CHAPTER 14

WATER QUALITY ASPECTS OF DREDGING AND DREDGED SEDIMENT DISPOSAL

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BACKGROUND

The dredging of U.S. waterways and harbors has been recognized by Congress as beneficial to the country as a whole. It is further recognized that dredging and dredged sediment

disposal practices as part of waterway and harbor navigation depth maintenance will have some impact on designated beneficial uses and water quality of the waters at the dredging and dredged sediment disposal sites. Congress, in the 1972 amendments to the Federal Water Pollution Control Act (PL 92-500), specified in Section 404 that the disposal of dredged sediments in U.S. waters may take place, provided that there is an avoidance of "adverse effects." It further stated that the disposal of dredged sediments should not result in violation of applicable water quality standards after considering dispersion and dilution, toxic effluent standards, and marine sanctuary requirements, and should not jeopardize the existence of endangered species. While Congress indicated that some adverse impacts associated with dredging and dredged sediment disposal are to be expected, these impacts are to be minimized. Further, federal and state water quality standards are not to be violated as part of dredged sediment disposal operations.

It is important to understand how U.S. Environmental Protection Agency (EPA) water quality criteria were developed and how they are conventionally used in water pollution control programs in order to understand problems with applying state water quality standards based on these criteria to evaluate the potential water quality impacts of dredging and dredged sediment disposal.

DEVELOPMENT AND USE OF U.S. EPA WATER QUALITY CRITERIA

The 1972 amendments to the Federal Water Pollution Control Act (PL 92-500) required that the U.S. EPA develop water quality criteria for determining excessive concentrations of constituents in ambient waters for various types of beneficial uses such as fish and aquatic life, domestic water supplies, agricultural use, etc. The first of these criteria were released in July 1976 as the "Red Book" criteria (U.S. EPA, 1976). Congress also required that the U.S. EPA develop a set of priority (most important) pollutants and water quality criteria for the priority pollutants. The U.S. EPA did not comply with this congressional mandate by the specified date, with the result that environmental groups filed suit against the agency. This led to a court order decree in which approximately 130 chemicals or groups of chemicals were designated as priority pollutants by the court.

It is recognized by many professionals in the water quality management field that this list of chemicals was not necessarily an appropriately developed list. However, these are the chemicals that receive primary attention in almost all current water pollution control programs. This list of chemicals is largely composed of compounds which are known or suspected animal and/or human carcinogens. Many of the chemicals on this list are chlorinated or brominated hydrocarbons. In November 1980, the U.S. EPA promulgated water quality criteria for the priority pollutants (U.S. EPA, 1980).

In July 1986, the U.S. EPA promulgated revised water quality criteria for a group of chemicals, principally heavy metals and ammonia, which can have a significant adverse impact on aquatic life. These criteria supersede the "Red Book" criteria. All of these criteria have been reissued by the U.S. EPA in what is called the "Gold Book" (U.S. EPA, 1987). The U.S. EPA (1996, 1999) updated the "Gold Book" criteria with new information that had been developed since the early 1980s for a number of the heavy metals and several other constituents that are of concern primarily because of their toxicity to aquatic life.

Congress in 1972 specified that the U.S. EPA water quality criteria are to serve as a guide to state pollution control agencies in developing the state water quality standards.

While typically U.S. EPA water quality criteria are not enforceable, state standards based on these criteria are enforceable and are the primary basis upon which the water pollution control programs in the United States are now formulated. Since the U.S. EPA must approve state water quality standards, typically the state standards are at least as protective of beneficial uses of receiving waters as U.S. EPA water quality criteria upon which they are based. States have the option of making their standards more protective than the U.S. EPA criteria. They, however, have great difficulties obtaining approval for less protective criteria.

In order to keep the state standards somewhat up-to-date with the recent information developed on the impact of constituents on aquatic life and other beneficial uses of water, the Federal Water Pollution Control Act specifies that the states must review their standards every 3 years and bring them up-to-date in accord with U.S. EPA requirements. The U.S. EPA must approve these standards every 3 years.

Federal and typical state water pollution control regulations specify that the objective of the regulations is the protection of designated beneficial uses of a waterbody. Frequently, this is translated into attainment of numeric, chemically based water quality standards. However, as discussed here, there are situations, especially associated with dredging and dredged sediment disposal, where the attainment of state water quality standards at the edge of a mixing zone associated with a dredging or dredged sediment disposal operation is unnecessarily overprotective of aquatic life. While congressionally mandated requirements associated with dredged sediment disposal operations specify that water quality standards shall not be violated, violation of standards would not necessarily be adverse to aquatic life at the dredged sediment disposal site. These violations would be "administrative" which relate to the worst-case characteristics of the U.S. EPA water quality criteria that serve as a basis for developing state water quality standards. It should be noted that the water quality impacts of dredging at the dredging site are considered to be *de minimis*.

The basic approach used today in water pollution control programs for regulation of wastewater discharges is to require that any discharger of constituents, such as a city, industry, etc., must obtain a National Pollutant Discharge Elimination System (NPDES) permit for the discharge. This permit typically specifies that the concentrations of various constituents that are likely to be in the wastewater discharge shall be no greater than the state water quality standard at the edge of a mixing zone for the wastewater effluent and the receiving waters. Each state has the ability to specify to some extent the size of the mixing zones that are allowed. Within the mixing zone, the concentrations of constituents can exceed a water quality standard.

Many states do not define a physically sized mixing zone, but instead define a minimum flow in the receiving water (river) such as a Q_{7-10}, which is the low flow in the river that occurs for 7 consecutive days once in 10 years. For this flow, the states will specify a load of various constituents in the wastewater discharge that will not exceed the water quality standard when the wastewater discharge is mixed into the receiving waters. Using this approach, the states do not have to define a mixing zone, nor is there need to measure the concentrations of various constituents in the receiving waters. All that needs to be done is to measure the concentrations of constituents in the effluent and determine the effluent flow. This provides sufficient information to determine whether the concentrations of constituents in the receiving water would exceed the water quality standard at the point where they are mixed into these waters.

In contrast to the mixing allowed under PL 92-500, wherein the size and nature of the mixing zone is left up to the individual states, specific constraints regarding mixing are required for ocean disposal under the Marine Protection, Research, and Sanctuaries Act (PL 92-532). In that instance the *mixing zone* is defined as that volume of water which

overlies the disposal site. Federal water quality criteria are not to be exceeded outside the boundary of the mixing zone nor anywhere in the marine environment 4 hours after disposal has taken place. Determination of whether the criteria are met is accomplished through the use of a mixing model provided with the ocean disposal implementation manual (U.S. EPA and U.S. CE, 1991). Where criteria do not exist for all constituents of concern or additive-synergism is suspected, the results of acute toxicity tests conducted with water column organisms are used to determine compliance with the criteria. Results of acute toxicity tests with water column organisms may also be used to determine compliance with state water quality standards under PL 92-500.

U.S. EPA water quality criteria are, in general, based on worst-case or near worst-case assumptions in which aquatic organisms have been exposed to essentially 100 percent available (toxic) forms of constituents for chronic or near chronic (extended) periods of time conditions (Lee, et al., 1982*a,b*). In some cases acute toxicity tests are conducted and chronic values mathematically derived through an acute chronic ratio, i.e., 10 percent, or 1 percent of acute value. These criteria are generally protective of aquatic life since, in many instances, chemical constituents exist in aquatic systems in a variety of chemical forms, only some of which are toxic to aquatic life. Further, there are many situations, especially near the point of discharge of constituents, where the duration of exposure of aquatic organisms to the constituents of concern is less than the duration of exposure applicable to the criterion values.

Figure 14.1 shows the typical relationship between the concentration of available forms of constituents and the duration of exposure that organisms may experience without adverse effects. As shown, there is a stippled area where aquatic organisms can be adversely impacted by available forms of constituents for a certain period of exposure. The U.S. EPA criteria are typically developed by extrapolating the horizontal line under the stippled area to the ordinate value. These criteria, therefore, are protective of aquatic life under all durations of exposure. However, it is well known that organisms can experience short durations of exposure in waters that contain concentrations of available forms of constituents well above the U.S. EPA criterion value without adverse impact on the organisms. Typically, it has been found that there is a factor of at least 10, and more commonly 50 to 100, between the concentrations of constituents that will kill half of the aquatic organisms in a 4-day exposure and the chronic safe concentration (U.S. EPA criterion value) for a chemical.

There is considerable concern about the use of the U.S. EPA worst-case–based water quality criteria as state standards, since this can lead to significant, unnecessary expenditures for chemical constituent control. While the U.S. EPA worst-case–based criteria are, in general, valid for the conditions for which they were developed, their mechanical application to most waters results in significant overprotection, which leads to unnecessary costs in controlling discharges. Beginning in the mid-1980s, the U.S. EPA (1994*a*) developed extensive guidance on modifying their worst-case criteria to consider site-specific conditions. In those situations where there are violations of the worst-case criteria/standards, site-specific investigations using U.S. EPA guidance will often enable adjustment of the standards to eliminate excessive expenditures for chemical constituent control over that needed to protect the designated beneficial uses of the waterbody.

APPLICATION OF U.S. EPA WATER QUALITY CRITERIA TO DREDGED SEDIMENT DISPOSAL PROJECTS

Figure 14.1 diagrammatically shows a typical water column duration of exposure for aquatic organisms associated with dredged sediment disposal operations. The typical dredged sediment disposal project is such that organisms in the water column for which

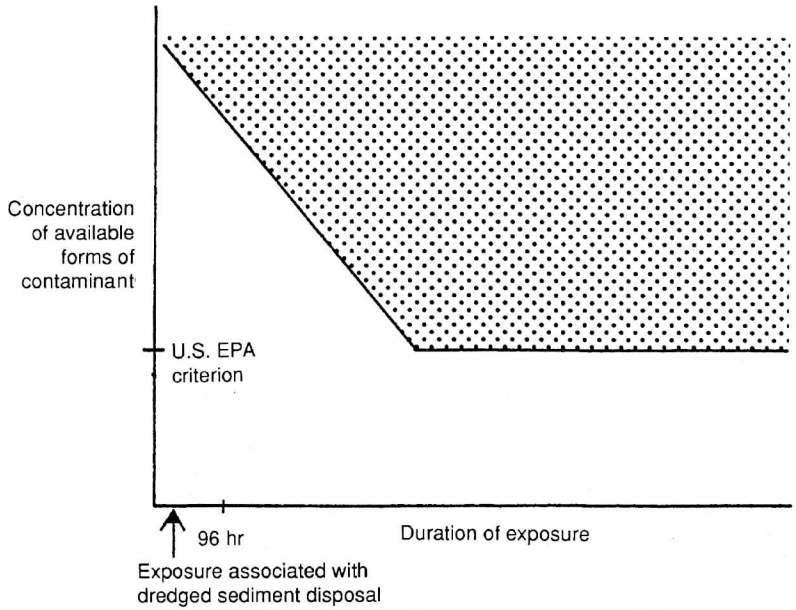


FIGURE 14.1 Aquatic toxicology.

U.S. EPA criteria, or state water quality standards developed on these criteria, are based can receive exposures of a few minutes to a few hours duration. There are few situations where the organisms would encounter a 4-day exposure, much less chronic exposure. Therefore, the application of U.S. EPA criteria or state water quality standards based on these criteria is unnecessarily overprotective of aquatic life in the water column associated with dredging and dredged sediment disposal operations.

Another significant problem with applying the U.S. EPA–based water quality criteria and standards to dredged sediment disposal projects is that the constituents of concern in dredged sediments are largely associated with particulate matter where they occur as precipitates or are sorbed (attached) onto sediment particles. It has been known for over 30 years that constituents associated with particulates are largely unavailable (nontoxic) to aquatic life. This situation makes U.S. EPA criteria and state water quality standards developed on these criteria, which are implemented based on total recoverable constituent concentrations, overprotective of aquatic life in a water column. It is, therefore, evident that applying U.S. EPA and state water quality standards based on these criteria to dredging and dredged sediment disposal projects is not a valid approach to determine potential adverse impacts of "excessive" concentrations of constituents. Concentrations of many constituents, such as heavy metals, can be present in the water column near dredging and dredged sediment disposal activities in concentrations many orders of magnitude above U.S. EPA criteria without adverse effects on aquatic life.

It is extremely important to understand that the U.S. EPA criteria and state standards are not applicable to the protection of aquatic life in sediments, including their interstitial waters, where the total concentrations of constituents in the sediments are compared to the criterion values. While some regulatory agencies are proposing to use this approach, it is technically invalid and should not be adopted. This issue is discussed further here.

In addition to concern about the direct toxic effects of chemical constituents in aquatic systems on aquatic life, there is also concern about the bioaccumulation of constituents in

aquatic organisms that occurs to a sufficient extent to cause these organisms to become unsuitable for use as food by humans or fish-eating animals. Beginning in the 1960s there was widespread concern about DDT, PCBs, mercury, and some other persistent chemicals' bioaccumulation within aquatic organism tissue to the extent that the edible tissue was judged unsafe for use for human food based on U.S. Food and Drug Administration (FDA, 1984) action levels. The FDA has promulgated acceptable concentrations (action levels) of certain chemicals in food. However, these action levels are based to some extent on economic considerations. In the past, and to some extent today, these action levels have been widely used as critical tissue concentrations for excessive bioaccumulation of constituents in aquatic organisms.

The U.S. EPA (1992*a*; 1995*a,b*; 1997*b*) has developed a risk-assessment approach for determining excessive concentrations of chemicals in fish tissue used as human food. This approach bases a critical tissue concentration on the current information on the hazard that ingestion of a certain chemical over a period of time represents to an individual. A key component of the risk-assessment approach is the fish consumption rate. In the early 1980s it was assumed that the average consumption rate of fish was about one meal per month. It is now recognized that there are people who eat one meal of fish per week taken from local waters. This consumption rate significantly decreases the allowed tissue residue. Typically, the critical concentrations that evolve from this approach are lower than those that have been developed by the FDA. The U.S. EPA has developed a number of its water quality criteria, such as for mercury, DDT, PCBs, dioxin, etc., based on worst-case assumptions about the potential for bioaccumulation of constituents present in water within the fish or other aquatic organisms' edible tissue. Lee and Jones-Lee (1996*a*) have discussed some of the problems of developing water quality criteria for bioaccumulatable chemicals based on worst-case assumptions.

The worst-case nature of these criteria assumes primarily that the bioaccumulation that occurs in relatively clean aquatic systems with little or no particulate matter present will occur in other situations where particulates are present. It is well known, however, that in the presence of particulate matter, the bioaccumulation factors that the U.S. EPA uses in developing their criteria are overly protective. Far less bioaccumulation occurs from constituents associated with particulates than for many dissolved constituents. Since dredging and dredged sediment disposal projects involve large amounts of particulate matter, U.S. EPA criteria and state water quality standards based on these criteria are overly protective of beneficial uses of water where the concern is the bioaccumulation of chemicals within aquatic life that can be used as food and are potentially impacted by the dredging or dredged sediment disposal operations.

