US Army Corps
of Engineers
Waterways Experiment
Station

Environmental Effects of Dredging

Section 01 - Aquatic Disposal Technical Notes EEDP-01-1 through EEDP-01-18

Compiled by

Dredging Operations Technical
Support Program

Section 01—Aquatic Disposal
Technotes EEDP-01-1 through EEDP-01-18

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EEDP-01-1
June 1985

Environmental Effects of Dredging Technical Notes

BIOMAGNIFICATION OF CONTAMINANTS IN AQUATIC FOOD WEBS AS A RESULT OF OPEN-WATER DISPOSAL OF DREDGED MATERIAL

PURPOSE: This note provides information regarding the potential extent of biomagnification (the tendency for contaminant concentrations in animal tissues to increase through successively higher trophic levels) of contaminants in aquatic food chains resulting from the open-water disposal of contaminated dredged material. The note also provides a technically sound perspective and offers general technical guidance on assessing the environmental importance of biomagnification in aquatic food chains as a result of open-water disposal of contaminated dredged material. It does not consider biomagnification in nonaquatic organisms.

BACKGROUND: Disposal of dredged material in open water is used extensively by the Corps of Engineers. Pesticides and pesticide residues, nutrients, organic wastes, heavy metals, and other contaminants entering waterways may associate strongly with particulate materials and eventually accumulate in the sediments. The presence of potentially toxic contaminants in some sediments has generated concern that dredging and open-water disposal of contaminated dredged material may cause the deterioration of the aquatic environment. It is felt that persistent chemical residues from the dredged material may accumulate within the tissues of aquatic plants and animals to levels that are in excess of the ambient concentrations in their environment. Most of these substances have no known biological function, and there is concern that some may accumulate to levels that could affect the growth, reproduction, or survival of the organism or its predators.

Although well documented in terrestrial ecosystems, the occurrence and extent of biomagnification in aquatic ecosystems is questionable and is the topic of considerable debate. In 1983, extensive independent literature reviews were prepared by the Corps of Engineers (Kay 1984) and the Environmental Protection Agency (Biddinger and Gloss 1984) to assess the magnitude of contaminant biomagnification in aquatic ecosystems. The Corps literature review was conducted as part of the Long-Term Effects of Dredging Operations (LEDO) Program.

There were some minor differences between the two reviews, but both reached very similar conclusions regarding biomagnification of contaminants in aquatic food webs. The findings of these literature reviews provide the basis of this Technical Note on biomagnification as a potential contaminant mobility problem originating from the open-water disposal of contaminated dredged material. The ecological consequences of any presumed biomagnification of contaminants are beyond the scope of this note. See Dillon (1984) for information regarding the consequences of contaminant accumulation in aquatic animals.

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The Phenomenon of Biomagnification

Many chemicals are present in the environment in extremely low concentrations, frequently near or below the levels readily detectable by routine analytical techniques. Living organisms may accumulate these chemicals to levels greatly in excess of the ambient concentrations in their environment. The ability to accumulate substances from the environment is biologically significant, for this is how living organisms obtain these substances commonly designated as "essential nutrients." However, nonessential chemicals (e.g. trace substances) also may be accumulated from the environment by natural biological processes. These substances have no known biological function and can accumulate to levels that may be detrimental to the organism.

Trace substances may enter living organisms in several ways. Both aquatic plants and animals accumulate trace substances by bioconcentration (direct adsorption and absorption from the sediments and water). Animals also accumulate trace substances by ingestion. The total process of accumulating substances by both ingestion and bioconcentration is called bioaccumulation. Occasionally, the concentrations of trace substances in living organisms continue to increase as the substances are passed on from lower to higher trophic levels. This phenomenon is called biomagnification.

The relative importance of food and bioconcentration as pathways for entrance of trace contaminants into aquatic organisms is the subject of considerable debate. The predominant route of entrance of a contaminant into a living organism depends on the nature of the environment inself and the relative level of exposure in the food and the external environment. Food becomes the primary source for contaminant accumulation only when bioconcentration from the external environment is minimal. Food-chain biomagnification as the result of dietary intake of contaminants is said to occur if the concentration of a substance increases at each successively higher trophic level as the result of dietary intake of food (prey) by a consumer (predator).

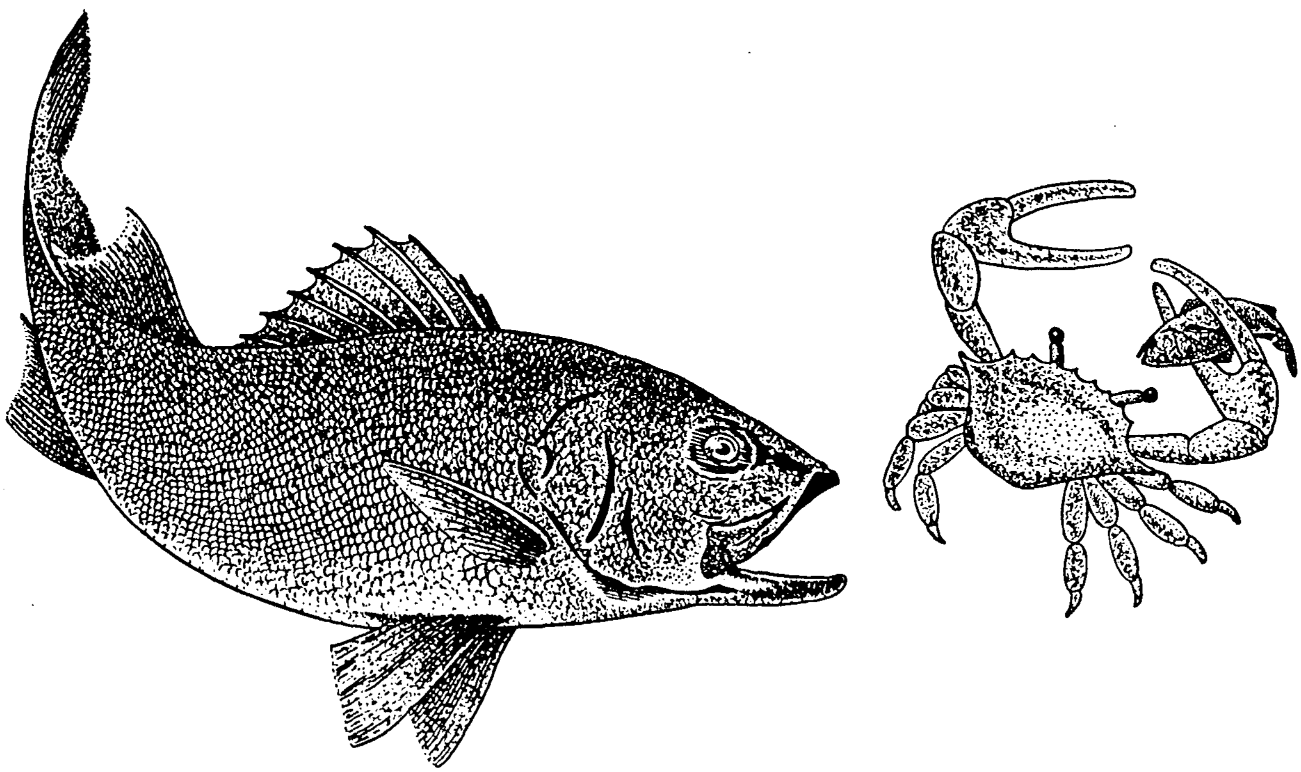
Biomagnification of contaminants may occur when all of the following conditions are met:

The chemical is persistent in biological systems (Macek 1970).

Direct uptake from the external medium in minimal.

The food pathway is essentially linear and highly structured, and the predominant energy flow is from lower to higher trophic levels.

Most aquatic (freshwater and marine) food webs are rather weakly structured, however, and do not have trophic levels as clearly defined as those of terrestrial systems. One species may occupy several trophic levels during its lifetime due to different feeding habits at different stages in its life cycle. Opportunistic omnivores also feed upon organisms occupying several trophic levels. Energy flow in aquatic food webs is multidirectional (for example, crabs are both prey and scavengers of fish), and a large component of the energy in aquatic systems is bound within the detritus.



Aquatic systems also rarely meet the criterion of minimal uptake from the external medium. Contaminant levels in the water may be low, but are usually higher than levels found in the atmosphere. In comparison to terrestrial animals (terrestrial is extended to include all animals that breathe air via lungs; shorebirds and "aquatic" mammals are considered as a special case of terrestrial animals living partially or wholly in water and are not covered herein), aquatic (water-breathing) animals have large respiratory areas in proportion to body size. The solubility of oxygen in water, especially seawater, is low. Therefore, ambient oxygen available for respiration is substantially less for most water-breathing aquatic animals than for their air-breathing counterparts. Large quantities of water must be passed over their gill surfaces to provide adequate oxygen for respiration, simultaneously increasing the uptake of other essential and nonessential substances from the surrounding medium. The body integuments (coverings) of aquatic animals, especially invertebrates, are usually more permeable than the integuments of terrestrial animals, allowing chemicals to pass readily into and from their tissues.

The combination of intimate physical contact with the external medium, due to relatively permeable body surfaces and respiration via gills, and a complexly interactive trophic web has led to the conclusion that trace contaminants probably do not increase nearly as much with trophic levels (i.e., biomagnify) in aquatic systems as in nonaquatic systems (Isaacs 1975). Thus diet generally is thought to be of minor importance as a source of most contaminants in the aquatic food web (Scura and Theilacker 1977; Macek, Petrocelli, and Sleight 1979; Narbonne 1979).

Summary of Findings of the Literature Reviews

Heavy metals

The majority of the data reviewed by Kay (1984) and Biddinger and Gloss (1984) indicated that most heavy metals except methylmercury do not biomagnify either in freshwater or marine food webs. A review of field and laboratory studies (Kay 1984) showed that food may be an important source for the bioaccumulation of toxic heavy metals, particularly those that are essential trace elements (copper, zinc, and selenium), but also some that have no known metabolic functions (chromium, arsenic, cadmium, mercury, and lead). These elements may be taken up from food, but do not biomagnify to any extent from one trophic level to the next within the food web. Concentrations of these elements generally were higher in the tissues of benthic herbivores and detritivores and, occasionally, planktivores than in the top-level carnivores.

In the case of methylmercury, laboratory evidence reviewed by Kay (1984) suggested that biomagnification would not occur, but was contradicted by the majority of the field studies, which indicated biomagnification. Both the Corps and the EPA reviews found that methylmercury has an affinity for muscle and tissues and apparently is biomagnified through the trophic web to the top predators. Consequently, higher, although not necessarily harmful, concentrations of methylmercury frequently are found in the large commercially valuable fishes than in invertebrates. However, the magnitude of increase from low trophic levels to high is on the order of one to ten times, not tens or hundreds of thousands of times as may occur in nonaquatic food webs. There is no satisfactory explanation for the contradictory results of laboratory and field studies with respect to methylmercury biomagnification.

Kay (1984) noted that inorganic mercury does not appear to biomagnify in aquatic food webs. Biddinger and Gloss (1984) also indicated that selenium and zinc might biomagnify; Kay (1984) noted that food was an important source for both metals, but did not indicate biomagnification.

Organic compounds

Food chain studies indicate that diet may contribute to the body burdens of a number of chlorinated and nonchlorinated organic compounds present in aquatic animals. Kay (1984) concluded that those compounds which appear to have potential for biomagnification in aquatic food webs were the polychlorinated biphenyls (PCB), kepone and mirex, benzo[a]pyrene, and naphthalenes. Biddinger and Gloss (1984) agreed on PCBs and added DDT to the biomagnification list. Kay (1984) found no strong evidence for biomagnification of DDT in water-breathing animals. However, where biomagnification occurred, it produced concentrations on the order of one to ten times higher in the upper trophic levels than in the lower ones, in contrast to the tens or hundreds of thousands of times higher as has occurred in such nonaquatic food webs as those involving DDT in fish-eating birds (Kay 1984).

As in the case of the heavy metals, the data on these organic contaminants sometimes were contradictory. Although top predatory fishes often contained higher levels of specific contaminants than other members of the food web, the relationship between contaminant levels in the tissues and an organism's position in the food web was not clear. The apparent contradictory nature of the data may reflect a number of factors, including the mobility of the top predators, age and size differences, inadequate understanding of the feeding habits of different species, particularly with respect to the changing of feeding habits at different stages of the life cycle, imprecision in the assignment of trophic levels, and inadequate sampling and analytical procedures.

The most obvious finding of both reviews was that few organic compounds appear to biomagnify; however, relatively little information was available regarding the behavior of many of the compounds in aquatic food webs. Consequently, any absolute statement regarding the occurrence of biomagnification of these contaminants must be reserved until further data are available.

Conclusions and Implications

The literature reviews were prepared independently, almost simultaneously, and covered a similar range of heavy metals and organic compounds, and reached similar conclusions. The information from the reviews was consistent with the findings of other investigators (Isaacs 1975; Scura and Theilacker 1977; Macek, Petrocelli, and Sleight 1979; Narbonne 1979).

The available information indicates that biomagnification of contaminants is not a dramatic phenomenon in marine and freshwater food webs. Without doubt, most heavy metals and organic compounds do not biomagnify substantially over several trophic levels in obligate aquatic food webs. Kay (1984) and Biddinger and Gloss (1984) agreed that those few contaminants that may have the potential to biomagnify definitely included methylmercury and PCBs; and that selenium, zinc, benzo[a]pyrene, DDT, naphthalenes, kepone, and mirex may possibly biomagnify. The apparent biomagnification of these contaminants in aquatic food webs usually is by small factors (1, 2, 3, 3, etc.) rather than by orders of magnitude (10, 100, 1000, etc.) from the lowest to highest trophic levels.

It is considered unlikely that dredging of contaminated sediment and immediate placement in an open-water disposal area would cause any significant long-term changes in the chemical characteristics of the sediments or substantially alter the bioavailability of the contaminants in the sediment. Contaminant uptake from sediments and mobility within the aquatic food chain should be similar regardless of whether those sediments were left undisturbed or were dredged and placed in an open-water disposal site. Therefore, based on existing literature, it appears unlikely that the open-water disposal of contaminated dredged material will cause any widespread ecological perturbations due to contaminant biomagnification in aquatic food webs. Further concern and expenditure for research on contaminant biomagnification originating from open-water disposal of contaminated dredged material appears to be unjustified.

Further attention should be given to evaluating biomagnification in food webs that include both aquatic and nonaquatic components, which was beyond the scope of these reviews. When food webs have major components in both aquatic and nonaquatic environments, such as the case of birds feeding on fish, biomagnification by large factors is possible and deserves serious consideration and evaluation. Placement of contaminated dredged material in a wetland or upland environment could impact associated nonaquatic portions of food webs.

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EEDP-01-2
September 1986

Environmental Effects of Dredging Technical Notes

FATE OF DREDGED MATERIAL DURING OPEN-WATER DISPOSAL

PURPOSE: This note summarizes published information on suspended solids transport into the water column during dredged material disposal by barge and hopper at open-water sites. The note provides an overview of field data referenced in the more widely quoted studies on open-water disposal and compares collection methods and results. The importance of using mass units of measurement rather than only volumetric units in accounting for the fate of dredged material is also discussed.

BACKGROUND: The many unknowns associated with the processes and impacts of open-water disposal of dredged material and the resulting environmental concern led to restrictions on the use of aquatic disposal sites in the late 1960s and early 1970s. This concern, however, fostered an expanded interest in research on the subjects, including a number of interrelated work units under the Corps' Dredged Material Research Program (DMRP). One of the principal focuses of the DMRP and later studies was the nature and effects of suspended solids (usually as turbidity) associated with dredging and disposal operations. Certainly no aspect of the subject was resolved completely, but considerable progress was made in the 1970s in describing, quantifying, and modeling the turbidity at disposal sites.

The use of open-water disposal sites subsequently increased, and turbidity has been less frequently cited as a concern in project planning. However, new questions are appearing concerning the movement of contaminated dredged material during disposal by surface release from barges and hoppers. Since contaminants are typically bound to the solid phase of sediment (particularly the fine-grained fractions), an understanding and predictive capability of the movement of these particles as suspended solids can lead to insight into the fate of the contaminants. This note will help to guide the direction of present and future investigations into contaminant fate by providing a state-of-the-science review of the literature and published data. Efforts were made to be thorough in the listing of studies and to use original references as sources. However, if there have been any omissions, the author (Clifford L. Truitt, Coastal Engineering Research Center) would welcome additional references.

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Overview of the Disposal Process and the Nature of Suspended Solids

Disposal process

The mechanics of the behavior of dredged material placed at an open-water site by instantaneous discharge from a barge or hopper have been described and/or modeled by a number of investigators (Clark et al. 1971, Koh and Chang 1973, Gordon 1974, Brandsma and Divoky 1976, Johnson and Holliday 1978, Bokuniewicz et al. 1978, and others). These descriptions typically divide the behavior of the material into three distinct transport phases or stages generally according to the physical forces or processes that dominate during each period. The most common terminology in use today for these stages is convective descent, dynamic collapse, and long-term or passive diffusion. Figure 1 is a schematic representation of these stages.

