

Assessing and Managing Contaminated Sediments: Part II, Evaluating Risk and Monitoring Sediment Remedy Effectiveness

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EDITOR'S NOTE:

This is the second of a two-part review article on the assessment and management of contaminated sediments. Part I, "Developing an Effective Investigation and Risk Evaluation Strategy," appears in both the print and online editions of IEAM Volume (1), Number (1).

ABSTRACT

This is the second of a two-part review of the current state-of-the-science pertaining to the assessment and management of contaminated sediments. The goal of this review is to introduce some of the major technical and policy issues stemming from the assessment and management of contaminated sediments, highlight a number of aspects of contaminated sediment assessment and management found to be successful, and, when appropriate, address the barriers that still exist for improving contaminated sediment management. In Part I (Apitz et al. 2005), the key elements of an effective investigation and risk evaluation strategy were reviewed, beginning with the development of a conceptual site model (CSM) and including a discussion of some of the key factors influencing the design of sediment investigations and ecological risk assessment of sediment-bound chemicals on aquatic biota. In this paper, Part II, various approaches are reviewed for evaluating sediment risk and monitoring sediment remedy effectiveness. While many of the technical and policy issues described in this review are relevant to dredged material management, the focus of this paper is on sediment assessment for environmental management.

Keywords: Sediment assessment Sediment management Ecological risk assessment Sediment quality

INTRODUCTION

It is widely acknowledged in the scientific community that pollutants such as polychlorinated dibenzo-p-dioxins (PCDDs), furans (PCDFs), biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs), brominated flame retardants (BFRs), heavy metals, and pesticides are found worldwide in many rivers, lakes, estuaries, and coastal areas and originate from point or nonpoint sources dating back to the early industrial era of modern society or earlier (Long et al. 1995; Iannuzzi et al. 2002). Although in many cases these sources have been reduced in the last few decades, sediments containing these and other contaminants act as secondary sources of organic and metal contamination, posing significant direct and indirect

environmental risks through bioaccumulation in aquatic organisms and by incorporation into aquatic and upland food webs (Salomons et al. 1987). Episodic physical redistribution of contaminated sediments within dynamic waterways over time can disperse such environmental risks, potentially affecting biological and water quality conditions far from the original sediment source (Reible and Savant-Malhiet 1993). The accurate measurement of chemicals in sediment, assessment of the potential ecological and human health risks, and the design and implementation of appropriate remedial strategies to reduce or eliminate the potential risks have become the focus of environmental regulatory agencies in several countries (USEPA 1998a; Chapman and Wang 2001; Apitz and Power 2002; den Besten et al. 2003).

In this two-part paper, the current state-of-the-science pertaining to the assessment and management of contaminated

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Table 1. Key considerations associated with different approaches to environmental sampling

The confidence with which the assumption of a normal distribution for site data can be justified (which is required when using statistically based, randomized sampling approaches)
The level of knowledge of site-specific conditions and characteristics
The relative ease of design implementation
The ability to focus sampling efforts in more critical areas of the site
The range in contaminant concentrations that may occur across the site
Relative cost-effectiveness

sediments is reviewed, including the various sediment assessment approaches that support remedy design such as conceptual site model (CSM) development; contaminant distribution, fate, and behavior, including the use of novel screening tools; linking sediment chemistry with biology, including toxicological and bioaccumulation studies; assessing the natural recovery potential for contaminated sediments; and predicting and monitoring remedy effectiveness. In Part I, the many key elements of an effective investigation and risk evaluation strategy were reviewed, beginning with the development of a CSM through the evaluation of environmental fate and the factors influencing the effects of sediment-bound chemicals on aquatic biota (Apitz et al. 2005). In Part II of this paper, various approaches are reviewed for evaluating ecological risk and monitoring sediment remedy effectiveness. The goal of this review is to introduce some of the major technical and policy issues stemming from the assessment and management of contaminated sediments, highlight a number of aspects of contaminated sediment assessment and management found to be successful, and address the barriers that still exist for streamlining contaminated sediment management. While many of the technical and policy issues described in this review are relevant to dredged material management, the focus of this paper is on sediment assessment for environmental management.

ASSESSING CONTAMINANT DISTRIBUTION, FATE, AND BEHAVIOR

Most tiered assessment frameworks begin with the compilation of available environmental sampling data and an evaluation of chemical levels in sediments. If the concentrations of contaminants are below sediment quality guideline (SQG) values, then the management options are less stringent and more flexible because and the sediments are perceived to pose negligible environmental risks.

Several statistical designs and procedures can be employed to determine the array, number, and locations for sediment data collection at a site, including simple random sampling, systematic sampling, stratification, or composite sampling (USEPA 1985; Gilbert 1987; USEPA 1991b, 2000b). Another sampling approach, referred to as judgmental (or biased) sampling, may also be employed and is generally based on historical information for the site (including knowledge of historical source locations), visual inspection, and professional judgment (USEPA 1991b); however, judgmental sampling is not a statistically based (randomized) approach to sediment

characterization, and does not generally support defensible conclusions about spatial conditions beyond the immediate locations from which samples were collected and tested (USEPA 2000b).

The U.S. Environmental Protection Agency (U.S. EPA) reviews many of the benefits and limitations associated with the use of different approaches to sampling design and, in particular, the significance of the environmental variability inherent in each approach (Table 1). Some of the uncertainties associated with a particular sampling design may be reduced through the collection of large amounts of data, but this may be neither cost-effective nor fit within the time frame of the investigation. In contrast, fewer samples or less data may be needed to adequately characterize the spatial extent of sediment contamination, if field-validated sediment transport and dispersion models are used (USEPA 1985).

The application of hydrodynamic and sediment particle transport and dispersion models, when integrated with available site information, are useful in assisting in the development of sampling design plans, particularly when attempting to simulate (or retroactively characterize) temporal influences on the spatial extent of contaminant distribution. One such model is ECOMSED (http://www.hydroqual.com/ehst_sed_trans.html). A common criticism of models, however, is that they are often too complex (and data-intensive) for most aquatic investigations and, unless well planned, a truly field validated hydrodynamic or sediment transport model represents a major effort and expense. A review of sediment transport models and approaches can be found in Ziegler (1999).

When dealing with sampling design, one major concern is the heterogeneity of the distribution of contaminants in the sediment. Any prior information about site heterogeneity can aid in the selection of a sampling scheme, and one of the goals of a sampling design is to sufficiently characterize this heterogeneity. However, because analytical chemistry costs can be high, the number of samples to be taken may be limited. A number of strategies aid in filling gaps between standard analytical sampling points, including the use of acoustic methods and Rapid Sediment Characterization (RSC) tools.