One of the areas of frequent concern associated with bioaccumulation of chemicals in aquatic life is what constitutes an excessive amount of accumulation for chemicals for which the FDA has not established an action level. Also of concern is the allowable amount of bioaccumulation in nonedible organisms (i.e., worms or other benthic/epibenthic organisms) or within specific organs of an organism such as the liver. Some regulatory agencies will apply FDA limits to these organisms or organs as being appropriate for determining excessive bioaccumulation. This approach is technically invalid. What constitutes an excessive concentration of a chemical in the human diet does not necessarily translate to an excessive body burden in benthic organisms or other forms of aquatic life, and is not a valid basis for determining an excessive concentration of a chemical in an aquatic organism's liver or some other organ unless that organ is, in fact, used as food in accord with the assumptions that were used by the FDA in developing the action levels.

Recently, two compilations of critical concentrations of chemical constituents in aquatic life tissue, as they may impact the aquatic life with elevated residue tissue levels, have been published. One is the U.S. Army Corps of Engineers (CE)/U.S. EPA (1998) "Environmental Residue-Effects Database (ERED)," and the other was developed by the

U.S. EPA (Jarvinen and Ankley, 1999). Therefore, in addition to considering whether there are excessive tissue residues as they may impact humans and wildlife that use the aquatic life as food, there is also need to evaluate whether the tissue residues are potentially harmful to aquatic life through a critical review of the information available in these databases.

In addition to considering the potential impact of tissue residues on humans who eat local contaminated aquatic life, as well as damage to the host organism for the tissue residue, there is a need to consider whether higher-trophic-level birds and animals that eat fish and other aquatic life can be exposed to excessive concentrations of hazardous chemicals. The U.S. EPA (1993*a*), as part of the Great Lakes initiative, developed a wildlife-based water quality criterion for PCBs. This criterion considers whole organism content, not just edible tissue, as well as the fact that wildlife, such as fish-eating birds and animals, may consume greater amounts of fish than humans, thereby having a greater exposure to hazardous chemicals. Fish-eating animals' and birds' diets may consist largely of local contaminated fish and shellfish.

In the early 1970s, the predecessor to the U.S. EPA and some state pollution control agencies adopted regulatory approaches for dredged sediments that significantly increased the costs of dredging projects by 20 to as much as 50 percent, based on "excessive" concentrations of certain regulated constituents in the sediments. This approach was based on an inaccurate analysis of the potential impacts of the constituents in the sediments by the regulatory agency personnel and resulted in a significant unnecessary expenditure of public funds in the name of pollution control associated with dredging projects. Further, in some instances, the alternative, more expensive approaches adopted, some of which are still in use today, would be expected to be more harmful to aquatic life than the less expensive methods. Similar kinds of problems are occurring today with respect to using the U.S. EPA water quality criteria to regulate dredged sediment management.

It is evident from the preceding discussion that there are considerable problems in using federal water quality criteria or state standards as a basis for determining "adverse impact" on constituents associated with a dredging or dredged sediment disposal operation. It can be generally, if not universally, assumed that with a dredging or dredged sediment disposal operation, if the U.S. EPA criteria and state standards are not violated, then there is little or no likelihood that the regulated (i.e., those for which there are water quality criteria) constituents present in the dredged sediments will have an adverse effect on aquatic life in the water column near the dredging site or at the disposal site. If, however, the criteria or standards are violated, then it is likely that such violations do not necessarily constitute an adverse impact on the beneficial uses of the water. As noted earlier, the objective of the U.S. water pollution control program is protection of beneficial uses, not attainment of standards. The U.S. EPA's criteria have some applicability to regulating wastewater discharges, but have little or no technical applicability to cost-effective regulation of dredging or dredged sediment disposal operations. Certainly, it would be inappropriate to cause a particular dredging project to alter the proposed approach that is to be used for dredged sediment disposal based on the finding that the U.S. EPA criteria and/or state water quality standards would be violated.

THE DEVELOPMENT AND USE OF HAZARD-RISK-ASSESSMENT APPROACHES FOR DREDGED SEDIMENT PROJECTS

Lee, et al. (1982*a*) discuss the importance of properly evaluating the water quality impact of constituents on aquatic life in formulating technically valid, cost-effective water pollution control programs. They recommended that U.S. EPA water quality criteria and state

standards based on these criteria be used as an indicator of potential water quality problems. For further discussion of this approach, consult Lee and Jones-Lee (1995, 1996*b*). With few exceptions, as a result of the overprotective character of these criteria/standards, if the concentrations of constituents in a water are less than the criteria/standards, there is little or no likelihood that the constituents will have an adverse effect on aquatic life. If, however, the concentrations exceed the criteria/standards, then there is the potential for adverse impact. However, it should not be assumed that violation of the criteria/standards represents an adverse impact on beneficial uses. Lee, et al. (1982*a*), and Lee and Jones-Lee (1995, 1996*a*) recommended that, in most instances, the discharger of constituents which cause apparent violations of criteria/standards should be given the opportunity to conduct site-specific studies to evaluate whether the violations of the criteria/standards represent adverse impacts on beneficial uses of the water. These site-specific studies represent an aquatic life hazard-risk-assessment evaluation in which the real hazard that a chemical constituent or group of constituents represents to aquatic life-related beneficial uses of a water are evaluated. The U.S. EPA (1992*b*) has adopted the risk-assessment approach as a standard approach for evaluating whether the discharge of a particular constituent has the potential to cause a significant adverse impact on aquatic life and other beneficial uses of a waterbody. For additional information on this approach, consult U.S. EPA (1992*b*) and Bender and Jones (1993).

One of the first steps in developing a hazard-assessment scheme for use as an environmental quality management tool is to define the potential adverse impact that should be considered; that is, the beneficial uses of the water/area in question desired by the public must be defined. The beneficial uses can include aesthetic enjoyment, fishery, boating, swimming, navigation, benthic worm farm, drinking water source, habitat for migratory waterfowl—anything the public wants for a waterbody and can convince others through the regulatory process that that use is appropriate. The hazard/risk assessment is then designed to evaluate the impact of the activity/discharge on the uses desired.

The hazard-risk-assessment approach involves a sequential, tiered evaluation of the expected concentrations of available forms of constituents and aquatic organisms' duration of exposure to these available forms compared to the critical concentration duration of exposure relationships which are known to have an adverse impact on a particular form of aquatic life. The first tier is typically a "back of the envelope" evaluation, which is based largely on a preliminary analysis of the situation utilizing existing information. Higher-level tiers require laboratory and/or field studies to develop the necessary information to determine whether an adverse impact is likely, compared to the critical concentration duration of exposure situation. As discussed here, Lee and Jones have applied hazard-risk-assessment techniques that they developed to several dredging projects with considerable success. While the hazard-risk-assessment approach typically involves the expenditure of more funds in field studies than are normally expended, when taken in the context of the cost of alternative dredged sediment disposal operations, often a few dollars spent in hazard-risk-assessment studies may save many tens of thousands to millions of dollars in dredging project costs.

FACTORS CONTROLLING RELEASE OF CONSTITUENTS FROM SEDIMENTS

Figure 14.2 is a diagrammatic representation of a sediment-water interface for deposited sediments associated with dredging projects. The sediment-water column consists of four regions. The uppermost region is the water column in which there is typically a small amount of suspended particles. The next region is just above the sediment-water interface,

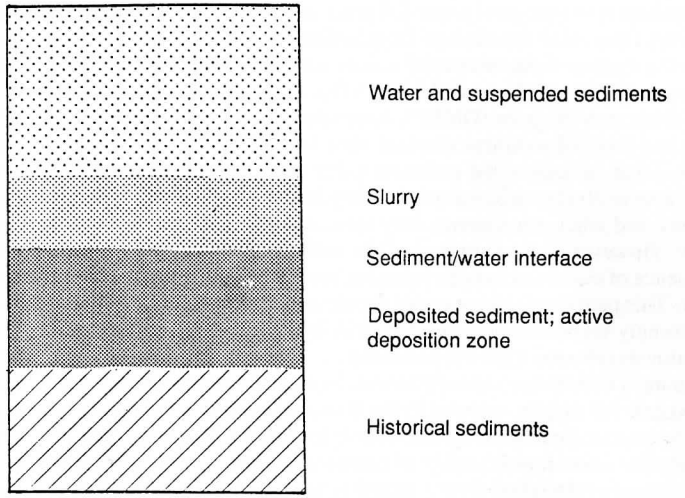


FIGURE 14.2 Characteristics of a sediment-water system.

where there is typically a significant increase in the number of suspended sediment particles. At times, this can be a slurry of suspended sediments. This is the area where constituents released from the bedded sediments is mixed into the overlying waters. This is also the area where those organisms present in the sediment that depend on oxygen obtain their oxygen as well as exposure to constituents present either in the water column or released from the sediments.

The sediment-water interface can be fairly diffuse, depending on the nature of the sediments and the turbulence of the water in this area. Below the sediment-water interface is the zone of active deposition in which there is active mixing of the sediments and their associated constituents within the sediments. This mixing can arise from a variety of causes such as wind, tidal or flow-induced currents, ship traffic, and organism activity. The latter includes biogenic formation of gases, such as methane and carbon dioxide, which would stir the sediments as the gas bubbles rise through them.

Below the active deposition zone are the historical sediments. Any constituents found in these sediments would not be expected to be brought to the surface through normal mixing processes. New work dredging, however, frequently involves removal of historical sediments. While these sediments are sometimes contaminated by human activity, in many instances, they are relatively clean and their impact on beneficial uses of water is largely restricted to physical impact. In some instances the characteristics of new work dredging project sediments is such that they have appreciable adsorption capacity that can remove constituents from the water column by this process on dredged sediment disposal.

As discussed by Lee (1970), the primary factor governing the release of constituents from sediments to the overlying waters is the mixing of the sediments into these waters. This mixing causes a change in the liquid-solid ratio which would tend to promote release of constituents from a sediment surface to water. Further, the stirring enables the constituents in the interstitial water to be brought to the surface at a significantly greater rate than typically occurs by diffusion-controlled processes. Since, in most instances, the concentrations of constituents in interstitial waters is greater than in the overlying waters, the primary factor controlling the release of constituents from sediments is the mixing or stirring process. Therefore, considerable attention should be focused on dredging activities, especially hydraulic dredging where the sediments are slurried in an approximate ratio of

one part sediment to four parts water (20 percent by volume). This slurrying would tend to result in rapid release of constituents from sediments due to the change in liquid-solid ratio and from the mixing of the interstitial waters with the waters used to slurry the sediments.

Lee, et al. (1978) and Jones and Lee (1978), as part of the Corps of Engineers Dredged Materials Research Program (DMRP), conducted extensive laboratory and field studies at dredging and dredged sediment disposal sites located in many parts of the United States. They found that, as long as the sediment water slurry was oxic (contained dissolved oxygen), of the over 30 chemical parameters they measured, including heavy metals, a variety of organics, and other constituents, only ammonia and manganese were released from the sediments. However, if the slurrying of the sediments with dredging site water took place in the absence of dissolved oxygen (anoxic), several potential pollutants were released into the water. This pattern of release under anoxic conditions and no release under oxic conditions is strongly supportive of the role of ferric hydroxide in acting as an efficient scavenger for constituents released from the sediments.

Iron in most sediments exists in a ferrous form. Upon contact with waters containing dissolved oxygen, it is rapidly oxidized by the dissolved oxygen to ferric iron which precipitates as ferric hydroxide. Freshly precipitated ferric hydroxide has a large surface which can sorb significant amounts of a wide variety of constituents. This is why few constituents present in interstitial water or desorbed during slurrying remain in the water column. While they are released, they are rapidly taken back to the sediments by the ferric hydroxide scavenging.

Lee (1975) pointed out that it is important to distinguish between the scavenging ability of freshly precipitated ferric hydroxide and aged ferric hydroxide precipitate. While freshly precipitated ferric hydroxide has a high sorption capacity, aged ferric hydroxide has limited sorption capacity and will release constituents sorbed at the time of formation of the ferric hydroxide, especially if it has become dry. This is an important phenomenon that occurs in some confined dredged sediment disposal projects that leads to the potential for water quality problems associated with this method of dredged sediment disposal.

Organism-Induced Release

In addition to a solubilization of precipitated constituents and desorption of constituents associated with sediment particles typically connected with stirring of the sediments into the overlying waters, sediment constituent release can occur in the intestinal tract of aquatic organisms that consume sediment. The pH, organic content, and other characteristics of the intestinal tract of some aquatic organisms is significantly different from many ambient waters, with the result that constituents that may not be released upon slurrying could be released within organisms. The possibility of this mode of release of constituents should be evaluated. While in some instances the release of constituents from aquatic organisms results in their being made available to the water column, most of the time this mode of release has been reflected in tissue residues which are best assessed through bioaccumulation studies.

The trophic level buildup of constituents in aquatic organisms where higher-level trophic-state organisms have greater concentrations of constituents in their tissue than lower-level organisms is of concern. Unlike terrestrial organisms, constituents in aquatic organisms tend to reach an equilibrium with the external environment because the organisms are permeable. Differences in constituent concentrations in organisms at different trophic levels are generally understandable in the context of lipid content, age, and other factors rather than trophic considerations. However, as noted by Kay (1984), there are a few constituents which have the potential for trophic magnification in aquatic environments. These include methylmercury, PCBs, and PAHs, possibly kepone and mirex. The bioaccumulation of constituents within organisms occurs from both direct uptake of

constituents from the water and the consumption of particles that have constituents associated with them as well as the consumption of other organisms. It is often difficult, if not impossible, to distinguish between these various modes of uptake. It is for this reason that water concentrations of constituents are often poor predictors of the amount of constituents that will accumulate within higher trophic-level organisms associated with a dredging or dredged sediment disposal project.

Another potential problem that has occurred with the interpretation of bioaccumulation data is the assumption that the concentrations in lower forms of aquatic life, such as polychaetes (worms), are representative of what would be present in higher trophic-level organisms such as fish. This is certainly not the universal case. Polychaetes could have high concentrations of a constituent without adverse effect to them or higher trophic levels which may feed on them. The significance of body burdens of constituents within lower trophic-level organisms is poorly understood at this time. While, as discussed previously, increasing information is developing on the impact of body burden of hazardous chemicals on the host organism for higher trophic-level organisms such as fish, limited information is available on the impact of elevated concentrations of hazardous chemicals in lower trophic-level organisms on the health of the host organism.

