation required, beyond those required by each state, to document potential adverse impacts from COPC to both human and ecological receptors. These evaluations may occur in Tiers II–IV until a decision can be reached concerning the acceptability and degree of certainty about the potential risks or impacts. Before initiating testing, it is essential that the information required under each tier be thoroughly understood and that the information necessary for interpreting results at the advanced tiers be assembled and communicated with regulators and stakeholders (USACE 2003). The *Upland Beneficial Use of Dredged Material Testing and Evaluation Annotated Bibliography* (GLC 2004b) includes many additional references that may be useful in evaluating upland beneficial uses of dredged material.

Table 6-1a. Summary of upland pathway procedures for environmental protection.

Tier	Ecological Exposure Pathways for Upland Placement Scenarios				
	Direct contact	Inhalation (volatiles or particulates)	Run-off to surface water (aquatic life)	Leachate (groundwater and surface water seepage)	Plant bioaccumulation and consumption
Tier I: Existing Information	Comparison to regional or reference unimpacted (*background*) sediment and also soil concentrations. Evaluate particle size.				
Tier II: Screening Level Assessment	Bulk sediment chemistry: Comparison to (adjusted[1]) ecological soil screening levels	Estimate volatile emissions using bulk sediment chemistry, total organic carbon, Kd, K_{OC}, Henry's Law constants, diffusivities in air, bulk density of dredged material	Bulk sediment chemistry, total organic carbon, Kd, KOC: Predict porewater concentrations[2] , apply basic mixing considerations, Compare to surface water quality criteria	Bulk sediment chemistry, total organic carbon, Kd, KOC: Predict porewater concentrations[2] , apply basic mixing considerations, Compare to surface water quality criteria	Diethylene-triaminepentaacetic acid (DTPA) extract
Tier III: Effects-Based Chemical and Biological Testing	Screening level ecological risk assessment	Conduct Volatile Flux Chamber Test	Modified Elutriate Test, Simplified Laboratory Runoff Procedure, or Synthetic Precipitation Leachate Procedure: Compare to surface water quality criteria	Sequential Batch Leaching Test: Compare to surface water quality criteria	Plant bioaccumulation test; Compare to screening levels derived according to Appendix C[2]
Tier IV: Site-Specific Risk Assessment and Relative Risk and Benefit Analysis	Site-specific assessment of ecological impacts				

[1] Please see Appendix C (Ecological Biota Screening Levels) *Assessment of Eco-SSLs for Determining Suitability of Dredged Material for Beneficial Use – Plant Pathway.*

[2] Estimations of contaminant release to the water column are described in Section 5.2. Alternatively, utilize laboratory analytical measurements of porewater (e.g., solid phase microextraction) or filtered elutriate samples.

Table 6-1b. Summary of upland pathway procedures for human health.

Tier	Human Health Exposure Pathways for Upland Placement Scenarios						
	Direct contact	Inhalation (volatiles or particulates)	Ingestion of crops	Ingestion of game	Drinking water (surface water source)	Drinking water (groundwater source)	Ingestion of fish (surface water runoff)[1]
Tier I: Existing Information	Comparison to regional or reference unimpacted (*background*) sediment and also soil concentrations, evaluate particle size.						
Tier II: Screening Level Assessment	Comparison to generic USEPA and state-specific risk-based soil screening levels[2] for residential and/or industrial use	Comparison to generic USEPA and state-specific risk-based soil screening levels for residential and/or industrial use, *inhalation pathway only*	DTPA extract	TBP calculation	Bulk sediment chemistry, total organic carbon, Kd, K_{OC}: Predict runoff concentrations[3], apply basic mixing considerations, and compare to USEPA Safe Drinking Water Act Levels	Bulk sediment chemistry, total organic carbon, Kd, K_{OC}: Predict porewater concentrations[3], apply basic mixing considerations, and compare to USEPA Safe Drinking Water Act Levels	Bulk sediment chemistry, total organic carbon, Kd, K_{OC}: Predict runoff concentrations, apply basic mixing considerations, and compare to surface water quality criteria for protection of human health, fish consumption
Tier III: Effects-Based Chemical and Biological Testing	Comparison to scenario—specifically modified soil screening levels	Conduct Volatile Flux Chamber Test	Plant bioaccumu-lation test	Animal bioaccumu-lation test	Modified Elutriate Test, Simplified Laboratory Runoff Procedure, or Synthetic Precipitation Leachate Procedure: Compare to USEPA Safe Drinking Water Act Levels	Sequential Batch Leaching Test: Compare to USEPA Safe Drinking Water Act Levels	Modified Elutriate Test, Simplified Laboratory Runoff Procedure, or Synthetic Precipitation Leachate Procedure: Compare to surface water quality criteria for protection of human health, fish consumption
Tier IV: Site-Specific Risk Assessment and Relative Risk and Benefit Analysis	Scenario and/or site-specific assessment of human health risks[4]						

[1] Runoff need only be considered if the dredged material isn't dewatered prior to placement.

[2] USEPA risk-based screening levels developed for assessing hazardous waste sites may be informative although not relevant from a regulatory perspective. State-specific values are listed in Appendix B.

[3] Estimations of contaminant release to the water column may be made via laboratory analytical measurements of porewater (e.g., solid phase microextraction) or filtered elutriate samples.

[4] Site-specific risk assessment should be performed according to USEPA Risk Assessment Guidance for Superfund protocols, although those may not be relevant from a regulatory perspective (USEPA 2016a and 2016b). State-specific guidance should also be consulted and incorporated into the assessment.

6.1.1 Tier I – Identifying existing information

The evaluation of potential impacts to ecological and human receptors is generally initiated in Tier I by determining what exposure pathways may be complete based on your site-specific conceptual site model (Section 4.3.2). This includes identification of any sources of contamination that may have impacted the sediments. In Tier I, grain size is also typically assessed. As explained in Section 4.4, if the sediments are to be dredged from an area free of contamination and also are predominantly sandy, then only limited testing may be warranted to document their environmental suitability for a beneficial use. In Tier I, constituents measured in dredged material are typically compared to ambient background concentrations at the placement site or a suitable reference site. This is akin to the initial evaluation phase for aquatic placement described in Section 5.2.1. If concentrations of constituents in dredged material exceed these background (*ambient*) or reference concentrations, then additional tiers of evaluation may need to be conducted.

6.1.2 Tier II – Screening level assessment

The Tier II evaluation begins to address COPC-exposure risk based on specific contaminant pathways—direct soil contact, inhalation, surface water, groundwater, plant uptake, and animal uptake—for ecological and human receptors.

Constituents of potential concern in dredged material that fail to pass soil screening values established by the state will require further evaluation if the option of upland beneficial use is pursued. The COPCs that exist at higher concentrations in the dredged material relative to the proposed upland placement site would be further evaluated via site-specific scenarios for dredged material placement and COPC exposure. It is important to recognize that risk management strategies that have a large impact on reducing potential risk to ecological receptors and human health can be incorporated into the beneficial use of dredged materials. However, beneficial use alternatives with limited exposures or avoidance of material exceeding screening criteria may be the best course of action. The choice of COPC screening values is dependent on the exposure pathways being considered and receptors, as developed in the CSM. For upland placement, three general CSMs were introduced in Sections 4.3.2 and 4.3.3. A list of potential screening level evaluations that may be used for each exposure pathway at a site is presented in Tables 6-1a and 6-1b.

Tier II screening level assessments rely on the bulk sediment chemical analysis expressed on a dry-weight basis (e.g., milligram/kilogram) along with other physical and site data that provide a better understanding of potential risk. If adequate data are not available, additional samples should be collected and analyzed. Tier II may also include diethylenetriaminepentaacetic acid (DTPA) extraction and analysis procedures designed to assess the availability of constituents for plant uptake, and environmental interactions of COPCs with existing soils (UTM Appendix H) (USACE 2003). Calculations for runoff and leachate pathways are available; interested parties may contact their local USACE office or ERDC* (Kreitinger et al. 2011).

6.1.3 Tier III – Effects-based chemical and biological testing

Where uncertainty exists following Tiers I and II evaluation of potential impacts to human health and the environment, or no screening criteria are available for evaluating chemistry data, it may be necessary to use Tier III to obtain more detailed information. The evaluations in Tier III include effects-based testing that require the exposure of biota to dredged material and reference/background soils or additional chemical/physical tests on dredged material to simulate site-specific conditions. When there are no defensible Tier I or Tier II procedures for predicting contaminant exposure and toxicity, it may be necessary to conduct Tier III testing to obtain more detailed information. Great Lakes states provide some guidance on Tier III-type tests that can be performed to address COPC risks in specific pathways. For instance, the Synthetic Precipitation Leaching Procedure can be used to predict potential impacts to groundwater based on direct chemical measurements rather than using predicted estimates of the fraction of COPC that may leach into underlying soils and groundwater following placement. However, the Synthetic Precipitation Leaching Procedure and other single pH leaching tests are based on a specified set of environmental conditions (e.g., final pH dictated by the tested material, liquid-to-solid ratio of 20). These conditions may be different in the environment (e.g., final pH may be dictated by the surrounding soil or by amendments, liquid to solid ratio may be much lower). This can dramatically affect the leaching behavior of

* Schroeder, P. R., T. N. Aziz, and S. E. Bailey. (in preparation) *Screening Evaluations for Open Water Disposal of Dredged Material*. ERDC TN DOER-RXX), US Army Engineer Research and Development Center, Vicksburg, MS. (preliminary titling and publication information)

some constituents. The USEPA has developed a Leaching Evaluation Assessment Framework for inorganic constituents (USEPA 2017). These methods and framework can be used to better predict leaching behavior under a range of environmental conditions.

6.1.4 Tier IV – Site-specific risk assessment, relative risk, and benefit analysis

Tier IV consists of a site- or project-specific evaluation of contaminant risk based on the unique characteristics of the dredged material and placement site. For example, a small increase in risk for mercury methylation and bioaccumulation may be considered insignificant and acceptable when below known effects levels and particularly where potential benefit to wildlife is considerable as when dredged material is used to create a wetland meadow within a former industrial waterfront. Considerable flexibility exists in how to conduct a Tier IV evaluation. The Tier IV evaluation consists of collecting and evaluating data that complement and reduce the uncertainty of the Tier II and III evaluations. In that way, the site-specific risk assessment relies on LOE from the various evaluations to develop a conclusion regarding the potential for unacceptable risks that may develop from exposure to dredged material. A Tier IV evaluation may include a detailed human health risk assessment or an assessment of relative risk of dredged material placement (see Section 4.1). In most cases, a Tier IV evaluation will determine whether the relative increase in risks or potential impacts to human health and the environment is considered acceptable.

Considering and weighing project benefits against project risks or uncertainties is an important principle that in certain circumstances can augment evaluations related to the beneficial use of dredged materials that would ultimately be incorporated into the NEPA evaluation that is required of all federal projects. The many potential ecological benefits resulting from a habitat improvement project may greatly outweigh the potential ecological risks associated with low levels of a sediment contaminant near background concentrations. The uncertainties in both realized benefits and potential risks posed by the dredged material placement should inform any risk management measures considered for the project (Section 7). These risk management measures can themselves be evaluated as part of Tier IV. For example, a pilot or demonstration project could be completed and additional monitoring could be performed to better define any potential risks and determine efficacy of proposed controls before full scale implementation of the dredged material management occurs. These additional monitoring results could then be considered as part of the Tier IV assessment.

6.2 Procedures for evaluating impacts through specific contaminant pathways

The contaminant pathways and associated exposure media being evaluated for an upland beneficial use project would be identified in a site-specific CSM (Section 4.3). Tables 6-1a and 6-1b present these pathways and associated exposure media in tabular form, following the guidance provided in the UTM (USACE 2003). The UTM provides guidelines for evaluating potential ecological impacts from COPCs in dredged material when placed in island, nearshore, or upland CDFs (USACE 2003). While the main purpose of the UTM is to identify the contaminant pathways that may result in exposure to receptors outside a CDF, the testing methods and tiered approach for evaluating risks in terrestrial habitats can also be applied to dredged material placed in unconfined settings, including beneficial use sites. Evaluating potential impacts using the UTM is driven by the need to comply with the CWA and NEPA. For projects where return flow does not exist, the regulatory authority for the beneficial use of sediment is driven by state rules for solid waste management and soil recycling. In the absence of state rules and regulations, or where standards do not address risks to terrestrial wildlife, the approaches presented in the UTM are useful for evaluating potential impacts.

6.2.1 Evaluating potential impacts from direct soil contact pathways

As indicated in Tables 6-1a and 6-1b, the evaluation of potential impacts to ecological and human receptors is generally initiated by comparing contaminants measured in dredged material to background concentrations, screening level limits, or other benchmark values. To evaluate potential ecological impacts, these screening level assessments may be supplemented by conducting laboratory soil toxicity and bioaccumulation tests under Tier III evaluations (per USACE 2003). Tier II and III evaluations can be further supplemented with Tier IV evaluations, if necessary. A Tier IV evaluation uses site-specific data to refine estimates of exposure and toxicity based on the unique site conditions, exposure scenarios, and receptors of concern (e.g., wildlife and avian species and residential and recreational land use for humans).

Evaluating the potential risk to humans is initiated with a Tier II evaluation that compares sediment chemistry results to screening values developed for protecting human health. Typically, regional screening levels (RSLs) developed by the USEPA (2020b) in conjunction with state-specific values (Appendix B) are used. USEPA risk-based screening levels developed for assessing hazardous waste sites may be informative although not relevant from a regulatory perspective. Many states have brownfield programs, and some Great Lakes states have programs in place specifically for regulating the beneficial use of dredged material (see Appendix B). These state-specific soil concentrations may be used in lieu of (or in addition to) the USEPA RSLs, depending on the state regulatory program in place. When the concentration of dredged materials exceeds the soil screening values for direct contact, ingestion, or inhalation (not for groundwater protection), a site-specific risk assessment (calculation of actual site- or setting-specific risks from exposure during beneficial use) may be performed during Tier IV.

Human exposure to contaminants in dredged material can result from dermal contact, ingestion of soil particles, inhalation of dust and/or vapors, and ingestion of ground or surface water impacted by leachate or runoff. Exposure and toxicity testing are not usually performed to determine risk to humans, so the Tier III analysis is performed via site-specific refinements to better estimate contaminant exposures rather than using generic/default and sometimes overly conservative estimates which are typically utilized during the screening-level phase (Tiers I and II) . A Tier IV evaluation of potential risk to human health may be warranted under a number of scenarios; for example, if the additive risk from a suite of contaminants could be significant. In this case, simple Tier II comparisons of individual contaminants to screening values underestimate the total risk to human health. In some cases, site-specific management plans could be developed to minimize exposure and risk over time, requiring the refinement of exposure estimates and additional consideration of toxicity factors. Finally, Tier IV analysis may be desired if the potential risk to human health before initiating a project is significant at the site, and the beneficial use of dredged material is designed to reduce the overall risk to human health.

6.2.1.1 Tier I – Initial evaluation of potential impacts via direct soil contact

Two states (New York and Ohio) have developed regulatory programs specifically for upland beneficial use of dredged material (Appendix B). In the other Great Lakes states, the absence of dredged-material specific regulations may result in dredged material being evaluated as a *waste* material under applicable state regulations to determine if the material meets the criteria to be excluded from state solid waste regulations and to allow the dredged material to be evaluated as a soil or soil material (e.g., aggregate, sand, or clay). Appendix B contains a summary of current criteria for each state and links to online sources of the most current information.

The upland Tier I screening of dredged material quality should also involve a comparison of ambient levels of constituents expected at the placement and/or reference site (Section 4.5.2). The USEPA has developed guidance and free software (ProUCL) to assist in the statistical comparison of site data to background concentrations, using a rigorous statistical approach covering a wide-range of data variability, distribution, skewness, and sample size (USEPA 2015). If the dredged material does not contain statistically significant higher concentrations of a constituent than what already would be expected at the placement site, that constituent generally does not need further evaluation. Appendix A provides sources of ambient concentrations for various constituents in soil and sediments across the Great Lakes region that may be used in conjunction with the USEPA guidance and ProUCL software to make this determination. Most of the established background data sets include only metals, although the New York State statewide soil survey includes analyses for organic constituents as well, and the Illinois data sets include PAHs (NYSDEC 2005; IPCB 2013).

The values provided in Appendix A are mainly regional values, although Ohio has established county-specific soil background data sets (Ohio EPA 2015). Obtaining a more site-specific set of reference or background data at the proposed placement site is also appropriate and may be more informative than using state-wide screening values. In addition, it may be very helpful to establish harbor- or site-specific ambient concentrations of certain organic compounds that have very low risk-based screening levels for protection of human health (e.g., benzo(a)pyrene and other PAHs). PAHs, as constituents generated by incomplete combustion of fossil fuels, are ubiquitous in the environment, so identifying screening levels (in Tier II) that do not account for ambient concentrations of PAHs already present at the placement site may complicate the determination of suitability of dredged material for upland beneficial uses. Guidance on performing sampling and analysis to characterize distributions of constituents at a placement, reference, or background location are provided by the USEPA (2002a and 2016b) and by some of the Great Lakes States (IEPA 2013; IDEM 2012; Ohio EPA 2009). The unbiased sampling for urban contaminants as performed in New York and Chicago may be useful examples of how to obtain these types of data for urban harbors in the Great Lakes (Azzolina et al. 2016; Tetra Tech 2003). Links to these and other sources of background and reference data sources are provided in Appendix A.

Caution must be used in assuming COPCs in dredged material that do not exceed background or reference soil levels pose no risk to ecological or human receptors. Dredged material usually consists of buried sediment that is anaerobic and has geochemical characteristics very different from aerobic soils. When used as soil in an aerobic environment, the solubility and bioavailability of some COPCs can be higher and result in potential toxicity. In some cases, additional testing and evaluation may need to be considered to assess potential risk to ecological receptors. Other factors that may warrant additional consideration as part of the Tier I evaluation include determination of physical and chemical parameters that influence COPC bioavailability, such as soil pH, total organic carbon, and clay content of the reference or background site and the dredged material. The speciation of the metallic form being evaluated should also be considered (e.g., total vs. methylmercury).

While the preceding discussion indicates exemptions may negate the requirement for additional testing, due diligence is required under NEPA to communicate potential risks associated with beneficial use of dredged material containing COPCs that may present a potential environmental or human health exposure risk. The evaluation in all tiers rests heavily upon proper identification of COPCs. Tier I also begins the process of eliminating COPCs from further concern, thus narrowing the potential COPCs to a more focused set of COPCs that warrants detailed evaluation and documents the reasons others do not warrant further consideration. This will result in a focused list of COPCs necessary and sufficient to thoroughly assess potential environmental problems associated with the proposed project.

6.2.1.2 Tiers II through IV – Evaluating direct soil contact exposures to ecological receptors

Tier II - Screening potential ecological impacts

The USEPA has developed ecological soil screening levels (Eco-SSLs) that can be used for Tier II assessments of dredged material (USEPA 2003). These screening values were originally developed for evaluating contaminated surface soils and identifying COPCs requiring further evaluation in baseline ecological risk assessments. These USEPA risk-based screening levels developed for assessing hazardous waste sites may be informative although not relevant from a regulatory perspective. The USEPA Eco-SSLs are contaminant concentrations considered protective of ecological receptors that commonly come into contact with soil or ingest biota that live in or on soil. Some of the Great Lakes state dredged material, brownfields, hazardous and/or solid waste programs also incorporate ecological soil screening levels (or soil concentrations considered protective of ecological receptors) into their assessment protocols (Appendix B). These state-specific values should be utilized in the Tier II evaluations as well. Generally, if a COPC in soil is below these conservative ecological screening levels, no further evaluation is warranted, and the evaluation for that COPC ends at Tier II.

However, these USEPA Eco-SSL screening values were not developed using dredged material, and their application to evaluating potential risks resulting from the beneficial use of dredged material in upland environments should be used with caution. For example, the bioavailability of metals in dredged material may be higher or lower than in soils due to the geochemistry of sediments. Dredged material will often have higher concentrations of sulfides, lower redox potential, and differences in the content and character of organic matter when compared to soils above the water table. The application of Eco-SSL screening values for chemicals whose bioavailability is strongly influenced by soil pH, sulfides, and reduced metal species should be reviewed before applying a Tier II evaluation. Therefore, the use of Eco-SSL may be more appropriate to use when evaluating dredged material that has already been dewatered in a CDF than newly dredged material.

Tier III - Testing potential ecological impacts

A Tier III evaluation can be used to provide additional information on the potential for soil toxicity, as well as uptake into terrestrial food webs. For many contaminants, a linear relationship does not exist between the concentration in dredged material and bioavailability to soil invertebrates or plants; thus, laboratory measurements are often conducted. The UTM recommends conducting bioassays on dredged material as well as on a reference sediment or soil for comparison. Standardized laboratory tests have been described for both soil invertebrates and plants (Appendices G and H in USACE [2003]). A Tier III procedure for evaluating plant growth and bioaccumulation uses *Cyperus esculentus*, a common sedge found throughout the Great Lakes region, also known as Yellow Nutsedge. Considerations for plant uptake, including choice of plant species tested, are further discussed in Section 6.2.3.2. Earthworms (*Eisenia fetida*) are typically used as a representative soil invertebrate that can also be used to evaluate toxicity (USACE 2003). The Tier III evaluation will reach one of two conclusions:

1. Growth and/or survival of plants and/or earthworms exposed to the dredged material is not statistically less than growth and/or survival or organisms exposed to the reference material. No further evaluation of plant or animal toxicity is necessary.
2. Growth and/or survival of plants and/or earthworms exposed to the dredged material is statistically less than growth and/or survival from the reference material. This conclusion may indicate the dredged material may not be suitable for placement where environmental impacts may be a concern, such as for habitat restoration and/or nature preserves. However, the magnitude of potential effects on receptors of concern (plants and/or animals) may be further considered via a Tier IV evaluation.

A discussion of bioaccumulation testing (uptake into plants and/or animals for higher level food chain evaluation) is presented in Section 6.2.3.

<u>Tier IV – Evaluating project-specific ecological exposures and line of evidence approach</u>

A Tier IV evaluation of potential ecological impacts through exposure to soil incorporates site or project-specific data to refine the evaluation of risk. The analysis generally compares existing baseline risk before execution of the project and the potential for increased risk that results from execution of the project. There is considerable flexibility on how to conduct a Tier IV evaluation. The Tier IV evaluation can range from collecting and evaluating new site-specific data that complements and reduces the uncertainty of the

Tier II and III evaluations through an LOE approach to a quantitative assessment of site-specific ecological risk following for example the USEPA Ecological Risk Assessment Guidance for Superfund (USEPA 1998b). General guidance for conducting ecological risk assessments has also been developed by the USEPA (see http://www.epa.gov/raf/publications/guidelines-ecological-risk-assessment.htm). These protocols developed for assessing hazardous waste sites may be informative although not relevant from a regulatory perspective (USEPA 2016b).

6.2.1.3 Tiers II through IV – Evaluating direct soil contact impacts to human health

Tier II – Screening for human health impacts

The USEPA has developed risk-based RSLs for evaluating the concentration of contaminants measured in soils at hazardous waste sites (USEPA 2020a). These USEPA risk-based screening levels developed for assessing hazardous waste sites may be informative, although not relevant from a regulatory perspective. The RSLs are used to help identify soil COPCs that may result in unacceptable risk to human health. For carcinogenic contaminants, the RSLs are based on a range of 1 in 1,000,000 (0.000001) to 1 in 10,000 (0.0001) excess lifetime cancer risk, which is considered an acceptable range of cancer risks for people. For contaminants which cause non-cancer toxicity, a metric called the hazard quotient is used. The hazard quotient is a ratio of estimated exposure to the exposure deemed acceptable. A hazard quotient of 1 or below is considered acceptable.

The USEPA has developed RSLs for screening soil contaminant concentrations that are protective of human health for residential and commercial/industrial exposures to soil, air, and tap water (drinking water). The USEPA RSL website provides tables of risk-based screening levels, calculated using the latest toxicity values and physical and chemical properties of contaminants, using a set of default exposure assumptions. The USEPA updates the RSLs approximately every 6 months to reflect updates in toxicity factors and updates to the USEPA risk assessment methodology guidance. Because they are updated so frequently, these screening level tables are not provided as part of this guidance manual, but they should be downloaded at the time of use from the USEPA website: https://www.epa.gov/risk/regional-screening-levels-rsls.

Each of the states have brownfields, hazardous waste, and/or solid waste regulations that utilize risk-based concentrations developed to be protective of human health (Appendix B). The USEPA RSLs may be used in conjunction with state-specific soil benchmarks in order to fully inform the potential for human health impacts from exposure to the dredged material. New York relies on its state-specific brownfield soil cleanup objectives in its process for permitting upland beneficial use of dredged material, while Ohio relies on the USEPA RSL in determining whether dredged material is suitable for upland beneficial use (Appendix B).

The development of a CSM, discussed Section 4.3, is necessary to appropriately choose and apply these RSLs and state-specific soil concentrations protective of human health. For example, if the CSM indicates that the dredged material would be placed in a residential setting, then the generic residential soil screening levels should be used. However, if the dredged material is slated for placement on a brownfield being redeveloped for commercial or industrial use, it would be more appropriate to use the generic industrial use RSLs. The only difference between these RSLs is the amount of exposure to soil for the different land use scenarios.

The assumptions used by the USEPA for the default residential or commercial exposure scenarios (e.g., hours per day someone spends outdoors at the dredged material placement site, the number of days per week or year that someone is exposed, the rate of incidental ingestion of soil that is assumed) is provided at the USEPA RSL website (USEPA 2020b). These screening values, however, are not specific to the Great Lakes; they have been developed to be conservative estimates of exposure applicable to the many different climate zones throughout the United States. Thus, exposure assumptions may or may not be appropriate for a specific beneficial use project and should be reviewed before conducting the evaluation of potential risk to human health. For example, residential RSLs provide a conservative estimate of potential risk that assumes a generic reasonable maximum exposure within a residential setting. To make these screening values conservative for southern climates, the USEPA assumes that residents wear short-sleeved shirts and/or shorts throughout the year and that exposure to soil occurs 350 days per year for 30 yr. These exposure assumptions would clearly not be appropriate for Duluth, MN, or other northern harbors. Exceedances of RSL values by dredged material should be reviewed in more detail, and assumptions should be verified as part of Tier

II evaluations. State-specific soil risk-based concentrations are often more appropriate to apply in the Tier II evaluation.

To perform a Tier II screen for protection of direct contact and inhalation exposure pathways for protection of human health:

1. Download the generic tables from the USEPA website and use the residential (or industrial) screening levels set at a target cancer risk of 1E–06 and hazard quotient of 0.1.
2. Compare average concentrations (or upper confidence limits on the mean concentrations*) of chemicals in the dredged material to the generic residential (or industrial) screening levels.

Constituents that fail the screening would be considered constituents of potential concern and carried forward to the next, more site-specific screening.

<u>Tier III – Human health impacts</u>

Tier III and IV project-specific or site-specific human toxicity data are not collected due to time, expense, and ethical considerations. Rather, these Tiers involve further desktop evaluations and calculations to further refine potential risk indicated by an exceedance of Tier II RSLs by certain COPCs.

Under Tier III the default RSL values developed for beneficial use of dredged material may be adjusted for site-specific exposure scenarios. The USEPA RSL online calculator allows adjustments to the cancer risk limit and target hazard index. For example, half of the Great Lakes states (Illinois, New York, Ohio, and Wisconsin) use or prefer a 1 in 1,000,000 (1E–06) cancer risk limit in developing their state-specific risk-based regulations and guidance addressing soil contamination for upland beneficial use of dredged material. The other Great Lake states (Indiana, Michigan, Minnesota, and Pennsylvania) use or prefer a 1 in 100,000 (1E-05) cancer risk limit in addressing soil contamination levels in their states (Table B.1-1).

* The USEPA ProUCL software mentioned in Section 6.2.1.1 can also be used to develop upper confidence limits on the mean of COPC concentrations in dredged material.

In addition, the online USEPA RSL calculator allows for the adjustment of various exposure factor values. The exposure factor values may be adjusted based on the ranges of ages of the people being exposed (e.g., infants, children, youth, and adults). The exposure factor values that can be adjusted include body weight, exposure duration (years of exposure), exposure frequency (days per year), exposure time (hours per event), parameters involved in the soil ingestion pathway (daily soil ingestion rate), parameters involved in dermal exposure assessment (soil adherence factor and area of exposed skin), and parameters involved in the inhalation exposure assessment (particulate emission factor values, such as climate zone, wind speed, acres exposed and vegetation cover, and volatilization factor values, such as soil characteristics).

For example, protection of a recreational user or park visitor can be estimated if the amount of time spent on the dredged material placement site is adjusted. The residential exposure frequency is assumed to be 350 days/year. A lower exposure frequency, for example, 2 hr per day for 90 days/year (e.g., recommended by Ohio EPA), would be appropriate for many recreational exposure scenarios.

Even in the absence of a state regulatory program specific to upland beneficial use of dredged material, state-specific exposure assumptions used in other upland regulatory programs would be appropriate to incorporate in Tier III evaluations. In some of the Great Lakes states, the state-specific risk-based soil cleanup objectives use exposure factor values that are more appropriate for the Great Lakes than the conservative nationwide exposure assumptions used by the USEPA in development of the RSL. For example, exposure to fugitive dust may be reduced during winter months. According to frost data from the Minnesota Department of Transportation and snow cover data from the Minnesota Office of Climatology, there is an average of 100 days per year in Minnesota when the ground is frozen and covered by 1 in. or more of snow (MPCA 2021a). During these days, it is not likely a human receptor will be exposed to outdoor soil via ingestion, dermal contact, or inhalation of fugitive dust or vapors. (However, a receptor may still be exposed to soil via ingestion of soil present in indoor dust during this 100-day time period.) The Commonwealth of Pennsylvania has also incorporated this assumption of the ground being frozen for 100 days a year in developing its regulations addressing soil contamination (PADEP 2022). Thus, it may be appropriate to reduce the outdoor soil exposure frequency by approximately 100 days a year when developing regional RSLs for the Great Lakes states.

To develop a table containing scenario and/or regional site-specific screening levels, follow these steps:

1. Use the USEPA online RSL calculator to generate site-specific RSLs for a recreational scenario.
2. Choose only those chemicals that failed the background screen (Tier I) as well as the generic RSL screen (Tier II).
3. Adjust the target cancer risk and/or hazard index if appropriate.
4. Adjust the exposure factors to reflect reasonable actual exposure that might be expected at the dredged-material placement site.
5. Download the new scenario and site-specific RSLs and repeat the screen of concentrations of constituents in the dredged material with the new scenario and site-specific RSLs.

These steps are performed to create *regional* residential and industrial-use scenario RSLs. As an illustration of this type of effort, the resulting RSLs for selected constituents (e.g., metals, PAHs, PCBs) were extracted into Tables B.1-2 and B.1-3 (residential and non-residential values, respectively, Appendix B) for comparison to state-specific soil criteria. Note that as stated earlier, the USEPA updates the RSLs approximately every 6 months to reflect updates in toxicity factors and updates to the USEPA risk assessment methodology guidance. The values in Appendix B may not reflect the most current values.

If no constituents exist in the dredged material at concentrations above the new screening levels, then it can be concluded that the dredged material upland placement scenario is protective of human health.

<u>Tier IV – Human health impacts</u>

A Tier IV evaluation of potential human health risks may involve a performance of a full site-specific risk assessment according to USEPA and relevant state guidance (USEPA 1989, Appendix B). This would include calculation of site-specific risks for COPCs that fail the Tier III screen, which may include a further refinement of site-specific exposure factors as well as an examination of toxicity criteria to ensure that the most appropriate toxicity criteria are being utilized. Some COPCs for which a further evaluation of toxicity criteria may be appropriate based on site-specific factors include PCBs (for which a range of toxicity criteria have been developed by the USEPA, depending on the PCB mixture present and exposure route), and COPCs for which toxicity criteria may not be established in the USEPA toxicity database, the Integrated Risk Information System (USEPA 2022. State-specific guidance regarding use of toxicity factors for assessing human health risks should also be consulted (Appendix B).

6.2.2 Evaluating potential impacts from surface water and groundwater pathways

The impacts of COPCs through runoff and leaching to nearby water bodies are a concern where nearshore aquatic plants and animals or groundwater may be adversely affected. The evaluation of potential impacts from exposure to surface water and groundwater is initiated by conducting Tier II comparisons of contaminants predicted to be in the runoff or leachate from dredged material placed in upland sites vs. suitable reference material. Some site-specific data and default values for model variables are used to calculate a predicted concentration for COPCs in leachate or runoff (e.g., Kreitinger et al. 2011). These results are then compared to state WQS for aquatic life or drinking water standards for potable sources of ground or surface water, based on a point of compliance. State-specific guidance regarding the groundwater to surface water pathway should be consulted. For those constituents for which these standards are lacking, the comparisons may be made to background concentrations, screening level limits, or other benchmark values. Additional Tier III laboratory data and analysis may be desired in some cases to improve predictions of contaminant concentrations in leachate or runoff. The Tier IV evaluations generally include use of more rigorous surface water and groundwater quality models that predict the fate and transport of a contaminants providing a final concentration at a selected regulatory point of compliance.

Leachate is the water with associated dissolved and colloidal materials that seep through dredged material and surrounding subsurface soils. Leachate from dredged material placed in an upland site is produced by three potential sources: gravity drainage of the original porewater, inflow of groundwater, and infiltration of precipitation. Solid particles are not generally transported with leachate, and the evaluation is limited to the dissolved (including fine colloidal fraction) concentration of contaminants. The leachate pathway is perhaps the most technically complex to evaluate, yet it rarely is of environmental concern for contaminant migration because of the physical characteristics of most dredged materials, nature of contamination, and low concentrations of contaminants typically encountered (USACE 2003). Precipitation that passes through dredged material and directly enters surface waters is also typically not a concern with regard to water column impacts since the rate of flow of leachate is low while dilution and attenuation through soils for most contaminants are great, resulting in contaminant concentrations near background levels.

Options for managing and controlling water when dredged material is placed upland for beneficial use are presented in Appendix E.

6.2.2.1 Tiered approach for evaluating water pathway exposures for ecological receptors

The approach for evaluating water pathway exposures is summarized in the “run-off” and “leachate” columns in Table 6-1a.

Tier II – Water pathway evaluation for ecological receptors

The Tier II screening for evaluating potential impacts to surface water quality is based on guidelines provided in Section 5 and Appendix G of the UTM (USACE 2003; Schroeder et al. 2004, 2008). (Note that groundwater would not be considered a medium of exposure for ecological receptors unless the groundwater discharges to surface water. At that point, the water is evaluated as surface water.) Specific methods and data requirements for evaluating potential impacts to groundwater quality are often provided by state regulatory agencies and guidance provided in Section 6 and Appendix D of the UTM (USACE 2003). To evaluate the potential risk of impacts to surface and groundwater quality resulting from placement of dredged sediment at upland sites, an initial Tier II screening is conducted by comparing the predicted porewater concentration of contaminants to state WQS for protection of aquatic life (Section 5.2.2). Constituents that are found to exceed state WQS are further evaluated depending on the exposure pathway.

The Tier II leachate screening procedure is based on equilibrium partitioning principles and conservative (i.e., environmentally protective) application of design and operating variables for upland sites (Schroeder 2000; Schroeder and Aziz 2004). The evaluation makes use of site-specific data provided by the user and default values for pertinent variables to calculate a predicted leachate concentration of contaminants in groundwater. A spreadsheet model is available to perform all necessary calculations. The model, along with documentation, can be downloaded as an Automated Dredging and Disposal Alternatives Modeling System module from the USACE Dredging Operations Technical Support website at https://dots.el.erdc.dren.mil/models.html. If desired, equations for manual screening calculations are available (Schroeder 2000). Model parameters may be specified by state regulatory agencies, and guidance is provided in Section 6 and Appendix D of the UTM (USACE 2003). To evaluate potential impacts to groundwater quality for those compounds that have a predicted soil porewater concentration exceeding drinking water standards, the peak concentration in porewater is predicted at the aquifer-vadose zone interface using simplified equations developed as part of the USACE Hydrologic Evaluation of Leachate Production and Quality model.

For those constituents that are not predicted to leach or runoff to a receiving water body of interest above levels of concern, no further evaluation is warranted, and the evaluation ends at Tier II.

Tier III – Water pathway evaluation for ecological receptors

Additional laboratory test data may be collected during Tier III evaluations and to refine estimates of COPC concentrations in leachate and surface water runoff. The potential for partitioning of contaminants from dredged material into soil can be directly measured using the USEPA Synthetic Precipitation Leaching Procedure (USEPA Method 1312), which is a preferred method by many state regulatory agencies. Some states may require the Toxicity Characteristic Leaching Procedure although the procedure was developed for characterizing hazardous waste to be placed in landfills (in an anaerobic and acidic environment) and is not applicable for dredged material that will be used beneficially as a soil.

Two laboratory tests are available for prediction of runoff water quality. They are the Simplified Laboratory Runoff Procedure and the Rainfall Simulator/Lysimeter System tests (USACE 2003, Appendix C). The results of these tests are compared to WQS for protection of aquatic life and may be used in a screening level risk assessment for other ecological or human health exposures. Simplified Laboratory Runoff Procedure testing examines freshly collected, anaerobic sediment for release of organic contaminants so as to avoid losses of organic constituents by volatilization during drying or by oxidation during creation of aerobic conditions. When testing freshly collected, anaerobic sediments for release of metals, the Simplified Laboratory Runoff Procedure oxidizes the sediment using hydrogen peroxide or enhanced drying to simulate aerobic soil conditions. This procedure is designed to increase the oxidation of metals, providing a more realistic evaluation of metal solubility in runoff from upland beneficial use sites. The results of the Tier III testing are typically compared to the jurisdictional state acute WQS for the protection of aquatic life.

Batch and column leaching along with simple modeling can be conducted to evaluate potential leachate quality (Myers et al. 1996; Brannon et al. 1994). The models originally designed for evaluating CDFs can be applied to evaluate leaching potential at upland beneficial use sites. The test options and procedures are defined by site conditions and the physical/chemical characteristics of the COPC. Equilibrium partitioning distribution coefficients are developed for the specific dredged material. The Sequential Batch Leaching Test and the Pancake Column Leach Test are described in the UTM (USACE 2003, Appendix D). However, the Sequential Batch Leaching Test is a simpler procedure that is recommended for freshwater environments. Additionally, the Sequential Batch Leaching Test is more cost and time effective than the Pancake Column Leach Test. Detailed discussions on selection of Sequential Batch Leaching Test vs. Pancake Column Leach Test and appropriate test conditions are provided in the UTM.

As mentioned in Section 6.1.3, the USEPA has developed new leaching protocols. Its Leaching Evaluation Assessment Framework for inorganic constituents can be used to better predict leaching behavior under a range of environmental conditions (USEPA 2017b).

The UTM (USACE 2003) provides guidance on evaluating the results of these runoff and leachate tests. Basically, if these Tier III tests do not produce water concentrations above levels of concern, no further evaluation is warranted and the evaluation ends at Tier III.

Tier IV – Water pathway evaluation for ecological receptors

Site-specific conditions can significantly alter the concentration of COPCs in surface water and groundwater through attenuation and dilution before exposure of receptors. Surface water and groundwater contaminant fate and transport models of varying sophistication may be used to refine and evaluate the potential exposure of ecological and human receptors to contaminants that may be present in surface water runoff and groundwater. Screening level models are generally sufficient to evaluate the exposure from leachate but three-dimensional groundwater and contaminant transport modeling is an option to improve the prediction of contaminant concentrations at the point of compliance or exposure of receptors as a function of time (USACE 2003).

6.2.2.2 *Tiered approach for evaluating water pathway exposures for human health*

The impacts of COPCs through runoff and leaching may be a potential concern where water may be used by humans for drinking water, industrial water supplies, or irrigation, as indicated in the "drinking water" exposure pathway columns in Table 6-1b. The approaches for conducting Tiers II, III, and IV evaluations of potential impacts to surface water and groundwater quality are the same as for ecological receptors discussed above. To evaluate the potential impacts to surface and groundwater quality resulting from placement of dredged sediment at upland sites, an initial Tier II screening is conducted by comparing the predicted porewater concentration of contaminants to state WQS for drinking water quality. The UTM (USACE 2003) provides a screening spreadsheet that will perform the comparison and include the impacts of dilution and attenuation if desired. When the groundwater or surface water may be a source of potable drinking water for human consumption, the most appropriate screening levels would be the USEPA maximum contaminant levels for drinking water, developed under the Safe Drinking Water Act (USEPA 1976). Constituents found to exceed state WQS are further evaluated depending on the specific exposure pathway using site-specific data. Although runoff and leachate discharge to surface water and subsequent consumption of fish from the receiving surface water body is identified as a potentially complete exposure pathway in Table 6-1b, the combined runoff and leachate discharge alone would likely not contain enough mass to affect fish from the water column alone. Therefore, this should be considered a very minor exposure pathway.

6.2.3 Evaluating potential impacts from plant and animal uptake pathways

Upland placement of dredged material can result in uptake of COPCs into plants and animals that are then consumed by ecological receptors and humans. As described above in Section 6.2.1.2, soil contaminants taken up by plants and animals can result in both direct toxicity and bioaccumulation into the terrestrial food web. Tier II evaluation of the potential for impacts to avian and wildlife species resulting from consumption of contaminated plants and soil invertebrates is based on comparing dredged-material contaminant concentrations to Eco-SSLs. Tier III analysis includes the direct measurement of contaminants in plant and invertebrate tissue following exposure to sediments in laboratory bioaccumulation tests. Tier IV evaluations can range in degree of effort from the collection and evaluation of additional data that complements and reduces the uncertainty of the Tier II and III evaluations (for example, by considering size of the project site in comparison to assumed area for exposure for either residents, commercial workers, and/or various ecological receptors) to conducting a formal site-specific ecological or human health risk assessment and assessment of increased risks resulting from the desired beneficial use of the dredged material. As for the previous pathway evaluations, methods for evaluating the plant and animal uptake pathways are presented in the UTM (USACE 2003).

6.2.3.1 Tier II – Plant and animal uptake pathway evaluations

As discussed earlier in Section 6.2.1.2., Eco-SSLs are screening values for soils that are protective of ecological receptors that commonly come into contact with and/or consume biota that live in or on soil. These screening values are derived separately for four groups of ecological receptors, including birds and mammals that consume plants and soil invertebrates. As such, these values are presumed to provide adequate protection of terrestrial ecosystems. The Eco-SSLs are derived to be protective of the full distribution of species' exposure and effects and are intended to be applied at the screening stage of an ecological risk assessment. These screening levels should be used to identify the COPCs that require further evaluation.