It can generally be assumed that much of sediment contamination occurring at a site is associated with finer-grained materials such as clays and silts, rather than coarser-grained deposits (including sands or gravels). Despite many exceptions to this rule (Apitz 1998), sediments in contaminated or urbanized areas are often made up of mixtures of contaminated fine-grained material and less contaminated coarse-grained material. Using this assumption, acoustic profiling (USEPA 1994b; NRC 1997), including side-scan or multi-beam sonar techniques, can be employed to remotely characterize the spatial distribution and degree of heterogeneity of different sediment types, including the finer-grained sediments that are most likely to retain the contamination. These methods, however, may have difficulties characterizing sediments containing a mixture of grain sizes and cannot distinguish sediments deposited in different time periods (e.g., before or after urbanization or industrial discharges).

The tendency for contaminants to associate with fine-grained sediment deposits means that contaminants from different sources will often co-associate. This is not always the case, however, and sampling and analytical plans must

be designed to address the possibility that different reaches of a river or harbor may be contaminated with different contaminants or mixtures of contaminants, or associated with different sediment deposits.

The use of field screening or rapid sediment characterization tools

Field screening or rapid sediment characterization (RSC) technologies are mobile screening tools that provide measurements of chemical, biological, or physical parameters on a real-time or near real-time basis. When used appropriately, these tools can streamline many aspects of the investigation process. The tools can be used to delineate areas of concern, to fill in information gaps, and to ensure that expensive, certified analyses have the greatest possible utility.

Rapid sediment characterization is defined as the utilization of near real-time screening techniques and technologies to rapidly delineate the extent of contamination, physical characteristics, and biological effects (CMECC 1996; USEPA 1997a). The U.S. EPA has compiled an on-line Field Analytical Technologies Encyclopedia (FATE; <http://fate.clu-in.org>) that is intended to provide information about technologies that can be used in the field to characterize contaminated soil and groundwater, monitor the progress of remedial efforts, and, in some cases, to support confirmation sampling and analysis after remediation is completed. Although not all of the technologies currently available are applicable to sediment sites, several RSC tools have been tested and demonstrated in sediments (Giesy et al. 1990; Filkins 1992; Kirtay et al. 1998; USEPA 1998b; ASTM 1999).

Table 2 lists the key questions that must be asked and answered when determining whether or not RSC tools are appropriate to use to assess sediment contamination. From the answers to several questions asked before sampling begins and consideration of the advantages and disadvantages of different techniques, appropriate decisions can be made on how best to implement an RSC technology or suite of technologies suitable to support a sediment investigation and risk assessment.

Several RSC technologies have been evaluated in the field and show some promise for future application in sediment assessments. Perhaps the most widely recognized is x-ray fluorescence (XRF), which measures the fluorescence spectrum of x-rays emitted when metal atoms are excited by an x-ray source. The energy of emitted x-rays reveals the identity of the metals in the sample; the intensity of the emitted x-rays can be related to concentration (Swift 1995; USEPA 1998b). An XRF spectrometer can analyze a large number of metals at concentrations ranging from parts per million to percent levels, and encompassing all of the metals typically found in soils and sediments. Field portable XRF (FPXRF) instruments provide near real-time measurements with minimal sample handling, allowing for extensive, semiquantitative analysis. Several examples can be found in the literature in which FPXRF has been used for the analysis of soils and sediments (Skei et al. 1972; Stallard et al. 1995; Kirtay et al. 1998). FPXRF has been certified by the U.S. EPA as a field screening method for metals in soils (USEPA 1998b).

Another technology is ultraviolet fluorescence spectroscopy (UVF), which is based on the measurement of the fluorescence observed following UV excitation of organic

Table 2. Key questions for determining the usefulness of Rapid Sediment Characterization (RSC) tools to assess sediment contamination

What are the goals of the investigation?
What are the contaminants of concern?
Are the contaminants known?
What are the action limits?
What are the strengths and weaknesses of the analytical methods being considered?
Do instrument detection limits meet action limit requirements?

solvent extracts of sediment. In general, this method is used to measure fluorescent organics (especially PAHs), though some care must be taken to reduce interferences from naturally occurring organic compounds (e.g., humics). Several studies have used UVF to assess total PAH levels in various types of sediment (Hargrave and Phillips 1975; Filkins 1992; Owen et al. 1995).

Laser Particle Scattering instruments (Sequoia Scientific, Bellevue, WA, USA) are available for in situ submersible contaminant measurements and measurements conducted in the laboratory or aboard a boat (Sequoia Scientific 1999). The operating particle size range of these instruments is typically between 1.25 microns and 250 microns. Grain size measurements are made because contaminants generally are associated with the fine-grained particles (Förstner 1987), and also because these data are essential for evaluations of settling velocity and other issues of particle transport.

Two RSC aquatic bioassays include the QwikLite and QwikSed bioassays (San Diego, CA, USA), which measure the inhibition of light emitted by marine bioluminescent dinoflagellates (e.g., *Gonyaulax polyedra*) exposed to effluents, elutriates, or sediment porewater (ASTM 1999). The bioassays are capable of measuring a response within 24 h of test setup and can be conducted for a standard 4-d acute test or 7-d chronic test. A third test, the Microtox® bioassay (Newark, DE, USA), measures the inhibition of light emitted by a bioluminescent microorganism exposed to sediment extracts, porewater, and, in a few cases, bulk sediment. Several comparative studies have examined the results obtained from different bioassays and found significant differences in the interpretations of test results (Giesy et al. 1990).

Although not generally classified as an RSC tool, sediment profile imaging (SPI) technology has been shown to provide rapid in situ imaging of the sediment matrix directly below the sediment-water interface. Sediment profile imaging cameras provide images of the biologically active sediment horizon, providing insight into benthic community behavior, health, and physical and geochemical interaction with the sediment (Rhoads and Germano 1986). The advent of digital time-lapse photography (Solan and Kennedy 2002) may enhance this technology by providing both spatial and temporal information on the interactions between organisms and sediment, a primary concern in ecological risk assessment (ERA).

At present, the key concerns limiting the use of RSC tools in sediment investigations are that some technologies are nonspecific, provide only semiquantitative data, or are overly sensitive to matrix interferences (USEPA 1993). Because

of these limitations, the U.S. EPA often considers the data generated by RSC tools as not equivalent to those generated using standard methods; RSC data are typically classified as either “screening data with definitive confirmation” or “definitive data” (USEPA 1993).

Another concern regarding the use of RSC tools is that, because they are not subject to the same QA/QC protocols and rigors as more standard procedures, RSC tools are vulnerable to regulatory or legal scrutiny. While these concerns are not trivial, a growing number of case studies indicate that project managers, regulators, and the user community have accepted RSC data as a critical, though not a stand-alone, line of evidence in the analytical and decision-making process (CMECC 1996; USEPA 1997a). Confirmation sampling using more traditional investigation methods (e.g., for 10% of samples) will likely be necessary for several more years until greater confidence in the use of RSC results in sediment assessments becomes more commonplace.