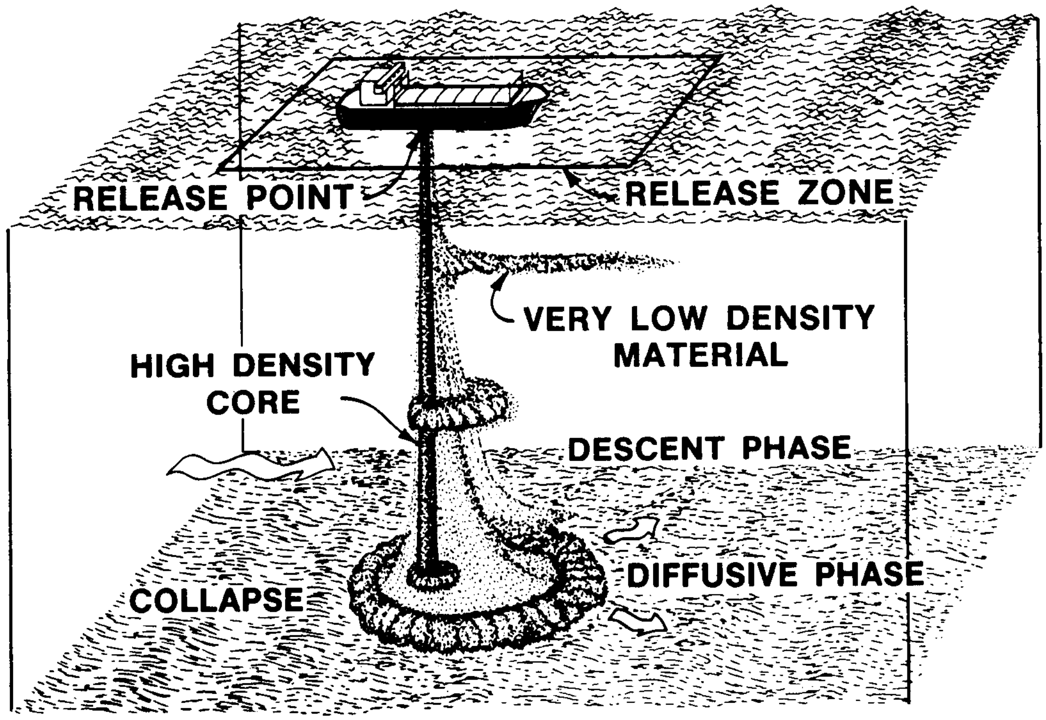

Figure 1. Transport processes during open-water disposal (adapted from Pequegnat et al. 1981)

When dredged material is released from a barge, it descends through the water column as a dense fluidlike jet. Within this well-defined jet, there may be solid blocks or clods of very dense cohesive material. Sustar and

Wakeman (1977) and Bokuniewicz and Gordon (1980) described the factors affecting this descent. Both concluded that the proportion of material that forms into clods in the discharge depends primarily on the mechanical properties of the sediment (especially moisture content and plasticity) and how those properties have been affected during the dredging operation. During the descent, large volumes of site water are entrained in the jet; as a result of several factors, including turbulent shear, some material is separated from the jet and remains in the upper portion of the water column. This so-called "lost" material (i.e., unaccounted for in the mass balance) transported out of the immediate site is frequently viewed with concern when dealing with contaminated sediments and is discussed in the following paragraph. To complete the stages of the disposal process, the descending jet and its core of cohesive material then collapse, usually as a result of impact on the bottom or, more rarely and at deeper sites, when it encounters a layer in the water column with ambient density equal to or greater than the jet. In the latter period of the collapse, that portion of the discharge that is not deposited when it impacts initially will move radially outward as a density/momentum-driven surge until sufficient energy is dissipated and the material begins to rapidly settle on the bottom. At this time diffusive processes dominate and any material remaining from the surge will be mixed with the lower water column and diluted and will continue to settle, although more slowly.

Suspended solids versus turbidity

The suspended solids concentrations in the water column and even those that comprise the surge are frequently reported as turbidity or a turbidity plume. As summarized by Stern and Stickle (1978), the term turbidity represents a complex composite of several variables that collectively influence the optical properties of water, and attempts to correlate turbidity with the weight concentration of suspended matter (suspended solids) are often impractical. Nevertheless, because of the time during which a disposal operation occurs (seconds to tens of minutes), considerable resources are needed to collect continuous water samples for gravimetric analysis. A majority of the data collected to date relies on some type of turbidity measuring device such as a transmissometer or other optical instrument. The approach most often used is to collect as many samples as possible for gravimetric analysis and to use those results to provide a local calibration for the turbidity values measured before and during the operation.

Field Investigations of Losses During Disposal

Long Island Sound

An early comprehensive field study of open-water disposal was reported by Gordon (1974). The results were based on observations of seven individual dumping operations at the New Haven site in Long Island Sound. The operations used clamshell equipment and bottom-dumping scows held stationary during discharge of the dredged material. Volumes of individual dumps ranged from approximately 1200 to 3000 cu yd. The project involved predominately maintenance dredging, and the dredged material was 60 to 90 percent in the silt- to clay-size range. Water depths at the disposal site were 60 to 65 ft, and measured bottom currents had maximum velocities of 0.5 to 1.0 ft/sec and minimums of 0.2 ft/sec.

A transmissometer calibrated with sediment from the study was used to observe the solids plumes. A number of techniques including profiles with depth at fixed stations and tracking of the disposal plume were used, and the results were composited for analysis.

Gordon calculated that approximately 1 percent of the total material exiting the barges remained suspended in the upper water column and was dispersed over a significant distance. The remaining material moved along the bottom in a very well-defined surge. He provided additional calculations of the flux of material in this bottom surge at various distances from the impact point and concluded that 80 percent of the original volume of material was deposited on the bottom within a radius of 100 ft and 90 percent within 400 ft. The surge was confined to the bottom in a layer 12 to 15 ft thick (a thickness equal to roughly 20 percent of the total water depth at the site).

San Francisco Bay studies

A second major source of information on open-water disposal is found in the reports of a comprehensive investigation, "Dredge Disposal Study: San Francisco Bay and Estuary," undertaken by the US Army Engineer District, San Francisco. In the main report, Sustar and Wakeman (1977) summarized and interpreted the results of several related investigations.

Releases were monitored in 1974 at three principal sites: barge operations at the Alcatraz site and at site LA-5 south of the Farallon Islands (the

100-fathom site*) and hopper-dredge operations at the Carquinez site. The deepwater Farallon site yielded no quantifiable data on losses in the water column, but surveys and underwater photographic coverage confirmed that, even in 600 ft, most of the material released could be subsequently identified on the bottom and that the spread was limited to an area 500 by 1000 ft. Preliminary measurements using a transmissometer were made at the Alcatraz and Carquinez sites to define plume behavior and refine the monitoring techniques.

The following year, an intensive monitoring program was conducted on hopper-dredge disposal operations at Carquinez. The dredged material was classified as silty clay to clayey silt and was discharged through twin 1300-cu-yd hoppers. Water depth during disposal was typically 45 ft and currents ranged from 0.3 to 0.8 ft/sec. Both transmissometers and gravimetric analysis were used to measure the suspended solids at the site.

The data from Carquinez, supported by observations and measurements at the other sites, indicated that concentrations in the range of grams per liter were recorded in a well-defined layer within 6 to 7 ft of the bottom (15 percent of the water depth). Twice during the study period, another instrument that was placed approximately 10 ft off the bottom registered concentrations higher than 300 mg/ℓ. Total unaccounted suspended solids in the upper portion of the water column above the surge were calculated to be 1 to 5 percent of the material released. Further, the report suggested that the source of much of the surface plume was spillage/overflow from the hoppers as the vessel turned on its disposal runs and from vessel disturbance of the released jet.

Dredged Material Research Program sites

Bokuniewicz et al. (1978) summarized several field studies of the mechanics of placing dredged material at various open-water sites. Results were reported for both hopper-dredge and barge/scow disposal operations under a variety of site conditions. A total of six sites were studied, including the previous New Haven study by Gordon (1974) and another site in the Long Island Sound area. A number of parameters were monitored in each study and considerable data on insertion, descent, and surge velocities were reported. A specially designed transmissometer was used to measure solids concentrations and was supplemented by water samples for gravimetric analysis. The work done

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* Units of measure are from original references. Hence both metric and nonmetric units occur.

during the study at a site off Seattle is especially notable because the water depths of over 200 ft were deeper than any other site studied.

Throughout a wide range of sediments, equipment types, and site conditions, the same basic description of the transport processes was found to be valid. Significant concentrations of solids were found only in a well-defined bottom layer, and impacts in the upper water column were minimal. The authors concluded that the amount of material in suspension transported through the upper water column during the placement process was very small (less than 1 percent in most cases). The thickness of the surge layer was confirmed to depend on total water depth at the site, and a further conclusion was presented on the effects of currents at the disposal site. Because of the large volume of water entrained by the descending jet, it will acquire the lateral speed of the (currents in the) receiving water. However, this was observed to result only in displacing the point of impact by a predictable distance, and no greater dispersion, disruption of the jet, or additional loss of material was noted.

New York Bight

In evaluating the losses associated with dredging, transporting, and disposing of material from New York Harbor, Tavolaro (1982, 1984) used a mass-balance approach rather than water-column sampling at the disposal site. The project involved both maintenance and new work, but both were dredged by clamshell equipment. Disposal took place at the Mud Dump site in New York Bight in 50 to 80 ft of water.

In addition to the innovative mass-balance approach, Tavolaro's monitoring work was exceptional in that he collected data from 229 barge loads representing over 800,000 cu yd of dredged material. Generally the procedure consisted of securing sufficient geotechnical sampling information so that volumetric measurements could be converted to units of dry mass for the in situ, barge, and postdisposal conditions. The volume (mass) at the site following disposal was calculated by comparing predisposal and postdisposal bathymetry. The losses during disposal were then inferred by subtracting the mass measured at the site from the mass in the barges. He concluded that 3.7 percent of the material mass was unaccounted for during the disposal operations.

Duwamish Waterway

The latest field study available on an open-water disposal operation was summarized by Truitt (1986). The results were part of a broader monitoring program conducted during a disposal demonstration project by the US Army Engineer District, Seattle. In summary, a single barge load of approximately 1100 cu yd of silty shoal material was discharged into a previously defined depression at the bottom of the Duwamish navigation waterway. Water depth ranged from 65 to 70 ft, and the bottom of the depression was about 6 ft below the surrounding bottom. Maximum sustained bottom currents were 0.2 ft/sec with occasional readings in the upper water column approaching 1.0 ft/sec. Stations were established along radials from the release point, and water samples were collected essentially continuously for subsequent gravimetric analysis to determine the concentrations of suspended solids. In order to provide a check of the results, a mass balance similar to that undertaken by Tavolaro was performed using replicate bathymetry and geotechnical data.

The results of the mass-balance calculation were presented within ranges representing estimates of the error associated with the bathymetry. These ranges overlapped, increasing confidence in the independent calculations. Between 7 and 14 percent of the material (as measured in the barge) was either transported out of the immediate vicinity or could not be accounted for in the mound. However, this amount (7 to 14 percent) represents the total flux of solids through the entire water column at a radius of approximately 100 ft from the disposal depression. It is therefore analogous to the sum of the material in the bottom surge layer <u>and</u> in the upper water column as reported by earlier investigators.

Figure 2 is an example of a profile of solids concentration with depth at one station. Notice that the maximum concentrations (700 mg/ℓ) in the near-bottom layer are lower than the values measured by Gordon (1974) and others. This is due to the confining effects of the depression. Little impact can be seen in the upper portions of the water column. Adjusting the loss calculations to reflect only the suspended solids passing through the water column above the bottom layer yields a value of 2 to 4 percent of the original mass that is likely to be dispersed over significant distances. The remaining material formed a surge layer in spite of the depression, but the concentrations in this layer are low. At 100 ft, they represent approximately 5 to 11 percent of the original material compared to 18 percent typically measured by Gordon (1974) at a site with a level bottom.

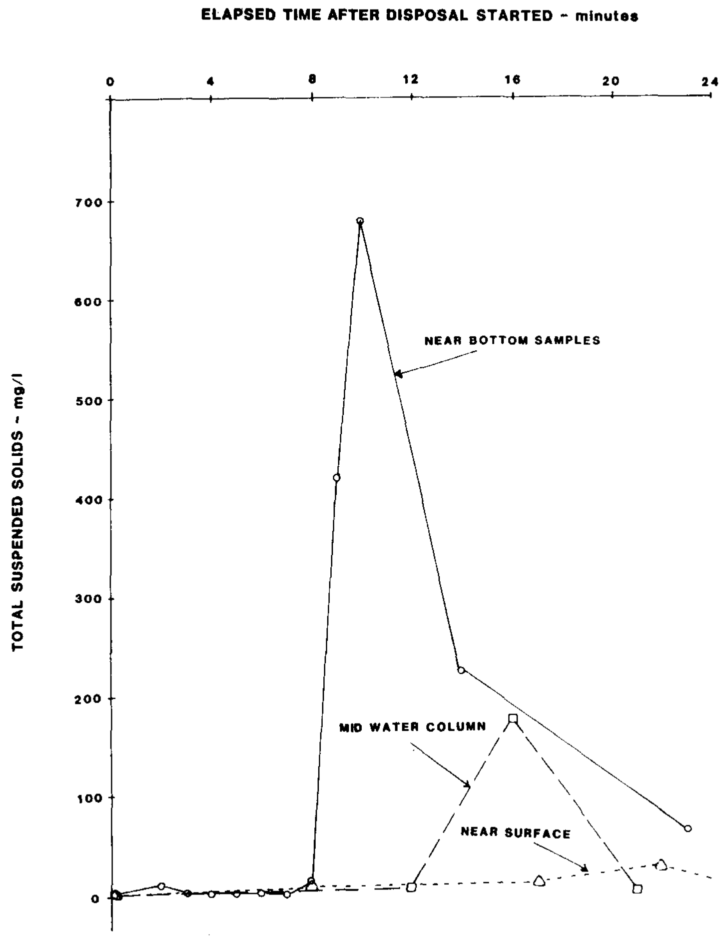

Figure 2. Time series of total suspended solids at three depths showing well-defined bottom layer and minimal effects in upper water column (Truitt 1986)

The study confirmed that only a small amount of suspended sediment is typically transported away from the jet through the upper water column during disposal. The principal transport mechanism at the disposal site was the bottom surge or density flow, and control measures such as disposal into a depression can be effective in arresting that transport.

Conclusions from field studies

The five studies discussed above appear to be the only reports of actual field measurements of short-term dispersion or loss of material resulting from open-water disposal of dredged material by barge or hopper operations. The data are summarized in Table 1. Each investigation confirmed the validity of the description of the transport processes suggested by Clark et al. (1971);

Table 1

Summary of Field Studies of Fate of Dredged Material

During Open-Water Disposal

		Site Characteristics		Dredging/Disposal Characteristics					
Data Source	Site	Water Depth, ft	Bottom Currents ft/sec	Dredged Sediment	Dredge Type	Disposal Type	Typical Volume cu yd	Monitoring Technique or Device	Sediment in Upper Water Column, Percent of Original
Gordon (1974)	Long Island Sound	60-65	0.2-1.0	Silt-clay	Clamshell	Scow	1200-3000	Transmissometer	1
Sustar and Wakeman (1977)	Carquinez*	45	0.3-0.8	Silt-clay	Trailing suction hopper	Hopper	1300	Transmissometer and gravimetric	1-5
Bokuniewicz et al. (1978)	Ashtabula (Lake Erie)	49-59	0-0.7	Sandy silt	Trailing suction hopper	Hopper	900	Transmissometer and gravimetric	1**
	New York Bight	85	0.2-0.8	Marine silt	Trailing suction hopper	Hopper	8000	Transmissometer and gravimetric	1**
	Saybrook (Long Island Sound)	170	0.7-2.3	Marine silt	Clamshell	Scow	1500	Transmissometer and gravimetric	1**
	Elliott Bay	220	0-0.7	Sandy silt	Clamshell	Scow	500-700	Transmissometer and gravimetric	1**
	Rochester (Lake Ontario)	55-150	0-0.7	Riverine silt	Trailing suction hopper	Hopper	900	Transmissometer and gravimetric	1**
Tavolaro (1982)	New York Bight	52-80	N/R	Silt and clay	Clamshell	Scow	1800-4000	Mass balance	3.7
Truitt (1986)	Duwamish Waterway	65-70	0.2	Silt-clay	Clamshell	Scow	1100	Gravimetric and mass balance	2-4

* Limited data from two additional sites included.
** Synthesis of all sites reported.

over a wide range of site conditions, materials, and operational and/or measurement techniques, the results shown in Table 1 are remarkably consistent.

Additional References

A number of other authors have quoted values for losses of dredged material during open-water disposal or have made conclusions without citing specific details or sources of information. The following authors, given with their sources, are perhaps the most frequently cited.

Bokuniewicz and Gordon (1980) stated that the amount of dredged material lost to the surrounding water during the placement process will be small, generally 1 to 5 percent of the amount released, regardless of the proportion of the material that forms into clods. Their conclusions were based on the work of Gordon (1974) and Sustar and Wakeman (1977). Bokuniewicz (1985), writing a chapter in the series, Wastes in the Ocean, again quoted the values of 1 to 5 percent of the released material remaining in suspension. Johanson, Bowen, and Henry (1976) also relied on the study by Gordon (1974) to conclude that the turbidity cloud contains less than 1 percent of the dumped material. Alden, Dauer, and Rule (1982) mentioned monitoring three test dumps as part of an investigation of the Norfolk open-water disposal site. Although no specific details or sources were given, they concluded that the disposal resulted in little change in the physical condition of the water column.

Mass and Volumetric Balances

In any discussion of losses during dredged material disposal, some consideration must be given to the manner, volumetric or mass, in which quantities are measured and compared. This is especially important when the data collection and analysis involve direct before-and-after comparisons. Tavolaro (1982, 1984) clearly established that apparent volumetric changes may not be true losses when evaluated solely on a mass basis. A known initial volume in a barge, say 1000 cu yd, and 900 cu yd identified in-place at the site following disposal does not imply that 10 percent of the original material was lost during placement. It is easy to see the problem with this approach, even during a short-term time frame, given the calculation by Bokuniewicz et al. (1978) that a descending jet may entrain a volume of site water equal to 70 times its original volume! After undergoing such a tremendous (and rapid) change, the volume in place has only a limited relationship to the original volume. Over longer periods of time, volatilization and consolidation further obscure the usefulness of considering only volumetric data for accounting for the fate of the material. Finally, the measuring capability of routine monitoring equipment and techniques is such that differences in the range of 1 to 5 percent are generally undetectable.

Summary

The published field data support the theoretical description of the transport phases in typical open-water disposal operations. The short-term impacts resulting from suspended sediment are confined to a well-defined layer near the bottom. The initial thickness of this layer before spread and diffusion is related primarily to the depth of water at the site. A thickness above the bottom equal to 15 to 20 percent of the total water depth was observed in the majority of the studies. Above this bottom layer, suspended sediment concentrations are one to two orders of magnitude less and the total amount of solids dispersed over longer distances is 1 to 5 percent of the original material. Any monitoring program designed to account for dredged material fate during disposal should include measurements of mass and not rely solely on volumetric balances.

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Environmental Effects of Dredging
Technical Notes

ENGINEERING CONSIDERATIONS FOR CAPPING SUBAQUEOUS DREDGED MATERIAL DEPOSITS -- BACKGROUND AND PRELIMINARY PLANNING

PURPOSE: The following two technical notes present information applicable to planning and constructing dredged material capping projects:

EEDP-01-3 Background and Preliminary Planning

EEDP-01-4 Design Concepts and Placement Techniques

This first note identifies and reviews field experiences with subaqueous capping of dredged material and discusses aspects of site selection.