Tier II characterization of the potential for uptake of nonionic organic compounds into earthworms and potential risk to avian and wildlife species can be evaluated using the TBP comparisons to reference soils. However, due to large differences in the bioavailability of COPCs in different soil types, Tier III direct exposure tests are preferred, especially for evaluating bioaccumulation by earthworms and plants that may be consumed by avian and wildlife species.

Tier II evaluations may also include chemical extraction methods or models to estimate bioavailable fractions for non-polar organic contaminants, such as PAHs and PCBs, two of the most common Great Lakes sediment contaminants. The UTM (USACE 2003) provides guidance for evaluating the TBP of a dredged material and estimating the potential bioaccumulation of non-polar organic contaminants to earthworms in terrestrial environments (Appendix G of the UTM [USACE 2003]). The application of the TBP involves comparison of bioaccumulation potential in the dredged material with bioaccumulation potential from a suitable reference site. If there is not a statistically significant increase in bioaccumulation potential from the dredged material as compared to the reference site, then no further evaluation for this COPC is warranted.

This procedure was originally developed for direct contact of aquatic organisms in sediment and has been used to predict uptake of contaminants by terrestrial worms in aerobic soils with some limitations. For example, a Tier II method to predict metal availability to earthworms has not been developed. It is, therefore, preferred to evaluate metal bioavailability to earthworms by conducting Tier III laboratory direct exposure tests that can then be used to estimate impacts to avian and wildlife species (see section below).

6.2.3.2 Tier III – Plant and animal uptake pathway evaluations

The UTM describes a Tier III procedure for evaluating plant bioaccumulation using *Cyperus* esculentus, a common sedge found throughout the Great Lakes region, also known as Yellow Nutsedge (UTM Appendix H [USACE 2003]). The test described uses an index plant representative of vegetation found in habitats in and around a CDF, as this species is able to persist under harsh growing conditions commonly found on CDFs. However, other plants that may be present at a beneficial use project may be used. For instance, turf grass, field crops including leafy vegetables, or species used for wildlife enhancement may be used. Choosing the appropriate plant species consistent with ecological management goals and/or human health exposure pathways of concern is key to adequately assessing this pathway if plant bioaccumulation of metals may be an issue. Plant uptake of metals is highly variable and dependent on plant species, soil/sediment properties, and environmental conditions (e.g., pH and redox) (McBride 2002). The same test species must be used in the test material, the reference soil, and the control. Results of the plant bioaccumulation testing are used to compare COPC uptake by plants on dredged material to COPC levels in plants in ambient or background conditions. Additionally, the results can be used to perform a risk assessment to determine potential exposure through the food chain and if that exposure could result in exposures to harmful concentrations impacting ecological receptors or human health.

Earthworms (*Eisenia fetida*) are used as a representative soil invertebrate that can accumulate a wide variety of contaminants from the soil in which it lives. The bioaccumulation assays provide information on (1) bioavailability and mobility of contaminants from soil to soil-dwelling earthworms, (2) bioavailability and mobility of contaminants from soil to soil-supported plants, and (3) the potential for contaminant movement to higher organisms (e.g., birds, mammals, amphibians, reptiles) that are linked to plants and earthworms in the food web. These bioaccumulation assays are described in detail in Appendix G of the UTM (USACE 2003).

Tissue concentrations of COPCs in earthworms and plants exposed to dredged material and a reference soil are statistically compared to determine if exposure to dredged material result in elevated tissue concentrations.

The Tier III evaluation will reach one of two conclusions:

1. Bioaccumulation from the dredged material is not statistically greater than bioaccumulation from the reference material. No further evaluation of plant or animal bioaccumulation is necessary.
2. Bioaccumulation from the dredged material is statistically greater than bioaccumulation from the reference material. Therefore, the magnitude of potential effects on receptors of concern (plants and/or animals) should be considered via a Tier IV evaluation for any bioaccumulative COPCs that have failed the earlier assessment tiers.

6.2.3.3 Tier IV – Plant and animal uptake pathway evaluations

Tier IV consists of a site- or project-specific evaluation of risk based on the unique characteristics of the dredged material and placement site and would only consider COPCs that have failed previous evaluation tiers. Considerable flexibility exists in how a Tier IV evaluation is conducted. The Tier IV evaluation can range from the collection and evaluation of data that complements and reduces the uncertainty of the Tier II and III evaluations through an LOE approach, or a Tier IV evaluation may include a formal human health site-specific risk assessment and assessment of increased risk resulting from the desired beneficial use of the dredged material.

For example, for ecological receptors, a statistically significant increase in tissue concentrations (relative to uptake from reference material) identified in Tier III indicates a greater potential for exposure; however, the biological significance to ecological receptors requires additional analysis of the data. For this purpose, the Ecological Biota Screening Levels (Eco-BSLs) have been developed where the data are available. The Eco-BSLs (Appendix C) were established by taking the Toxicity Reference Values (TRVs) for the shrew (mammal) and the woodcock (avian) used in the calculation of Ecological Soil Screening Levels (USEPA 2003), and determining a maximum acceptable tissue concentration in worms and plants using the following formula:

$$Eco\text{-}BSL = (TRV \times BW)/(F \times CR) \tag{6-1}$$

where

TRV = Toxicity Reference Value (mg dry weight/kg body weight per day)
BW = the body weight of target receptor (kg)
F = the fraction of tissue consumed
CR = the consumption rate (kg dry weight tissue per day).

Parameters used for body weight per day, fraction of tissue consumed, and consumption rate were as specified by the Ecological Soil Screening Level documentation.

In most cases, a Tier IV evaluation will incorporate the evaluation of the relative increase in risks to human health and the environment compared to the relative increase in benefits to human health and the environment. Ultimately, the potential risks and uncertainties identified in Tier IV along with potential management actions needed should be compared to the benefits derived by the project. As indicated in Section 4.1, weighing the importance of potential benefits vs. potential negative impacts and evaluating the uncertainties are stakeholder-driven processes. A discussion of ways to manage and/or mitigate potential risks posed by dredged material placement options is presented in Section 7.

7 Risk Management

Risk management is required to control unacceptable adverse impacts identified in Sections 5 and 6 from exposures to contaminants. In addition, beneficial use projects often have risk and uncertainty that may arise in the design and construction. There may also be other non-contaminant related risks associated with the execution and success of a beneficial use project which are a function of the geotechnical or physical attributes of the dredged material or the placement site, or engineering considerations related to the dredging and placement processes. The engineer manual *Dredging and Dredged Material Management* provides a discussion of these attributes and processes which may be helpful in addressing non-contaminant related beneficial use project execution risks (USACE 2015). Risk management consists of the implementation of controls, tools, and approaches to reduce risk and address uncertainty.

7.1 Uncertainty and risk management in beneficial use projects

Risk management is the process to manage and control contaminant-related exposures. Uncertainty related to the adequacy of site characterization or contaminant bioavailability are two key factors that must be taken into consideration in assessing and managing risk. Uncertainties may require measures to control variables in design or placement operations; however, the tiered testing approach was designed to provide multiple lines of evidence regarding the effects of exposure. Thus, effective risk management addresses exposure. A major aim is to transparently integrate the context, assumptions, and uncertainties involved in characterizing and managing any risks identified during beneficial use evaluation (USEPA 2016a).

<u>Contaminant-related risk uncertainties</u> – Contaminant-related risks are evaluated based upon a scientifically defensible assessment of potential ecological or human health risks associated with exposure to constituents of concern in the dredged material. Uncertainties associated with contaminant-related risks may include, but are not limited to, the following:

- Completeness and accuracy of site characterization/variability of dredged material
- Estimated vs. actual exposures

- Uncertainties in contaminant effects
- Bioavailability of contaminants in dredged material.

<u>Operational uncertainties</u> – Operational uncertainties may include the following:

- Dredged material placement issues (i.e., constructability due to material separation, slope instability, etc.)
- Material migration increasing the potential for unanticipated impacts outside of the placement site boundaries.

An adaptive management plan provides a means for anticipating and managing identified risks and uncertainties associated with a project. Adaptive management has been used in the context of ecosystem management for many years (Holling 1978; Walters 1986), including use on large scale remediation and restoration projects (Convertino et al. 2013). Within the context of the beneficial use of dredged sediment, however, adaptive management as a formalized process is underutilized. Opportunities for adaptively managing the beneficial use of navigation dredged sediment exist; however, guidance for developing an adaptive management plan addressing common uncertainties associated with beneficial use of dredged material is lacking. An adaptive management plan consists of a framework of decision criteria and response actions to be implemented if and when unexpected conditions or outcomes are encountered. These decision criteria and response actions are developed based on identified exposure pathways with unacceptable risk and site-specific uncertainties related to these pathways and are informed by field operations or ongoing monitoring as the project progresses (see Section 7.5). Response actions could include, for example, design modifications necessary to achieve a target exposure concentration such as adding silt curtains to reduce the transport of suspended solids (SS) outside of the intended project area or the placement of silt curtains to protect sensitive aquatic communities near the placement site.

The opportunities presented for an integrated adaptive/risk management approach are dependent upon the type and quality of relevant information available, the ability to accurately monitor conditions in a manner that informs operations in a timely fashion, and the ability to utilize this information throughout the progression of the project in order to improve project outcomes or to avoid undesirable impacts.

7.2 Management of contaminant exposure risks during beneficial use projects

The need for risk management associated with exposure to contaminants in the dredged sediment should be established based on a considered evaluation of the likelihood and anticipated degree of exposure to receptors of concern. The ecological and engineering goals established for the project (Section 4.2) should be used in conjunction with the conceptual model (Section 4.3) to identify contaminant exposure pathways of concern and to inform the need for management to prevent unacceptable levels of exposure. While the conceptual site model informs the need for management, the selected management alternatives can alter the conceptual site model by reducing or eliminating exposure pathways. The assessment thus is often iterative in nature.

Risk management is an inherent part of the environmental evaluation and project-design process and may be within the initial design to manage risks identified during the site characterization, testing and design process, or reactively to manage unexpected conditions or risks encountered during or following implementation. The risk management options presented in this section generally need only be considered if potentially unacceptable ecological or human health risks are identified during testing, design, construction or follow-on monitoring, or there is a high degree of uncertainty in the testing, evaluation or variability in the materials be dredged.

The principal options available for management of contaminant-associated risks include the following:

- Operational, engineering, and institutional controls
- Water treatment
- Sediment treatment.

Most contaminant-related concerns can be addressed at beneficial use sites by selective sourcing of dredged material based on prior assessment of material suitability and the use of appropriate operational, engineering, and institutional controls. Alternatively, where economic and effective treatment alternatives are available, marginally unsuitable or marginally suitable dredged sediment and associated process water streams could be treated to reduce the exposure concentrations. Treatment eliminates the need for such long-term controls—which may require maintenance or monitoring—and provide definitive risk reduction, while broadening the volume of sediments considered suitable for beneficial use. Operational, engineering, and institutional controls are further discussed in the following sections while advanced water and sediment treatment alternatives not representative of risk management measures taken at beneficial use sites are presented in Appendix D.

7.3 Operational, engineering, and institutional controls

Operational, engineering, and institutional controls provide a means for managing risks associated with dredged sediment placement, including the following:

- Direct exposure to contaminants in sediment or release of pore water or carrier water
- Indirect exposure to contaminants from the site through the food chain
- Direct impacts of placement, such as inadvertent sedimentation on sensitive environmental areas
- Uncertainty associated with site conditions such as the potential for dredged material migration.

Contaminant concentrations considered acceptable and safe in dredged material intended for beneficial use placement may vary from state to state. The use of controls may or may not be considered sufficient, from a regulatory perspective, to allow beneficial use of dredged material with contaminant levels exceeding specified thresholds. Dredged material with high levels of contaminants is generally considered unsuitable for beneficial use, with or without the use of contaminant controls. However, controls may still be considered as a risk management or adaptive management measure for materials otherwise determined to be suitable for beneficial use to minimize potential exposure impacts in receptors of particular concern or to address uncertainties associated with dredged material variability, contaminant characterization, and bioavailability.

Operational controls include actions that are implemented when placing the material to improve accuracy of placement or minimize water column impacts. Examples include the following:

- Reducing the rate of material discharge at the placement site to achieve more accurate placement or more effective distribution

- Reducing rate of discharge to minimize water column impacts, such as elevated turbidity, and increase effective dilution of dissolved contaminants
- Sequential placement of less acceptable material first and then covering it with cleaner material
- Changing dredging/placement equipment.

Engineering controls involve physical barriers or specialized equipment, to prevent inadvertent material losses or prevent unacceptable levels of contaminant release from the dredged sediment, or isolate contaminated materials from potential receptors. Examples include the following:

- Submerged diffusers
- Berms
- Capping materials and vegetation
- Treatment of return water–settling aids, filters, amendments
- Runoff controls–detention basins, silt fences
- Fences.

Institutional controls include actions to prevent disturbance of the site that might result in contaminant releases or to limit access to the site or indirect exposures, and may include, but are not limited to, the following:

- Fish consumption advisories
- Restrictions on digging/excavation
- No wake zones
- No anchorage zones.

Engineering controls may require long-term monitoring to confirm that they continue to perform as intended and to inform maintenance or adaptive management actions, which may include the need to implement additional institutional controls. In any application where controls are employed as a risk management measure, an appropriate monitoring, maintenance, and adaptive management plan should be developed and implemented.

Because it is not always appropriate, feasible, or beneficial to implement controls, the decision to employ controls should consider the trade-offs between costs, timeliness, and environmental protection, both locally and globally. For example, the use of silt curtains can reduce turbidity outside of the placement area. However, the associated restrictions on movement in and out of the contained area can increase project duration, thus increasing the project's carbon footprint, impacting air quality, and increasing the overall disturbance to the ecosystem. In addition, an increase in project costs may limit the volume that can be placed for beneficial use, reducing the benefits derived from the placement.

7.4 Controls for aquatic placement

7.4.1 Water column exposure pathway controls

There are two primary exposure pathways relevant to aquatic placement of dredged sediments, the water column pathway and the benthic pathway. An overview of typical operational and engineering controls for mitigating short- and long-term risks to the water column and benthic pathways for aquatic placement is presented in Figures 7-1 and 7-2. Quantifying the effectiveness of these techniques as risk reduction measures generally requires modeling to compare exposures and risk with and without a control.

The impact of these controls on the placement operations should also be considered; in some cases, optimum placement operations will conflict with controls necessary to limit environmental exposures. For example, a high discharge rate may be needed to distribute material over a sufficient area, but it may also result in elevated SS and total contaminant concentrations in the water column. Ideally, operations would be balanced to achieve adequate risk reduction and optimum placement operations, but the interdependency of many of these processes—as shown in Figure 7-1—will likely dictate some operational compromises and adjustments.

The greatest number of controls is available for turbidity/SS in the water column (Figure 7-1). Use of various devices and pipe configurations to minimize energy of discharges such as submerged discharge, tremie tubes and diffusers; use of high solids pumps; use of mechanical dredging and mechanical placement; proper inspection and maintenance of scow seals; maintaining adequate vessel clearance; and limiting traffic over the site while material is in an unconsolidated state are among the operational and engineering alternatives for turbidity/SS control. Physical barriers, such as sheet pile and silt/turbidity curtains, can limit the movement of SS, minimizing the area impacted by elevated levels of turbidity/SS.

Daily or more frequent inspection by construction staff is needed to observe and monitor levels of turbidity, wind direction, weather, and wave conditions; if turbidity/particulate-associated contaminant issues are anticipated, sensors mounted on buoys may be used to track turbidity. When elevated turbidity levels are observed, predetermined actions are then taken, such as sampling at a higher frequency, changing operations, or shutting down placement. Physical samples may also be taken for comparison to predetermined thresholds and determination of requisite action, as indicated by the adaptive management plan developed for the project. Conditions and operations before and at the time of sampling should be documented to determine possible causes of threshold exceedances, including boat movement, traffic, and operations and environmental factors, such as wind, wave direction, and energy.

7.4.2 Benthic exposure pathway controls

There are also several effective and demonstrated controls available to manage risk associated with exposure of benthic organisms to contaminants in dredged sediment and to reduce the potential for bioaccumulation and trophic transfer. Sequential placement, a form of capping, could be used to place dredged sediments requiring benthic exposure pathway controls first and then covering them with dredged sediment that do not require benthic controls. Of the controls shown in Figure 7-2, capping is a well-demonstrated method for isolating contaminants in sediment from the aquatic environment; there are many examples of this technology being used for remediation of even highly contaminated sediment sites.

A comprehensive guidance document is under development that will synthesize the current knowledge base pertaining to in situ remediation—primarily capping[*]. The ITRC also recently published a technical document, *Remedy Selection for Contaminated Sediments* (ITRC 2014), which includes a section on conventional and amended capping. The ITRC (2014) cites several references, but in particular USEPA (2005d) and Palermo et al. (1998), which contain comprehensive guidance on capping.

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[*] Schroeder, P., K. Gustavson, D. Reible, C. Ruiz, T. Fredette, P. Gidley, T. Estes, M. Palermo, S. Bailey, T. Borrowman, M. Channell, Martin, and D. Averett. (in preparation). *Technical Guidelines for In-Situ Sediment Remediation*, USEPA/USACE, USEPA Office of Superfund, Washington, DC. (preliminary titling and publication information)

More recent technology developments may not be reflected in these references, including the use of reactive amendments in caps. Direct amendment of sediment with activated carbon to sequester contaminants in forms unavailable for biotic uptake in the aquatic/benthic environment has been demonstrated in numerous field-scale, and some full-scale projects. A summary is provided in Patmont et al. (2015) along with considerations relevant to selecting and implementing this technology are also discussed in this paper. The use of amendments is described in more detail in Appendix D.

Figure 7-1. Operational and engineering controls relevant to water column exposure pathway for aquatic placement.

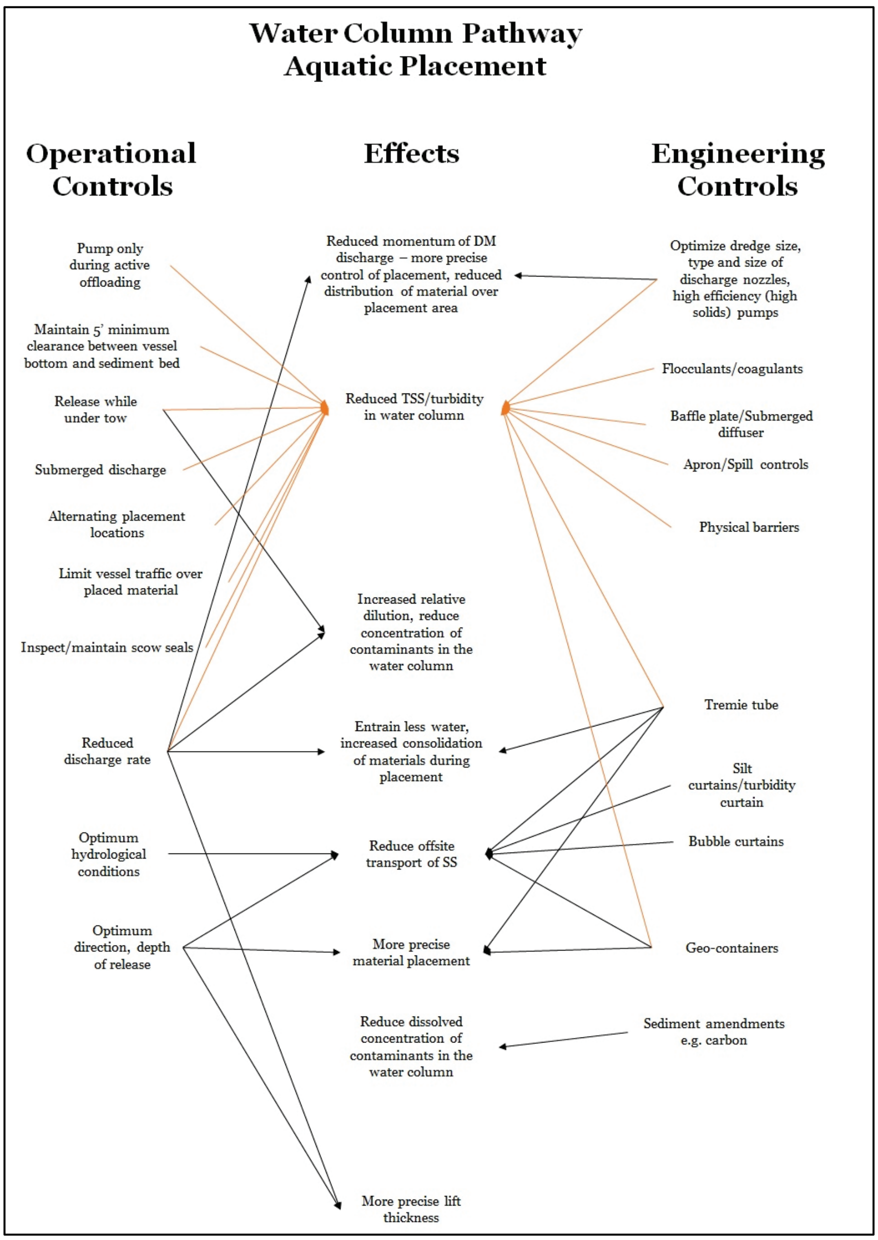
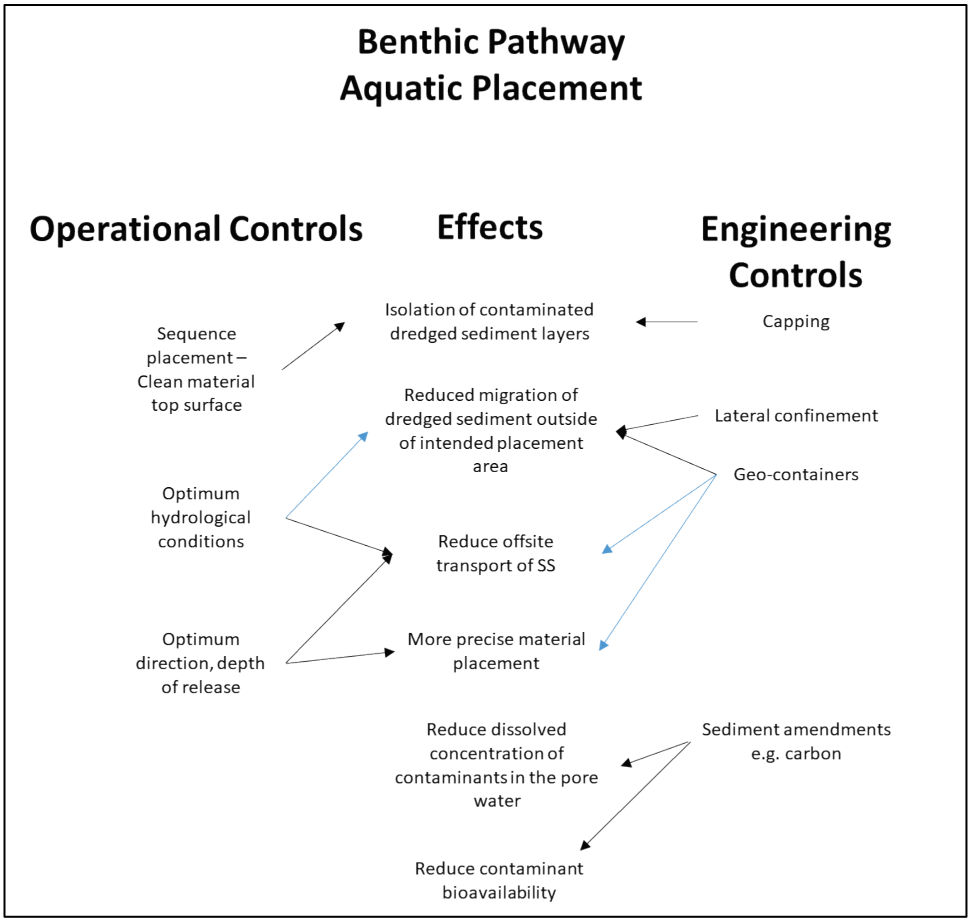


Figure 7-2. Operational and engineering controls relevant to the benthic pathway for aquatic dredged-material placements.



7.5 Controls for upland placement

Exposure pathways relevant to upland disposal (e.g., presented in Tables 6-1a, 6-1b) may include any or all of the following (additional pathways such as direct contact would also be relevant for upland beneficial use placement):

- Effluent discharges
- Runoff discharges
- Volatilization
- Plant and animal uptake
- Leachate.

Placement of material in an upland environment for beneficial use purposes may closely resemble disposal in a confined disposal facility (at least during initial placement operations), with material placed inside a diked area to contain the solids—initially in a semifluid state if hydraulically dredged—and to manage associated water. In some cases, however, upland placement may be unconfined, especially for beneficial use. Particularly in the case of unconfined placement—but sometimes also with confined placement—controls are needed to eliminate a contaminant pathway, reduce or prevent release of contaminants to surrounding water, air, or organisms in direct contact with the sediment, or to treat process streams (e.g., return water or runoff) for dissolved contaminants or SS prior to release to the surrounding environment. Figures 7-3 through 7-7 illustrate various operational and engineering controls that might be considered for upland placement. Definitions for three of the engineering controls referenced in the figures are as follows:

- Package water treatment refers to a combination of water treatment processes—in this case, modular and portable—that may include unit operations for SS removal, contaminant or nutrient sorption or degradation, clarification, and solids dewatering, depending upon the particular treatment need.
- Ponded water amendments—sorbents such as activated carbon may be broadcast for sorption of contaminants from ponded water present in a containment area.
- Sediment amendments—sorbent or reactive materials applied to or mixed with the sediment to sorb, transform, or limit solubility of contaminants released from the sediment to the surrounding pore water and surface water.

The appropriateness, opportunity, and need for controls in either setting (confined or unconfined beneficial use placement) is dependent upon several factors, including the following:

- Condition of the dredged sediment at the time of placement (dry, slurried, or high solids, but flowable) and the need to contain the material within specific boundaries.
- Grain size distribution and other geotechnical properties of the material, such as plasticity and angle of repose (coarse material is stackable and drains readily by gravity; fine material is less stable when wet, may flow, and generally dewaters very slowly).
- Volume of water produced during placement (hydraulic vs. mechanical, discharge directly to the beneficial use placement site vs. following an

intermediate dewatering step at a processing facility or confined disposal facility).

- Contaminant concentrations in the sediment, porewater, and return water.
- Partitioning characteristics of contaminants in the sediment.
- Volatility of contaminants in the sediment and produced water.
- Relevance of an exposure pathway in a specific setting (shallow, potable groundwater vs. deep or nonpotable groundwater underlying the placement site, and planned end-use of the site).
- Particulate losses from dried material may also be of greater concern when material is placed close to public areas and habitation.
- Stability, accessibility, and nature of amendments applied for contaminant reduction/immobilization in sediment or ponded water.

The need for controls and the feasibility of implementation is dependent upon the condition of the material when placed at the site. Hydraulically placed material may not be trafficable for an extended period—typically several months. Distributing materials over the surface of the dredged sediment generally cannot take place until after the material has consolidated substantially.

Figure 7-3. Effluent pathway controls—upland placement.

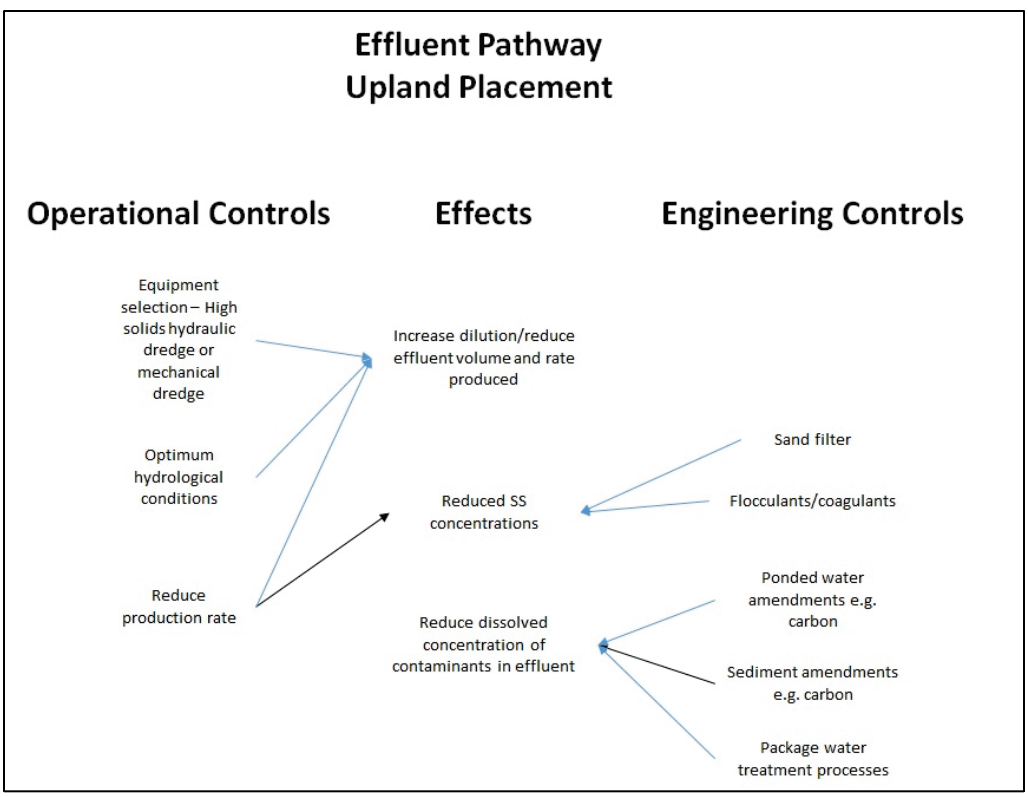


Figure 7-4. Runoff pathway controls—upland placement.

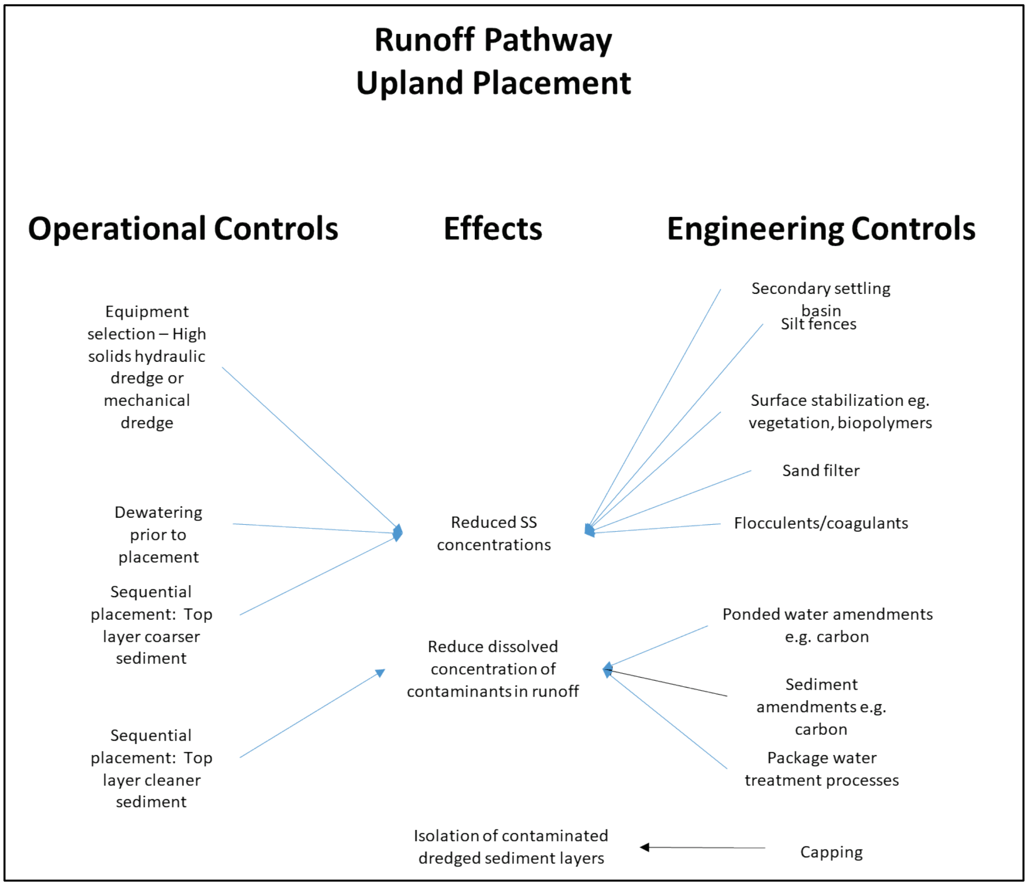


Figure 7-5. Volatilization pathway controls—upland placement.

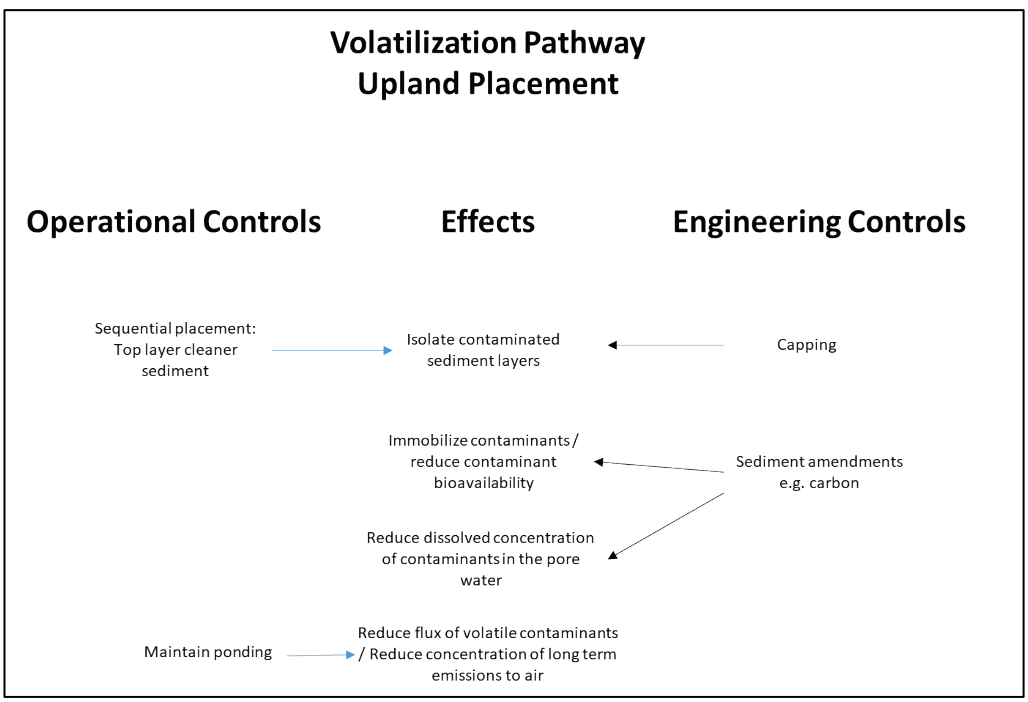


Figure 7-6. Plant and animal uptake pathway controls—upland placement.

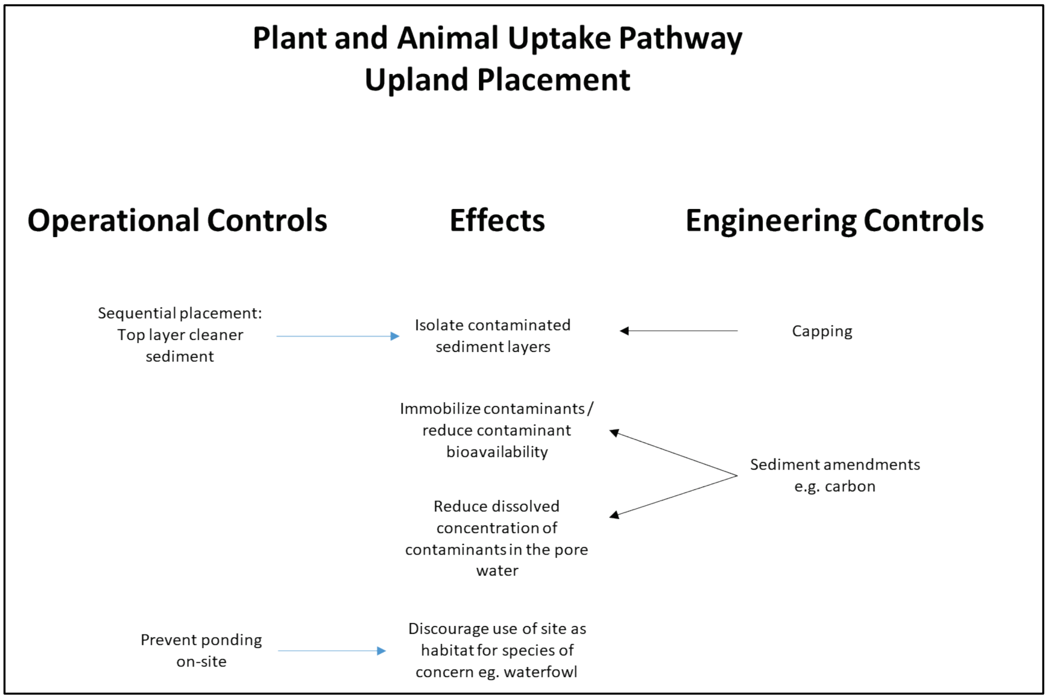
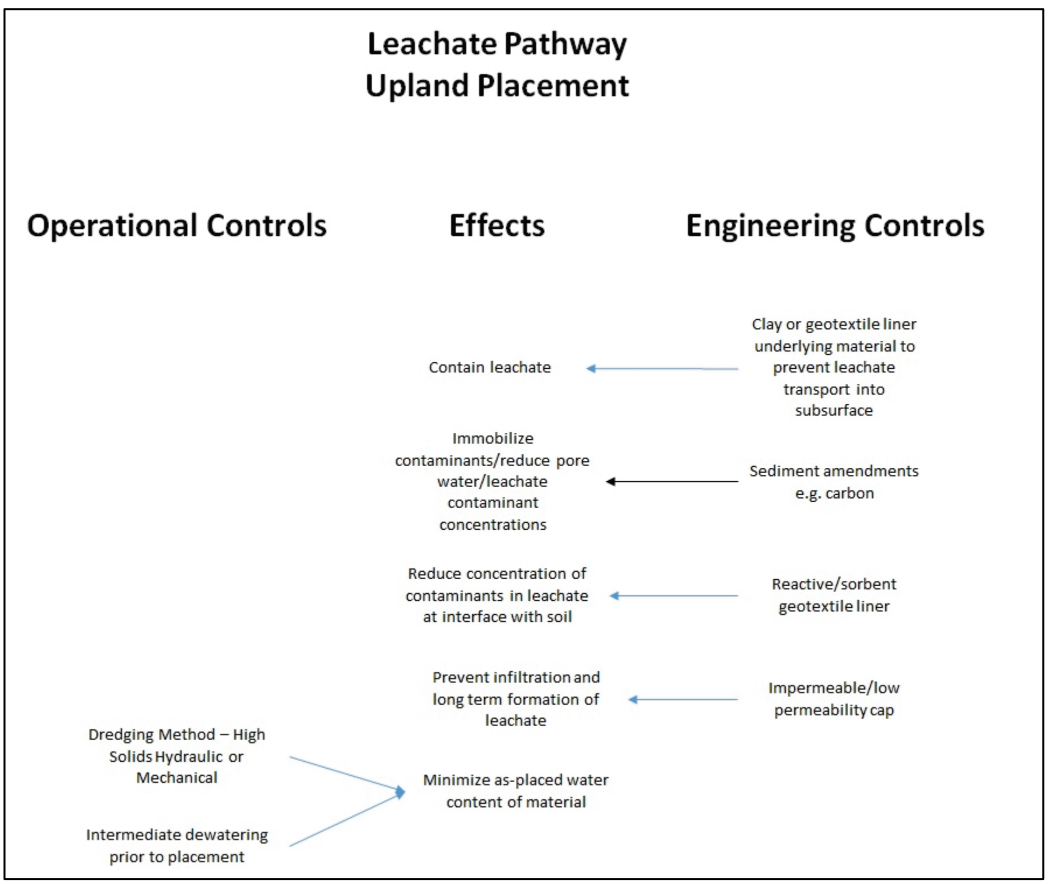


Figure 7-7. Leachate pathway controls—upland placement.



7.6 Use of adaptive management

Adaptive management is a process by which the impact of uncertainties associated with all aspects of a project are considered and managed through development of well-defined decision points and operational response actions (Convertino et al. 2013; Holling 1978; Walters 1986). Through an adaptive management plan, the uncertainties associated with a project and its outcomes are identified before implementation. Decision points and actions are identified as part of the adaptive management plan in response to intermediate outcomes or resolution of various uncertainties so that costs and environmental impacts are minimized and benefits maximized. Operational resilience, reduced project costs, reduced environmental impacts, and greater certainty of meeting project objectives are benefits of a well-developed and executed adaptive management plan, and are consistent with the Engineering With Nature initiative (Section 1.1).

Adaptive management includes explicit measurement metrics and clearly articulated response actions. Adaptive management is intrinsically integrated into the development of risk management objectives, identification of alternatives, site and material characterization, and implementation activities. The uncertainties specific to each exposure risk are identified (e.g., addressing questions about contaminant releases to the water column; the discharge rate; the entrainment of water with the discharge; and dilution modeling predictions of the discharge with the ambient water). The operational and environmental implications of these uncertainties are considered and performance indicators developed to inform management responses.

Uncertainty, performance metric and management actions that might be part of an adaptive management plan are defined as follows:

- Uncertainty: Magnitude of contaminant release from sediment during placement
- Performance metric: Monitoring of evaluation parameters at a specified compliance point and monitoring interval
- Management action: Implementation of pre-selected operational or engineering controls.

The specified alternatives or adaptive management response actions would be triggered by monitoring relevant parameters at specified locations and time intervals and comparing the monitoring results to predefined thresholds/performance specifications, otherwise known as decision points.

An initial alternative may be chosen and the adaptive management plan implemented through a series of incremental stages. Each of these stages provides an opportunity to further inform and refine the process through observation and monitoring. This ensures environmental protectiveness without overly conservative measures, optimizing project design and processes for both environmental effectiveness and economic efficiency.