SIGNIFICANCE OF THE CONCENTRATIONS OF CONSTITUENTS IN SEDIMENTS

By far, the greatest cause of regulatory agencies' development of inappropriate dredged sediment management approaches is the failure of those who develop those approaches to understand that there is no relationship between the concentrations of constituents in sediments and the potential impact that constituents could have on the beneficial uses of the water associated with the sediments. Typically, the concentration of a constituent in water shows an increasing impact with increasing concentration. A concentration of constituents in sediments, however, does not always show this pattern. The difference is that concentrations in water are on a mass per unit volume of water basis. The water matrix (typically 1 L) is of constant composition (H_2O). However, the concentrations of constituents in sediments is on a mass per unit mass of sediment basis. The unit mass of sediment is of variable composition. In some cases it can be an oily waste or residue. In others, it is a quartz sand or a calcium carbonate precipitate. In other cases, it can be a clay or combination of types of clay. Each of these and other sediment matrices affects the impact that constituents present in them have on aquatic organisms and many other beneficial uses of the water. Since the impact that the constituent associated with the sediment has on aquatic organisms is related to the release of constituents from the sediments, it is not surprising that there is no general relationship between the concentration of constituents in sediments of various types and the impact these constituents in sediments have on aquatic life and water quality.

The authors have found that some sediments of a region with lower concentrations of constituents had a greater impact on water quality than those with the higher concentrations. This arose out of the fact that the basic sediment matrix for the two types of sediment were different. The sediments with the higher concentrations of constituents tended to hold the constituents much more strongly and therefore did not release them upon slurrying with water.

However, there may be situations where a region will have all the same types of basic sediment matrix. Under these conditions, since the binding power of constituents to the sediments is a constant, a relationship may be developed between the concentrations present in this sediment and their impact on water quality. It would be rare, however, that such a situation would exist and it should not be assumed, as it is often done, that there is a

relationship between the concentration of constituents in sediments and their impacts on water quality.

Since the late 1960s, various regulatory agencies have been trying to develop dredged sediment regulations using the bulk composition of sediments as a regulatory parameter. Many millions of dollars of public funds have been unnecessarily expended because of this approach as a result of regulatory agencies requiring that more expensive methods of dredged sediment disposal be used because the concentrations of constituents in the sediments exceeded some arbitrary "critical" concentration. While it has been known for over 20 years that this approach is technically invalid and wasteful of public funds, as well as possibly doing more harm to the environment than would occur otherwise, some regulatory agencies are still regulating dredging practices using bulk sediment chemical composition in developing dredged sediment disposal criteria. At this time, the U.S. EPA and some state regulatory agencies are actively pursuing numerical chemical sediment-based quality criteria/guidelines for the purpose of regulating the discharge of constituents to surface waters and of assessing the water quality significance of constituents associated with sediments. A number of approaches have been adopted and/or proposed which involve the use of numeric sediment quality criteria/guidelines as regulatory requirements that must be met. As discussed here, these approaches could ultimately cause significant, unnecessary expenditure of public funds associated with dredging projects.

POTENTIAL WATER QUALITY PROBLEMS ASSOCIATED WITH DREDGING PROJECTS

Congress mandated that the U.S. Army Corps of Engineers (CE) maintain the navigation depth for commercial and public transport of approximately 25,000 miles (40,000 km) of U.S. waterways. To accomplish this, the U.S. CE, either directly, or indirectly through contractors, dredges or issues permits to dredge about 500 million m^3/yr of sediment from these waterways. Generally, the least expensive and most expedient method to dispose of sediments once dredged is to discharge them in open water off-channel, or dump them in open water at a location removed from the dredging site. In the early 1970s, considerable concern was expressed about the potential water quality impact of open-water disposal of chemically contaminated dredged sediments.

There are several distinct aspects of water quality problems associated with dredging and dredged sediment disposal projects. It is important to distinguish among them to properly evaluate whether constituents in a sediment that is to be dredged in a certain manner could have a significant impact on the beneficial uses of water at the dredging or the dredged sediment disposal sites.

In evaluating the potential adverse impact of disposal of dredged sediments, it is important to distinguish between physical and chemical impacts. Within the potential chemical impact, it is important to distinguish between water column and redeposited sediment impacts. For the chemical impact of redeposited sediments, consideration should be given not only to toxicity to aquatic organisms but also to the potential for bioaccumulation of constituents present in the dredged sediments that would be adverse to using edible aquatic organisms as a source of human food, as well as to aquatic and terrestrial wildlife. A summary of the information available in each of these topic areas is presented in this section.

Physical Impact of Dredging and Dredged Sediments

At both the dredging and disposal sites, consideration should be given to the physical impact of dredging and dredged sediment disposal. These physical aspects range from

burial of organisms due to the settling of the suspended or dumped sediment to physical abrasion or clogging of gills or other organs by suspended sediment. Dredging, especially dredged sediment disposal, can have an adverse impact on certain organism populations. Great care must be exercised in dredging in the vicinity of special organism habitats, such as coral reefs or spawning areas, to be certain that the suspension and deposition of sediment is not adverse to the organisms. Problems of this type are primarily caused by the presence of sediment and can be largely independent of its chemical characteristics.

While redeposited dredged sediments, especially at the disposal site, can result in the burial of aquatic organisms leading to their death, the U.S. Army Corps of Engineers Dredge Materials Research Program studies showed that many organisms that live in sediments are able to migrate through appreciable depths of sediments dumped on them. The key factor seems to be the hydrogen sulfide concentration of the sediments since this chemical is fairly toxic to aquatic life. It also removes oxygen from the sediment column and therefore could deprive oxygen needed by organisms that are attempting to migrate to the surface of the sediment for a sufficient period to cause their death. Phenomena of this type are to be expected at designated dredged disposal sites where large amounts of anoxic sediments are dumped onto mobile aquatic organisms. There is little that can be done about this problem except to be certain that the designated disposal areas do not represent ecologically important areas for the region.

The physical impact of grain size and texture of the sediments must be considered in evaluating the impact of a dredged sediment disposal operation. In the studies conducted by the authors and their associates (Lee, et al., 1978) near the Galveston, Texas, Bay entrance channel in the Gulf of Mexico, it was found that dumping of large amounts of contaminated sediments resulted in little long-term impact on the aquatic organism populations in and upon the sediments that could be explained by other than physical grain-size changes. This site was a geologically high-energy disposal site where the dumped sediments were rapidly dispersed throughout the region. This dispersion quickly diluted the potentially hazardous constituents to the point where they had little or no impact.

While some regulatory agencies assert that increased suspended sediment concentrations are significantly adverse to aquatic life, as discussed by Jones and Lee (1978), work that was done under the sponsorship of the U.S. Army Corps of Engineers DMRP showed that high concentrations of suspended sediment, in some cases approaching grams per liter, can be present without adversely affecting the organisms due either to abrasion or organ blockage. The relative insensitivity of many organisms to high concentrations of suspended sediment is not surprising considering that these organisms have evolved in an environment where they have been frequently exposed to high concentrations of suspended sediment due to natural causes such as storms, flow-induced currents, tides, etc.

It is also sometimes asserted that dredging activities increase the water column turbidity to the point where the amount of photosynthesis that can occur in the water is curtailed, resulting in an overall impairment of the function of the ecosystem due to limiting algal growth. While dredging projects can, and usually do, increase water column turbidity for a short period of time, it would be rare for the decreased photosynthesis associated with this situation to have a significant adverse impact on the overall functioning of an ecosystem. It would also be rare for the suspended sediment associated with dredging and dredged sediment projects to have a significant adverse effect on aquatic life–related beneficial uses of a waterbody.

Impact on Water Movement, Bay Circulation, and Waste Assimilative Capacity

One of the potential consequences of dredging channels in harbors and other waterways is a change in the water transport and circulation patterns in the area. In an estuarine situation,

the dredged channel can allow saltwater to move further into the estuary than would occur otherwise. The dredged channels can also allow a change in the location of impacts of municipal and industrial wastewater discharges due to a more rapid transport of pollutants down the channel than would occur in the shallow, or nondredged, area.

The dredging of a channel can also affect the flow of groundwater into a bay or river as well as the flow of saltwater out of the estuary to the groundwaters of the area. The dredging of deep channels in areas where the groundwaters of an area have been protected from saltwater by a confining (low-permeability clay) layer within the sediments can result in a greater potential for saltwater intrusion into coastal freshwater aquifers. Obviously, before a channel is dredged or significantly deepened near an area that serves, or could serve, as a freshwater water supply source, fairly detailed hydrogeological investigations should be conducted to be relatively certain that problems of this type will not occur.

While there are known physical impacts of dredging and dredged sediment disposal of the types described previously, evaluation of the potential significance of them has to be done carefully on a site-specific basis. Further, it is not always found that the physical effects of dredged channel impacts are adverse to water quality.

Impact of Chemical Constituents in Dredged Sediments on Water Quality

In the late 1960s, it became generally known that many U.S. waterway sediments, especially those in urban and industrial areas, were contaminated with a wide variety of chemicals that, if released from the sediments, could have a significant adverse impact on the beneficial uses of the water. This finding caused pollution control agencies at the federal and state levels to attempt to develop criteria by which the water quality significance of dredged sediment–associated constituents could be judged. At the federal level, the Federal Water Quality Administration (a predecessor of the U.S. EPA) developed what became known as the *Jensen criteria* (Boyd, et al., 1972), which were based on bulk sediment composition. These "criteria" specified the total concentrations of a few water pollution indicator parameters or constituents, such as COD, volatile solids, zinc, etc., which, if exceeded, indicated that the sediments were unsuitable for open-water disposal and would have to be disposed of by alternative methods. At the time of the adoption of these criteria, it was known by many who were working on the water quality aspects of sediment-associated constituents that the total concentration of a constituent in a sediment is rarely a reliable indicator of the amount of the constituent that is available to affect aquatic life or water quality. Thus, the fundamental assumption of the Jensen criteria was invalid.

Nonetheless, the Jensen criteria had a pronounced impact on the U.S. CE's dredging operations as they were used by many U.S. EPA regions and states as the principal basis for evaluating potential dredged sediment disposal impacts. As a result, sediments that for many years had been dredged and disposed of in nearby open water were judged by these criteria to be polluted and alternative methods of disposal had to be undertaken. In the U.S.–Canadian Great Lakes region where there was considerable concern about "polluted" sediments, Congress authorized a $250 million construction program to undertake so-called "confined" disposal of dredged sediment as an alternative to open-water disposal. Because many of the U.S. CE districts faced significant increases in the cost of dredging operations associated with the implementation of the Jensen criteria and other factors, including the Clean Water Act (CWA) amendments of 1972, Congress authorized the U.S. CE to conduct a $30 million, 5-year Dredged Materials Research Program designed to evaluate, among other things, the significance of constituents associated with dredged sediments in terms of water quality.

Through this research program, which was conducted by the U.S. CE Waterways Experiment Station in Vicksburg, Mississippi, and completed in 1978, several hundred

reports were generated and many professional papers were published on the results of these studies. They can be accessed on the Internet at *www.wes.army.mil/el/dots*. In addition to the DMRP studies, a number of the U.S. CE districts conducted studies specifically designed to address problems of concern to them that were not being addressed by the DMRP. It is estimated that the total funding devoted to various aspects of dredged sediment disposal investigations and research in the 1970s was in excess of $40 million. About 25 to 30 percent of this amount was directed specifically to evaluating the water quality significance of constituents associated with dredged sediments.

The authors and their associates were involved in several laboratory and field studies, both as part of the DMRP and with several Corps districts, devoted to evaluating the water quality significance of constituents associated with dredged sediments, with particular emphasis on open-water disposal of these sediments (Lee, et al., 1978; Jones and Lee, 1978; Lee and Jones, 1977; Lee and Jones, 1981*a*). In this work, the reliability of the elutriate test, developed by the U.S. CE and the U.S. EPA as an alternative to bulk sediment criteria as a measure of release of constituents to the water column during open-water disposal of dredged sediment, was evaluated. An aquatic organism bioassay-toxicity test procedure was also developed and evaluated for its suitability to assess the potential toxicity of sediment-associated constituents to organisms living on or within the redeposited sediments. Part of this work was devoted to evaluating the potential for bioaccumulation within edible aquatic organisms of sediment-associated constituents that could render these organisms unsuitable for use as human food or be adverse to higher trophic-level organisms such as fish-eating birds.

Elutriate Test

The results of the DMRP studies conducted by the authors (Lee, et al., 1978; Jones and Lee, 1978) confirmed what was known at the time of the start of the DMRP (Lee and Plumb, 1974), namely, that the total concentration of a constituent or indicator parameter in a dredged sediment is not related to the release of the constituent in available forms to the water column during open-water disposal or to the toxicity of the sediment to aquatic life. This is because a variety of physical, chemical, and biological factors, principally sorption, reactions with the iron system, and hydrodynamics, control constituent uptake and release in sediment-water systems (Lee and Jones, 1987). Thus, the concentration of a constituent in a sediment cannot be used directly to estimate the potential impact of the sediment on aquatic life or other beneficial uses of a disposal site water.

Because of the importance of physical and chemical factors in controlling the release of sediment-associated constituents during disposal, the elutriate test was developed by the U.S. EPA and the U.S. CE to imitate in the laboratory conditions that could exist during a hopper dredging–open-water disposal operation. The procedure involves mixing dredged sediment with site water (20 percent sediment by volume) for 30 min with compressed air, allowing the mixture to settle under quiescent conditions for 1 hr, filtering and analyzing the filtrate for the chemicals of interest. The factors influencing the release of constituents during the elutriate test and the ability of this procedure to mimic the release of chemical constituents during actual dredged sediment disposal operations were evaluated by Lee, et al. (1978) by conducting more than 300 elutriate tests on a variety of waterway sediments from across the United States and monitoring about 20 dredged sediment disposal operations. It was found that when conducted under oxic conditions as prescribed, the elutriate test generally predicted both the direction and the approximate magnitude of constituent release upon open-water disposal of dredged sediments; failure to maintain oxic conditions during the test, however, renders the results of the test uninterpretable.

Ammonia was consistently released during elutriation of the dredged sediments. Concentrations in the elutriates were often higher than existing U.S. EPA water quality

criteria and many state water quality standards. However, as discussed by Jones and Lee (1978), this does not necessarily mean that water quality problems would result from the open-water disposal of these sediments. It does indicate that site-specific hazard assessments of the potential water quality problems associated with ammonia should be conducted when "high" concentrations are found in elutriates to determine the likelihood that sufficiently high concentrations would occur in association with the particular disposal operation, causing an impairment of the beneficial uses of the water at the disposal site.

Generally, the studies by Lee, et al. (1978) showed that zinc, iron, nitrate, copper, lead, cadmium, and phosphate, as well as a variety of chlorinated hydrocarbon pesticides, were not released during an oxic elutriate test. Some relatively clean sediments did show small amounts of PCB releases; however, as expected, contaminated sediments with large amounts of organics did not show PCB releases.

