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A Review of Risk-Based Approaches to the Development of Screening Criteria for Soils and Application to Beneficial Use of Dredged Material

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PURPOSE: Beneficial use of dredged material is critical to sustainable dredged material management. Although dredged material is slowly becoming recognized as a resource, employment of beneficial use practices is limited by a number of factors, including the lack of consistent screening criteria specific to beneficial use of dredged material. Determining appropriate criteria (or the approach to criteria selection/development) is a challenge due to the wide range of beneficial use applications with varying potential for exposure, human health impacts, and environmental effects. Currently, criteria for soils vary among the states, and in many instances there is either a lack of criteria applicable to dredged material beneficial use or overly-restrictive criteria established based on the "precautionary principle." Overly-restrictive criteria constrain beneficial use of dredged material unnecessarily. Furthermore, the lack of consistency debilitates planning efforts, which leads to missed opportunities and increased cost. A scientifically-defensible method for determining the suitability of dredged material for beneficial use applications is needed. This technical note discusses considerations for applying criteria to beneficial use of dredged material and summarizes approaches that have been used within and outside the United States to develop risk-based screening criteria for soils that could potentially be used or adapted to develop screening criteria for beneficial use of dredged material.

BACKGROUND: The U.S. Army Corps of Engineers (USACE) and its contractors have dredged an average of approximately 250 million cubic yards of sediment annually over the last decade to fulfill USACE's mission to construct and maintain the Nation's navigable waterways (USACE Institute for Water Resources 2010). Beneficial use of dredged material is not a new concept; Engineer Manual 1110-2-5026 (USACE 1987) emphasizes the importance of dredged material as a manageable, valuable soil resource, and the need for project planning to take this into account at project inception. Options for placement of dredged material have become more limited as a result of increasingly stringent environmental restrictions on open water disposal and diminishing capacity in existing confined disposal facilities (CDFs). Due to costs to site and construct new disposal facilities and to more restrictive environmental regulations, sustainable dredging and disposal operations will depend on developing a capability to reclaim material already stored in CDFs for beneficial use (Bailey et al. 2010).

Beneficial Uses. Dredged material can be used for a wide range of beneficial uses in urban, agricultural, industrial and natural settings. The U.S. Environmental Protection Agency (USEPA) and USACE identify the following general categories for beneficial use (USEPA/USACE 2007):

- Habitat restoration and development
- Beach nourishment

- Parks and recreation
- Agriculture, forestry, horticulture, and aquaculture
- Strip-mine reclamation and solid waste management
- Construction/industrial development
- Multi-purpose activities—a series of applications, such as a park over a landfill with a final dredged material cover

These major categories can be subdivided into a number of more specific applications. Testing dredged material for the various beneficial uses generally includes assessment of geotechnical, engineering, chemical, and biological properties. Technical information needed to assess different beneficial uses may differ from information needs for disposal alternatives due to differences in structural requirements, exposure, potential contaminant pathways, and the receptors of concern. Winfield and Lee (1999) and Brandon and Price (2007) identify and describe characterization tests for determining the suitability of dredged material for beneficial uses. While the geotechnical and engineering properties for specific beneficial uses have been documented, the environmental suitability of material for specific uses is more complex and is more difficult to define.

Procedures for Evaluation and Testing of Dredged Materials. Evaluating environmental effects of dredged material management alternatives are laid out in the "Technical Framework" (USEPA/USACE 2004), the Upland Testing Manual (USACE 2003), the Inland Testing Manual (USEPA/USACE 1998) and the Ocean Disposal Manual (Green Book) (USEPA/USACE 1991). Originally published in 1992, and updated in 2004, the Technical Framework provides a consistent stepwise approach for evaluation of dredged material for open water disposal, confined disposal, and beneficial use alternatives. This approach was designed to meet the statutory requirements of various federal laws and regulations governing dredging and dredged material disposal. Complementing the framework are the ocean, inland, and upland testing manuals, which prescribe four tiers for the assessment and testing of environmental effects of dredged material (USACE 2003):

- Tier I – Review existing available information to determine the need for further testing to evaluate pathways or contaminants of concern.
- Tier II – Determine the need for management actions derived from very conservative techniques that use the chemical, physical, and biological characteristics of the dredged material and basic information about management and disposal options.
- Tier III – Obtain more detailed information through effects-based testing.
- Tier IV – Perform case-specific studies or formal quantitative risk assessment designed to answer specific, well-defined questions.

In all of these guidance documents, the evaluation begins with collecting and examining available information to establish if there is a "reason to believe" that the material is contaminated prior to conducting additional testing (Tier I). If there is "reasonable assurance that the proposed discharge material is not a carrier of contaminants" (33 U.S.C. 1344, Section 404(b)(1) guidelines) the assessment ends at Tier 1. The tiered approach uses time and financial resources efficiently by constraining testing to the level required to make a definitive determination regarding impacts of disposal, reserving the more costly testing and analysis required for higher tiers—or cases where the lower level evaluations are not sufficiently definitive. The Framework and Testing Manuals provide procedures for evaluating environmental impacts associated with disposal. A similar approach would be reasonable to streamline evaluation of the suitability of dredged material for beneficial use.

> The Great Lakes Beneficial Use Task Force has included in its report the following recommendation (Great Lakes Commission 2001):
> "Risk-based guidance that establishes contamination thresholds or parameters for different beneficial use applications, based on the physical and chemical properties of the dredged material and its end use, should be developed. This guidance should use a comparative, risk-based approach instead of strict numerical standards, yet could allow for case-specific determinations to consider the range of physical and chemical characteristics of dredged material and exposure pathways associated with its end use."

Definition of Contamination. To effectively apply the tiered evaluation approach, it is necessary to establish what level is contaminated. Most sediment dredged for navigation or cleanup purposes is inherently contaminated at some level. The "Technical Framework" (USEPA/USACE 2004) defines contaminated sediments or contaminated dredged materials as "those that have been demonstrated to cause an unacceptable adverse effect on human health or the environment." The context for evaluating this definition can include dredged material placed in an upland site, as well as contaminants in a waterway. Identifying contaminants of concern or potential concern is one of the first steps in evaluating disposal options and in evaluating beneficial use options. Sediment contaminants include polynuclear aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), pesticides, metals, and other contaminants that may have been introduced into the watershed by anthropogenic sources, both recent and historical. Considered from a risk perspective, many beneficial uses for dredged material do not require completely clean or uncontaminated material. Requiring that dredged material from a site be "uncontaminated" in order for the material to be used beneficially is overly restrictive, and would likely rule out use of navigation sediments where the need to use them is greatest. Most of the navigation waterways where beneficial use will relieve the limitations on dredged material disposal are in commercial or industrial areas where contaminants, at some level, are commonplace. Whether a particular contaminant is a potential concern depends on its amount, mobility, distribution, toxicological importance, bioavailability, receptors, geochemistry and other factors. While detailed risk assessment is one way to establish the viability of beneficial use as an alternative for a given dredged material — to make use of the efficiency of lower tier evaluations — benchmarks would be needed to screen material for specific beneficial use applications without extensive testing.

The intent of screening-level criteria is to conservatively identify contaminant concentrations below which no adverse effects, or unacceptable risks, are anticipated. In such cases where concentrations fall below the screening level, the material is deemed suitable, and no further testing or risk assessment would be needed. Exceedance of the screening criteria, however, does not indicate that the material is unsuitable; rather, it indicates the need to move to a higher tier of effects-based testing, as emphasized by Peddicord and Lee (1998). Screening, as a decision point in the evaluation process, is illustrated in Figure 1. A screening value should not be used as a cleanup level or as an indication of adverse impacts.

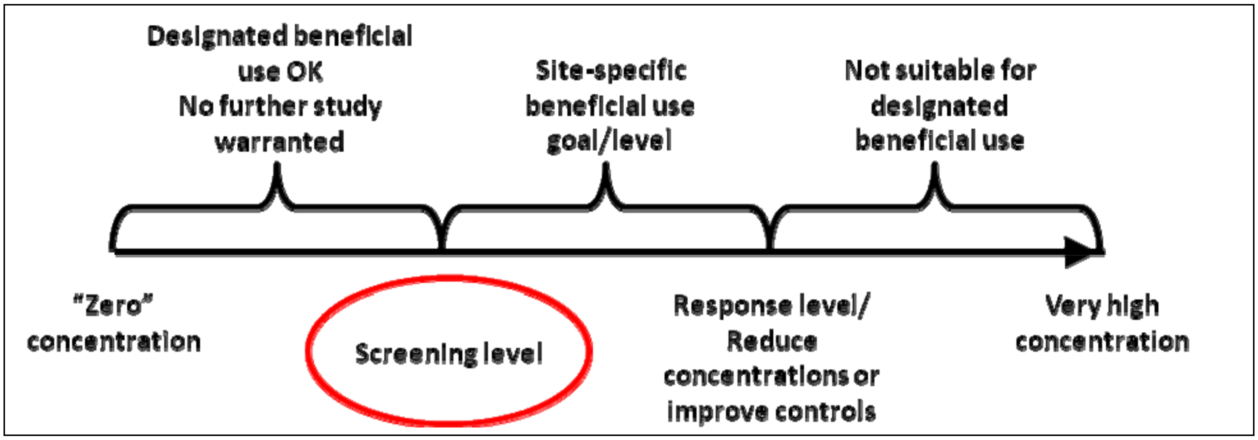

Figure 1. Screening level application in decision making (adapted from (USEPA 1996)).