7.6.1 Components of an adaptive management plan

Adaptive management plans (CEDA 2015; Convertino et al. 2013; Lillycrop et al. 2011) may differ in the number of steps by which the process is formulated, or the specifics of the performance criteria developed to support decision-making, yet all adaptive management plans contain essentially the same major elements within an iterative framework. The following are the major elements of a formalized adaptive management plan. The iterative nature of an adaptive management plan is especially manifested in Steps 7 through 10.

- Step 1: Establish/refine goals and objectives
- Step 2: Construct system model
- Step 3: Develop alternatives
- Step 4: Identify uncertainties
- Step 5: Develop decision points and response actions
- Step 6: Implement selected alternative
- Step 7: Monitor
- Step 8: Interpret monitoring data
- Step 9: Collect feedback and conduct decision-making
- Step 10: Adapt
- Step 11: Complete project
- Step 12: Operation and monitoring of project.

In the context of beneficial use of dredged sediment, adaptive management may facilitate broader use of dredged sediment by addressing the uncertainties associated with potential environmental impacts and assuring compliance with regulatory requirements. Although most sediments in navigation channels are not heavily contaminated, they are not pristine, either. As a result, there may be concerns about contaminant-related impacts associated with in-water beneficial use placement in sensitive areas, even though the sediments have been determined to be suitable through appropriate characterization and testing (e.g., compliance with the CWA). A monitoring program would inform the need for engineering controls to ensure environmental protectiveness; an adaptive management plan would specify the parameters triggering the need for engineering controls and the action to be taken. Conversely, monitoring may show that environmental impacts are less than expected, eliminating the need for engineering controls and resulting in cost savings for the project. For example, if predicted turbidity levels result in the use of silt curtains as part of the design and subsequent monitoring shows that actual turbidity levels within the enclosed area are below levels of concern, the use of silt curtains could potentially be discontinued, streamlining operations and shortening the construction period.

Adaptive management as a formalized process is not a regulatory requirement and does not replace or supplant the regulatory requirements under which the beneficial use of dredged sediment occurs. Adaptive management is simply a structured means for anticipating the uncertainties associated with a project and utilizing information as it becomes available to improve operational efficiency and to acceptably manage risk within the context of regulatory compliance, without unnecessarily conservative measures.

7.6.2 Opportunities for adaptive management in beneficial use placement scenarios

While the focus of this manual is environmental compliance and risk management, adaptive management can address the overall objectives of a dredged sediment beneficial use project: to optimize the process, ensure environmental protectiveness/compliance and maximize anticipated benefits. The objectives of the beneficial use can be ascribed to three major categories: engineering considerations, environmental compliance, and ecological benefits.

The engineering objectives of aquatic beneficial use placement include consideration of functional components related to sediment placement. The final surface elevation of placed sediment often must fall within very specific target elevations. Water depth is a critical factor for establishing desired vegetation and habitat, and since water levels may vary over time, this factor may include a great deal of uncertainty. Accurate modeling of consolidation and precise sediment placement are, therefore, key to meeting overall project objectives and should be emphasized. For example, the uncertainties associated with consolidation rate and placement accuracy may require phasing of dredged sediment placement to allow time for the dredged sediment and foundation materials to undergo the major part of the expected consolidation, followed by a second phase in which shallower, final lifts are placed to achieve the final desired elevation in some cases.

Engineering uncertainties associated with aquatic beneficial use projects that may be addressed by adaptive management include the following:

- Unforeseen variability in physical, geotechnical, and engineering properties of material being placed.
- Uncertainty about placement accuracy (equipment limitations and environmental factors, such as site hydrodynamics).
- Uncertainty about stability of material during placement, when relatively unconsolidated, and following placement (with long-term exposure to erosive processes).
- Uncertainty about the final elevation achievable, due to consolidation of material placed in combination with the resulting consolidation of foundation materials—frequently soft and highly compressible.
- Uncertainty and variation in hydrodynamic transport characteristics of the site, affecting loss of material during placement and long-term stability following placement.
- Uncertainty in water levels and fluctuations over time, including short timeframe changes such as waves and seiches, and longer timeframe changes in water levels including those associated with climate change.

In addition to the magnitude of contaminant release from the placed materials, there are corresponding uncertainties related to environmental compliance, particularly with total suspended solids (TSS)/turbidity levels in the surrounding waters during and following placement. An adaptive management plan provides for operational modifications or engineering controls to conditions that exceed pre-established thresholds, through reduction in production rate, use of silt curtains, or other appropriate means. These response actions, or operational and engineering controls, are discussed for aquatic and upland placements in Tables 7-1 and 7-2 and the supporting text following the tables.

To ensure that the context of these tables is clear, it is worth reiterating that an adaptive management plan prescribes response actions to conditions identified in the field that were not identified or quantifiable in the original site characterization and design phases due to an absence of information or unanticipated variability in materials or site conditions. The field monitoring parameters used to determine the need for controls during placement of dredged material may be different from those used to verify conditions following placement, or to assess long-term impacts, due to the need to obtain information rapidly and inform operations in the field as they are taking place. For example, sampling and re-analyzing sediment as it is placed may be logistically problematic and of little benefit for the following reasons:

- Contaminant concentrations in sediment are not the only determinants of the magnitude of contaminant impact/release from the sediment.
- The presumption is that the original dredged sediment characterization was well-conceived and sufficient to characterize the sediment and associated contaminant releases/impacts to the degree possible—additional field sampling would be subject to sources of uncertainty as the original site characterization samples. In the absence of significant evidence that the pre-construction characterization missed some significant parameter or material variation, confirmatory sediment sampling during or following placement is unlikely to provide new information with which to inform the need for controls beyond those identified during the design phase. Furthermore, analyses are most likely performed on composite samples of the dredged material, which masks the spatial variability. Finally, an important consideration is that ecological effects are typically the result of exposure to the site as a whole rather than to individual sampling point concentrations.
- The turnaround time for sediment sampling and analysis is generally too long to facilitate decision-making during active operations, particularly with respect to the need to employ contaminant controls to address unanticipated contaminant releases to the water column.
- Readily identifiable impacts, such as turbidity or dissolved contaminant concentrations exceeding water quality criteria, may better inform the need for operational decisions/changes in the field.

In contrast, it may be useful to characterize the post-placement surface sediment concentrations (after construction is complete) to serve as a baseline to facilitate identification and correction of any problems that may arise from unforeseen impacts after project construction.

> Operational or engineering controls may be considered in the design phase to manage *anticipated risk*. *Adaptive management* informs the need for controls in *response to conditions realized in the field*.

Many of the same engineering controls—and perhaps some operational controls, such as production rate—may be considered during the design phase and incorporated based on sediment characterization and risk assessment, to manage the anticipated risk. Their use as prescribed in the adaptive management plan takes place within the context of managing unanticipated risks or greater than anticipated impacts identified during or following project construction/material placement. Adaptive management informs the need for controls and the controls to be employed in response to conditions realized in the field. Although the following tables largely reflect field conditions requiring additional controls for risk management, the converse can also be true; if impacts are less than anticipated, some aspects of a design could potentially be simplified or eliminated because they are not needed to achieve adequate risk management. The engineer manual *Dredging and Dredged Material Management* (USACE 2015) provides a comprehensive summary of dredging and material placement equipment and techniques that may assist in addressing the uncertainties listed in the tables below.

Table 7-1. Environmental compliance and engineering uncertainties, monitoring parameters and response actions for beneficial uses of dredged sediment in aquatic placements.

Uncertainty	Potential Sources of Uncertainty	Monitoring Parameters	Decision Point	Response Actions (Operational and/or Engineering Controls)
Environmental Compliance				
Contaminant release during placement	Sediment variability, sampling inadequacy, insufficiently or poorly characterized sediment, unknown or variable sediment-specific containment partitioning behavior	Total or dissolved contaminant concentration in receiving waters at specified compliance point	Monitoring parameter exceeds specified threshold at compliance point, at a magnitude, frequency or duration requiring action as specified in applicable regulations.	• Reduce discharge rate to increase effective dilution within the receiving water. • Skip affected dredged material reaches. • Divert to alternative placement site (with higher dilution capability or water management options) for affected dredging reaches. • Employ containment and/or treatment measures to reduce total or dissolved contaminant concentrations before discharge.
Benthic toxicity/ unacceptable bioaccumulation in upper trophic levels	Sediment variability, sampling inadequacy, insufficiently or poorly characterized sediment, unknown or variable sediment-specific containment partitioning behavior, synergistic effects due to site conditions or contaminants	Receptor tissue concentrations, Evidence of mortality, population reduction, diversity reduction of site-dwelling organisms over time (after recovery from disturbance)	Statistical differences between pre- and post-placement parameters confirming adverse effect of placement	• Add clean capping layer to isolate affected areas. • Add amendments to sediment to sequester contaminants in phases that are not biologically available.
Engineering Uncertainties				
Dredged material settling and transport properties (is the material going where we want it and staying where we put it?)	Representative measurement of grain size, plasticity, bulk density, solids content, specific gravity, organic content	Suspended solids (turbidity) Areal coverage achieved	• Suspended solids concentration exceeds criteria outside mixing zone. • Materials fail to settle or remain in target area. • Materials fail to flow as necessary to achieve delivery over target area.	• Reposition discharge to optimize flow path across placement site. • Modify discharge rate to optimize hydraulic retention time within placement site. • Deploy silt curtains, or temporary sheet pile for SS control. • Employ diffusers to reduce discharge momentum. • Reposition discharge to achieve desired distribution of material.

Table 7-1. Continued.

Uncertainty	Potential Sources of Uncertainty	Monitoring Parameters	Decision Point	Response Actions (Operational and/or Engineering Controls)
Engineering Uncertainties				
Dredged material stability (is the material stacking to the desired elevation in a stable form, resistant to sloughing or further flow?)	Representative measurement of grain size, plasticity, bearing capacity, angle of repose	Bathymetry/surface elevation changes within and outside of the placement area	Solids (mudflow) migrating outside placement area or back into navigation channel	• Perimeter underwater containment dikes to prevent migration off site. • Position discharge sufficiently within the interior of the site to accommodate the angle of repose of the material around the perimeter. • Reduce discharge rate or lift depths to allow strength gain before full loading.
Placement accuracy (are we able to place material accurately and to the desired tolerances?)	Prevailing current direction and velocity, sediment properties influencing settling velocity and transport, equipment capabilities/limitations	Bathymetry changes during and shortly following placement	Target material elevation met, material accumulation outside of intended boundaries	• Impose placement grid over the site to ensure uniform distribution. • Reposition discharge to accommodate influence of prevailing currents. • Modify equipment or rate of delivery for greater vertical and horizontal precision.
Current carrying capacity	Variable current direction and velocity, sediment properties influencing settling velocity and transport	Suspended solids, current velocity and direction	Suspended solids concentrations exceed criteria outside mixing zone, current velocity above or below predefined thresholds.	• Schedule operation for most favorable prevailing conditions. • Reposition discharge to optimize dredged material flow path within placement site. • Modify discharge rate to optimize flow energy/momentum. • Pause dredging temporarily if current velocity falls outside predefined thresholds (higher or lower than desired).

Table 7-1. Continued.

Uncertainty	Potential Sources of Uncertainty	Monitoring Parameters	Decision Point	• Response Actions (Operational and/or Engineering Controls)
Engineering Uncertainties				
Current direction	Variable wind direction and speed, tidal variations	Suspended solids plume and deposition, wind and current direction	Suspended solids concentrations exceed criteria outside mixing zone, wind or current direction outside of tolerances.	• Schedule operation for most favorable prevailing conditions. • Reposition discharge as necessary to accommodate changes in direction of flow. • Modify discharge rate to optimize momentum for current conditions. • Pause dredging temporarily if current direction creates unfavorable condition that operational modification (repositioning of dredged discharge) cannot compensated for.
Site capacity	Combined consolidation of foundation materials and dredged sediment	Bathymetry changes during placement, elevation changes in emergent areas	Target elevations met throughout the site	Divert to alternate site if capacity exceeded.

Table 7-2. Environmental compliance and engineering uncertainties, monitoring parameters and response actions for various beneficial uses of dredged sediment in upland placements.

Uncertainty	Potential Sources of Uncertainty	Monitoring Parameters	Decision Point	Response Actions (Operational and/or Engineering Controls)
Environmental Compliance				
Contaminant releases from sediment to effluent, runoff or leachate	- Sediment variability, sampling inadequacy, insufficiently or poorly characterized sediment, unknown or variable sediment-specific containment partitioning behavior - Attainable dilution attenuation in receiving waters	Contaminant concentrations in effluent, runoff, and porewater	Monitoring parameter exceeds specified threshold at compliance point, at a magnitude, frequency or duration requiring action as specified in applicable regulations.	• Stage placement of material so that cleaner dredged material is on top. • Amend sediment/ponded water with carbon or other suitable media to sequester dissolved contaminants to solid phase so that they will settle out of water column with the sediment solids.
Risk to receptors via direct contact pathway	Sediment variability, sampling inadequacy, insufficiently or poorly characterized sediment, unknown or variable sediment-specific containment partitioning behavior	Receptor tissue concentrations, observed toxicity or necrosis	Statistical increase in receptor tissue concentrations, or other indicators of adverse effect potentially attributable to sediment-associated contaminates.	• Place additional clean material on the surface as a barrier to direct contact. • Amend sediment to sequester contaminants in a nonbioavailable phase.
Risk of bioaccumulation to higher trophic level receptors	Sediment variability, sampling inadequacy, insufficiently or poorly characterized sediment, unknown or variable sediment-specific containment partitioning behavior	Tissue concentrations in receptors of concern at the site	Statistical increase in receptor tissue concentrations	• Place additional clean material on the surface as a barrier to direct contact. • Amend sediment to sequester contaminants in a nonbioavailable phase.
Magnitude of particulate losses from dewatered dredged material	Sediment variability, sampling inadequacy, insufficiently or poorly characterized sediment	Airborne particulates (total dust loading) on-site Evidence of off-site airborne particulate transport	On-site airborne particulate levels exceed worker safety levels Off-site particulate transport exceeds specified threshold	• Dried dredged material may be wetted (sprayed) during placement. • Vegetative or artificial barriers may reduce offsite transport. • Seeding the surface of the material for long term particulate control. • Surface stabilizing treatment, such as biopolymer.

Table 7-2. Continued.

Uncertainty	Potential Sources of Uncertainty	Monitoring Parameters	Decision Point	Response Actions (Operational and/or Engineering Controls)
Engineering Uncertainties				
Dredged material settling properties	Sediment variability, sampling inadequacy, insufficiently or poorly characterized sediment	Slow or insufficient clarification	Effluent TSS exceeds discharge limits after allowance for mixing/dilution in receiving water	• Reduce production rate to increase hydraulic retention time. • Employ flocculants to reduce SS concentrations.
Dredged material dewatering rate	Assumptions about prevailing climatic conditions, including evaporation rate and rainfall	Water content profile, evidence of desiccation (surface cracking)	Rate of dewatering impacting project timeline	Amend sediment with pozzolanic materials to enhance dewatering.
Dredged material compressibility	Sediment variability, sampling inadequacy, insufficiently or poorly characterized sediment	Magnitude of associated settlements	Settlements incompatible with design objectives	Adjust dewatering, placement, and compaction methods to achieve adequate density and reduced compressibility
Dredged material shear strength	Sediment variability, sampling inadequacy, insufficiently or poorly characterized sediment	Shear strength estimated via laboratory and/or field testing	Shear strength insufficient for design objectives	Adjust dewatering, placement, and compaction methods to achieve adequate density and increased shear strength
Dredged material index properties (see Table 4-1 for listing of index properties)	Sediment variability (Presence of anomalous material), sampling inadequacy, insufficiently or poorly characterized sediment Material segregation occurring during placement	Visual monitoring and periodic sampling/testing to assess index properties	Index properties incompatible with design objectives	Amend sediment to achieve compatible index properties (addition or organic material, sand, etc.).
Dredged material compaction characteristics	Same as above	In situ moisture/density testing and index testing	Compaction insufficient to achieve density required for adequate performance	Adjust dewatering, placement, and compaction methods to improve density
Dredged material permeability	Same as above	Permeability	Permeability incompatible with design objectives	Adjust dewatering, placement, and compaction methods to achieve desired permeability.
Dredged material durability (freeze-thaw) and corrosivity	Same as above	Sampling/testing to assess index properties and corrosivity	Durability/corrosivity incompatible with design objectives	Account for material freeze-thaw and corrosivity characteristics in design

7.7 Dredged material performance enhancements

The term *performance enhancement* pertains to the amendment or modification of a sediment to achieve a desired functional characteristic, such as fertility or shear strength, as prescribed by the requirements of the intended end use of the sediment. One example is the development of a product called *manufactured soil*, which involves the blending of dredged sediment, wastewater treatment plant solid residuals, and other amendments to produce a friable, fertile base material for agriculture or gardening (Lee 2001; Lee et al. 2007; Sturgis and Lee 1999; Sturgis et al. 2001; Sturgis et al. 2002). A number of studies were conducted at ERDC to develop this concept and a market for the product.

Another example is the addition of pozzolanic materials to produce processed dredged material to improve the engineering properties of the raw dredged material (e.g., strength, compressibility, and durability). The New Jersey Department of Transportation conducted extensive studies evaluating the relative dose/response for various amendment/sediment mixtures and published a report summarizing the findings of these studies and the resulting “processed dredged material recipe” (Maher et al. 2013). This technique may be appropriate where materials are intended for a structural use, such as construction fill, but may not be appropriate for ecological applications due to changes in the permeability and texture of the sediment, increased pH, contaminants contributed by the pozzolonic materials themselves, and increased mobility/toxicity of certain contaminants under highly alkaline conditions.

7.8 Summary

This section provided an overview of risk management tools and approaches with specific relevance to the beneficial use of sediment and illustrates the relationships between uncertainty, risk, and adaptive management. When environmental evaluations outlined in preceding sections indicate a need for risk management due to anticipated contaminant related impacts, or uncertainties associated with the site conditions or project design, some form of adaptive management may provide a useful approach to ensure a successful project outcome. The procedures outlined here for determining the need for managing risks and uncertainties and the development and use of an adaptive management plan are discussed in more depth in the various references cited, which provide additional details that may be applied to a specific beneficial use project.

8 Summary and Recommendations

The *Great Lakes Beneficial Use Testing Manual* is the first guidance document developed by USACE for evaluating the environmental suitability of dredged material for any beneficial use placement. The procedures in this manual are based on USACE's extensive research and on established procedures for evaluating dredged material management alternatives. Additional evaluations not described in any previous guidance may be necessary to fully characterize the risk and benefits for a proposed dredged material management alternative. This Manual expands upon the environmental evaluation protocols outlined in previous guidance documents by proposing a holistic, risk-based approach which considers environmental suitability based on potentially impacted environments (aquatic or upland), jurisdictional authorities (federal or state), receptors at risk (human or ecological), and pathways of exposure (water, sediment, soil, air, biota, etc.). The *Great Lakes Beneficial Use Testing Manual* draws from the existing testing manuals to the extent possible for beneficial use assessments to avoid unnecessary additional testing or duplication of testing. In this manual, environmental suitability is defined as meeting current ecological and human health protection requirements at both the federal and state levels, based on chemical and biological assessments, and as meeting physical requirements for the beneficial use proposed. The *Great Lakes Beneficial Use Testing Manual* incorporates testing and evaluation guidance from the USACE, USEPA, and state resource/regulatory agencies to provide descriptions of placement options and restrictions. This will ensure that managers of beneficial use projects better understand dredged material suitability.

Although the final language in this Manual reflects USACE's scientific recommendations, resource managers looking to implement beneficial use of dredged material projects at individual sites can develop Quality Assurance Project Plans that provide more site- and project-specific details to their environmental evaluations. These site-specific project plans can also identify the methods for interpreting and evaluating data generated, and the resulting decisions to be made. Site-specific plans should reflect the local, project-specific stakeholder partnership priorities or needs addressing dredged material management at that individual site. Development of this regional approach supports rather than precludes the implementation of site-specific plans for making those shared dredged material management decisions.

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Abbreviations

A/C	Activated Carbon
AOC	Area of Concern
ARCS	Assessment and Remediation of Contaminated Sediment
ATAR	Autoheating Thermophilic Aerobic Reactor
BAF	Bioaccumulation factor
BSAF	Biota-sediment accumulation factors
CDF	Confined disposal facility
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFR	Code of Federal Regulations
COC	Constituent of concern
COPC	Constituent of potential concern
CoSSTEP	Contaminated Sediment Treatment Technology Program
CSM	Conceptual site model
CWA	Clean Water Act
CZMA	Coastal Zone Management Act
CZMP	Coastal Zone Management Program
DMMP	Dredged material management plan
DMMU	Dredged material management unit
DOER	Dredging Operations and Environmental Research
DTPA	Diethylene-triaminepentaacetic acid
Eco-BSL	Ecological Biota Screening Level
Eco-SSL	Ecological soil screening level
EPA	Environmental Protection Agency

ERDC	US Army Engineer Research and Development Center
ESA	Endangered Species Act
EU	European Union
GLTM	*Great Lakes Testing Manual*
HREG	Harbor Resource Environmental Group
ITM	Inland Testing Manual
ITRC	Interstate Technology and Regulatory Council
LOE	Line of evidence
MEP	Multiple extraction procedure
MOD	Magnitudes of difference
MPRSA	Marine Protection, Research, and Sanctuaries Act
NAPL	Non-aqueous phase liquids
NEPA	National Environmental Policy Act
NHPA	National Historic Preservation Act
NJDOT	New Jersey Department of Transportation
NPDES	National pollutant discharge elimination system
NPOC	Non-polar organic contaminant
NY/NJ	New York/New Jersey
PAH	Polycyclic aromatic hydrocarbon
PCB	Polychlorinated biphenyl
RCRA	Resource Conservation and Recovery Act
REF	Reference
ROD	Record of Decision
RSL	Regional screening level
S/S	Solidification/stabilization
SITE	Superfund Innovative Technology Evaluation

SMCRA	Surface Mining Control and Reclamation Act
SS	Suspended solids
SVOC	Semi-volatile organic compound
TBP	Theoretical bioaccumulation potential
TCLP	Toxicity characteristic leaching procedure
TOC	Total organic carbon
TRV	Toxicity Reference Value
TSCA	Toxic Substances Control Act
TSS	Total suspended solids
USACE	US Army Corps of Engineers
USEPA	US Environmental Protection Agency
UTM	Upland Testing Manual
WQS	Water quality standards
WRDA	Water Resources Development Act

Appendix A: Sources of Soil and Sediment Background (Reference) Concentrations in the Great Lakes States

Table A-1. Available data describing ambient/background concentrations of chemicals in the Great Lakes region.

Location	Medium	Citation	Metals ?	PAH ?	Other organics?
New York	Soil	New York State Department of Environmental Conservation 2005, Brownfield and Superfund Regulation, 6 NYCRR Part 375 - Environmental Remediation Programs, Technical Support Document Appendix D, Concentrations of Selected Analytes in Rural New York State Surface Soils: A Summary Report on the Statewide Rural Surface Soil Survey, August. http://www.dec.ny.gov/docs/remediation_hudson_pdf/appendixde.pdf	Yes	Yes	VOCs, SVOCs*, Pesticides, PCBs
		Azzolina, N.A.; Kreitinger, J.P.; Skorobogatov, Y.; and Shaw, R.K. 2015. Background Concentrations of PAHs and Metals in Surface and Subsurface Soils Collected throughout Manhattan, New York. Environmental Forensics 17:4, 294-310	Yes	Yes	No
Ohio Erie, Lorain, Lucas and Cuyahoga counties†	Soil	Ohio Environmental Protection Agency 2020. Evaluation of Background Metal Soil Concentrations. Developed in Support of the Ohio Voluntary Action Program. Ohio EPA Division of Emergency and Remedial Response. http://epa.ohio.gov/portals/30/vap/docs/Evaluation%20of%20Background%20Metal%20Soil%20Concentrations.pdf	Yes	No	No
Ohio	Sediment	Ohio Environmental Protection Agency Division of Emergency and Remedial Response 2018. Guidance for Conducting Ecological Risk Assessments. DERR-00-RR-031. http://www.epa.state.oh.us/portals/30/rules/RR-031.pdf	Yes	No	No
Michigan	Topsoil clay sand	Michigan Department of Environmental Quality, Waste and Hazardous Materials Division, 2005, updated 2015. Michigan Background Soil Survey. https://www.michigan.gov/documents/deq/deq-rrd-MichiganBackgroundSoilSurvey_495685_7.pdf	Yes	No	No

* Semi-volatile organic compound

† Ohio counties along Lake Erie for which county-specific background values have been established.

Table A-1. Continued.

Location	Medium	Citation	Metals?	PAH?	Other organics?
Michigan	Soil	Cleanup Criteria Requirements for Response Activity (Formerly the Part 201 Generic Cleanup Criteria and Screening Levels) 2018. Table 2, Table 3. https://www.michigan.gov/egle/0,9429,7-135-3311_4109-251790--,00.html	Yes	No	No
Illinois, Chicago area, metropolitan areas, non-metropolitan areas	Soil	Illinois Pollution Control Board 2013. Environmental Protection Act Subtitle G: Waste Disposal Chapter I: Pollution Control Board Subchapter f: Risk Based Cleanup Objectives Part 742. Tiered Approach to Corrective Action Objectives. Appendix A Table G (metals), Appendix A Table H (PAHs). ftp://www.ilga.gov/jcar/admincode/035/03500742sections.html	Yes	Yes	No
		Electric Power Research Institute (EPRI) 2004. Polycyclic Aromatic Hydrocarbons (PAHs) in Surface Soil in Illinois: Background PAHs. Report no. 1011376. Palo Alto, CA, Milwaukee, WI, and Springfield, IL: EPRI, We Energies, and IEPA.	No	Yes	No
		Tetra Tech, 2003. Polynuclear Aromatic Hydrocarbon Background Study, City of Chicago, Illinois. City of Chicago, Chicago, IL.	No	Yes	No
Wisconsin	Soil	United States Geological Survey (USGS) 2012, Distribution and variation of arsenic in Wisconsin surface soils, with data on other trace elements: USGS Scientific Investigations Report 2011–5202, Prepared in cooperation with the US Department of Agriculture, Natural Resources Conservation Service, Wisconsin Department of Natural Resources, and Wisconsin Department of Health Services http://pubs.usgs.gov/sir/2011/5202/	Yes	No	No
Minnesota	Soil	Minnesota Pollution Control Agency (MPCA) 2021b, Background Threshold Value Evaluation. Remediation and Environmental Analysis Outcomes Divisions https://www.pca.state.mn.us/waste/risk-based-site-evaluation-guidance	Yes	Yes	Dioxin, benzo(a) pyrene equivalents

Table A-1. Continued.

Location	Medium	Citation	Metals?	PAH?	Other organics?
Minnesota	Sediment	Minnesota Pollution Control Agency (MCPA) 2016, Ambient Sediment Quality Conditions in Minnesota https://www.pca.state.mn.us/sites/default/files/tdr-g1-19.pdf	Yes	Yes	PAHs, Biphenyls, PCBs, Dioxins/ Furans (PCDD/F) PBDEs
		LimnoTech. 2016. Final Memorandum. Upper Tolerance Limits of Least Impacted Sediment Assessment Areas in the St. Louis River Area of Concern. From: Tim Towey, LimnoTech, Inc, Revised November 2016.	Yes	Yes	PCBs, Dioxins/ Furans (PCDD/F)PBDEs and chlorinated pesticides
		US Army Engineer Research and Development Center (ERDC). 2017. Technical Analysis Memorandum: Evaluation of Polychlorinated Dibenzo-p-dioxins/dibenzofuran Concentrations and Toxic Equivalents in Habitat Restoration Sites and Other Sediment Assessment Areas within the Minnesota Portion of the St Louis River Area of Concern.	No	No	Dioxins/ Furans (PCDD/F)
Nationwide	Streambed sediment	United States Geological Survey (USGS) 2008, The National Geochemical Survey – Database and Documentation. Open File Report 2004-1001, county-by-county averages. http://tin.er.usgs.gov/geochem/doc/home.htm	Yes	No	No

Table A-2. Comparison of Great Lakes State-specific background concentrations for soil. All values are mg/kg.

Compound	CAS Numbers	New York[1]	Ohio - Cuyahoga County[2]	Ohio - Lorain County[2]	Ohio - Erie County[2]	Ohio - Lucas County[2,3]	Michigan 2005[4]	Michigan 2015[5]	Illinois[6] - Within Metropolitan Areas	Illinois[7] - Outside Metropolitan Areas	Wisconsin[8]	Minnesota[9]
<u>Metals</u>												
Aluminum	7429-90-5	16,400	NA	11,521	11,300	NA	6,900	11,237	9,500	9,200	28,721	19,000
Antimony	7440-36-0	NA	NA	NA	NA	NA	NA	NA	4	3.3	NA	NA
Arsenic	7440-38-2	13	24	19.1	16.7	9.7, 2.42	5.8	14.9	13	11.3	8.3	9
Barium	7440-39-3	350	98.9	82.6	111	90.1, 41.0	75	118	110	122	364	210
Beryllium	7440-41-7	1.1	NA	0.61	0.85	NA	NA	0.69	0.59	0.56	NA	NA
Cadmium	7440-43-9	2.5	0.834	NA	NA	NA	1.2	2	0.6	0.5	1.07	NA
Chromium (Total)	7440-47-3	30	21.1	18.2	17.6	23.2, 7.14	18	32.7	16.2	13	43.5	27
Cobalt	7440-48-4	12.8	NA	15.7	13.23	NA	6.8	7	8.9	8.9	22	12
Copper	7440-50-8	33	NA	26	42	NA	32	57.1	19.6	12	35.4	NA
Iron	7439-89-6	26,200	NA	36,177	27,567	NA	12,000	23,185	15,900	15,000	34,314	29,000
Lead	7439-92-1	63	51.7	29.5	29.6	17, 12.2	21	37.1	36	20.9	51.6	NA
Manganese	7439-96-5	1,600	NA	1,504	1,164	NA	440	1,302	636	630	2937	NA
Mercury		0.18	0.097	0.0513	0.071	0.045	0.13	0.27	0.06	0.05	NA	NA
Nickel	7440-02-0	25	NA	30.4	22.5	28.5, 6.3	20	24.9	18	13	30.8	NA
Selenium	7782-49-2	3.9	0.943	1.79	1.1	NA	0.41	1.3	0.48	0.37	NA	NA
Silver	7440-22-4	0.7	NA	NA	NA	NA	1	1.4	0.55	0.5	NA	NA
Thallium	7440-28-0	NA	NA	0.966	0.6	0.44, 0.067	NA	<1.0	0.32	0.42	NA	0.29
Vanadium	7440-62-2	33	NA	25.5	22.7	NA	NA	32.1	25.2	25	85	62
Zinc	7440-66-6	109	NA	73.6	71.1	NA	47	115	95	60.2	150	NA

Table A-2. Continued.

PAHs												
Acenaphthene	83-32-9	NA	NA	NA	NA	NA	NA	NA	0.13	0.04	NA	NA
Acenaphthylene	208-96-8	NA	NA	NA	NA	NA	NA	NA	0.07	0.04	NA	NA
Anthracene	120-12-7	NA	NA	NA	NA	NA	NA	NA	0.4	0.14	NA	NA
Benzo(a)anthracene	56-55-3	1	NA	NA	NA	NA	NA	NA	1.8	0.72	NA	NA
Benzo(a)pyrene	50-32-8	1	NA	NA	NA	NA	NA	NA	1.3 / 2.1	0.98	NA	2
Benzo(b)fluoranthene	205-99-2	1	NA	NA	NA	NA	NA	NA	2.1	0.7	NA	NA
Benzo(ghi)perylene	191-24-2	NA	NA	NA	NA	NA	NA	NA	1.7	0.84	NA	NA
Benzo(k)fluoranthene	207-08-9	0.8	NA	NA	NA	NA	NA	NA	1.7	0.63	NA	NA
Chrysene	218-01-9	1	NA	NA	NA	NA	NA	NA	2.7	1.1	NA	NA
Dibenzo(a,h)anthracene	53-70-3	NA	NA	NA	NA	NA	NA	NA	0.42	0.15	NA	NA
Fluoranthene	206-44-0	0.087	NA	NA	NA	NA	NA	NA	4.1	1.8	NA	NA
Fluorene	86-73-7	NA	NA	NA	NA	NA	NA	NA	0.18	0.04	NA	NA
Indeno(1,2,3-cd)pyrene	193-39-5	NA	NA	NA	NA	NA	NA	NA	1.6	0.51	NA	NA
Naphthalene	91-20-3	0.014	NA	NA	NA	NA	NA	NA	0.2	0.17	NA	NA
Phenanthrene	85-01-8	NA	NA	NA	NA	NA	NA	NA	2.5	0.99	NA	NA
Pyrene	129-00-0	0.17	NA	NA	NA	NA	NA	NA	3	1.2	NA	NA

Notes

Please see Table A-1 for a complete listing of references for soil background values in the Great Lakes States.

[1] Identified as the unrestricted use Soil Cleanup Objective [6NYCRR375-6.8(a)], or 95th percentile from NY State Habitat Soil Survey Table D.1 of 6 NYCRR375, Appendix D
[2] Ohio county specific values. The background mean plus two standard deviations is the maximum allowable limit or upper limit for normally distributed data. If the data follows a lognormal, nonparametric, or gamma distribution, the upper limit was calculated with USEPA's ProUCL program to determine the 95% upper prediction limit based on the best fit distribution.
[3] Lucas county <50% sand, >50% sand
[4] Michigan Statewide default background value, except for arsenic, for which regional criteria have also been developed. 2005 Michigan Background Soil Survey values are incorporated into Cleanup Criteria Requirements for Response Activity (Formerly the Part 201 Generic Cleanup Criteria and Screening Levels)
[5] Values derived from Topsoil table in the Michigan Background Soil Survey, updated 2015. Values are combined statewide data based on the mean plus two standard deviations of the data distribution.
[6] Illinois values from within metropolitan areas. Note that for PAHs, Illinois has an additional data set for Chicago metropolitan area only; values for benzo(a)pyrene are for the City of Chicago and other metropolitan areas. Values are the upper tolerance limits of the data set.
[7] Illinois values from outside metropolitan areas. Values are the upper tolerance limits of the data set.
[8] Wisconsin background threshold values are provided in their residual contaminant levels (RCL) tables and are the maximum levels in surface soil.

[9] Minnesota statewide background threshold values finalized in April 2021.

Appendix B: Great Lakes State Environmental Guidance and Regulations for Beneficial Use of Dredged Material

New York

1. Responsible State agency(s) for upland beneficial use:

New York State Department of Environmental Conservation (NYSDEC), Division of Materials Management

2. Point of contact within the agency:

- Kathleen Prather, Division of Materials Management
    - (518) 402-8678
    - kathleen.prather@dec.ny.gov

3. Process for determining suitability of dredged material for upland beneficial use:

Dredged materials are usually reviewed in accordance with NYSDEC's Solid Waste Management Facilities Regulations, 6 NYCRR Part 360-369, specifically for beneficial use of any material: Subdivision 360.12(e). Once reviewed pursuant to these regulations, if suitable for upland beneficial use, the dredged material in question will be granted a beneficial use determination (BUD). The BUD may specify use of the dredged material at a specific location as fill, cover, topsoil, or aggregate, or may allow its general sale or distribution in one or more of these uses. Note that two pre-determined beneficial uses (no review required by NYSDEC) can be found in Subdivision 360.12(c); one is for coarse dredged materials with low organic carbon; the other is for excavated clay, till or rock that may be dug or blasted to deepen channels on some projects, provided these materials are kept separate from overlying sediment.

4. Applicability of State regulations to upland beneficial use of dredged material:

NYSDEC's solid waste regulations apply to management of dredged materials, including disposal or beneficial use. An exclusion exists in 6 NYCRR Part 360.2(a)(3)(xi) for dredged materials which are managed under a NYSDEC Dredging Permit or Clean Water Act 404 Water Quality Certification. However, most upland placement of dredged material is not managed under dredging permits but rather through BUDs granted pursuant to 6 NYCRR 360.12(e).

Other regulations which may apply to dredged materials include the 6 NYCRR Part 370 Series, which includes identification of hazardous waste and its treatment, management and disposal, and also remediation of Superfund and Brownfield sites.

5. State-specific soil criteria (from above State regulation, or other):

Recent revisions to beneficial use regulations in Sections 360.12 and 360.13, incorporate soil cleanup objectives (SCOs) in 6 NYCRR Part 375, Environmental Remediation Programs Regulations, to evaluate soils and soil-like materials such as dredged material in soil-like uses, especially as fill and cover or topsoil. Dredged materials are evaluated on a case-specific basis, but if meeting new "General Fill" criteria, i.e., Public Health-Residential Land Use and Groundwater Protection SCOs, the BUD may allow general sale or distribution of dewatered dredged material in place of fill, cover or topsoil.

6. Long-term web links providing this information:

Beneficial Use Determinations: https://www.dec.ny.gov/chemical/8821.html

DEC Regulations Portal (Chapter IV-Quality Services):
https://www.dec.ny.gov/regs/2491.html

Pennsylvania

Please see the information provided in Tables B.1-1 and B.1-2. No other guidance on using dredged material for upland beneficial use has been provided by Pennsylvania.

Ohio

Please see the information provided in Tables B.1-1 and B.1-2.

1. Responsible State agency for upland beneficial use:

   - Ohio Environmental Protection Agency

2. Point of Contact within the agency:
   - : Vanessa Steigerwald Dick, Ph.D.
     Environmental Scientist

     Standards and Technical Support and Dredge Program

     Division of Surface Water

     Northeast District Office

     Phone: 330-963-1219

     Vanessa.SteigerwaldDick@epa.ohio.gov

3. Process for determining suitability of dredged material for upland beneficial use:
   - Ohio EPA's upland placement of dredge material process, for material excavated or dredged from a federal navigational channel (Lake Erie) during harbor or navigation maintenance activities, can be found at Ohio Environmental Protection Agency Division of Material and Waste Management 2016 Ohio Administrative Code (OAC) Chapter 3745- 599: Beneficial Use and Harbor Sediment Authorization rules (Final) and associated general and individual permits for material excavated or dredged from a federal navigational channel during harbor or navigation maintenance activities (Effective 2/25/2019)

Michigan

1) Responsible State agency(s) for upland beneficial use:

- Michigan Department of Environment, Great Lakes, and Energy (formerly Michigan Department of Environmental Quality – Office of Waste Management and Radiological Protection)

2) Point of contact within the agency:

- Duane Roskoskey
- RoskoskeyD@Michigan.Gov
- 517-582-3445

3) Process for determining suitability of dredged material for upland beneficial use:

- Michigan's dredge procedure can be found at http://www.michigan.gov/documents/deq/deq-policy-09-018_414753_7.pdf .
- Uncontaminated dredge material is not a regulated solid waste and can be used upland, without restriction from the solid waste regulations. If a material tests over 90% sand (except for material from the Tittabawassee and Saginaw Rivers) it is considered to be uncontaminated.
- Dredge material that does not meet the 90% sand criteria or is from the two rivers listed above must test the material for the contaminants found at http://www.michigan.gov/documents/deq/lwm-dredge-criteria-charts_254601_7.pdf . Any dredge material that meets the Part 201, Environmental Remediation, generic residential criteria is considered to be uncontaminated and can be used upland, without restriction from the solid waste regulations.
- Dredge material that fails the criteria listed above can still be used upland, but under certain restrictions, which include:
    - Being placed upland adjacent to where it was dredged
    - Capped with clean soil
    - Placing a deed restriction on the property

4) Applicability of State regulations to upland beneficial use of dredged material.

- Contaminated soils are considered to be a solid waste under Part 115, Solid Waste Management, of the Natural Resources and Environmental Protection Act, 1994 PA 451, as amended. The solid waste statute can be found at http://www.legislature.mi.gov/documents/mcl/pdf/mcl-451-1994-ii-3-115.pdf .

5) State-specific soil criteria (from above State regulation, or other).

- The Part 201 Residential criteria can be found at http://www.michigan.gov/documents/deq/deq-rrd-Rules-Table1GroundwaterResidentialandNon_447070_7.pdf and http://www.michigan.gov/documents/deq/deq-rrd-Rules-Table2SoilResidential_447072_7.pdf .

6) Long-term web links providing this information.

- Dredging page - http://www.michigan.gov/deq/0,4561,7-135-3307_29692_24403-10906--,00.html
- Solid Waste page - http://www.michigan.gov/deq/0,4561,7-135-3312_4123---,00.html

Indiana

1) Responsible State agencies for upland beneficial use:

- Indiana Department of Environmental Management (IDEM)
- Indiana Department of Natural Resources (IDNR)

2) Point of contact within the agencies:
Indiana Department of Environmental Management (IDEM)

- Marty Maupin
    - mmmaupin@idem.in.gov
    - 317-233-2471
- Anne Remek
    - aremek@idem.in.gov
    - 317-233-0447

Indiana Department of Natural Resources (IDNR)

- Steve Davis
    - sdavis@dnr.IN.gov
    - 219-874-8316

3) Process for determining suitability of dredged material for upland beneficial use:

To evaluate potential chemical-related risks arising from future use of dredged materials placed in upland areas, IDEM uses, at a minimum, its non-rule policy document, Remediation Closure Guide [WASTE-0046-R1-NPD] available at https://www.in.gov/idem/cleanups/files/remediation_closure_guide.pdf . This document is strictly guidance and not regulation. Responsible parties may propose approaches that do not appear in the guidance.

4) Applicability of State regulations to upland beneficial use of dredged material:

The statutory basis for the closure guide is IC 13-12-3-2 and IC 13-25-5-8.5.

5) State-specific soil criteria:

IDEM has published several sets of screening levels. Annual updates began in 2012. Some of these screening levels appear in each table are suitable for evaluating risk from soil direct contact and others are for ground water or vapor. The Remediation Closure Guide referenced above contains human health risk screening levels. Current and past versions of the tables are at https://www.in.gov/idem/cleanups/2392.htm . Again, these tables are guidance and not regulation. By statute, responsible parties may propose site specific screening levels, including those that take into account institutional and engineering controls. For ecological screening levels, the USEPA Region 4 Regional Ecological Risk Assessment Supplemental Guidance may be used.

https://www.epa.gov/risk/regional-ecological-risk-assessment-era-supplemental-guidance

Illinois

1) Responsible State agency(s) for upland beneficial use.