No relationship was found between the bulk chemical content of the sediments and the release of constituents in the elutriate test. Field studies showed that the elutriate test results predict the direction and approximate magnitude of constituent release during open-water disposal of dredged sediments. It is important, however, that the elutriate tests be conducted under oxic conditions (with air mixing) in order to reliably assess the potential for constituents released from the sediments to have an adverse impact on aquatic organisms at the dredged sediment disposal site water column. The elutriate test was found to be a valuable tool for testing the impact of potential water column constituents present in dredged sediments during open-water disposal operations.

The U.S. EPA and U.S. CE developed two testing manuals, *Evaluation of Dredged Material Proposed for Ocean Disposal—Testing Manual* (U.S. EPA, U.S. CE, 1991) and *Evaluation of Dredged Material Proposed for Discharge in Waters of the U.S.—Inland Testing Manual* (U.S. EPA, U.S. CE, 1998). The *Ocean Testing Manual* is designed to implement Section 103 of Public Law 92-532 (the Marine Protection, Research, and Sanctuaries Act of 1972). In accord with this section, the U.S. EPA is to develop regulations governing ocean disposal of dredged sediments which are to be used as a basis for evaluating the suitability of such disposal. The U.S. EPA regulations governing ocean disposal of dredged sediments, Title 40 of the Code of Federal Regulations, Parts 220-228 (40 CFR 220-228), contained a number of significant technical errors when they were first promulgated in the late 1970s. Unfortunately, these regulations have not been brought up to date. As a result, the U.S. EPA/U.S. CE guidance manual, which presents testing procedures for ocean disposal of dredged sediments, continues to use an inappropriate approach of applying U.S. EPA water quality criteria at the edge of a mixing zone or anywhere 4 hours postdisposal to determine if the criteria have been exceeded. These criteria are not appropriate for such an assessment. As discussed earlier, exceedance of the U.S. EPA criteria at the edge of a mixing zone associated with a dredging project rarely, if ever, would represent impairment of the beneficial uses of the waters of the region where the dredged sediment disposal would take place. This arises from the fact that the criteria were developed for extended periods of exposure compared to those that an aquatic organism could receive in the water column where ocean disposal of dredged sediment occurs.

Water column toxicity tests on the elutriates are used when there are no criteria for all constituents or additive synergism is expected to evaluate whether constituents present in dredged sediment elutriates could cause acute toxicity in the water column at the dredged sediment disposal site. It is important to emphasize that such an approach must properly consider the duration of exposure that organisms could receive at a particular dredged sediment disposal site. The duration of exposure of the acute toxicity test significantly exceeds the duration of exposure that organisms could receive at a dredged sediment disposal site involving open-water dumping of the sediments. The exposures that are possible under these conditions are on the order of minutes. The acute toxicity testing approach of elutriates is overprotective of what is needed to protect aquatic organisms at the dredged

sediment disposal site water column from toxicity that would be found in the elutriates. It is indeed rare, if ever, that ocean or inland water disposal of dredged sediments involving dumping could cause toxicity to water column aquatic life. The inappropriateness of the approach in this section again arises out of the outdated approaches adopted by the U.S. EPA in the mid-1970s regulating the ocean disposal of dredged sediments.

The U.S. EPA and U.S. CE (U.S. EPA and U.S. CE, 1998) published the *Inland Waters Testing Manual,* which presents testing protocols that provide guidance for use in conducting tests of dredged material to assess the potential for chemical constituent-related impact associated with dredged material disposal into open waters covered by Clean Water Act regulations. The testing is conducted in order to assist the permitting authority in making determinations regarding the effect of the discharge on the aquatic ecosystem and in determining whether the discharge will comply with the Clean Water Act 404(b)(1) guidelines.

A review of how the *Inland Waters Testing Manual* is being implemented in several areas of the country shows that large amounts of unnecessary testing is being done as part of developing data in each of the areas where there is concern about water quality impacts such as chemical characteristics, toxicity tests, bioaccumulation, etc. This testing is expensive and serves little purpose, beyond that of enabling the regulatory agencies, such as the Corps of Engineers district, the U.S. EPA region, and state water quality management agencies, to have data on record to show that the testing was done. This testing basically represents file cabinet fodder that is not required. The *Inland Waters Testing Manual* allows for variance from testing if previous information shows that the sediments from an area do not represent a significant threat to water quality when disposed of in a certain way. To mechanically test, just to have data, is contrary to the appropriate use of public funds.

During the 30 years there has been concern about the water quality significance of chemical constituents in dredged sediments, there has not been a single documented case where maintenance dredging of a waterway and disposal of the dredged sediments in open waters has been found to be significantly adverse to the beneficial uses of the receiving waters for the dredged sediments due to the presence of chemical constituents in the sediments. In the 1970s it was felt that there was need for a comprehensive testing program for aquatic life toxicity and bioaccumulation to protect the beneficial uses of receiving waters from significant adverse impact of chemical constituents in dredged sediments. Today, after over 20 years of intensive testing in many locations of the United States, it is appropriate to critically review the need for the extensive testing of waterway dredged sediments that is now being done in many areas. This does not mean that there are not contaminated sediments at some locations such as Superfund sites where large amounts of highly hazardous chemicals have been discharged/dumped into a bay or other area where there is limited circulation/mixing. However, the likelihood of significant adverse impact of chemical constituents in waterway dredged sediments that, when disposed of by open-water disposal, will be significantly adverse to the beneficial uses of the receiving waters for the dredged sediments is remote.

There is need in several areas of the country, such as San Francisco Bay, New York Harbor, etc., where there is heightened concern about the open-water disposal of contaminated sediments, to critically examine what are the real, significant water quality problems that have occurred or could occur in the future due to open-water disposal of highly contaminated dredged sediments. One of the most significant deficiencies of the CE/U.S. EPA dredged sediment management program is the failure of these agencies to require that detailed postdredging/disposal monitoring be conducted to evaluate what was the real impact of the disposal of contaminated sediments on the beneficial uses of the receiving waters. Basically, rather than mechanically continuing the extensive testing and filing of the test data in a file cabinet or computer database, it would be far more appropriate to critically monitor receiving waters in the region of open-water disposal of contaminated sediments to determine if any real, significant water quality impact occurred.

An evaluation monitoring approach of the type described by Jones-Lee and Lee (1998) should be used to examine such issues as whether the toxicity found in a dredged sediment in accord with *Inland Waters Testing Manual* procedures resulted in a significant adverse impact on the beneficial uses of the receiving waters for the open-water disposal of the dredged sediments. Similarly, how do the results of the *Inland Waters Testing Manual* bioaccumulation evaluation relate to the excessive bioaccumulation that is occurring in the receiving waters for the dredged sediments? By critically examining the existing beneficial uses of a waterbody that is receiving contaminated dredged sediments through follow-up monitoring of dredged sediment disposal, it will be possible to determine whether the current toxicity and bioaccumulation testing has relevance to cost-effective protection of the beneficial uses of waterbodies.

There have been a number of studies of this type conducted over the years (see Jones and Lee, 1978, and the publications of the DMRP and post-DMRP activities, available from *www.wes.army.mil/el/dots.)* Each time the results of the study showed that the open-water disposal of highly contaminated sediments has failed to cause significant adverse impact on the beneficial uses of the receiving waters for the dredged sediments. Ultimately, through an evaluation monitoring program, it should be possible to develop a more meaningful, cost-effective waterway dredged sediment management program that will be protective of the beneficial uses of receiving waters for open-water disposal of dredged sediments, without significant unnecessary expenditures for alternative, more expensive dredged sediment management.

Bioassay-Toxicity Sediment Testing

In the DMRP studies conducted by the authors, about 30 chemical parameters were measured in the dredged sediment elutriates. These tests showed that, except for ammonia, there was little or no likelihood of constituent release from dredged sediments that would be adverse to aquatic life at the disposal site water column. This conclusion is based on examining the relationship between the concentrations of constituents released and the concentration duration of exposure relationships that exist for aquatic organisms at a dredged sediment disposal site involving dumping of dredged sediments. In order to assess whether unmeasured chemicals could have an adverse impact on water column organisms, as well as organisms that might colonize the redeposited dredged sediments shortly after deposition and whether it was possible for additive-synergistic adverse impact of various chemicals to occur, Lee, et al. (1978) developed a dredged sediment elutriate bioassay screening toxicity test.

The dredged sediment bioassay-toxicity tests developed by Lee, et al. (1978) were generally conducted by proceeding through a standard oxic elutriate test procedure and introducing the test organisms into the settled elutriate. Grass shrimp (*P. pugio*) were used for marine conditions and daphnids were used for freshwater. The survival of the organisms in the test systems and controls were recorded over a 96-hr (or longer) period. Thus, the organisms were exposed not only to the constituents released from the sediment during the elutriate test but also to any constituents released over the 4-day test period and obtained by the organisms directly from the sediment particles present in the bottom of the test system. These test conditions represent worst-case conditions, more harsh than would likely occur at a dredged sediment disposal site since there was limited dilution of the constituents released from the sediments in the test system compared to what would occur at a typical dredged sediment disposal site .

The work of Lee, et al. (1978) on the potential toxicity of U.S. waterway sediments to aquatic life showed that many sediments, especially those near urban and industrial areas, contained constituents which caused them to be toxic to aquatic life under the conditions of the laboratory test. Typically, from 10 to 50 percent of the test organisms were killed in the 96-h test period. Some tests were carried out over a period of 21 days; they did not, in general, show significantly greater toxicity than that observed over the 4-day test period. The toxicity found in sediments obtained near urban-industrial centers was, however, much less than what would be predicted based on the concentrations of constituents that were present in the sediments. Further, it was found that there was no relationship between the bulk chemical content of constituents in sediments, either individually or collectively, and the toxicity of the sediments to aquatic organisms. While many sediments had high concentrations of heavy metals and chlorinated hydrocarbons of various types, these constituents were present in the sediments in nonavailable, nontoxic forms. In the case of New York Harbor sediments, Jones and Lee (1988) and Jones-Lee and Lee (1993) found that the toxicity was due to ammonia. Subsequently, other investigators have also found that ammonia is one of the principal causes of sediment toxicity in aquatic life.

Recent work by the U.S. EPA and others directed toward developing numeric sediment quality criteria is developing information that is helpful in explaining why chemical constituents in sediments are not toxic to aquatic life. While this has been known to some extent for some time, it is now clear that particulate forms of constituents are typically not available-toxic to aquatic life, including benthic and epibenthic forms. When the chemical characteristics of sediments are examined in light of possible chemical reactions that could cause chemical constituents of concern to be converted to particulate nontoxic forms, it is found that aquatic sediments typically contain a wide variety of constituents which would convert some of the most hazardous chemicals, such as heavy metals, chlorinated hydrocarbon pesticides, and PCBs, into particulate, nontoxic forms. For example, it has been known for many years that many aquatic sediments contain high concentrations of sulfides and polysulfides primarily in the form of iron sulfides. Almost all of the heavy metals, such as Cu, Zn, Cd, Ni, Pb, etc., tend to form insoluble sulfides. The solubility of many of these sulfides is less than iron sulfide and therefore would replace iron as a precipitating metal for the sulfide species. The recent studies by the U.S. EPA and others have shown that heavy metal sulfides are nontoxic to aquatic life. Since the concentrations of noniron heavy metals in sediments would rarely exceed the concentrations of sulfides in sediments on a molar basis, it would be unusual for a sediment to contain sufficient concentrations of sulfides to detoxify all of the heavy metals present in them.

It should be noted, however, that there are a variety of other mechanisms for detoxification of heavy metals in sediments that must also be considered. These include carbonates, hydrous metal oxides, solid-phase organic complexing agents, clays, etc. Therefore, it would not be surprising to find that even if the total noniron heavy metals in a sediment exceeded the sulfides and polysulfides measured as acid volatile sulfides (convertible to H_2S under acid conditions) that the sediments would be nontoxic due to heavy metals.

The work of the U.S. EPA and others has included examining the role of particulate organic matter present in sediments measured as total organic carbon (TOC) in binding potentially toxic, nonpolar organic chemicals to sediment particles. The chemicals of greatest concern are the chlorinated hydrocarbon pesticides (DDT, aldrin, dieldrin, chlordane, etc.), PCBs, and the polynuclear aromatics (PNAs-PAHs). All of these chemicals can be toxic to aquatic life under water column conditions. However, in sediments they would tend to bind to the organic matter through sorption reactions. The U.S. EPA studies have shown that high TOC-containing sediments tend to be much less toxic for given concentrations of these chemicals than low TOC-containing sediments. It is reasonable to explain this observation by understanding that the high TOC sediments, which have a higher amount of organic matter present, can bind the organic toxicant in such a way as to detoxify it. It is, therefore, evident that one of the primary reasons why the chlorinated hydrocarbon pesticides and other organic chemicals present in sediments are, in general, not toxic or have limited toxicity to aquatic life is because they are bound to particulate organic matter.

In the 1970s, the U.S. EPA and U.S. CE developed an unnecessarily complex approach for sediment bioassay testing (U.S. EPA and U.S. CE, 1977). This approach applies to ocean disposal of dredged sediments pursuant to the MPRSA and is still required by 40 CFR parts 220–228. It requires individual tests on various component parts of the elutriate, using three different types of test organisms. Not only did this requirement increase the cost from a few hundred dollars to several thousand dollars per sample but also the results were not readily interpretable. Essentially the same amount of useful information can be obtained from a simple screening toxicity test using a moderately sensitive, benthic/epibenthic organism, such as grass shrimp or amphipods for estuarine and marine waters and daphnids or amphipods for freshwater, as is obtained using three different kinds of organisms (Lee and Jones, 1977). Although the regulations have not been revised, current guidance (U.S. EPA and U.S. CE, 1991, 1998) recommends using combined liquid/suspended phase (rather than separately) but, in accord with regulations, three organisms continue to be used.

If the screening toxicity tests show potentially excessive toxicity, then site-specific follow-up testing could be undertaken on an array of organisms, various sediment-water ratios, and under other conditions that may influence the results, in a hazard assessment format to assess the potential for adverse impact under the particular conditions being considered; thus the mechanical conduct of an expensive array of tests may be reduced. Similarly, if the screening toxicity tests indicated that there was essentially no toxicity under worst-case conditions, testing could be terminated and considerable time and money saved.

One of the controversial issues that has still not been appropriately resolved for either marine or freshwater disposal of dredged sediments is what is an allowable percent mortality in a sediment toxicity test. It is certainly inappropriate to assume, as has been done in some parts of the country, that if there is a statistically significant difference in organism deaths in the test systems and the control systems, open-water disposal of the sediments would result in a significant adverse effect on aquatic life at the disposal site. It is important to understand that there may be little or no relationship between a 10, 20, or some other percent mortality in the sediment toxicity test and what will actually occur at the dredged sediment disposal site.