BACKGROUND: In recent years the search for alternatives to expensive and limited upland containment areas for contaminated sediment has centered on in-water capped disposal. This interest was further reinforced when the convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (the London Dumping Convention) accepted the capping concept, subject to monitoring, as an appropriate technology for rapidly rendering harmless the contaminants of concern in dredged material. Subsequent detailed investigations (e.g., Brannon et al. 1985, O'Connor and O'Connor 1983) have confirmed that capping can be effective in chemically and biologically isolating contaminated dredged material from the overlying aquatic environment.

However, in order to ensure this effectiveness, capping projects cannot be treated simply as a modification of conventional disposal practices. A capping project must be thought of as an engineered structure with design and construction requirements that must be met, verified, and maintained over the design life. This is not to say that traditional equipment and operational methods cannot be applied to capping contaminated materials. In fact, they have been used with good success. Technologies must, however, be applied in a sytems context and with careful control and monitoring.

ACKNOWLEDGEMENT: The author of this note is Clifford L. Truitt of the WES Coastal Engineering Research Center.

ADDITIONAL INFORMATION AND QUESTIONS: Contact Dr. Michael R. Palermo (601) 634-3753 (FTS 542-3753) or the manager of the Environmental Effects of Dredging Programs, Dr. Robert M. Engler (601) 634-3624 (FTS 542-3624).

The Capping Concept

A simple definition of in-water or subaqueous capping is the controlled accurate placement of contaminated materials at a disposal site, followed by a covering or cap of clean isolating material. Figures 1 and 2 are schematics of

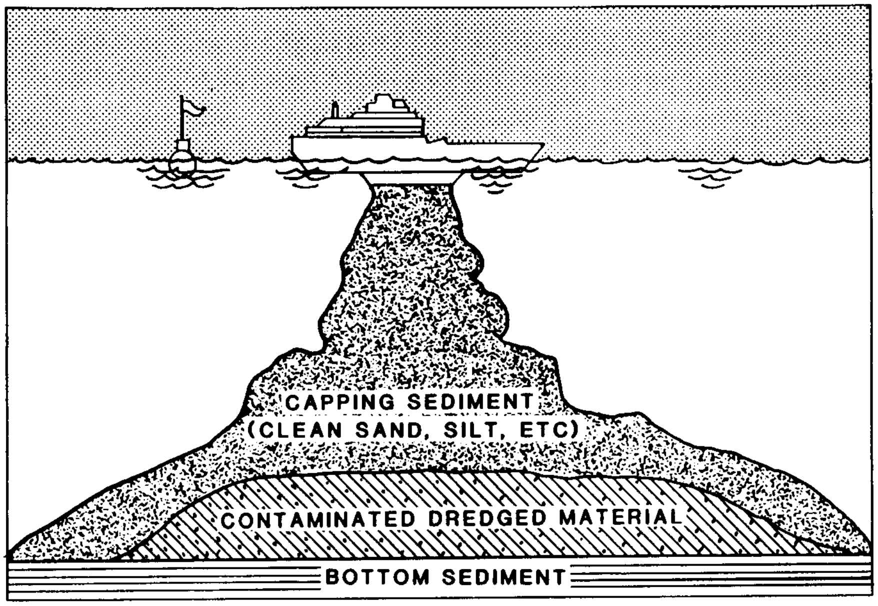

Figure 1. Schematic of typical level-bottom capping operation (adapted from Shields and Montgomery 1984)

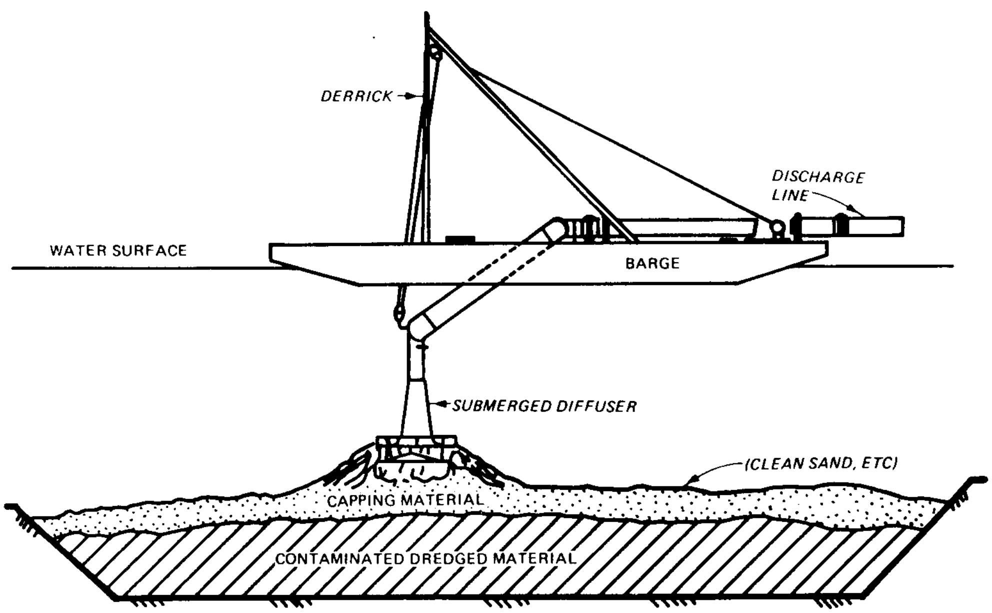

Figure 2. Schematic of contained aquatic disposal (CAD) project also showing use of a submerged diffuser for placement

two types of capping projects, level-bottom capping and contained aquatic disposal (CAD). As the name suggests, level-bottom capping projects attempt to place a discrete mound of contaminated material on an existing flat or very gently sloping natural bottom. A cap is then applied over the mound by one of several techniques, but usually in a series of disposal sequences to ensure adequate coverage. CAD is generally used where the mechanical properties of the contaminated material and/or bottom conditions (e.g., slopes) require positive lateral control measures during placement. Use of CAD can also reduce the required quantity of cap material and thus the costs. Options might include the use of an existing depression; preexcavation of a disposal pit; or construction of one or more submerged dikes for confinement.

It is evident that capping projects must be characterized by a high degree of interaction among various operational factors. Table 1, from Shields and Montgomery (1984), demonstrates these interrelationships and emphasizes the need for a systems approach to planning.

TABLE 1. Considerations for Planning Capping Operations

Number	Decision Decription	Impacted by Decision
1	Dredge equipment selection	
2	Selection of disposal and capping site	
3	Placement method for contaminated material	1,2
4	Method for transporting contaminated material to disposal site	1,2,3
5	Selection of capping material	1,2,3,4
6	Placement method for cap	1,2,3,4,5
7	Dredge plant for obtaining cap material	1,2,3,4,5,6
8	Method for transporting cap material to disposal site	1,2,3,4, 5,6,7
9	Method for navigation and positioning at site	2,4,8
10	Method for monitoring site	2,9

Overview of Existing Capping Projects

Field experience with subaqueous capping is certainly limited in comparison to the decades of upland disposal site design. However, a sufficient number of capping projects have been completed to establish that the concept is technically and operationally feasible. Table 2 describes the salient features of the major capping projects reported in the literature.

Level-bottom capping projects

The majority of the reported projects were the level-bottom design in which contaminated fine-grained sediment was excavated by mechanical dredge and placed by conventional bottom-dumping barges or scows. The cap material was typically silt and/or fine sand that was placed over the mounds by either scows or a conventional hopper disposal. None of the reports noted any difficulty in producing well-defined discrete mounds.

In general, descriptions of the projects indicated that the sediment formed a very steep-sided central mound with a radius of 400-500 ft and a height of several feet (Table 2). Following a sharp break in slope, material continued in a deposit up to several inches thick over an annular area extending an additional 400 to 500 ft. In these projects, no attempts were made to cover the mound with a cap of uniform thickness. Coverage was achieved by point placement of relatively large volumes (at least 2 to 3 times the underlying mound volume) of capping sediment. In the few reported cases where, in the opinions of the investigators, the disposal project was not considered entirely successful (e.g., Central Long Island Sound Cap Site No. 1 and No. 2), the difficulties were traced to problems with positioning or control rather than to equipment or design.

In summary, experiences at several heavily monitored level-bottom capping projects indicate that mechanically dredged sediment can be deposited in discrete mounds and successfully capped. Conventional equipment and operational techniques can be used, provided special attention is given to precise positioning and overall control of the operation.

CAD projects

Design objectives. Applications of the CAD design have been limited; and, because the projects involved variations in equipment and technique, generalizations are difficult. As noted, CAD is typically used where positive lateral control of a contaminated dredged material is desired during placement. In planning these types of projects, it is important to clearly identify the reasons for the desired increase in confinement so that proper alternatives are evaluated.

Three existing CAD sites are listed in Table 2; however, in none of the

TABLE 2. Descriptions of Capped Disposal Projects from the Literature

Project: Location (Date)	Project: Site Characteristics	Contaminated Material: Volume of Material yd³*	Contaminated Material: Dredging Method	Contaminated Material: Placement Method	Capping Material: Volume, yd³* (Type)	Capping Material: Thickness of Cap, ft	Capping Material: Placement Method	Capping Material: Positioning Method	Data Source
Duwamish Waterway Seattle, WA (1984)	Existing subaqueous depression ~70 ft deep	1,100	Clamshell	Scow	3600 (sand)	1-3	Sprinkling from scow	Surveying instruments	Truitt 1986, Sumeri 1984
Rotterdam Harbor, The Netherlands (1981-1983)	Phase I: Botlek Harbor Excavated to ~98 ft deep	1,200,000	Trailing suction hopper	Pumpout-submerged diffuser	- - (clay)	2-3	Scow, then leveled over site	Surveying instruments	d'Angremond et al. 1986
	Phase II: 1st Petroleum Harbor Excavated to ~80 ft deep	620,000	Matchbox suction	Pipeline submerged diffuser	- - (clay)	2-3	Scow, then leveled over site	Automated dredge and suction head positioning equipment	d'Angremond et al. 1986
Hiroshima Bay, Japan (1979-80)	Contaminated bottom sediment overlaid in situ with capping material ~70 ft deep	N/A	N/A	N/A	- - (sand with shell)	1.6	Conveyor to gravity-fed submerged tremie	Surveyed grid and winch/anchor wires	Kikegawa 1983
							Suction/ pumpout thru submerged spreader bar		Togashi 1983
New York Bight (1980)	Generally flat bottom ~80-90 ft deep	860,000 (mounded to 6 ft thick)	Clamshell	Scows	1,800,000 (majority fine sand)	Average 3-4 Maximum 5-9	Scow, hopper dredge	Buoy, real-time navigation electronics	Freeland 1983, Mansky 1984, O'Connor and O'Connor 1983, Suszkowski 1983
Central Long Island Sound Disposal Area (1979)	Stamford-New Haven, North Generally flat bottom ~65 ft deep	34,000 (mounded 3-6 ft thick)	Clamshell	Scows	65,400 (sand)	up to 7-10	Hopper dredge	Buoy, Loran-C coupled positioning system	Morton et al. (eds.) 1984, O'Connor and O'Connor 1983
(1979)	Stamford-New Haven, South Generally flat bottom ~70 ft deep	50,000 (mounded 4-6 ft thick)	Clamshell	Scows	100,000 (cohesive silt)	up to 13	Scow	Buoy, Loran-C coupled positioning system	

(Continued)

* All volumes are approximate, usually based on estimated in-scow measurements. Dash entries indicate volume of capping either unknown or not reported.

TABLE 2. **Descriptions of Capped Disposal Projects from the Literature** (Concluded)

Project		Contaminated Material			Capping Material				
Location (Date)	Site Characteristics	Volume of Material yd	Dredging Method	Placement Method	Volume, yd^3 (Type)	Thickness of Cap, ft	Placement Method	Positioning Method	Data Source
Central Long Island Sound Disposal Area (1979) (Continued)									Morton et al. (eds.) 1984, O'Connor and O'Connor 1983
(1981)	Norwalk Generally flat bottom ~65 ft deep	92,000 (multiple mounds up to 8-12 ft thick)	Clamshell	Scows	370,000 (silt and sand)	up to 6-7	Scow	Buoy	
(1982-3)	Mill-Quinnipiac Generally flat bottom ~65 ft deep	40,000	Clamshell	Scows	1,300,000 (silt)	Multiple broad area placement. Estimated final avg 6-10	Scow	Buoy	
(1983)	Cap Site No. 1 Generally flat 60 ft deep	33,000 (mounded 3 ft thick)	Clamshell	Scows	78,000 silt	Incomplete coverage	Scow	Buoy, Loran-C	
(1983)	Cap Site No. 2 Generally flat ~56 ft deep	40,000 (low mound, 2 ft thick)	Clamshell	Scows	40,000 sand	Irregular - maximum 4.5, areas as little as 0.6	Scow	Buoy, Loran-C	

three did the engineering characteristics of the dredged material directly dictate the use of a CAD design. The principal design influence in these projects was the need to produce a disposal site with sufficient volume below navigable depths in an existing waterway. The secondary objective was to reduce the number of migration pathways through which contaminants could find their way into the environment (i.e., increase the contaminant isolation).

The interactive processes shown in Table 1 were particularly demonstrated in the Rotterdam Harbor projects (d'Angremond, de Jong, and de Waad, 1986). The use of the CAD alternative provided the required volume within existing waterways and reduced the total number of contaminant migration pathways. However, because the depth of the excavation would have placed the contaminated material closer to critical groundwater resources, that single pathway actually became the greatest concern and resulted in a decision to deposit clay to line the excavation as well as to cap the contaminated material. But the decision to use CAD also allowed dredging to be performed by a hydraulic dredge with pipeline transport at significant time and cost savings.

Cap placement at existing CAD sites. The method and/or rate of placing capping material over a CAD site, especially one in which hydraulically dredged sediments have been disposed, has been cited as a concern. Point dumping of cap material over these unconsolidated deposits is likely to result in displacement of the contaminated material. The reviewed projects in Hiroshima Bay (Togashi 1983 and Kikegawa 1983) demonstrated technologies that have application to this problem. Both projects involved the overlaying of contaminated bottom sediment in situ with clean capping sand. In one case, a telescoping tremie (gravity-fed downpipe) was extended through the water column and capping sand fed into it by a conveyor/barge system. In the second test, a submerged spreader bar with diffuser ports was used to apply the cap. Both projects resulted in the controlled placement of a uniform cap approximately 20 in. thick.

The Duwamish Waterway capping project (Sumeri 1984, Truitt 1986b) demonstrated the use of a conventional split-hull barge with operational modifications to place the cap. Contaminated sediment had been dredged mechanically and accurately placed in an existing depression used as the CAD site by a precisely positioned and controlled barge operation. The cap was then placed by incrementally opening, over a period of tens of minutes, the split hull of another barge filled with clean sand. The sand exited slowly and was sprinkled through the water column onto the site. Dispersion was minimal and three discrete, but overlapping, disposal sequences were used to ensure adequate coverage.

A third procedure was tested in the Rotterdam Harbor projects. At these sites the excavation of the CAD areas produced a surplus of clean cohesive clay that was incorporated into the design to be used as a reduced permeability capping material. The combination of unconsolidated hydraulically-placed contaminated sediment and the very cohesive mechanically dredged capping material precluded conventional point dumping of the cap. Barge loads of the clay were deposited on the bottom adjacent to the disposal site and the material subsequently raked over the contaminated sediment using a towed drag. This technique is not recommended because of the localized increase in suspended solids during construction, but it did demonstrate that a cap could be effectively placed and supported.

Considerations for Capping Site Selection

At least six considerations can be identified that are important in evaluating the engineering acceptability of a proposed open-water dredged material disposal site:

- Bathymetry (bottom contours)
- Currents (velocity and structure)
- Average water depths
- Salinity/temperature (density) stratifications
- Bottom sediments
- Operational requirements (location/distance, surface sea state, etc.)

In general, these considerations are no different for a site intended for capping. Probably the most important (physical) goal in selecting an open-water site for disposal and capping of contaminated dredged material is long-term stability of the deposited material. However, site selection normally involves a compromise or trade-off among the desirable criteria for each site characteristic.

Influence of Site Conditions on Capping Projects

Bathymetry. If the bottom in a disposal area is not horizontal, a component of the gravity force will influence the energy balance of the bottom surge. It is difficult to estimate the effects of slope alone, since bottom roughness plays an equally important role in mechanics of the spreading process. Gordon (1974) described the results of monitoring barged disposal operations at a level bottom site on Long Island Sound and concluded that 81 percent of the original volume of sediment released was deposited within a radius of 100 ft from the point of impact and 99 percent was deposited within a radius of 400 ft. Disposal into an existing depression approximately 150 by 300 ft was monitored during the Duwamish capping demonstration project (Truitt 1986b). Measurements of sediment in the water column at a distance of 100 ft from the center of impact showed that 93 percent of the original mass could be accounted for within this radius and confirmed the positive effect of using existing or constructed confining features at a disposal site.

Currents. Basic current information should be collected at prospective disposal sites to identify site-specific conditions. However, based on observations at several sites, Bokuniewicz et al. (1978) concluded that the principal influence of currents in the receiving water is to displace the point of impact of the descending jet of material with the bottom (by a calculable amount). They stated that even strong currents observed at a Great Lakes site need not be a serious impediment to accurate placement, nor do they result in significantly greater dispersion during placement. Further, currents do not appear to affect the surge phase of the disposal (see Truitt (1986a) for a description of the overall disposal processes at open-water sites).

Long-term effects of currents at a prospective site may still need to be investigated, and little information is available on the transport of sediments from disposal mounds. Storm-induced currents are also of interest in the long-term stability of the site. However, disposal operations would be halted during storms, so the designer need consider only near-bottom currents not water-column currents. Measured current data can be supplemented by estimates for external events using standard techniques; e.g., see the Shore Protection Manual (Coastal Engineering Research Center 1984).

Average water depths. Aside from the effect that depth has on currents, there appears to be little additional short-term influence on disposal.

Bokuniewicz et al. (1978) observed the same general physical processes resulting from placement of dredged material at different sites with water depths ranging from approximately 50 to 200 ft. In deeper water, more entrainment occurs in the descent phase, and there is more bulk dilution of the dredged material before it reaches the bottom. However, there is no increase in the jet impact speed, nor does the bottom surge spread at a faster rate. The initial thickness of the spreading surge above the bottom has been shown to be a function of water depth. Again, the total water depth at a site has more favorable impact on long-term stability than unfavorable impact during the disposal process.