Upland beneficial use for mechanically dredge material placed away from surface water and with no discharge to waters of the State does not require a permit from Illinois EPA. It is the responsibility of the generator of the dredge material and user of the dredge material to determine that the upland beneficial use will not cause violations of the Illinois Environmental Protection Act or the regulations under the Act. The Illinois Environmental Protection Agency administers inspection, compliance and enforcement programs regarding violations of the Illinois Environmental Protection Act and regulations under the Act.

2) Point of contact within the agency(s).

For dredging projects with discharges to waters of the State or hydraulic dredging projects – Facility Evaluation Unit- Permit Section – Division of Water Pollution Control- Illinois EPA – Ph. 217-782-0610. For other types of projects, the Facility Evaluation Unit and/or the Permit Section of Illinois EPA's Bureau of Land (Ph.-217/524-3300) may be an appropriate point of contact.

3) Process for determining suitability of dredged material for upland beneficial use.

Upland beneficial use for mechanically dredged material placed away from surface water and with no discharge to waters of the State does not require a permit from Illinois EPA. It is the responsibility of the generator of the dredge material and user of the dredge material to determine that the upland beneficial use will not cause violations of the Illinois Environmental Protection Act or the regulations under the Act.

For dredge materials that are hydraulically dredged or discharged to waters of the state, testing procedures are outlined in 35 Ill. Adm. Code 395. A sieve size analysis is required and additional testing (supernatant or elutriate for particular parameters) is required when the material is either fine grained, the dredge material is contaminated or discharge of the dredge material may cause water quality violations. Mechanically dredged material that is contaminated, for instance from a cleanup site, may be subject to state permit requirements for containment/treatment of the material or if taken to a landfill subject to landfill regulations administered by the IEPA Bureau of Land. The Illinois EPA reviews projects when required to do so under its permitting authorities on a case by case basis.

4) Applicability of State regulations to upland beneficial use of dredged material.

Upland beneficial use for mechanically dredge material placed away from surface water and with no discharge to waters of the State does not require a permit from Illinois EPA. For other types of projects, a 401 water quality certification issued under 35 Ill. Admin. Code 395 and the Clean Water Act and or a state permit must be obtained.

5) State-specific soil criteria (from above State regulation, or other).

The Illinois EPA does not have criteria specific to beneficial use of dredge material to upland areas when there is no discharge of the dredge material to surface waters. However in evaluating projects with a discharge to surface waters, hydraulic dredging projects or contaminated dredge material placement the following criteria may be used to determine suitability of the material for placement/use.

35 Ill. Adm. Code Subtitle C Water Quality Standards and related regulations 35 Ill Adm. Code 395 certification regulations.

EPA-823-R-97-008, September 1997 sediment screening criteria

Ingersoll, C.G., MacDonald, D.D., Wang, N., Crane, J.L, Field, L.J., Haverland, P.S., Kemble, N.E., Lindskoog, R.A., Severn, C., Smorog, D.E., 2000, Prediction of sediment toxicity using consensus-based freshwater sediment quality guidelines: June 2000: EPA 905/R-00/007

MacDonald, D.D., Ingersoll, C.G., Berger, T.A., 2000, Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems: Archives of Environmental Contamination and Toxicology 39(1):20-31

35 Ill. Adm. Code 742 –Tiered Approach to Cleanup Objectives Background soils data

Background lake and stream sediment data. 35 Ill. Adm. Code 620 groundwater standards

The Groundwater Standards and Subtitle C water quality standards are required to be met. The other criteria noted above may be used in evaluating if a project will meet the groundwater or Subtitle C Water Pollution regulations.

In addition for beneficial use of sand for beach nourishment in Lake Michigan asbestos criteria is found in this document - Illinois Beach State Park (IBSP): Determination of Asbestos Contamination in Beach Nourishment Sand Final Report of Findings, University of Illinois – Chicago - June 20, 2006

6) Long-term web links providing this information.

Subtitle C Parts 301, 302, 303, 304, 309, 395 at
https://pcb.illinois.gov/SLR/IPCBandIEPAEnvironmentalRegulationsTitle35

https://pcb.illinois.gov/documents/dsweb/Get/Document-33352/

https://pcb.illinois.gov/documents/dsweb/Get/Document-33354/

https://pcb.illinois.gov/documents/dsweb/Get/Document-16952/

https://pcb.illinois.gov/documents/dsweb/Get/Document-33356/

https://pcb.illinois.gov/documents/dsweb/Get/Document-12064/

Tiered Approach Cleanup Objectives – 35 Ill. Adm. Code 742 at
https://pcb.illinois.gov/documents/dsweb/Get/Document-38408/

Groundwater Standards – 35 Ill Adm. Code 620 at
https://pcb.illinois.gov/documents/dsweb/Get/Document-33425/

Attorney General's Asbestos Task Force Study - Illinois Beach State Park (IBSP): Determination of Asbestos Contamination in Beach Nourishment Sand Final Report of Findings http://www.lb7.uscourts.gov/documents/09-15353.pdf

Wisconsin

Please see the information provided in Tables B.1-1 and B.1-2. No other guidance on using dredged material for upland beneficial use has been provided by Wisconsin.

Minnesota

1) Responsible State agency for upland beneficial use:
- Minnesota Pollution Control Agency

2) Point of contact within the agency:
- Emily Schnick
  - Emily.Schnick@state.mn.us

3) Process for determining suitability of dredged material for upland beneficial use:

Environmental Risk Assessment
1. Physical Test
When 93% or more of the dredged material is retained on a #200 sieve the material is considered management level 1. No further testing required.
2. Review of past industrial activities/sources of pollutants
Further information included in the guidance document, 'Managing dredge materials in the State of Minnesota' http://www.pca.state.mn.us/index.php/view-document.html?gid=12959
3. Chemical Analysis (baseline list and those pollutants identified in #2)
Further information included in the guidance document, 'Managing dredge materials in the State of Minnesota'

Management Levels
Based on the results of completed sediment characterization, the dredged material is categorized into three management levels. The management level of a dredged material dictates the appropriate disposition of the material. If chemical analysis is done because sediments do not meet the definition of sand (200# sieve), management levels are determined by comparison to the appropriate Soil Reference Values (SRVs). SRVs are used to evaluate potential human health risks from soil exposure.

Dredged Material is categorized into three Management Levels:

▪ Level 1: suitable for use or reuse on properties with a residential or recreational use category (meet the limits of Residential/Recreational SRVs);
▪ Level 2: suitable for use or reuse on properties with an industrial use category (meet the limits of Industrial SRVs); and,
▪ Level 3: Contact MPCA staff for additional information on regulatory requirements for disposal. Characterized as having significant contamination, as demonstrated by one or more analyte concentrations being greater than the Level 2 SRV. This material is not suitable for reuse and is typically landfilled.

4) Applicability of State regulations to upland beneficial use of dredged material:
• Dredged material includes material that is excavated at or below the Ordinary High Water Level (OHWL) of water basins, watercourses, public waters, or public waters wetlands, as defined by Minn. Stat. ch. 103G.005. Examples of dredged material include sediment from the maintenance of dams and other hydraulic control structures; sediment from habitat improvement projects and other construction activities; sediment from the navigational dredging for shipping cargo and freight in Minnesota's commercial ports; and, dredge projects that require the removal of sediment from Minnesota waters at marinas and recreational boating areas.
• Dredged material is defined as a "waste" and "other waste material" by Minn. Stat. 115.01. It is therefore the duty of the Minnesota Pollution Control Agency (MPCA), as set forth in Minn. Stat. 115.03, subd. 1(e), to regulate the management and disposal of dredged material.
• The MPCA's permitting role is the regulation of the disposal of the dredged materials, not the dredge activity itself.

A State Disposal System (SDS) permit may be required to store, treat, dispose and/or reuse dredged materials on-land in Minnesota if the dredged material originates from pollution remediation projects or from navigational channels and associated bays, harbors, docks and marinas from the following areas and is greater than 3,000 cubic yards:

• Mississippi River downstream of River Mile 857.6 (which is approximately at the Soo Line Rail crossing near St. Anthony Parkway in Minneapolis)
• Minnesota River downstream of River Mile 27 (which is approximately two miles upstream of the CSAH 101 crossing at Shakopee)
• St. Croix River downstream of River Mile 26 (which is approximately three miles upstream of the East Chestnut Street crossing at Stillwater)

- St. Louis River downstream of the State Highway 23 crossing
- St. Louis bay or Duluth/Superior Harbor
- Out of state projects

If the project is not located in the areas above, an SDS permit for the management of dredged material is not required. However, the project proposer is recommended to follow the guidance document, 'Managing dredge materials in the State of Minnesota' and submit a notification form to the MPCA.

5) State-specific soil criteria (from above State regulation, or other):
SRVs and sieve test

6) Long-term web links providing this information. Dredge Materials Management website: https://www.pca.state.mn.us/water/dredged-materials-management

Table B.1-1. Comparison of development of generic risk-based soil concentrations by the Great Lakes states for protection of human health.

State Name	Cancer Risk Limit[1]	Follows USEPA Toxicity Criteria Hierarchy[2]?	Modifies USEPA RSL[3] Exposure Assumptions?	Includes Specific Guidance for Recreational Scenario?	Includes Food Uptake/Consumption Pathways (Agricultural or Unrestricted Scenario)?	Includes Background Values for Metals and PAHs?	Year of Last Update	Expected next update
New York	1E-06	No[4]	Yes[5]	No	Yes	Yes, metals, PAHs[6]	2006	NA[7]
Pennsylvania	1E-05	Yes	Yes[8]	No	No	No	2018	Periodic[9]
Ohio	1E-06	Yes	No	Yes [10]	No	Yes, metals[11]	2020	Bi-annually
Michigan	1E-05	Yes	Yes	Yes[12]	No	Yes, metals[13]	2018	Periodic
Indiana	1E-05	Yes	Yes[14]	Yes	No	No	2021	Yearly
Illinois[15]	1E-06	Yes[16]	No[17]	No	No	Yes, metals, PAHs	2013	Periodic
Wisconsin	1E-06	Yes	No[18]	Yes[19]	No	Yes, metals	2018	Bi-annually
Minnesota	1E-05	No[20]	Yes[21]	Yes[22]	No[23]	Yes, metals, PAHs	2021[24]	TBD

1 In addition to limiting cancer risks, all of the States employ a hazard quotient of 1 to limit adverse health effects from non-carcinogenic constituents.

2 Human Health Toxicity Values in Superfund Risk Assessments OSWER Directive 9285.7-53 https://semspub.epa.gov/src/document/11/136.

3 USEPA's Regional Screening Levels for Chemical Contaminants at Superfund Sites https://www.epa.gov/risk/regional-screening-levels-rsls.

4 New York state reviewed toxicity criteria available from all sources, including those outside the USA (e.g., World Health Organization), in choosing toxicity criteria.

5 The NYSDEC unrestricted scenario accounts for consumption of home grown produce and meat, and also employs an additional "relative source contribution" factor.

6 Also included analyses for other organics such as semi-volatile compounds, pesticides, and PCBs in the New York state-wide soil survey.

7 The regulation has not been updated since its inception and no indications are given regarding updates.

8 Assumes that the ground is frozen or snow covered for 100 days/year, reduces exposure frequency for direct soil exposure pathways.

9 The regulation has already undergone periodic updates since its inception, but no indication is given when the next will occur.

10 Although generic recreational soil cleanup values are not provided, Ohio EPA has recommended that the exposure frequency reduced to 90 days (relative to residential).

11 Ohio provides background/reference sediment values that vary by region, and also county-specific soil background values for metals.

12 Reduces residential and non-residential dermal exposure frequency to 245 and 160 days/year, respectively, adult body weight to 70 kg, nonresidential exposure duration to 21 years, and adjusts skin surface area.

13 The regulation indicates that soil cleanup values protective of a recreational scenario can be developed using the framework of the regulation.

14 In addition to state-wide general metal background concentrations, Michigan has metal background concentrations for 3 different soil types: topsoil, sand, and clay.

15 Reduces exposure frequency for residential to 250 days/year.

16 There are no regulations in Illinois specific to determining the suitability of dredged material for upland use. The regulations which would be used as guidance in evaluating the suitability of dredged material for upland use are identified in Item 5 in Section B-1 of this Appendix (above). While 35 Ill. Admin. Code 742 (the regulation identified in this table) would be used as a tool in evaluating a given use, other guidance and information would also be evaluated, as these regulations do not take into account the potential impact to surface water.

17 The only exception to this is for methyl tertiary-butyl-ether, for which Illinois EPA derived its own oral reference dose to support a groundwater standard.

18 Uses the USEPA's Soil Screening Guidance exposure assumptions.

19 Selects Chicago, IL, as the climatic zone for PEF calculations.

20 The regulation indicates that soil cleanup values protective of a recreational scenario can be developed using the framework of the regulation.

21 Minnesota department of health values take precedence if available over USEPA IRIS toxicity criteria.

22 Is included as part of the residential scenario.

23 Generic soil concentrations protective for consumption of home-grown food not provided, but framework acknowledges these can be complete exposure pathways and a state representative should be consulted.

24 A revised draft of the new Minnesota soil reference values was released for review in June 2015 and then finalized in April 2021 (MPCA 2021a). The older 2009 SRV values are still being cited in the 2014 "Managing Dredge Materials in the State of Minnesota."

- New York Department of Environmental Conservation: Beneficial Use Determinations (BUDs) 6 NYCRR Part 360.12 https://www.dec.ny.gov/chemical/8821.html
- Pennsylvania Department of Environmental Protection: Administrative Code, 2011, Chapter 250, Administration of Land Recycling Program https://www.dep.pa.gov/Business/Land/LandRecycling/Standards-Guidance-Procedures/Pages/default.aspx
- Ohio Environmental Protection Agency: Ohio Administrative Code 3745-599 Beneficial Uses of Wastes and Harbor Sediment Authorization https://codes.ohio.gov/oac/3745-599
- Michigan Department of Environment, Great Lakes, and Energy: Cleanup Criteria Requirements for Response Activity https://www.michigan.gov/egle/0,9429,7-135-3311_4109_9846-251790--,00.html
- Indiana Department of Environmental Management: Indiana Code (IC 13-12-3-2 and IC 13-25-5-8.5) Risk Based Closure, Screening and Closure Level Tables https://www.in.gov/idem/cleanups/resources/technical-guidance-for-cleanups/idem-screening-and-closure-level-tables/
- Illinois Environmental Protection Agency: Title 35 Illinois Administrative Code, Risk Based Cleanup Objectives Part 742. Tiered Approach to Corrective Action Objectives. https://pcb.illinois.gov/documents/dsweb/Get/Document-38408/
- Wisconsin Department of Natural Resources: Environmental Protection Chapter NR 720 Soil Cleanup Standards http://dnr.wi.gov/topic/Brownfields/Professionals.html#tabx2
- Minnesota Pollution Control Agency: Environmental Response and Liability Act, Minnesota Statue 115B.01 to 115B.24, Risk Based Site Evaluation, Soil Reference Values https://www.pca.state.mn.us/sites/default/files/c-r1-05.pdf

Table B.1-2. Comparison of residential (non-industrial) risk-based concentrations for soil, protection of human health via direct contact [1]. All values are mg/kg.

Compound / State	CAS	New York [2]	Pennsylvania [3]	Ohio [4]	Michigan [5]	Indiana [6]	Illinois [7]	Wisconsin [8]	Minnesota [9]	USEPA Adjusted RSL[10]
Metals										
Aluminum	7429-90-5	NA	1.90E+05	7.70E+04	5.00E+04	1.00E+05	NA	7.75E+04	1.90E+04	1.08E+05
Antimony	7440-36-0	NA	8.80E+01	3.10E+01	1.80E+02	4.30E+01	3.10E+01	3.13E+01	6.20E+00	4.38E+01
Arsenic	7440-38-2	1.60E+01	1.20E+01	6.80E-01	7.60E+00	9.50E+00	13.0 / 11.3 (background)	6.77E-01	9.00E+00	9.47E+00
Barium	7440-39-3	3.50E+02	4.40E+04	1.50E+04	3.70E+04	2.10E+04	5.50E+03	1.53E+04	3.00E+03	2.15E+04
Beryllium	7440-41-7	1.40E+01	4.40E+02	1.60E+02	4.10E+02	2.20E+02	1.60E+02	1.56E+02	3.10E+01	2.18E+02
Cadmium	7440-43-9	2.50E+00	1.10E+02	7.10E+01	5.50E+02	9.90E+01	7.80E+01	7.11E+01	1.60E+00	9.96E+01
Chromium (III)	16065-83-1	3.60E+01	1.90E+05	1.20E+05	7.90E+05	1.00E+05	1.20E+05	1.00E+05	2.30E+04	1.64E+05
Chromium (VI)	18540-29-9	2.20E+01	4.00E+00	3.00E-01	2.50E+03	4.20E+00	2.30E+02	3.01E-01	1.10E+01	4.22E+00
Cobalt	7440-48-4	NA	6.60E+01	2.30E+01	2.60E+03	3.20E+01	4.70E+03	2.34E+01	1.20E+01	3.28E+01
Copper	7440-50-8	2.70E+02	8.10E+03	3.10E+03	2.00E+04	4.30E+03	2.90E+03	3.13E+03	2.20E+03	4.38E+03
Iron	7439-89-6	NA	1.50E+05	5.50E+04	1.60E+05	7.70E+04	NA	5.48E+04	2.90E+04	7.67E+04
Lead	7439-92-1	4.00E+02	5.00E+02	4.00E+02	4.00E+02	4.00E+02	4.00E+02	4.00E+02	3.00E+02	4.00E+02
Manganese	7439-96-5	2.00E+03	1.00E+04	1.80E+03	2.50E+04	2.50E+03	1.60E+03	1.83E+03	2.10E+03	2.57E+03
Total Mercury		8.10E-01	3.50E+01	1.10E+01	1.60E+02	3.10E+00	1.00E+01	3.13E+00	3.10E+00	1.52E+01
Nickel	7440-02-0	1.40E+02	4.40E+03	1.50E+03	4.00E+04	2.10E+03	1.60E+03	1.55E+03	1.70E+02	2.17E+03
Selenium	7782-49-2	3.60E+01	1.10E+03	3.90E+02	2.60E+03	5.50E+02	3.90E+02	3.91E+02	7.70E+01	5.47E+02
Silver	7440-22-4	3.60E+01	1.10E+03	3.90E+02	2.50E+03	5.50E+02	3.90E+02	3.91E+02	7.70E+01	5.48E+02
Thallium	7440-28-0	NA	2.00E+00	7.80E-01	3.50E+01	1.10E+00	6.30E+00	7.82E-01	2.90E-01	1.10E+00
Vanadium	7440-62-2	NA	1.50E+01	3.90E+02	7.50E+02	5.50E+02	5.50E+02	3.93E+02	6.20E+01	5.51E+02
Zinc	7440-66-6	2.20E+03	6.60E+04	2.30E+04	1.70E+05	3.20E+04	2.30E+04	2.35E+04	4.60E+03	3.29E+04

Table B.1-2 Continued.

VOCs										
Benzene	71-43-2	2.90E+00	5.70E+01	1.20E+00	1.80E+02	1.70E+01	8.00E-01	1.60E+00	9.40E+00	1.62E+01
Ethylbenzene	100-41-4	3.00E+01	1.80E+02	5.80E+00	2.20E+04	8.10E+01	4.00E+02	8.02E+00	1.90E+02	8.09E+01
Toluene	108-88-3	1.00E+02	1.00E+04	4.90E+02	5.00E+04	8.20E+02	6.50E+02	8.18E+02	8.20E+02	6.84E+03
Xylene	1330-20-7	1.00E+02	1.90E+03	5.80E+01	4.10E+05	2.60E+02	3.20E+02	2.60E+02	2.60E+02	8.07E+02
PAHs										
Acenaphthene	83-32-9	1.00E+02	1.30E+04	3.60E+02	4.10E+04	5.00E+03	4.70E+03	3.59E+03	4.50E+02	5.02E+03
Acenaphthylene	208-96-8	1.00E+02	1.30E+04	NA	1.60E+03	NA	NA	NA	NA	NA
Anthracene	120-12-7	1.00E+02	6.60E+04	1.80E+03	2.30E+05	2.50E+04	2.30E+04	1.79E+04	2.80E+03	2.51E+04
Benzo(a)anthracene	56-55-3	1.00E+00	6.00E+00	TEF of 0.1 [11]	2.00E+01	1.50E+01	9.00E-01	1.14E+00	B(a)P equivalent of 2	1.58E+01
Benzo(a)pyrene	50-32-8	1.00E+00	5.80E-01	1 [11]	2.00E+00	1.50E+00	9.00E-02	1.15E-01		1.61E+00
Benzo(b)fluoranthene	205-99-2	1.00E+00	3.50E+00	TEF of 0.1 [11]	2.00E+01	1.50E+01	9.00E-01	1.15E+00		1.61E+01
Benzo(ghi)perylene	191-24-2	1.00E+02	1.30E+04	NA	2.50E+03	NA	NA	NA	NA	NA
Benzo(k)fluoranthene	207-08-9	1.00E+00	4.00E+00	TEF of 0.01 [11]	2.00E+02	1.50E+02	9.00E+00	1.15E+01	B(a)P equivalent of 2	1.61E+02
Chrysene	218-01-9	1.00E+00	3.50E+01	TEF of 0.001 [11]	2.00E+03	1.50E+03	8.80E+01	1.15E+02		1.61E+03
Dibenzo(a,h)anthracene	53-70-3	3.30E-01	1.00E+00	TEF of 1 [11]	2.00E+00	1.50E+00	9.00E-02	1.15E-01		1.61E+00
Fluoranthene	206-44-0	1.00E+02	8.80E+03	2.40E+02	4.60E+04	3.40E+03	3.10E+03	2.39E+03	2.00E+02	3.35E+03
Fluorene	86-73-7	1.00E+02	8.80E+03	2.40E+02	2.70E+04	3.40E+03	3.10E+03	2.39E+03	3.90E+02	3.35E+03
Indeno(1,2,3-cd)pyrene	193-39-5	5.00E-01	3.50E+00	TEF of 0.1 [11]	2.00E+01	1.50E+01	9.00E-01	1.15E+00	B(a)P equivalent of 2	1.61E+01
Naphthalene	91-20-3	1.00E+02	1.60E+02	2.00E+01	1.60E+04	2.80E+01	1.70E+02	5.52E+00	8.10E+01	2.81E+01
Phenanthrene	85-01-8	1.00E+02	6.60E+04	NA	1.60E+03	NA	NA	NA	NA	NA
Pyrene	129-00-0	1.00E+02	6.60E+03	1.80E+02	2.90E+04	2.50E+03	2.30E+03	1.79E+03	2.20E+02	2.51E+03

Table B.1-2 Continued.

PCBs										
Aroclor 1242	53469-21-9	NA	9.00E+00	2.30E-01	NA	3.20E+00	NA	2.35E-01	NA	3.22E+00
Aroclor 1248	12672-29-6	NA	9.30E+00	2.30E-01	NA	3.20E+00	NA	2.36E-01	NA	3.18E+00
Aroclor 1254	11097-69-1	NA	4.40E+00	1.20E-01	NA	1.70E+00	NA	2.39E-01	NA	1.64E+00
Aroclor 1260	11096-82-5	NA	9.00E+00	2.40E-01	NA	3.40E+00	NA	2.43E-01	NA	3.36E+00
Total PCBs	1336-36-3	1.00E+00	NA	2.30E-01	NA	3.20E+00	1.00E+00	2.34E-01	8.10E-01	3.19E+00
Pesticides										
Chlordane[2a]	12789-03-06	9.10E-01	5.30E+01	1.70E+00	3.10E+01	2.40E+01	1.80E+00	1.74E+00	9.50E+00	2.40E+01
DDD	72-54-8	2.60E+00	7.80E+01	1.90E-01	9.50E+01	2.70E+00	3.00E+00	1.90E+00	1.90E+01	2.65E+00
DDE	72-55-9	1.80E+00	5.50E+01	2.00E+00	4.50E+01	2.80E+01	2.00E+00	2.00E+00	2.20E+01	2.77E+01
DDT	50-29-3	1.70E+00	5.50E+01	1.90E+00	5.70E+01	2.70E+01	2.00E+00	1.89E+00	7.30E+00	2.64E+01
Dieldrin	60-57-1	3.90E-02	1.20E+00	3.40E-02	1.10E+00	4.80E-01	4.00E-02	3.40E-02	1.10E-01	4.75E-01
Endosulfan	115-29-7	4.80E+00	1.30E+03	4.70E+01	1.40E+03	6.60E+02	4.70E+02	4.69E+02	1.30E+02	6.57E+02
Endrin	72-20-8	2.20E+00	6.60E+01	1.90E+00	6.50E+01	2.70E+01	2.30E+01	1.90E+01	4.00E+00	2.65E+01
Heptachlor	76-44-8	4.20E-01	4.00E+00	1.30E-01	5.60E+00	1.80E+00	1.00E-01	1.40E-01	1.60E+00	1.88E+00
Heptachlor epoxide	1024-57-3	NA	2.00E+00	7.00E-02	3.10E+00	9.80E-01	7.00E-02	7.20E-02	2.80E-01	9.87E-01
Hexachlorocyclohexane, beta	319-85-7	7.20E-02	1.00E+01	3.00E-01	5.40E+00	4.20E+00	NA	3.01E-01	2.50E+00	4.22E+00
Lindane (hexachlorocyclohexane, gamma)	58-89-9	2.80E-01	1.70E+01	5.70E-01	8.30E+00	8.00E+00	5.00E-01	5.68E-01	4.30E+00	7.95E+00

Notes

[1] Please see Table B.1-1 for the source of the state-specific soil values and additional notes on their derivation.

- The risk values are for protection of human health, direct soil contact only.
- Some states may have lower soil values for some constituents for protection of ecological receptors and/or groundwater resources.
- For some states, if the risk-based concentration in this table is lower than the background concentration (identified from sources provided in Appendix A), the background value may be used.

[2] New York soil cleanup objectives are the values from 6NYCRR375-6.8(b), residential direct contact values.

[2a] New York chlordane (alpha) [CAS # 5103-71-9] value used.

[3] The Pennsylvania medium-specific concentrations for residential land use apply to soil from 0 - 15 feet below ground surface (2018).

[4] Ohio EPA recommends using the USEPA generic RSLs at a cancer risk of 1E-06 and an HQ of 0.1, except for PAHs, which should use a cancer risk of 1E-05, and metals, which should use an HQ of 1 or background soil concentrations (see Appendix A for background concentrations).

[5] Michigan values are direct contact generic cleanup criteria from their residential Table 2 for soil (2018).

[6] Indiana values are risk integrated system of closure screening values for direct contact, residential land use values (2021). Total PCB value in this table is for high risk exposure.

[7] Illinois soil remediation objectives are the lower of the ingestion or inhalation exposure-route specific values for residential land use.

[8] Wisconsin soil residual contaminant levels are the non-industrial direct contact not-to-exceed values (2018). Total PCB value in this table is for high risk exposure.

[9] Minnesota final chronic soil reference values for residential/recreational land use (2021 version).

[10] USEPA's generic May 2021 RSLs were adjusted for residential exposure in the Great Lakes region by

- choosing target cancer risk of 1E-05, and a target hazard quotient of 1 for non-carcinogens
- choosing Chicago as the climate region for calculating a PEF for inhalation
- reducing exposure frequency from 350 to 250 days/year (to account for frozen/snow covered soil during winter)

[11] Ohio EPA indicates that the total benzo(a)pyrene equivalents should not exceed 1 mg/kg (representing an excess cancer risk of 1E-05). The concentration of other carcinogenic PAHs indicated by this footnote should be multiplied by the Toxicity Equivalency Factor (TEF) for the purposes of summing the PAHs.

NA indicates that values are not available for this compound.

Table B.1-3 Comparison of industrial (non-residential) risk-based concentrations for soil, protection of human health via direct contact.

Compound/State	CAS	New York [2]	Pennsylvania [3]	Ohio [4]	Michigan [5]	Indiana [6]	Illinois [7]	Wisconsin [8]	Minnesota [9]	USEPA Adjusted RSL[10]
Metals										
Aluminum	7429-90-5	NA	1.90E+05	1.10E+06	3.70E+05	1.00E+05	NA	1.00E+05	1.00E+05	1.13E+06
Antimony	7440-36-0	NA	1.30E+03	4.70E+02	6.70E+02	4.70E+02	8.20E+02	4.67E+02	9.30E+01	4.67E+02
Arsenic	7440-38-2	1.60E+01	6.10E+01	3.00E+00	3.70E+01	3.00E+01	13.0 / 11.3 (background)	3.00E+00	9.00E+00	3.00E+01
Barium	7440-39-3	1.00E+04	1.90E+05	2.20E+05	1.30E+05	1.00E+05	1.40E+05	1.00E+05	4.10E+04	2.19E+05
Beryllium	7440-41-7	2.70E+03	6.40E+03	2.30E+03	1.60E+03	2.30E+03	2.10E+03	2.30E+03	4.20E+02	2.30E+03
Cadmium	7440-43-9	6.00E+01	1.60E+03	9.80E+02	2.10E+03	9.80E+02	2.00E+03	9.85E+02	2.30E+01	9.85E+02
Chromium (III)	16065-83-1	6.80E+03	1.90E+05	1.80E+06	1.00E+06	1.00E+05	1.00E+06	1.00E+05	1.00E+05	1.75E+06
Chromium (VI)	18540-29-9	8.00E+02	2.20E+02	6.30E+00	9.20E+03	6.30E+01	4.20E+02	6.36E+00	6.20E+01	6.36E+01
Cobalt	7440-48-4	NA	9.60E+02	3.50E+02	9.00E+03	3.50E+02	1.20E+05	3.47E+02	6.90E+01	3.47E+02
Copper	7440-50-8	1.00E+04	1.20E+05	4.70E+04	7.30E+04	4.70E+04	8.20E+04	4.67E+04	3.30E+04	4.67E+04
Iron	7439-89-6	NA	1.90E+05	8.20E+05	5.80E+05	1.00E+05	NA	1.00E+05	1.00E+05	8.18E+05
Lead	7439-92-1	3.90E+03	1.00E+03	8.00E+02	9.00E+02	8.00E+02	8.00E+02	8.00E+02	7.00E+02	8.00E+02
Manganese	7439-96-5	1.00E+04	1.50E+05	2.60E+04	9.00E+04	2.60E+04	4.10E+04	2.59E+04	2.60E+04	2.59E+04
Total Mercury	7439-97-6	5.70E+00	5.10E+02	4.60E+01	5.80E+02	3.10E+00	1.60E+01	3.13E+00	3.10E+00	4.56E+01
Nickel	7440-02-0	1.00E+04	6.40E+04	2.20E+04	1.50E+05	2.20E+04	2.10E+04	2.25E+04	2.60E+03	2.25E+04
Selenium	7782-49-2	6.80E+03	1.60E+04	5.80E+03	9.60E+03	5.80E+03	1.00E+04	5.84E+03	1.20E+03	5.84E+03
Silver	7440-22-4	6.80E+03	1.60E+04	5.80E+03	9.00E+03	5.80E+03	1.00E+04	5.84E+03	1.20E+03	5.84E+03
Thallium	7440-28-0	NA	3.20E+01	1.20E+01	1.30E+02	1.20E+01	1.60E+02	1.17E+01	2.30E+00	1.17E+01
Vanadium	7440-62-2	NA	2.20E+02	5.80E+03	5.50E+03	5.80E+03	1.40E+04	5.84E+03	6.20E+01	5.84E+03
Zinc	7440-66-6	1.00E+04	1.90E+05	3.50E+05	6.30E+05	1.00E+05	6.10E+05	1.00E+05	7.00E+04	3.50E+05

Table B.1-3. Continued.

VOCs										
Benzene	71-43-2	8.90E+01	2.90E+02	5.10E+00	8.40E+02	5.10E+01	1.60E+00	7.07E+00	4.20E+01	5.09E+01
Ethylbenzene	100-41-4	7.80E+02	8.90E+02	2.50E+01	7.10E+04	2.50E+02	4.00E+02	3.54E+01	4.80E+02	2.54E+02
Toluene	108-88-3	1.00E+03	1.00E+04	4.70E+03	1.60E+05	8.20E+02	6.50E+02	8.18E+02	8.20E+02	4.68E+04
Xylene	1330-20-7	1.00E+03	8.00E+03	2.50E+02	1.00E+06	2.60E+02	3.20E+02	2.60E+02	2.60E+02	2.49E+03
PAHs										
Acenaphthene	83-32-9	1.00E+03	1.90E+05	4.50E+03	1.30E+05	4.50E+04	1.20E+05	4.52E+04	6.80E+03	4.52E+04
Acenaphthylene	208-96-8	1.00E+03	1.90E+05	NA	5.20E+03	NA	NA	NA	NA	NA
Anthracene	120-12-7	1.00E+03	1.90E+05	2.30E+04	7.30E+05	1.00E+05	6.10E+05	1.00E+05	4.20E+04	2.26E+05
Benzo(a)anthracene	56-55-3	1.10E+01	1.30E+02	2.10E+02	8.00E+01	2.10E+02	8.00E+00	2.08E+01		2.06E+02
Benzo(a)pyrene	50-32-8	1.10E+00	1.20E+01	2.10E+01	8.00E+00	2.10E+01	8.00E-01	2.11E+00	B(a)P equivalent of 3	2.11E+01
Benzo(b)fluoranthene	205-99-2	1.10E+01	7.60E+01	2.10E+02	8.00E+01	2.10E+02	8.00E+00	2.11E+01		2.11E+02
Benzo(ghi)perylene	191-24-2	1.00E+03	1.90E+05	NA	7.00E+03	NA	NA	NA	NA	NA
Benzo(k)fluoranthene	207-08-9	1.10E+02	7.60E+01	2.10E+03	8.00E+02	2.10E+03	7.80E+01	2.11E+02		2.11E+03
Chrysene	218-01-9	1.10E+02	7.60E+02	2.10E+04	8.00E+03	2.10E+04	7.80E+02	2.11E+03	B(a)P equivalent of 3	2.11E+04
Dibenzo(a,h)anthracene	53-70-3	1.10E+00	2.20E+01	2.10E+01	8.00E+00	2.10E+01	8.00E-01	2.11E+00		2.11E+01
Fluoranthene	206-44-0	1.00E+03	1.30E+05	3.00E+03	1.30E+05	3.00E+04	8.20E+04	3.01E+04	2.70E+03	3.01E+04
Fluorene	86-73-7	1.00E+03	1.30E+05	3.00E+03	8.70E+04	3.00E+04	8.20E+04	3.01E+04	5.80E+03	3.01E+04
Indeno(1,2,3-cd)pyrene	193-39-5	1.10E+01	7.60E+01	2.10E+02	8.00E+01	2.10E+02	8.00E+00	2.11E+01	B(a)P equivalent of 3	2.11E+02
Naphthalene	91-20-3	1.00E+03	7.60E+02	5.90E+01	5.20E+04	8.60E+01	2.70E+02	2.41E+01	2.80E+02	8.56E+01
Phenanthrene	85-01-8	1.00E+03	1.90E+05	NA	5.20E+03	NA	NA	NA	NA	NA
Pyrene	129-00-0	1.00E+03	9.60E+04	2.30E+03	8.40E+04	2.30E+04	6.10E+04	2.26E+04	3.20E+03	2.26E+04
PCBs										
Aroclor 1242	53469-21-9	NA	4.60E+01	9.50E-01	NA	9.50E+00	NA	9.72E-01	NA	9.50E+00
Aroclor 1248	12672-29-6	NA	4.60E+01	9.40E-01	NA	9.40E+00	NA	9.75E-01	NA	9.39E+00
Aroclor 1254	11097-69-1	NA	4.60E+01	9.70E-01	NA	9.70E+00	NA	9.88E-01	NA	9.72E+00
Aroclor 1260	11096-82-5	NA	4.60E+01	9.90E-01	NA	9.90E+00	NA	1.00E+00	NA	9.91E+00
Total PCBs	1336-36-3	2.50E+01	NA	9.40E-01	NA	9.40E+00	1.00E+00	9.67E-01	1.00E+01	9.42E+00

Table B.1-3. Continued.

Pesticides										
Chlordane[2a]	12789-03-06	4.70E+01	2.60E+02	7.70E+00	1.50E+02	7.70E+01	1.60E+01	7.76E+00	1.00E+02	7.66E+01
DDD	72-54-8	1.80E+02	3.80E+02	2.50E+00	4.00E+02	2.50E+01	2.40E+01	9.57E+00	1.00E+02	2.46E+01
DDE	72-55-9	1.20E+02	2.70E+02	9.30E+00	1.90E+02	9.30E+01	1.70E+01	9.38E+00	1.30E+02	9.28E+01
DDT	50-29-3	9.40E+01	2.70E+02	8.50E+00	2.80E+02	8.50E+01	1.70E+01	8.53E+00	8.70E+01	8.53E+01
Dieldrin	60-57-1	2.80E+00	6.00E+00	1.40E-01	4.70E+00	1.40E+00	4.00E-01	1.44E-01	1.50E+00	1.44E+00
Endosulfan	115-29-7	9.20E+02	1.90E+04	7.00E+02	4.40E+03	7.00E+03	1.20E+04	7.01E+03	1.90E+03	7.01E+03
Endrin	72-20-8	4.10E+02	9.60E+02	2.50E+01	1.90E+02	2.50E+02	6.10E+02	2.46E+02	5.40E+01	2.46E+02
Heptachlor	76-44-8	2.90E+01	2.00E+01	6.30E-01	2.30E+01	6.30E+00	1.00E+00	6.54E-01	8.90E+00	6.26E+00
Heptachlor epoxide	1024-57-3	NA	1.00E+01	3.30E-01	9.50E+00	3.30E+00	6.00E-01	3.38E-01	4.20E+00	3.30E+00
Hexachlorocyclohexane, beta	319-85-7	1.40E+01	5.10E+01	1.30E+00	2.50E+01	1.30E+01	NA	1.28E+00	1.40E+01	1.28E+01
Lindane (hexachlorocyclohexane, gamma)	58-89-9	2.30E+01	8.30E+01	2.50E+00	4.20E+01	2.50E+01	4.00E+00	2.54E+00	2.50E+01	2.54E+01

Notes

[1] Please see Table B.1-1 for the source of the state-specific soil values and additional notes on their derivation.

- The risk values are for protection of human health, direct soil contact only.
- Some states may have lower soil values for some constituents for protection of ecological receptors and/or groundwater resources.
- For some states, if the risk-based concentration in this table is lower than the background concentration (identified from sources provided in Appendix A), the background value may be used.

[2] New York soil cleanup objectives are the values from 6NYCRR375-6.8(b), industrial (restricted) land use. New York also has another set of soil cleanup objectives for commercial land use to which imported materials for fill may be limited even at an industrial land-use site, unless justified on a case-specific basis.

[2a] New York chlordane (alpha) [CAS # 5103-71-9] value used.

[3] The Pennsylvania medium-specific concentrations for non-residential land use apply to surface soil, 0 - 2 feet below ground surface (2018).

[4] Ohio EPA recommends using the USEPA generic RSLs at a cancer risk of 1E-06 and an HQ of 0.1, except for PAHs, which should use a cancer risk of 1E-05, and metals, which should use an HQ of 1 or background soil concentrations (see Appendix A for background concentrations).

[5] Michigan values are direct contact generic cleanup criteria from their non-residential Table 2 for soil (2018).

[6] Indiana values are risk integrated system of closure screening values for direct contact, commercial/industrial land use values (2021). Total PCB value in this table is for high risk exposure.

[7] Illinois soil remediation objectives are the lower of the ingestion or inhalation exposure-route specific values for industrial/commercial land use.

[8] Wisconsin soil residual contaminant levels are the industrial direct contact not-to-exceed values (2018). Total PCB value in this table is for high risk exposure.

[9] Minnesota final chronic soil reference values for commercial/industrial land use (May 2021 version).

[10] USEPA's generic May 2021 RSLs were adjusted for the composite worker exposure in the Great Lakes region by

- choosing target cancer risk of 1E-05, and a target hazard quotient of 1 for non-carcinogens
- choosing Chicago as the climate region for calculating a PEF for inhalation

  (no reduction in exposure frequency was made for the worker scenario)

NA indicates that values are not available for this compound.

Appendix C: Assessment of Ecological Soil Screening Levels (Eco-SSLs) for Determining Suitability of Dredged Material for Beneficial Use — Plant Pathway

Purpose

This appendix discusses the use of ecological soil screening levels in determining suitability of dredged material for upland habitat creation for wildlife use. Various soil screening level guidance has been provided by state resource agencies, the US Environmental Protection Agency (USEPA) and others for comparing total soil contaminant levels to numerical criteria protective of ecological or human health. The basis of screening levels varies and may include protection of human health through residential or industrial use exposures, groundwater protection, and protection of ecological health. Such screening levels can be used in Tier 1 evaluations of dredged material to determine if any given COPC is present at concentrations that would require further evaluation of potential risks. However, there is considerable uncertainty in the use of soil screening levels that were not developed using data from dredged material exposure because mobility (fate and transport) of constituents from dredged material may differ from mobility in soil.

This work focuses on the soil-to-plant pathway of contaminant risk to ecological receptors. Soil screening level evaluations were compared to plant exposure testing results to determine the effectiveness in predicting potential exposure risks. The purpose of this appendix is to summarize and discuss the results and implications of soil screening application to beneficial use evaluations.

Background

There are many CDFs currently operated under state Section 401 water quality permits in the Great Lakes that will be filled to capacity in the near future. Once reaching capacity, many of these CDFs will be used for habitat/recreation use and will be used by wildlife unless measures are implemented to prevent habitats from developing or to discourage wildlife use. Potential risk to ecological receptors that colonize or use the CDF and are exposed to contaminants is an issue and, in many cases, has not been thoroughly evaluated. Dredged material removed from CDFs and used beneficially outside the CDF that may pose no risks to human receptors may pose risks to sensitive wildlife receptors or could result in higher exposure to wildlife than if the material remained in the CDF. It is the responsibility of the USACE under NEPA to address these ecological concerns using the best available science necessary to ensure risks are communicated and understood. The suitability of dredged material for upland beneficial use is ultimately made in consultation with the state regulatory agency in whose jurisdiction the beneficial use occurs. Any restrictions are generally based on grain size and/or concentrations of COPCs. Whether defined as soil *screening levels*, soil *criteria* or soil *standards*, there is a myriad of approaches used by state regulatory agencies to determine acceptable levels of potential contaminants in dredged material used for terrestrial beneficial uses.