The test protocol should be designed in such a way as to properly consider the "noise" in the test results. In this regard, there seems to be considerable confusion about the difference between "statistical" significance and "water quality" significance. As discussed by Jones and Lee (1978), it is important to clearly distinguish between these two concepts. The former is simply a comparison of numbers and is a function of sample size, degree of confidence, etc. While of importance in ensuring the integrity of sampling and analytical programs and in describing the variability of a system, the statistical significance has essentially no relationship to water quality significance. The concentration of a chemical can and does increase by a statistically significant (at some prescribed confidence level) amount without having an environmentally significant impact. The former indicates that there are sufficient data to indicate that an increase in organism impact has occurred at the prescribed confidence level. If the confidence level is changed, the increase may no longer be "significant" statistically. If the increase has water quality significance, it means that the increase is sufficient to cause an adverse impact on beneficial uses of the water.

Jones and Lee (1978) discussed the approach that should be used in interpreting the dredged sediment toxicity test data. As conducted, this test estimates the impact on fairly sensitive aquatic organisms that could occur if the organisms spent 4 days in settled water that is the same as that in the dredge discharge pipe. Not considered in this assessment are the dilution of the released constituents with area water and incoming water, the dilution/mixing of the dredged sediments with other nontoxic sediments of the disposal region, the mobility and avoidance behavior of many types of organisms, or the myriad physical and chemical reactions that typically take place at a disposal site. The physical and chemical reactions, principally the scavenging of chemicals by ferric hydroxide, tend to render many

chemicals less toxic. As a result, the toxicity observed during the worst-case elutriate toxicity tests is likely to be different from, i.e., less than, that exhibited by a dredged sediment dumped within a disposal area.

Laboratory-based toxicity tests can represent well beyond the worst-case situations that can occur under field conditions (Jones and Lee, 1988). Under field conditions, constituents released to the sediment-water interface have the opportunity for significant dilution. Organisms that derive their constituents through uptake of water at or near the sediment-water interface would typically be exposed to significantly lower concentrations under field conditions than in the laboratory test. Therefore, significantly lower toxicities would more likely be observed in the field than in the laboratory.

During the past few years, increasing use of partial chronic toxicity tests has been made to assess whether a particular water has the potential for toxicity to aquatic life. Some tests of this type involve the use of recently hatched fish embryos which are exposed to the water and constituent for about a 1-week duration. The survival and growth of the embryos are assessed during this week-long exposure. The results of such tests are found to provide similar levels of toxicity to those that are typically found with full-scale chronic aquatic organism testing involving several years of exposure. The U.S. EPA has published two guidance manuals for conducting these tests (U.S. EPA, 1994*b*; Lewis, et al., 1994). These tests are relatively simple to conduct and relatively inexpensive. When testing an ambient water, they provide significant information on whether there is toxicity in the water that could be of significance to aquatic life in the area. It is important that their use, however, be conducted in such a way as to properly mimic the exposure that aquatic organisms could receive associated with the dredging project. It is virtually impossible for any dredged sediment disposal operation to provide an exposure of aquatic organisms to constituents in a water column for a week's duration. Therefore, the use of these tests should involve shorter exposure periods to the dredged sediments or dredged sediment elutriates than the typical test period and, in accordance with the regulations, be interpreted given the mixing that occurs at the disposal site.

Bioaccumulation

By far the greatest difficulty with the U.S. EPA–U.S. CE (1991, 1998) dredged sediment testing manual is its prescribed test for bioaccumulation. The U.S. EPA and the U.S. CE (1991) use a laboratory-based, 28-day bioaccumulation test for organic constituents. The lengthening of the time from the previously used 10 days to 28 days for bioaccumulation is appropriate for some chemicals, such as dioxins, since it appears that 10 days is too short an exposure to allow full accumulation. The lengthening of the test period does not, however, address the major problem with this test of not being able to extrapolate the laboratory results to field conditions.

All that can be deducted from the results of a bioaccumulation test is that, under the test conditions, bioaccumulation to a certain degree occurred in the test organisms. This provides little or no useful information on the bioaccumulation that will actually occur in edible organisms, such as fish, in the field at a dredged sediment disposal site. With few exceptions, although it could occur, the accumulation that will occur under laboratory conditions will greatly exceed what will occur in the field. Therefore, the bioaccumulation test, whether conducted for 10 or 28 days, is not a valid test to determine whether excessive concentrations of constituents present in a dredged sediment will accumulate in edible tissues of organisms of concern with respect to their use as human food in excess of U.S. Food and Drug Administration or U.S. EPA guidelines. Since the relationships between actual bioaccumulation under field conditions and the bioaccumulation found under various sediment-water-organism biomass couplings are not defined, it is difficult, if not impossible, to

extrapolate the results of such laboratory tests to a field situation. On the other hand, this test is environmentally conservative since field exposure is typically far less than that achieved in the laboratory under the conditions used.

Lee and Jones (1977) and Jones and Lee (1978) have described an approach to evaluate bioaccumulation with open-water disposal of dredged sediments. It involves collecting organisms from a disposal site which has received some of the sediments that are in question. These organisms should be analyzed for the various constituents of potential concern and the results compared to FDA action levels and U.S. EPA screening values for these constituents. The organisms should be collected at least twice a year, e.g., spring and fall, to detect major seasonal differences. Also, care must be exercised in selecting the types of organisms for analysis. A distinction should be made between those organisms which are transitory, i.e., migrating through the area, and those that tend to stay near the disposal area for a sufficient period of time to accumulate the constituents of concern. While it is of potential interest to see if organisms in a particular area of concern contain higher concentrations of constituents not on the FDA or U.S. EPA lists than those in other areas, at this time, the interpretation of the significance of body burden data for these chemicals to the organism or to organisms that use these organisms as food is not reliably possible.

Hazard-Assessment Approaches for Dredged Sediment Disposal

A need exists to utilize a combination of elutriate tests, dredged sediment bioassays, and knowledge of the physical, chemical, and biological characteristics of the disposal area to determine the potential for adverse impact of constituents present in a particular dredged sediment on the beneficial uses of a disposal site water. Lee and Jones (1981*a*) developed a hazard-assessment approach to provide a framework which can be used to identify information needs, integrate the information as it is generated, and provide guidance in the integration and interpretation of elutriate test, dredged sediment bioassay, and disposal site characterization results in making dredged sediment management decisions.

Figure 14.3 presents a hazard-assessment scheme that was developed by the authors for assessing the potential impact of dredging and disposal in the upper Mississippi River (Lee and Jones, 1981*a*). It shows how elutriate tests, dredged sediment bioassays, and the physical, chemical, and biological characteristics of the disposal site can be used in a tiered hazard-assessment scheme to evaluate dredging and disposal alternatives.

With reference to Fig. 14.3, the first feasible methods for dredged sediment disposal should be identified and evaluated in order of cost. If the least expensive method does not have an unacceptable hazard associated with it, it would likely be the method of choice. Generally, open-water disposal is the least expensive disposal alternative. The proposed disposal site should be examined to be certain that it is not an ecologically sensitive area such as a fish breeding ground, oyster bed, etc. This is generally of greatest concern if the area has not received dredged sediments in the recent past. If it is a sensitive area, another site should be selected and evaluated. If it is known to not be a sensitive area, other aspects of potential concern with respect to dredged sediment disposal can begin to be evaluated. If the sensitivity of the proposed disposal site is not known and cannot be readily determined, the potential impact may be evaluated by examining the proposed dredged sediments. Under the latter two conditions, the testing continues in the first tier by evaluating the dredging site conditions. If the disposal site is not a sensitive area and the sediments are not being derived from an area near a source of constituents, the technical recommendation to the public would likely be that there should not be a significant impact on beneficial uses of the disposal site water.

If the sediments were identified in the tier 1 evaluation as being from near a source of constituents, the potential for the release of chemicals from them should be examined in a more advanced, tier 2, testing level. That is, insufficient information was obtained from the

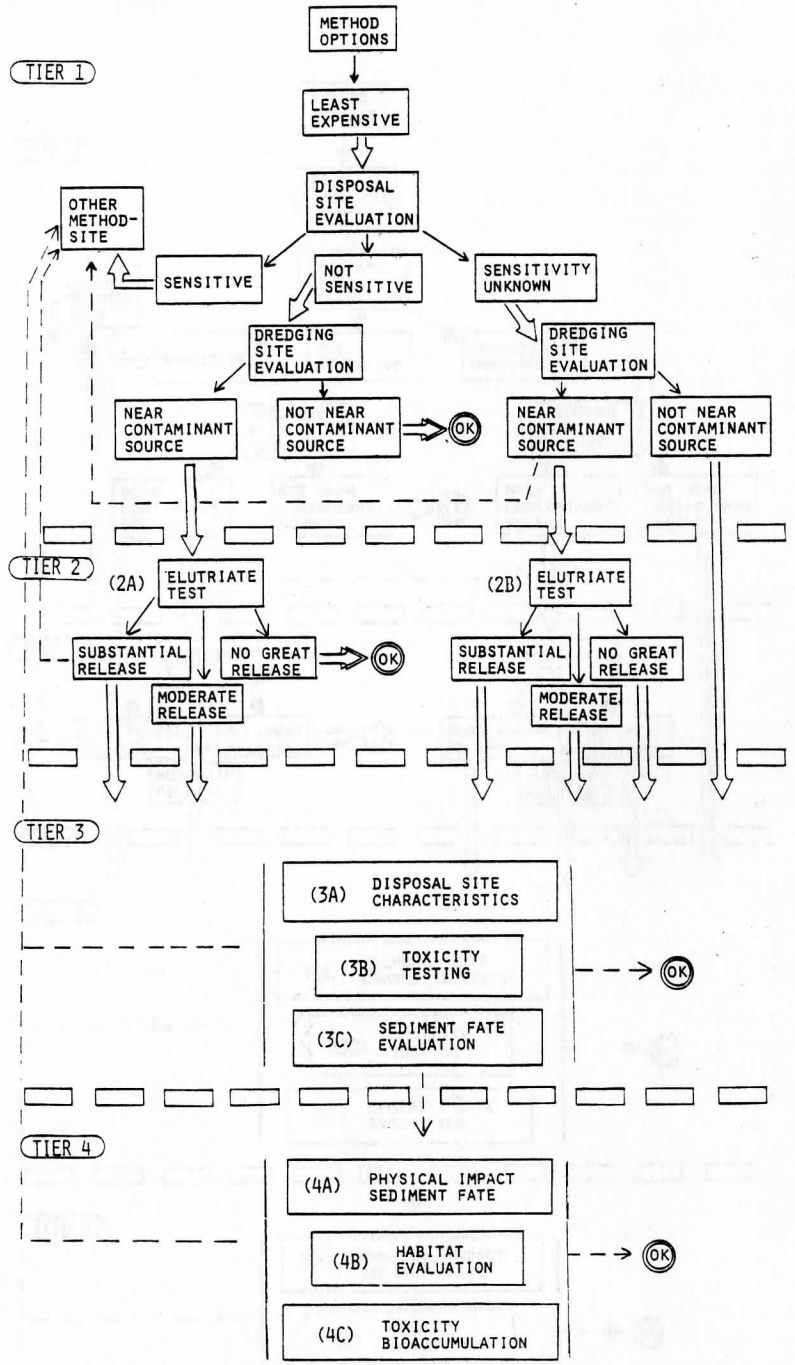


FIGURE 14.3 Abbreviated diagram of environmental hazard assessment for dredged sediment disposal.

tier 1 investigation to make a recommendation regarding the potential adverse impact of dredged sediment disposal. Tier 2 testing, in the scheme outlined in Fig. 14.3, consists of elutriate tests run under oxic conditions. Depending on the types and amounts of chemicals released during elutriation (a worst-case procedure), additional testing should be undertaken, other methods of disposal considered, or a recommendation should be given that the expected hazard should be considered acceptable.

Where there are still questions about the potential impact of the particular dredging/disposal scenario being considered after the tier 2 tests, it may be desirable to undertake the next level of testing. In this and subsequent levels, the tests become much more site-specific, involved, and expensive. Additional information is collected to fill gaps in understanding the disposal site characteristics such as dilution and dispersion, the acute toxicity of chemicals that were released in the elutriate tests and in a screening bioassay, and the fate and persistence of the sediments at the disposal site. Depending on the results of these tests, a decision could be made that another method of dredging/disposal should be considered, a recommendation should be made that the risk should be considered acceptable, or additional information is needed in order to make the technical assessment of potential impact.

Under the last condition, tier 4 testing and the cost of conducting those types of assessments compared to the cost of evaluating other methods would be considered. Tier 4 testing would focus on the more subtle potential impacts of dredged sediment disposal at the particular area and would be largely in the form of experimental dredged sediment disposal operations using the site/method evaluated through tier 3, but with detailed monitoring of the disposal site. In undertaking tier 4, those responsible must carefully evaluate whether irreparable damage could be done to the ecosystem of the regions as a result of deposition of dredged sediment in the area. Based on past studies, it appears that there will be few situations where irreparable damage will be done to an aquatic ecosystem by the disposal of a single or several sediment loads at a site; normally such potential situations would have been identified in earlier tiers of testing.

This hazard-assessment scheme is an illustration of an approach that should be developed for a particular area; it is not intended to be an all-inclusive statement. The particular tests prescribed depend on the information known and lacking, etc. Before it is used, the back-up papers and reports (Lee and Jones, 1981*a,b*) should be read and the subtleties and implications of the approach understood.

It is important to distinguish between a tiered hazard assessment of the type described previously and a tiered assessment that is being used by regulatory agencies such as the U.S. EPA and the U.S. CE for evaluating the potential water quality impact of constituents in dredged sediments. Typically, the first tier of the U.S. EPA–U.S. CE focuses on evaluation of existing information to determine if there is reason to believe that the sediments have been contaminated. Part of tier 2 focuses on bulk sediment inventory in which an evaluation is made as to whether the dredged sediments are more contaminated than the disposal site sediments. While rarely is this the case, if this is found, caution should be exercised in making any judgments about the potential impact of dredged sediment disposal based on bulk sediment chemical analyses. The relative levels of contamination between the dredging site and disposal site sediments or for that matter any other sediments, such as a reference sediment, provide no reliable, useful information on the potential impact of constituents in these sediments on beneficial uses of water at the disposal site.

Another part of tier 2 involves the use of the elutriate test in which a comparison is made between the amounts of constituents released in the test with water quality criteria and standards that would be applied at the edge of a mixing zone. Again, as discussed previously, it is technically invalid to determine potential impacts of constituents released in the elutriate test on water quality at an open water dredged sediment disposal site by comparison to water quality criteria/standards. Violation of such criteria/standards would rarely, if ever, represent an adverse effect on water quality at the dredged sediment disposal site.