As mentioned, there is a wide range of potential beneficial uses. Because of differences in exposures, receptors, environmental pathways, and other ecosystem variables, which in turn define "risk," the contaminant concentration that is acceptable will vary for different locations and uses. For example, dredged material used for habitat restoration may place organisms in direct contact with dredged material contaminants; whereas dredged material used for construction may be buried or encapsulated isolating the organisms from contact with the contaminants. From a human health perspective, use of dredged material for beach nourishment or recreational areas may have a greater probability of risk to human health than strip-mine reclamation. Accordingly, it is difficult to develop a single criterion that is conservative for all scenarios without being overly conservative for the majority of uses. Therefore, separate screening criteria may be needed for different categories of uses to account for varying degrees of exposure.

OBJECTIVE: The objective of the work presented here was to review existing risk-based criteria and their basis, in order to establish a criteria development methodology specifically for evaluation of dredged material for beneficial use. The wide range of beneficial use applications with varying potentials for exposure, as well as the heterogeneous nature of the sediments and contaminants themselves, presents a significant challenge in the development of a universal criteria development methodology. The wide range of values observed in the limited existing criteria, and the lack of consistency in the basis of these criteria support the urgent need for such criteria. As criteria specifically for dredged material are lacking, the criteria reviewed were developed for soils.

In order to gain widespread acceptance and use, the criteria to be developed should address the following attributes:

- Transparency. How the criteria are selected/generated should be readily apparent and scientifically defensible. Where possible they should be adapted from already accepted procedures.
- Risk-based. Since risk assessment is an accepted scientific method that links a given exposure with a given effect, risk principles should be employed in the development of criteria.

- Bioavailability. Because bioavailability determines the extent to which contaminants are taken up by organisms, the mere presence of a contaminant does not constitute a risk. This factor should be considered in estimating risk and in establishment of criteria development.
- Specific to characteristics of dredged material. The criteria should consider the unique characteristics of dredged material that affect contaminant mobility, including organic carbon, response to change in redox conditions, grain size, and other geochemical properties.
- Range of potential beneficial uses. There is a broad spectrum of potential beneficial uses with diverse environmental conditions, pathways and receptors. These may include aquatic, upland or wetland uses in saline or freshwater environments and in industrial, residential, or sensitive ecological areas. Further, many sites are transitional. Both immediate and future uses of the site must be considered and evaluated as part of the criteria development process.
- Regulatory and public acceptance. The criteria must be scientifically sound such that they can gain the acceptance of federal and state regulatory agencies, as well as being understandable and adequately protective from the public's point of view.
- Strategy. A clear strategy or procedure for applying the criteria is needed for successful and efficient implementation.

Risk Approach. An understanding of what is meant by environmental risk is fundamental to development of criteria for beneficial use. The USEPA and the USACE have adopted the risk assessment paradigm (National Research Council 1983) for the evaluation and management of contaminated dredged material. The risk paradigm relies on risk assessment approaches to determine whether contaminated media pose unacceptable risks to human health and the environment (Interstate Technology and Regulatory Council (ITRC) 2005). A risk assessment involves evaluation (descriptive and quantitative) of contaminant sources, transport pathways, routes and duration of exposure, receptors, exposure and effects assessments, toxicity evaluations for plant and animal (non-human) populations, carcinogenic and non-carcinogenic effects estimates for human populations, and a statement of acceptable levels of impacts (toxicity, reproductive success, cancer, etc.). The risk paradigm provides a conceptual framework needed for a consistent and meaningful review of criteria for beneficial use of dredged material.

Risk assessment, a multidisciplinary effort, provides valuable and sometimes essential information about a project, but consumes considerable time and resources. It is generally structured to involve the four major elements listed in the text box. The terms used for human health vs. ecological risk assessment differ slightly, but the overall approach is similar. Use of risk assessment in dredged material management has been discussed in detail in (PIANC 2006), (USEPA/USACE 2004), (Great Lakes Commission 2004), (ITRC 2005), (Moore et al. 1998), and (Cura et al. 1999).

Elements of risk assessment:
- Hazard identification/ Problem formulation
- Exposure assessment
- Dose response assessment/ Effects assessment
- Risk characterization and uncertainty

Cura, et al. (1999) describes the relationships between human health and ecological risk assessment. Ecological risk assessment focuses on potential risk to nonhuman biota, whereas human health risk assessment focuses on carcinogenic and non-carcinogenic risk to humans. Physical and chemical processes that drive the distribution of contaminants — i.e., the exposure assessment — will not change between the two types of risk assessment. Linking the two are the estimates of contaminant uptake by biota (evaluated in the ecological risk assessment) to humans if people eat that organism. They diverge where and how toxicological processes and endpoints are treated for the receptor species and how these processes relate to potential effects. Development of criteria for beneficial use must consider both human health and ecological risk.

One of the first tasks in scoping or formulating the problem is to identify contaminants, contaminant transport pathways, exposure routes, receptors, links between contaminants, pathways, and receptors, and assessment and measurement endpoints. An analysis plan must also be identified. Taken together, these comprise what is known as the conceptual site model (CSM). Common contaminant transport pathways for dredged material placed at an upland (not covered) site for beneficial use include volatilization, fugitive dust, runoff, leaching and infiltration, groundwater discharge, and plant and animal uptake (Great Lakes Commission 2004). Exposure routes of most importance include human dermal contact, human ingestion, human inhalation, biota ingestion, biota bioaccumulation, and plant toxicity. Pathways and receptors for a shoreline site where dredged material is used for beach nourishment could include resuspension of sediment particulates and transport by stream flow or wave action. Exposure routes would be similar in nature but with different types of organisms.

The development of screening criteria applies aspects of the exposure effects and risk characterization steps of the risk assessment paradigm. Contaminant concentrations below which the human health and ecological risks are acceptable are determined based on algorithms for carcinogenic, non-carcinogenic and mutagenic effects for human health and on toxicological endpoints for plants and animals. In a reversal of the normal risk assessment process, the acceptable exposure concentration is determined to be the screening value against which exposure assessment results can be compared for a given project.

CONSIDERATIONS SPECIFIC TO BENEFICIAL USE OF DREDGED MATERIAL

Exposure. Exposure of receptors to contaminants in dredged material used beneficially will vary depending on the properties of the dredged material, proximity of the placement (e.g., residential or industrial locations), and potential for contact or contaminant transport, during processing and placement, or after placement. Exposure potential will also be influenced by the nature of the placement (surface vs. subsurface, for example) and engineering controls employed to stabilize the material or limit access. Appropriate criteria will reflect differences in exposures and receptors.

The goal and the challenge is to provide reasonable criteria that are conservative enough to be protective, yet do not screen out a majority of uses where exposure potential is clearly limited. To standardize a procedure to evaluate exposure risks, beneficial uses may be grouped into categories of similar exposure potential. While exposure will be case specific, some conservative assumptions could be made to develop criteria for several categories of beneficial use.

The potential for exposure of human and ecological receptors should be considered in development of beneficial use criteria categories and in subsequent screening investigations. Proximity, land use, engineering controls and access are some of the relevant considerations in making this determination. For example, human exposure may not be a concern for remote locations, or locations with limited access, but will likely be a concern where direct access or material/contaminant transport (such as leaching of contaminants into a drinking water aquifer) is a possibility. For construction activities where dredged material may be used for fill purposes, the dredged material may be covered by layers of clean soil, asphalt or concrete, or building structures which would severely limit exposure potential for both human and ecological receptors by eliminating pathways. Project scale is also a consideration; the risk associated with small areas or short-term exposures should be less than that for large area or long-term exposures.

Exposure evaluations should be based on the final beneficial use product as opposed to the original dredged material, as properties may be significantly different. Also, as calculations for exposure are based on given assumptions about potential land use, it is important to consider the potential for changes over time. For instance, while material may initially present little exposure beneath a parking lot, the exposure scenario would change if the pavement is removed. As is commonly done at Brownfields and in other contaminated scenarios, sometimes it may be necessary to impose deed restrictions to ensure a change in land use does not increase exposure and, therefore, the risk associated with dredged material contaminants. Also, transformations resulting from processing, changes in environment or aging may alter the contaminant concentrations, forms and mobility. For instance, incorporation of organic matter during soil manufacturing may dilute contaminant concentrations and stabilize contaminants with a high affinity for organic matter. Conversely, increased solubility, mobility and bioavailability of metals may be expected for upland uses where the material becomes oxidized over time.

Bioavailability. As defined by the National Research Council (NRC 2003), "bioavailability processes are the individual physical, chemical, and biological interactions that determine the exposure of plants and animals to chemicals associated with soils and sediments." If only a fraction of a contaminant is available to be taken up by an organism, then the fraction that is unavailable does not contribute to risk. That which is not bioavailable does not pose a risk, unless there is the potential to later become bioavailable due to a change in conditions. While ignoring bioavailability is conservative, it may be overly so, resulting in unnecessary restrictions or increased cost for further investigation. Criteria based on bioavailable contaminant concentrations rather than total concentrations are more likely to be adequately protective, without being overly conservative.