Ecological soil screening levels

The USEPA has developed Eco-SSLs (USEPA 2003). Eco-SSLs are concentrations of COPCs in soil that are protective of ecological receptors that commonly come in contact with the soil or ingest biota that live in or on soil. The Eco-SSLs were developed for four groups of ecological receptors: soil invertebrates, plants, birds, and mammals. The latter two groups of ecological receptors are represented by mammalian herbivore (vole), insectivore (shrew), carnivore (weasel) and avian herbivore (dove), insectivore (woodcock), carnivore (hawk). Since Eco-SSLs are intended to represent soils on a national scale, it may seem possible to use these for screening contaminants in dredged material for most ecologically based upland beneficial uses. Eco-SSLs were specifically developed based on exposures to soil, and one could argue application to dredged material evaluations is not supported by the data used to develop Eco-SSLs. Dredged material placed in an upland environment and allowed to colonize by plants and soil invertebrates will develop all the characteristics of a normal soil over time. However, the question is how much time and what are the differing physical-chemical effects on COC bioavailability at any given time ecological exposure occurs. Extensive testing and validation of the use of Eco-SSLs in screening dredged material for beneficial use is needed to ensure their use in dredged material evaluations provides adequate ecological protection. For purposes of this study, the only pathway being evaluated is that of the mammalian and avian herbivore.

Guidance provided by the Upland Testing Manual (UTM)

The UTM (ERDC/EL TR-03-1 [USACE 2003]) provides guidance for the evaluation of potential ecological/human health impacts from the placement of dredged material into CDFs that result in terrestrial environments. Guidance in the UTM considers dredged material disposed into a CDF and evaluates potential contaminant transport outside the CDF through various pathways (e.g., into receiving water, groundwater, air, or by ingestion and transfer through plant and animal food chains to outside receptors). Transport of contaminants outside the CDF through any given pathway would require engineering controls to manage that pathway if the transport could potentially result in unacceptable adverse effects or failed endpoints such as water quality standards. The UTM uses a tiered approach with cost and intensity increasing with each higher tier. If a decision cannot be made at the end of a lower tier, additional evaluations are performed at a higher tier. Guidance within the UTM is designed to address the disposal of dredged material within a CDF and provide evaluation protocols as necessary to determine no impact outside the CDF is expected or engineering controls are necessary to prevent such impact from occurring. Impacts associated with beneficial site use or beneficial use of dredged material removed from CDFs is specifically not addressed by the UTM. However, the UTM testing procedures to determine potential contaminant bioaccumulation or migration may be used in an evaluation of potential risks associated with beneficial uses.

Although the UTM does not include specific guidance for comparing dredged material COPCs to a soil screening level criteria or soil standards to determine suitability for upland placement or need for risk evaluation, it is suggested in the Tier I or Tier II evaluation. Eco-SSLs or other soil criteria may be useful in initial tier evaluations for CDF placement to determine the need for further evaluation and are generally the first step required by state regulatory agencies for beneficial reuse approval. It is recognized existing soil screening or soil standards by design may be overly conservative or restrictive in the protection of ecological and human health. For instances where dredged material COPCs exceed soil screening levels or soil standards it may be appropriate to conduct biological exposure or other testing to determine COPC bioavailability and/or transport kinetics from the material in question.

Biological exposure testing described in the UTM provides reliable assessment of contaminant bioavailability from soil to receptors. The bioaccumulation of contaminants by terrestrial plants and animals is not directly governed by any specific regulations or limitations of COPC in tissues. The UTM guidance suggests biological testing on both the dredged material and on a reference or background soil for comparison of tissue concentrations. While this provides an assessment of bioaccumulation from the dredged material relative to surrounding conditions, this alone does not provide an assessment of risk to receptors. This must be determined by comparing the tissue COPC concentration to a concentration known to cause an adverse effect. However, such information is not readily available to address the wide range of contaminants that are encountered in dredged sediments or the exposure risks to many different receptors. Early studies by Environmental Laboratory (1987), Lee et al. (1992, 1993), and others addressing plant uptake of heavy metals from dredged material compared plant tissue concentrations to US Food and Drug Administration and World Health Organization established action levels for poisonous or deleterious substances in human food and animal feed or developing European Union (EU) directives on undesirable substances in foodstuffs and animal feeds sold in EU countries. These action levels were primarily associated with shellfish, processed foods, and or vegetable products for human consumption. While these action levels may have been useful at the time as a guide for COPCs concentrations in foodstuffs that were of concern for human health, they did not provide sufficient guidance for risks to ecological health.

A more relevant approach today may be derived from the TRV food chain model used in development of the USEPA Eco-SSLs for herbivores (mammals and birds) for exposure to soil COCs. Using guidance provided by USEPA for Eco-SSL development, a plant COPC limit can be determined and compared to actual tissue COPC concentrations by exposing plants to dredged material. This provides a screening level endpoint using site-specific exposure results not otherwise provided in the UTM and provides a means of evaluating the effectiveness of the Eco-SSL approach for evaluating potential risks to wildlife exposed to dredged material colonized by plants.

Approach

To evaluate the potential ecological impacts from contaminants in dredged material used beneficially to support wildlife habitat, two approaches were evaluated. A screening level approach compared site soil metals to Eco-SSL soil limits for the mammalian and avian herbivore only, and a greenhouse study exposed plants to the CDF soils and to local reference soils. Plant uptake of metals from CDF soils was compared to uptake from reference soils and to a plant concentration upper limit derived from the toxicity reference values and food chain model used in the Eco-SSL development.

Methods and materials

Dredged material was randomly collected to a maximum depth of 40 cm from several locations within the upland areas of CDFs located in Cleveland, Lorain, and Toledo, OH. In addition, for each CDF location, a reference (REF) site was selected, and soils were collected from these sites as well. The materials were sealed in plastic buckets and shipped overnight delivery to the ERDC, Vicksburg, MS. The dredged material from each CDF was thoroughly mixed and screened through a 2 cm screen to remove any debris. Test materials were characterized for moisture content, pH, and grain size following the methods in Appendix H of the UTM (USACE 2003). Total organic carbon was determined by SW-846 Method 9060 and total metals (As, Cd, Cr, Cu, Pb, Ni, Ag, and Zn) were determined by SW-846 Method 3050b and 6010 (USEPA 1998a). This subset of metals was chosen for analyses because these are the metals that failed a Tier I screen of constituents for potential environmental impacts at the CDFs evaluated (USACE Buffalo District 2007a,b,c and 2008). Results were reported on a dry weight basis. Total soil metals were numerically compared to Eco-SSLs (USEPA 2003) to determine if any of the metals exceeded acceptable levels for ecological exposure.

The Eco-SSLs for limits of soil metal to herbivore receptors assumes a bioaccumulation factor for each metal based on published literature. To evaluate the soil-to-plant-to-herbivore pathway actual plant uptake by exposure of plants to test materials was determined in a double bucket bioassay apparatus as described Folsom and Price (1991) and USACE (2003). Three replicates of each were prepared and planted with *Cyperus esculentus*. After 45 days of growth, aboveground plant tissues were harvested, washed, and weighed. Tissue was digested and analyzed for As, Cd, Cr, Cu, Pb, Ni, Ag, and Zn following SW-846 Method 3050b and 6010 (USEPA 1998a). Results were reported on a dry weight basis.

Based on the guidance provided for development of Eco-SSLs (USEPA 2003), acceptable concentrations of contaminants in plant tissues that would be ingested by receptor herbivore species were calculated based on the following equation:

$$C_{plant} = (I \times BW)/(F \times CR)$$

where

I = the acceptable daily intake of contaminant (mg dry weight/kg body weight per day) (also equivalent to the toxicity reference value).
BW = the body weight of target receptor (kg)
F = the fraction of vegetation consumed
CR = the consumption rate (kg dry weight plant per day).

For example, the TRV provided for the surrogate receptor group (mammalian herbivore) for cadmium is 0.770 mg dry weight per kilogram of body weight per day (USEPA 2005b). Using the surrogate species (meadow vole) with a body weight of 17 g (0.017 kg) and assuming the diet is 100% plant tissue at a rate of 0.0875 kg plant/kg body weight (0.0014875 kg/day), then one has the following:

$$C_{plant} = (0.770 \text{ mg/kg} \times 0.017 \text{ kg}) / (1 \times 0.0014875 \text{ kg})$$

$$C_{plant} = 8.80 \text{ mg/kg}$$

The resulting value of 8.80 mg kg^{-1} of cadmium in plant tissue would be the concentration below which the mammalian receptor group would not exceed a daily dose exposure known to produce an adverse response. Comparison of actual plant tissue cadmium, determined by plant bioassays in the test material, to the calculated C_{plant} value can determine if bioaccumulation of cadmium by plants exposed to test materials may pose a risk to receptor herbivores. An acceptable plant concentration (C_{plant}) was calculated for all metals shown in Table C-1 using the TRVs and the food ingestion rate provided by USEPA for each metal (USEPA 2003) and a receptor bodyweight of 17.0 g for the herbivore meadow vole (USEPA

1993b). Table 2 shows the same calculation for the avian herbivore (mourning dove) with a bodyweight of 96.0 g.

Table C-1. TRVs used in Eco-SSL determinations[1] and calculated acceptable plant concentrations for mammalian herbivores (DW = dry weight; BW = body weight).

Contaminant	Food Ingestion Rate (kg DW/kg BW/day)	Toxicity Reference Value (mg DW/kg BW/day)	USEPA Soil ECo-SSL, Mg/kg	C-PIANT, mg/kg
	Plant to Mammalian	Plant to Mammalian	Mammalian	Mammalian
Arsenic	0.0875	1.04	170	11.9
Cadmium	0.0875	0.770	73	8.8
Chromium[2]	0.0875	2.40	380	27.4
Copper	0.0875	5.6	1100	64.0
Lead	0.0875	4.70	1200	53.7
Nickel	0.0875	1.70	340	19.4
Silver	0.0875	6.02	1500	68.8
Zinc	0.0875	75.4	6800	861.7

[1] From USEPA 2003
[2] Trivalent (CR III)

Table C-2. TRVs used in Eco-SSL determinations[1] and calculated acceptable plant concentrations for avian herbivores.

Contaminant	Food Ingestion Rate (kg DW/kg BW/d)	Toxicity Reference Value (mg DW/kg BW/d)	USEPA Soil ECo-SSL, Mg/kg	C-Plant, mg/kg
	Plant to Avian	Plant to Avian	Avian	Avian
Arsenic	0.19	2.24	67	11.8
Cadmium	0.19	1.47	28	7.7
Chromium[2]	0.19	2.66	78	14.0
Copper	0.19	4.05	76	21.3
Lead	0.19	1.63	46	8.6
Nickel	0.19	6.71	210	35.3
Silver	0.19	2.02	69	10.6
Zinc	0.19	66.1	950	347.9

[1] From USEPA 2003
[2] Trivalent (CR III)

Results from plant bioassays

Physical soil data are shown for the test materials in Table C-3. Soil pH levels are typical for soils in northern Ohio watersheds and are within the optimum range for supporting plant growth. The lowest pH of 5.7 in the

Cleveland REF soil (which was collected from a lakeside city park) is typical of soil that may have received excessive nitrogen fertilizer and insufficient lime to offset acidity. While the pH range shown would not be expected to significantly affect the growth of most plants, even subtle differences can be expected to influence chemical reactions in the soil rhizosphere, including metals extractable by plant roots. Greater differences in soil clay content and total organic carbon are shown for Cleveland soils from the Cuyahoga watershed. Depending on nutrient levels, these characteristics could have an effect on potential plant growth and potential uptake of metals. Plant yields for the tested materials are shown in Figures C-1 through C-3 and confirm the lower yield expected for the Cleveland REF and highest yields for the Toledo CDF and REF. The obviously lighter-green appearance of the Cleveland REF is an indicator of lower nutrient availability; however, nutrient levels in the soil were not tested, and all test materials produced more than sufficient biomass to assess uptake of metals.

Table C-3. CDF and REF soil characteristics.

Sample	Field Moisture, %	pH	Particle Size Analysis (%)			TOC mg/kg
			Sand	Silt	Clay	
Lorain CDF	32.2	7.2	25.0	53.3	21.7	12,000
Lorain Reference	22.3	6.9	31.7	52.5	15.8	31,000
Cleveland CDF	23.6	6.8	35.8	55.0	9.2	7,500
Cleveland Reference	14.9	5.7	78.3	14.2	7.5	4,500
Toledo CDF	24.7	7.5	17.5	44.2	38.3	13,000
Toledo Reference	24.6	6.4	16.7	40.8	42.5	15,000

Figure C-1. Plant growth in Cleveland CDF (l) and REF soils (r).

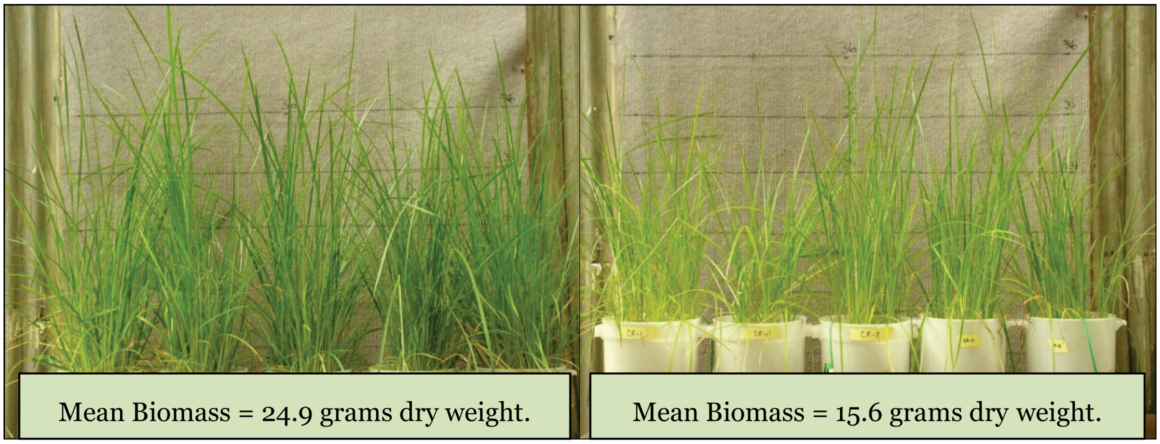


Figure C-2. Plant growth in Lorain CDF (l) and REF soils (r).

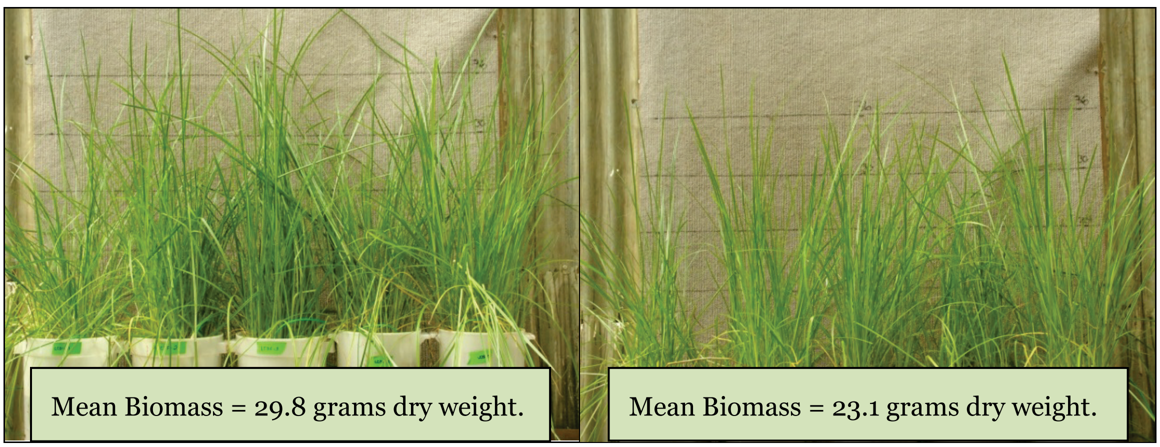
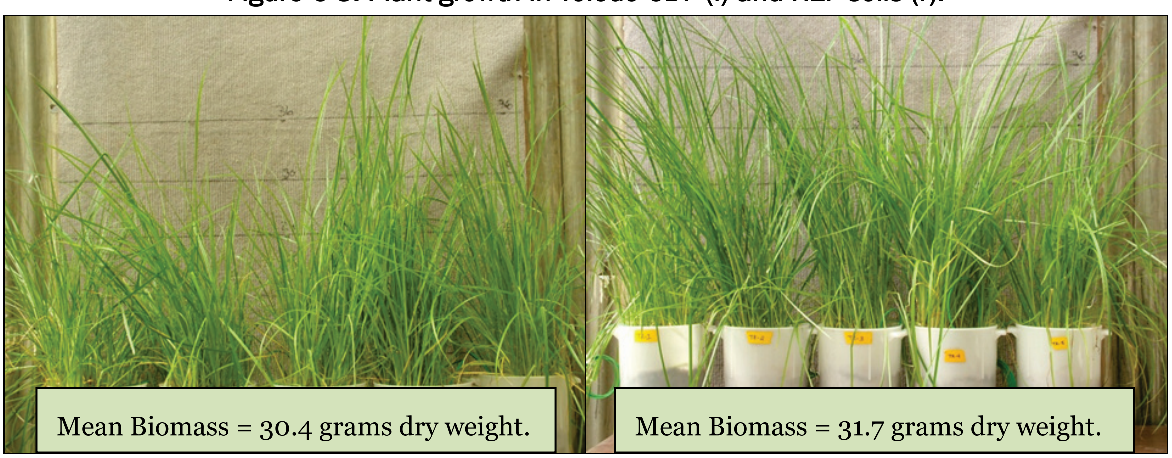


Figure C-3. Plant growth in Toledo CDF (l) and REF soils (r).



Soil and plant tissue concentrations of metals from the Lorain CDF and Reference Soil (REF) bioassays are shown in Table 4. Although soil concentrations of Cd, Cr, Cu, Pb, Ni, and Zn were all higher in the Lorain CDF than the Lorain REF soil, none of the metals exceeded the Eco-SSLs for mammalian or avian exposures. Actual plant exposures to the test soils showed greater plant tissue concentrations of As, Cd, Cr, Pb, and Ni from CDF vs REF soil. In addition, tissue concentration of Pb exceeded the C_{plant} limit for avian exposure even though the soil concentration was less than the soil Eco-SSL. The last 2 columns in Table C-4 provide the soil-to-plant bioaccumulation factor (BAF). The bioavailable fraction of the total soil metal concentration, which is controlled by complex chemical reactions in the soil rhizosphere, is more important than soil concentration alone for controlling plant uptake and the results indicate why simple numerical comparisons of soil metal concentrations are often not as protective as expected.

Table C-4. Comparison of Lorain CDF and REF soils to acceptable limits of metals in soil and plant tissues for ecological exposure, all values in mg kg^{-1}.

COC	CDF Soil	REF Soil	Eco-SSL Plant to:		CDF Plant	REF Plant	Soil to Plant BAF	
			Mammal	Avian			CDF	REF
As	12	14	170	67	0.543	<0.50	0.05	ND
Cd	5.4†	3.8	73	28	6.797	4.87	1.26	1.28
Cr	33	21	380	78	0.823	<0.50	0.02	ND
Cu	47	27	1100	76	10.057	10.4	0.21	0.39
Pb	39	26	1200	46	10.62††	3.93	0.27	0.15
Ni	37	28	340	210	1.54	1.013	0.04	0.04
Ag	0.3	<1	1500	69	<0.50	<0.50	ND	ND
Zn	189	140	6800	950	72.83	92.663	0.39	0.66

† CDF soil/plant concentration in bold exceeds REF concentration.

†† Plant tissue concentration exceeds the C_{plant} limit for avian consumption presented in Table 2.

The influence of bioavailability is shown extremely well for the Cleveland soils in Table C-5. All the tested metals (except Ag) in the CDF soil exceeded the REF soil. However, the higher soil concentrations did not result in higher plant tissue concentrations as none of the CDF plant concentrations exceeded the REF soil plants. The bioavailability of Cd and Pb was significantly higher in the REF soil and resulted in tissue concentrations of Pb over four times higher than the C_{plant} limit for avian herbivore exposure.

Table C-5. Comparison of Cleveland CDF and REF soils to acceptable limits of metals in soil and plant tissues for ecological exposure, all values in mg kg^{-1}.

COC	CDF Soil	REF Soil	Eco-SSL Plant to:		CDF Plant	REF Plant	Soil to Plant BAF	
			Mammal	Avian			CDF	REF
As	13†	6.4	170	67	<0.50	0.92	ND	0.14
Cd	2.6	0.99	73	28	1.29	3.67	0.50	3.71
Cr	20	7.8	380	78	0.07	1.9	0.00	0.24
Cu	39	10	1100	76	8.13	9.69	0.21	0.97
Pb	34	9.3	1200	46	3.64	35.3††	0.11	3.80
Ni	28	24	340	210	1.04	2.65	0.04	0.11
Ag	<1	<1	1500	69	<0.50	<0.50	ND	ND
Zn	186	47	6800	950	61.49	82.41	0.33	1.75

† CDF soil/plant concentration in bold exceeds REF concentration.

†† Plant tissue concentration exceeds the C_{plant} limit for avian consumption presented in Table 2. (Note however that the concentration of lead from the reference soil is below the 50th percentile of background/ambient concentrations of lead in unimpacted soil in the Eastern United States, USEPA 2003, 2005b).

Results shown in Table C-6 shows the Toledo CDF exhibited the highest soil concentrations of Cr, Cu, Pb, Ni, and Ag, and Pb and all metals tested in the Toledo CDF exceeded the Toledo REF. The Pb concentration was the only metal tested that exceeded a soil Eco-SSL pathway for herbivores (46 mg kg^{-1} for avian exposure). However, plant uptake of 4.2 mg kg^{-1} Pb did not exceed the C_{plant} upper limit of 8.6 mg kg^{-1} and the BAF was the lowest of all the materials tested. Although all CDF soil metal concentrations exceeded the REF, only Cd, Cu, and Ni in CDF plant tissues exceeded the REF tissues. Cadmium concentrations in Toledo CDF plant tissues exceeded the C_{plant} upper limit for both avian and mammalian herbivores of 6.9 and 8.8 mg kg^{-1}, respectively.

Table C-6. Comparison of Toledo CDF and REF soils to acceptable limits of metals in soil and plant tissues for ecological exposure, all values in mg kg^{-1}.

COC	CDF Soil	REF Soil	Eco-SSL Plant to:		CDF Plant	REF Plant	Soil to Plant BAF	
			Mammal	Avian			CDF	REF
As	9.6†	6.6	170	67	<0.50	<0.50	ND	ND
Cd	5.1	2.2	73	28	10.27†‡	1.45	2.01	0.66
Cr	60	27	380	78	<0.65	<0.50	ND	ND
Cu	55	30	1100	76	11.75†	10.01	0.21	0.33
Pb	50ssl	32	1200	46	4.217	6.64	0.08	0.21
Ni	46	20	340	210	1.71†	0.993	0.04	0.05
Ag	0.7	<1	1500	69	<0.517†	<0.50	ND	ND
Zn	186	96	6800	950	87.12†	61.06	0.47	0.64

† CDF soil/plant concentration in bold exceeds REF concentration.

ssl Soil concentration exceeds the Eco-SSL for avian exposure.

†‡ Plant tissue concentration exceeds the C_{plant} limit for avian and mammalian consumption, ,as presented in Tables 1 and 2.

Results of these studies showed that total soil concentrations of metals in the test soils exceeded the Eco-SSL limits for avian and mammalian herbivores only for Pb in the Toledo CDF. Actual plant exposure to Toledo CDF soil did not result in a plant tissue concentration that would equal or exceed an Eco-SSL-based C_{plant} concentration limit. However, plants grown in Cleveland REF and Lorain CDF soils did exceed the C_{plant} Pb limit for avian herbivore exposure. In addition, plants grown in the Toledo CDF soil produced tissue concentrations higher than the C_{plant} Cd limits for both avian and mammalian herbivores. These results indicate that screening-level approaches based on soil contaminant concentrations such as the Eco-SSLs may not be reliable for determining potential exposure to ecological receptors and possible impacts to human health from the use of dredged material as soil to support vegetation for habitat development whether it remains in or is removed from the CDF. The results of plant bioassays indicate plant colonization on Lorain CDF and Toledo CDF soils may pose potential risks of lead and cadmium exposure to avian herbivores that spend most of their time feeding on these sites.

Discussion

These results have shown that the Eco-SSL guidance based on herbivore exposure can be both overprotective and underprotective when evaluating the ecological risks associated with bioaccumulation of metals from dredged material into plant tissues. The uncertainty exists by the use of a single BAF in the Eco-SSL development to model the soil-to-plant extractable fraction of metals that varies only by total soil concentration. The soil-to-plant BAF can vary by orders of magnitude and is affected more by various soil physical and chemical properties vs. total metal soil concentration. The complexity of the synergistic and antagonistic effects of various properties on the plant extractable fraction of most metals makes it difficult to develop a simple yet reliable model based on median or regression BAF. As an example, the Eco-SSL for cadmium in soil as provided by USEPA (2005a) is derived based on a cadmium uptake equation described in Table 4a in (USEPA 2005b) as follows:

$$\ln(C_p) = 0.546 * \ln(C_s) -0.475$$

where

C_p = Concentration in plant tissue
C_s = Concentration in soil.

Inserting a soil concentration (Cs) of 1.33 into the equation results in a predicted plant concentration (Cp) of 0.73 mg kg^{-1} with a resulting BAF of 0.546. The equation results in a decreased bioavailability as the soil Cd concentration increases. Substituting the Toledo CDF soil concentration of 5.1 mg kg-1 from Table C-6 discussed previously, results in

$$\ln(C_p) = 0.546 * \ln(5.1) -0.475$$

$$C_p = 1.51$$

The above theoretical plant concentration of 1.51 mg kg^{-1} results in a theoretical soil-to-plant BAF of 0.3. However, actual tissue concentrations of cadmium in *Cyperus esculentus* exposed to Toledo CDF soil averaged 10.3 mg kg^{-1} with a soil-to-plant BAF of 2.0. Using the Eco-SSL for soil cadmium in this case underestimates the risk to mammalian and avian herbivores due to an underestimation of predicted cadmium in plant tissues. In Table C-7, plant uptake of lead determined by plant bioassays are compared to Eco-SSL derived exposure limits for lead in soil and estimated plant uptake. The total soil concentration for each test material was entered in Eco-SSL uptake equation for lead and was compared to the

Eco-SSL and actual plant uptake [$\ln(C_p) = 0.561 * \ln(C_s) - 1.328$] (USEPA 2005c, 2007a). The 50 mg kg^{-1} in the Toledo CDF exceeded the Eco-SSL of 46 mg kg^{-1} for avian exposure via plant uptake. However, both the predicted plant concentration of 5.58 mg kg^{-1} and the actual concentration from the bioassay of 4.22 mg kg^{-1} was below the acceptable concentration (C_{plant}) of 7.7 in Table C-2. Additionally, actual plant uptake of Pb from the Cleveland REF and Lorain CDF soils exceeded the C_{plant} limit despite lower soil concentrations of Pb. These results demonstrate the potential frequency of failure in using USEPA Eco-SSLs to make decisions regarding use of dredged material for habitat use where receptors are expected to be exposed to metals in colonizing plants.

Table C-7. Comparison of Eco-SSL predicted to plant bioassay results for lead.

Site Name	C_s[1]	Predicted C_p[1]	Predicted BAF	Bioassay C_p[1]	Bioassay BAF
Lorain CDF	39	4.86[ue]	0.12	10.62‡	0.27
Lorain REF	26	3.87	0.15	3.92	0.15
Cleveland CDF	34	4.50	0.13	3.64	0.11
Cleveland REF	9.3	2.17[ue]	0.23	35.3‡	3.80
Toledo CDF	50†[oe]	5.58	0.11	4.22	0.08
Toledo REF	32	4.35	0.14	6.67	0.21

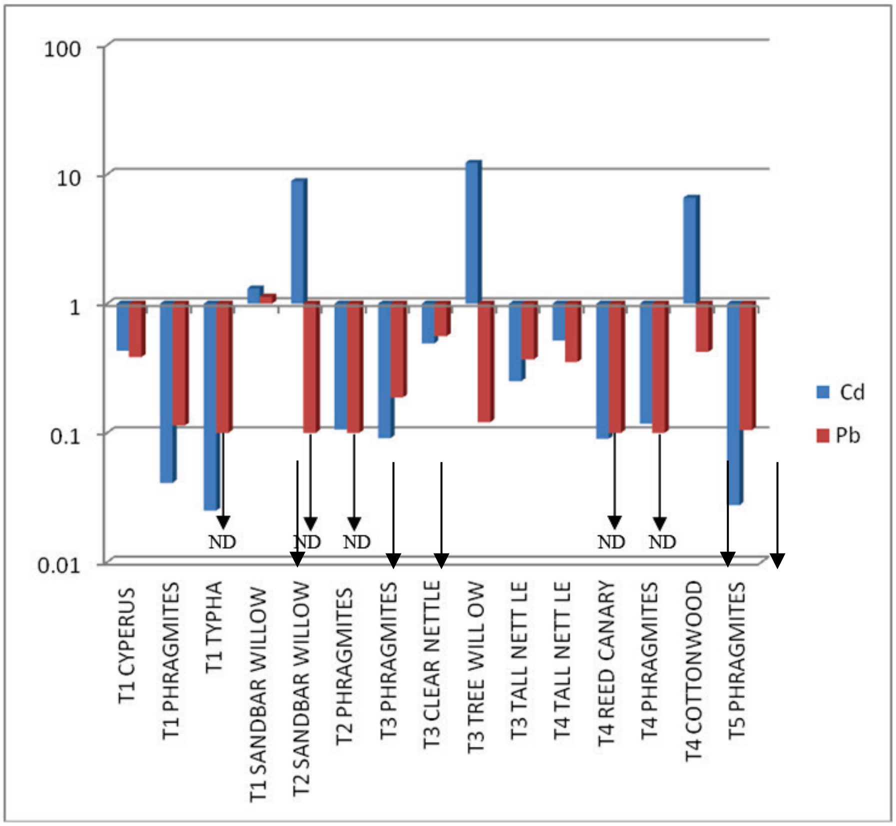
[1] Concentrations in mg kg^{-1}.
† Soil concentration exceeds Eco-SSL for avian herbivore, Table 2.
‡ Concentration exceeds C_{plant} concentration from Table 2.
[oe] Eco-SSL over estimates risks through soil-to-plant pathway.
[ue] Eco-SSL estimated Cp underestimates risks through soil-to-plant pathway.

Since plants are a primary route of exposure from soil to wildlife, a thorough assessment of risk associated with this pathway is necessary for any use of dredged material for vegetated habitat. It is undetermined specifically the factors influencing the elevated Cd and Pb in plant tissues from the test soils discussed above, but soil concentration alone was certainly not a factor. It is clear that simple screening level evaluations may not provide the necessary level of accuracy and factors that influence extractable metals in soils — soil pH, cation exchange capacity, redox potential, fertilization, synergistic/antagonistic interaction between metals and others — may be too complex to model for any application above a watershed level.

Another factor in the plant pathway affecting the potential risk to wildlife is the plant itself. While the plant bioassay provides a conservative approach in assessing potential risks to exposed wildlife it is also recognized the test plant (*Cyperus esculentus*) is not representative of all plants that may be present on a CDF or other site where dredged material may be used beneficially. The ability of plants to extract metals can vary from plant to plant. Figure C-4 shows a CDF near Detroit, MI, where indigenous plants were identified (Price et al 2005) along five distinct transects (T1–T5) of colonization and leaf tissues of predominant species were analyzed for metals. Results for Cd and Pb (Figure C-5) show uptake of lead for all species ranged from <0.1 to 0.56 mg kg^{-1} except for *Salix exigua* (sandbar willow) in the more anaerobic T1 transect, which accumulated 1.1 mg kg^{-1} of lead. Cadmium accumulation was significantly higher in the woody Salix species (sandbar and tree willows) and *Populus deltoids* (eastern cottonwood) when compared to herbaceous species growing in transects T2, T3, and T4 and would exceed the C_{plant} limit for herbivores. Price (2005) noted heavy deer grazing of sandbar willow, *Typha glauca* (Typha) and *Phragmites australis* (Phragmites) and evidence of other mammalian herbivores. Given the more than two orders of magnitude difference in Cd concentrations between these three plants, it becomes important to identify plant species present and being consumed, as well as rate of plant consumption by relevant wildlife receptors, when assessing the risks to wildlife for evaluating use of dredged material for wildlife habitat.

Figure C-4. Distribution of plant species along transitional elevations within a CDF.

Figure C-5. Tissue concentrations of cadmium and lead growing in a CDF near Lake Erie in Michigan.



Note that this is not a critique of the usefulness of Eco-SSLs. It is a critique of the use of Eco-SSLs for evaluating the suitability of dredged material for ecological beneficial uses. Eco-SSLs were developed specifically to be used during Step 2 of the Superfund ecological risk assessment process (USEPA 2003) to determine if additional ecological site study was warranted. Eco-SSLs were not designed to be used as clean-up levels for hazardous waste sites nor should be adopted as generic clean-up standards. As stated previously, Eco-SSLs were not developed using exposure effects data from COC concentrations in dredged material, and it is expected that the different physico-chemical and biological properties of dredged material compared to naturally occurring soil, hazardous or otherwise, would result in differing effects on COC availability to plants. It may be appropriate to use Eco-SSLs as a screening tool in a Tier 1 or Tier 2 level evaluation of COCs in dredged material to determine the need for Tier 3 biological exposure evaluations when dredged material is simply being disposed in a CDF. If bioaccumulation of COCs is a concern, CDFs can be managed to eliminate pathways of COC exposure, mobility, and transport outside the CDF. However, if dredged material has the beneficial purpose of providing habitat for ecological receptors, it would be inappropriate to use a screening tool developed from a database not relevant to the material for which long-term risks are being evaluated. Currently, a COPC database for exposure/effects in dredged material has not been developed for terrestrial receptors on upland beneficial use of dredged material.

Conclusions

The more than 250 Mcy of sediment dredged annually from federal navigation channels provides opportunity for beneficial use, particularly for habitat restoration/creation. While only 3 to 5 Mcy of sediment is removed annually from Great Lakes navigation channels, the volume of dewatered sediment in Great Lakes CDFs provides readily available, nutrient-rich soil material that can be removed to restore degraded upland lands or can be used in situ to provide productive wildlife habitat. Each dredged material proposed for habitat use, whether as part of dredging operations or removed from CDFs, must be evaluated to ensure bioaccumulation of COPCs poses no unacceptable ecological risk, especially to sensitive species. Screening-level approaches can provide rapid, low-cost alternatives to more time-intensive laboratory exposure studies. Eco-SSL screening of soil metals potentially mobile through the plant uptake pathway to wildlife receptors was performed on three dewatered sediments from CDFs and three reference soils in the Great Lakes. Plant bioassays were performed, and tissue concentrations of metals in exposed plants were compared to concentrations of unacceptable risks to avian and mammalian herbivores. In the Eco-SSL evaluation, Toledo CDF soil Pb was shown to exceed the soil limit that would result in plant bioaccumulation posing ecological risks to herbivorous receptors. This was not confirmed by the actual tissue concentrations from exposed bioassay plants. However, plant tissue concentrations of Pb from Lorain CDF and Cleveland REF soils did exceed ecological risk levels for herbivorous birds, even though soil concentrations did not. None of the soils tested exceeded soil Cd Eco-SSL risk levels, but Toledo CDF exposed plant tissue levels did. These results demonstrate the significant uncertainty in utilizing Eco-SSLs to determine ecological risk in dredged material beneficial use evaluations. Note that the focus of this study was on herbivorous receptors (mammals and birds). However, for some of the metals considered in this study, the USEPA ecological soil screening evaluations indicated that insectivorous receptors were more sensitive and may require lower Eco-SSLs for their protection. At the onset of a screening level ecological risk assessment, the lowest Eco-SSLs are typically used to screen for potential risks. One potential drawback to using the lowest Eco-SSL for screening purposes is that several of those hypothetically calculated Eco-SSLs are within the lower range of background or ambient concentrations of metals in soils across the United States (USEPA 2003).

In addition, the demonstrated variability of bioavailable soil metals in the CDF and REF soils tested and variability of metal uptake by various plants in previous dredged material studies provides evidentiary support for exposure testing as a more confident evaluation to determine suitability of dredged material for wildlife habitat. More research is needed to develop a better understanding of factors driving plant extractable metals in dredged material so Eco-SSLs or other screening level approaches can be adapted with greater confidence to Great Lakes dredged material evaluations. This may be more easily adapted at a watershed level and until such time plant bioassays, in conjunction with acceptable tissue concentration consumption limits, can provide a more definitive assessment and communication of ecological impacts in determining beneficial use suitability and habitat management.

Appendix D: Treatment of Impaired Sediments

Introduction

This appendix contains a synopsis of available sediment treatment technology alternatives, a short history of their development, and key operational characteristics; this information was included as an appendix to the testing manual to provide a complete overview of the current status of sediment treatment and its potential relevance to treatment of sediment intended for beneficial use, as of the time of preparation of this manual. Technically, all of the technologies discussed here have potential relevance to treatment of dredged material to facilitate beneficial use. From a logistical and economical perspective, however, the choices are more limited. A key issue is the limited processing capacity of smaller, mobile plants necessary to provide treatment within proximity of dredging or disposal sites, versus the cost to construct and operate a larger, fixed-based plant and to transport material to the plant to sustain operations. From a business model perspective, consistent and high volumes of feed material are needed over an extended period (typically 15 or 20 yr or more) to recover cost of equipment and provide a profit motivation. This has not been a good fit for management of navigation dredged materials, produced in multiple, diverse locations, on an intermittent basis, with uncertain funding levels from year to year. The second and larger issue is the cost of treatment, which is generally too high to justify use for management of moderately contaminated materials where contaminant exposures can be managed by other means. Nevertheless, treatment remains an attractive goal with the potential to facilitate beneficial use of significant volumes of dredged material otherwise considered unsuitable for beneficial use. There are technologies available that are sufficiently economical for this application and, as more is learned about ways to optimize treatment, other alternatives may be available in the future.

Considerations in technology selection and feasibility evaluations are discussed in this appendix, in addition to the technical and logistical challenges and limitations and approximate unit cost of treatment for different technology types; actual treatment costs are somewhat site specific, and the costs presented in this appendix are intended primarily to provide a general cost range to inform preliminary feasibility level evaluations. Case studies are provided where available; most were taken from a remediation setting as this is the context within which treatment has been most viable, from a cost perspective, and necessary from an environmental perspective.

A brief history of sediment treatment technology development

A synopsis of the history of sediment treatment is provided to convey the real challenges involved in development of effective and economic sediment treatment and the limitations of the treatment processes themselves; while conceptually a very desirable objective, technically and economically the obstacles have proven to be difficult to overcome in all but very specific applications — most in the remediation setting. Despite a significant investment in technology development over the past 35 yr, treatment processes capable of contaminant reduction or destruction have only recently begun to approach commercialization in the United States (Estes et al. 2011). Treatment, while theoretically desirable, has proven to be difficult to implement in the context of navigation dredging for various reasons (Estes et al. 2011), including the following:

- Complexity of the sediment matrix and the presence of multiple contaminants
- Logistical issues, including disparate treatment and dredge production rates, large staging area and storage requirements, treatment plant siting restrictions, lack of treatment mobility
- Cost and logistics of dredged material transportation, process pretreatment and management of secondary waste streams generated during treatment
- Treatment cost versus cost of disposal
- Limited beneficial use opportunities for some decontaminated sediment products, market uncertainty for products, and the lack of a uniform treatment standard
- High initial capitalization costs coupled with short-term, intermittent, or scattered demand
- Uncertainty regarding technology performance and cost.

Sediment treatment was considered by USACE as early as the 1970s to minimize impacts of open water disposal of contaminated dredged material. Multiple technology development programs followed (Estes and McGrath 2014), including the following:

- Assessment and Remediation of Contaminated Sediment (ARCS)
- Superfund Innovative Technology Evaluation (SITE)

- Contaminated Sediment Treatment Technology Program (CoSSTEP)
- Water Resources Development Act (WRDA) Sediment Decontamination Demonstrations
- The New Jersey Department of Transportation (NJDOT) Office of Maritime Resources Sediment Technology Decontamination Demonstration Program.

A brief description of each of these programs is provided in Appendix A of Estes et al. (2011). The majority of the research conducted on sediments under these programs was conducted on a small (bench or pilot) scale. The largest sediment-specific demonstrations were conducted under WRDA and the NJDOT Sediment Decontamination Demonstration Programs. Even these demonstrations were relatively limited in terms of the volume of sediment treated and the period of operation, although some of the equipment may have been of sufficient scale to serve as part of a full-scale plant. Only the physico-chemical process was operated at a full-scale processing rate (190,000 m^3/yr, or 250,000 yd^3/yr), and even in that case, continuous operations were limited to a few days at a time. The maturity, applicability, and commercialization potential of these processes were evaluated under the Dredging Operations and Environmental Research program (Estes et al. 2011).

The economics and operating conditions are somewhat more favorable to treatment for sediment remediation projects, where the cost of hazardous waste disposal has provided a more level playing field between treatment and disposal, and production rates can be reduced. While typical navigation dredge production rates may range from 800 to 1500 m^3/day (1,000 to 2,000 yd^3/day), an environmental dredge may produce only 80 to 400 m^3/day (100 to 500 yd^3/day). Even at this reduced production rate, the environmental dredge may be limited to operating at 40% to 50% of capacity to keep the scale of land-based operations reasonable. Significant surge capacity and/or storage areas are typically required to address this disparity (Estes et al. 2011).

Recent developments suggest that there may be dewatering technologies available in the near future that are capable of handling full-scale dredge production; however, this has not been confirmed[*]. As dewatering is generally the rate limiting step for physical separation processes that may

[*] Michael Hodges. 13 July 2015. Personal communication. Chief Technology Officer. Genesis Water.

be needed for some beneficial use placement options, this could represent a significant step forward operationally; however, associated water treatment requirements and costs likely will limit usefulness of technologies with a discrete water separation step to very specific cases.