Viollier et al. (2003) review several other emerging technologies that have the potential to revolutionize sediment investigations by providing real-time data on spatial and temporal variability in contaminant behavior and fate. These relatively new in situ tools measure, monitor, or image the benthic interface and include in situ microelectrodes that measure selected constituents at the millimeter scale; benthic flux chambers that measure contaminant, nutrient, metal, and oxygen flux across the sediment-water interface; two-dimensional oxygen optodes that provide images of oxygen in surface sediments over time; and seepage m that measure advective fluxes at the sediment-water interface.

Evaluating natural contaminant fate and transport processes

Natural fate and transport processes normally control the recovery of unremediated contaminated sediments, the effectiveness of in situ remedial processes, and the amount and fate of any residual contamination after disturbance of the sediment (Reible et al. 1991). The significance of natural processes is influenced heavily by site-specific characteristics. These characteristics must be adequately assessed prior to the selection, design, and optimization of any sediment management options and the assessment must reflect fully the aquatic environment in which contaminated sediment is found. The relative importance of release, transport, and fate processes differ significantly between lacustrine, riverine, estuarine, and coastal environments (Reible et al. 1991).

Martin et al. (1996) provide a comprehensive review of hydrodynamic processes affecting sediment fate and transport, while Reible et al. (2004) describe how to address these processes in models to support contaminated sediment management. Key factors are the energy of the overlying flow and whether the system is net erosional or depositional. Under high flow conditions, the bed sediment tends to be coarse-grained and noncohesive with little sorptive capacity and low depositional rates. Significant amounts of sediment and associated contaminants may be suspended in the water and the dynamics of the sediment may largely define the stability of the contaminants. Because most persistent sediment contaminants are associated with the solid phase, any mobilization of this phase dramatically increases contaminants mobility. Yet high suspended sediment loads do not necessarily mean that the bed is unstable because the source of the suspended sediments may be surface runoff or

simply transport from upstream. In low energy environments, contaminant fate and transport are not controlled by sediment erosion and resuspension because deposits are typically fine-grained, providing high sorptive capacity and significant slowing of advection and oxygen transport.

One of the most important processes within stable or depositional sediments is bioturbation, the mixing associated with the normal life-cycle activities of sediment-dwelling organisms (François et al. 2001). Many benthic organisms, especially deposit feeders, tend to prefer fine-grained sediments, enhancing uptake and bioturbation. In the absence of significant bioturbation, physicochemical and microbial processes largely govern the release or fate of contaminants from the bed sediment. Important physicochemical processes include advection, diffusion, and sorption and desorption. The high sorption capacity associated with fine-grained sediments may also increase the relative importance of boundary layer mass transfer processes in the water near the sediment-water interface. This occurs by a mechanism similar to the enhanced importance of water-side mass transfer processes in the evaporation of highly volatile compounds (i.e., compounds that strongly prefer movement to the atmosphere).

In the United States, it is estimated that approximately one-third of all Navy hazardous waste sites and many Army and Air Force landfills have coastal groundwater infiltrating the waste (Chadwick et al. 2003a, 2003b), and that 75% of all Resource Conservation and Recovery Act (RCRA) sites and Superfund sites are located within a half mile of a surface water body. Such water bodies can be of various types, including riparian, lacustrine, estuarine, and marine. A small body of work addresses such processes in freshwater systems (Palmer 1993; Stanford and Ward 1993; Hill 1997), but a growing body of evidence indicates that groundwater-surface water interactions (GSIs) in estuarine and marine systems represent an important, complex, yet often-neglected migration pathway for natural and anthropogenic constituents entering coastal waters (Millham and Howes 1994; Lendvay et al. 1998a, 1998b; Paulsen et al. 2001).

Little is known about the behavior and fate of contaminants (such as petroleum hydrocarbons) as they move downstream in soil and groundwater systems and ultimately mix with saltwater and sediment systems. As a result, at a number of coastal sites in the United States, conservative soil cleanup criteria (lowered by orders of magnitude to address uncertainty) have been imposed to be protective of putatively downstream sediment benthic biota. In the case, for example, of soil cleanup levels developed for the region around San Francisco International Airport in the 1990s, total petroleum hydrocarbon cleanup levels were developed that were below background levels.

At present, few studies integrate the relationship between groundwater-borne contaminants and estuarine ecological risk. While natural attenuation is often invoked as a rationale for ignoring surface water effects, few have quantitatively assessed this attenuation or the processes underlying the attenuation (Lee 2000). Recognizing this issue, federal and state regulatory agencies in the United States are beginning to stress requirements that include the consideration of the transition zone at sites where contaminated groundwater is suspected to contribute to contaminant loading in surface waters (Duncan et al. 2000). Almost no scientific basis exists upon which to design an approach to such considerations, and fundamental knowledge of the processes that regulate

the fate and effects of groundwater-borne contaminants in aquatic environments needs to be developed. A method of accounting for contaminant flow, sorption, and weathering is also needed, as contaminants move toward the hyporrheic zone and mix with sediments. How this affects bioavailability and, ultimately, the derivation of sufficiently protective biological criteria, needs to be addressed.

Linking sediment chemistry with biological effects

Contaminant bioavailability is controlled by the relationship between the concentration of a chemical in an environmental matrix and its accessibility for incorporation into biological receptors (Alexander 2000). Reduced bioavailability via chemical sequestration plays an important role in soil and sediments systems in which the toxicity of a chemical in aqueous or gaseous phases has been shown to be mitigated once the chemicals come in contact with soil or sediments (Hrudey et al. 1996). Because bioavailability depends on the chemical and the environmental matrix, sediments of different compositions and histories, but the same bulk contaminant concentrations, may have very different toxicities.

Furthermore, a number of factors may change contaminant bioavailability over time. Evidence suggests that long-term sequestration can be altered by changes in the physical-chemical-biological environment of sediments systems (Ghosh et al. 2001). Changes in redox state, salinity, and organic matter diagenesis have been shown to affect contaminant binding and dissolution (NRC 2003). Because contaminants immobilized in soil and sediment systems may be released back to the aqueous environment, one of the key issues concerns the long-term stability of the nonbioavailable fraction, and the rate of transfer between available and nonbioavailable fractions.

Although the concepts of bioavailability are well established (Mark 1995; Kelsey and Alexander 1997; Reible et al. 1999), formally and consistently incorporating these principles into risk assessments and into regulatory frameworks have proven problematic (NRC 2003). Until the complex mechanisms influencing contaminant bioavailability are better understood, bioavailability assays should only be one line of evidence in sediment assessments, and should be supplemented by direct measures of biological impact.