The bioavailability of contaminants in soils or sediments is impacted by chemical processes such as sorption/desorption, transformation/degradation, and oxidation/reduction. Various soil components have the ability to absorb contaminants such that they are inaccessible for biouptake. Some components that may impact bioavailability include organic carbon or black carbon, and clay minerals. Geochemical parameters such as redox state, pH, salinity or presence of acid volatile sulfides can also influence contaminant bioavailability. For instance, oxidation of sediments may cause solubilization of metals. Geochemical changes are particularly important to consider for sediments originating from a reducing environment and being placed in an upland environment where they will become oxidized. In establishing and applying criteria, one must consider the properties of the material not only at the time of placement, but also over time, due to the potential for such changes to occur. Again, the site-specific beneficial use must be designed to control these potential transformations over time.

In order to apply bioavailability concepts one needs some measure of bioavailability. Unfortunately, the degree to which contaminants are bioavailable varies greatly among contaminants, soil types, organisms and environmental conditions. Ehlers and Luthy (2003) discuss tools to evaluate bioavailability. Instruments, such as X-ray diffraction, scanning electron microscopy, nuclear magnetic resonance and others, are available to explore the geochemical compartments that contain the contaminant; however, due to their sophistication they tend to be better suited as research tools. Numerous simple empirical extraction tests using solid-phase sorbents are used to estimate the bioavailable fraction of contaminants. This includes diffusive gradient in thin films (DGT), solid-phase microextraction (SPME), semi permeable membrane devices (SPMD) and C-18- and Tenax-containing disks or beads (Cornelissen et al. 2005). In vitro extractions that mimic mammalian digestive processes have also been proposed. Bioassays are used to study both influential biological processes and physical and chemical processes. NRC (2003) notes that the various tools available have both strengths and weaknesses and multiple tools may be necessary for a "weight-of-evidence" approach. Default adjustment factors are available for chemical absorption in human health risk assessment (NRC 2003), but NRC warns that their use may not be protective and appropriate for all circumstances and recommends the use of site-specific measurements.

There is a significant amount of variation in the manner in which bioavailability is presently incorporated in various screening evaluations. The ITRC Contaminated Sediments Team (*http://www.js3design2.com/con_sed_web_jws/consed_3.htm*) discusses how bioavailability is considered in the evaluation and remediation of contaminated sediment sites. For site-specific investigations, bioavailability is considered during the scoping process in developing a conceptual site model, but is rarely taken into consideration during the screening process. In risk assessment, bioavailability is usually factored in by use of default values or site-specific data inserted into exposure equations. Human health risk assessment considers either absolute or relative bioavailability as a factor in exposure assessments; absolute bioavailability refers to the fraction of the applied dose that is absorbed, while relative bioavailability reflects the difference between uptake of solid-bound contaminant vs. contaminant in the dosing medium used for the toxicity study (NRC 2003). Bioavailability considerations have influenced cleanup goals at a number of sites; NRC (2003) cites seven cases where bioavailability adjustments ranged between 10 % and 80 % from the default value. However, explicit bioavailability assessments are not a regular feature of site-specific risk assessment. Ecological risk assessment is more complex due to the many species. Direct contact is the pathway most frequently driving ecological risk assessment for invertebrates and wildlife. For direct contact, partitioning techniques such as acid volatile sulfide and biota-soil/sediment-accumulation factors may be used to predict partitioning between phases (solid – not bioavailable, aqueous or within an organism), but these techniques have substantial uncertainties. As NRC (2003) warns, when applied in the development of cleanup goals, bioavailability should only be used to adjust criteria when site conditions are unlikely to change substantially over time. To incorporate bioavailability concepts into beneficial use screening evaluations, bioavailability should probably be considered on a case-by-case basis, comparing the bioavailable concentrations of contaminants in specific materials to criteria that are based on conservative assumptions regarding bioavailability.

SYNOPSIS OF AVAILABLE SCREENING CRITERIA. There is not a uniform or national set of criteria directly applicable to beneficial use of dredged material. However, a number of U.S. and international agencies have developed contaminant criteria, screening levels, or cleanup goals for air, water, soils, and sediments related to human health or ecological receptors that have potential applicability to beneficial uses of dredged material. Development for most of these guidelines was driven by the need for assessment of risks at contaminated sites identified for potential remediation. Building upon commonly accepted methodologies is always good practice when developing evaluation procedures. It fosters acceptance at the scientific, regulatory and public level. Even if the specific contaminant concentrations are not comparable, the processes used to derive these values may be adapted to certain dredged material beneficial uses. Reports documenting criteria development are summarized below.

Authors for a number of literature references have considered criteria developed by national, state, and international organizations. Brandon and Price (2007) and Great Lakes Commission (2004) listed state and national criteria that may be potentially applicable to dredged material beneficial uses. Barron and Wharton (2005) surveyed methodologies for developing media (surface water, soil, sediment, and tissue) screening values for ecological risk assessment. Fishwick (2004) reviewed international approaches to setting soil screening values for use in the screening phase (Tier 1) of the United Kingdom Environment Agency's ecological risk assessment framework. A survey of the screening values from 15 European Union nations was summarized for the European Commission by Carlon (2007). Environmental Planning and Toxicology, Inc. (1999) reviewed methods for developing ecological soils quality guidelines and criteria in support of the development of ecological soil screening levels for the USEPA. Friday (2005) provides ecological screening values developed by others for surface water, sediment, and soil. Friday describes how the screening values were derived and recommends benchmarks that can be used for ecological risk assessment at the Department of Energy (DoE) Savannah River Plant.

Most of these reviews focused on screening level approaches or values presented in the same source documents. For human health assessments, state environmental agencies and U.S. federal agencies generally rely on the approach developed by the USEPA for risk assessment at Superfund sites as originally presented in USEPA 1991. Ecological risk values usually refer to USEPA's ecological screening level document (USEPA 2005), although only a small number of states include ecological levels in their publications. On the international stage, Canada and the Netherlands offer comprehensive documentation for how they develop screening values. Many European countries have followed the lead of the Dutch in this respect. Table 1 presents an overview of the ecological and human health screening value processes for the USEPA, Canadian Council of Ministers of the Environment (CCME) and the Netherlands. Because these agencies offer some originality in their screening level development, their approaches will be discussed in some detail in the paragraphs that follow. Then, descriptions of values available from other selected federal, state, and local organizations will be briefly summarized. Our goal is to identify features of available screening criteria that can be applied or adapted to developing screening criteria for dredged material beneficial uses.

Table 1. Overview of features for soil screening levels developed on a national scale.

	Land use				Receptors							Exposure route					
	Residential	Parkland	Industrial	Commercial	Humans--Adult	Humans--Child	Plants	Invertebrates	Wildlife	Microbes	Livestock	Direct contact	Ingestion	Inhalation	Dermal	Groundwater	Bioaccumulation
USEPA Eco-Soil Screening Levels							•	•	•			•					•
USEPA Regional Screening Levels	•		•		•	•							•	•	•	•	
CCME—Environmental	•	•	•	•			•	•	•		•	•	•			•	•
CCME—Human Health	•	•	•	•	•	•							•	•	•	•	•
The Netherlands Environmental Risk Limits					•		•	•	•	•	•[1]	•	•	•	•	•	•
NOAA Screening Values (SQuiRTs)							•	•	•	•							•
ORNL Eco Screening Benchmarks							•	•	•	•		•	•				•

[1]Values based on human consumption of milk and meat

USEPA Ecological Soil Screening Levels (Eco-SSLs). Probably the most extensively documented screening levels were developed by the USEPA (2005) for the purpose of "conserving resources by limiting the need for EPA and other risk assessors to perform repetitious toxicity data literature searches and data evaluations for the same contaminants at every [contaminated] site." USEPA's intentions are that the procedures and processes used to develop the screening levels are sufficiently transparent for use by others to derive values for other contaminants. This transparency will enable adaptation of the processes to improve their applicability to dredged material contaminants, pathways, and receptors.

Four steps were used in the general approach to deriving a contaminant-specific screening level as shown in Table 2 (USEPA 2005). As indicated by the brief overview in Table 2, the process was quite detailed and included an exhaustive literature search. USEPA has catalogued relevant data for other contaminants and receptors for public use. Twenty-three contaminants have been identified as priority for development of Eco-SSLs, including 17 metals, dieldrin, hexahydro - 1,3,5-trinitro-1,3,5-triazine (RDX), trinitrotoluene (TNT), 1,1,1-trichloro-2,2-bis (p-chlorophenyl)ethane (DDT) and metabolites (DDE and DDD), pentachlorophenol, and polycyclic aromatic hydrocarbons (PAHs). Polychlorinated biphenyls (PCBs) were originally on the list, but USEPA concluded that development of a PCB soil screening value was not warranted because of the known high persistence and toxicity of PCBs, and the conservative nature of the Eco-SSLs. USEPA's stance is "if PCBs are detected in soil above background levels, the PCBs are probably site related and therefore should be included as a contaminant of potential concern in the baseline risk assessment." Detailed documentation for derivation of Eco-SSLs for these contaminants may be found at http://www.epa.gov/ecotox/ecossl/index.html.

Table 2. Process for development of Eco-SSLs (USEPA 2005).