What constitutes treatment?

Legal and regulatory requirements, public perception, and risk reduction objectives all influence what processes qualify as *treatment*. The most rigorous and commonly accepted definition of treatment refers primarily to processes capable of achieving contaminant extraction or destruction. A broader definition would also include any processes that achieve site-specific treatment objectives and risk reduction by reducing contaminant exposure, mobility, or bioavailability.

From a technical perspective, there are a variety of technologies that can be employed to remove, immobilize, or destroy contaminants. Not all will be feasible logistically or economically, and the effectiveness of a given treatment is typically very site specific. The number and type of contaminants present, the composition of the sediment, the feed and infrastructure requirements of the process, the residuals produced, the treatment objectives, and the attendant costs will influence the suitability of a specific treatment process to a specific material. A feasibility analysis will consider these factors, supported by the results of bench and pilot testing, to identify the best candidates and inform the optimum approach and process configuration.

A treatment train will typically employ a combination of processes to prepare the material and optimize the treatment, although very simplified treatment operations may be feasible in some cases. The primary treatment types of relevance to sediment treatment include the following:

- Separation/volume reduction
- Soil washing
- Stabilization
- Contaminant destruction.

These processes are well described in multiple publications, including (among others) USEPA (1999), Estes et al. (2004), USEPA (1996), ITRC (2011a) and Estes et al. (2011).

There are multiple processes that can be employed for each treatment type, depending upon the complexity of the separation and the target treatment objectives. The aforementioned sediment treatment types and relevant examples are discussed briefly in the following.

Separation

Separation involves the use of technically simple unit operations originally adapted from the mining industry to separate a *clean* and *contaminated* sediment fraction, with the objective of reducing the volume of contaminated material requiring more costly management.

A typical separation circuit will include equipment to remove oversize material (cobbles, gravel, plastic, debris), followed by operations to separate the sediment solids by size and/or density.

<u>Screening</u> – Screening is primarily a size-dependent process, for example, and can be accomplished on dry material or on slurried material, although efficiency is generally higher for wet processes. Some screens are equipped with spray bars to break down agglomerated clays and wash fine particles from the surface of coarser material as the feed passes over and through the screen.

<u>Hydro-separation</u> – By contrast, a hydro-separator achieves a specific separation based upon the size *and* density of the individual particles. Hydrocyclones and upflow/teeter bed separators are examples of hydroseparators. Typically, this type of separation will not achieve as definitive a *cut* as screening — a portion of the fine material will be carried over into the coarse process stream and vice versa, as a function of the individual particle size and density. A fine sand particle might be *seen* by the process as equivalent to a larger, less dense organic particle — such as coal fragments — for example. The greater the density differences between the particles, the easier and more efficiently they can be separated.

Separation, while relatively straightforward, does have some limitations, including the following:

- Size limitations on the material that can be fed to the plant.
- Mechanical dewatering can be costly, problematic, and rate limiting.
- Separation is not a high efficiency process — effective separation requires distinguishable fractions.

- Contaminants are concentrated in a reduced process stream, but not destroyed.
- Wet processes have a high associated wastewater volume that must also be managed.

Often, the treatment objective is to recover the sand fraction — assumed to be least contaminated, and readily dewatered by gravity — and to dewater and dispose of the fines — assumed to be more contaminated and requiring long settling times or mechanical dewatering prior to disposal or off-site transport. The dewatering step is usually the rate-limiting process; thus, the availability of a CDF for management of the water and fines is a significant advantage in terms of processing rates and cost. (See Estes et al. [2002] for further discussion of cost/benefit analysis for physical separation processes.)

Erie Pier

A simple separation process has been in place at the Erie Pier CDF, in Duluth, MN, for many years. At the Erie Pier CDF, mechanically dredged sediment is offloaded on the high side of the CDF and washed down a sluiceway graded into the settled material, using water pumped from the pond on the lower end. Sand settles out within the sluiceway and is recovered with a front-end loader.

At Erie Pier, the lack of a discharge permit for release of the process water limits the total volume of material that can be processed each year. Although the useful life of the CDF is still finite, the sand separation and removal operation has effectively extended the useful life of the facility well beyond the original design life.

Miami River project

A modular separation and dewatering plant was designed and constructed by Boskalis Dolman, a Dutch sediment processing company, and was employed to handle and process sediments dredged from the Miami River (Averett and Estes 2011). Due to contaminant levels in the sediments, the material was determined to be unsuitable for open-water disposal or beneficial use. There were also no suitable areas nearby available for long-term upland confinement. The Miami-Dade Department of Environmental Resources Management approved disposal of the material at a solid waste landfill, and the local sponsor made available an 8.5-acre staging area for dewatering and rehandling of the dredged material prior to transport for disposal.

A backhoe dredger was used to excavate the sediment, which was barged to the processing facility. Figure D-1 illustrates the sequence of unit operations in a block flow diagram. Oversize material and debris were removed by a grizzly and trommel. Following this, gravel was separated on a scalping screen. The remaining slurry was processed through hydrocyclones, with the sand (underflow) then passing over a dewatering screen and the fines (overflow) mechanically dewatered with belt filter presses. Makeup/wash water was added at several points in the process. Aerial views of the plant are shown in Figures D-2 and D-3.

An estimated total of 720,000 yd^3 of sediment was removed at a cost of approximately $80 million, with the final total cost pending remaining contract settlements (Averett and Estes 2011). More than 90% of the sand produced by the plant was clean enough for use as a final landfill cover; the dewatered fines were determined to be acceptable for use as daily landfill cover (Averett and Estes 2011).

Figure D-1 Block flow diagram, Miami River dredging project (Averett and Estes 2011).

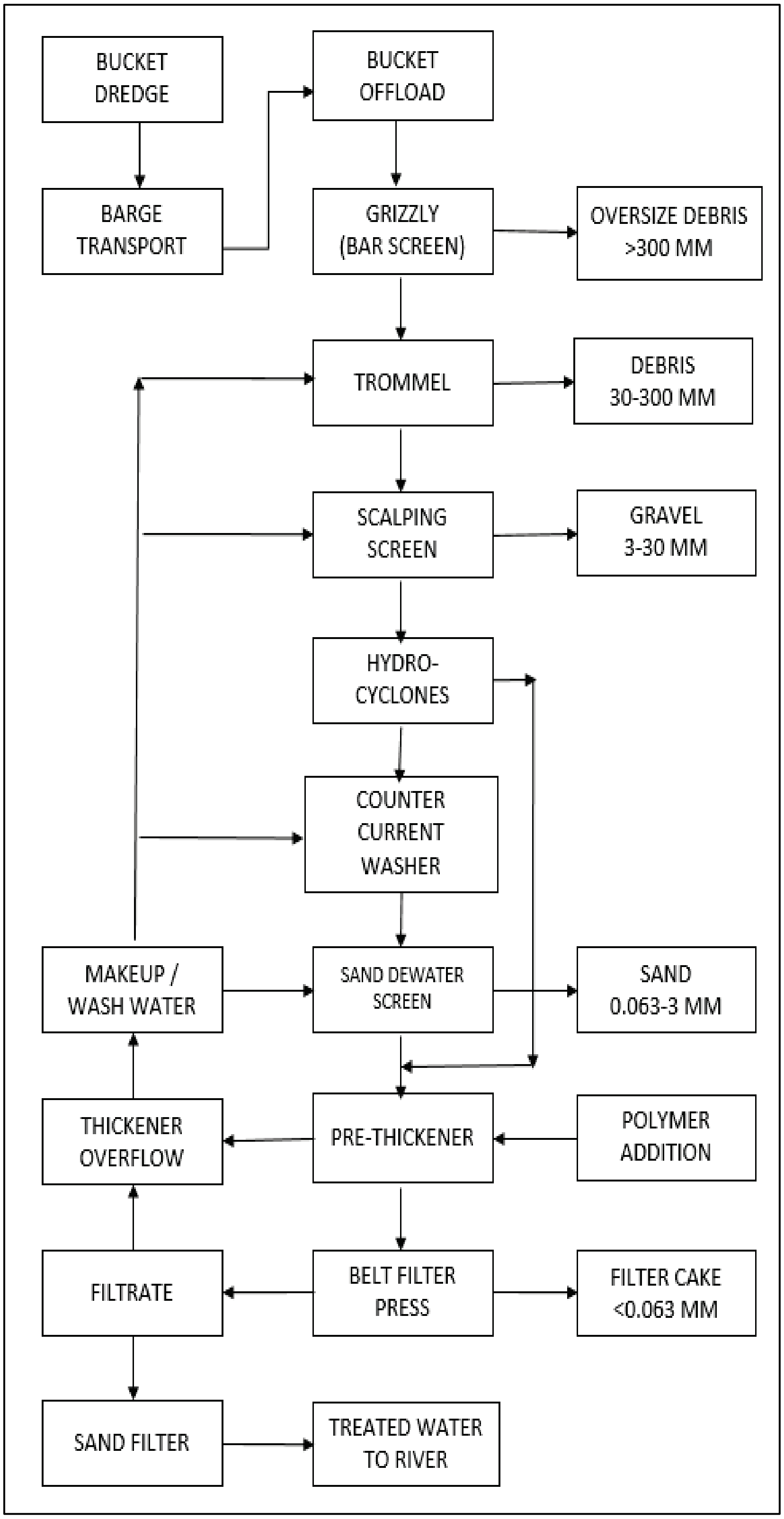


Figure D-2. Photograph of Boskalis-Dolman physical separation system, Miami River, Florida (courtesy Bastiaan Lammers, Boskalis Dolman).

Figure D-3. Photograph of Boskalis-Dolman vibrating screens, hydrocyclones, and sediment processing system, Miami River, Florida (courtesy Bastiaan Lammers, Boskalis Dolman).

Marina Del Rey

Another example of a navigation project employing physical separation is Marina Del Rey, a small-craft harbor located approximately 2 mi north of Los Angeles International Airport. In 2008, a contract was awarded to dredge up to 50K yd^3 of sediment from the harbor's entrance and main channel. The sediment was coarse grained in character, with generally less than 30% passing a No. 200 sieve (75 µ). Contaminants of concern in the sediment included PCBs, pesticides, and heavy metals (Averett and Estes 2011).

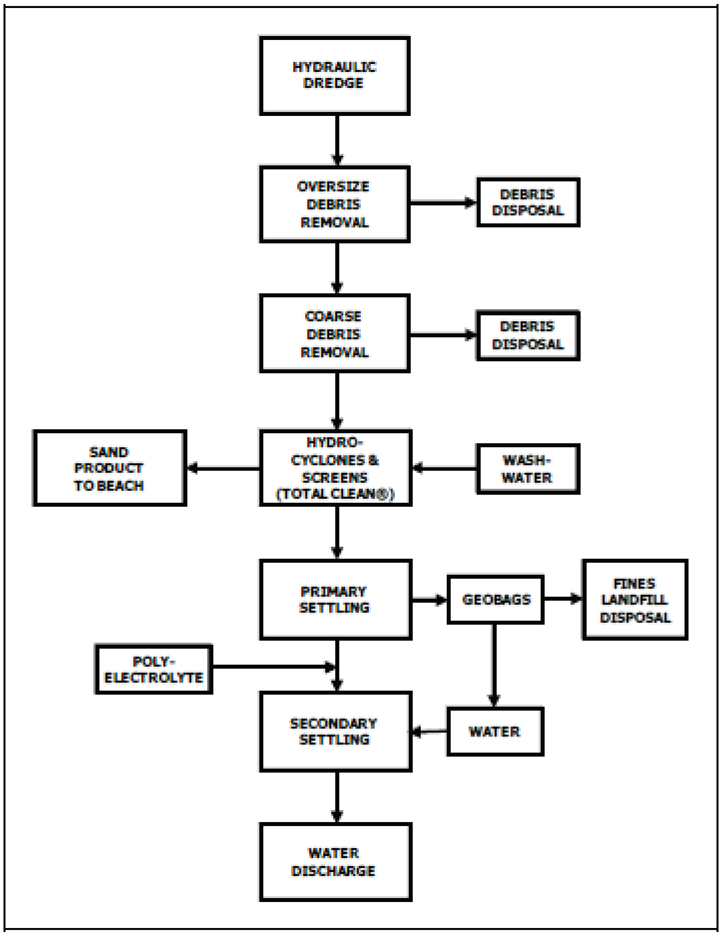
Contaminant levels precluded the sediment from open-water disposal. The US Army Corps of Engineers, Los Angeles District, contracted with CJW Construction, Inc., Santa Ana, CA, to dredge the channel and to employ a physical separation system to recover sand for placement on Dockweiler State Beach, adjacent to and south of Marina del Rey.

An Innovative Material Systems Versi-Dredge, Model 7012, nominal production capacity 350 yd^3 (270 m^3) per hour through a 25 cm discharge line was used; the actual production rate for this project was much less than this, however. The dredge pumped material through a floating pipeline to the shore where it emptied into a coarse screen fixed to the bottom of a basket, attached to a track-mounted crane. The screen collected mostly trash, particularly plastic bags and stringy debris, which had proved difficult to remove from other screening devices tested in the earliest stages of the project. The screen was operated in a batch-wise manner; when a thin layer of debris accumulated, the crane lifted the basket out of the flow path and emptied the material into a roll-off dumpster. Underflow from the screen was picked up by a centrifugal pump and transported 2 mi along the beach to the physical separation plant.

The heart of the physical separation operation was a package unit — Total Clean® TCW-3000 — consisting of hydrocyclones, vibrating screens, settling tank, and feed/recirculating pumps, furnished by Del Tank and Filtration Systems. The Total Clean incorporates a V-shaped tank with baffle system and screw that functions as a clarifier. A shaftless screw at the bottom of the tank moves settled solids to the suction of the hydrocyclone feed pumps. The hydrocyclones remove the coarsest materials — sands and silts — as underflow, which is subsequently dewatered on vibrating linear screens. The hydrocyclone overflow contains fine and light particulates and the majority of the process water; the underflow is recycled to the tank for additional settling and particulate capture prior to overflowing the tank. A schematic of the process is provided in Figure D-4. A more detailed process description can be found in Averett and Estes (2011).

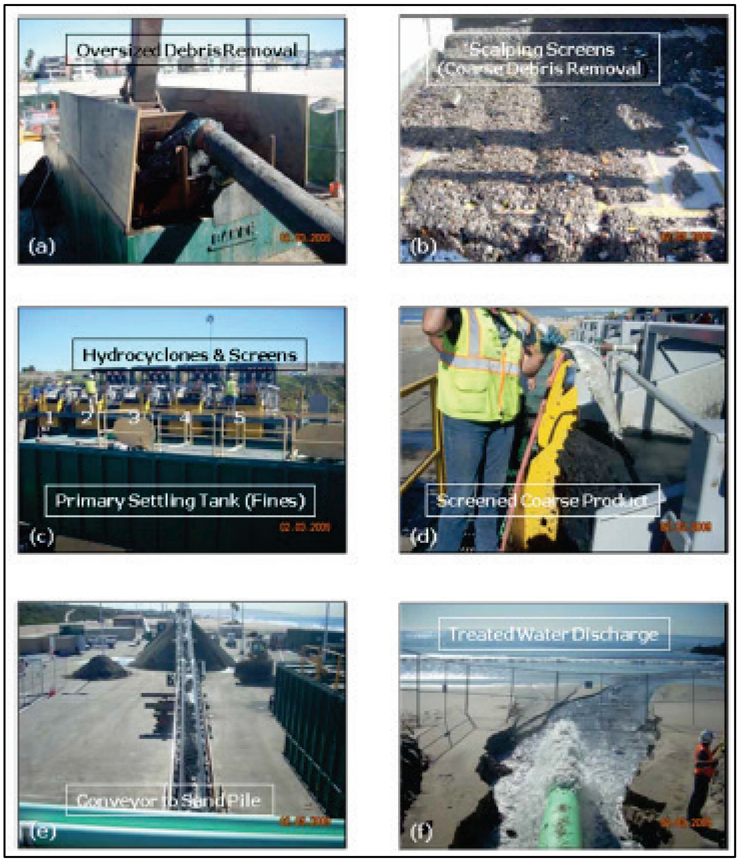
Sandy material from each screening unit discharged onto a common conveyor belt (Figure 1.5(e)), transporting the sand to a paved storage area for stockpiling and gravity dewatering, although very little free water was released. Although residual concentrations of chlordane in the sand fraction were above some regulatory thresholds, toxicity testing showed limited bioavailability and the intent is to place the sand on the beach as originally planned (Averett and Fields 2009).

Figure D-4. Process schematic for physical separation plant at Marina Del Rey.



Overflow from the hydrocyclones and screening unit discharged to two Baker tanks for primary settling (Figure D-5 (c)). Overflow from the primary settling tanks flowed into eight 80 m^3 Baker tanks for secondary settling and final clarification before discharging to the Pacific Ocean (Figure D-5 (f)), following confirmation that the effluent met regulatory criteria. Underflow from the primary settling tanks was initially pumped to a belt filter press for dewatering. However, operational problems with the belt filter and with pumping the settled material from the settling tanks, resulted in replacement of the filter press with geobags for dewatering of the fine grained materials.

Figure D-5. Individual unit operations, feed, and process streams for physical separation plant at Marina Del Rey.



Dredged material processing began in late December 2008 and concluded in March 2009. The dredge operated on an intermittent basis due to start-up, operational, and regulatory issues. The contractor dredged an estimated 8,000 m^3 of sediment from the channel and recovered approximately 70% of the dredged material as sand. While the goal was to operate on a 24 hr/day schedule, the project did not advance beyond 12 hr/day operations due to noise restrictions and operational issues at startup.

Dealing with debris was one of the major start-up issues. In particular, stringy materials like plastic bags, wire, and cables tended to initially hang up or get wrapped around the dredge's horizontal cutter and the bar rack over the dredge's pump intake. The dredge pump had to be shut down and the material manually removed. Some of this material was entrained in the dredged material slurry and ultimately pumped to the processing plant. Stationary screens, requiring manual cleaning, were used at first. These were abandoned in favor of the crane-mounted basket discussed in the process description, which proved to be effective in capturing the oversize debris with limited operational difficulty and without interruption of dredging.

Difficulties in pumping the material after it had consolidated in the primary settling tanks and unsatisfactory operation of the belt filter press was another significant issue. The settling tank did not include any kind of raking mechanism to keep material moving to the pump intake; as material accumulated, it became too consolidated to effectively pump to the filter press. (Percent solids of the material fed to a belt filter press is an important process variable; the material must be sufficiently porous to release water but sufficiently flocculated to prevent it from passing through the filter belts.)

New Bedford Harbor

New Bedford Harbor, in New Bedford, MA, was listed as a National Priorities List site in 1983, as a result of PCBs discharges associated with area electrical manufacturing. Dredging of a 5-acre *hot spot* at the northern end of the Acushnet Estuary took place from 1994 to 1995. The final Record of Decision (ROD) issued in 1998 identified dredging and shoreline containment of approximately 450,000 yd^3 sediment and wetlands soils as the remedy for the upper and lower harbor areas. Four CDFs were to be constructed to hold the dredged sediments.

Issues related to the proposed CDF construction sites led to a change in the ROD, with sediments instead being sent to an off-site landfill. Sediment was dredged hydraulically, using an 8 ft Ellicott Mudcat auger dredge, and the sediment was dewatered in preparation for transport off site. Full-scale dredging began in 2004; 2011 marked the USEPA eighth season of hydraulic dredging in New Bedford Harbor (https://www.epa.gov/new-bedford-harbor). By 2012, approximately 230K yd^3 of a projected 900K yd^3 of contaminated sediment had been dredged.

The dewatering process – Separation and dewatering processes differ primarily in treatment objectives; often the unit operations are very similar, if not the same. A separation process is operated with the objective of recovering a target sediment fraction, and dewatering is a necessary component of waste stream management. A dewatering process is operated with the primary objective being to reduce the water content of the sediment but will typically employ some type of separation to remove coarse sediments that can be effectively dewatered by gravity and that are also abrasive and damaging to the dewatering equipment.

At New Bedford Harbor, large debris such as tires, bricks, etc., are removed from the harbor bottom prior to dredging to avoid any damage to the equipment. Dredge slurry is pumped to a de-sanding circuit, consisting of a series of screens. The coarse material is stockpiled prior to transport offsite to a licensed disposal area. The fine material and process water is pumped to filter presses for dewatering; filtrate is treated and discharged back to the harbor, and fines are transported to a licensed PCB landfill in Michigan. (A short video is available on the USEPA website https://www.epa.gov/new-bedford-harbor.)

In some sediments, sand is relatively uncontaminated compared to the fine sediment fraction. At the time that the New Bedford dredging project was being planned, off-site transportation and disposal costs were approximately $60/ton less for materials with PCB concentrations below 50 mg/kg (under the TSCA, this is the threshold defining hazardous materials). One of the initial processing objectives was to produce a sand fraction with PCBs below the TSCA level, such that less restrictive disposal or possibly beneficial use could be considered.

The coarse fraction contained fine sand and coarse organic particulates, the relative size and density of which proved difficult to separate sufficiently to achieve the target PCB reduction in the coarse fraction. The original plant was equipped with hydrocyclones discharging onto dewatering screens; two different hydrocyclone types were ultimately tested in an effort to improve the separation. Based on information now available on the USEPA project site, only the fines are now being placed at a PCBs disposal facility, suggesting that subsequent processing modifications were successful.

Lower Fox River

The Fox River, in Green Bay, WI, is a remediation project involving several miles of PCB-contaminated river sediments, through a combination of dredging, capping and monitored natural recovery (Feeney et al. 2011). A full-scale processing plant was erected over a 6-acre area on a site in Green Bay, WI, to separate and dewater the nearly 4M yd^3 of sediment scheduled to be dredged from Operable Unit (OU 2-5). A key element of this aspect of the project is the formal integration of general, marine and sediment processing contractors. Given the interdependence of dredging and plant processing rates, this is potentially a significant advantage in terms of cost avoidance and operational efficiency.

The plant is fed by a 12 in. cutterhead hydraulic dredge and two 8 in. hydraulic dredges; the plant itself has a maximum capacity of 9000 gpm, at a 5%–10% solids concentration in the feed. The processing plant was designed with redundancy in the sediment separation and dewatering operations, as well as the water treatment unit, to minimize impact [of equipment breakdown or batch cycle times] on dredging rates. Over a period of 6.5 months, approximately 545K yd^3 of contaminated sediment were dredged and processed in the first year of operation (2009). More than 720K yd^3 were dredged and processed in the following year.

Figures D-6 – D-9 depict various components of the plant. The dredge discharge enters the plant and passes over the vibrating screens where oversize material (>6 mm is removed). The <6 mm particles pass through the screen and are then processed through multiple hydrocyclones, producing coarse and fine sand fractions (150 µm to 6 mm and 63 µm to 150 um). The sand fractions are further polished in upflow classifiers, which remove additional fine and lower density particles that may contain elevated levels of contaminants. The fine fraction produced by the hydrocylones goes through a thickening step and is dewatered in the membrane plate and frame presses having a capacity of 14 yd^3 solids/hour/press with a cycle time of 75 min.

Dewatered fines with PCBs above the TSCA threshold are transported by truck to a licensed facility in Michigan, over 400 mi away. Materials with PCBs below the TSCA limit are disposed of at a location approximately 30 mi from the plant. Process water is recycled through the plant; excess water is treated and discharged to the river following confirmatory chemical analysis.

Figure D-6. Vibrating screens receive discharge from hydraulic dredges, removing oversize materials prior to pumping to the hydrocyclones.

Figure D-7. Hydrocyclones for sand separation.

Figure D-8. The plant is equipped with eight, membrane plate and frame filter presses with a production capacity of 14 yd^3 solids/hour/press and a cycle time of 75 min.

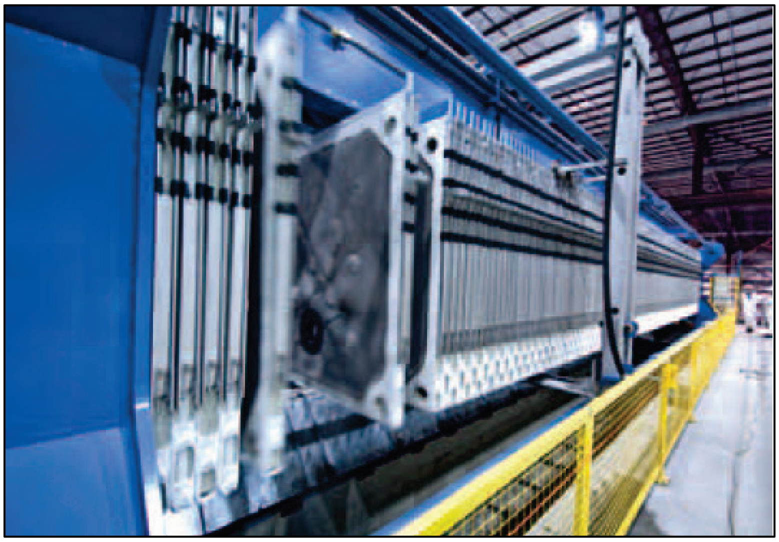


Figure D-9. Water treatment plant, 9000 GPM capacity, sand, bag, and carbon filtration processes.

Soil washing

USEPA defines soil washing as “a technology that uses liquids (usually water, sometimes combined with chemical additives) and a mechanical process to scrub contaminants from soils” (USEPA 1996). Soil washing has also been considered as a means of desalinating marine sediments for beneficial use (Estes et al. 2002). Like separation, the soil-washing process separates target size and density fractions using many of the same unit operations. In addition, soil washing transfers contaminants from the solid phase to the aqueous phase through addition of surfactants or other reagents and attritioning. Some degree of washing will occur with any separation process conducted on slurried material; the large associated wastewater stream is considered to be one of the chief disadvantages of separation and soil washing. The soil-washing process differs only in that additional reagents and processes are employed to enhance physico-chemical removal of the surficial contaminants from specific sediment fractions. Various pieces of equipment are available to enhance attritioning and are generally a necessary part of an efficient soil-washing process.

As in simple separation processes, the first step is to remove oversize materials and debris. The remainder of the material then proceeds through an attritioning process and further separation and dewatering processes, selected based on the sediment characteristics, contaminant distribution and the processing objectives. Treatment cost may be reduced and process efficiency increased if the target sediment fraction undergoes a preliminary separation from non-target fractions, thus reducing the quantity of reagent required in subsequent cleaning steps. Typically, there will be solid and wastewater residual waste streams that also require management/disposal, in addition to the clean product produced; often, wastewater is recycled through the plant to minimize make-up water and wastewater volumes.

Soil washing is a relatively mobile and scalable process. Soil washing is most effective in cleaning the coarse fraction of the sediment, although processes have been developed that also treat the fine fraction of the sediment. Where recovery of a clean sand fraction is the primary treatment objective, the initial sand content of the sediment is an important consideration in evaluating sediment treatability. Higher sand content suggests a sediment more suited to soil washing than low sand content; however, it is also possible that high sand content materials could be used beneficially without treatment in less restrictive applications, such as those where some type of containment will be used. An example would be use as a subfill layer, to be covered with clean material and then asphalt. These considerations are discussed in more detail in the Feasibility/Pilot Testing section of this chapter.

Saginaw River demonstration

One of the early demonstrations of soil washing was conducted under the ARCS program with a barge-mounted pilot scale plant, nominal capacity

5 tons/hr. "Approximately 800 yd3 of partially dewatered mechanically dredged material was barged to the site and used as plant feed" (Estes and McGrath 2014). The plant operations included the following:

- Conveyors for transport and manual debris removal, later replaced with a 2 in. grizzly with attritioning
- A rotary trommel
- 9 in. Linatex separators (hydrocylones)
- A Linatex hydrosizer or dense media separator
- An attrition scrubber (surfactant sand washing step)
- A second set of hydrocyclones
- Sand recovery and dewatering screens
- A clarifier with flocculent treatment.

Because this was a pilot test conducted to increase understanding regarding the effectiveness of various unit operations, plant operations were not optimized to the specific sediment being processed. Some process inefficiencies noted during this pilot test could successfully be addressed with appropriate sediment characterization and equipment selection and sizing. Larger hydrocyclones would likely achieve more efficient sand/fines separation. The clarifier was determined to be the rate-limiting step in the process but would be more difficult to scale up to accommodate higher flows due to the cost and footprint required. Where a disposal facility is available for settling and consolidation of fines, this could be a significant cost advantage over projects requiring a thickening and dewatering step.

Ultimately, 80% of the feed solids were recovered as washed sand, and PCBs were reduced by 82% in this fraction. TOC was reduced by 79%, metals from 55% to 88%, and fines <75 µm by 77%. Projected costs for a 50 ton/hr plant were provided in the project report — ranging from $23 to $54/cy for 100,000 and 10,000 yd^3, respectively, but more recent and comprehensive cost estimates for soil washing are available in Estes et al. (2011).

Stabilization

Solidification/stabilization (S/S)

S/S is a well-established remediation technology for treatment of contaminated soil, sediment, sludge, and waste (i.e., contaminated material). S/S is a process of blending treatment reagents into contaminated material to impart physical and/or chemical changes that result in reduced environmental impact of the contaminated material to groundwater and/or surface water (ITRC 2011a). Generally, metals immobility is achieved through pH and alkalinity changes, reducing the solubility of cationic metals. Anions are more difficult to bind in insoluble compounds, however. A number of amendments may also be added to sorb organic contaminants, including organically modified clays, activated carbon, ferric hydroxide, and rubber particulate, among others (Maher et al. 2005). While effective at immobilizing contaminants within the sediment matrix, S/S does not achieve any contaminant destruction. One case study evaluating the combination of chemical oxidation and S/S was conducted on Newark Bay dredged material; more information on that demonstration can be found under S/S case studies, below.

Until recently, one of the principal shortcomings of S/S for use in remedial applications was a lack of long-term performance data and performance standards. A recent guidance document was developed and published by the ITRC (2011a) to address these deficiencies. The ITRC document is focused on in situ application of S/S in which pozzolonic materials are added to a soil or sediment, resulting in solidification of the matrix, reduced hydraulic conductivity and reduced contaminant leaching.

In situ S/S has also been effectively demonstrated on remediation sites and is one of the most commonly used in situ technologies at Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) sites for source control (ITRC 2011a). S/S has also been used to control migration of non-aqueous phase liquids. Only a few states have established regulations pertaining to the use of S/S, and no state-specific guidance was available at the time of publication of the ITRC (2011a) report. As a result, the performance criteria most commonly used (strength, hydraulic conductivity and leaching potential) are not consistently applied.

S/S is also perhaps the most commonly used *treatment* technology used for navigation sediments; in this context, S/S is generally used to effectively dewater dredged material prior to transport offsite for disposal. Sediments can be amended directly in the transport barge with conventional mixing equipment (Figure D-10).

Figure D-10. Mixer for in-barge mixing of amendments for sediment S/S.

Stabilization can also be done prior to dredging to minimize water quality impacts of more highly contaminated sediments.

The application of S/S to dredged material as a treatment prior to beneficial use differs somewhat from the remedial context given differences in the level of contamination in the sediments, and differences in the treatment objectives. Sediments sufficiently contaminated to be classified as hazardous waste would not normally be considered for any type of beneficial use. For a remedial site, containment and risk reduction are the primary, though not necessarily the only, objectives. For beneficial use, there are usually engineering and ecological site functions that must also be considered. S/S significantly alters the pH, permeability, and friability of the sediment matrix, such that S/S treated sediments may not be suitable for use in ecological restoration settings where more *soil-like* characteristics are needed.

For a more complete description of the S/S technology, behavior of stabilized in the environment, performance parameters, design and implementation and long term monitoring requirements, the reader is referred to the ITRC (2011a) document. A number of S/S points of contact can also be found in the appendices of that document.

S/S Case studies

New York/New Jersey (NY/NJ) Harbor In-Situ S/S of Soft Sediments — A demonstration was conducted on NY/NJ harbor sediments (Maher et al. 2005) to assess the efficacy of deep soil mixing with pozzolonic materials on contaminant mobility, sediment strength, amendment dosages, dispersion occurring during treatment, and impact of high organics content in the sediments. Sediments in the test area were watery, very soft silt with organic content ranging from 12% to 14%. Three cells, approximately 18 ft long × 14 ft wide × 10 ft deep were mixed with three different dosages of cement slurry (100, 150, and 200 kg/m^3); cement was the preferred additive for solidification because it can be mixed in slurry form.

The researchers noted an increase in shear strength, as measured in laboratory tested samples, ranging from 770 to 1,870 psf at 40 days and 1,996 to 2,533 psf at 1 yr. An average moisture content reduction of 40% was also observed. TSS increases occurring in the water column during mixing were confined to within approximately 125 ft of the mixing point (although a site specific evaluation of the need for TSS controls would be needed for other field applications). Improved erosion and resistance to re-suspension of the contaminated sediments was expected, although this was not measured.

The treated sediments had a consistency of hard silt/clay (penetration test *N* values increased from weight of rod to over 40). Maher reported that cement contents of 14% and possibly more could be used without rendering the sediments too stiff to dredge with a conventional bucket dredge following treatment. Maher et al. (2005) postulated that even long-term strength gains would not render the sediment too stiff to be dredged and that S/S could therefore be a viable interim risk reduction measure while the full range of remedial alternatives was being investigated. Cement dosage likely would need to be optimized to take this factor into account, however.

PCBs, PAHs, and dioxins were successfully immobilized in the treated sediment. The high organic content was reportedly an impediment to cement hydration; Maher et al. (2005) concluded that this could be adjusted for with higher cement dosages, based on appropriate pretreatment testing. While not statistically significant, there appeared to be a correlation between elevated TSS levels and elevated pH.

The results of this study are potentially relevant to the use of moderately contaminated sediments as fill for harbor improvement projects or possibly even upland environments. The workability of the material following S/S was not addressed, however. In addition, this study preceded more recent guidance for S/S. A number of important parameters, such as long-term leaching behavior, structural integrity, volatilization, and process optimization were also not addressed in the study. ITRC (2011a) should be consulted for more information relevant to these, and other, performance and implementation parameters.

Harbor Resources Demonstration – This was a demonstration project conducted under the NJDOT sediment decontamination demonstration program. A pilot scale study was initially conducted on approximately 650 gal of dredged material from the Stratus Petroleum site in Newark were treated. A larger-scale project followed, using commercial scale equipment, with an objective of processing approximately 2,400 yd^3 sediment from the Darling International Site. The Harbor Resource Environmental Group (HREG) Dredged Material Process is described as "chemical oxidation for contaminant reduction, moisture removal or dewatering, and beneficial use conditioning through the addition of cement." Potassium permanganate in aqueous solution was used as the oxidant, introduced to the sediment slurry containing 15%–30% solids by weight, at a dosage of approximately 10,000 ppm. Following a 6 hr contact time, the slurry was pumped to a belt filter press for dewatering. A polymer was added to improve dewatering characteristics. Cement was then added at a dose of approximately 7.6% by weight, using a screw-type ribbon blender (HREG 2005). A 7-day start-up/shake-down period was followed with a 5-day processing period. Ultimately, the plant was able to process only a little more than 300 tons of treated material in that time.

Based on average concentrations over the treatment period, analysis of the feed and treated sediment samples was inconclusive with respect to Dioxin reduction. On the single treatment day with the highest observed feed concentrations, the highest semi-volatile organic compound (SVOC) and PCB reductions were also observed (70% and 65%, respectively).

Requirements for the selected beneficial use — fill material for the EnCap Golf development project — included the following:

- Minimum compressive strength of 2,000 lb/ft^2
- Maximum hydraulic conductivity of 1 × 10–5 cm/sec
- Maximum particle size of 4 in.

The treated material met the hydraulic conductivity requirements and exceeded the unconfined compressive strength by roughly 50%. Reportedly, the material was also very friable. Additional findings by HREG (HREG 2005):

- "The results of the multiple extraction procedure (MEP) test revealed that out of approximately 114 data points for PAHs and metals, only Lead exceeded groundwater criteria and only in three of the samples.
- Despite the higher concentrations of Manganese in the treated dredged material, the results of the MEP test indicate that Mn was not leachable (below groundwater criteria).
- All compounds passed TCLP."

Estimated processing costs, based on an annual production capacity of 500,000 yd^3, over a 30 yr period, were estimated to range between $34 and $45/cy (2005 $US), depending upon the contaminant concentrations in the feed and the amount of chemical oxidant required.

In-situ reactive amendments

In situ treatment has long been considered the most desirable approach to remediation of contaminated sediments but also the most difficult to implement and to monitor, particularly for technologies intended to achieve contaminant destruction. Further, treatment reagents may also have unintended environmental consequences.

<u>Activated Carbon (A/C) Amendment</u> — A/C amendment of sediments has emerged as one of the most viable in situ treatment approaches available for sediments contaminated with organic compounds. By reducing contaminant concentrations in the pore water of bioactive zones, A/C application can potentially provide rapid risk reduction at a remediation site, with effectiveness improving over time as a greater proportion of sediment-associated contaminants become irreversibly bound to the A/C.

Other amendments have also been considered that are potentially effective for stabilization of both organic and metal compounds, including organoclay, apatite, biochar, coke, zeolites, and zero valent iron, among others. Calcium nitrate has been field demonstrated for reduction of H_2S in highly polluted waterways, a problem at some CDFs and beneficial use placement sites (Estes and McGrath 2014), although the potential water quality impacts of this technology do not appear to have been studied.

In situ stabilization with A/C has been extensively demonstrated at bench and pilot scale, with over 25 field sites in the United States and Europe; properly implemented, the technology is less costly, less disruptive, and more effective than other alternatives such as dredging or capping (Patmont et al. 2015). In situ stabilization has also been incorporated as a tool in isolation capping as well, however, with thin, reactive caps emerging as preferable to thick isolation caps in some cases. Apatite, Aquablok, sand, and coke breeze were demonstrated in a field capping demonstration on the Anacostia River, with monitoring taking place over several years following placement. Findings from that study were reported in Lampert et al. (2013) and showed, despite higher contaminant transport than anticipated, overall lower pore water concentrations than in control areas. A more recent demonstration on the Grand Calumet river involved partial dredging and capping of approximately 1.3 mi of riverbed[*], which was highly contaminated with heavy metals and PAHs, with a reactive cap composed of sand with a layer of organoclay (final report pending). These and other projects illustrate that in situ amendment of caps and sediments is emerging as customary practice for remediation sites; these technologies have potential also for effective, long-term management of contaminants in dredged material intended for beneficial use in a restoration setting.

Amendments can be applied directly, as a thin layer on the sediment surface, as part of a cover material or mixed into the sediment itself. The specific character of the amendment under consideration will determine the need for some type of carrier, such as sand, to facilitate accurate placement and long-term stability and acceptability of benthic and aquatic exposure to the amendments themselves.

Geochemical Controls – By understanding the biogeochemistry of the contaminants, particularly metals, there is potential to exploit natural

* http://www.epa.gov/greatlakes/sediment/legacy/grandcal/

mechanisms to maintain contaminants in less mobile and less bioavailable forms. This would be particularly valuable in settings where sediments are subject to cyclic wetting and drying, which tends to produce increased contaminant releases during periods of change in the redox state of the sediment.

Although not an entirely new concept, this approach to management of contaminants in a beneficial use application is presently undergoing further research at ERDC. Sediments themselves have a natural ability to control contaminant releases to a greater or lesser degree, depending upon the composition of the sediment. Natural carbonates provide buffering capacity, preventing pH changes that can occur as a result of redox changes in sediments subject to cycling wetting and drying, and leading to mobilization of metal contaminants. Numerous other naturally occurring compounds, such as iron (Fe) and manganese (Mn) complex with metals under certain pH and redox conditions to render them insoluble and biologically unavailable. Zeolites are natural minerals with high ion exchange capacity that may also be useful to provide short-term, rapid-risk reduction as related to metals toxicity.

Natural materials containing key constituents in these chemical reactions are under evaluation in bench scale testing to assess the effect of the amendments on solution pH as a function of dose, as well as the impact on dissolved metals concentrations in pore water and overlying surface water, for varying redox conditions. Because microbial activity also affects the chemical environment in sediments, the integration of microbially mediated reactions is also being considered in developing a template for application of in situ geochemical controls. A holistic approach is needed to exploit the complex mechanisms at work in a sediment under varying conditions. The objective of this research is to develop alternatives applicable not only to in situ treatment at beneficial use sites but also to treatment of contaminated sediments in CDFs.

Contaminant destruction

Contaminant destruction is the gold standard of treatment but also the most costly and difficult treatment objective to achieve. Theoretically, there are multiple processes that can be employed to achieve contaminant destruction, and these generally fall into one of three categories:

- Chemical oxidation
- Thermal incineration/vitrification
- Biological degradation.

Chemical oxidation

Chemical oxidation involves the addition of chemical reagents for the purpose of degrading organic compounds in the sediment. Several examples of oxidation-process demonstrations can be found in the reports of the various technology demonstration programs (Estes and McGrath 2011), including the following:

- Wet Air Oxidation (Zimpro) was demonstrated at bench scale under the ARCS program.
- Electrochemical oxidation was demonstrated at pilot scale under the SITE program (by Weiss Associates).
- Biogenesis physico-chemical (e.g., soil washing) process was demonstrated at bench scale under (CoSTTEP) and at bench, pilot, and full-scale (capacity) under WRDA. Large pilot demonstrations were also conducted on sediments in Europe and the Passaic River, in Lyndhurst (NorthJersey.com[*]).

Although the concept of mineralization of organic contaminants (breaking them down to CO_2 and water) is appealing, the effectiveness of chemical oxidation in sediment has not been clearly demonstrated; a significant reason for this is the non-specificity of the reagents. Reagent dosages must be sufficient to oxidize not only the target contaminants but also competing materials contained in the sediment. Bacterial biomass, total organic carbon, iron, manganese, hydrogen sulfide, and carbonates (ITRC 2005), common constituents of natural sediments, can all be problematic. Natural organic matter, carbonates, humic acids, by-products of oxidation and the reagents themselves (in peroxide overdosing) can all act as scavengers resulting in higher required reagent dosages. Carbonates may also provide significant buffering, limiting the effectiveness of pH dependent processes (Estes and McGrath 2014).