Salinity/temperature (density) stratification. A sufficiently great density gradient in sufficiently deep water can result in arrest of the descending jet. The depth at which this occurs can be calculated. Bokuniewicz et al. (1978) suggested that although highly stratified conditions may be encountered, it is most unlikely that water depths would be great enough at most sites to cause collapse in the upper water column. Johanson, Bowen, and Henry (1976), reporting on work discussed by Brooks (1973), presented a simple empirical equation to estimate when a descending jet would penetrate a stratified layer. In addition to the relative differences in density, the depth to the interface of the density layers in the water column (not total water depth) and the initial volume of the jet are the important terms.

Operational requirements. Among the operational criteria that should be considered in evaluating potential capping sites are: volumetric capacity of site; nearby obstructions or structures; haul distances; bottom shear due to ship traffic (in addition to natural currents); and ice influences. The effects of shipping are especially important since bottom stresses due to prop wash and/or direct hull contact at shallow sites are typically of a greater magnitude than the combined effects of waves and other currents. A windowing or templating technique has been used successfully in several Corps districts to overlay the effects of each site-selection parameter in an area, identifying graphically the optimal sites.

Modeling site influences

Numerical models have been developed (Johnson 1986) that can be used to estimate the initial configuration of a dredged material disposal mound on the sea floor. These models incorporate the dredged material characteristics and features of each of the six site evaluations considerations described earlier.

The models allow rapid and economical comparisons of the influence of site conditions at several locations under consideration for a disposal project or prediction of the effects of variations in operational technique or equipment at a selected site. A recent application, for example, allowed assessment of the effects of very deep water at a Puget Sound disposal site on the descent of the jet from a conventional surface release versus a submerged discharge.

Summary

Capping is the controlled accurate placement of contaminated dredged material at a disposal site, followed by a covering or cap of clean isolating material. Capping projects are typically described as level-bottom placement or contained aquatic disposal. Field experience with subaqueous capping is limited, but eleven sites have been identified where the technique has been applied in one form or another (Table 2).

Site-selection considerations for capping projects are similar to those for any open-water disposal. The influences of several types of site characteristics on capping have been identified and discussed. Modeling methods are available to aid in evaluating sites and designs. Additional information on cap materials, placement, and monitoring are provided in Technical Note EEDP-01-4.

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Environmental Effects of Dredging
Technical Notes

ENGINEERING CONSIDERATIONS FOR CAPPING SUBAQUEOUS DREDGED MATERIAL DEPOSITS -- DESIGN CONCEPTS AND PLACEMENT TECHNIQUES

PURPOSE: The following two technical notes present information applicable to planning and constructing dredged material capping projects:

EEDP-01-3 Background and Preliminary Planning

EEDP-01-4 Design Concepts and Placement Techniques

This second note discusses the selection of cap material and presents the results of recent equipment and technique demonstrations. Monitoring guidelines are also described.

BACKGROUND: In order to ensure the effectiveness of capping, such projects cannot be treated simply as a modification of conventional disposal practices. A capping project involves an engineered structure with design and construction requirements that must be met, verified, and maintained over the design life. This is not to say that traditional equipment and operational methods cannot be applied to capping contaminated materials. In fact, they have been used with good success.

ACKNOWLEDGEMENT: The author of this note is Clifford L. Truitt of the WES Coastal Engineering Research Center.

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Considerations for Cap Materials and Cap Design

One of the principal design decisions in a capping project is the nature and thickness of the cover material to be placed over the contaminated dredged material. The cap provides the isolation necessary to control the movement of contaminants out of the dredged material into the overlying water column and to prevent direct contact between aquatic biota and the contaminants. The cap also performs the important physical function of stabilizing the dredged material and protecting it from transport and dispersion away from the site. The design of the cap, therefore, requires a two-fold approach. It must result in a capping layer with properties and thickness such that it functions as an adequate seal, yet the layer must remain unbroken and resist resuspension and transport by the bottom shear stresses at the site.

Shields and Montgomery (1984) suggested that potential capping materials can be classified as inert, chemically active, or sealing agents. They, as well as Johanson, Bowen, and Henry (1976), reviewed characteristics and applicability of several types of material. Although chemically active materials and sealing agents (e.g., grouts, polymer films) have some attractive capping properties, general experiences with them are limited and specific cases of use on subaqueous dredged material deposits are nonexistent. As shown in Table 2 of Technical Note EEDP-01-3, all projects to date have used inert materials (clean sand and silt) for capping, and it is unlikely that this trend will change in the immediate future. Sufficient volumes of clean sediment are usually available even in contaminated reaches, and techniques and equipment for placing such materials as capping are also readily available.

Contaminant isolation

The effectiveness of inert sediment as a contaminant-isolation technique has been evaluated by Brannon et al. (1985). Their experiments used modified flow-through reactor units containing contaminated sediment and capping material. To assess effectiveness, they performed chemical analyses on water samples from the reactor water columns and monitored contaminant uptake in indicator clams and polychaetes. In their testing matrix, samples of a sand, silt, and clay were evaluated at various thicknesses and both with and without the presence of bioturbation organisms. Results indicated that the cap materials with the higher percentages of clay and silt were generally more effective than sand in preventing the movement of contaminants into the water column. The thickness of the cap, however, especially in the presence of bioturbation, is apparently as important as the type of material since thicker caps of each of the three materials were equally effective. Certainly additional work in the general area of contaminant isolation is suggested and testing of specific contaminated sediments is advisable for design.

The effective thickness necessary for isolation must be specified considering any incorporation into the underlying sediment and must be maintained over the life of the project. However, given the difficulty of constructing and maintaining a conformal cap within a tolerance of inches (e.g., conventional fathometer accuracy is on the order of 6 in.), practical cap thicknesses specified as an operational requirement are going to be on the order of 3 ft. It is likely that for all but the most unusual case, constructability and erosional considerations will control the minimum cap thickness.

Cap erodibility

Sediment behavior. The cap design must specify the necessary thickness and materials that will maintain that thickness under the effects of long-term erosion and transport. Sediment transport is a complex process made even more complicated by the mechanical effects of the dredging on the sediment and by the configuration of the disposal mound. Although sediment can be classified in a number of meaningful ways, the information most commonly available in dredging projects is particle size (percent sand, silt, and clay) and some indication of the plasticity (e.g., inferred from Atterberg limits, USCS class, or possibly shear strength data).

Noncohesive sediment (sand and some silt) transport as individual grains typically in a continuing series of discrete erosion and deposition events. The transport is primarily dependent on the size, shape, and weight of the sediment particles and on the magnitudes of the fluid forces exerted on them.

For sediment generally classified as cohesive (silt and clay), the potential erodibility is more dependent on the condition of the cohesive bonds between the particles than on the characteristics, especially size, of the individual particles. Since fine-grained sediment has such poor settling properties, the particles are not easily redeposited once suspended and tend to move in a suspended layer above the bottom or to remain stationary in such a layer (i.e., fluff). Their hydrodynamic behavior is complicated by the effects of flocculation. In addition, the initial bonding is influenced by the method of dredging and placement, and the longer-term surface cohesion is related to the nonlinear, time-dependent consolidation process.

Subaqueous caps constructed predominantly of plastic clay-sized sediment are feasible and, in fact, have been used (i.e., Rotterdam Harbor project listed in Table 2 of Technical Note EEDP-01-3). Once placed, such material is more resistant to erosion than noncohesive sediment and can provide an effective seal. However, because of the difficulty in handling and uniformly placing such materials, this must be thought of as an exception to a typical project. It is more likely that a cap would be constructed of some combination of sand and silt with low to moderate plasticity. It must be noted that, for such deposited material, the apparent grain size presented to the fluid may be different than that observed in laboratory classification. It is common for mixtures to undergo initial sorting and winnowing that results in a surface layer having an average grain size much larger and less likely to transport than the remaining material. In addition, biological activity is known to aggregate grains of sediment providing a degree of self-armoring and apparent cohesion in relatively short periods of time.

Predictive methods. There are four principal approaches that can be applied to predicting the resuspension and transport of material from a capped mound (Dortch 1986): steady-state analytical methods; time- and rate-dependent analytical methods; physical and numerical modeling; and field and laboratory measurements. Randall (1986), summarizing the work of Dortch (1986), described the applications of each method as follows.

The first approach assumes steady or constant conditions and is representative of long-term average conditions. Such an analysis is the simplest to apply but fails to show results that can occur during episodic events such as storms. A steady-state analytical method developed for dredged material disposal mounds and applied to a site in San Francisco Bay was reported by Trawle and Johnson (1986).

The second approach is more difficult to apply, but it includes the effects of extreme events and variations in rate-dependent processes. Continuous physical processes are discretized into a series of distinct events for analysis. A time- and rate-dependent analysis of a dredged disposal site in Tampa Bay, Florida, was conducted by Williams (1983). Trawle and Johnson (1986) also extended their method to nonsteady conditions.

The application of numerical models to disposal mound transport can yield valuable information and detail, but also requires significant effort and potentially high cost for the more sophisticated multi-dimensional versions. Such methods generally require the use of both a hydrodynamic model and a sediment transport model either in coupled or uncoupled form.

Little information is available on the application of field or laboratory measurements to the study of the long-term fate of dredged material placed in subaqueous disposal sites. (For a summary of investigations of short-term fate, see Technical Note EEDP-01-2.)

In all these predictive methods, the focus is on resuspension and transport (typically based on incipient motion of individual grains) of mound or cap material. However, the net effect on cap stability must consider the eventual fate of resuspended cap (and adjacent bottom) material. It would be a rare site that experienced net transport in all directions away from the mound. Certainly some sites may experience gradual losses in volume over time and storm events can result in significant, temporary profile lowering at a mound; but verified general models for predicting the net effects of resuspension, transport, and redeposition are not yet available. The provision of an increased thickness of cap material at initial construction (advance nourishment) together with monitoring and maintenance are recommended as interim measures to ensure that the effective cap thickness is provided for the design life of the disposal area.

Placement Equipment and Techniques

This discussion of placement techniques applies equally to the contaminated dredged material to be capped as well as to the capping material itself. However, the intent of various techniques may differ between the two. Previous investigations (see Technical Note EEDP-01-2) have demonstrated that dredged material released at the water's surface, both by instantaneous discharge from barges or hopper dredges and by continuous hydraulic pipeline discharge, tends to descend rapidly to the bottom as a dense jet with minimal short-term losses to the overlying water column. Potentially undesirable effects can still result from impact, scour, and spread of the material over the bottom. Two objectives for the placement of both cap and underlying dredged material are control and accuracy. In all cases, accurately controlled placement reduces required areas, confines benthic impacts, results in economy of materials, and can reduce monitoring effort.

In the case of some contaminated dredged material, an additional objective necessary may be to isolate the material from the water column during at least part of its descent. This isolation can minimize mixing and potential chemical releases; significantly reduce entrainment of site water, thereby reducing disposal volumes; and negate any possible effects of currents during disposal. Technologies to accomplish these objectives are described in the following paragraphs, but they should be viewed as conservative measures and their need on a specific project should be clearly established. Experience has shown, for example, that contaminated silt and clay that have been dredged by clamshell will tend to remain in clumps during descent, offer little time or surface area for chemical release (certainly at in interstitial level), and form nonflowing discrete mounds on the bottom.

Specific additional considerations for placement of clean inert capping material focus more on controlling the rate of its application to the contaminated material. Conventional point dumping of moderately cohesive capping material may produce sufficient impact energy to displace soft deposits of underlying contaminated dredged material. Variables include the depth of water, rate of release, likelihood of clod formation versus transition to discrete particle sedimentation, and the strength of underlying material.

Modified surface release

Conventional equipment can be used to place cap material in many cases with only minor modifications. In the Duwamish contained aquatic disposal (CAD) demonstration (see Technical Note EEDP-01-03, Table 2), clean sand was successfully sprinkled over the contaminated dredged material by slowly opening a conventional split-hull barge over a time frame of just under one hour. The sand descended in a generally continuous manner with no displacement of the dredged material. Three barge loads were applied in an overlapping pattern to produce the necessary coverage. Clean coarse capping material could also be applied by surface discharge of a conventional hydraulic pipeline or by spray-booms analogous to side casting.

Submerged discharge

The use of a submerged discharge or closed conduit of some type to place the dredged material and/or the cap is a further level of control that is available. To the extent that the conduit extends through the water column and physically isolates the discharge, it can meet the objectives described for handling some contaminated material. If it is combined with a diffusive head to reduce velocities and place material near the bottom, it can meet the objective for capping. A number of conduit technologies are available or have been suggested to place dredged material and/or capping material through the water column.

Submerged diffuser.

A submerged diffuser (Figure 1), originally designed as part of the Corps' Dredged Material Research Program, has been successfully field tested in the Netherlands at Rotterdam Harbor and as part of an equipment demonstration project at Calumet Harbor, Ill. (McLellan and Truitt 1986). The diffuser

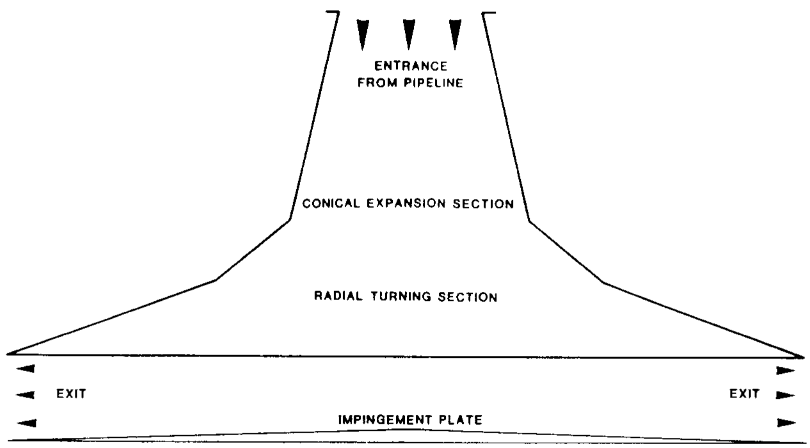

Figure 1. Schematic of submerged diffuser

minimizes upper water column impacts and especially improves placement accuracy and controls sediment spreading, which in turn reduces benthic impacts. By routing the slurry first through a conical expansion and then a combined turning and radially divergent diffuser section, the discharge is released parallel to the bottom and at a lowered velocity. The design of the diffuser section can be modified to suit project needs.

Results of the Calumet diffuser demonstration showed that the discharge velocity was reduced to 25 percent of the measured pipeline velocities. At a distance of 15 ft from the diffuser, the velocity was 5 percent of the average pipeline value (Figure 2). The discharged material was confined to the lower 20 percent of the water column with no increase in suspended solids above that point.

The diffuser could be employed as a direct connection to a pipeline dredge or as a modification to hopper dredged or mechanically dredged material

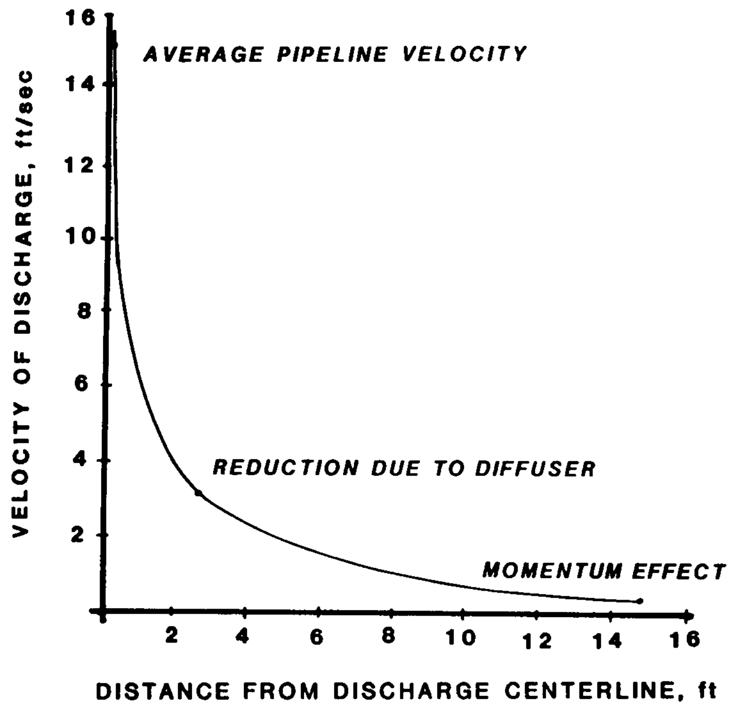

Figure 2. Changes in velocity of dredged slurry through diffuser and adjacent water column

disposal techniques (Figures 3 and 4). For the latter cases, a reslurring pump-out capability would be required. The pipe connecting the surface/support barge to the submerged diffuser head can be of relatively small diameter (conventional pipeline sizes) and can be semirigid or flexible if the head is controlled independently by cable or other moorings.

Gravity-fed downpipe (tremie). This technology consists of a large-diameter conduit extending from the surface through the water column to some point near or above the bottom. Dredged material would be placed into it either as a slurry or by being mechanically removed from a scow. Isolation from the water column is achieved, and placement accuracy is improved. However, little reduction in momentum or impact energy takes place over conventional bottom dumping. Because the conduit has a large cross-sectional area and is a rigid structure, site conditions (e.g., currents, water depth, sea state) would exert considerable influence on its use and cost.