Step	Plant and soil invertebrates	Wildlife (birds and mammals)
Conduct electronic literature search	22 exclusion criteria (e.g., medical studies, field studies, modeling, irrelevant data, etc.) used to select appropriate papers 7,600 titles/abstracts screened 5,200 papers acquired	Searched for dose-response literature for mammals and birds 44,000 papers identified 66 rejection criteria
Determine acceptability of study for use in deriving Eco-SSL	11 study acceptance criteria (e.g., scientific reporting, primary sources, adequacy of adequate study details, chronic toxicity studies, organic matter ≤10%, endpoints relevant to ecological receptors, relevant endpoints, etc.) Endpoints are growth, physiology (plants only), population, and reproduction 7% of 5,200 papers passed	Accepted studies with oral route of exposure, at least two exposures (control and contaminant) Endpoints are behavioral, biochemical, growth, mortality, pathology, population, physiology, and reproduction Only chronic studies Include NOAEL[4] and LOAEL[5]
Extract, evaluate, and score data from accepted studies	9 study evaluation criteria (e.g., bioavailability, experimental design and methods, statistics reported, organisms used, etc.) Studies scoring 11 of 18 or higher (61%) retained	Evaluated and scored 10 attributes of toxicological study (e.g., chemical form, dose calculation, route of exposure, duration of exposure, statistical power, and adherence to test guidelines) on the basis of relevance of setting a TRV[6] Studies scoring 65 out of 100 (65%) retained
Derive value	Derive according to established procedure, including sorting data by bioavailability score (higher bioavailability favored), completing quality assurance review by expert panel, and calculating value Value is geometric mean of EC_{20}[1], MATC[2], or EC_{10}[3] values (order of preference)	Hazard quotient=1.0 Generic food chain model applied TRV set as equal to the geometric mean of NOAEL values for growth and reproduction or the highest bounded NOAEL below the lowest bounded LOAEL for growth, reproduction or survival Considered NOAEL and LOAEL endpoints and included unbounded NOAEL values but not unbounded LOAEL values.
Notes	[1]EC_{20}=Effects concentration 20% [2]MATC=Maximum acceptable toxicant concentration [3]EC_{10}= Effects concentration 10%	[4]NOAEL=No-observed adverse effect level [5]LOAEL=Lowest-observed adverse effect level [6]TRV=Toxicity reference value

Eco-SSLs primarily address existing or placed terrestrial soils from the surface down to the plant root depth or the depth accessed by burrowing animals. The process addresses only complete exposure pathways, which may be defined as the ability of a contaminant to travel from the source to ecological receptors and be taken up by the receptors via one or more exposure routes. If natural habitat for a given receptor is not available, such as in an industrial area, or the contaminant is buried below the biologically active depth, there may not be a complete exposure pathway. Exposure pathways addressed by Eco-SSLs currently available are listed in Table 3.

Dermal and inhalation routes for wildlife were considered but deemed less significant for the contaminants being evaluated than those routes in Table 3 (USEPA 2005).

Table 3. Exposure pathways addressed by USEPA Eco-SSLs.

Receptor	Pathways
Birds and mammals	Ingestion of soils during grooming, feeding, and preening
	Ingestion of food contaminated as a result of uptake of soil contaminant
Plants	Direct contact
Soil invertebrates	Direct contact
	Soil ingestion

The Eco-SSLs for plants and soil invertebrates apply to soils where the pH falls between 4.0 and 8.5, and where the organic matter content is less than or equal to 10%[1]. Although these values were derived for upland soils, USEPA indicates that they may also be appropriate for some wetland soils because the wildlife receptors could be representative for mammals and birds. However, reptiles and amphibians have not yet been included in the Eco-SSLs. The plant and invertebrate exposures and effects could be used although the reduced bioavailability of contaminants in wetland soils by virtue of higher organic carbon and reduced metal solubility would make the Eco-SSLs even more conservative. The values are not recommended where the soils are regularly flooded (sediments), or for soils with organic matter greater than 10% or with pH less than 4. Also not addressed are groundwater and surface runoff or surface water pathways.

Eco-SSL values for plants and invertebrates were based directly on the NOAELs and LOAELs. However, the values for birds and mammals also considered bioaccumulation through food intake. The wildlife risk model used for deriving the Ecol-SSLs is mathematically shown in the equation below:

$$HQ_j = \frac{\left\{\left[Soil_j \times P_s \times FIR \times AF_{js}\right] + \left[\sum_{i=1}^{N} B_{ij} \times P_i \times FIR \times AF_{ij}\right]\right\} \times AUF}{TRV_j} \tag{1}$$

where:

HQ_j = Hazard quotient for contaminant (j) (unitless),

$Soil_j$ = Concentration of contaminant (j) in soil (mg/kg dry weight),

N = Number of different biota types in diet,

B_{ij} = Concentration of contaminant (j) in biota type (i) (mg/kg dry weight),

P_i = Proportion of biota type (i) in diet,

FIR = Food ingestion rate (kg food [dry weight]/ kg BW [wet weight] /day),

AF_{ij} = Absorbed fraction of contaminant (j) from biota type (i) (for screening purposes set equal to 1),

AF_{js} = Absorbed fraction of contaminant (j) from soil (s) (for screening purposes set equal to 1),

[1] Organic matter for sediments is generally less than 10%. However, sediments are generally higher in organic matter than soils.

TRV_j = Toxicity reference value (mg/kg BW/day),

P_s = Soil ingestion as proportion of diet,

AUF = Area use factor (for screening purposes set equal to 1).

After substituting the parameters assumed to equal 1, assuming HQ equal 1, and substituting the following expression for B_{ij} , the Ecological Soil Screening Level (*Eco—SSL*) can be calculated.

$$B_{ij} = BAF_{ij} \times Soil_j \tag{2}$$

where BAF_{ij} = Soil-to-biota bioaccumulation factor (*BAF*) for contaminant (*j*) for biota type (*i*),

$$Eco - SSL = Soil_j = \frac{TRV}{FIR \times \left(P_s + BAF_{ij}\right)} \tag{3}$$

Eco-SSLs may be calculated for all relevant pathways, and a screening value selected by choosing the lowest of the calculated SSL values.

USEPA Regional Screening Levels for Human Health. Several USEPA Regional Offices have developed tables for contaminant screening of soil, water, and air for human health risk assessments at Superfund sites: Region 3 published Risk Based Criteria, Region 6 published Human Health Medium-Specific Screening Levels, and Region 9 published Preliminary Remediation Goals. Recently (2008), these regions — with the assistance of the Oak Ridge National Laboratory — joined forces and developed a consensus table accompanied by a web-based calculator for risk-based screening levels using the latest toxicity values, default exposure assumptions and physical and chemical properties of the contaminants. Screening levels found in these tables are based on potential Applicable or Relevant and Appropriate Requirements (ARARs) or on "risk based calculations that set concentration limits using carcinogenic or systemic human toxicity values under specific exposure conditions." http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/index.htm

The regional screening levels (RSLs) developed by these regions are based on default exposure parameters and factors that represent Reasonable Maximum Exposure conditions. Background for calculating the screening levels is available in (USEPA 1991), (USEPA 1996), and (USEPA 2002), but the RSLs are updated semiannually. The basis for these calculations is toxicity values published in the literature or in readily available data bases. USEPA defines a hierarchy of human health toxicity values for use in calculating the screening levels.

1. USEPA's Integrated Risk Information System (IRIS)
2. USEPA's Provisional Peer Reviewed Toxicity Values (PPRTVs) (restricted to USEPA users, but other users may request access)
3. Agency for Toxic Substances and Disease Registry (ATSDR) minimal risk levels (MRLs)
4. California Environmental Protection Agency Office of Environmental Health Hazard Assessment's Chronic Reference Exposure Levels and Cancer Potency Values
5. Screening toxicity values in certain PPRTV assessments
6. Health Effects Assessment Summary Tables (HEAST) website (restricted to USEPA users)

The equations used by USEPA are based on an average daily dose or exposure to a receptor—either a human adult or a child. The average daily dose is a function of chemical concentration, ingestion rate, time of exposure, and body weight and is calculated by the general equation below:

$$Average\ Daily\ Dose = Chemical\ Concentration \times \left\{ \frac{Ingestion\ Rate \times Exposure\ Duration \times Exposure \times Frequency}{Body\ Weight \times Averaging\ Time} \right\} \quad (4)$$

The dose calculations become complicated because several of these variables may not be measureable directly, but require a mathematical expression or model to estimate the concentration accessible to the user, e.g., the concentration in air as the contaminant volatilizes from soil or water. For a given environmental pathway, the ingestion rate, exposure times, and body weight may be specified by the user. Averaging time is the period over which the exposure is averaged. The risk of the exposure to the receptor takes two forms—non-cancer and carcinogenic. The non-cancer risk is expressed by the equation:

$$Hazard\ Quotient = \frac{Average\ Daily\ Dose}{Reference\ Dose} \quad (5)$$

The hazard quotient is usually set to equal 1 so that the screening level is medium (soil, air, water) concentration producing an average daily dose equal to the reference dose. The carcinogenic risk is expressed by:

$$Risk = Average\ Daily\ Dose \times Cancer\ Slope\ Factor \quad (6)$$

Substituting and rearranging terms in the average daily dose and cancer risk equations yields a screening level equation of the form:

$$Risk\ Based\ Screening\ Level = \frac{Body\ Weight \times Averaging\ Time \times Risk\ Level}{Ingestion\ Rate \times Exposure\ Frequency \times Exposure\ Duration \times Cancer\ Slope\ Factor} \quad (7)$$

The non-cancer reference parameters and the cancer slope factors used for the above equations are selected from the following, depending on the type of risk and the exposure route or pathway:

- Reference dose—daily oral exposure to the human population likely to be without an appreciable deleterious risk during a lifetime.
- Reference concentrations—concentration where continuous inhalation exposure to the human population is likely to be without an appreciable deleterious risk during a lifetime.
- Cancer slope factor—upper bound estimate of the probability of a response per unit intake of a chemical over a lifetime.
- Inhalation unit risk—upper bound excess lifetime cancer risk estimated to result from continuous exposure to an agent at a concentration of $1 \mu g/m^3$ in air.