If contaminants were uniformly bioavailable and uniformly toxic, regardless of exposure scenario, then contaminated sediments could be regulated, ranked, and managed based on bulk sediment concentrations using SQGs in a manner similar to many water management programs. However, a variety of factors mitigate (or enhance) contaminant availability and toxicity in complex sediment systems. Thus, sediments are evaluated based not only on bulk concentrations (which reflect only potential exposure), but also on a number of tools that assess bioavailability (which reflect actual exposure), bioaccumulation, and biological effects (e.g., toxicity and community impacts). Sediment toxicity tests, benthic macroinvertebrate community analysis, and histopathology typically are used to evaluate potential biological effects. Biomarkers are another set of tools that are often employed for measuring exposure, although some tools are also used to hypothesize direct adverse effects. A detailed discussion of the linkage between bioaccumulation and sediment quality is provided by the U.S. EPA (2000a, 2000c).

Tissue chemistry measurements can directly assess bioavailability and integrate exposure over time from the water,

sediment, and food web pathways. However, an uncertain relationship exists between tissue residues and bioeffects and metabolism or bioregulation may result in no linkage between exposure concentrations and tissue residues (Burmester et al. 1991). In addition, much variability often exists between species and individuals of the same species. Furthermore, it is difficult to determine exposure for mobile species. Despite these shortcomings, databases have been developed to help evaluate tissue residue values and to predict potential effects (Jarvinen and Ankley 1999; see e.g., <http://www.wes.army.mil/el/ered/index.html>). The U.S. EPA (2000c) has published a lengthy report that associates the presence and quantity of potentially bioaccumulative chemicals in sediment with uptake in the tissues of aquatic and terrestrial organisms, and with the effects of those chemicals on the organisms.

Histopathological indices have been useful in assessing the toxicological effects of contaminated sediments (Burton 1992). Effects can be measured directly and quantified in terms of frequency of incidence (Ingersoll et al. 1997). However, histopathological indices may be influenced by factors other than chemical contaminants.

Biological testing of sediments

Bioassay, defined as the use of biological media (whole organisms, cells, etc.) to measure contaminant concentration, potential biological response in environmental samples, or both, has become an integral component of tiered environmental assessments. Several procedures have been standardized for the evaluation of whole sediment effects (USEPA 1994a, 1994c, 1994d; ASTM 2001a). In general, the majority of these tests focus on (acute) lethality in whole organisms (e.g., typically benthic infaunal species) following short-term or acute exposures (<14 d). More recently developed protocols focus on the measurement of (chronic) sublethal responses (e.g., reduced growth or reproduction [or both] following longer-term exposures). Both acute and chronic tests provide useful measures of potential toxicity to benthic biota. The selection of appropriate procedures for a specific application depends on the questions being addressed, the nature of the environment and sample matrix, the nature of the contaminant or contaminants of concern, and the behavior of the test organisms. Evaluating the toxicity of in-place sediments should be approached differently from examining the potential effects of sediments that are to be removed, relocated, or treated as part of a remedial action.

Solid-phase test procedures have a broad range of applicability to sediment management and remediation activities. Such tests are most commonly used to help delineate the extent and magnitude of contamination in the initial assessment and in post-remedial monitoring. In the initial assessment, toxicity test results, in combination with bulk sediment chemistry, are used to delineate the real extent of bioavailable contamination. In addition, the application of more specialized techniques such as toxicity identification evaluations (TIEs) can be used to help identify the contaminants or contaminant classes most likely responsible for toxicity at the site or to exclude potentially confounding factors (e.g., ammonia) (USEPA 1991a). It should be noted, however, that TIE procedures for whole sediments are not fully developed nor routinely applied at this time. Sediment toxicity tests may be used to assess comparative risks associated with various management alternatives (capping, natural attenuation, etc.). They can also be used to establish

the efficacy of different proposed remediation technologies (e.g., soil and sediment washing, biotreatment, etc.). Because toxicities of many of the intermediate products are not known, whole sediment tests may provide a useful tool for the integration of potential effects of intermediates formed by a specific treatment process. They can also reveal effects of a remedial strategy on nontarget contaminants; for example, whether treatments may mobilize previously unavailable nontarget compounds. However, the ecological relevance of such an application would be limited unless the remediated material is to be returned to an aquatic environment, or if in situ treatment were being considered.

Sediment bioassays that test the effects of exposure to sediment porewater are increasingly advocated to evaluate the potential for toxicity of some chemicals in sediment. Porewater is generally extracted via centrifugation of the sediment sample. However, other methods such as squeezing a sediment sample by applying positive pressure with an inert gas in a specially designed chamber, or passive diffusion through a semipermeable membrane device placed in the sediments also have been used to collect sediment porewater (Ankley and Schubauer-Berigan 1994). Recently, in situ sampling of porewaters using specialized probes has been demonstrated (Chadwick et al. 1999).

Interest in the evaluation of sediment porewater as a test matrix stems from the belief that porewater chemistry is more representative of the bioavailable fraction of sediment-associated contaminants than is bulk sediment chemistry. However, the validity of such an assumption is highly dependent on the organism being examined and how it interacts with sediments (Chapman et al. 2002). Currently, only a small number of sediment porewater tests have been developed, though in theory any aquatic organism could be used to evaluate sediment porewater toxicity. The small volume of sample that is practicably obtained through conventional porewater extraction methods limits the application of porewater tests to species life-history stages that can be evaluated in small volumes (e.g., <50 ml).

Additionally, the exposure of epibenthic or pelagic species at different life-history stages to sediment porewater is generally viewed as providing little relevant ecological information; such organisms may express heightened sensitivity to sediment-associated factors other than contaminants, thus being poor surrogates for those aquatic species that might directly contact porewater (Chapman et al. 2002). For example, evidence suggests that sea urchin (*Strongylocentrotus purpuratus*) embryos lack the cellular processes found in other invertebrates that serve as protective mechanisms to eliminate organic chemicals from tissues (Toomey and Epel 1993; Galgani et al. 1996). This would explain the high degree of sensitivity often reported in the sea urchin porewater bioassay in comparison to other bioassays.

In general, sediment porewater tests have a high propensity to be affected by factors other than sediment-associated contaminants (Chapman et al. 2002). Consequently, porewater tests are perhaps best suited as screening tools in initial assessments. However, because TIEs are better developed and more easily performed on water samples, porewater is a useful matrix for conducting sediment investigations. Results of porewater TIEs can be used in conjunction with other lines of evidence (chemical analysis, site history information, etc.) to identify the contaminants of concern at a site and help focus selection of the best remedial alternatives.