Hopper dredge pumpdown. Some hopper dredges have pump-out capability by which material from the hoppers can be discharged like a conventional hydraulic pipeline dredge. In addition, some have further modifications that allow pumps to be reversed so that material can be pumped down through the dredge's

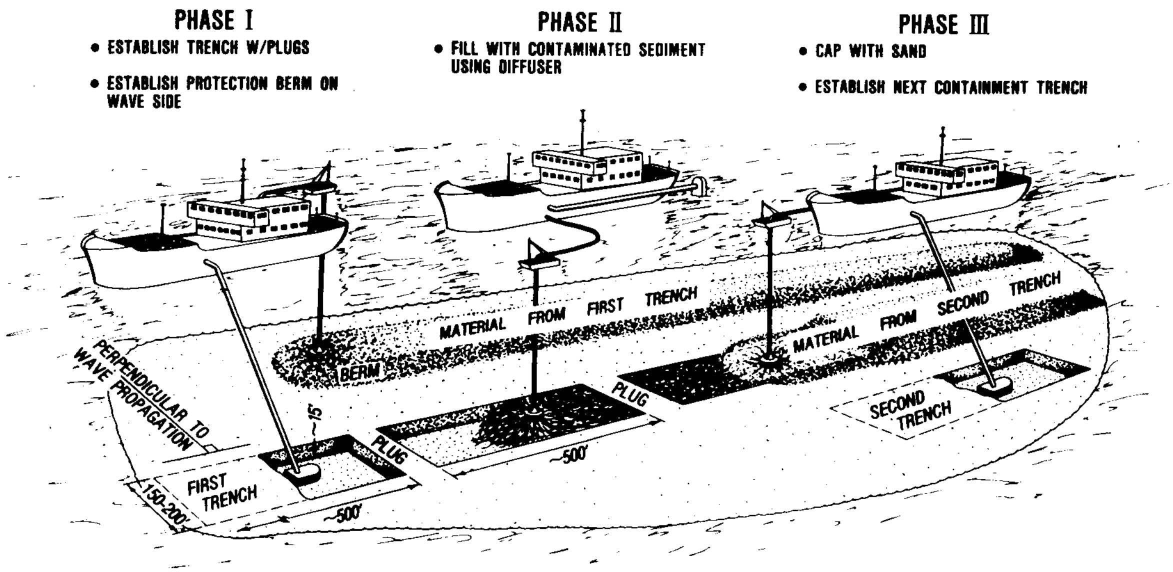

Figure 3. Conceptual design of CAD site using hopper dredge and submerged diffuser

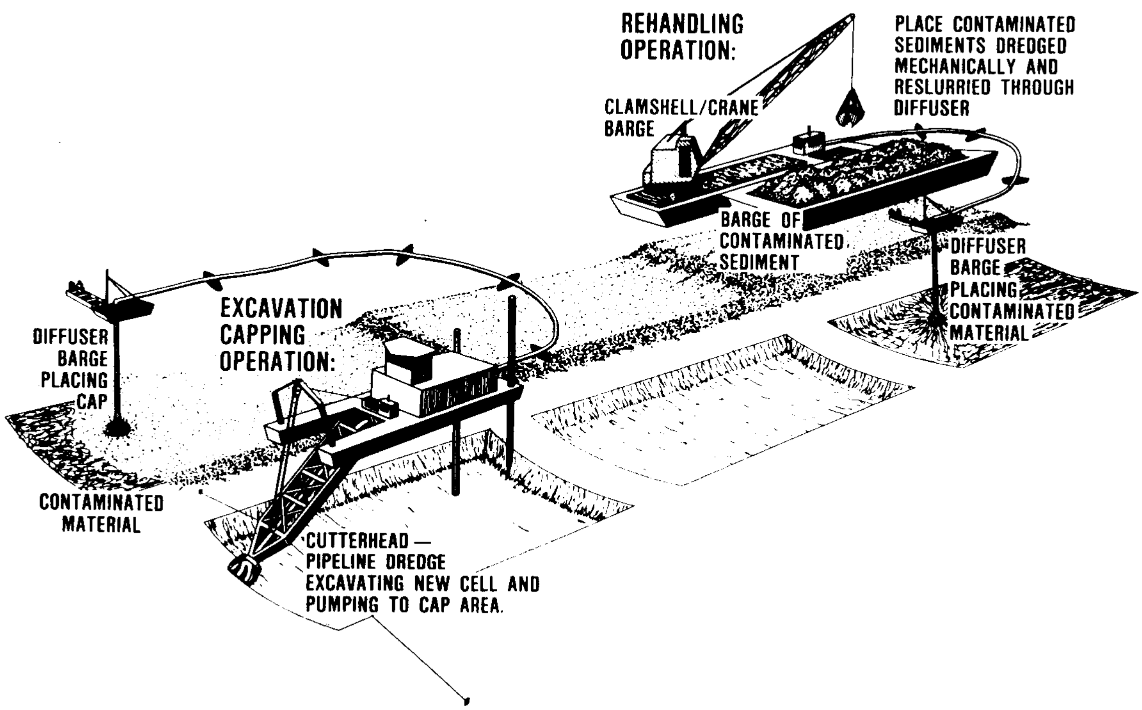

Figure 4. Conceptual design of CAD site using barge pumpout and submerged diffuser

extended dragarms. Because of the expansion at the draghead, the result is similar to use of a diffuser section. Pumpout depth is limited, however, to the maximum dredging depth of the hopper.

Monitoring

Monitoring at the disposal site must address both contaminant migration and physical condition of the site and must do so over time. Three basic categories of monitoring are suggested based on their time frames and intent.

1. Construction monitoring. Monitoring should take place before, during, and immediately following the construction operation. Background chemical characterization of the site will be necessary to serve as a baseline for comparisons. Water samples should be take during the placement of the contaminated dredged material and during capping primarily for monitoring resuspension in the area. However, the focus of the construction monitoring should be on bathymetry, accurate positioning during discharge, and accounting for the volume/mass of sediment handled. Moored buoys will be required at the site together with a real-time and recording positioning system. Replicate soundings must be taken frequently during placement of the dredged material and the capping material. Side-scan sonar and video equipment could also be used to verify conditions. Cores should be taken through the completed cap to verify its thickness and for sediment chemistry characterization.

2. Long-term monitoring. Similar water column sampling and sediment core series should be completed periodically after construction. Bathymetry and consolidation should also be measured at these intervals.

3. Contingency plans. In addition to the above regular monitoring, specific contingency plans should be developed to complete a similar monitoring series after a prespecified threshold storm event or ship incident.

Summary

A properly designed and placed cap provides the isolation necessary to control the movement of contaminants out of deposited dredged material into the overlying water column, and to prevent direct contact between aquatic biota and contaminants. It also performs the physical function of stabilizing the dredged material mound and protecting it from transport. Laboratory test methods are available to estimate the cap thickness required for isolation. However, this thickness is considered a minimum requirement and must be maintained in spite of erosion at the site.

Equipment and techniques for placing both dredged material and cap should consider the objectives of control and accuracy. Technologies such as the submerged diffuser are available to provide controlled accurate placement and to accomplish the additional benefit of isolating the material from the water column during descent.

Monitoring is an important aspect of construction verification and site management. Typical monitoring includes chemical characterization of site and deposited materials, bathymetry, mound consolidation, and cap thickness.

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EEDP-01-5
August 1989

Environmental Effects of Dredging Technical Notes

MONITORING DREDGED MATERIAL CONSOLIDATION AND SETTLEMENT AT AQUATIC DISPOSAL SITES

PURPOSE: This technical note provides information on methods for monitoring the consolidation and subsequent settlement of dredged material deposited at aquatic disposal sites. Information is given on methods that have been used by the Corps of Engineers (CE) at various aquatic disposal sites around the United States. Other methods are discussed that may prove useful in monitoring the consolidation and subsequent settlement of subaqueous dredged material deposits.

BACKGROUND: Each year approximately 120 million cu yd of dredged material are deposited at designated aquatic disposal sites around the United States. Placement of uncontaminated dredged material is typically conducted at level-bottom subaqueous disposal sites and results in the formation of a mound of material on the floor of the water body. Contaminated dredged material placed in aquatic disposal sites may be chemically and/or biologically isolated from the overlying water column by capping with clean dredged material.

Placement and subsequent capping of contaminated dredged material may be accomplished either at level-bottom disposal sites or in contained aquatic disposal (CAD) sites. CAD sites are natural or constructed depressions into which contaminated dredged material is placed and subsequently capped. The CAD disposal may be more effective in containment of contaminated material since lateral movement of the material is restricted and less surface area is exposed to the water column. Level-bottom disposal and CAD concepts are illustrated in Figures 1 and 2, respectively. Aquatic dredged material disposal sites have typically been located in water depths of 20 to 150 ft.

In conjunction with any of these aquatic disposal options for confining contaminated material, postdisposal monitoring of the dredged material deposit should be conducted. Monitoring of the behavior of constructed aquatic deposits is necessary to evaluate and predict the long-term physical and chemical stability of the deposit and to assist in determining the remaining disposal site capacity. Several methods are available for monitoring the settlement characteristics of subaqueous deposits of dredged material.

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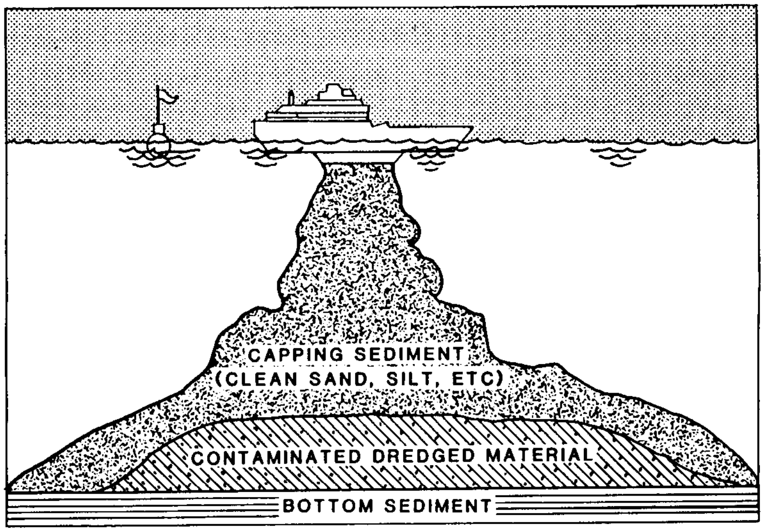

Figure 1. Schematic of typical level-bottom capping operation

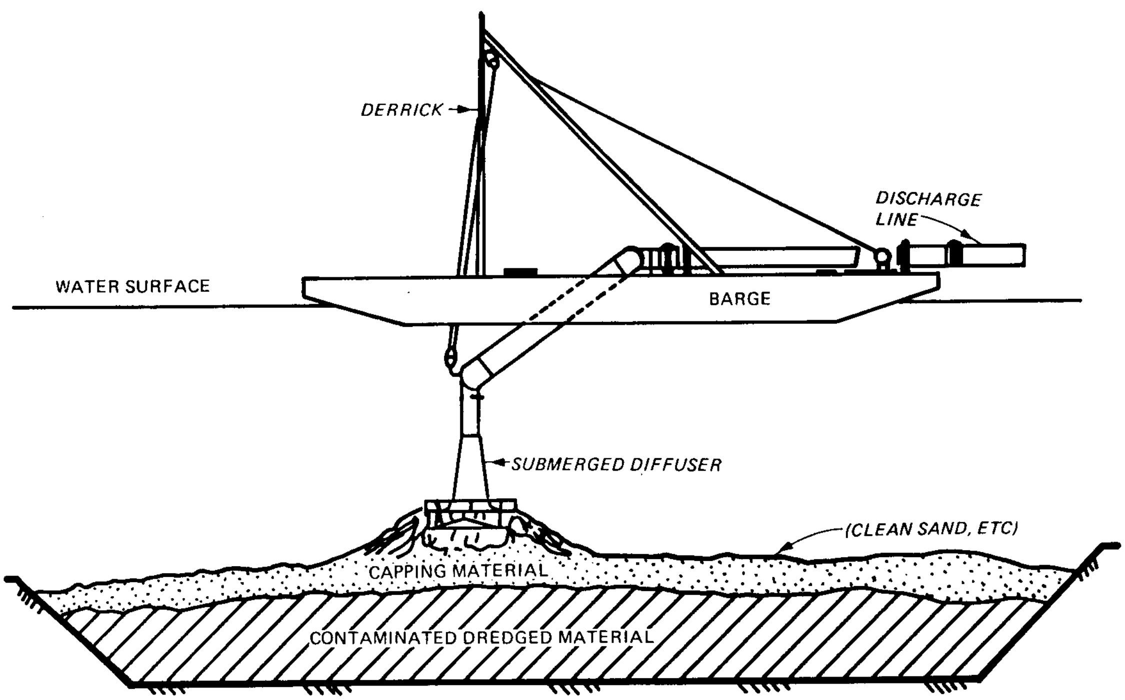

Figure 2. Schematic of CAD project showing use of a submerged diffuser for placement

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Postdisposal Behavior

After the dredged material and cap, if one is used, are placed at an aquatic disposal site, the material will undergo consolidation and may be exposed to erosion and transport from the disposal site (Poindexter 1988). Consolidation can occur in any one of or all three materials (if they are compressible): the capping material, the dredged material, and/or the underlying bottom sediments. As consolidation occurs in these materials, pore water is extruded from the deposit, and the shear strength of the material is increased. Extrusion of pore water results in a volume decrease of the deposited dredged material; this volume reduction is exhibited by settlement of the mound's upper surface. The increase in shear strength results in a deposit less susceptible to slope instability and to erosion.

During an investigation of the consolidation behavior of a dredged material deposit, the behavior of all compressible soil layers at that particular site should be considered and evaluated. Not only must the dredged material and any capping material be investigated and monitored, but any compressible foundation soil must also be evaluated. This is necessary so that any changes in elevation of the deposit's surface can be accounted for. It is not adequate to merely assume that a particular amount of consolidation will occur in the foundation soil. Instead, field and laboratory investigations should be conducted to determine whether compressible foundation or capping materials are present, and, if they are, consolidation tests should be run to enable prediction of the amount of consolidation that can be expected. The disposal site should then be monitored to discriminate any foundation consolidation from dredged material and/or cap consolidation.

Erosion and transport of the deposit's exposed surface material may occur if the disposal environment is such that current velocities exceed the critical shear stress for the material. The more cohesive an exposed material is, or the larger individual exposed particles are, the more resistant a material is to erosive/transport forces and, therefore, the more stable the mounds or deposits are. When planning for an aquatic disposal site deposit, the disposal site environment should be considered; e.g., bottom surface, depth of water, currents, and eroding versus accreting location, as well as properties of the material that will be on the surface of the deposit, whether it is dredged material or capping material (Shields and Montgomery 1984; Truitt 1986a, b, c; Dortch 1986; and Randall 1986).

Methods of Monitoring

A number of methods are available for monitoring the postdisposal behavior of subaqueous dredged material deposits. The various methods provide different types, quantities, and accuracies of information. The monitoring method(s) used should be selected to provide the required information and the desired level of accuracy for a particular disposal project.

The three most common methods of monitoring that have been successfully used by the CE (hydrographic surveys, settlement plates, and sediment sampling) are discussed in the following paragraphs. The type of equipment needed, its installation and use, the data provided, and the advantage/limitations of each monitoring method are included. Other commercially available monitoring techniques are then briefly mentioned.

Hydrographic surveys

By far, the most commonly used technique for monitoring settlement of subaqueous deposits is the hydrographic survey. Surveys of this type are typically used to monitor the changes in and condition of subaqueous features. Within the CE, this technique is most often used to evaluate the need for dredging and to verify the effectiveness of the dredging process in shipping channels, harbors, and turning basins. The technology of the hydrographic survey can be applied directly to monitoring the settlement characteristics of dredged material deposits.

Hydrographic surveys measure the depth of water between the survey boat and floor of the body of water. These surveys are usually conducted along parallel transects with equidistant spacing between the transects. The distance between readings taken on the transects and the spacing between adjacent transects determines the resolution of the grid of data collected. By correctly accounting for tidal fluctuations during the survey, elevation of the subaqueous sediment surface can be monitored and changes in elevation over time can be documented. More detailed information on planning and conducting hydrographic surveys can be found in another WES document (Fredette et al. in preparation).

The advantages of using the hydrographic survey are that the necessary equipment is generally available and the technique is applicable in the depths of water that may be encountered at aquatic dredged material disposal sites. A major disadvantage is the level of accuracy that can be attained. The typical accuracy of depth measurements from hydrographic surveys using standard CE equipment is ±6 to 12 in. at best (Clausner and Hands 1988). With this level of accuracy, it is difficult to make reliable measurements of changes in height of dredged material when the changes in height may range from a few inches to 1 to 2 ft. Horizontal positioning accuracy of the survey vessel is another factor which may affect the quality of the survey data.

An additional disadvantage is that hydrographic surveys provide only the total change in elevation of a deposit. The surveys give no indication of the consolidation of individual layers (foundation, dredged material, and capping material) present at a disposal site. Also, the method cannot be used to delineate between changes in mound height due to consolidation and those resulting from surface erosion of the deposited material.

<u>Settlement plates</u>

Settlement plates have been used for a number of years to monitor changes in thickness of various layers of dredged material in confined upland disposal sites. Periodically, settlement plates have been incorporated into the monitoring plans for aquatic disposal sites. The settlement plates described in the following paragraphs were used at an aquatic disposal site that was part of a capping demonstration project on the Duwamish Waterway in the Seattle District (Truitt 1986a, Poindexter 1988).

Telescoping settlement plates were used to measure changes in height of individual material layers at an aquatic dredged material disposal site (Figure 3). The lower tier plate was placed on the foundation soil before the dredged material was deposited. After dredged material disposal, the second tier settlement plate was slipped over the riser pipe of the lower tier and came to rest on the surface of the dredged material. After placement of the cap, the third tier settlement plate was placed over the riser pipe of the second tier, and the plate rested on the surface of the cap. Readings were subsequently made to determine changes in individual layer thicknesses. This provided settlement data for both the dredged material and the capping material. Since the elevation of the lower tier riser pipe had not been determined relative to a

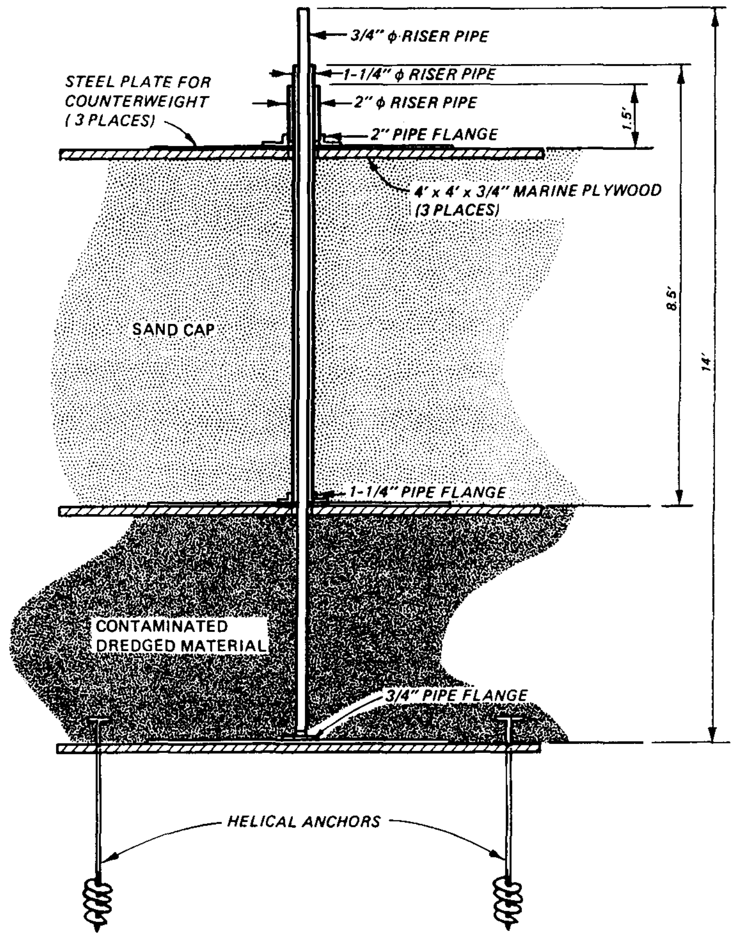

Figure 3. Tiered settlement plate that measures changes in height of individual layers of material (design 1)

stationary benchmark outside the disposal site boundaries, settlement of the foundation soils could not be determined.