USEPA presented the conceptual site model diagram (Figure 2) to illustrate the exposure routes evaluated for the RSLs. The specific exposure scenarios with numbers applicable to beneficial use of dredged material are listed in Table 4.

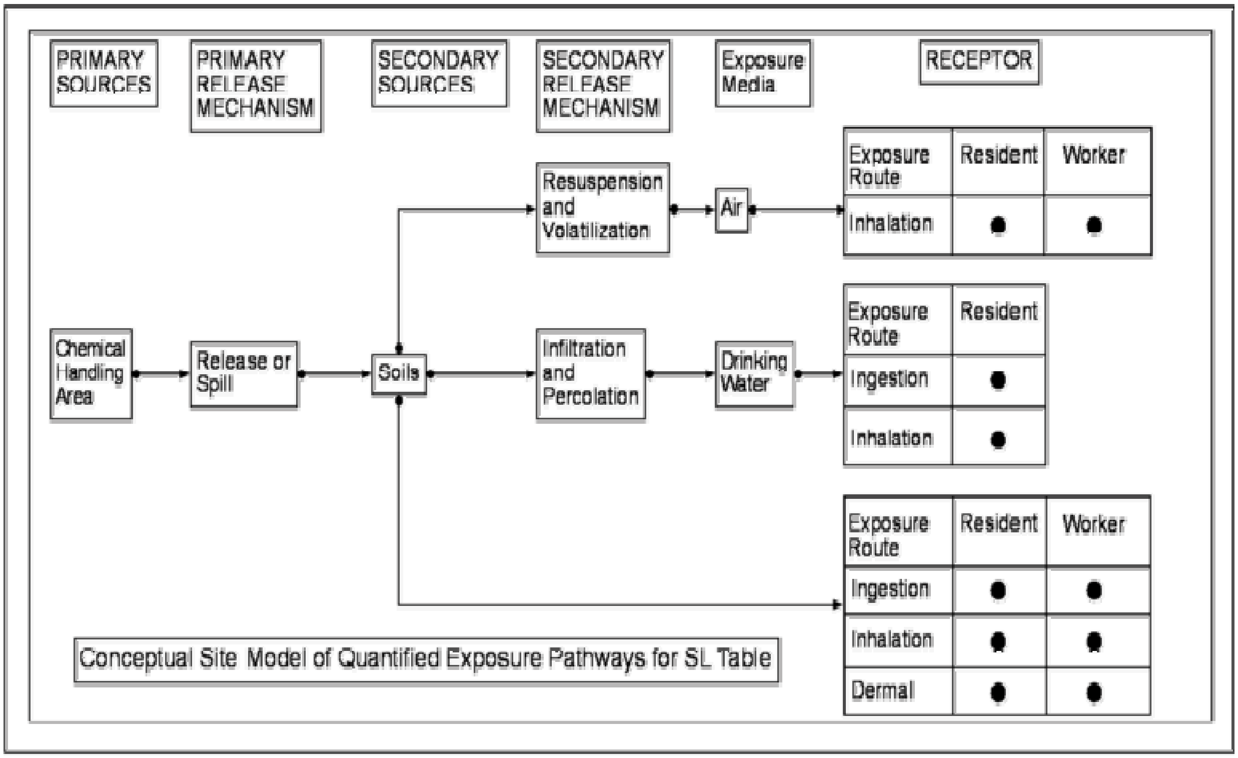

Figure 2. Conceptual site model used for calculation of USEPA regional soil screening levels (USEPA 2010).

Canadian Council of Ministers of Environment (CCME) Environmental and Human Health Soil Quality Guidelines.

These guidelines were developed for the protection of ecological receptors in the environment and/or for the protection of human health associated with the following land use categories: agricultural, residential/parkland, commercial and industrial. ***Environmental guidelines*** are aimed at the protection of the terrestrial ecosystem, including wildlife and livestock, by considering adverse effects resulting from direct contact exposure to soil contaminants and from ingestion of contaminated soil and food. Also included in these guidelines are indirect exposures through use of contaminated groundwater for agricultural purposes, migration to nearby surface waters, and migration to adjacent properties due to wind and water erosion (Canadian Council of Ministers of the Environment 2006). ***Human health guidelines*** consider direct and indirect exposure to soil contaminants. A generic human exposure scenario is assumed for each land use.

Environmental guidelines. Acceptable data from the literature are reviewed to determine the environmental health and behavior of the contaminant and to derive an effects-based soil quality guideline for invertebrates, plants, and microbes from toxicity data. Environmental pathways that consider ingestion of contaminated soil and food are described for livestock and terrestrial wildlife. Groundwater pathways are addressed by modeling partition of contaminants into soil pore water and subsequent transport into an aquifer.

Table 4. Exposure pathways for soils evaluated in development of USEPA Regional Screening Levels for human health.

Exposure Pathway No.	Locale	Medium	Release Mechanisms	Exposure Route	Receptor	Risk Type	SL Source	Toxicity Value*
1	Residential	Soil	Direct	Ingestion	Human-child	Non-cancer	USEPA RSL	RfD_O
2	Residential	Soil	Direct	Ingestion	Human-adult	Carcinogenic	USEPA RSL	CSF_O
3	Residential	Soil	Direct	Ingestion	Human-adult	Mutagenic	USEPA RSL	CSF_O
4	Residential	Soil	Dust	Inhalation	Human-child	Non-cancer	USEPA RSL	RfC
5	Residential	Soil	Dust	Inhalation	Human-adult	Carcinogenic	USEPA RSL	IUR
6	Residential	Soil	Dust	Inhalation	Human-adult	Mutagenic	USEPA RSL	IUR
7	Residential	Soil	Direct	Dermal	Human-child	Non-cancer	USEPA RSL	RfD_O
8	Residential	Soil	Direct	Dermal	Human-adult	Carcinogenic	USEPA RSL	CSF_O
9	Residential	Soil	Direct	Dermal	Human-adult	Mutagenic	USEPA RSL	CSF_O
10	Composite worker	Soil	Direct	Ingestion	Human-adult	Non-cancer	USEPA RSL	RfD_O
11	Composite worker	Soil	Direct	Ingestion	Human-adult	Carcinogenic	USEPA RSL	CSF_O
12	Composite worker	Soil	Dust	Inhalation	Human-adult	Non-cancer	USEPA RSL	RfC
13	Composite worker	Soil	Dust	Inhalation	Human-adult	Carcinogenic	USEPA RSL	IUR
14	Composite worker	Soil	Direct	Dermal	Human-adult	Non-cancer	USEPA RSL	RfD_O
15	Composite worker	Soil	Direct	Dermal	Human-adult	Carcinogenic	USEPA RSL	CSF_O
16	Indoor worker	Soil	Direct	Ingestion	Human-adult	Non-cancer	USEPA RSL	RfD_O
17	Indoor worker	Soil	Direct	Ingestion	Human-adult	Carcinogenic	USEPA RSL	CSF_O
18	Indoor worker	Soil	Dust	Inhalation	Human-adult	Non-cancer	USEPA RSL	RfC
19	Indoor worker	Soil	Dust	Inhalation	Human-adult	Carcinogenic	USEPA RSL	IUR
20	Outdoor worker	Soil	Direct	Ingestion	Human-adult	Non-cancer	USEPA RSL	RfD_O
21	Outdoor worker	Soil	Direct	Ingestion	Human-adult	Carcinogenic	USEPA RSL	CSF_O
22	Outdoor worker	Soil	Dust	Inhalation	Human-adult	Non-cancer	USEPA RSL	RfC
23	Outdoor worker	Soil	Dust	Inhalation	Human-adult	Carcinogenic	USEPA RSL	IUR
24	Outdoor worker	Soil	Direct	Dermal	Human-adult	Non-cancer	USEPA RSL	RfD_O
25	Outdoor worker	Soil	Direct	Dermal	Human-adult	Carcinogenic	USEPA RSL	CSF_O
26	Any	Soil	Groundwater	Ingestion	Human	Non-cancer	USEPA RSL	RfD_O / MCL
27	Any	Soil	Groundwater	Ingestion	Human	Carcinogenic	USEPA RSL	CSF_O / MCL

* CSF_O = Chronic oral slope factor
RfD_O = Chronic oral reference dose
IUR = Chronic inhalation unit risk
RfC = Chronic inhalation reference concentration
MCL = Maximum contaminant concentration (Drinking water standard)

Note: See http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/usersguide.htm for additional information

These guidelines acknowledge that contaminant fate and transport and bioavailability are dependent of soil physical characteristics. Where data are available, separate guidelines are developed for coarse textured soils (sand and gravel—median grain size greater than 75 microns) and fine textured (silt and clay—median grain size less than 75 microns).

The overall procedure used to develop Canada's soil quality environmental guidelines is illustrated in Figure 3. The purpose of the literature search is to collect published and nonproprietary data; the literature is examined and papers that can be scientifically verified are selected. Development of a soils contact guideline applies to all four land uses (previously listed). Methods used in order of priority are the weight of evidence method, lowest observed effect concentration method, and the median effects method (CCME 2006). The preferred approach is to derive a threshold effects concentration using the 25th percentile of the compiled effects-endpoints data distribution divided by an uncertainty factor, for agricultural and residential/parkland uses.