Sediment extract tests such as the Microtox bioassay fall under a special class of test procedures that evaluate extracts prepared using either water or an organic solvent containing the sediment-associated contaminants. Many of the procedures for assaying genotoxic potential of sediment-associated contaminants, such as the Ames, Mutatox, H4IIE, and the P450RGS bioassays, rely on similar methods of extraction for testing (Inouye 1999). The most common criticism of these procedures is that bioavailability is confounded by the extraction method, which may substantially alter the biological responses of test organisms relative to in situ conditions.

Sediment–water interface (SWI) tests are designed to evaluate the effects on epibenthic species as a result of contaminant flux from the sediment surface to overlying waters (Anderson et al. 1996). The tests involve the collection of intact sediment cores and the placement of test organisms in a screened tube that rests on the surface of the sediment core. The screen prevents the animals from directly contacting the sediment, but permits exposure to contaminant flux from the sediment surface. Originally developed to assess the potential effects of contaminant flux on echinoid development (specifically the sea urchin), the same procedure can be applied to other epibenthic species (Anderson et al. 1996). The approach represents a reasonable approximation of actual exposure conditions in the field, assuming quiescent conditions. A common concern, however, is that a sediment core in a laboratory is subject to diffusional fluxes, but not advective fluxes from processes such as groundwater flow, wave pumping, and bioirrigation that dominate transport in situ (Apitz and Chadwick 1999; Apitz et al. 2003). The potentially confounding influences of factors other than sediment associated contaminants affecting overlying water quality (e.g., salinity, pH, ammonia, and sulfide) are of great concern as well.

Sediment elutriate tests were originally designed to assess potential water column effects associated with suspended sediments and dewatering discharges related to dredging and disposal activities (USEPA/USACE 1991, 1998). Since these types of activities are of generally short duration, nearly all of these tests have focused on effects in pelagic species associated with short-term or acute exposures.

Two general types of endpoints exist for these tests: standard lethality of whole organisms and effects on larval development. As with whole sediment tests, the selection of the most appropriate type of test for a particular application depends on the questions being addressed, the nature of the environment and sample matrix, and the nature of the contaminant or contaminants of potential concern. For example, such tests are probably not useful for assessing potential effects of whole sediment on infaunal invertebrates. Elutriate tests are better suited for the evaluation of potential water column effects during removal and disposal, or during other natural or anthropogenic short-term resuspension events.

According to Word et al. (2004) and others, the two principal confounding factors in elutriate tests are ammonia and total organic carbon (TOC). Most of the pelagic species and nektonic life-history stages commonly employed in sediment elutriate tests never encounter ammonia levels commonly associated with sediment matrices. Consequently, they tend to be more sensitive than benthic organisms to ammonia toxicity. Because ammonia is a relatively refractory constituent and generally

not considered a persistent contaminant of potential concern at sediment sites, it is important to distinguish between effects induced by ammonia and effects possibly induced by the contaminants of potential concern.

High levels of TOC have also been shown to potentially interfere with interpretation of elutriate tests (Bridges et al. 1996). With the exception of larval mussels and echinoderms, which are not fed during the course of exposure, TOC can represent a potential food source for species requiring exogenous sources of food. Thus, while in some cases it is possible to observe effects on organisms exposed to elutriates with both low TOC and low contaminant levels, in other cases it is possible not to observe effects in organisms exposed to elutriates with both high TOC and high contaminant levels, possibly due to interferences. By measuring these parameters and running appropriate controls, one can account for the influence of these potentially confounding factors on test results.

The American Society for Testing and Materials (ASTM 2001b) provides guidance for the selection of resident species as test organisms. While the use of standardized test procedures and test species offers consistency and permits comparison across sites over time, the use of indigenous test organisms can provide useful information about potential effects to resident species.

In situ tests are increasingly used as an alternative to laboratory bioassays to assess environmental effects (Rice and White 1987; Ireland et al. 1996). Methods for conducting in situ exposures are determined largely by the site characteristics and species being evaluated, and test selection should be carefully linked to both data quality objectives (DQOs) and a conceptual site model.

Bioaccumulation testing

Bioaccumulation of contaminants may result in adverse effects to resident species; models to predict such uptake from nonpolar organic chemicals and mercury from sediments have been developed, and should be applied in a tiered framework when bioaccumulative chemicals are present at levels of potential concern. The use of biota-sediment accumulation factors (BSAF) has been proposed as a means to estimate risk and, in some cases, to derive cleanup levels in sediment (Tracey and Hansen 1996; Burkhard 2003; NRC 2003). The BSAF is defined by U.S. EPA (USEPA 1994b) as the ratio of the chemical concentration in the organism on a lipid-normalized basis to the chemical concentration in the sediment on an organic-carbon basis; hence the units are grams of organic carbon per gram of lipid. In theory, sediment cleanup targets can be determined for chemicals by dividing a maximum allowable tissue level (MATL) by the BSAF. However, this assumes that: (1) the MATL is known, (2) the BSAF values are defined by accepted measures (either site-specific or literature values), and (3) a constant relationship (i.e., steady-state) exists between the exposed organism and the sediment. According to Wong et al. (2001), the variability in BSAF estimates among fish and benthic biota can be large, which likely limits its utility even under in situ riverine conditions as a first-level screening tool for predicting bioaccumulation. Other considerations such as aquatic fate and effect processes and trophic transfers of contaminants are typically not considered in the application of most simulation models, including the BSAF approach (Koelmans et al. 2001). The U.S. Army Corps of Engineers have used

BSAFs to develop a theoretical bioaccumulation potential to estimate the equilibrium concentration of a chemical in the tissues of an organism exposed to contaminated sediment. The BSAF approach can be linked with a simple food web model to further evaluate the effects of sediment-bound chemicals on aquatic organisms (Thomann and Komlos 1999).

An alternative approach to BSAFs are the more advanced mechanistic food web models first developed by Thomann et al. (1992) and Gobas (1993) that predict bioaccumulation of hydrophobic organic chemicals in aquatic organisms. These models incorporate assumptions regarding the bioaccumulation process, food web structure, bioenergetics, and toxicokinetics, and strive to minimize the uncertainties in the model results. According to Burkhard (1998), the differences between the two models stem from methods linking aqueous and sediment chemistry to concentrations in lower food chain organisms.

Aside from laboratory protocols for measuring the bioaccumulation of sediment-associated contaminants such as the 28-d bioaccumulation protocol for the freshwater oligochaete, *Lumbriculus* (USEPA 2000c), several in situ techniques have been used to evaluate the bioavailability of sediment-associated contaminants, specifically caged biota studies and semipermeable membrane devices (SPMDs). In general, these techniques are intended to provide a relatively short-term, reproducible indicator of contaminant bioavailability.