Two telescoping settlement plate designs were used at the Duwamish Waterway Site (Poindexter 1988). The major difference in the settlement plates was in the diameter of the riser pipes. Design 1 used a 3/4-in.-diameter pipe as the center pipe, while design 2 used a 2-in.-diameter pipe in the center. Design 2 also used polyvinyl chloride (PVC) pipe on the second and third tier settlement plates since the required diameters of these risers were large and weight of the entire settlement plate assembly needed to be kept to a minimum. Settlement plates in the second and third tiers were designed and fabricated to have a unit weight approximating that of water so that the plates would not sink through the soft dredged material or cause consolidation of the underlying material by acting as a surcharge load.

The two settlement plate designs were used to evaluate the effectiveness of each with regard to withstanding the forces of dredged material disposal and minimizing surface scour after material deposition. Design 1 pipes performed satisfactorily in both aspects. Because of problems encountered during settlement plate installation, no definitive information was obtained on Design 2 pipes.

The advantage of using tiered settlement plates is that exact changes in thickness (settlement) of the various layers of deposited material can be obtained. Furthermore if the elevation of riser pipe from the lower tier settlement plate is related to a known elevation outside of the disposal site, then settlement of compressible foundation soil can also be monitored. When noncompressible material is used as the cap, any changes in cap thickness can be attributed to erosion. Disadvantages of this method are that divers must be used to place the plates and to obtain the settlement readings and the riser pipes/settlement plates may be accidentally disturbed or removed by anchors, cables, or fishing nets. They may also be damaged by the disposal process.

Sediment sampling

After placement of dredged material and capping material, core borings can be taken at specified time intervals to determine profiles of engineering properties. This provides a means of monitoring temporal changes in physical characteristics at the capped site.

Core borings of the sediment to be dredged and deposited dredged material provide information concerning types of material involved in the disposal operation; this information is useful in predicting anticipated behavior of the material as well as in interpreting and understanding observed field behavior (e.g., rate of consolidation and possible erodibility of the material). Sampling also provides data on water contents/void ratios of the material at various times during the dredging/disposal operation; this will allow determination of the effect of various dredging/disposal activities on sediment characteristics. Void ratio data provide needed information about conditions during the consolidation process.

Several methods are available for obtaining samples of sediment before dredging or after deposition of the dredged material at the disposal site. The most commonly used sampling devices are the Vibracore sampler and the gravity piston sampler (also known as the drop tube sampler). However, the sampling method that provides the best undisturbed sample is the Osterberg sampler.

Vibracore sampler. The Vibracore sampler is a device that has been used successfully to obtain samples of sediment from aquatic or open-water environments. Typically 3-in.-diameter cores are taken. The individual sample length is typically 20 ft, although some small devices are only capable of taking 5- to 10-ft samples. Some larger devices may be modified to take samples of 30- or 40-ft lengths. The Vibracore is generally used to sample sands; it has been used to sample some fine-grained material, but the success rate has not been as great for these materials.* A typical Vibracore sampler is shown in Figure 4.

The Vibracore consists of a steel barrel with a plexiglass liner for sample collection and a vibratory driving mechanism mounted on a four-legged tower guide and platform (US Army Engineer District, Savannah 1967). The entire assembly is lowered through the water to the substrate surface by a crane/cable hoist system. After the device has been accurately positioned on the bottom through

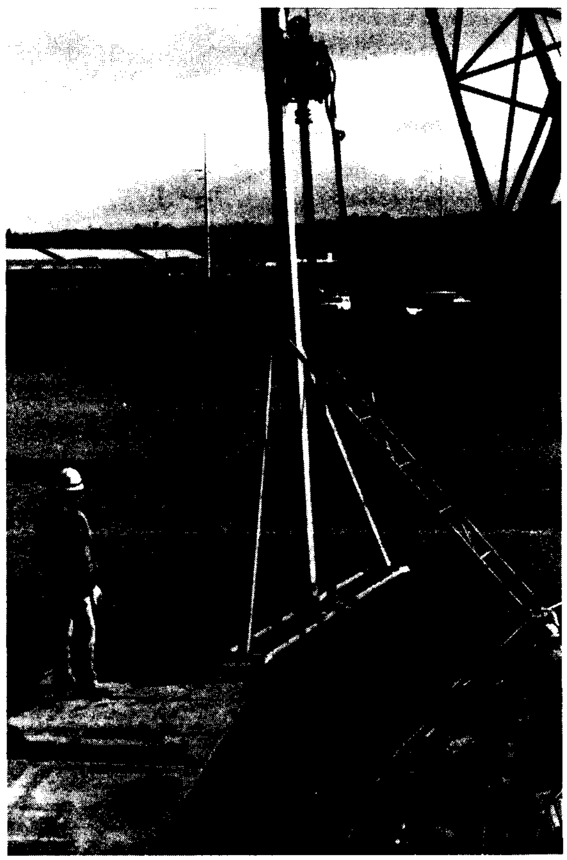

Figure 4. Typical Vibracore sampler being lowered to a subaqueous sampling site

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* Patrick A. Douglas, US Army Engineer District, Mobile, personal communication.

the use of standard surveying equipment or navigation-positioning equipment, compressed air is supplied to the vibratory unit through flexible hoses extending from the floating plant down to the Vibracore. Upon application of the compressed air, the oscillating hammer (vibrator) propels the core barrel into the subbottom material. The Vibracore can be equipped with a penetration-recording device that provides a record of the penetration depth and time. After the core barrel has been extended to its full length or until it resists further penetration, the sampler is retracted from the substrate and returned to the floating plant deck. The plastic core barrel containing the sample is then removed from the sampling device, and the ends are capped for sample preservation. The core barrel is later cut open longitudinally to expose the material for visual inspection and collection of specimens for laboratory testing.

Advantages of the Vibracore sampler are the ease, speed, and low cost of sampling by this method. Typically eight to twelve 20-ft cores may be obtained in one day by an experienced sampling crew (US Army Engineer District, Savannah 1967). The major disadvantage is that the vibratory method of driving the sample tube can cause changes in the density of materials sampled: loose sand and silt may be densified while dense material may be loosened during sampling. An additional problem may be encountered if a soft material is overlain by a firmer stratum. In this case, the soft material will be pushed aside instead of entering the sample tube if the shear strength of the soft material is less than the force required to overcome the friction between the firmer material and the sample tube. If a Vibracore sampler is to be used to collect samples from aquatic disposal sites, it is recommended that the penetration-recording device be acquired and used to provide definitive information on depth of penetration.

Gravity core sampler. The gravity core sampler has been used on a number of disposal area monitoring projects. The diameter of the sample typically varies from 3 to 6 in., with the 3-in. diameter being more common. The length of sample retrieved can range from 3 to 20 ft, depending upon the particular equipment used (Stanton, Demars, and Long 1985).

The gravity core sampler consists of a core barrel, penetration weights, and stabilizing fins. The core barrel is equipped with a plastic liner and has a cutting head on the lower end. As the sampling device is lowered through the water, a triggering device is held in place by the tension in a line that is attached to a weight. When the weight reaches the substrate surface, the tension in the line is released and the sampler drops to the bottom and penetrates the

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Other methods

Several remote methods of sampling may prove to be useful in monitoring aquatic disposal sites. These methods include side-scan sonar, subbottom profiling, and various other settlement/pore-pressure monitoring techniques.

In side-scan sonar systems, acoustic energy is projected laterally from a pair of transducers mounted in a cylindrical body (called the "fish") that is towed behind a boat. Electrical energy applied to the piezoelectric transducers in the fish causes them to vibrate, creating pressure waves that travel out through the water. The energy is reflected back from the seabed or structure, picked up by the transducers, and recorded to produce a sonograph. Transducers typically vibrate at 50 to 500 kHz, with 100 and 500 kHz being most common. The 100-kHz frequency provides greater range, up to 1,500 ft on either side, and is most often used for sea-bottom mapping and locating objects. A frequency of 500 kHz gives a shorter range, up to 300 ft on either side, but provides greater detail (Clausner and Hands 1988; Truitt 1986; Coastal Engineering Research Center 1983).

A subbottom profiler operates in the same manner as the side-scan sonar, but it uses a lower frequency acoustic pulse which penetrates the sediments on the bottom. A 3.5- to 14-kHz frequency pulse is typically used for these instruments. The subbottom profiler is pointed straight down and produces an image that delineates the sediment surface and the sediment layers below the surface. In order for the various layers to be distinguishable, there must be a significant difference in material types and the various layers must be at least 2 ft thick. Additional detailed information on acoustical surveying and monitoring techniques may be obtained from Clausner and Hands (1988).

Various techniques have been used on land to investigate both the stratigraphy of an area and the consolidation settlement that occurs. Some of these techniques might be applicable to aquatic dredged material disposal sites. Techniques that might prove useful include settlement probes, liquid settlement systems, and pore-water pressure probes.

Settlement probes of various types can be used as a downhole tool in a borehole. When inserted into a borehole, a settlement probe measures the depth/location of particular objects outside the borehole or attached to the casing; these objects are stationary relative to the adjacent soil. Periodic monitoring can be used to document the consolidation of various layers.

Liquid settlement systems monitor changes in pressure head in a closed system to measure any settlement that occurs. A transducer would normally be installed at the point of interest within the disposal site and the reference liquid reservoir would be installed in a stable location outside the site. Hydraulic lines are needed to connect the transducer to the reference reservoir. A separate system would be required for each point to be monitored within the dredged material deposit.

A pore-pressure probe measures the pore-water pressure existing with depth throughout a soil deposit as the probe is pushed through the soil. Instantaneous readings provide accurate data in sand or other free-draining deposits. In fine-grained materials, probe-induced pore pressures will build up during the process of pushing the probe to the desired location for a reading; therefore, a short time delay must be allowed before the pore-pressure reading is taken in order to obtain an accurate reading.

Summary

When an aquatic site is used for disposal of dredged material, a postdisposal monitoring program should be established to evaluate the stability of the deposit, provide site-capacity data, and expand the available knowledge of the behavior of these deposits for future predictive purposes. The consolidation behavior of all compressible materials, including the dredged material, cap, and foundation soil, should be monitored. A number of monitoring techniques are available. The most commonly used methods are the hydrographic survey, settlement plates, and sediment sampling. Other methods are available but have not been proven in the aquatic dredged material disposal site environment.

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EEDP-01-6
March 1987

Environmental Effects of Dredging Technical Notes

COMPUTERIZED DATABASE FOR INTERPRETATION OF THE RELATIONSHIP BETWEEN CONTAMINANT TISSUE RESIDUES AND BIOLOGICAL EFFECTS IN AQUATIC ORGANISMS

PURPOSE: This note provides initial information on the format and potential application of a computerized database in conducting certain types of literature searches. The ultimate goal of this task is to provide Corps elements with numerical as well as descriptive guidance so that they can relate contaminant tissue residues to biological effects in aquatic organisms in a more accurate, consistent, and technically defensible fashion.

BACKGROUND: Over the last 10 years, only a small number of sediments evaluated in regulatory testing programs have been found to be acutely toxic. Consequently, decisionmakers have relied less on toxicity and more heavily on bioaccumulation information. Unfortunately, there is little generally accepted interpretive guidance regarding the biological importance of bioaccumulation in aquatic organisms (Peddicord and Hansen 1983). In an effort to provide some initial guidance in this area, an assessment of the literature was conducted under the Long-Term Effects of Dredging Operations (LEDO) Program in which the association between bioaccumulation and biological effects in aquatic organisms was examined (Dillon 1984). Major findings of this initial literature review and assessment can be summarized as follows:

a. Only 6 percent of 2181 publications reporting biological effects information also contained contaminant residue data. This narrow database effectively limits numerical identification of specific biological threshold concentrations.

b. Of all the available biological end points to consider, reproductive effects as well as some measure of growth in aquatic organisms appear to be the best candidates for the subacute bioassessment of dredged material in a regulatory program.

c. Whole-animal (organismic) evaluations represent a reasonable and technically defensible compromise between biochemical assessments, which are potentially more sensitive, and population/community assessments, which generally have more ecological relevance.

d. A majority (67 percent) of 2181 publications reported exposure of aquatic organisms to contaminants dissolved in aqueous solution. Only 7 percent evaluated the biological effects after exposure to contaminated sediment.

Based on results of the initial review of the literature, a more intensive look at the reproductive end point was undertaken by Dillon and Gibson (1985). They reported on the most frequently examined contaminant (cadmium), organism (fish), and reproductive end point (hatching success). In an effort to increase the number of reports considered for review (see subparagraph a above), published bioconcentration factors (BCF) were used by Dillon and Gibson to estimate tissue concentrations from data in those papers containing biological effects data but no tissue residue information.

Both of these assessments of published literature and the numerical calculations contained within them were performed manually using index cards and desk calculators. To provide a more rapid and comprehensive information retrieval system, computer hardware and appropriate software were acquired. This technical note describes progress to date in application of the system to develop and test a database from the literature on sublethal effects of contaminants in aquatic organisms.

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The Computerized Database: An Overview

In the spring of 1986, appropriate computer software to carry out an automated literature assessment was identified. The program was initially designed in 1983 and evolved into two software components: the Manager and the Searcher. A third component, the Editor, was added in 1986.

The program is the first text management system designed especially for bibliographic citations. Each citation entered can be any length and format and can include any additional descriptive notes of interest. Searches can be conducted by specifying any character, word, name, or phrase that is found anywhere in any entry or by specifying unique accession numbers assigned to the entries.

The Manager lets the operator manage information with minimum effort and maximum flexibility. Data are organized into records, which are units of related information contained in specific fields. The names of the fields are stored in a template, which can be easily designed or customized by the user to fit specific needs or preferences. A bibliographic template ordinarily consists of fields such as author, title, year, journal, keywords, etc. Additional fields can be added to build a powerful database of information gleaned from the literature. An example is described in the next section.

Using the file system, information can be stored, sorted, searched, and retrieved quickly and efficiently. The Manager can search an entire record or selected fields for specific words and phrases, sort records by as many as six criteria (or keys), and generate reports.

Each of the three software products is designed to work independently of the others; at the same time, all three can be integrated to organize and process information in countless ways. The Searcher enables the user to access hundreds of on-line databases available from five different on-line services: BRS, DIALOG, NLM, ORBIT, and QUESTEL. Records from these databases can be downloaded and then transferred to user files with the Manager. The Editor lets the operator develop formats for bibliographic citations and use references stored by the Manager or downloaded by the Searcher from on-line databases. References can be easily formatted to meet organization- or journal-specific literature citation requirements.

Computerized Literature Review

The program is being used to conduct an assessment of the literature on sublethal effects of contaminants on aquatic organisms and to evaluate the software system, which was installed on a microcomputer running under PC-DOS. The input of data for the review, which began in May 1986, and the system evaluation are described in the following paragraphs.

Before a keyboard entry can be made of information retrieved from the open literature, a file of records must be created. An example of the input format for a record of the BIOCON (biological consequences) user file in the Manager is shown in Figure 1. For every paper that is reviewed, the following information is recorded into a template: author, reference (REF); biological response; contaminant (CONTAM); species; phylogeny (PHYLOG); aquatic medium; tissues; route of exposure; whether tissue concentrations were reported or estimated (R.VS.E.); life stages; contaminant exposure time; exposure concentration (EXP CON); tissue concentration (TIS CON) if reported or estimated; and any observed change in reproductive activity (EFFECTS).

Each publication is examined for reports of the highest tissue

User File: BIOCON Template: 14 Accession Number: 17

AUTHOR	Biesinger, K. E., L. E. Anderson and J. G. Eaton 1982			
REF	Chronic effects of inorganic and organic mercury on *Daphnia magna*: toxicity, accumulation and loss. Arch. Environ. Contam. Toxicol. 11:769-744			
RESPONSE	reproduction			
CONTAM	mercury			
SPECIES	water flea (*Daphnia magna*)			
PHYLOG	Arthropoda, Crustacea			
AQ.MED.	fresh water			
TISSUES*	whole body, wet weight			
ROUTE	water			
R.VS.E.	R.			
STAGES	12 ± 12 hr			
TIME	21 days			
EXP CON	reproduction	$HgCl_2$	water flea	0.36-0.72 μg/ℓ 1.28 2.70
TIS CON	reproduction	$HgCl_2$	water flea	8.13-17.15 μg/g 28.90 60.93
LEC	reproduction	$HgCl_2$	water flea	28.90 μg/g
HNEC	reproduction	$HgCl_2$	water flea	17.15 μg/g
EFFECTS	reproduction $HgCl_2$:	0.36-0.72 μg/ℓ - no effect on number of young produced 1.28 μg/ℓ - decreased number of young produced 2.70 μg/ℓ - 100% mortality		
COMMENTS				

* Tissue concentrations are reported in units of micrograms per gram (μg/g) wet weight. Data originally reported on a dry-weight basis were converted to wet weight assuming 80-percent body water.

Figure 1. Sample format for record in the Manager

concentration at which no effect on reproduction was observed, as well as for the lowest tissue concentration at which an effect was observed. These values are entered as the Highest No Effects Concentration (HNEC) and the Lowest Effects Concentration (LEC). Tissue concentrations are expressed on a wet-weight basis. Exposure concentrations are expressed in micrograms per liter (parts per billion) unless noted otherwise.