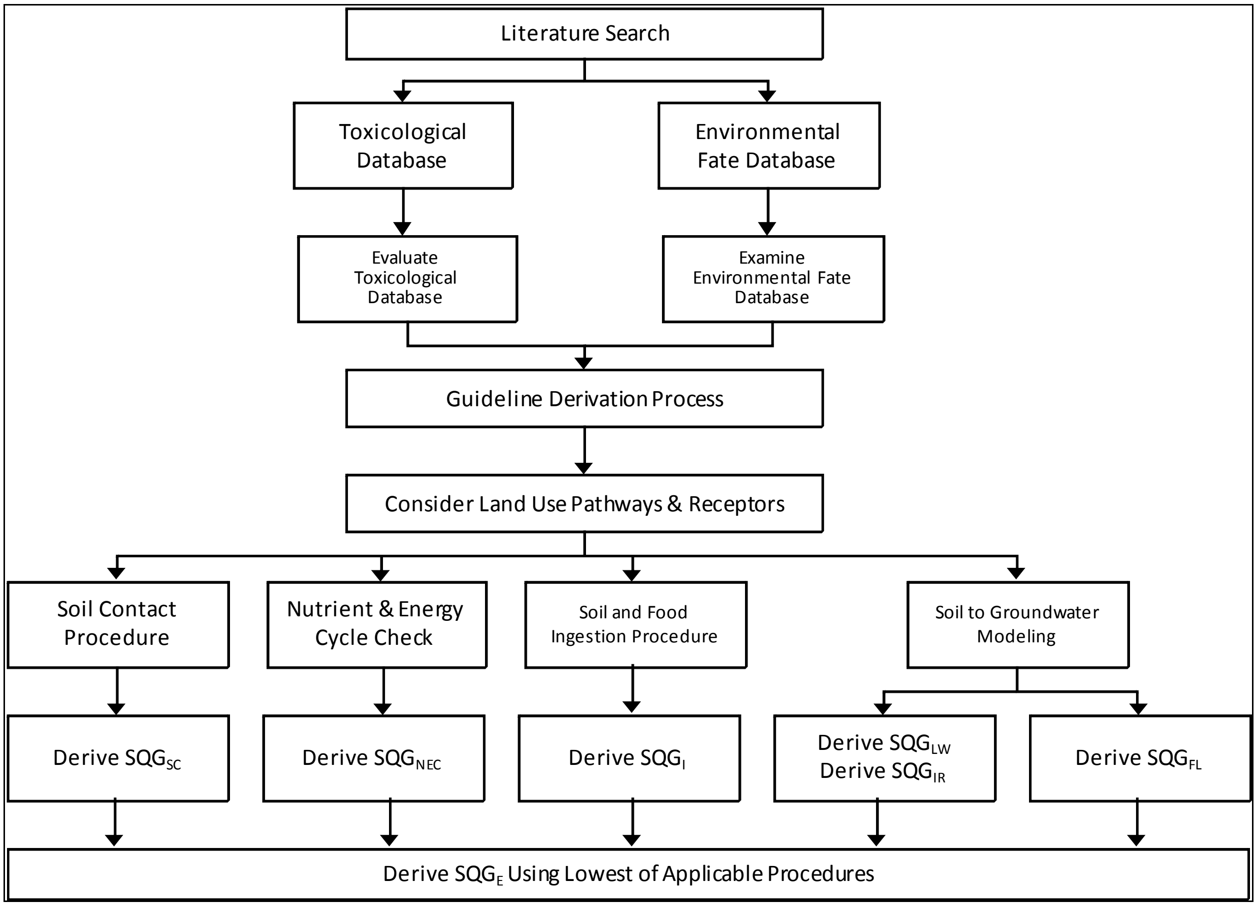

Figure 3. CCME soil guideline development procedure (CCME 2006).

The 50th percentile of the compiled effects-endpoints distribution is used for commercial and industrial land uses. The nutrient and energy cycle check uses professional judgment to assess the effect of contaminants on the ecological functions of the soil, including decomposition, respiration and organic nutrient cycles. Guidelines for soil and food ingestion first determine the daily threshold effect dose (DTED) for primary, secondary, and tertiary consumers and applying an uncertainty factor to the LOAEL for the most threatened species. The soil quality guideline for ingestion by primary consumers (SQG_{1C}) is calculated using an equation of form similar to Equation 3 above. Canada also considers leaching of contaminants into the groundwater and subsequent transport to a surface water body. Pore water concentration for organic contaminants is calculated using a distribution coefficient, and concentration 10 meters from the source is estimated using groundwater modeling; this concentration is compared to surface water quality criteria to determine the soil guideline value. Mixing and dilution in the surface waters is not considered, nor are inorganic contaminants. Guidelines for agricultural land uses (where a well is placed in a contaminated area) are estimated using the same partitioning and mixing models as for effects on fresh water life. The livestock guideline is calculated based on livestock DTED, body weight, and ingestion rate. Irrigation guidelines are based on water quality criteria for irrigation waters. The guideline for offsite migration from commercial or industrial sites due to erosion is derived using the universal soil loss equation to protect adjacent areas that may be more environmentally sensitive. The final environmental soil quality guideline is selected as the lowest contaminant concentration for the applicable land use and receptors or pathways.

Human health guidelines. CCME guidelines for protection of human health consider direct soil exposure (ingestion, inhalation, and/or dermal), migration of soil contaminants into drinking water supply, and volatilization of contaminants into indoor air. Additional checks are made for food ingestion and offsite migration due to wind and water erosion. Guidelines are developed by considering exposure through all relevant pathways and applying scientifically derived information, backed by professional judgment, where data gaps occur.

Human health soil guidelines threshold toxicants (for which there is a dose/concentration below which no adverse effects are expected to occur) and non-threshold toxicants (for which there is considered to be some probability of human harm at any level of exposure) are differentiated, taking into account daily background exposure from air, water, soil, food, and consumer products. Indirect exposure pathways are evaluated conservatively by applying simplified transport and redistribution models using generic site characteristics for a variety of site conditions (CCME 2006). Human exposure scenarios related to each land use are evaluated.

The NOAEL from toxicological studies involving experimental animals or from epidemiological studies of human populations is the preferred endpoint for threshold contaminants. If an NOAEL is not available, then a LOAEL may be used. Uncertainty factors are applied to the NOAEL or the LOAEL to derive a tolerable daily intake (TDI) to which a person can be exposed over a lifetime without harmful effects (CCME 2007). For non-threshold contaminants, the TDI is based on risk-specific doses (RSD) potentially causing cancer in 10^{-5} to 10^{-6} of the population. The estimated daily intake accounts for contributions from background concentrations (CCME 2007).

The soil quality guideline for human health (SQG_{HH}) is set by working backwards from the TDI or the critical RSD. Various routes of exposure for each land use are considered as shown in Table 5. Direct and indirect exposure pathways from soil are considered. CCME (2006 & 2007) present equations used to calculate the soil quality guideline for threshold and non-threshold substances. The primary indirect exposure pathways are migration of soil contaminants into groundwater used as drinking water and inhalation of contaminants volatilized indoors. As for the environmental guidelines, two check mechanisms (off-site migration and ingestion of food grown on contaminated soils) are available where data for other pathways are incomplete or of limited confidence. The lowest value derived from the applicable calculations, (i.e. most protective) is selected for each of the land uses as the final SQG_{HH}.

Table 5. Receptors and exposure pathways considered in the derivation of CCME's human health soils quality guidelines (CCME 2007).

Route of exposure	Agriculture	Residential/parkland	Commercial	Industrial
Sensitive receptor (Threshold contaminant)	Toddler	Toddler	Toddler	Adult
Sensitive receptor (Non-threshold contaminant)	Adult	Adult	Adult	Adult
Exposure period	24 hours/day 365 days/year	24 hours/day 365 days/year	10 hours/day 5 days/week 48 weeks/year	10 hours/day 5 days/week 48 weeks/year
Direct soil exposure pathways	Ingestion Dermal contact Inhalation	Ingestion Dermal contact Inhalation	Ingestion Dermal contact Inhalation	Ingestion Dermal contact Inhalation
Indirect soil exposure pathways	Groundwater Indoor air Produce, meat, & milk ingestion	Groundwater Indoor air Backyard produce	Groundwater Indoor air Off-site migration	Groundwater Indoor air Off-site migration

The final overall soil quality guidelines are selected as the lower of the two guidelines obtained for human health and the environment for each land use. In the final step, professional judgment regarding management considerations, plant nutritional requirements, geochemical background, and practical quantitation limits may override or modify the final SQG.