Tier 3 focuses on acute aquatic organism bioassays using the U.S. EPA–U.S. CE multiple-species multiple-test approach. This is an overly complex, overly expensive approach that for most dredging projects unnecessarily and significantly increases the cost of testing. Tier 3 also focuses on bioaccumulation assessment involving laboratory testing of sediments and various organisms. This testing approach, however, does not provide reliable information on the bioaccumulation that will occur at a dredged sediment disposal site.

As part of developing updated guidance for implementing regulations governing ocean disposal of dredged sediments, the U.S. EPA and U.S. CE (1991) have used a tiered-assessment approach for determining whether dredged sediments may be disposed of in the ocean. This approach suffers from some of the same deficiencies described earlier. The U.S. EPA–U.S. CE tiered-assessment approach is not a properly developed hazard-assessment approach of the type described by Lee and Jones (1981*a*). Considerable testing funds could be saved at many sites through the use of the Lee and Jones approach. An example of such savings is provided by the work that they did for the Norfolk District of the Corps of Engineers (Lee and Jones, 1981*b*), where through a true selective tiered hazard assessment, it was possible with limited additional testing to demonstrate to regulatory agencies that the ban that had been imposed on dredging in part of the Intercoastal Waterway due to the presence of certain constituents in the sediments was technically invalid.

IMPACT OF EQUIPMENT USED FOR DREDGING AND DREDGED SEDIMENT DISPOSAL

There are basically two types of equipment used to dredge U.S. waterway sediments. These are characterized as mechanical and hydraulic dredging. Mechanical dredging typically involves use of a clamshell or dragline to remove sediments in a bucket which are typically deposited in a barge for disposal. Normally, mechanical dredging and dredged sediment disposal has the least potential for water quality impact due to constituents associated with the sediments. This is because there is little mixing of the sediments with water and therefore there is limited opportunity for release of constituents from the sediments to the water.

Hydraulic dredging, on the other hand, involves slurrying the sediments with water in a one-part sediment to four-parts water mixture where this mixture is typically pumped as a slurry to either open water or to confined upland disposal or to a hopper of a hydraulic dredge. The slurrying of sediments results in mixing of the sediments with water which tends to promote the release of constituents from the sediments and present in the interstitial water into the slurry water (see Fig. 14.4). However, it is well documented that such release does not occur for most elements once the slurried waters are in contact with dissolved oxygen because of the scavenging of the released constituents by ferric hydroxide.

There are basically two types of disposal operations associated with hydraulic dredging. One of these involves pumping the slurry into a hold/hopper of a ship (hopper dredge) where the dredged sediments are transported to the disposal site. Typically, hopper dredging involves open-water disposal of the sediments, although it is possible to pump from the hoppers to on-land disposal. Figure 14.5 is a diagrammatic representation of the principal components of a hopper dredged dumping of sediments. Basically, when the dredge is at the appropriate location, the doors on the hoppers, which are on the underside of the ship, open and the contaminated sediments are allowed to fall out. Most of these sediments descend rapidly to the bottom as a cohesive mass. While the sediments were placed into the hoppers as a 20 percent slurry, typically the excess water is allowed to drain off at the dredging site or in transport to the disposal site, with the result that the slurry characteristics of the sediment are rapidly lost after being placed in the hopper.

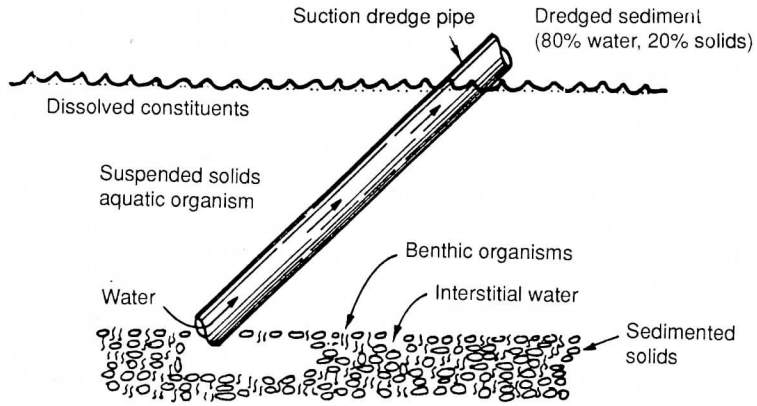
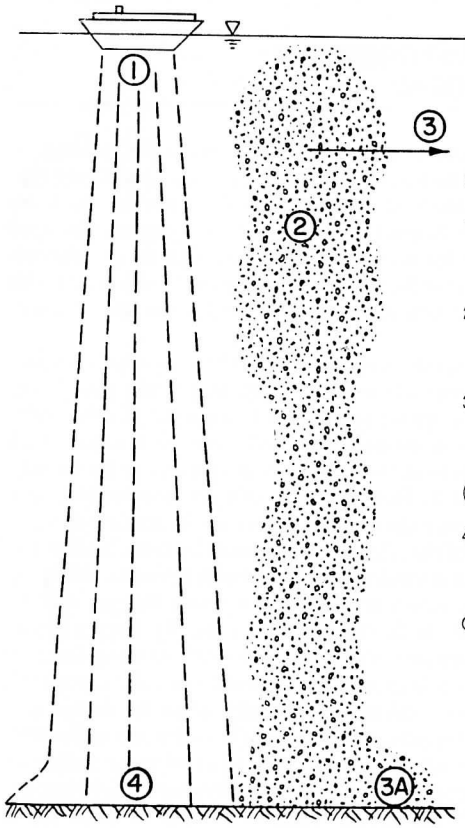


FIGURE 14.4 Hydraulic dredging—sediment removal.



Phases

1. Dump-descent of dumped sediments
   Short time: seconds
   Entrainment surface waters
2. Formation of turbid plume
   Small amount of suspended solids
   in water column
3. Transport of turbid plume
   Dilution to nondetectable in an
   hour or so

(3A). May have density current near bottom

4. Erosion of deposited sediments
   Mixing of dumped sediments with
   sediments of region

Overall: Change sediments from anoxic-reducing
oxic-oxidizing conditions, which reduces
availability of contaminants

FIGURE 14.5 Hydraulic dredging—hopper dredge dumping.

As indicated in Fig. 14.5, small amounts of water column turbidity are associated with the descent of the dredged sediments from the hopper to the bottom of the water column. This forms a turbid cloud with the waters of the region. Based on the studies by Lee, et al. (1978), while this turbid cloud does show some release of chemical constituents, its size and the magnitude of the release is such that no water quality problems have been found, and they are not expected due to constituent release associated with the disposal operations.

In the studies conducted by Lee, et al. (1978), the only constituent released of potential concern was ammonia. Figure 14.6 shows the typical passage of the turbid plume associated with the open-water disposal of dredged sediments. As indicated in Fig. 14.6, near the surface (2-m depth) the turbidity persisted at a location a few tens of meters down current from the dump for about 2 min. Near the bottom at 14 m, the turbid plume turbidity persisted for about 7 min. The dissolved oxygen (DO) depletion for this dumping operation near the surface is shown in Fig. 14.7. The DO depletion due to the oxygen demand in the sediments, while measurable, would not have an adverse effect on aquatic life because of the short duration and limited magnitude.

The release of ammonia from two hopper dredged dumps of contaminated sediments in open waters that took place about 2 hours apart is shown in Fig. 14.8. It is evident from Fig. 14.8 that the release of ammonia would present no adverse impact on aquatic life unless the organisms were able to stay in the rapidly moving turbid plume for a considerable period of time and the plume persisted for a long period of time. Studies by Lee, et al. (1978) at several locations showed that, typically, the turbid plume was no longer identifiable after about 1 hour following the dump. The concentration of released ammonia and the duration of exposure that organisms encounter associated with release are sufficiently short (minutes) so that no water quality problems due to ammonia release would be expected.

It may be concluded that the hopper dredged disposal of even contaminated sediments would not be expected to cause any water column water quality problems because of the short

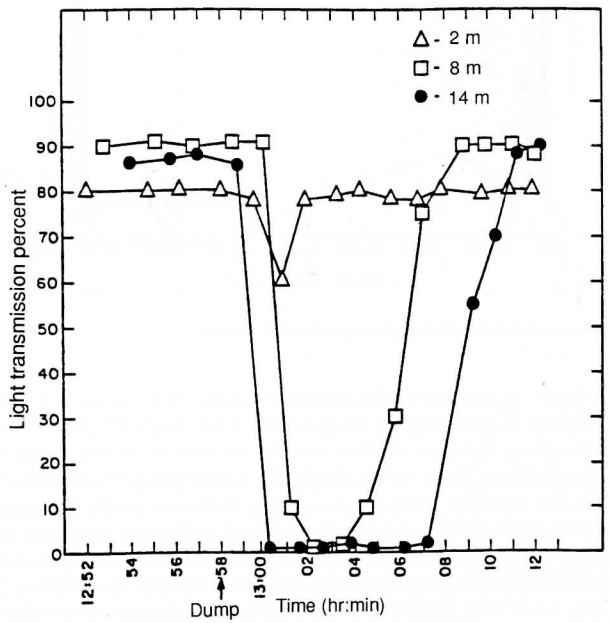


FIGURE 14.6 Passage of the turbid plume.

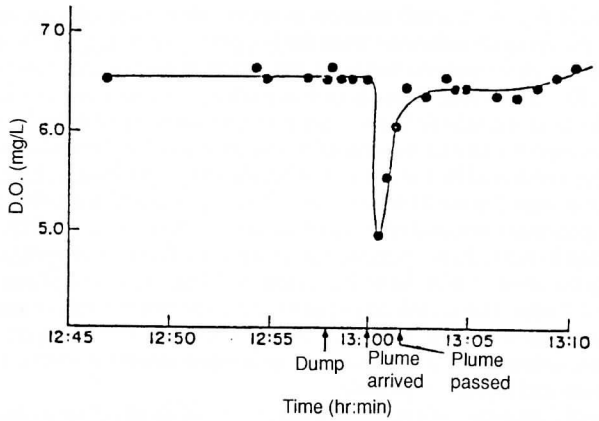


FIGURE 14.7 Dissolved oxygen depletion during passage of turbid plume.

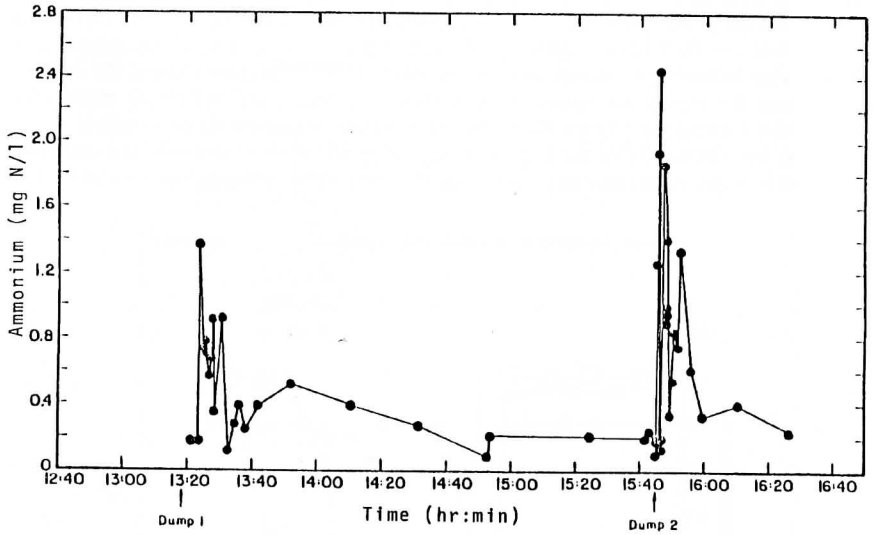


FIGURE 14.8 Ammonia release during passage of turbid plumes.

exposures that aquatic organisms could experience from such releases. A similar situation exists for the dumping of mechanically dredged sediments in open waters. The magnitude of release of constituents to the water column and the duration of exposure that water column organisms could experience in the turbid plume that forms during the descent of the dredged sediments is such that no water column water quality problems would be expected.

The open-water disposal of hydraulically dredged sediments involving pipeline transport to the disposal site represents a significantly different situation than those found for dumping of mechanically or hydraulically dredged sediments in open waters. This situation is pictured in Fig. 14.9. The approximately 20 percent sediment slurry discharge from a hydraulic pipeline transport operation quickly forms a density current which moves along

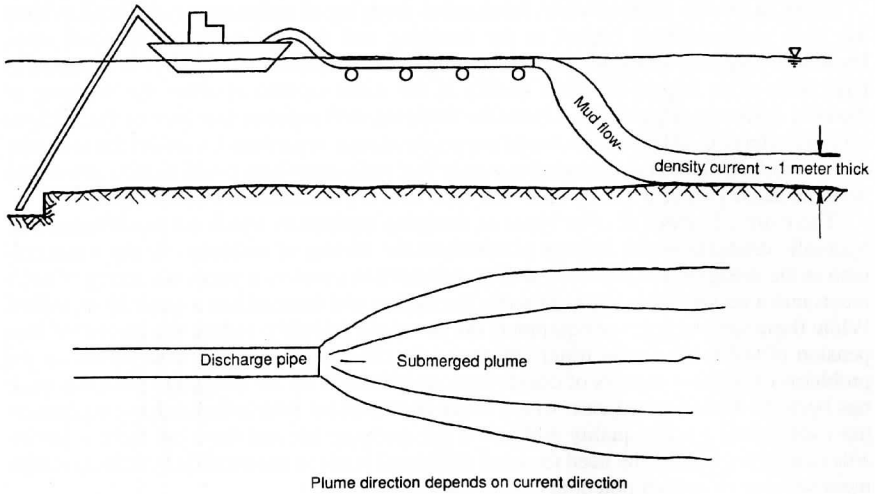


FIGURE 14.9 Hydraulic dredging—open-water pipeline disposal.

the bottom in about a 1-m-thick layer in the down-ambient-water-current direction from the discharge. This density current has been found to persist for thousands of meters from the point of discharge. It is typically devoid of dissolved oxygen and could, therefore, represent a significant adverse impact on aquatic organisms residing on the bottom in the path of the current due to low DO and/or the release of constituents such as ammonia. This would be especially true if the density current were to persist in one location for a period of many hours to a day or so. Ordinarily, however, such a situation does not occur due to mechanical problems which cause the dredging operation to have to be shut down. Further, due to the movement of the dredge, as well as accumulations of sediments near the point of discharge, a frequent need exists to change the point of discharge, thereby shortening the exposure time that organisms located on the bottom could experience due to hydraulic dredging and pipeline transport operations.

There is concern about whether the dredging of contaminated sediments causes the release of "pollutants" in the water column and, therefore, is significantly adverse to aquatic life and other beneficial uses of the waters in which dredging occurs. Studies by Lee, et al. (1978) showed that while the water column near a dredging site is often more turbid than the waters outside of the area, there is no significant release of chemical constituents to the water column associated with this turbidity that would be adverse to aquatic life through toxicity, bioaccumulation, etc. These results are in accord with the results of Lee, et al. (1978) and Jones and Lee (1978) elutriate test studies which showed that the stirring of the sediments into overlying waters did not release constituents other than ammonia and manganese II to the water column. There is typically sufficient dilution associated with dredging projects at the dredging site so that any ammonia release does not represent toxic conditions to aquatic life. Further, manganese II has a low toxicity to aquatic life.