Mussels or clams are frequently used in caged studies in both freshwater and marine environments because they are ubiquitous and sedentary and tend to concentrate contaminants to levels that are orders of magnitude greater than environmental concentrations (Green et al. 1980). In general, the exposure period used in caged studies must be sufficient for organisms to attain steady-state or equilibrium with respect to contaminant uptake, but short enough to reduce the chances of mortality due to disease or lack of food. For example, Jones and Sloan (1989) reported data that suggested a 4- to 6-week exposure period was sufficient to achieve steady-state in fathead minnows exposed to PCBs in the Hudson River. Rice and White (1987) reported uptake maxima occurring within 20 to 30 d in fathead minnows and 9 d in fingernail clams exposed to PCBs. Curry (1977) proposed that caged mussels (*Elliptio complanata*) could be used to detect trace organic chemicals in water after 4 to 6 weeks of exposure.

An SPMD is a passive sampling device designed to mimic the uptake of organic chemicals from water by aquatic organisms. Typically, SPMDs consist of polyethylene tubing or membranes filled with a solvent (e.g., hexane, triolein) and mounted on a frame suspended in the water column. Hydrophobic organic chemicals passively diffuse into the solvent over periods of time similar to, or less than, those used in caged biota studies. The use of SPMDs is especially valuable for detecting chemicals present at low or variable concentrations in the water column. Similar devices have been designed for inorganic contaminants (Davison and Zhang 1994). A significant drawback, however, is the relative lack of calibration data to relate SPMD data to environmental concentrations (Meadows et al. 1998). Most importantly, however, SPMDs do not address potentially significant processes affecting bioaccumulation in organisms, most notably uptake via feeding and metabolism and elimination.

Establishing reference areas

In benthic macroinvertebrate community assessments, generally the third leg of the sediment quality triad introduced by Chapman (1990) along with sediment chemistry and sediment toxicity, effects are judged when comparing the community of organisms that inhabit a contaminated sediment site to those that inhabit an uncontaminated reference location. Because most contaminated sediment sites are evaluated after the initial chemical impact has occurred, *in situ* biological assemblages may be quite different than those that initially inhabited the site (Tannenbaum 2002, 2003). Therefore, it can be difficult to establish a link, if any, between the presence of contaminants and ecosystem conditions. For this reason, most sediment decision frameworks require multiple lines of evidence to establish the need for action.

In a triad-based sediment decision framework (Chapman 1996), contaminant levels are compared to one or more chemical benchmarks based either on literature values or site-specific conditions; sediment toxicity is compared to selected controls or reference materials, and sediment benthic community structure is compared to those in a reference site. Other triad frameworks have been proposed that focus on site-specific bioavailability, bioaccumulation, and other parameters (Chapman et al. 1997), but the three lines of evidence described above are the most common components of a triad analysis. While sediment chemistry and toxicity values are often (but not always) compared to standard conditions, benthic community structure is quite sensitive to a number of natural and anthropogenic conditions, so this critical leg of the triad should be based on the careful selection of a reference site for comparison (Chapman et al. 1997).

The selection of an appropriate reference area should be based on a consideration of several factors (USEPA 1994b). For example, contaminant behavior and bioavailability and benthic community structure can be affected by sediment physical characteristics such as grain size and mineralogy; thus, the physical nature of sediments should be considered in the reference area. Total organic carbon, salinity, acid volatile sulfides (AVS), oxidation-reduction potential and gradients, and pH also are important considerations when selecting a reference area because these factors can affect both contaminant behavior and benthic community structure. These parameters are relatively simple to assess, and provide important information that can be utilized in the decision-making process later on. Because sediments sites may exhibit a range of characteristics, matching all physical, chemical, and biological parameters to those at a reference area can be difficult; in some cases, a single reference site may not be sufficient to perform an adequate assessment.

Hydrodynamic considerations also are important, especially for site and reference areas located in river and stream environments. It would, for example, be inappropriate to compare sediment or community data collected from high-flow river systems to those in more flow-restricted lakes or ponds (Wenning and Ingersoll 2002). In estuarine and marine systems, it is important to compare erosional or depositional areas with reference sites of similar sedimentation regimes. An area where sediments can be scoured and transported would not offer the same type of physical bottom characteristics, contaminants, or biological features as one where such flows were restricted.

Different strategies have been proposed to address situations in which matching site and reference areas is not possible. One approach is to select a set of reference sites that emphasize different physical or biological characteristics. For example, Stronkhorst (2003) has proposed two sets of reference areas to evaluate sediments in Dutch Harbors; one set of reference areas is based on fine-grained sediments and the other set is based on coarse-grained sediments. A similar approach has been used in risk assessments of sediments in San Diego Bay, California, USA (K. Richter, SPAWAR Systems Center, San Diego, CA, USA, personal communication).

Often, statistical comparisons are made between study and reference conditions, with statistically significant differences between reference and study sites signifying “pass” or “fail” of chemical, toxicity, or benthic community criteria. Germano (1999), however, warns against the ritual of null hypothesis significance testing (NHST) with mechanical dichotomous decisions around a sacred 0.05 criterion, suggesting, instead, that it is important to ensure that the mechanisms driving ecosystem health are well understood.

Ultimately, the parameters measured at sites of concern and reference sites, how they are to be compared, and the way the data are to be used in a decision are more important than a perfect reference site, and their selection should ultimately be driven by the characteristics of the sites available. For example, in San Francisco Bay, California, USA, sediment chemistry has been compared to both regional reference values and literature-based SQGs. Because in this region the estuarine sediments are strongly affected by salinity and introduced species, a relative benthic index (RBI) also has been developed to evaluate pollution tolerance using three opportunistic species that thrive under polluted conditions and three sensitive species that thrive under pristine conditions. Sites in the bay were characterized with a weight-of-evidence approach that used threshold values for each of the triad metrics (Hunt et al. 2001). Other approaches to interpretation of lines of evidence include detailed decision matrices (Chapman 1996), some of which can have weighting factors based on multiple factors, including uncertainty, the relevance of measures to the endpoints of concern, and various aspects of study design including reference conditions (e.g., Johnston et al. 2002).

Assessing the potential for natural recovery

The pertinent natural processes most often associated with the consideration of the potential for natural recovery of water bodies affected by contaminated sediments include sediment deposition (burial), dispersion, mixing, irreversible adsorption, and chemical and biological reactions. When used in the context of a sediment remedy alternative in the United States, the term “monitored natural recovery” (MNR) is often used. MNR differs from “No Further Action” (NFA) because assessment, modeling, and long-term monitoring are required to verify that recovery is taking place, whereas the selection of NFA is generally based on the assumption that the potential risks are so low that none of these tasks are necessary.