Oak Ridge National Laboratory Screening Levels, Oak Ridge National Laboratory (ORNL) Ecotoxicological Screening Benchmarks. In the mid 1990s, the U.S. Department of Energy's ORNL derived — or compiled from other sources — preliminary remediation goals for use as an ecological risk assessment tool. These ecological screening benchmarks identify contaminants, media, and receptors that may be at risk and that may require further investigation during an environmental risk assessment. The values for soils were developed primarily by the ORNL staff at a time when screening levels were not available from the regulatory community. Benchmarks were developed or obtained for the following types of exposure and classes of endpoint groups (Sample, et al. 1998):

- Exposure of aquatic biota to chemicals in water (National Ambient Water Quality Criteria or toxicity to piscivorous wildlife)
- Exposure of benthic biota to chemicals in sediments (values for sediment and pore water derived from seven sources)
- Exposure of terrestrial plants to chemicals in soil
- Exposure of soil invertebrates to chemicals in soil
- Exposure of soil functional groups to chemicals in soil
- Exposure of wildlife to chemicals in orally ingested materials (derived by iteratively calculating exposure estimates using different soil concentrations and soil-to-biota contaminant uptake models)

The ORNL guidance notes that "Remedial goals for soils should be modified based on the bioavailability of the contaminants of concern." (Efroymson, Sutter II, et al. 1997). Soil benchmarks for invertebrates were derived in a manner similar to the Effects Range Low (ERL) procedure outlined by Long and Morgan (1990) for sediment screening. The ERL is the tenth percentile of the distribution of toxic effects thresholds. ORNL rank-ordered the LOEC values for earthworms and microbial heterotrophs from the literature and picked a soil concentration that approximated the 10^{th} percentile of effects. Procedures are described for selecting values where the data sets are limited. (Efroymson, Will et al. 1997).

Screening values for terrestrial plants growing in soil were based on literature reports of toxicity tests of individual chemicals in laboratory, greenhouse, or field settings. The method for deriving the value was similar to the ERL method used for invertebrates. The phytotoxicity benchmarks were derived by rank-ordering the LOEC values and then picking a number that approximated the 10th percentile. This approach was justified by assuming that the phytotoxicity of a chemical in soil is a random variable, the toxicity of contaminated soil at a particular site is drawn from the same distribution, and the assessor should be 90% certain of protecting plants growing in the site soil. The major source of bias was noted as the use of soluble metal salts in the toxicity tests (Efroymson et al. 1997a).

The Netherlands Screening Values. The Netherlands National Institute for Public Health and the Environment (RIVM) derive environmental risk limits (ERLs) for protection of humans and environmental receptors from contaminants in surface water, groundwater, sediment, soil, and air. RIVM prescribes the following levels of protection (Van Vlaardingen and Verbruggen 2007):

- Negligible concentration (NC)
- Maximum permissible concentration (MPC) — the concentration at which no harmful effects are to be expected
- Maximum acceptable concentration (MAC) — the concentration protective against acute toxic effects exerted by exposure to short-term or transient peak concentrations
- Serious risk concentrations (SRC) — the concentration where serious risks are to be expected

For water and sediment, the RIVM uses the same methodology as that prescribed to meet requirements in the European Water Framework Directive (Leeper 2005). For soil, they follow the methodology for the European risk assessment for new and existing substances and biocides (European Commission 2003). Risk limits not covered by these references (e.g. NC and SRC), which are required to comply with Dutch environmental policy, are derived by Dutch procedures (Van Vlaardingen and Verbruggen 2007), (Janssen et al. 2004).

The Dutch guidance document (Van Vlaardingen and Verbruggen 2007) provides detailed guidance on the physical, chemical and toxicological parameters needed to derive ERLs, as well as procedures for determining the various ERLs for each of the compartments or media that may be impacted.

The MPC is defined as the concentration in a given environmental compartment that:

- has no adverse effect on ecosystems;
- has no adverse effect on humans (non-carcinogenic substances); and
- has no more than a probability of 10^{-6} per year of death (for carcinogenic substances).

Van Vlaardingen and Verbruggen (2007) describe the following procedure for determining the MPC_{eco} for organic compounds in soil:

1. When no toxicity data are available for soil organisms, the equilibrium partitioning method is applied.
2. When only one test result with soil dwelling organisms is available (earthworms or plants), the $MPC_{eco,\ soil}$ is calculated both on the basis of this result, using assessment (safety) factors, and by using the equilibrium partitioning method, with the $MPC_{eco,\ water}$ as input. The lowest value of the two is chosen as final $MPC_{eco,\ soil}$ value.
3. When toxicity data are available for a producer and/or a consumer and/or a decomposer, the $MPC_{eco,\ soil}$ is calculated using assessment factors (Table 6).
4. An $MPC_{eco,\ soil}$ is calculated on the basis of the lowest determined effect concentration (e.g. NOEC, EC10 or L(E)C50).
5. Calculation of an $MPC_{eco,\ soil}$ using statistical extrapolation techniques can be considered when sufficient data are available. The minimum data set to calculate a species sensitivity distribution should contain chronic toxicity data for at least 10 species from different taxonomic groups.

Table 6. Assessment factors applied to determine $MPC_{eco,\ soil}$.

Available test result	Assessment factor
L(E)C50 short-term toxicity test(s) (e.g. plants, earthworms, or micro organisms)	1000
NOEC for one long-term toxicity test (e.g. plants)	100
Two long-term NOECs from species representing two trophic levels	50
Long-term NOECs from at least three species representing three trophic levels	10
Species sensitivity distribution (SSD method)	5 to 1 (to be fully justified on a case by case basis)

For $MPC_{eco,\ soil}$ derivation for metals, the added risk approach taking into account background soil levels is followed. The maximum permissible addition (MPA_{eco}) is the contaminant amount that when added to background the contaminant level is the soil is MPC.

Following (European Commission 2003) guidance, the assessment of secondary contamination (biouptake) via the terrestrial food chain is triggered by the following compound properties:

- The compound has a log $K_{ow} \geq 3$, or
- The compound is highly adsorptive, or
- The compound belongs to a class of substances known to have a potential to accumulate in living organisms, or
- There are indications of bioaccumulation from structural features of the compound
- There is no mitigating property such as hydrolysis (half-life less than 12 hours)

Four different routes contributing to human exposure were incorporated: consumption of leafy crops, root crops, milk and meat. First, the concentration in the leaf, root, milk or meat is calculated as a 10% fraction of the TDI, taking into account the daily dietary intake of these products. The concentration in leaf, root, milk and meat are then each recalculated to a concentration in soil: $MPC_{human,\ soil,\ leaf}$, $MPC_{human,\ soil,\ root}$, $MPC_{human,\ soil,\ milk}$ and $MPC_{human,\ soil,\ meat}$. The lowest of the four values is selected and is the final $MPC_{human,\ soil}$.

Ecotoxicological ERLs for the groundwater compartment are derived based on ecotoxicological data for the surface-water compartment.

For non-ionic organic compounds, it is assumed that bioavailability is determined by organic matter content only. The Technical Guidance Document (TGD) (European Commission 2003), advises recalculating data from terrestrial toxicity experiments to the standard soil. Within the framework of the International and National Environmental Quality Standards for Substances in the Netherlands (INS) (Van Vlaardingen and Verbruggen 2007), this recalculation of results from individual tests (LC50s, EC50s, EC10s, NOECs) to Dutch standard soil and sediment is performed according to Equation 8 with the organic matter content (F_{om}) of Dutch standard soil and sediment:

$$TEST\ RESULT_{Dutch\ std\ soil} = TEST\ RESULT_{Experimental\ soil} \times \frac{Fom - Dutch\ std\ soil}{Fom - Experimental\ soil} \tag{8}$$

The TGD states the following with respect to normalization to standard soil:

"It should be noted that this recommended normalization is only appropriate when it can be assumed that the binding behavior of a non-ionic organic substance in question is predominantly driven by its log K_{ow}, and that organisms are exposed predominantly *via* pore water." However, no guidance is given for those compounds to which the above statement does not apply; e.g., ionizable organic compounds (Van Vlaardingen and Verbruggen 2007).

National Oceanic and Atmospheric Administration (NOAA) Screening Values. NOAA assembled from various sources a set of "Screening Quick Reference Tables (SQuiRTs) (Buchman 2008). These screening values were developed for use in preliminary evaluations of substances that may threaten natural resources of concern to NOAA. Screening values are presented for inorganic and organic contaminants in sediment, surface water, groundwater, and soil. Ground water values are based on USEPA primary and secondary MCLs for drinking water supplemented by values from Canada and the United Nations World Health Organization, surface water values are primarily from USEPA Ambient Water Quality Criteria, and sediment values are from multiple sources. Soil values include Dutch standards and the USEPA Eco-SSLs.

State Published Screening Values. A number of states have published risk-based screening values for soils, as well as other environmental compartments. Since most states are primarily responsible for enforcement of federal, as well as state, environmental laws, state values will often control where dredged material can be placed for beneficial use. The states have generally published screening values developed using USEPA approaches to meet their needs rather than deriving values from the literature based on risk-based testing procedures. However, states may adjust the values to be more or less conservative and to be more applicable to state environmental conditions or state policies and priorities. Brandon and Price (2007) reviewed state criteria and extracted the soil screening level concentrations from a number of states. The

Great Lakes Commission (2001) abstracted state criteria specifically — or potentially — applicable to beneficial uses of dredged material for the states that border on the Great Lakes. The California Center for Land Recycling (CCLR) surveyed state soil and groundwater cleanup levels to support the California EPA in developing screening values for its Brownfields program, and compiled a comprehensive data base for all states with promulgated levels. The cleanup level can vary widely. One example comparison among 43 state residential soil screening values for arsenic showed a variation in concentration range of almost five orders of magnitude *http://www.cclr.org/programs/policy*.

The ITRC (2005) surveyed and reviewed soil screening values for thirteen states to document differences in screening values, methods, and rationales used to derive those values. The focus of the comparison was on basis for the development of the criteria and how the screening values are utilized by the thirteen states. The paragraphs below highlight the ITRC findings.