The use of the program in performing a literature review makes data entry quick and efficient. Moreover, records formatted with templates may be searched and sorted in numerous ways. For example, one may be reviewing the results of a 10-day bioaccumulation study involving contaminated sediment and polychaete worms. To assist in interpreting the results of the test, the database would be searched for all citations dealing with polychaetes, and a summary of the results could be requested. Likewise, if one would like to know the relative toxicity of various contaminants, relevant literature could be accessed and summarized. The search can be tailored to individual needs such as fresh water versus salt water, specific organisms, and/or various contaminants, etc.

The program lacks one capability that proved to be a limitation during the present literature review. All characters, including numbers, are treated as text and thus cannot be used for arithmetic computations. A user could retrieve all HNEC data relating to a specific contaminant, for example, but could not use the program to calculate a mean of the HNEC values. This limitation also means that comparison operators (<, >, etc.) are not available for searches. A user would not be able to search for all HNEC greater than a specified value, for example. It is hoped that this limitation will be addressed in future product upgrades. For the present, this disadvantage is considered to be minor and is far outweighed by the power, simplicity, and modest cost of the system.

Future Plans

Efforts will continue on establishing the database and making the system available to Corps field offices. Meanwhile, persons requiring numerical and/or descriptive information relating tissue residues to biological effects in aquatic organisms may contact the authors of this technical note.

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EEDP-01-7
March 1987

Environmental Effects of Dredging Technical Notes

USE OF *DAPHNIA MAGNA* TO PREDICT CONSEQUENCES OF BIOACCUMULATION

PURPOSE: Results reported herein represent a portion of the laboratory research evaluating the relationship between mercury and cadmium tissue residues and biological effects in the freshwater crustacean, *Daphnia magna* (commonly known as the water flea). Procedures presented here for a 28-day *Daphnia magna* toxicity test could be used in screening for water-column toxicity resulting from open-water disposal of a specific dredged material.

BACKGROUND: As a part of its regulatory and dredging programs, the U. S. Army Corps of Engineers often conducts, or requires to be conducted, an assessment of the potential for bioaccumulation of environmental contaminants from sediment scheduled for dredging and open-water disposal. There is, at present, no generally accepted guidance available to aid in the interpretation of the biological consequences of bioaccumulation. To provide an initial basis for such guidance, the Environmental Laboratory is conducting both literature database analyses and experimental laboratory studies as part of the Long-Term Effects of Dredging Operations (LEDO) Program.

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Materials and Methods

Laboratory cultures of *Daphnia magna* were maintained in 1-ℓ culture dishes (eight adult *Daphnia* per dish) set in a constant temperature water bath at 20.0° C with a 14-hr photoperiod. Reconstituted water with a hardness of 180 mg/ℓ, as $CaCO_3$, and a pH of 8.2 was used in the culture medium (Dunbar et al. 1983). The *Daphnia* were fed every day (except Sunday) from a laboratory culture of the green algae, *Ankistrodesmus falcatus*, at a ration equivalent to 1.71 mg of dry algae for each container.

Exposure to mercury and cadmium began with <24-hr-old neonates having a mean dry weight of 21.0 μg and a mean length of 1.58 mm. The *Daphnia* were exposed to 0.0, 0.05, 0.1, 0.5 and 1.0 μg/ℓ mercury and, separately, to 0.0, 0.1, 0.5, 1.0 and 5.0 μg/ℓ cadmium under static renewal conditions. There were 12 replicate beakers per concentration and each contained 200 ml of water and 2 *Daphnia*. Each beaker was covered with a black petri dish. Reconstituted hard water was used throughout the test, and temperature, photoperiod, and feeding ration were identical to that used to maintain the laboratory *Daphnia* culture.

All beakers were checked daily for mortality, neonate production, and any abnormal behavior. Mortality was defined as cessation of all visible signs of movement of the second antennae, respiratory appendages, and the postabdomen after 5 sec of observation (Buikema et al. 1976). When discovered, neonates and dead adults were removed, counted, and discarded. Test solutions in all beakers were renewed each Monday, Wednesday, and Friday. Microliter volumes of mercury or cadmium were added from stock solutions prepared with mercuric chloride and cadmium chloride, respectively, dissolved in reverse osmosis (R.O.) water.

At the termination of the test, the *Daphnia* were rinsed three times in R.O. water. Lengths were determined by measuring from the top of the head to the base of the caudal spine using a dissection scope equipped with an ocular micrometer. The *Daphnia* were then individually placed in preweighed aluminum foil pans and dried for 24 hr at 70° C. After cooling in a desiccator for 2 hr, the *Daphnia* were weighed and dry weights were obtained to the nearest 1.0 μg. Eight daphnids were pooled per sample (three samples per treatment) in 20-ml vials containing 1.0 ml of 50-percent nitric acid, HNO_3. After digestion at 70° C for 24 hr, volumes were adjusted to 6.0 ml with R.O. water and analyzed for total mercury via atomic absorption after gold amalgamation formation. Cadmium tissue and water samples were analyzed via atomic absorption spectroscopy. Water samples were collected immediately after one renewal period during the experiment. Water from four replicate beakers was combined to yield one pooled water sample of 125-ml volume. There were three such pooled water samples per treatment utilizing water from all 12 replicate beakers. Each water sample was acidified to a pH of <2.0 with 1.0 ml of concentrated HNO_3.

To evaluate whether mercury and cadmium were quantitatively affecting the food source and therefore introducing a nontreatment bias to the biological endpoints, an algal toxicity test was conducted with the green algae food source concurrently with the *Daphnia* exposure. All exposure conditions were identical to the *Daphnia* test except that there were three replicate beakers per metal concentration and no *Daphnia* were present in the beakers. At the end of 48 hr, algae were spun down in a centrifuge at 6089 g's for 10 min. Excess water was siphoned off, and algal pellets were resuspended in 10 ml of R.O. water. Samples were counted in a Neubauer counting chamber at 40X.

Treatment effects on mortality, growth, and reproduction were evaluated by one-way analysis of variance. The Waller-Duncan K-ratio t-test was used to separate means. Differences were considered statistically significant for $p < 0.05$.

Results

Survival

Mercury. *Daphnia* exposed to the highest mercury concentration (1.0 μg/ℓ) showed only 17-percent survival by the end of the 28-day experiment. Except for one *Daphnia* in the control, all those exposed to 0.0, 0.05, and 0.1 μg/ℓ survived the 28-day test (Figure 1). Survival of the *Daphnia* in the 0.5-μg/ℓ concentration at day 28 was intermediate (75 percent) to the other treatments.

Approximately half of the *Daphnia* that died in the 1.0-μg/ℓ treatment exhibited complete loss of setae and distal segments of the second antennae (locomotor appendages) approximately 3 to 5 days prior to death. The diminutive antennae could not maintain the *Daphnia* in the water column, its normal habitat. Affected organisms in the 1.0-μg/ℓ treatment propelled themselves along the bottom of the beaker in short jerky motions.

Cadmium. *Daphnia* exposed to the highest cadmium concentration (5.0 μg/ℓ) exhibited 100-percent mortality by day 21, while all those exposed to the three lowest concentrations (0.0, 0.1, 0.5 μg/ℓ) survived the 28-day experiment (Figure 2). Not surprisingly, survival of *Daphnia* in the remaining exposure concentration, 1.0 μg/ℓ, was intermediate to the other treatments.

Growth

Mercury. When expressed as mean lengths, growth was not significantly

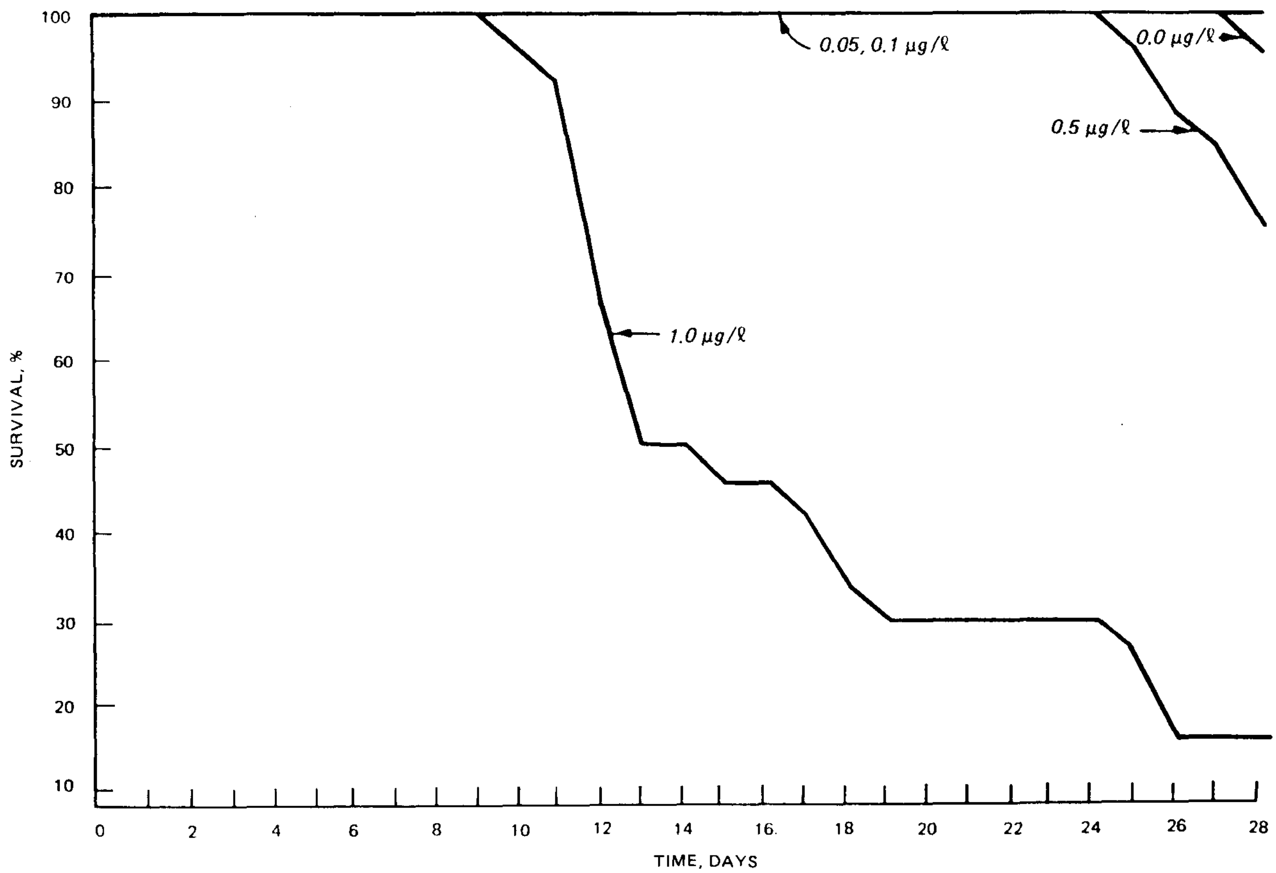

Figure 1. Survival of *Daphnia magna* during 28-day exposure to mercury

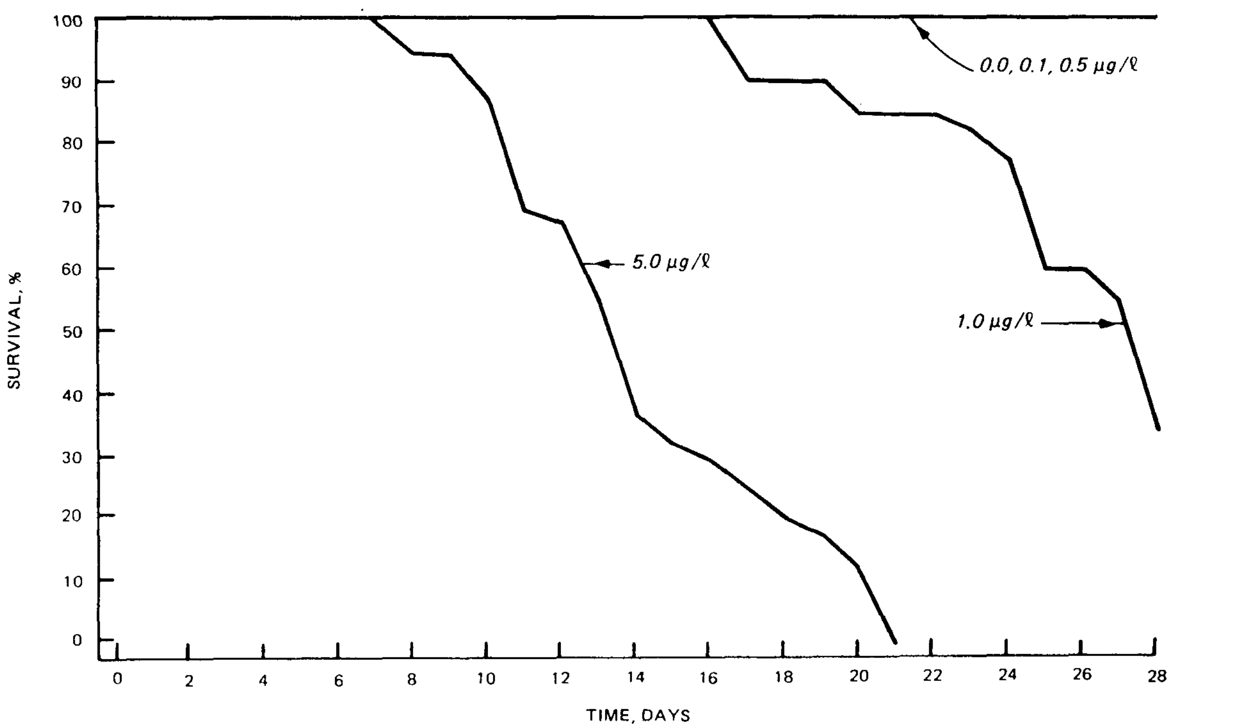

Figure 2. Survival of *Daphnia magna* during 28-day exposure to cadmium

affected by mercury exposure (Table 1). Mean lengths ranged from 5.12 to 5.33 mm. However, growth, expressed as dry weight, was significantly affected in an unexpected dose-related manner. The mean dry weight of *Daphnia* in the highest treatment (1.0 μg/ℓ) was 795 μg, which was significantly greater than the dry weights observed in the three lowest exposure concentrations (0.0, 0.05, 0.1 μg/ℓ). In those three groups, mean dry weights ranged from 511 to 558 μg. The mean dry weight of *Daphnia* exposed to 0.5 μg/ℓ was intermediate (610 μg) and significantly different from all other treatments.

Cadmium. Growth was significantly reduced in *Daphnia* exposed to 1.0 μg/ℓ cadmium compared to those in the three lower exposure concentrations (Table 2). Although growth data were not collected for *Daphnia* in the 5.0-μg/ℓ treatment due to high mortalities, daily observations indicated that these organisms were much smaller than those in the other treatments. Both measures of growth (mean lengths and dry weights) were significantly lower in the 1.0-μg/ℓ treatment (4.94 mm and 405 μg, respectively) compared to the three lower concentrations. In those three groups, mean lengths ranged from 5.20 to 5.26 mm and dry weights from 508 to 544 μg and were not significantly different from each other.

Reproduction

Mercury. There were no significant differences among mercury treatments for two measures of reproduction, i.e., time to first egg production or total neonates produced per female (Table 1). Total neonates produced per female ranged from 42 to 35. There was a significant depression in the total number of neonates produced per beaker in *Daphnia* exposed to the highest mercury concentration (1.0 μg/ℓ). This reduced production was due to mortality of the adults and not to a reduction in reproduction, per se.

Mean time to first appearance of eggs in the brood chamber ranged very narrowly from 5.9 to 6.0 days and was also not significantly affected by the mercury exposure (Figure 3). These similar mean values imply that egg production was extremely synchronous. This synchrony persisted throughout the experiment as evidenced by the appearance of successive broods (Figure 3). Six distinct broods were produced during the 28-day experiment with peaks in production occurring on days 9, 12, 16, 19, 23, and 26. Mercury does not appear to affect the timing of successive broods production.