The use of MNR as a protective alternative is based on the assumption that the risk posed by a contaminant is closely associated with its spatial and temporal proximity to receptors, and that natural processes can function to eliminate or limit that proximity (NRC 1997). In most cases (barring degradation) contaminants remain in the environment, albeit

sequestered from the biota. While this can be unsettling for some stakeholders, many other remedial strategies (barring those which achieve destruction of contaminants) also work by isolating or removing contaminants from the food chain, and often by removing them to other environments (e.g., confined disposal facilities, landfills, etc.).

The application of MNR as a remedial alternative for sediments is not practiced as often as monitored natural attenuation (MNA) at terrestrial groundwater sites (USEPA 1997b; SAB 2000). The selection of MNR is largely based on the identification of in situ factors controlling the bioavailable fraction of the chemicals to the sensitive receptor groups. Sensitive receptor groups and the natural processes that either ameliorate or exacerbate their exposure to chemicals are specific to both the contaminant and the site. Consequently, the success of MNR as a remedial strategy is dependent on an accurate understanding of the contaminated environment and the site-specific conditions that control the fate and mobility of the chemicals in that environment.

In the United States and elsewhere, the majority of contaminated sediment sites occur in shallow, coastal areas, and are much more likely than offshore environments to be affected by advective processes such as groundwater flow, tidal and wave pumping, and by resuspension via ship and storm activity. While these processes are recognized in the oceanographic community as having significant effects on chemical fluxes (Moore 1999), they are largely unstudied in contaminated systems. The relative magnitudes of these processes, as compared to the traditionally assessed processes such as diffusion and bioturbation, have not been determined. These issues must be addressed for near-shore sediments. If contaminated sediments are to be left in place, it is critical to evaluate potential pathways by which contaminants might pose an ecological or human health risk, and to monitor, minimize, or eliminate these pathways. On the other hand, the relative importance of these pathways as mechanisms of sediment recovery must also be determined (Apitz et al. 2003).

A number of interacting physical, chemical, and biological processes contribute to natural recovery in sediments. Burial and sequestration of the contaminated sediments often reduces chemical bioavailability. For instance, in Lavaca Bay, Texas, USA, total mercury extends deeper into the sediment than methylmercury, suggesting that the buried total mercury is less available for mobilization through microbial methylation than mercury found closer to the sediment-water interface (Gill et al. 1999). Additionally, Frazier et al. (2000) suggest that gradual burial of mercury-contaminated sediments in Fairhaven Bay and the Sudbury River reservoirs located in Massachusetts, USA, has reduced the amount of mercury available for methylation in both sites. If sedimentation is anticipated to be an important aspect of MNR, then it will be important to determine if the aquatic environment is capable of providing a source of clean sediments adequate to cover the contaminated sediment layer. Hydraulic characteristics of the environment that limit the resuspension and transport of sediment will also contribute to minimizing chemical mobility in the environment.

Significant research is under way to evaluate the "irreversible" sorption of contaminants on sediments (Huang and Weber Jr 1997; Kan et al. 1998; Chefetz et al. 2000). It has been demonstrated by a number of methods that not all contaminants in sediment are easily leachable, degradable,

or bioavailable. Partitioning processes often result in the sequestration of chemicals in inaccessible microsites or as residues covalently coupled to the organic fraction of the sediment particles. This binding often reduces the overall bioavailability, toxicity, and bioaccumulation potential of the contaminant. For example, chloroanilines have been shown to react and bind with sediments as demonstrated by their resistance to subsequent extraction with solvents. These strong interactions are likely the outcome of covalent bonding between the amino groups and the oxygen-containing group of the humic fraction of the sediment (Beyerle-Pfnur and Lay 1990). What is not clearly understood is whether this binding limits bioavailability for all benthic organisms.

The surface characteristics, redox potential, pH, and chemistry of the sediments are also important determinants of chemical speciation and will influence the partitioning of contaminants between the solid and aqueous phases of the sediment (NRC 2003). For example, the toxicity of metals is often related to their interstitial porewater concentrations rather than total mass in sediments. In many sediments, the concentration of acid-volatile sulfides (AVS) is a key factor in determining metal concentrations in interstitial water because AVS-bound metals form insoluble sulfide complexes that minimize biological activity (Ankley et al. 1996). The stability of these sulfide complexes is generally contingent upon the reductive capacity of the sediment (vandenHoop et al. 1997); aeration due to mixing and resuspension, bioturbation, or bioirrigation may affect metal availability. Vollier et al. (2003) review several assessment tools available for examining sediments in situ.

Microbial processes have been shown to support the goals of MNR in sediments (e.g., Atlas et al. 1981). For example, in situ biological degradation of hexachlorobenzene (HCB) has been shown to occur over a 16-year time period in anaerobic sediments (Beurskens et al. 1993). The reported half-life for HCB was determined to be approximately 7 years and HCB degradation to di- and trichlorobenzenes was shown to be biologically mediated (Beurskens et al. 1993). Natural degradative processes have been shown to play an important role in the recovery of estuarine and marine sediments contaminated with DDT (1,1,1-trichloro-2,2-bis[*p*-chloro-phenyl]ethane) and DDE (1,1-dichloro-2,2-bis[*p*-chlorophenyl]ethylene), a degradation product of DDT (Quensen et al. 1998). Microbial degradation of many PAHs and aliphatic and aromatic hydrocarbons in marine and freshwater sediments has been found in both laboratory and field studies under aerobic (Atlas et al. 1981) and anaerobic (Coates et al. 1997; Zhang and Young 1997) conditions. The methods for extrapolating measured degradation rates, either in the laboratory or the field, to predictions of absolute rates of in situ recovery of contaminated sediments must still be developed and validated.

Site-specific characteristics that inhibit MNR include insufficient deposition of clean sediments and a potentially high-energy environment in which deposited sediments have the potential to be resuspended and transported (NRC 1997). Sediments with low organic content may not bind contaminants, leaving them available for diffusion to the water column (NRC 2003). Microorganisms may dissolve solid phase minerals that can be responsible for precipitating and sequestering contaminants (vandenHoop et al. 1997). Microbial processes may transform contaminants into more soluble or bioavailable forms (Jernelöv 1970). For example,

Table 3. Key factors to consider during the selection, design, and optimization of sediment remediation technologies

Ultimate effectiveness of the technology with contaminated sediments
Potential risks associated with application of the technology or resulting from the residual contamination left by the technology
Methods for measuring and evaluating the performance of the technology when applied to contaminated sediments
Economics of the technology when applied to contaminated sediments, including the effects of scale and the balance between capital and operating costs
Uncertainties in application of the technology or in evaluating its potential risks and benefits

the methylation of inorganic mercury in surficial sediments to produce the more toxic and bioaccumulative methylmercury is an example of a microbial process that does not support MNR goals (Jernelöv 1970).