Considerable public and regulatory attention is often given to the turbidity that is associated with dredging at the dredging site. Dredging projects are sometimes required to curtail dredging if turbidity becomes excessive. Further, many dredging projects are required to utilize silt curtains to reduce the spread of the turbid waters. While this turbidity is readily visible, other than a short-term aesthetic problem of turbid water, it would not be expected to be significantly adverse to the beneficial uses, including aquatic life, of the waters where the dredging is taking place.

From an overall point of view, mechanical dredging of sediments would tend to have the least environmental impact at the dredging and dredged sediment disposal sites. Hydraulic dredging with open-water dumping of the sediments would also be expected to have little or no impact on water quality in the water column at either the dredging or dredged sediment disposal site. Hydraulic dredging with pipeline transport of the dredged sediment slurry could have an adverse impact on aquatic organisms located at the sediment water interface due to the extended exposure that such organisms could receive associated with the density current arising from this type of dredged sediment disposal operation.

There are a number of other types of dredging equipment which are modifications of hydraulic dredging which attempt to minimize the stirring of sediments in the water column at the dredging site. There is also equipment that involves a vacuum sucking of sediments into a vessel which is then raised to the surface and dumped into a barge for transport. While these various types of equipment do, or would probably, reduce the amount of suspension of sediments in the water column at the dredging site and thereby minimize the problems of adverse impacts of constituents associated with the dredging operation, there has been no demonstrated case where using conventional hydraulic dredging equipment has ever caused a water quality problem at the dredging site and therefore there is justifiable reason to question the need to spend additional funds to use modified dredging equipment because of alleged potential environmental impacts.

UPLAND-"CONFINED" DISPOSAL OF DREDGED SEDIMENTS

While it has been commonly assumed that confined disposal of dredged sediments, which involves placing them on land or behind dikes to form an island in water, is less environmentally damaging, these assumptions are not necessarily correct. In fact, it is now beginning to be more widely recognized that at some locations upland disposal of contaminated dredged sediments has a greater potential for adverse environmental impact than open-water disposal of dredged sediments. As generally practiced today, "confined" disposal does not truly confine deposited sediment, but rather provides a settling area where the larger, more dense particles are removed and the water associated with the hydraulic dredging operation is allowed to enter the watercourse. With few exceptions, confined disposal operations have overflows to the nearby watercourse where any constituents released from the sediments during the dredging operation, as well as those constituents associated with the fine materials that are not removed in the disposal area, enter the water. Since it is the fine materials that often have the greatest affinity for constituents, if the constituents are or become available from the fines, then more ecological/water quality damage could result from so-called confined disposal than with open-water disposal, because the area where the confined disposal overflow occurs, i.e., nearshore, is generally the most ecologically sensitive area of the waterbody.

Studies conducted at the U.S. CE Waterways Experiment Station (Palermo, 1986) have shown that the sediments in a confined disposal area, which when wet retain the constituents such as heavy metals, upon drying tend to release these heavy metals. This appears to be related to the oxidation of the amorphous sulfides present in the sediments which are keeping the heavy metals in particulate form and the development of acidic conditions. It may also be related to the aging of the ferric hydroxide precipitate (hydrous oxides). While freshly precipitated ferric hydroxide has a substantial binding power for constituents, aged precipitates, especially those that dry out, lose some of this binding power. It is, therefore, not surprising to find that heavy metals associated with dredged sediments, when placed in a confined disposal area that is allowed to dry out, will show appreciable release of heavy

metals to the overflow waters the next time that dredged sediments are introduced into the area. This situation can also occur with atmospheric precipitation drainage and runoff from these areas. It is important that those who advocate upland disposal of contaminated dredged sediments conduct a proper, critical review of the adverse impacts of the constituents associated with the sediments which leave the confined disposal area during overflow during dredging operations or via drainage from the disposal site. pH changes from neutral to acidic probably have the most profound effect on metals release.

A dredged sediment management issue of concern in some areas is the regulation of the use of contaminated dredged sediments for flood control levee enhancement to improve structural stability and/or to develop a shallow water habitat for restoring/improving aquatic ecosystems. This is a major area of activity in the Sacramento River–San Joaquin River delta, as well as elsewhere. It is somewhat unique in that it can represent a mixture of open-water disposal and upland disposal. These types of dredged sediment management operations tend to have runoff associated with the dredged sediment placement, as well as from leaching from the dredged sediments placed in the water below the water line and from the sediments above the water line. Some of those interested in beneficial use of contaminated dredged sediments are experiencing significant impediments to their use for these purposes as the result of state and local regulatory agencies mechanically applying U.S. EPA Clean Water Act requirements that are designed for regulating municipal and industrial wastewater discharges to runoff/leaching from this type of dredged sediment management approach.

The regulatory approach being used requires that runoff associated with the beneficial use of dredged sediments not cause, or contribute to, an exceedance of a water quality standard at the edge of a defined mixing zone, if one is allowed, by any magnitude for a regulated parameter more than once every 3 years. The regulatory situation for this type of beneficial use for contaminated dredged sediments has a number of characteristics similar to the regulatory issues that are now being applied to urban area and highway stormwater runoff. When U.S. EPA worst-case–based water quality criteria are used as state standards, which is typically the case, it is found that this approach greatly overregulates many of the regulated constituents in urban area and highway stormwater runoff. The overregulation is also associated with some dredged sediment beneficial use projects. As discussed by Lee and Jones-Lee (1998*a*), the intermittent nature of urban area and highway stormwater runoff, coupled with the fact that most of the potential pollutants in such runoff are in particulate forms which are nontoxic/nonavailable, causes the runoff to be significantly overregulated when worst-case–based water quality criteria/standards are applied, either to the runoff at the point of entry into receiving waters or at the edge of a mixing zone, if one is allowed within the receiving waters for the runoff. The same situation occurs for the runoff/drainage from areas where contaminated dredged sediments have been used for levee enhancement or shallow water habitat development. Such runoff will likely be intermittent in nature and of short duration in which many of the potential pollutants in the runoff, such as heavy metals are in nontoxic, nonavailable forms. Further, and most importantly, with few exceptions, the runoff/seepage from such areas will be rapidly diluted in the receiving waters below critical concentrations to adversely impact aquatic life.

Rather than overregulating the beneficial use of contaminated dredged sediments by overly restrictive requirements, those interested in beneficial uses of dredged sediments should work with those concerned with waterway navigation maintenance, regulatory agencies, environmental groups, and members of the public to develop and implement site-specific studies to determine what, if any, real water quality use impairments are associated with the beneficial uses of contaminated dredged sediments for levee enhancement and/or shallow water habitat development. The overall approach that could be followed is that described by Jones-Lee and Lee (1998) that was originally developed for assessing the water quality impacts associated with urban area and highway stormwater runoff. This

approach, called evaluation monitoring, is based on a stakeholder-driven consensus of the studies that are needed to define whether there are real significant water quality problems associated with beneficial uses of contaminated dredged sediments. Where such problems are found, the stakeholders should develop a consensus on how best to control these problems to promote contaminated dredged sediment beneficial use without the significant unnecessary costs associated with this use. After a few stakeholder-developed demonstration projects are completed, it should be possible to significantly reduce the amount of monitoring/evaluation needed to gain approval for beneficial use of contaminated dredged sediments.

NEW AND PENDING REGULATIONS

The current U.S. EPA regulation governing disposal of dredged sediments in freshwater, estuarine, and marine waters are badly out of date and require a number of technically invalid approaches to be used to assess the potential water quality significance of chemical constituents in the sediments. A need exists for the U.S. EPA to update the regulations governing both the ocean disposal and inland water disposal of contaminated dredged sediments to reflect the large amount of information that has been developed over the past 25 years on the impact of such sediments and associated constituents on water quality as well as the approaches that should be followed to evaluate this impact on a site-specific basis. There are also significant inconsistencies in the regulatory approaches governing the ocean disposal of dredged sediments and disposal in "inland" waters of the United States.

The current regulations governing the ocean disposal of dredged sediments prohibit disposal of contaminated sediments that could have a long-term adverse impact on the numbers and types of organisms present at the disposal site. Short-term impacts due to physical effects, such as sediment grain size, etc., are allowable. Long-term impacts, such as those that might be associated with persistent chemical constituents, are prohibited. There is no significant justification for such an approach. Ocean disposal of dredged sediments occurs at designated disposal areas. Such areas are well-defined and, contrary to the current U.S. EPA regulations, there should be no requirement that the dredged sediment–associated constituents have no impact on the numbers and types of aquatic organisms present within these areas. It has been well established that changing the grain size of the uncontaminated sediments can have a dramatic impact on the numbers and types of benthic organisms present in an area. The impact on the numbers and types of organisms within the designated disposal area due to chemical constituents should also be considered acceptable. Such impacts can readily occur without having a significant adverse impact on the near-shore marine water quality and aquatic life resources of the area.

One of the areas of greatest concern in the regulations governing ocean disposal of dredged sediment is the limitation on the toxicity of these sediments to aquatic life. The approach that should be followed in sediment toxicity tests is to evaluate the real or potential impact at the edge of the designated disposal area, where the dredged sediments will be mixed with the natural sediments of the region. Dilution of contaminated sediments with small amounts of noncontaminated sediments typically greatly reduces the toxicity found for the contaminated sediments.

If large amounts of toxicity are found under laboratory test conditions which simulate field conditions, then additional studies should be done to determine whether the simulated conditions of the laboratory tests appropriately mimic the field conditions with respect to dilution of the sediments and the dilution of any constituents released from the sediments with the waters of the type that would occur at the edge of the designated disposal area. It is only with testing of this type that a proper evaluation can be made of what is an allowable

percent mortality of the test organism in a sediment toxicity test which is needed to protect the designated beneficial uses of the waters outside of a designated disposal area.

Reference Sediment

The U.S. EPA and U.S. CE (1991, 1998) in their testing manuals governing the disposal of dredged sediments make use of a reference sediment for testing purposes. This reference sediment approach is part of the U.S. EPA's regulations governing the disposal of dredged sediments. This approach arose out of the regulatory requirements that dredged sediment disposal shall not cause a significant adverse impact at the disposal site. By comparing the dredged sediment impact to that of a reference site was determined to be the best way to implement current regulatory requirements. As currently implemented, the same tests are performed on the reference sediments as on the sediments scheduled to be dredged. However, the behavior of a constituent in a reference sediment, as well as the toxicity to or bioaccumulation in an organism exposed to a reference sediment, cannot be used to evaluate the water quality impact that a particular dredging or dredged sediment disposal operation will have on water quality beneficial uses of the area. The current regulatory requirements need to be changed to focus on absolute impacts of disposal of dredge sediments at a particular location rather than the relative impact on some reference sediment.

Sediment Quality Criteria/Guidelines

Until recently, for about 10 years, the U.S. EPA attempted to develop numeric sediment quality criteria as a means of regulating the water quality impact of chemical constituents that become associated with aquatic sediments. Basically, this effort is an outgrowth of the Jensen criteria discussed earlier, in which the bulk chemical composition or some modification is used to determine "excessive" concentrations of chemical constituents in sediments. While it is well known that numeric criteria based on the bulk chemical composition of sediments is technically invalid, since there is no relationship between total concentration of a constituent and toxic/available forms, the U.S. EPA attempted to find a way to normalize sediment chemical data so that a single numeric value can be developed which can be used to determine "excessive" concentrations of constituents in sediments.

One of the normalization procedures that was aggressively pursued by the U.S. EPA (1993*b*) was the Equilibrium Partitioning (EqP) approach in which the amount of organic carbon present in the sediments is used to estimate the detoxification of the sediments for nonpolar organic chemicals such as chlorinated hydrocarbons and PAHs. This approach involves the use of U.S. EPA water column criteria to determine excessive concentrations of constituents in the waters in equilibrium with the sediments based on a sediment-water partition coefficient. This approach has considerable merit in explaining why chemicals of this type are not toxic in sediments. It has limited utility, however, in predicting whether a sediment will be toxic or not since there are a variety of factors other than TOC that influence the toxicity/availability of chemical constituents in sediments.

Another approach being used to normalize constituent concentrations in sediments in an attempt to try to develop numeric criteria is the use of acid volatile sulfides (AVS). DiToro, et al. (1990) reported that when heavy metals such as cadmium, zinc, copper, and lead, are present in sediments that have, on a molar basis, more simultaneously acid-extractable sulfides in the sediments than noniron heavy metals, the heavy metals are nontoxic to aquatic organisms. As discussed previously, this is to be expected since these heavy metals form insoluble sulfides which should be nontoxic. While sulfide precipitation can be used to explain why heavy metals present in sediments are nontoxic, a variety of other chemical

reactions occur with heavy metals that cause them to also become nontoxic. Therefore, trying to establish numeric criteria for heavy metals in sediments based on their concentrations relative to the sulfide concentrations of the sediments, as the U.S. EPA originally proposed to do, can readily lead to significant overestimates of toxicity of the heavy metals to aquatic life. Recently, the U.S. EPA has indicated that, while excess heavy metals over sulfides cannot be used to predict toxicity of the metals, excess sulfides over the metals can be used to predict nontoxic metals.

Another approach being investigated for possible use to judge sediment quality based on chemical analyses is through the use of the chemical characteristics of sediment pore (interstitial) waters. The aquatic chemistry of interstitial pore water is significantly different from the water from which benthic aquatic organisms typically obtain exposure to chemical constituents. Almost all interstitial water associated with chemical constituents is anoxic, i.e., oxygen-free, and therefore is not a suitable environment for aquatic organisms which depend on oxygen for respiratory purposes. Further, many of the aquatic organisms that burrow into the sediments develop protective tubes which tend to isolate them from the sediments and interstitial water-associated constituents.

There is no relationship between the concentration of interstitial water-associated constituents and the concentrations that would be present at the sediment-water interface. Typically, the concentrations in interstitial waters are much higher than that present at the sediment-water interface, where most organisms would be exposed to the constituents of potential concern. This situation is the result of several factors, one of which is dilution that occurs at the sediment-water interface. Another factor is those anoxic interstitial pore waters typically have elevated concentrations of iron in the ferrous form which upon contact with oxygen in the overlying waters is oxidized to ferric iron and precipitates as ferric hydroxide. The ferric hydroxide is a known, efficient scavenger for most constituents, making them less available if not unavailable for organism uptake. While interstitial water chemical measurements, if done properly, where the waters are not exposed to oxygen, would yield information on the "dissolved" and colloidal forms of constituents in sediments and, therefore, represent better estimates of potentially toxic forms, the likelihood of developing reliable chemically based assessments of aquatic life toxicity, based on interstitial water measurements, is small.