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[*] http://www.northjersey.com/news/despite-pilot-test-failure-lyndhurst-s-river-cleanup-will-continue-1.510770

Note, however, that although competition from natural organics for reagents is acknowledged as a major factor limiting the success of chemical oxidation, the majority of organic contaminants will reside in the organic materials. Destruction of the natural organics may be necessary to access the associated contaminants rather than being a secondary and undesirable reaction. Separation of the organic fractions prior to reagent addition has the potential to improve effectiveness in treating the larger non-organic sediment fraction but generates a separate (though typically much smaller) waste stream requiring disposal and requires an additional treatment process (separation) (Estes and McGrath 2014). In practice, chemical oxidation has not yet been shown to be an effective and economical means of treating/destroying particulate associated contaminants ex situ.

Zimpro Wet Air Oxidation — The Zimpro process was evaluated for effectiveness in treating PAHs in sediments from the Grand Calumet River (USEPA 1994). Although high efficiency in reduction of PAHs was reported, the results also demonstrate one of the key issues with chemical oxidation in sediments — competition from non-target constituents. Post-treatment analysis showed that oil and grease were reduced by 90%, and TOC and volatile solids were reduced by 50%. PCBs were reduced by 30% (although the process was not designed or expected to treat PCBs). The pH of the sediment was also noted to have dropped from approximately 7.6 to 6.5, suggesting that potential for dosage related metals mobilization could be an incidental effect of the process and should be considered. The other key issue is the difficulty that has historically been shown to achieve the same efficiencies demonstrated at bench scale, in full-scale processing. Given the relative ineffectiveness of chemical oxidation in subsequent demonstrations (Biogenesis — Estes et al. 2011; HREG 2005), it seems unlikely that the Zimpro process would fare differently. Cost to treat sediment with the Zimpro process was estimated at approximately \$133/yd^3 (assumed 1994 \$US cost basis) based on the assumptions provided in the report, which did not include site excavation, civil work, pre-screening needs, and overall site management and disposition of the residuals. Clearly, this in itself would preclude use for treatment of sediments for beneficial use. More cost-effective and robust approaches are needed.

Biogenesis — Biogenesis is a more recently developed soil washing technology that also incorporates a chemical oxidation step to achieve contaminant reduction. Biogenesis has been demonstrated at a larger scale and under more sustained operation than any other sediment contaminant destruction technology to date. The process has continued to evolve since it was evaluated in Estes et al. (2011), eliminating the associated wastewater stream to create a completely self-contained process. These are all positive developments. Based on all currently available data and literature on the project, however, significant reduction of organic contaminants continues to be problematic. In 2012, biogenesis was engaged to conduct bench scale testing to evaluate effectiveness in decontaminating Passaic River sediments taken from near Riverside County Park in Lyndhurst, an area which contains some of the most concentrated amounts of dioxin in the river[*]. Another company, Pear Technologies, also participated in this demonstration. Note that the treatment objective in this case was to decontaminate the bulk sediment, not just the sand fraction, a very difficult treatment challenge. The results of the bench scale testing showed some reduction of contaminants of concern but were not successful in meeting treatment targets that would have justified the cost of treatment. USEPA subsequently elected not to employ the technology for the relatively small volume of sediment to be dredged from River Mile 10.9 but left the door open for potential use at other sites in the future[†].

Thermal incineration/vitrification

Three high-temperature thermal treatment technologies were demonstrated under the WRDA Sediment Decontamination Demonstrations program and underwent a detailed evaluation reported in Estes et al. (2011), Minergy (glass furnace technology), Rotary Kiln, and Cement Lock. All three technologies are high-temperature processes, capable of destroying organic contaminants and immobilizing metals. The characteristics of the treated sediment vary with the conditions of processing and the additives used to control the melt. The following paragraphs describe each technology in brief, the beneficial use materials produced, scale of demonstrations, technology maturity, and estimated treatment costs.

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* http://www.northjersey.com/news/feds-say-river-cleanup-in-lyndhurst-fails-to-deliver-1.441008

† http://www.northjersey.com/news/despite-pilot-test-failure-lyndhurst-s-river-cleanup-will-continue-1.510770

Minergy Glass Furnace — The Minergy process was originally developed for treatment of wastewater solids from the paper pulp industry; the process produces a glass aggregate suitable for construction fill and other "beneficial uses such as hot mix asphalt, construction fill, cement substitute, and ceramic floor tiles" (Estes et al. 2011). The process flow diagrams below (Figures D-11 and D-12), were taken from Estes et al. (2011) and illustrate the different stages of material preparation required prior to thermal treatment, and the treatment process itself. "After thermal drying to remove most of the water in the sediment, the solids are melted at high temperature (1,600°C (2,900°F)) in a refractory-lined melter, producing a glass aggregate and effectively encapsulating metals and destroying organic contaminants. Flux materials are added to control melt temperatures and improve the qualities of the molten glass. The molten glass is then quenched to produce a glass aggregate" (Estes et al. 2011). Residence time in the furnace is approximately 6 hr.

Figure D-11. Pretreatment processes for glass furnace technology (taken from Estes et al. 2011).

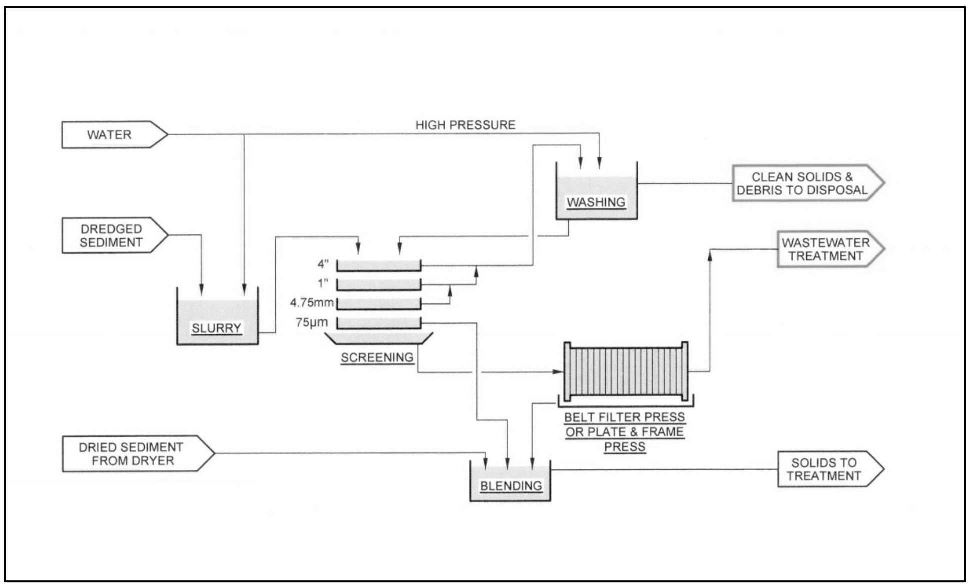
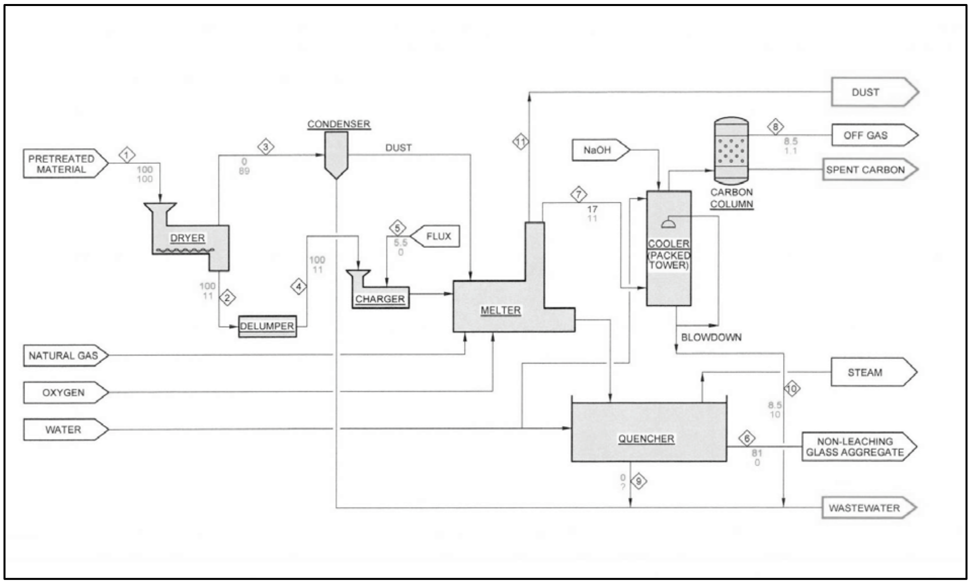


Figure D-12. Process flow diagram for Minergy glass furnace technology (taken from Estes et al. 2011).



Cement Lock® — Cement Lock is a rotary kiln based thermal technology, described as follows: “In the Cement-Lock® process, a mixture of sediment and modifiers is charged to a rotary kiln (the “Ecomelt® Generator”), which is the core of the treatment process. The integrated treatment process consists of two distinct phases: (i) solid separation (size reduction), dewatering and drying of the raw feed, and (ii) the rotary kiln (treatment) phase” (Estes et al. 2011). The Cement Lock process produces two distinctly different products, depending upon the mode of operation. In slagging mode, the product is call Ecomelt, which is pulverized and blended with lime or Portland cement creating a blended cement product. In non-slagging mode, the process yields a product called EcoAggMat, a low specific gravity material that can be used as fill or as a partial replacement for sand in concrete.

The plant used in the WRDA demonstration is pictured in Figure D-13; a process flow chart follows in Figure D-14.

Figure D-13. Cement-Lock® demo plant (Mensinger 2008).

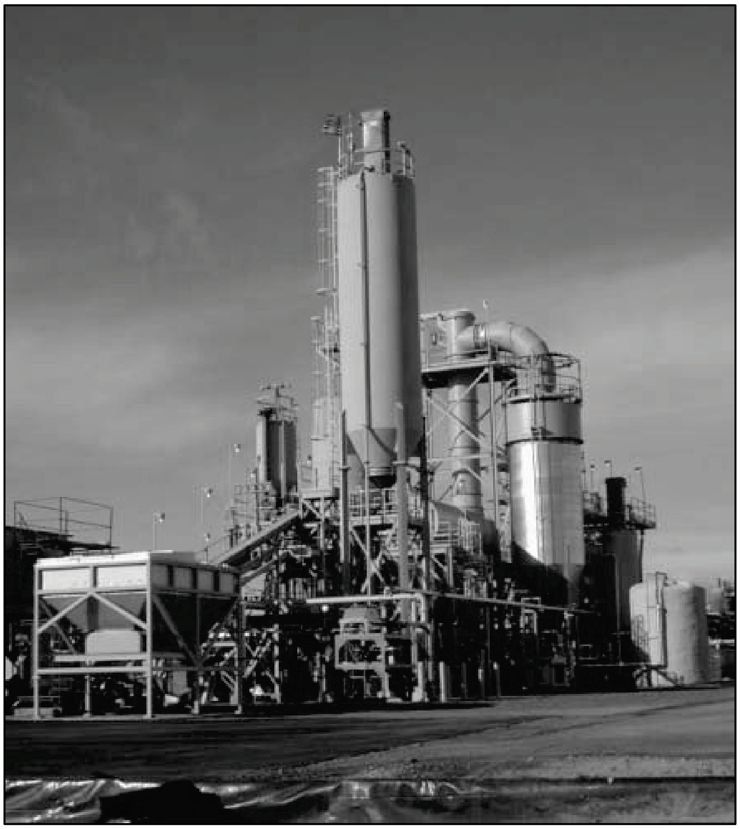
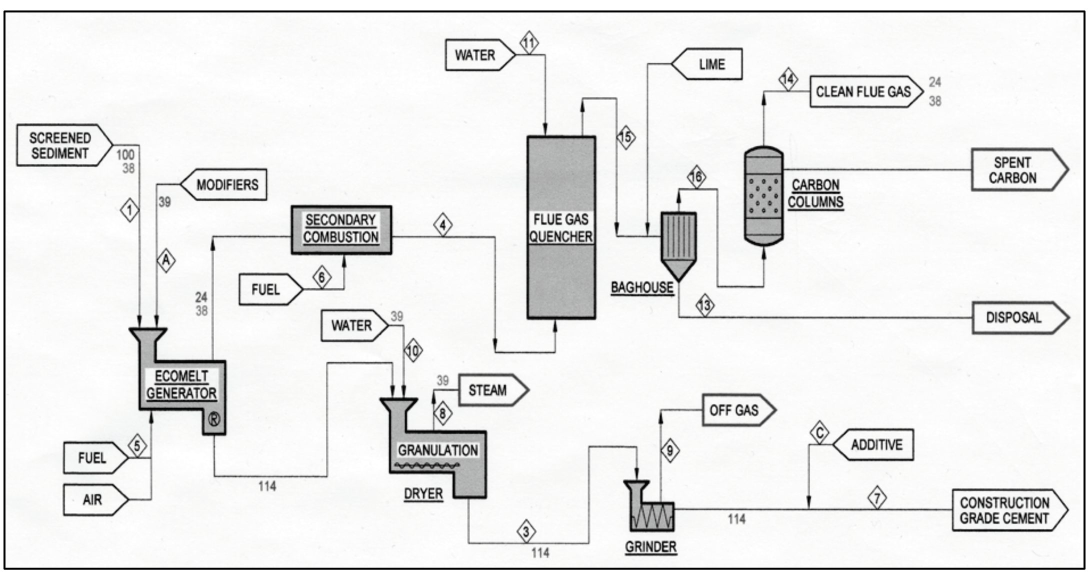


Figure D-14. Process flow diagram for Cement-Lock treatment (taken from Estes et al. 2011).



For both WRDA demonstrations, processing volumes were small, and sustained operation in treating sediment has not been demonstrated as of the time of this publication. Cement Lock has been demonstrated on sediments several times. Approximately 100 tons of Stratus Petroleum Refinery Site sediments were treated during a 15-month pilot ending in March 2005, but equipment problems prevented continuous operation. During a second phase of testing, a feed rate of 1,900 lb/hr was established, treating a total of 5.1 tons of Stratus Petroleum sediments. An extended duration test followed in December 2006 on Passaic River sediments, during which 16.5 tons of sediment were processed at 1,500 lb/hr. A second extended duration test processed 18 yd^3 of sediment over a 16 hr period, but equipment problems ultimately resulted in termination of the test. The controlling interest in Cement-Lock was purchased in 2007 from the Gas Technology Institute by Volcano Partners, a private concern that planned to redesign and re-engineer the concept to bring it up to commercial readiness*.

The capacity of the Minergy plant used in the 2006 sediment demonstration was approximately 200 lb/hr of dewatered dredge material. Processing capacity of a full-scale plant was projected to range from 200 to 1,600 metric tons dewatered sediment/day. A full-scale plant constructed for treatment of wastewater solids generated by paper mills became operational in January 1998 (Estes et al. 2011), but the process has not been demonstrated at full scale on sediment, to date.

Key logistical and cost issues for the thermal technologies are the disparate production rates between the dredge, pretreatment plant, and kiln, which will necessitate staging and storage areas sufficient to minimize dredging interruptions, provide for continuous, long-term operation of the process, and prevent product variations that may result from processing interruptions.

An economic analysis constructed from available data and uniform operational assumptions (Estes et al. 2011) estimated unit cost for Minergy and Cement Lock at between \$72/yd^3 and 101/yd^3 (December 2009 \$US), with no allowance made for potential revenues from product. At the time of publication of Estes et al. (2011), Minergy was selling the glass

* Al Hendricks. Personal communication. January 13, 2010. Managing Member, Volcano Partners, LLC, Maiyland, FL.

aggregate for a nominal fee of approximately \$1/yd^3; Cement Lock projected a value of approximately \$60/yd^3 (December 2009 \$US), but the actual market value had not been established.

Biological degradation

Numerous studies are available in the literature pertaining to the effectiveness of various organisms in the degradation of organic contaminants, nutrient delivery systems, and process management (Estes and McGrath 2014). Full scale bioremediation of sediments as not been achieved to date, for a variety of reasons. Limitations can arise from a number of sources, including the following:

- Toxicity of the target contaminant to the microorganisms
- Toxicity of non-target contaminants to the microorganisms
- Inability of the microorganisms to utilize the target compound, or preferential feeding on other available substrates
- Unfavorable conditions for the target consortia
- Limited contaminant bioavailability.

As with most treatment technologies, effective process control is more difficult at full scale than at bench scale, and many successful bench tests have been followed by unsuccessful pilot projects for this reason. However, bioremediation remains an appealing alternative because of the potential for essentially *passive* treatment that does not require significant rehandling of the material and that under ideal conditions can be implemented in situ. Despite continuing interest, in situ sediment bioremediation has progressed little due to the inherent operational difficulties in implementing, monitoring, and controlling the process in a subaqueous environment, in addition to the difficulty in demonstrating a necessary level of effectiveness in the treatment of truly recalcitrant contaminants.

Case studies

<u>Sheboygan River</u> — The Sheboygan River Pilot was conducted in the mid-1990s on 2500 yd^3 of PCB-contaminated sediments (Estes and McGrath 2011; Bishop 1996). The pilot was conducted ex situ, in a 14,000 ft^2 facility divided into two control cells and two treatment cells. Nutrients, oxygen, and other amendments were added through an underdrain system; oxygen

was added as oxygenated water and as dilute hydrogen peroxide solution. Initial concentrations of highly chlorinated PCBs in the sediment were low. High sediment oxygen demand prevented establishment of aerobic conditions for degradation of less chlorinated PCBs, even with the oxygen delivery system, and anaerobic dechlorination was not observed in the other cells. Limited bioavailability of PCBs in the sediments may have reduced effectiveness of treatment, and sediment variability likely complicated interpretation of the results. Ultimately, complete degradation was not achieved in the demonstration for any of the conditions tested — aerobic, anaerobic, aerobic/anaerobic cycling.

<u>Hudson River</u> — An in situ biodegradation demonstration was conducted in the Hudson River (Estes and McGrath 2011; Bishop 1996), by driving low-mix (3 rpm rakes) and high-mix (40 rpm turbine) caissons into the sediment, adding amendments (ammonium and phosphate nutrients, biphenyl, and hydrogen peroxide) to supplement indigenous organisms (two low-mix caissons) and a culture of PCB degraders (one low-mix and one high mix caisson). At 73 days, PCB losses of 38% to 55% were observed in all amended caissons; however, the PCB degraders did not enhance the degradation and were not competitive with indigenous organisms. Significant degradation of mono- and di-congeners was reported.

Among the conclusions drawn from this study (Bishop 1996) were the need to accomplish the following:

- Combine appropriate microbial pathways, biochemistry, and function of natural microbial communities.
- Overcome contaminant recalcitrance to biodegradation.

<u>Jones Island and Bayport CDF</u> — Both of these demonstrations were composting projects on sediments contaminated with PAHs and PCBs, one taking place at the Bayport CDF in Green Bay, WI, and the other at the Jones Island CDF in Milwaukee, WI. These demonstrations were conducted specifically with the objective of enabling safe beneficial use of these dredged materials. Figure D-15 is a picture of the scat turner, turning over the windrows of compost at the Bayport facility.

Figure D-15. Scat turner maintaining compost piles.

No decrease in PAH concentrations were observed in the Jones Island sediments (Estes and McGrath 2011); PAH concentrations decreased between 30% and 35% in Bayport sediments, but differences were not statistically significant. The opposite was observed for PCBs, with decreasing concentrations in Jones Island sediments but no apparent trend in the Bayport sediments. However, the data were highly variable, even showing increases over time in some samples, complicating interpretation of the data.

Processing challenges included the following:

- Moisture management was critical but difficult to maintain at field conditions; degradation is limited at moisture contents below 40%, while handling properties deteriorate at moisture contents above 50%.
- Heap size and biosolids addition (frequency and amount) needed to be balanced to maintain optimum temperatures in the composting heaps.
- Amendments contributed contaminants and also acted as a preferred food source for the microbes.
- Limited contaminant bioavailability may have contributed to the low effectiveness of the processes.

These findings suggest that contaminant availability should be one of the factors considered in feasibility testing and toxicity reduction should be the basis for determination of treatment success, as opposed to, or in addition to, target concentration reductions.

Autoheating Thermophilic Aerobic Reactor (A-TAR) — The A-TAR technology was evaluated in 1993 under the Great Lakes 2000 Cleanup Fund (Environment Canada 1997; Estes and McGrath 2011). The process was demonstrated at pilot scale on Hamilton Harbor sediments. The process takes place in a reactor and maintains temperatures at 45°C to 65°C without a heat source as a result of the ongoing microbial processes. The process reportedly achieved 95% removal of PAHs, 80% of total petroleum hydrocarbons, and 70% of oil and grease over an 8-day retention time. The technology is particularly interesting for the following reasons (Estes and McGrath 2011):

- Maintaining adequate temperatures to achieve biodegradation identified as a critical control variable in the Jones Island and Bayport CDF composting demonstrations (Myers and Bowman 1999; Myers et al. 2003).
  - The technology has been applied to digestion of wastewater sludges, which have also been used to amend sediments for composting and soil manufacture. This suggests a possible treatment synergy that could be exploited to achieve higher sediment treatment levels.
  - During wastewater sludge testing, the technology was observed to destroy pathogens, reduce oxygen requirements for aerobic digestion (due to elimination of nitrification), and to destroy weed seeds (Jewell and Kabrick 1980), a significant impediment to beneficial use in some areas of the country. Pathogens, while not regulated in sediments at this time, could potentially still be of concern in some beneficial use applications.

Feasibility/pilot testing

Basic characterization

The success of any treatment process is heavily dependent upon careful and adequate characterization of the sediment to be treated. Over and above the customary chemical characterization that is done to determine what contaminants are present and at what concentrations, the physical and mineralogical characteristics of the sediment can be equally important in selecting and designing a treatment process. All aspects of the proposed treatment should be considered in determining what sediment properties are of particular relevance. Feed-size restrictions suggest a need to understand the grain-size distribution of the sediment, for example, in addition to the relative proportions of the different size fractions so that equipment can be appropriately sized for the anticipated process streams. Relative capacity and processing rates of different pieces of equipment will inform the need for surge capacity, holding tanks, or stockpiling areas, for example. The characterization requirements, and the respective analysis conducted with the data, will differ somewhat depending upon the treatment technology being considered. The following is not a comprehensive listing but includes parameters that are generally useful to treatment feasibility evaluations.

Basic characterization should include a thorough evaluation of the geotechnical properties of the material, including, but not limited to the following:

- Grain-size distribution
- Percent organic, TOC, percent volatiles
- Solids specific gravity
- Bulk density
- Water content
- Clay content
- Mineralogical composition.

For separation processes in particular, the greater the physical differences in the target fraction as compared to the remainder of the sediment, the easier it is to separate the fractions efficiently for separate treatment or management. A generalization for size and density dependent processes is that a difference in specific gravity of at least 1 is required to separate two fractions effectively. This is not a mandated requirement, but the more similar two fractions are, the more overlap of the different fractions in the process streams. Specific sediment characteristics and treatment target objectives will determine whether this is acceptable.

Mineralogical composition of a sediment is important in terms of how tightly contaminants are bound to the sediment matrix and may inform how the sediment will interact with any process reagents, as in the case of carbonate competition for chemical oxidants, for example.

Understanding the physical and geotechnical characteristics of the sediment will inform not only the overall process selection but also the pre-treatment that might be necessary to prepare the material to ensure successful treatment. Separating the organic fraction from the sediment, for example, may be advantageous in more economical and effective oxidation of residual organics and contaminants on the mineral fraction of the sediment.

Fractionation testing

Fractionation testing involves a sequence of size and density separations conducted to determine where in the sediment the contaminants reside. A preliminary assessment of whether a simple size separation will be adequate to produce a clean sand fraction, for example, can be accomplished with bench scale fractionation testing. The need for additional cleaning steps to remove coarse organic materials from sand may be inferred from the results of the fractionation testing. It may also be useful to perform physical and geotechnical characterization on relevant sediment size or density fractions to assess separability of the fractions.

Process simulation

Bench scale testing representative of the intended field processes can be very helpful in conducting a preliminary evaluation of the feasibility of treatment with a specific process, assessing reagent dosages required or sensitivity of the process to various operational parameters, and identifying potential problems at a small scale where they are more easily addressed. Bench or small scale testing equipment exists for a variety of processes, ranging from biological reactors to separators and dewatering equipment.

Pilot testing

Rarely is bench scale characterization sufficient to draw meaningful conclusions regarding the potential success of a treatment at full scale. Bench testing can provide relevant and important information from which to structure a proposed plant or process design, but this should almost always be followed by pilot scale testing to better assess the effects of sediment variability on the treatment process. Processing issues that were not evident at bench scale often quickly reveal themselves on scale up, providing a more reliable basis for transitioning to a full-scale operation. Common issues include greater mass transfer limitations, less efficient mixing and reagent contact, greater material heterogeneity, and differences between pilot *make-do* equipment and full-scale equipment.

Considerations and limitations

Sediment presents a particularly challenging matrix for contaminant treatment, in part due to the characteristics of the sediment itself, the challenges of the aqueous environment to accomplishing in situ treatment, and the presence of multiple contaminants in the matrix. Sediments are also typically relatively heterogeneous, which makes accurate and complete characterization and treatment verification difficult. Fine-grained material is especially difficult to treat effectively and to handle within the process.

Besides the characteristics of the sediment, and the contaminants present in the sediment, there are a number of other factors to take into consideration when considering whether or not to treat, and in selecting the most appropriate treatment.

- Site conditions can significantly affect access, implementation, and stability of in situ treatments. Most processes require debris to be removed prior to treatment; size restrictions on the feed may require separation of gravel and objects as small as 3/8 in. prior to treatment. Some, but not all, thermal processes require removal of metallic debris of any size.
- Dewatering is typically the rate limiting step in processes requiring dewatering of the feed, or of the residual process streams. Appropriate staging areas and mixing/surge tanks may be required to address disparities in equipment capacity and dredge production rates.
- All processes produce some residuals — solid, aqueous, or both; the cost to manage the residuals must be taken into consideration in the overall cost and efficiency analysis.
- Most reagents, such as chemical oxidants, are not selective. The sediment must be dosed with sufficient reagent to address the demand from all competing materials, in addition to the target contaminants.
- The efficiency of the process required to meet process objectives should be considered; the most efficient process is not necessarily needed or justifiable to achieve sufficient risk reduction. Further, a process may

not perform well with one sediment but be relatively effective with another, depending upon contaminant levels, sediment characteristics, and location of contaminants in the matrix.

- Unit cost of treatment is dependent on multiple factors. Uniform cost assumptions were developed for the three thermal and one soil-washing technology evaluated in Estes et al. (2011), and sensitivity of the unit cost to assumptions regarding the cost of specific operations was also evaluated. Factors considered included the following:
  - Plant capacity
  - Energy requirements
  - Magnitude of residuals process streams and treatment or disposal requirements
  - Number of operators and skill level required
  - Infrastructure and equipment costs
  - Plant operating costs
  - Assumed value of product

The qualitative results of the sensitivity analysis are summarized in Table D-1.

Table D-1. Cost sensitivity of treatment processes to operational parameters.

Technology	Plant Capacity	Energy Costs	Residuals Treatment Cost	Labor Costs	Capital Costs	Operating Costs	Beneficial Use Product
Rotary Kiln	Moderate	Moderate	Low	Low	Moderate	Moderate	Moderate
Cement-Lock®	Moderate	High	Low	Low	High	Moderate	Moderate
Minergy	Moderate	Moderate	Low	Low	Moderate	Moderate	High
Biogenesis℠	High	Low	High	High	Moderate	High	High

- Several of the available treatment technologies that are near commercialization produce products suitable for various beneficial use applications. In most cases, the real market value of those products has not been demonstrated due to the limited scale of the treatment technologies to date. Potential issues with achieving value comparable to competing projects in the open marketplace include the potential concerns regarding a product generated from highly contaminated materials and the ready local availability of competing materials that do not suffer from this perception and that can be produced at lower cost. For example, sandy sediments are found in areas where sand is generally abundant; some type of cost motivation may be needed to motivate buyers to utilize the sand produced from a contaminated sediment, as compared to sand available from more pristine local sources.

- In evaluating treatment feasibility, the complete, integrated treatment train must be considered, from dredging to disposal of residuals. The relative capacity and feed requirements of the different unit operations must be considered within the context of the connecting processes and provision made to address discrepancies to ensure smooth and continuous operations.
- Presently, available cost estimates are based on limited scale operations, resulting in a relatively high level of uncertainty. Less mature and more *sensitive* processes are likely to have higher cost uncertainty than well-demonstrated and robust technologies that can be easily adapted to unexpected or changing conditions.
- Small sediment volumes typically cost substantially more to treat on a unit cost basis due to the fixed cost of infrastructure, staging area, etc.
- A comprehensive pilot scale evaluation requires sufficient sampling to enable determination of where in the process treatment is occurring, whether a specific unit operation is of benefit, or needed, and to discriminate between treatment and uncontrolled material or contaminant losses, or simple phase changes. Obtaining enough data to construct a mass balance for all materials and contaminants at each unit operation is critical. In addition, both total and leachable concentrations should be measured in treated product; the absence of a leachable fraction does not necessarily equate to destruction, and this may be important in assessing future risk associated with the material in a beneficial use placement.
- Highest decontamination efficiency is not necessarily the highest overall process efficiency — residuals must be considered as part of the overall process efficiency.

Appendix E: Practical Considerations for Dredged Material Management: Water Management for Upland Placement of Dredged Material

Water management for upland placement of dredged material

For upland unconfined sediment placement, including temporary dewatering prior to use at another site or placement directly on the use site, a number of fundamental issues may strongly influence the operation. These issues include water handling, water treatment needs, land area, enclosure requirements, time, and erosion control. The influence of these issues on the final outcome of the project cannot be emphasized enough. The practical considerations are what control upland placement and cost and are the *make or break* factors for the project.

Water management

A key issue for upland placement of dredged material is water management. There are essentially three options for managing water. These are summarized below and in Table E-1.

1. Allow direct return to waters of the United States. This would require a 401 water quality certification and is a suitable approach when placing coarse-grained, low-pollutant (low nutrient) sediment upland near the original material source. The 401 requirements are similar to what would be expected for in-water placement of the sediment. Turbidity and nutrients are likely to be key determinants.
2. Pond or collect the water for *treatment* and have a point discharge. This would require a 402 or National Pollutant Discharge Elimination System (NPDES) point discharge permit. The water would be held on site, likely in a diked impoundment or collection basin. Some water would evaporate or infiltrate. The remaining water could be treated using typical water/wastewater unit processes or could be held for a longer time in the pond to allow natural processes to improve water quality (through volatilization, settlement, biological transformation). If sufficient land and time are available, this later approach can be very effective. Six months or more of settling and quiescent ponding will generally improve water quality to the point that little or no treatment

is needed prior to discharge. Effectively, the impoundment is a temporary treatment lagoon.

3. Have all the water infiltrate, no water treatment or discharges. This approach works well if sufficient time and land are available to allow infiltration and also if the amount of water placed upland is limited from the start (i.e., this approach is more successful for mechanically placed materials.) Considerations include the water content of the original materials, the local precipitation amount verses evaporation, the soil types into which infiltration would occur (porous soils such as sands are best while clay soils are likely to have very low infiltration rates), the depth of the local groundwater table, and future land use and the timeframe for such uses.

Estimating the amount of water needed to discharge or treat is not straightforward. Generally, mechanically dredged sediment can be estimated to have a water content similar to the in situ condition Palermo et al. 2008). Depending on the grain size and sediment compaction, this may be less than 50% (although this will vary). Hydraulically dredged sediment will have much more water; a hydraulically dredged stream may average 5%–10% solids (although this also varies widely) during the dredging operation (which includes water discharged during line *blow down* and operational start up and shut down) so that the volume of water needing handling will be much greater. One way to get the benefits of hydraulic placement (less labor intensive, easier to distribute material) without the very large volume of water is to mechanically dredge the sediment then place the material hydraulically but recycle the off-loading water. This is a more sophisticated approach and would require a diked structure and controlled water management.

Table E-1. Comparison of water management approaches for upland sediment placement.

	Direct Return Water Stream	Indirect or Point Return Water Stream	No Discharge
CWA regulatory program	401 (non-point or wetland permit)	402 (NPDES or point discharge permit, for either temporary or permanent discharge, depending on the length of time for the operation)	none
Timeframe for discharges	Immediate dewatering, short timeframe to dewater freely released water	Can be short or long, depending on type of treatment. Plant treatment can be sized to match sediment discharge rate. Ponded or passive treatment can take months.	Generally longer, taking months for dewatering. Working the material can help speed drying, but infiltration will depend on the soil properties.
Water quality considerations	Best for high quality sediment with coarse grain size, low nutrients, very low anthropogenic compounds.	Can be adapted to water quality so is suitable for waters with high nutrients, anthropogenic compounds.	Not suitable for high levels of anthropogenic compounds but acceptable for higher nutrients.
Dredging type	Can be matched with either mechanical or hydraulic dredging, however high discharge rates may have turbidity issues due to sediment erosion with the discharge.	Can be matched with either mechanical or hydraulic dredging.	Best for mechanical dredging and relatively smaller amounts of water.
Land requirements	Moderate; sediment can be piled and managed concurrently with dewatering so that a large space may not be needed.	Small to moderate: treatment options can be matched to dewater sediment rapidly (filter presses) and treat discharge, so that material can be used dry in a short time and with a smaller footprint for stockpiling. Larger if a treatment pond approach is to be used and the water held for a long time prior to discharge.	Large. Generally, this approach works best when the sediment can be placed in a relatively thin lift and left to passively dewater in a large area to maximize infiltration and evaporation.

Water treatment

Water treatment may be needed for any concentrated discharge stream. As with all dredging, turbidity is often a concern due to the visual impact of a *muddy plume*, but the real water quality issues are typically the dissolved constituents. Sediment with high nutrients will readily release those nutrients into the entrained water. If this water is discharged in a concentrated stream from upland, it will be subject to Section 402 of the CWA or point discharge (NPDES) requirements. One straightforward method of handling this is to pond the water over the course of several months, which will allow natural biological transformation of the nutrients and/or uptake by bacteria and algae. This is cost effective from a treatment standpoint but requires time and space.

Nutrients, solids, and anthropogenic compounds (metals, large organic compounds) can be removed from the water stream using traditional water/wastewater treatment processes, including coagulation/flocculation, filtration, carbon filtration, biological treatment, oxidation processes (chlorination). These unit processes can be rented as a *package plant* for a short duration treatment and can be sized to meet the water generation rate, especially for hydraulic dredging when a large volume of water may be generated. Some companies design treatment trains specifically to match hydraulic dredging, starting with filter presses to dewater the sediment and following a typical wastewater treatment scheme, to provide a timely and small space alternative to produce dry sediment for disposal or use on another site, and treated water. This approach could be applied to a wide variety of sediment and water qualities but has the limitation that it is generally a more costly approach than passive dewatering.

Land Requirements

The land (space) requirements for direct upland placement come directly out of the water handling requirements and also the future land or sediment use. In general, a larger space and longer timeframe is needed for passive dewatering and water treatment approaches. For areas that are space constrained, active dewatering will be needed. If the sediment is to be left on the dewatering site, for example as clean fill, then some consideration needs to be given at the outset to the grade impacts and what a reasonable final contour will look like, depending on the future land use.

In general, sediment being dewatered passively should be placed in a fairly thin lift, such as a 3 ft layer so that the water held in the sediment can move through the layer. Depending on the sediment and site characteristics, the sediment may be able to be actively worked within a week or two by starting at the thin outer edge (but not by driving on it). Working the sediment (windrowing it, spreading it, compacting it) will help dewater it and also will minimize the space ultimately needed, however some space is needed for actively working the sediment. In general, it is best to plan that the sediment off-loading operation will need a large area to distribute the sediment initially, and then a second operation will take place using earthmoving equipment to help dewater, and distribute the sediment for further dewatering, to grade the material, or to move the material off-site.

Enclosure requirements

The wet sediment will generally need to be placed in some type of enclosure to contain the material, at least temporarily. The type of enclosure depends on the dewatering method. If water will be ponded, then a diked structure will be needed, with dikes constructed of clay or other stable material. It may be possible to use on-site soils for dikes; however, natural soils are unlikely to be suitable for ponding water for any length of time from a structural perspective.

If the sediment will be placed and allowed to passively dewater, such as by allowing infiltration, then site soils could be used for creating dikes. In this case, the dikes would be for temporary water containment or control but would not hold a pond of standing water for any length of time. Another alternative is to use jersey barriers lined with silt fence. In this case, the barriers are not impervious, and some small amount of water may be released through the joints between barriers, but the main role of the barriers is to delineate the placement area, control it, and prevent large amounts of erosion or material washing off site.

At the very least, an upland sediment placement site will need active erosion control at the boundaries. It is recommended that silt fence be properly installed around the entire area if dikes or some other structural enclosure is not used or needed.

Time

The choices made for land requirements, water management and treatment, enclosure requirements depend on and influence the time needed to complete the operation. In general, if the land is available for at least a full year (if not longer), it will allow sufficient time to prepare the site, dredge, dewater, final grade, and stabilize the site for erosion control. After those actions are completed, the site and material can be turned over to the property owner for the desired future use. If the land is not available for a sufficient timeframe, then active dewatering methods will need to be used to speed up that portion of the project; sediment dewatering is typically the rate-controlling step in the process.

Other considerations

Some other considerations include geotechnical properties of the material, final stabilization, dust control, grading, the edge of the pile or site and transition to surrounding land, future land uses, liability associated with the land, and defining when the work is done. Geotechnical properties of the sediment may be needed if the material is to be used upland, beyond simply the grain size. Information on compaction, strength, density is needed if the sediment will be used for structural fill, such as beneath roads.

Final stabilization typically requires planting (seeding) for erosion control, even if the site is to be developed in the future. Floral species native to the region should be used for this purpose. An upland placement project will require a stormwater (construction) NPDES permit, which is typically a general rule permit program. Under these programs, any site that will not be actively worked for more than 7 or 14 days will require at least temporary stabilization. Although a variety of stabilization methods are possible, typically grass seed is used for stabilization since the grass is an effective erosion control method on gentle slopes and can easily be graded off in the future. To close out the construction NPDES permit, final stabilization of the site is required, regardless of whether the entity doing the upland soil management is USACE, a contractor, or a local partner.

Dust control is part of site stabilization and becomes a concern as the dredged material dries. Placement of silt fence, sprinkling, and grass planting are all effective dust-control methods.

The final grade for the site and the transition of the *pile* at the edge of the site is also a concern. An ideal situation is that the dredged material will remain at the site where it is placed to dewater, minimizing transportation costs. Grading the material to facilitate dewater can help distribute the material into an even layer suitable for some future development or other use. It is recommended that the dredged material placement and dewatering plan include requirements for the final site grading that would be consistent with the future site use. In general, an upland sediment placement project is not *finished* until the sediment is dewatered sufficiently that the material can be graded or moved to the final location, all material is moved or graded, any supporting features (barriers, water treatment equipment) are removed, the site receives final stabilization, and all erosion control is complete.

It may be desirable to use dredged sediment as a clean cover material for a site that requires cover prior to use or development. In this case, the regulatory status as well as the potential presence of contaminants may dictate sediment placement requirements. For example, if a site has groundwater issues, the impact of infiltration of ground water on a contaminant plume must be considered. Liability associated with site issues also would require discussion.

Appendix F: Interpreting Laboratory Bioaccumulation Test Results on Dredged Sediment Proposed for Open-Water Placement

Purpose

This document explores how a number of sources of variability in benthic bioaccumulation data need to be considered toward interpreting laboratory bioaccumulation test results on dredged sediment in comparison to placement site or reference sediments. The overall purpose is to explain a technical weight-of-evidence approach for evaluating laboratory bioaccumulation test results from dredged sediment that indicate the possibility for increased bioaccumulation in the field upon placement in the open-water, including for aquatic beneficial use. A decision-making framework for evaluating differences beyond simple statistical significance is presented, emphasizing the magnitude of an observed difference in bioaccumulation and biologically meaningful benchmarks for interpretation. Three case studies using existing Great Lakes basin benthic bioaccumulation data are used to demonstrate the importance and utility of evaluating variance and MODs when interpreting benthic bioaccumulation results.

Background

To evaluate the bioaccumulation of an identified COPC from dredged sediment proposed for placement at an open-water placement site, formal federal guidance requires standardized laboratory benthic bioaccumulation testing in Tier 3 and at times Tier 4 to determine compliance with Clean Water Act (CWA) Section 404(b)(1) Guidelines (USEPA/USACE 1998a,b). Basically, the results of laboratory bioaccumulation tests for COPCs on dredged sediment and open-water placement site sediment are compared to decipher the resulting differences. The decisive intent of this evaluation is to predict direct bioaccumulation in benthos resulting from dredged sediment placement to evaluate sediment-associated COPCs that may accumulate sufficiently in the tissues of benthic predators (e.g., receptors, such as fish) with subsequent uptake at higher trophic levels (e.g., piscivorous fish) to elicit adverse effects (Suedel et al. 1994). Thus, while evaluating bioaccumulation in this context scrutinizes a single (benthic) pathway from sediment by quantifying a COPC in a benthic macroinvertebrate test species, the fundamental goal is to infer whether any associated increases in benthic bioaccumulation would potentially result in unacceptable adverse effects in predators in the field.

Bioaccumulation from benthos in this paradigm assumes dietary exposure through a food web link, whether it be direct or indirect, to a fish receptor. This assumption tends to be conservative because it cannot account for the complex and dynamic predator-prey interactions in the field. For example, in the Great Lakes, direct exposure of benthos to a fish receptor in relatively shallow-water areas (e.g., yellow perch [*Perca flavescens*] [Griswold and Tubb 1977; Knight et al. 1984; Smith 1985], brown bullhead [*Ameiurus nebulosus*] [Smith 1985]) can be fairly common. Great Lakes benthivorous fish show a strong preference for and higher site fidelity to the more complex habitats in the littoral zone, such as areas of rocky substrates, bottom slope, and submerged macrophytes, all of which provide a higher diversity and abundance of benthic prey compared to areas of less complex habitat, such as soft substrates (e.g., yellow perch [Janssen and Luebke 2004; Truemper et al. 2006; Duncan et al. 2011; Creque et al. 2010; Kovalenko et al. 2018]). Based on analysis of 13 different major fish taxa in Lake Superior, Sierszen et al. (2014) determined that the overall importance of benthic food web pathways to fish was highest in nearshore species, with offshore species depending more on planktonic food web pathways. Contributions of benthic pathways to fish declined with the depth of their habitat, corresponding with declining abundance of benthic invertebrates with increasing depth. With respect to deeper-water areas, while benthos-to-fish receptor connections can be partially direct (e.g., steelhead trout [*Oncorhynchus mykiss*] [Smith 1985; Rand et al. 1993]), they are usually indirect and oftentimes small (e.g., walleye [*Stizostedion vitreum*] [Griswold and Tubb 1977; Knight et al. 1984; Smith 1985; Hartman and Margraf 1992]). Most pre-existing dredged sediment open-water placement sites for fine-grain sediment in the Great Lakes are located in deeper zones with low habitat complexity (e.g., deep-water depositional areas with uniform, open mud-based bottoms), thus resulting in lower fish receptor exposure to sediment relative to shallow-water areas. In addition to these shallow- and deep-water food web complexities affecting fish exposure in the Great Lakes, the home range of fish receptors and/or their migrating prey foraging on benthos is not explicitly considered in this paradigm, despite being a basic and integral element of dietary exposure (e.g., von Stackelberg et al. 2005; Melwani et al. 2012; von Stackelberg et al. 2017).