Predicting and monitoring remedy effectiveness

Table 3 lists several key factors to consider during the selection, design, and optimization of sediment remediation technologies. Even before a remedy is selected and implemented, it becomes crucial to design a post-remedy plan to monitor whether the remedy is operating as expected. The monitoring plan should identify specific measurements and describe how those measurements will be interpreted regarding proper performance or remedy ineffectiveness. Unfortunately, many of these factors are difficult to measure. More importantly, however, are the lack of standardized methods to measure and evaluate the performance of remediation technologies and a frequent lack of adequate baseline monitoring before and during remedy implementation.

The primary goal of the application of sediment management or treatment technologies is the reduction of human health and ecological risks (USEPA 2002). The ultimate goal of any remedial monitoring program is to identify the success of the remedy at protecting or restoring the resource and protecting human and ecological receptors (NRC 1997; Swindoll et al. 2000; USEPA 2002). Because the ultimate goal may require many years to achieve, interim goals often are required to evaluate the success or failure of the sediment management approach. Thus, the monitoring program must be directed toward the measurement of these interim goals as well as the ultimate goal. In addition, monitoring programs should include efforts to evaluate the effectiveness of the remedy implementation. Monitoring remedial effectiveness involves three activities: (1) monitoring to assess effectiveness of remedial action in achieving ultimate goal (i.e., protection or recovery of the resource at risk); (2) identification of interim goals and monitoring to evaluate the effectiveness of the remedial action; and (3) monitoring implementation of the remedial action to evaluate effectiveness of meeting both engineering and environmental protection goals.

Because of the difficulty and time required to define success in restoring or protecting the primary resource at risk, interim goals are needed to provide more immediate feedback on the progress of the remediation or management

strategy. Interim goals are often based on relatively short-term chemical measurements and the results of bioassay tests. Remediation or management technologies generally seek to reduce risk by reducing contaminants that are or may be introduced to the biologically active zone, including the upper layers of sediment (which may change with time or as a result of storm events), the water column, and the air above and surrounding the water body. Monitoring of physical, chemical, and biological parameters within the biologically active zone often serves to evaluate effectiveness in meeting interim goals. Examples may include chemical concentration measurements in the water column or in the upper layers of sediment, or density and diversity of benthic organisms. Contaminant uptake in species that respond rapidly to sediment contamination can also be useful as an interim measure. The dynamics of the response of organisms high in the food web, such as fish or birds, may be too slow for these ecological endpoints to be used as interim measures.

A growing body of evidence suggests that sediment removal can at times result in more ecological damage, or, after great expense, not show measurable ecological improvement (Thibideaux et al. 1999). Because of volumes and costs involved, it seems clear that some sediment sites will be managed in place. While sediment guidance recommends an evaluation of site-specific risks and benefits of management strategies in the feasibility study process, technology-specific and site-specific data on risks or benefits of sediment remedial or management strategies (especially in-place strategies) are sparse.

In-place management or containment strategies and in situ remediation technologies need to be validated, as do a toolbox of analytical and modeling methods in support of the feasibility process for such in-place management, containment, and remediation strategies. Without such data, it is unlikely that a regulators and stakeholders will embrace in-place treatment or containment processes. Unfortunately, there currently seems to be little incentive for contractors, regulators, stakeholders, or remedial program managers to streamline the assessment and remediation process because of the potential risks posed by the use of innovative methods. If innovative in-place remedial strategies are to be accepted, efforts must be made to balance the risks and benefits using teams of experts that bring various assessment activities to sites. The U.S. EPA SITE and ARCS programs have served this function well in the United States, primarily for soil and freshwater environments, but more sediment-focused efforts are needed.

In summary, three types of monitoring are useful in supporting a sediment remediation program: (1) long-term monitoring of resources driving the remediation; (2) short-term monitoring of interim measures of remedial success; and (3) short-term monitoring of implementation of remedial technology. Closure of the material balance on sediment and contaminants during implementation of a remedial technology is recommended as a means of maximizing the understanding of the remedy effectiveness and improving the ability to undertake similar projects in the future. A clear definition is needed of the goals of the remediation and how these goals will be measured help to ensure the effectiveness of remediation. This information will improve the selection, design, and optimization of remedial or management approaches at other sites in the future.

Table 4. Six key elements of a sediment assessment and management strategy

Develop and periodically refine a conceptual site model
Design transparent, scientifically sound, and site-specific approaches to assessment and decision-making
Carefully select reference areas that will be used to judge the site under study
Work to define the area(s) of contamination by the judicious and tiered use of chemical and biological tools
Understand the physical, biological, and chemical processes that drive sediment and contaminant risk, behavior, and fate
Recognize the complexity of making risk management decisions at sediment sites which might span various regulatory environments and geographical scales

SUMMARY AND CONCLUSION

In this review, the second of a two-part series on sediment assessment and management strategies, the authors have endeavored to provide a broad-based, balanced discussion of the key elements of an effective strategy for assessment of contaminated sediment sites, and the policy and management issues that arise before, during, and after the remedy selection and implementation process. Many site-specific chemical, physical, biological, and regulatory issues need to be considered in assessments of contaminated sediments. This article addressed the six key elements listed in Table 4 that all sediment sites appear to have in common.

Recently, the concept of basin-scale sediment management has been proposed (European Water Framework Directive, WFD 2000/60/EC), which adds additional complexity to the issues discussed in this paper. Such an approach requires not only source control for both sediments and contaminants within the sediment cycle, but also within the industrial, agricultural, developmental, and other processes that affect that cycle. The dynamic nature of sediment systems will call for new approaches to sediment management and require a framework in which sediment transport processes, the volumes of concern, and sediment quality goals are explicitly addressed throughout the framework. Apitz and White (2003) describe one such approach in which both regional-scale and site-specific risks are balanced with regulatory and socioeconomic goals explicitly derived for watersheds.

At present, several gaps remain in the current state-of-the-science of the fate of contaminants in sediment, and the effects of in-place and ex situ remedial strategies. As is true for most environmental issues, no single correct pathway exists to address sediment management. Approaches should be driven by the ecological, political, and economic goals of interested stakeholders and balance knowledge, uncertainty, and policy. The numerous biological and chemical factors that influence sediment assessment and management are so varied that no regulatory guidance or engineering solution can anticipate every permutation. Thus, sediment remediation is nearly always a matter for professional judgment and the use of site-specific information. Sediment managers, scientists, and decision-makers increasingly make use of a number of valuation approaches and economic models, including cost-benefit analysis, ecological risk assessment, and technology feasibility studies to support remedy decision-making.

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