The complexity of aquatic sediments as chemical systems requires that an effects-based approach be used to assess whether chemical constituents in the sediments could have an adverse impact on aquatic life and other beneficial uses of the nearby waters. As discussed by Wright (in Chapter 13), as well as Wright, et al. (1992) and Wright and Wilson (1996), the U.S. EPA–U.S. CE evaluation procedures for determining the suitability of open-water disposal of contaminated sediments are, to a considerable extent, effects-based, e.g., sediment toxicity testing and bioaccumulation. Rather than trying to utilize arbitrarily developed chemically based numeric criteria, where exceeding a particular numeric value is related in some way to the concentration of constituents in sediments, the U.S. EPA–U.S. CE chose to directly measure the potential impacts such as measuring toxicity, bioaccumulation, etc. This is the most technically valid approach and the one that should be followed. It provides a much more appropriate estimate of the potential impact of constituents in sediments than chemically based sediment quality criteria/guidelines.

The use of total organic carbon and/or sulfides as parameters for normalization of chemical concentrations can both under- and overestimate the toxicity that a sediment can have to aquatic life. The studies by Jones and Lee (1978) provided a database on the chemical characteristics of sediments from many different U.S. waterways. A review of this database shows that, typically, sufficient sulfides exist in the sediments to precipitate all the noniron-heavy metals and sufficient organic matter exists in the sediments to bind all the nonpolar organic chemicals so that neither of these groups of chemicals should be toxic. However, there is also sufficient ammonia in many of the sediments to cause toxicity to some forms

of aquatic life. The U.S. EPA's approach of trying to develop sediment quality criteria based only on acid volatile sulfides and organic carbon can lead to erroneous conclusions on the potential impact of the constituents in the sediments to aquatic life. For many sediments it will underestimate the detoxification of sediment constituents for heavy metals and nonpolar organic chemicals. Further, it does not consider the role of ammonia as a toxicant in the sediments to aquatic life. The reliable way to proceed is through an effects-based testing program using sediment bioassays to directly assess potential toxicity. It is concluded that numeric sediment criteria of the type that the U.S. EPA has been developing, including the equilibrium partitioning TOC and acid volatile sulfide normalization approaches, have limited reliable applicability to assessing the real impact that a dredging or dredged sediment disposal will have on the beneficial uses of the waters in the vicinity of where the dredging and disposal activities take place.

Empirical Sediment Quality Guidelines—Cooccurrence-Based Values

At several locations in the United States, especially in the Puget Sound, Washington, area, regulatory agencies have been attempting, for a number of years, to use empirically based sediment criteria to regulate dredging and dredged sediment disposal activities. The U.S. EPA (1997*b,d*; 1998), as part of its National Sediment Inventory and its Contaminated Sediment Management Strategy, has used sediment quality guidelines that are based on cooccurrence-based approaches for relating the total concentrations of certain constituents in sediments to sediment toxicity. This approach could be of great significance to many dredging projects for waterways near urban-industrial centers where, because of the use of inappropriately based sediment quality guidelines, the potential impact of constituents associated with dredged sediments will be overestimated, causing alternative, usually more expensive, methods of dredged sediment management to be adopted. Wright, et al. (1992) and Wright and Wilson (1996) have discussed the problems with attempting to use numeric chemical criteria for regulating dredging projects. These reviews on this topic should be consulted for further information on it.

Some of the approaches being used to attempt to judge sediment quality and therefore regulate dredging projects include the numeric Triad and the Apparent Effects Threshold (AET) approaches. These approaches are similar in that they make use of a combination of bulk sediment chemical characteristics, benthic organism community structure (numbers and types of organisms present), and sediment bioassays to develop numeric values which are purported to be an index of sediment quality. Basically, both of these approaches are formulated on technically invalid assessments of the potential impact of sediment-associated chemical constituents on aquatic organisms. They make use of bulk chemical analyses as a foundation for assigning a numeric value to a sediment. Further, the AET numeric values used in the Puget Sound, Washington, area are based, to some extent, on the use of the Microtox testing of sediment extracts, using the effect of the extracts on photoluminescent bacteria as a measure of toxicity to aquatic life. This procedure has been known to be unreliable for evaluating aquatic life toxicity for many different aquatic systems. It is inappropriate to assume, as has been done in the AET approach, that the Microtox measured toxicity is related to the toxicity that would be present for aquatic organisms of interest in fresh or marine systems.

Another problem with the AET approach is that it is assumed that the numbers and types of organisms present in a sediment are controlled by the chemical constituents present. It is well known that a variety of factors, such as grain size and organic matter content, can significantly affect the numbers and types of organisms present in a sediment.

Advocates of the AET approach for the evaluation of sediment quality in the Puget Sound, Seattle, Washington, area claim that the use of the AET approach has been found

to be successful in regulating dredging projects. The fact that the AET approach has been used ("successfully applied") to make management decisions, does not mean that the approach is reliable or appropriate. A critical examination of how AET has been used in regulating waterway dredging projects in that area shows that it is not a reliable basis upon which to determine the potential water quality significance of chemical constituents in sediments as they may impact dredged sediment disposal. Sediments that pass the AET screen used and are thus exempt from further testing, could be toxic or otherwise significantly adverse to aquatic life at a dredged sediment disposal site due to constituents present in the sediment that are not considered in the AET evaluation.

Those sediments that do not pass the AET screen are subject to the more technically reliable biological effects-based testing in accord with U.S. EPA and U.S. CE (1991, 1998) procedures. It would be far more reliable and cost effective to forgo the unreliable AET screen and proceed directly with appropriate biological effects-based testing to determine whether measured, as well as unmeasured, constituents in sediments are potentially toxic to aquatic life.

The AET and numeric Triad values which are supposed to reflect sediment quality do not provide a proper evaluation of the potential significance of constituents in sediments in affecting water quality and aquatic life resources at a dredging or dredged sediment disposal site. Therefore, these approaches have little technical validity in regulating the impacts that chemical constituents associated with sediments can have on the water quality and other beneficial uses of the waters of the area where the sediments are found or are deposited in a dredging project.

In the early 1990s, the state of California Water Resources Control Board Bay Protection and Toxic Clean-up Program (BPTCP) staff proposed to develop sediment quality criteria based on the geometric mean of the AET, sediment spiking toxicity test, and the equilibrium partitioning approach. This numeric value is to be multiplied by an "uncertainty factor" to develop an overall numeric value that would be used to characterize the so-called sediment quality as part of designating toxic "hot spots" in the state's waters. As discussed by Lee and Jones (1991), this approach is not technically valid for properly characterizing the water quality significance of constituents in sediments. It can lead to a misclassification of sediment quality since it has in it a number of components which are known to be invalid approaches, such as bulk sediment chemical analyses, spiked sediment bioassays, etc., which are not related to the impact of the constituents in the sediments on the beneficial uses of the waters in the vicinity of the sediments.

Lee and Jones (1991) recommended, rather than using an empirical approach for assessing sediment quality, that the state of California should adopt an effects-based assessment of the potential water quality significance of constituents in sediments utilizing sediment toxicity test, aquatic organism assemblage information, and bioaccumulation of hazardous chemical constituents from the sediments. In 1997 the state of California BPTCP staff abandoned their sediment quality criteria development approach in favor of a narrative approach in which a nonnumeric sediment quality triad is used to characterize sediment quality. This triad is composed of sediment toxicity, organism assemblage, and chemical information to assess whether chemical constituents in the sediments are significantly adverse to the beneficial uses of a waterbody in which the sediments are located to designate the area as a toxic hot spot and require sediment remediation and/or the control of the input of the constituents that accumulate in sediments to "excessive" levels.

As discussed by Lee and Jones-Lee (1994), a number of individuals have compiled information on the concentrations of specific chemicals in sediments, by constituent, and the results of some type of assessment of biological response to those sediments. The "toxic responses" reported are responses to the wide variety of constituents that can be present in a sediment, and not necessarily a response to the specific constituent being considered. Thus, independent of the cause of the biological response, the same degree of response was

associated with the concentration of each individual chemical constituent measured in a sediment. Tables of such "cooccurrences" of the concentration of a chemical and a "response" developed by Long and Morgan (1990), Long, et al. (1995, 1998), Long and MacDonald (1998), and MacDonald, et al. (1992) have formed the foundation of what are termed cooccurrence approaches for sediment quality evaluation. For a given chemical, the sediments are ranked by concentration and presented with the associated assessment of the "effect" of the sediments, measured in any number of ways, without consideration of what conditions or constituents were causing or influencing the "effect." In regulatory applications, cooccurrence information has been used, albeit incorrectly, to establish various "effects threshold" values based on statistical manipulation. Fundamental, but refutable, presumptions made are that there is a causal relationship between the concentration of each constituent considered in a sediment and the water quality impact of that sediment (as were bulk sediment criteria), and that the "effect" reported for each sediment was caused independently by each of the measured chemical constituents in that sediment.

Examples of cooccurrence applications are the Apparent Effects Threshold (AET), the ER-L and ER-M values developed from Long and Morgan's (1990) data presentation, and the Probable Effects Levels (PEL) values derived from MacDonald, et al.'s (1992) cooccurrence compilations. If a sediment contains a chemical in concentrations above the AET, ER-M, PEL, or similar value, the sediment is considered by some regulators or in proposed regulations to be "polluted," and to require special consideration such as "remediation," alternate methods of dredged sediment disposal, or additional control of discharges of the chemical to the waterbody. While administratively simple to implement, this approach is not technically valid for assessing the potential impact of sediment-associated constituents and thus should not be used to regulate the dredging of contaminated sediments; it does not reliably consider the aqueous environmental chemistry of chemicals in aquatic sediments as they may impact aquatic life-related beneficial uses of waterbodies. Lee and Jones-Lee (1993, 1994, 1996*c,d,e*) have reviewed the significant technical deficiencies in the use of cooccurrence-based sediment quality criteria to evaluate and manage contaminated sediments. As discussed, a number of individuals and groups have evaluated the reliability of the Long and Morgan cooccurrence sediment-based guidelines where they have found that flipping a coin is a more reliable method of predicting sediment toxicity than exceeding the guideline values.

O'Connor (1999*a,b*) has recently reviewed the National Oceanic and Atmospheric Administration (NOAA) Status and Trends database, as well as U.S. EPA EMAP database, with reference to the reliability of the cooccurrence-based sediment quality guidelines and concluded that,

> All these criteria are better than random selections in identifying toxic sediment but they are not reliable. They are all more often wrong than right and should not be used, by themselves, to imply anything about biological significance of chemical data.

In an attempt to try to improve the reliability of this approach, advocates of cooccurrence approaches are developing quotients of Long and Morgan guideline values, where multiple parameters are considered in an attempt to establish the sediment quality. While this approach tends to provide greater statistical reliability in predicting toxicity than individual Long and Morgan or MacDonald guideline values, it still suffers from the inherent, fundamentally flawed basis of assuming a cause-and-effect relationship between the total concentration of a constituent considered by Long and Morgan in developing their cooccurrence table and the biological effect reported for the sediment. The biological effects found could readily be due to a nonconsidered parameter, such as ammonia, which is one of the most common causes of sediment toxicity to certain forms of aquatic life.

One of the most inappropriate examples of the use of Long and Morgan cooccurrence-based values in characterizing sediment quality is the approach adopted by the state of California Water Resources Control Board BPTCP staff (WRCB, 1998) in which these values are to be used as the chemical component of the sediment quality triad for designating toxic hot spots in the state's waters. The exceedance of one or more guideline values is proposed by the BPTCP staff as reliable information for developing an "association" between a chemical constituent in the sediment and the cause of the toxicity, bioaccumulation, or altered organism assemblages. Such an approach, while easily administered, is obviously technically invalid, where responsible parties can be designated to help pay for toxic hot spot cleanup in the state's Water Resources Control Board's aquatic sediment Superfund–Aquafund. Further, the responsible party could have their NPDES permit modified to control the input of the constituents to the waterbody that accumulate in sediments that cause an exceedance of a Long and Morgan cooccurrence-based value. As discussed by Lee and Jones-Lee (1998*b*) and Lee (1998), ultimately public and private interests could be trapped into spending large amounts of money in the name of water pollution control that will have little or no impact on the beneficial uses of the waterbodies where the Long and Morgan values have been used to associate a chemical constituent as the cause of a toxic hot spot.

The BPTCP contains specific restrictions on dredging in areas where the sediments are designated as a toxic hot spot. The BPTCP staff's proposed approach of using the Long and Morgan cooccurrence-based guideline values to associate an impact with a particular chemical could greatly increase the cost of dredging in some areas, such as San Francisco Bay, since the whole bay has been designated as a toxic hot spot due to excessive concentrations of mercury and certain chlorinated hydrocarbons, such as PCBs, in fish.

Lee and Jones-Lee (1998) and Lee (1998) have recommended that the chemical component of the sediment quality triad be based on aquatic chemistry evaluations, not total chemical concentrations, where, through the use of sediment-based toxicity investigation evaluations (TIEs), site-specific studies are conducted to determine whether a particular constituent in the sediment is the cause of the toxicity. Further, the TIE approach can be used to identify the cause of toxicity, and thereby properly associate, through true cause and effect, the chemical constituent responsible for the impact.

CONCLUSIONS

U.S. waterway sediments, especially near urban-industrial centers, were found in the late 1960s to contain large amounts of chemical constituents. If these constituents are released in the waterway or during dredging/disposal, they could have significant adverse effects on the beneficial uses of the water in the waterway, near open-water disposal sites, as well as in areas where confined disposal overflow occurs. Studies conducted during the 1970s under the auspices of the U.S. CE Dredged Material Research Program reaffirmed that bulk sediment criteria based on total constituent concentration are not technically valid for judging the toxicity of sediment-associated constituents to aquatic life. Elutriate tests and dredged sediment bioassay-toxicity tests provide an approach for evaluating the potential water quality significance of constituents associated with dredged sediments. Significant problems have developed with the use of these procedures in making dredged sediment disposal management decisions because of problems in the interpretation of the results of the tests. A hazard assessment approach, such as described in this chapter, should be used for this purpose, in which the hazard that the constituents present in the dredged sediment represent to the beneficial uses of the water at the disposal site is evaluated. A need exists to update the U.S. EPA regulations governing dredged sediment disposal in fresh and marine water systems to properly consider the vast amount of information that was generated since 1970 on the water quality significance of constituents associated with sediments.

Caution must be exercised in using chemically based approaches such as the cooccurrence approaches for assessing the water quality significance of chemical constituents in aquatic sediments. Such approaches can readily increase the cost of dredging associated with navigational channel maintenance with little or no water quality benefits. The biological effects-based approaches should continue to be used to regulate the dredging and disposal of contaminated sediments.

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