While the approach described above is necessary and seemingly straightforward given the nature of a proposed discharge of dredged sediment within the framework of formal federal guidance and regulations (USEPA/USACE 1998a,b), it can also serve to diminish or overlook the various other pathways and factors affecting bioaccumulation in aquatic organisms. Bioaccumulation is a complex and dynamic process, being the net result of contaminant uptake, biotransformation, and elimination. The main pathways in fish include uptake through the water column via gill and skin absorption, and pelagic diet; uptake from sediment through benthic diet; and depuration via gills, fecal egestion, excretion, and metabolism (e.g., Gobas et al. 1999; Barron 2003). Although bioaccumulation is a dynamic process, it is typically measured and tends to be administratively regulated through analytically determined concentrations. These measurements simply provide estimates at a given point in time and are unable to reflect the numerous physicochemical, physiological, ecological, and environmental factors continuously influencing bioaccumulation in aquatic ecosystems (von Stackelberg et al. 2002; Selck et al. 2012; Kim et al. 2016). With respect to non-polar organic contaminants (NPOCs), abiotic and biotic factors, in addition to concentrations in prey, influence bioaccumulation in higher trophic levels organisms such as fish. For example, physiochemical properties of NPOCs such as hydrophobicity (e.g., Junqué et al. 2018), whole-body lipid content (e.g., Pastor et al. 1996), NPOC concentrations in the water column (e.g., Hebert and Haffner 1991; Qiao et al. 2000; Fadei et al. 2015), growth (e.g., Madenjian et al. 1994; Paterson et al. 2016), dietary shifts (e.g., Paterson et al. 2006), foraging ecology (e.g., Burtnyk et al. 2009; Paterson et al. 2016), reproductive offloading (e.g., Fisk and Johnston 1998) and seasonal weight loss (e.g., Daley et al. 2014) can all contribute to the variability observed in tissue residues of higher predators that consume infaunal invertebrates (McLeod et al. 2014). Recently, Hites and Holsen (2019) found close correspondence between PCB and DDT* concentration trends in air and fish in the Great Lakes, suggesting that fish tissue residues are closely linked to contaminant levels in the atmosphere, a large source of contaminant input into the Great Lakes. The overall point is that a number of these factors can ultimately drive overall bioaccumulation variability,

* dichlorodiphenyltrichloroethane

which does not manifest itself in the laboratory. As such, they can obfuscate predictions of bioaccumulation in the field inferred from standardized laboratory bioaccumulation test data generated for the evaluation of dredged sediment proposed for open-water placement.

Bioaccumulation evaluations of dredged sediment proposed for open-water placement initially emphasize bulk sediment concentrations of contaminants, with elevated concentrations in a dredged sediment in comparison to the placement site sediment potentially triggering further evaluation. While necessary in this context, this approach is conservative because contaminant bioaccumulation from sediment tends to be poorly predictable from bulk sediment concentrations of bioaccumulative contaminants, even when other sediment characteristics are considered. In fact, positive relationships between paired bulk sediment concentration and standard laboratory benthic bioaccumulation test results across narrow sediment concentration ranges are oftentimes weak or insignificant. The conservative effect of this approach is underscored when bulk sediment concentrations are more similar than different, such as situations commonly encountered when comparing relatively minor differences among contaminant concentrations in maintenance-dredged sediment and open-water placement site sediment. Therefore, inferences in bioaccumulation at higher trophic levels based on either bulk sediment concentration or tissue residues derived from laboratory benthic bioaccumulation tests need to consider that a number of variables may have a greater influence on the bioaccumulation process than contaminant concentrations.

The bioaccumulation of NPOCs from sediment, whether direct or indirect, is inherently variable, and this becomes of the essence when extrapolating laboratory-derived benthic bioaccumulation test data to the field scenario. Note that these laboratory data are generated under standardized and controlled conditions. Under such conditions, comparably fewer factors (e.g., sediment properties and heterogeneity, test organism biology and health, chemical analyses) contribute to the variability of the bioaccumulation endpoint. Laboratory-testing variability is expected to be considerably lower than the variability associated with the field environs. Consequently, laboratory tests yield relatively repeatable and precise results. Moreover, the variability yielded through single sets of laboratory test data can be so small compared to the actual variation (in replicated laboratory tests and in the field) that smaller observed differences that imply potential increases in bioaccumulation should not by themselves be assumed to indicate actual increases in bioaccumulation in the field, regardless of whether such small differences are statistically significant in the traditional sense (USEPA/USACE 1998a,b). The ultimate need is to extrapolate the laboratory results to predict the potential effects of the complex and dynamic process of bioaccumulation in receptors in the field where bioaccumulation is influenced by numerous other factors.

The objective of this report is to detail how bioaccumulation of a COPC from dredged sediment is evaluated according to formal federal guidance (USEPA/USACE 1998a,b) pursuant to CWA Section 404(b)(1) Guidelines. This evaluation applies to any open-water placement alternative, including those for aquatic beneficial use. In particular, this paper examines the evaluation of laboratory benthic bioaccumulation results that have been observed to be greater from dredged sediment than from sediment at the open-water placement site, with an emphasis on the MOD. The MOD is the mean tissue concentrations in benthic macroinvertebrates exposed to dredged sediment divided by mean tissue concentrations in benthic macroinvertebrates exposed to placement site or reference sediment. Data are presented for three case studies from the Great Lakes basin that demonstrate the utility of evaluating MODs for interpreting benthic bioaccumulation results.

Evaluating bioaccumulation from dredged sediment proposed for open-water placement

Approach

To evaluate the bioaccumulation of a COPC from dredged sediment proposed for placement at a specified open-water site, formal federal guidance requires standardized laboratory benthic bioaccumulation testing in Tiers 3 and/or 4 to determine compliance with CWA Section 404(b)(1) Guidelines (USEPA/USACE 1998a,b). To accomplish this, sediment samples of the dredged sediment and open-water placement site sediment are subjected to standard 28-day *Lumbriculus variegatus* bioaccumulation testing in the laboratory (USEPA/USACE 1998a,b). These tests generate tissue concentration data for the COPC on replicated masses of *L. variegatus* exposed to the sediment samples. These data are used to predict and evaluate any increases in bioaccumulation of the COPC from the dredged sediment in benthic macroinvertebrates after it is placed at the open-water site. The conceptual model is to assess whether there may be any ecologically meaningful increases in bioaccumulation of the COPC from benthic macroinvertebrates to receptors in the aquatic ecosystem after open-water placement of the dredged sediment. As discussed earlier in this report, examination of this single pathway does not address the other pathways and many other factors influencing the bioaccumulation process in organisms at the open-water placement site.

Data interpretation

Hypothesis testing and MODs

Interpreting bioaccumulation-related impacts begins with a simple comparison of the mean tissue concentration in *L. variegatus* exposed to the dredged sediment to the mean tissue concentration in *L. variegatus* exposed to the open-water placement site sediment. This yields one of two possible scenarios—either requiring or not requiring statistical comparison of the data (USEPA/USACE 1998a,b)—leading to three different evaluation outcomes as summarized in the following table:

<table>
<tr><th>Data Scenario</th><th>Statistical Comparison Required?</th><th>Based on Comparison, Is Further Evaluation Required and Why?</th></tr>
<tr><td>The mean tissue concentration in L. variegatus exposed to the dredged sediment is less than the mean tissue concentration in L. variegatus exposed to the open-water placement site sediment</td><td>No</td><td>No further evaluation is required because the data predict that no increase in bioaccumulation of the COPC from the dredged sediment would occur in the aquatic ecosystem as a result of open-water placement.</td></tr>
<tr><td rowspan="2">The mean tissue concentration in L. variegatus exposed to the dredged sediment is greater than the mean tissue concentration in L. variegatus exposed to the open-water placement site sediment</td><td>Yes</td><td>The difference is found to not be statistically significant. No further evaluation is required because the data predict that no statistically significant increase in bioaccumulation of the COPC from the dredged sediment would occur in the aquatic ecosystem as a result of open-water placement.</td></tr>
<tr><td>Yes</td><td>The difference is found to be statistically significant. This requires further evaluation because the data signify that it is possible that significant increases in bioaccumulation of the COPC from the dredged sediment may occur in the aquatic ecosystem as a result of open-water placement.</td></tr>
</table>

Only the last scenario in the preceding table requires further evaluation. Since standardized laboratory tests often generate low variability among replicates (McQueen et al. 2020b), it is not uncommon to observe statistically significant differences between dredged sediment and open-water placement site sediment datasets with relatively similar means (i.e., small MODs). As described earlier in this report, laboratory-based variability is expected to be considerably lower than the variability associated with the field environs. Consequently, statistically greater benthic bioaccumulation from the dredged sediment observed in the laboratory may not represent ecologically relevant tissue increases in receptors in the field.

Recognizing the limitations of relying on statistical significance alone (e.g., Johnson 1999; Hobbs and Hilborn 2006; Burnham and Anderson 2014; Amrhein et al. 2019), USEPA/USACE (1998a) prescribes assessing various factors, including MODs, when assessing benthic bioaccumulation beyond statistically significant differences. In fact, the concept of a MOD is inseparable from the concept of statistical significance testing among means, which sets out to identify a hypothesized difference by considering the probability of the observed difference (or a larger difference) occurring under the null hypothesis of no difference (i.e., a *P*-value) (Johnson 1999; Burnham and Anderson 2014). That probability is directly dependent on the MOD, the replicate data variance, and the sample size. Different laboratory benthic bioaccumulation tests can observe the same MOD but have considerably different *statistical significance* due to data variance, requiring an evaluation of the variance to interpret meaningful differences (McQueen et al. 2020).

Statistical significance alone is thus insufficient for understanding and evaluating the importance of the difference. Fundamentally, a *P*-value cannot inform on the evidence either against a null or for a hypothesized difference.* In addition, a *P*-value does not illuminate the size—and therefore importance—of such a difference. The essential point being that

* A *P*-value cannot be used as a quantitative measure of evidence for the central reason that it conditions on a model (the null hypothesis) instead of on the data (Berger and Sellke 1987; Royall 1997; Wagenmakers 2007). A *P*-value includes the probability of data hypothesized (under the null) but never observed (the more extreme differences), and therefore offers no formal evidence concerning even the null hypothesis itself (Goodman and Royall 1988). Moreover, because this conditionality is limited to the null, it does not permit a direct evaluation of the alternative hypothesis or the evidence associated with it (Burnham and Anderson 2014).

it is more informative to focus on the observed difference rather than to categorize it by *significance* (Amrhein et al. 2019). Avoiding conclusions deduced solely from statistically significant differences has the companion benefit of empowering the biologist and risk assessor to weigh lines of evidence in decision-making. An example of using relative weights of evidence to evaluate benthic bioaccumulation results from dredged sediment is presented for Cleveland Harbor data in the following section of this appendix: “Benthic bioaccumulation variability in the Great Lakes Basin,” item “b”.

Most researchers and practitioners now advocate that the interpretation of statistical results in ecology emphasize estimating the size of the difference and evaluating the risks and consequences associated with the expression of such a difference on the system of study. This is the basis behind the resource management framework adopted by the Food and Agriculture Organization for fisheries of the United Nations, which in part necessitates that the probability of competing hypotheses be evaluated and that the magnitude of difference is estimated (Hobbs and Hilborn 2006). Overall, a complete framework for statistical analysis would include evaluating the data within the context of prior results (e.g., multi-year datasets), the study design (e.g., the low variance among laboratory replicates from a single data source), and the underlying mechanisms of the biological process (Amrhein et al. 2019). The MOD serves as one key line of evidence for evaluating bioaccumulation risk within this context (McQueen et al. 2020b).

Identifying a benchmark

The requirement to evaluate the MOD in bioaccumulation begets the need for benchmarks that frame the ecological relevance of bioaccumulation in the field. To this end, Section 12.1.2.1 “Minimum Detectable Difference” of ASTM International (2019) states the following: “Although there is no consensus concerning what constitutes an acceptable minimum difference, it is suggested that the bioaccumulation experiment be designed to detect a two-fold difference between tissue residues in the test and control sediments or the test and reference sediments. A two-fold difference should provide a sufficient signal for ecological and human health concerns in most cases." This criterion was also proposed by USEPA (1993a), where it is written "...a 2-fold difference should provide a sufficiently precise result to address ecological and human health concerns."

Recently, McQueen et al. (2020b) highlighted the statistical limitations of bioaccumulation testing and the consideration and use of MODs and sample variability when discerning biologically meaningful differences of contaminant tissue concentrations. Based on their analysis of laboratory benthic bioaccumulation datasets, McQueen et al. (2020b) proposed the application of a threshold MOD of 2 as a line of evidence for interpreting bioaccumulation test results. These references indicate that a two-fold difference in mean bioaccumulation between a given test sediment and reference sediment should be sufficiently adequate to capture ecologically important differences in bioaccumulation in the field. Furthermore, these references also suggest that a less than two-fold difference in bioaccumulation that is statistically significant may result in false-positive inferences when extrapolating comparisons of laboratory bioaccumulation data to the field environs. With respect to statistically non-significant differences (the second scenario presented in the above table), applying a benchmark such as a two-fold difference is generally unnecessary because the very low variance associated with a single set of laboratory replicates creates a conservative evaluation. Statistically non-significant results with a MOD >2 are rarely observed, especially in the Great Lakes, and when they occur it is with unusually high variance (McQueen et al. 2020b). Such high variance would only be compounded when extrapolating to the field and to higher trophic levels. That is, these cases are generally a result of noise in the bioaccumulation process rather than a hidden signal of ecologically meaningful differences between sediments.

Ancillary evidence in the literature is consistent with the use of a factor of 2 to decipher potential differences in benthic bioaccumulation. In examining the predictive ability of bioaccumulation factors (BAFs) and biota-sediment accumulation factors (BSAFs), Hydroqual Inc. (1995) found that within a homogenous group of compounds (e.g., PCB congeners), BAFs and BSAFs can be predicted within a factor of 10. In an investigation comparing co-located laboratory and field *L. variegatus* PCB bioaccumulation experiment data, Beckingham and Ghosh (2010) found agreement to be within a factor of 2. In a bioaccumulation model field verification effort, Burkhard et al. (2003) found that greater than 90% of predicted BAFs were within a factor of 5 of their measured values, of which approximately 60% were within a factor of 2. Data from Burkhard et al. (2011) suggested correspondence of paired laboratory and field *L. variegatus* BSAFs to be within a factor of 2. Using passive samplers, Jonker et al. (2018) recently found 10-fold differences in samples of dissolved sediment porewater (bioavailable) PCB and polycyclic aromatic hydrocarbon (PAH) concentrations split among laboratories. Laboratory and field bioaccumulation data on Great Lakes basin sediments also illustrate similar levels of variability for benthic bioaccumulation and are presented in the following section.

Benthic bioaccumulation variability in the Great Lakes Basin

Three examples from the Great Lakes region will be used to demonstrate the overall variability in both laboratory and field-derived *L. variegatus* bioaccumulation data. These examples provide data that support the use of a factor of 2 to evaluate bioaccumulation test data generated for dredged sediment open-water placement evaluations in Tier 3 or 4.

a. <u>Benthic bioaccumulation of total PCBs from Lake Erie reference area sediments offshore of Ashtabula, Ohio</u>—This example is a compilation of all standard 28-day total PCB *L. variegatus* bioaccumulation test data generated by USACE on Lake Erie reference area sediments offshore of Ashtabula, OH, in 1993, 2010, and 2016. These data are illustrated as follows:

(1) Figure F-1* is a scatter plot of mean total PCB residues measured in *L. variegatus* tissue versus bulk sediment concentration. This graph illustrates two main points. First, it does not show any significant positive linear relationship between total PCB concentration in tissue and sediment (R^2=0.02). This indicates that bulk sediment PCB concentration alone played little role in PCB bioaccumulation variability, suggesting that other factors drove bioaccumulation. Second, it shows that PCB bioaccumulation from Lake Erie reference sediments within a relatively narrow total PCB concentration range (in this case, 43 to 160 µg/kg) varies between a factor of 1.1 (29.5/28.1) to 8.4 (168/19.1), with the average difference among means being a factor of 3.2. When this evaluation was repeated using for total PCB sediment concentrations normalized to TOC content, the same results were found (R^2=0.01). For the relatively low and narrow range of PCB concentrations being addressed in this paper, which are common in dredged sediment contaminant evaluations, normalizing to TOC content typically does not improve bulk sediment PCB concentration as an indicator of benthic bioaccumulation. These types of datasets accentuate the uncertainties of inferring benthic bioaccumulation directly

* Figures F-1 through F-4 are located at the end of this appendix.

from bulk sediment concentration values, independent of the cumulative uncertainties with assessing bioaccumulation within the aquatic food web.

(2) Figure F-2 is a bar graph of mean total PCB BSAFs (tissue PCB concentration normalized to lipid content divided by sediment PCB concentrations normalized to organic carbon content) derived from the bioaccumulation test data. This graph shows that the mean BSAFs in some cases showed little difference but varied by up to a factor of 11.6 (2.55/0.22), with the average difference among means being a factor of 4.1. Consistent with BSAF theory, the BSAF values were independent of bulk sediment total PCB concentration.

The salient point from this example is that laboratory bioaccumulation of total PCBs in *L. variegatus* from these sediment samples within a narrow bulk concentration range typically varied by more than a factor of 2. This indicates that use of the two-fold difference in ASTM International (2019) provides an equitable and protective criterion to evaluate MODs in benthic bioaccumulation from sediment.

b. Benthic bioaccumulation of total PCBs from Cleveland Harbor (Upper Cuyahoga River Channel) sediments, Cleveland, Ohio compared with offshore open-water placement area/reference sediments—This example is a compilation of standard 28-day total PCB *L. variegatus* bioaccumulation test data generated by USACE on Cleveland Harbor (Upper Cuyahoga River Channel) and offshore open-water placement area/reference sediments in 2012, 2014, 2015 and 2017. These bioaccumulation data were generated across four investigations on samples of the same dredged material management unit (DMMU) and proposed open-water placement site/reference sediments, all of which possessed similar bulk sediment concentrations of total PCBs (in this case, 68 to 157 μg/kg). The overall variability of these bioaccumulation test data was evaluated as follows:

(1) Bioaccumulation MODs across the investigations are illustrated in Figure F-3. When compared to the proposed placement site sediments, mean bioaccumulation of total PCBs from the DMMU sediments across the consecutive investigations was either (1) lower; (2) greater but not statistically significant; or (3) statistically greater but within a two-fold difference. The MODs ranged from 0.5 to 1.9, with the overall average MOD of 1.1. This MOD is below or comparable to a factor of 1.3 at which the general onset of statistically significant differences in standard 28-day *L. variegatus* laboratory bioaccumulation tests are typically observed (as supported through numerous USACE ERDC bioaccumulation datasets

using a replication of 5; McQueen et al. 2020). Therefore, these results illustrate that the use of a factor of 2 to evaluate potentially ecologically meaningful bioaccumulation across several years of samples of the same DMMU sediments, when grouped, ultimately approached no absolute or statistically significant difference based on laboratory data alone.

(2) Another approach to evaluating bioaccumulation variability in this case compares mean total PCB residues in *L. variegatus* exposed to each harbor or open-water placement site/reference sediment across the four investigations. Among all sites, year-to-year differences in laboratory bioaccumulation, which could not be readily explained through bulk sediment concentration, ranged from a factor of 1.0 (DMMU-1, 156 µg/kg [2014]/153 µg/kg [2015]) to 4.2 (DMMU-2b, 181 µg/kg [2015]/43.4 µg/kg [2017]). Within each site, maximum bioaccumulation differences among years were consistently greater than two-fold, ranging from 2.4 (DMMU-1, 156 µg/kg [2014]/64.6 µg/kg [2017]) to 4.2 (DMMU-2b, 181 µg/kg [2015]/43.4 µg/kg [2017]). These year-to-year differences could not be readily explained through bulk sediment concentrations, which remained similar between years. Another analysis of these same data using a mixed effects model (which includes parameters for both year and site) demonstrated that the variance attributable to year-to-year differences was very similar to the variance attributable to differences between sites (USACE 2018). This mixed effects model is a straightforward example of evaluating the relative evidence for competing hypotheses or models outside of a null hypothesis testing framework (see section "Hypothesis testing and MODs" in this appendix). Not only do these data demonstrate the inherent variation in laboratory bioaccumulation test data, but they also exemplify that testing and evaluation prescribed in USEPA/USACE (1998a,b) simply yield *point-in-time* estimates of bioaccumulation, making it imperative to assimilate expected temporal variation in the evaluation of dredged sediment for open-water placement. Similar to the bioaccumulation data presented in the example in Figure F-1 (benthic bioaccumulation of total PCBs from Lake Erie reference area sediments offshore of Ashtabula, OH), these data indicated no significant positive linear relationship between total PCB concentration in *L. variegatus* and bulk sediment (R^2=0.05).

The salient point from this example is that MODs derived from laboratory total PCB benthic bioaccumulation data on sediments collected from the same areas typically varied within a factor of 2 across sampling years. This example also lends a temporal dimension to assessing benthic bioaccumulation. This indicates that use of the two-fold difference in ASTM International (2019) provides an equitable and protective criterion to evaluate MODs in benthic bioaccumulation from sediment.

c. Benthic bioaccumulation of total PCBs from Grasse River sediments near Massena, New York, laboratory versus field test results—This example includes a compilation of 14-day total PCB *L. variegatus* bioaccumulation test data generated by Beckingham and Ghosh (2010). This example is different from the previous two examples as the study sampled a small, 0.5-acre area during a single event and compared the results of bioaccumulation experiments conducted in the laboratory (ex situ) and field (in situ). Sediments from a total of seven discrete sites, including M1 through M6 and BG, were subjected to bioaccumulation testing. Figure F-4 plots all of the bioaccumulation test generated data against bulk sediment concentration. The bioaccumulation data illustrate considerable spread relative to the single small area sampled. The following information can be gleaned from these results:

(1) Total PCB bioaccumulation from sediment samples in this area varied by up to a factor of 4 to 5. Across all samples, average total PCB bioaccumulation differed up to 2.4-fold.

(2) Average total PCB residues in *L. variegatus* yielded in the bioaccumulation tests ranged from 1.4 to 3.0 µg/g in the field and 2.1 to 3.3 µg/g in the laboratory, with correspondence among the two experimental results being within a factor of 2 (Beckingham and Ghosh 2010).

(3) A statistically significant difference was found between the field and laboratory bioaccumulation data. However, Beckingham and Ghosh (2010) noted that the M1 and BG results had the greatest influence for the statistical significance, concluding that the differences among laboratory and field bioaccumulation test results were not exceptional.

(4) Bulk sediment total PCB concentration, which ranged from 2.1 to 3.9 µg/g (mean 3.0 µg/g) appeared to have a weak influence in net bioaccumulation (simple linear regression, all data - R2=0.35; P<0.001). A careful examination of the data shows that while the bulk sediment total PCB concentrations in the M6 and BG sediment samples were essentially the same (~3.6 µg/g), average bioaccumulation results varied by up to a factor of 1.7. In addition, the laboratory bioaccumulation data show considerable overlap across the full range in bulk sediment concentration. For example, the very similar bioaccumulation test results for M3 field and

M4 field (respective means of 2.1 and 2.3 µg/g) were associated with lowest and highest bulk sediment concentrations (2.1 and 3.9 µg/g, respectively).

The salient point from this example is that general agreement among *L. variegatus* PCB bioaccumulation test data, whether they are generated in the laboratory or field, was within a factor of 2.

Summary

In the Great Lakes, COPC bioaccumulation evaluations utilize standard laboratory benthic bioaccumulation testing to ultimately decipher whether a proposed discharge of dredged sediment would result in an ecologically meaningful (or unacceptable adverse) effect in fish receptors at proposed open-water placement sites. It has been frequently assumed that body residues in fish will change in direct proportion to the increase in the body residue in benthic prey; however, this is not ultimately the case. This paradigm cannot represent the several other COPC pathways in fish, nor can it account for the various other biotic and abiotic factors influencing the complex bioaccumulation process and bioaccumulation variability in fish. Moreover, it does not consider fish receptor habitat preference or home range and the oftentimes indirect relationship between contaminant concentrations in sediment, benthos, and fish. Finally, bioaccumulation is inherently variable, and this variability tends to substantially increase in the field and with higher trophic levels in the food web. Employing a weight-of-evidence approach, standard laboratory benthic bioaccumulation test data generated to determine differences in COPC bioaccumulation from a dredged sediment versus placement site sediment need to be interpreted within the backdrop of such factors and variability.

Figure F-1. Laboratory total PCB *L. variegatus* bioaccumulation data on all Lake Erie background sediments offshore of Ashtabula, OH.

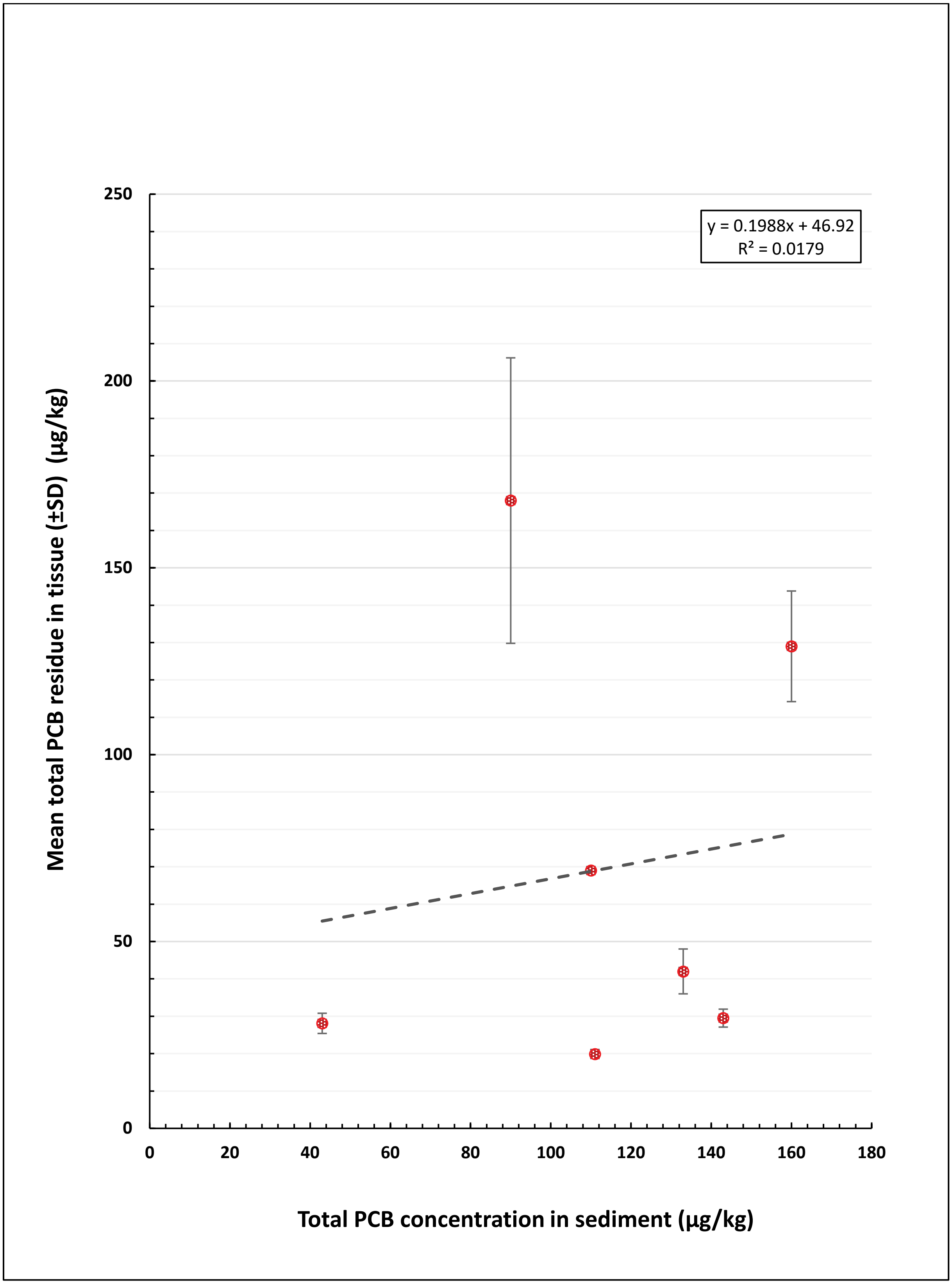
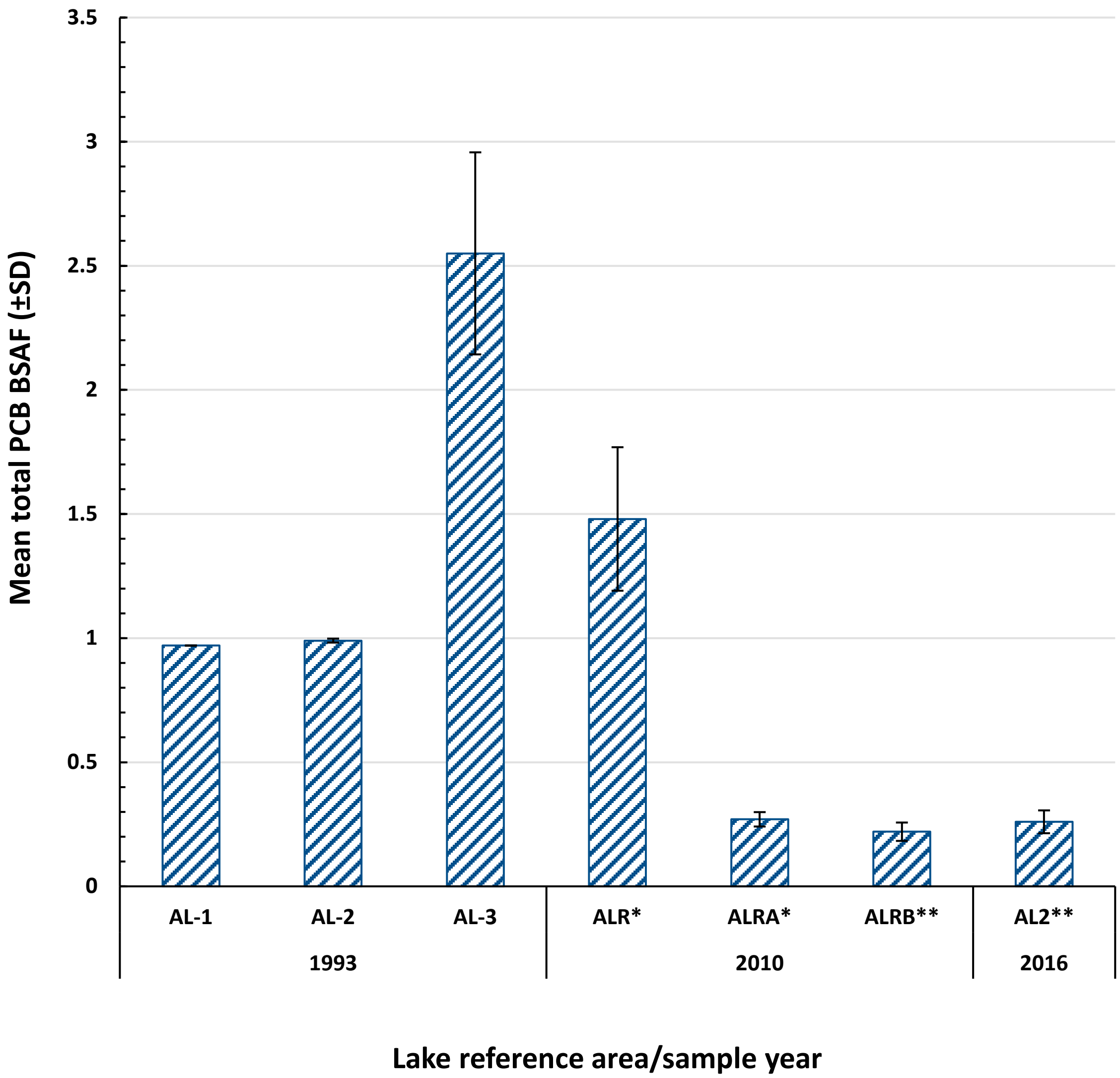
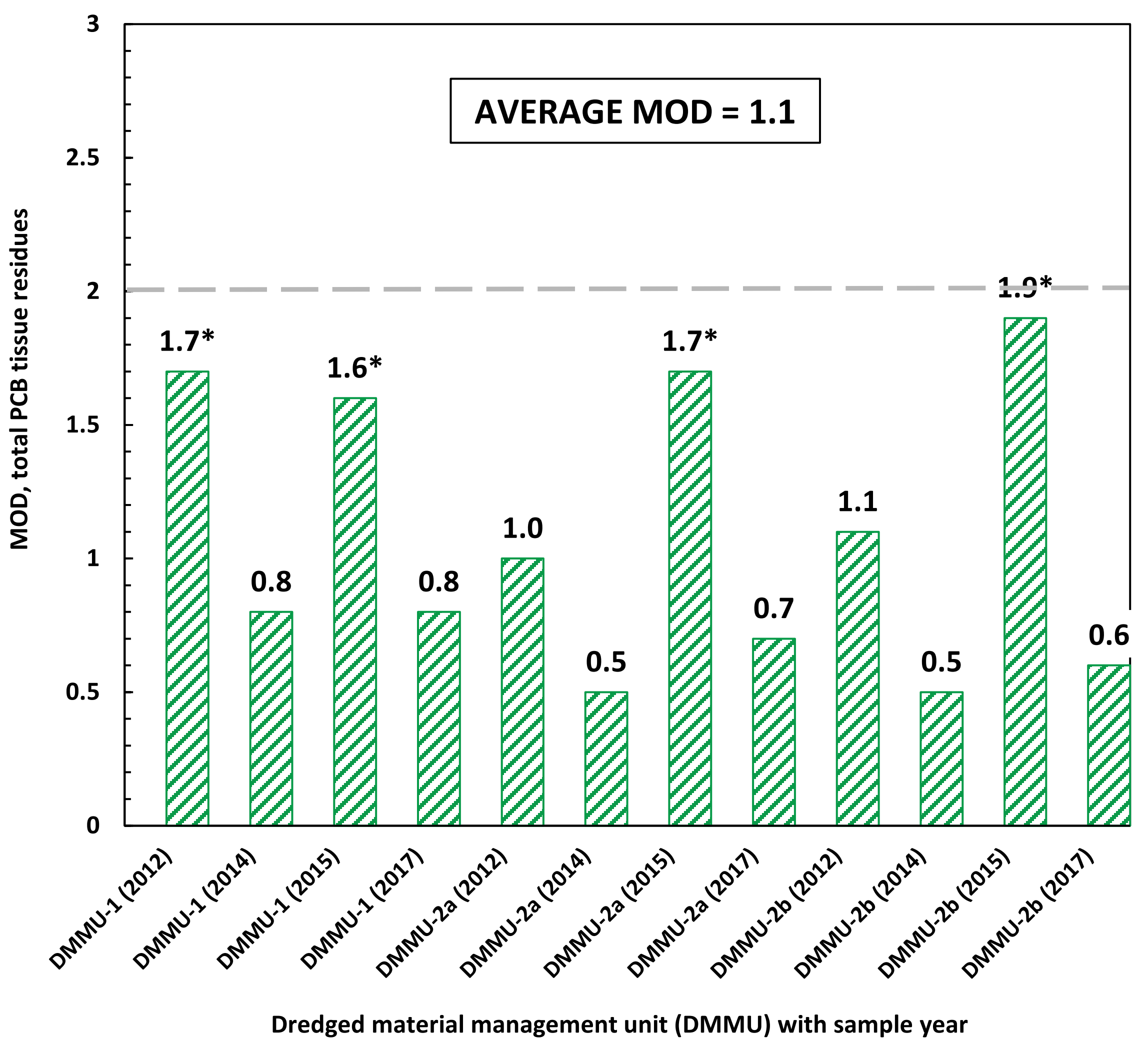


Figure F-2. Mean total PCB BSAFs on all Lake Erie reference sediments offshore of Ashtabula, OH.



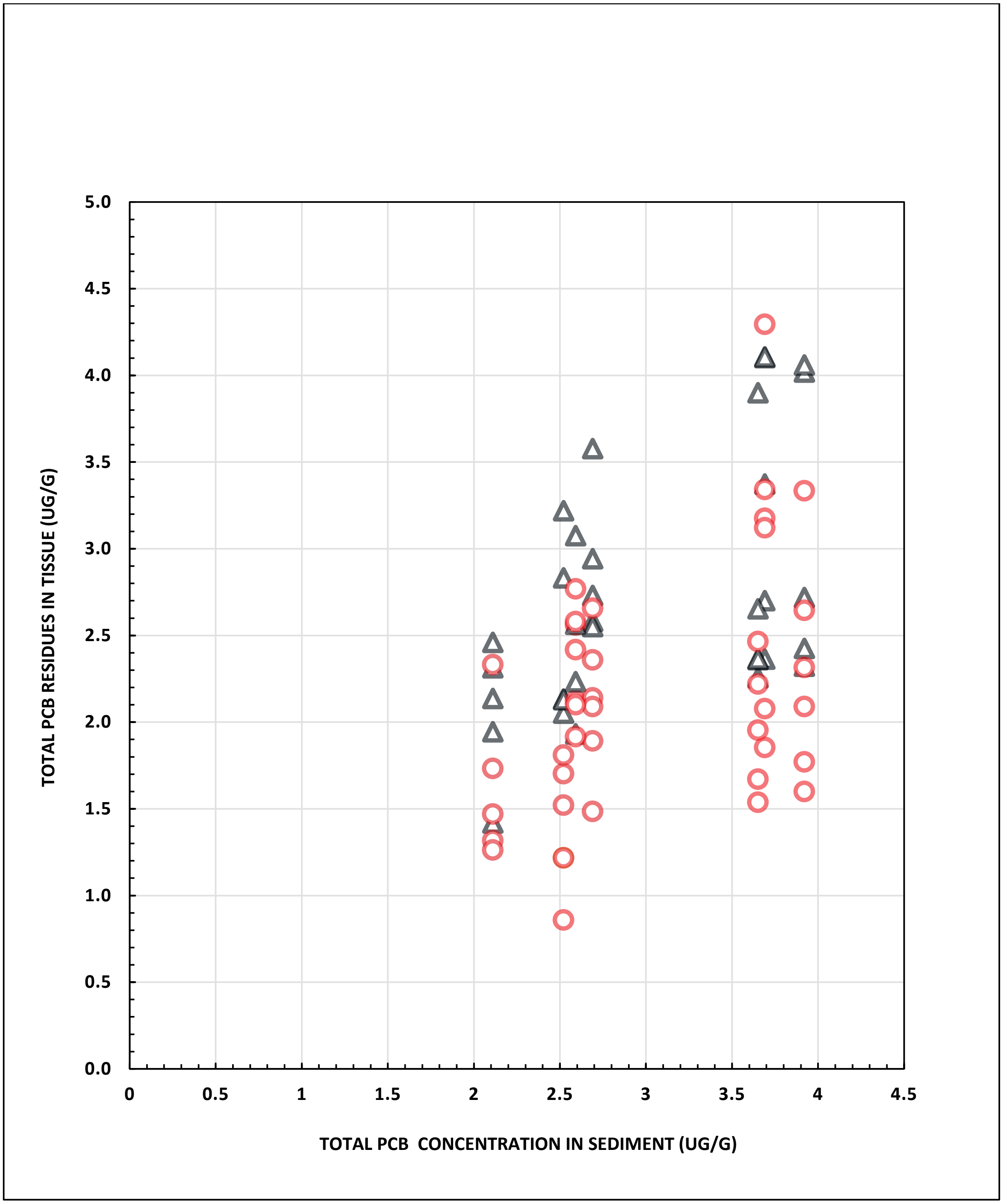
NOTE: Asterisks (* and **) designate identical reference areas.

Figure F-3. Magnitudes of difference (MODs) in laboratory *L. variegatus* total PCB bioaccumulation from channel sediments relative to proposed Lake Erie placement area sediments.



NOTE: An asterisk (*) designates that bioaccumulation from the dredged sediment was statistically higher than that from proposed placement area sediment.

Figure F-4. Plot of *L. variegatus* total PCB bioaccumulation versus sediment concentration using all data from Beckingham and Gosh (2010) (note: red circles – in situ; gray triangles – ex situ).



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14. ABSTRACT
The Environmental Evaluation and Management of Dredged Material for Beneficial Use: A Regional Beneficial Use Testing Manual for the Great Lakes (a.k.a. Great Lakes Beneficial Use Testing Manual) is a resource document providing technical guidance for evaluating the suitability of dredged sediment for beneficial use in aquatic and terrestrial environments in the Great Lakes region. The procedures in this manual are based on the Environmental Laboratory extensive research, working with US Army Corps of Engineers (USACE) Great Lakes districts, state resource agencies, and local stakeholders seeking to develop dredged material beneficial use alternatives consistent with regional needs and goals. This manual is the first guidance document developed by USACE for evaluating the environmental suitability of dredged material specifically for beneficial use placements. It provides a tiered framework for evaluating the environmental suitability of aquatic and upland beneficial uses consistent with the Inland Testing Manual and the Upland Testing Manual. This manual is intended to serve as a regional platform to increase collaborative problem-solving and endorse a common understanding of the scientific and institutional practices for evaluating dredged material for any beneficial use. Dredged sediment may be managed as a valuable resource, with great potential to create economic, environmental, and social benefits.

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