The ITRC compared screening levels for the various states for five contaminants of interest: arsenic, lead, benzo(a)pyrene, polychlorinated biphenyls, and trichloroethylene. The values reported by the states for residential soil screening were mostly derived with a health-based approach. Even though several of the states used the USEPA Region 6 or Region 9 screening methods, they still differed in their reported screening values. Tennessee, for example, used its statewide background level for arsenic in residential and industrial soil rather than the much lower value derived from the USEPA equations. California based its levels on specific state code or guidance and conducted its own research for certain exposures. The screening values with the most variance were those for the protection of groundwater resources. For some, the drinking water MCL was used as the criterion after applying leaching and groundwater transport models; for others, a concentration was used that calculated for protection of human health using toxicology equations. Different states also used a different dilution attenuation factor (DAF) in the groundwater transport estimate in concert with DAF values of 1 and 20 presented in earlier versions of the Region 9 PRG table.

As described by ITRC (2005), states inconsistently applied the USEPA exposure values in the risk equations.

> Of the 13 states participating in the survey, five (California, Colorado, Florida, Kansas, and Michigan) develop their own residential soil screening levels. Two states (Arkansas and Oklahoma) use levels developed by EPA Region 6. Two other states (Nevada and South Carolina) use EPA Region 9 PRGs without modification. Kentucky uses most Region 9 PRGs values except for soil adherence, soil absorption, and values for ages 7 to 18 for site-specific evaluations. The remaining three states (Alabama, Georgia and Tennessee) modified the EPA Region 9 PRGs by dividing the PRGs for non-carcinogens by a factor of ten. This is functionally equivalent to setting the acceptable HQ to 0.1 instead of 1.0.

Those who developed their own screening values calculated different average daily dose values because of different assumptions for body weight for children, skin surface area for adults, and other factors. Published screening levels for a chemical can differ from state to state by several orders of magnitude and the reason for these differences is not always apparent (ITRC 2005).

The ITRC (2005) made the following recommendations to those developing risk-based screening values:

- Publish the basis of the development of each criterion.
- Make the underlying assumptions and values transparent.
- Publish the intended use and application along with screening values.
- Provide training and communication tools.

A brief summary of the basis for criteria from various states is presented in Table 7. Included in this table are notes pointing out features of state guidance that may be important to the development of screening values for beneficial use of dredged material.

Table 7. Features and sources for state soil screening levels. (continued)

State	State "look-up" tables				State guidance		Human health value source							Eco value source			
	Residential	Commercial	Industrial	Other	Refers to USEPA tables	No specific guidance	USEPA Regional Screening	USEPA RAGS/SSL (1996)	USEPA Region 3	USEPA Region 5	USEPA Region 6 HMSSLs	USEPA Region 9 PRGs	ORNL Screening Levels	USEPA Eco-SSLs	USEPA Region 4 Eco-SSLs	ORNL Screening Levels	Notes
Alabama	X	X										X		X	X		Small source <270 sq yds; large source 1 acre Surface soils 0-1 ft; subsurface 1 ft bgs to water table
Arkansas					X						X				X		No specific screening values provided — USEPA sites referenced
California	X	X	X									X					USEPA toxicity factors adjusted to be more stringent References San Francisco Bay (SFB) Regional Water Quality Board screening Commercial & industrial one land use
Colorado	X	X						X									SESOIL and ATD123D used to model groundwater
Florida	X	X						X				X					Groundwater based on equilibrium partitioning model
Georgia	X		X					X				X			X		Commercial & industrial one land use
Indiana	X		X				X	X									Residential and industrial "closure levels"
Kansas																	Risk based standards under revision Aug 2010
Kentucky	X		X									X					PRGs copied for values
Maryland	X			X				X	X								"Non-residential" land use values provided Values called "cleanup standards"
Michigan	X	X	X					X		X							SSG 1996 used for deriving soil values for groundwater contamination Values presented for 4 levels of commercial land use
New York	X			X													"Restricted residential," and "protection of ecological resources included for "soil cleanup objectives"

Table 7. (concluded).

New Jersey	X			X				X									"Non-residential" land use values provided Values called "remediation standards"
Nevada												X					
Ohio								X									
Oklahoma	X	X	X					X			X						USEPA SSL 1996
Oregon	X	X	X	X	X		X										Occupational, construction worker, and excavation worker scenarios addressed. State risk based criteria for petroleum sites supplemented with USEPA RSLs State terrestrial screening levels available
South Carolina				X					X			X			X		Risk-based screening levels for petroleum contaminants (UST Program) Addresses different soil textures Surface (0-3 ft) and sub surface (> 3 ft)
Tennessee												X			X		
Texas	X	X	X											X			Values for 0.5 acre source area and for 30-acre source area Ecological benchmarks described
Washington	X	X						X									Terrestrial ecological screening levels provided Cleanup levels and risk calculations (CLARC) data base

None of the states prescribed screening values for ecological receptors, although several referred to USEPA regional guidance to address risks to plants, invertebrates, or wildlife.

California Regional Water Quality Control Board—San Francisco Bay Region (CRWQCB--SFBR). This organization has published "Screening for Environmental Concerns at Sites with Contaminated Soil and Groundwater" (CRWQCB--SFBR 2008). The environmental screening levels for soil address protection for the following pathways: human health direct exposure, vapor intrusion into buildings, leaching and subsequent impacts to groundwater, terrestrial biota, and adverse nuisance conditions. A tiered approach for use of the screening levels in environmental risk assessments is recommended. The first tier compares site sampling data directly to the screening levels to determine if further investigation (Tier 2) is needed. The second tier reviews one or more components (chemicals, assumptions, pathways, risk levels, etc.) of the Tier 1 analysis to account for site-specific considerations. Where the Tier 2 analysis indicates potential risk based on the screening approach, a more traditional risk assessment is required as Tier 3. CRWQCB--SFBR emphasizes that the screening levels are not regulatory cleanup standards. Lookup tables are provided for the pathways listed above, for residential land use, for commercial/industrial land use only, and for four situations related to proximity to drinking water source. The drinking water categories are shallow soils overlying drinking water source, shallow soils not overlying drinking water source, deep soils overlying drinking water source, and deep soils not overlying drinking water source. Shallow soils are defined as less than or equal to 3 meters below ground surface, and deep soils are defined as greater than 3 meters below ground surface.

SUMMARY. Risk-based screening values for soils, sediment, and water environments have been a subject of considerable interest in the U.S., Canada, and Europe for more than twenty years. Most of the research and development has targeted criteria to be used for cleanup activities—either to determine if contaminated land site remediation is needed or if remediation goals have been met. Although beneficial use of dredged material should not be considered in the context of remediation, the scientific approach used to develop appropriate screening values for beneficial uses should not differ substantially from those used on contaminated soils. These remediation risk evaluations have been accepted by the scientific and regulatory communities, as well as the public. The primary difference is in the risk of exposure for various beneficial uses, which may range from residential scenarios that expose humans and ecological receptors to contaminants, to construction scenarios where the dredged material may be capped with clean fill and have essentially no exposure at the surface.

Some of the features found in the reviewed criteria that would be favorable are shown in the Table 8 below.

Table 8. Favorable features of various criteria.

Criteria	Risk based	Recognizes different pathways,	Accounts for different exposures	Incorporates both human and environment,	Considers bioavailability
USEPA Eco-Soil Screening Levels	x	x			x
USEPA Regional Screening Levels	x	x			x
CCME—Environmental	x	x	x	x	
CCME—Human Health	x	x	x	x	
The Netherlands Environmental Risk Limits	x	x	x	x	x
NOAA Screening Values (SQuiRTs)				x	
ORNL Eco Screening Benchmarks	x	x	x		x
California Regional Water Quality Control Board	x	x	x	x	

Development of screening values for beneficial use should consider the following factors:

- Existing risk-based soil screening levels developed for human health and ecological receptors can be adapted to beneficial uses
- Application of risk-based screening levels should follow consideration of the degree of contaminant exposure, which often differ from screening levels developed for generic use applications

- Bioavailability and geochemical characteristics of dredged material will require adjustment of screening values developed for soils with higher bioavailability and contaminant mobility
- Many beneficial uses may incorporate controls to reduce exposure of receptors to contaminants
- Many beneficial uses are set in an industrial location where exposure by ecological receptors and human access may be limited
- In some cases dredged material contaminant concentrations are reduced by dilution when mixed with other materials (manufactured soil)
- Published screening values are not available for all contaminants generally found in dredged material in the chemical forms that are typically analyzed, suggesting sediment-specific calculations may be required

Given these considerations, it is clear that criteria need to be developed specifically for beneficial use of dredged material. While the methodology should be basically the same as that used to generate other risk-based criteria, the values would differ based on different assumptions of exposure, toxicity and contaminant mobility. With widely varying exposure scenarios for different potential beneficial uses, it makes sense to develop different criteria for several use categories depending on exposure.

Scientifically defensible, risk-based criteria are needed to allow conservative and efficient decision processes for beneficial use of dredged material. Understanding existing criteria and the methods by which they are generated is a first step for development of criteria specifically for that purpose. Additional efforts are underway to develop a structured methodology for generating criteria specific to dredged material beneficial use applications. Generation of conservative screening values for varying exposure levels will allow rapid determination of the suitability of materials that clearly do not pose a significant threat to the environment while allowing further evaluation of materials with contaminants that pose a potential threat. The use of screening criteria will streamline the decision process while allowing the flexibility to fully explore the beneficial use potential of dredged material.

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