Cadmium. The level of cadmium exposure affecting reproduction was similar to that observed for growth and survival. Again, there were no

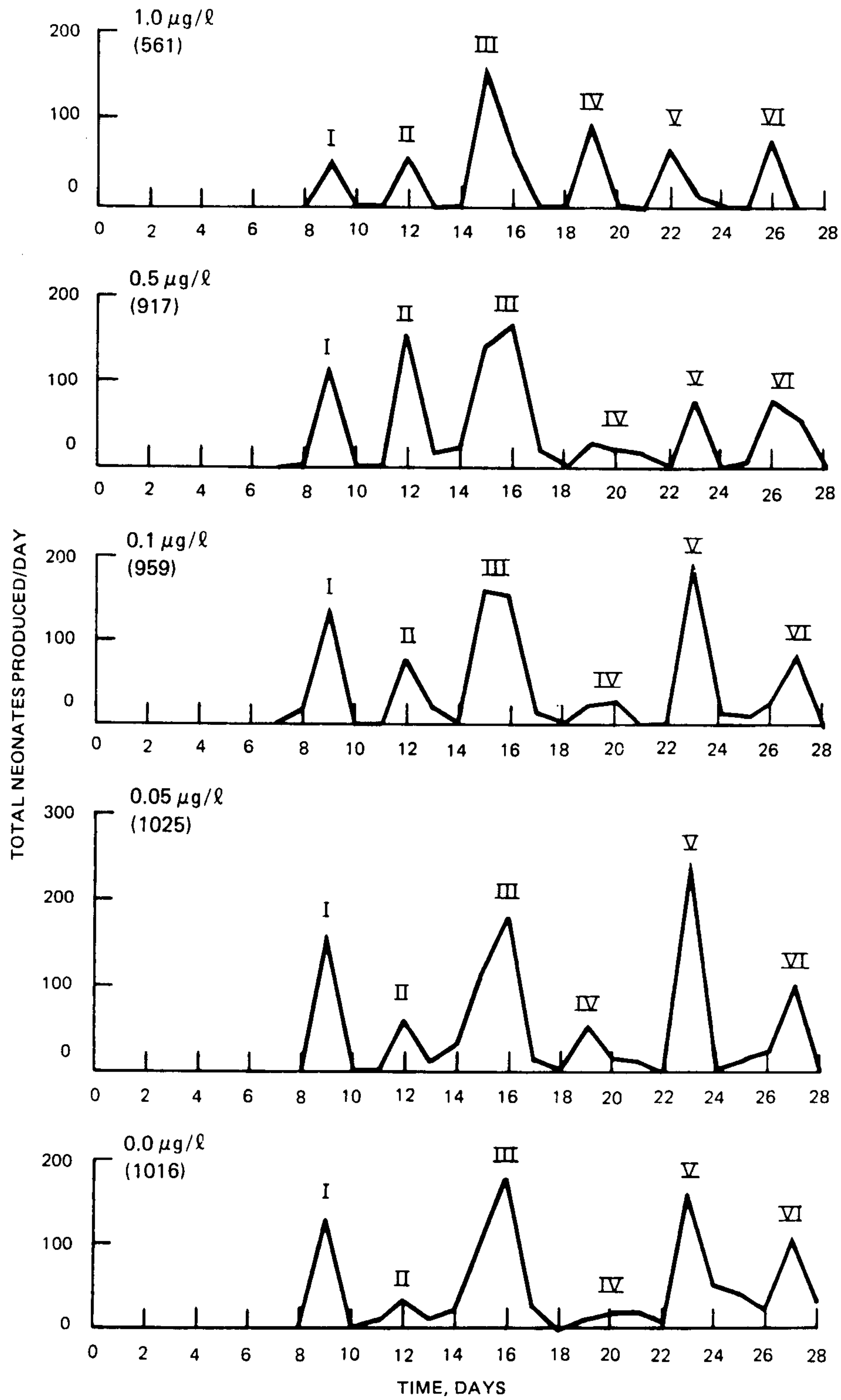

Figure 3. Daily neonate production of *Daphnia magna* during 28-day exposure to mercury. Values in parentheses represent the total production through 28 days. Roman numerals signify peaks in successive broods

significant differences among the 0.0-, 0.1-, and 0.5-μg/ℓ treatments for either measure of reproduction, i.e., time to first egg production or total neonates produced per female (Table 2). Reproduction was significantly depressed in *Daphnia* exposed to 1.0 and 5.0 μg/ℓ cadmium. Total neonates per female for *Daphnia* exposed to 1.0 and 5.0 μg/ℓ were 7.1 and 0.7, respectively. These values are significantly less than those for *Daphnia* in the lower exposure treatments (0.0, 0.1, and 0.5 μg/ℓ) in which mean values ranged from 39 to 42.

There was a significant depression in the total number of neonates produced per beaker exposed to 1.0 and 5.0 μg/ℓ cadmium. Time to first appearance of eggs in the brood chamber was also slightly but significantly delayed in *Daphnia* exposed to 1.0 and 5.0 μg/ℓ cadmium (6.4 days) compared to all other treatments (6.0 days) (Table 2). These statistically significant differences between very similar mean values are probably not biologically important but do imply that egg production was extremely synchronous. Indeed this synchrony persisted throughout the experiment as evidenced by the appearance of successive broods (Figure 4). Six distinct broods were produced during the 28-day experiment with peaks in brood production occurring on days 9, 12, 16, 20, 23, and 27. Cadmium does not appear to affect the timing of successive broods production.

Tissue residues

Mercury. Mean mercury tissue concentrations for *Daphnia* exposed to 0.0, 0.05, 0.1, 0.5, and 1.0 μg/ℓ, expressed on a dry-weight basis, were < 0.20, 0.69, 1.92, 5.47, and 5.47 μg/g, respectively. Due to high mortalities, only one sample consisting of four surviving *Daphnia* was available in the 1.0-μg/ℓ treatment at day 28. However, the concentration of mercury in this tissue sample along with that in the 0.5-μg/ℓ treatment were significantly greater than observed in the three lower treatments.

Cadmium. Measured water concentrations of cadmium in the exposure beakers were very similar to calculated concentrations (Table 3). Mean cadmium tissue concentrations for *Daphnia* exposed to 0.0, 0.1, and 0.5 μg/ℓ, expressed on a dry-weight basis, were 3.55, 4.49, and 6.58 μg/g, respectively. None of these values was significantly different from one another. Due to high mortalities, no tissue samples were collected from the 5.0-μg/ℓ treatment, and only a single sample consisting of the eight surviving *Daphnia* was available in the 1.0-μg/ℓ treatment at day 28. However, cadmium in this

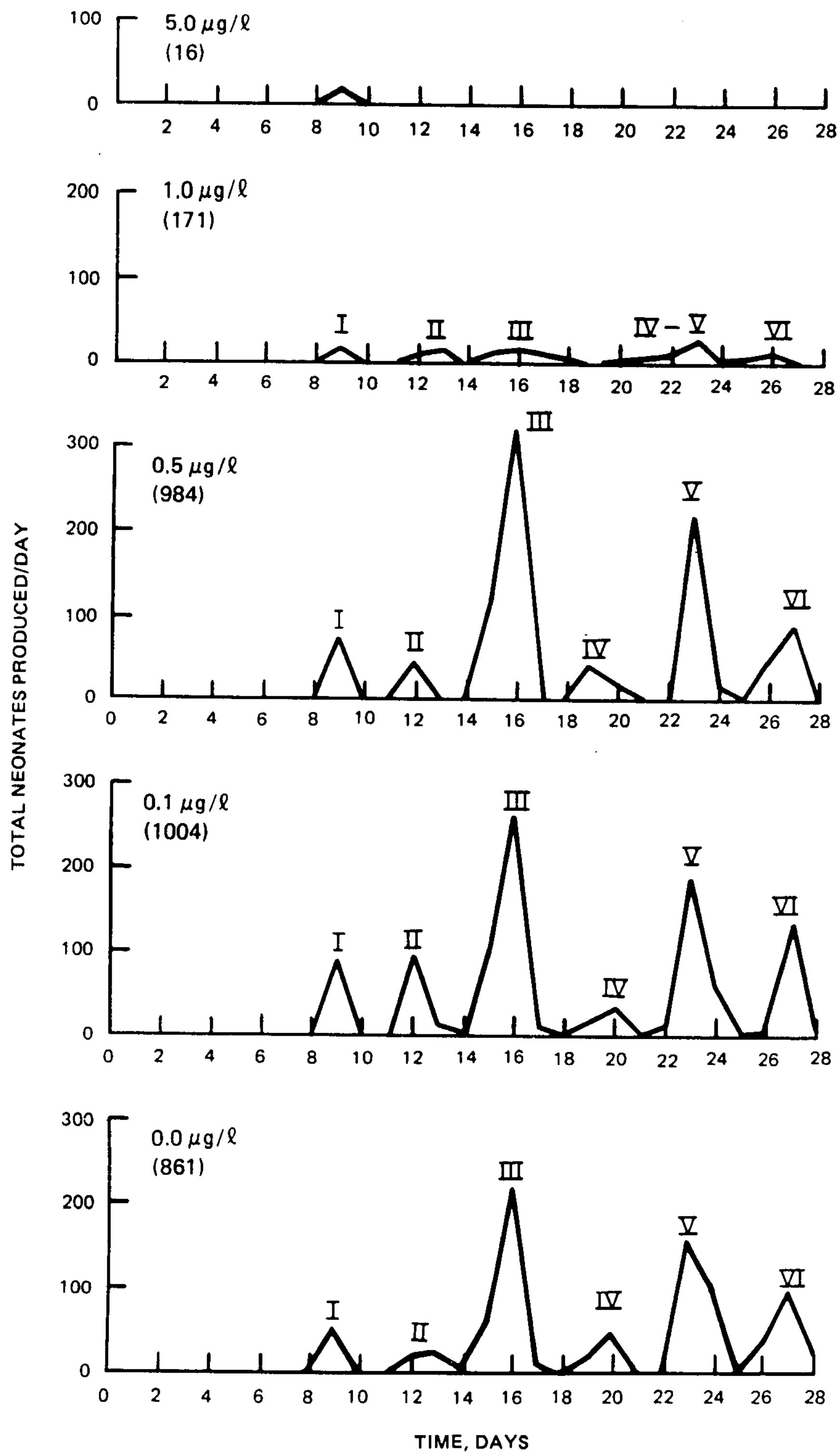

Figure 4. Daily neonate production of *Daphnia magna* during 28-day exposure to cadmium. Values in parentheses represent the total production through 28 days. Roman numerals signify peaks in successive broods

tissue sample appeared to be greater (11.8 μg/g) than found in the other three treatments.

Algal assay

Results of the algal toxicity test showed that there were no significant effects of either mercury or cadmium on the green algae used as the *Daphnia* food source.

Discussion

Mercury

Of the three biological effects examined (survival, growth, and reproduction), only survival was significantly affected in a detrimental manner when *Daphnia* were exposed to mercury for 28 days. Behavioral and morphological observations may help explain the unexpected dose-response pattern in dry weights. Since there was a very thin film of algae covering the bottom, the animals in the 1.0-μg/ℓ treatment were in direct contact with a spatially concentrated food source. The affected *Daphnia* were assumed to be feeding very well as evidenced by bright green digestive tracts and cleared feeding trails behind the *Daphnia* as they propelled themselves along the bottom of the beakers.

Some of the *Daphnia* exposed to 0.5 μg/ℓ were similarly affected, but the frequency of occurrence was greatly reduced compared to those in the 1.0-μg/ℓ treatment. Diminutive antennae were not observed in any *Daphnia* from the other (0.0, 0.05, and 0.1 μg/ℓ) treatments. It is speculated, therefore, that the observed pattern of dry weights was a combined consequence of reduced energetic costs associated with not having to maintain position in the water column coupled with increased caloric intake resulting from feeding on a spatially concentrated food source.

Cadmium

Results reported herein demonstrate a clear dose-response for cadmium-exposed *Daphnia magna*. Data for all the biological effects examined (survival, growth, and reproduction) indicated that *Daphnia* exposed for 28 days to 1.0 and 5.0 μg/ℓ cadmium were significantly affected in a detrimental manner, relative to *Daphnia* exposed to 0.0, 0.1, and 0.5 μg/ℓ. Similar results have been reported for the congener *Daphnia galeata mendotae*, in which growth and reproduction were impaired when exposed for 22 weeks to 4 μg/ℓ cadmium or higher (Marshall 1978). Marshall reported mean tissue concentrations, expressed on a dry-weight basis, of <8.0, 17.6, 28.3, 42.8, and 51.9 μg/g ppm for *Daphnia* chronically exposed to 0, 1, 2, 4, and 8 μg/ℓ cadmium, respectively.

Conclusions

This study provided information about the sensitivity of *Daphnia magna* to mercury and cadmium. *Daphnia* with tissue levels less than 1.9 μg/g mercury or 6.6 μg/g cadmium were not adversely affected. However, *Daphnia* with tissue concentrations greater than or equal to 5.5 μg/g mercury or 11.8 μg/g cadmium exhibited diminished survival, growth, and reproduction.

Results suggest that a 28-day *Daphnia magna* toxicity test might be used in screening for water-column toxicity resulting from open-water disposal of a specific dredged material. The test may be used to predict safe and harmful levels of mercury and cadmium for *Daphnia magna* when survival, growth, and reproduction are measured.

Daphnia magna offers a short-term alternative test species with predictive values for the establishment of chronic-effects data for freshwater invertebrates. The relatively short life cycle of the species and the 28-day duration of the test, the small volume of water used in the tests, and the ease in handling and high fecundity of the organism make *Daphnia* an appealing alternative to the conduct of studies with organisms that require a longer term study that involves much greater volumes of water and complex laboratory equipment.

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Table 1. Survival, Growth, and Reproduction of *Daphnia magna* Exposed to Mercury for 28 Days under Static Renewal Conditions[a]

Nominal Exposure Concentration µg/ℓ	Proportion of Adults Surviving to Day 28	28-day Growth: Adult Length mm	28-day Growth: Adult Dry Wt µg	Reproduction: Time to First Appearance of Eggs in Brood Chamber days	Reproduction: Total Neonates[b] per Female	Reproduction: Total Neonates[b] per Beaker
0.0	0.96 (0.04) n=12	5.24 (0.03) n=12	511 (10.0) n=23	6.0 (0.10) n=24	42 (1.9) n=12	85 (3.8) n=12
0.05	1.00 (0.00) n=12	5.21 (0.04) n=12	540 (13.0) n=24	6.0 (0.06) n=24	43 (1.5) n=12	85 (3.0) n=12
0.1	1.00 (0.00) n=12	5.12 (0.04) n=12	558 (10.7) n=24	5.9 (0.09) n=24	40 (1.1) n=12	[illegible] (2.3) n=12
0.5	0.75** (0.10) n=12	5.21 (0.04) n=12	610* (42.0) n=17	5.9 (0.10) n=24	39 (2.6) n=12	76 (5.3) n=12
1.0	0.17** (0.09) n=12	5.33 (0.19) n=4	795** (122.8) n=4	5.9 (0.06) n=24	35 (9.5) n=12	47** (12) n=12

[a] Entries for each exposure concentration are consecutively mean value, (standard error), and n=number of replicates. Significantly different values are identified as follows:
* Significantly different from control treatment.
** Significantly different from all other treatment means ($p < 0.05$).

[b] Adjusted for daily records of mortality and assumes equal neonate production by each of two *Daphnia* per beaker.

Table 2. Survival, Growth, and Reproduction of *Daphnia magna* Exposed to Cadmium for 28 Days under Static Renewal Conditions[a]

Nominal Exposure Concentration µg/ℓ	Proportion of Adults Surviving to Day 28	28-day Growth		Reproduction		
		Adult Length mm	Adult Dry Wt µg	Time to First Appearance of Eggs in Brood Chamber days[b]	Total Neonates per Female[c]	Total Neonates per Beaker
0.0	1.00 (0.00) n=12	5.26 (0.06) n=12	508 (22.9) n=23	6.0 (0.0) n=23	39 (3.1) n=12	72 (2.4) n=12
0.1	1.00 (0.00) n=12	5.24 (0.05) n=12	514 (17.2) n=24	6.0 (0.0) n=2	42 (1.9) n=12	84* (3.8) n=12
0.5	1.00 (0.00) n=12	5.20 (0.03) n=12	544 (8.7) n=24	6.0 (0.0) n=24	41 (1.4) n=12	82 (2.7) n=12
1.0	0.33** (0.15) n=12	4.94* (0.06) n=8	405* (42.1) n=8	6.4* (0.2) n=22	7.1** (4.0) n=12	14** (7.9) n=12
5.0	0.00** (0.00) n=12	No Data	No Data	6.4* (0.1) n=14	0.7** (0.3) n=12	13** (0.7) n=12

[a] Entries for each exposure concentration are consecutively mean value, (standard error), and n=number of replicates. Significantly different values are identified as follows:
 * Significantly different from control treatment.
 ** Significantly different from all other treatment means ($p < 0.05$).

[b] Includes only those Daphnia for which any egg production was observed.

[c] Adjusted for daily records of mortality; assumes equal neonate production by each of two *Daphnia* per beaker.

Table 3. Mercury and Cadmium in Tissue and Water Samples[a]

Nominal Water Concentrations µg/ℓ	Measured Water Concentrations µg/ℓ	*Daphnia* Tissue Concentrations µg/g
Mercury:		
0.0	0.023 (0.02) n=3	0.2 (1.63) n=3
0.05	0.06 (0.14) n=3	0.69 (0.02) n=3
0.1	0.06 (0.03) n=3	1.92* (0.74) n=3
0.5	0.25 (0.02) n=3	5.47** (0.22) n=2
1.0	0.73 (0.09) n=3	5.47** (0.0) n=1
Cadmium:		
0.0	<0.10 n=3	3.55 (1.63) n=3
0.1	0.10 (0.0) n=3	4.49 (0.59) n=3
0.5	0.50 (0.0) n=3	6.58 (0.91) n=3
1.0	0.97 (0.09) n=3	11.8**[b] n=1
5.0	4.5 (0.53) n=3	No data[b]

[a] Entries for each exposure concentration are consecutively mean value, (standard error), and n=number of replicates. Significantly different values are identified as follows:
*Significantly different from control treatment,
**Significantly different from all other treatment means ($p < 0.05$).

[b] Insufficient tissue for replicate samples due to low percent survival at day 28.

EEDP-01-8
March 1987

Environmental Effects of Dredging Technical Notes

SIMPLIFIED APPROACH FOR EVALUATING BIOAVAILABILITY OF NEUTRAL ORGANIC CHEMICALS IN SEDIMENT

PURPOSE: This note outlines a tiered approach for evaluation of bioavailability of neutral organic contaminants in sediment and provides a method for the numerical expression of bioavailability. The first tier is a simple mathematical calculation, from sediment chemistry, of maximum potential bioaccumulation. If Tier I calculations indicate potential bioaccumulation of neutral organic contaminants to concentrations of concern, then Tier II laboratory tests could be conducted to determine the actual amount of bioaccumulation. In the second tier, bioaccumulation is assessed in laboratory exposures of organisms to the contaminated sediment. Comparison of the actual bioaccumulation at projected steady state to the calculated maximum potential bioaccumulation results in a measure of bioavailability.

BACKGROUND: Public laws regulating dredged material disposal (Section 404 of the Clean Water Act and Section 103 of the Ocean Dumping Act) require ecological evaluation prior to the permitting of operations. Assessment of the potential for bioaccumulation of chemical contaminants in sediment is required as part of the evaluation process. Current methodology (USEPA/CE 1977) involves exposure of aquatic organisms for a period of 10 days to sediment deposited in aquaria. Analysis of tissues of surviving organisms at the end of the exposure period indicates whether detectable bioaccumulation occurred and thus whether the sediment contains specific chemicals of concern in bioavailable forms. The approach is applied empirically on a case-by-case basis and is limited to simple demonstration of uptake. The procedure yields no information concerning concentrations that would actually be accumulated by organisms given prolonged exposure to contaminated sediment, i.e., the projected achievable bioaccumulation. Nor is there any means of using analyses of chemical contaminants in sediment to estimate the concentrations that could theoretically occur in exposed organisms, i.e., the potential for bioaccumulation.

Sediment evaluations that are more effective and informative than the simple 10-day bioaccumulation test can be accomplished using a tiered approach. In the tiered approach, the potential for bioaccumulation is estimated; if estimates of potential bioaccumulation are high enough to be of concern, then the projected achievable bioaccumulation at steady state is determined. This two-tiered method for evaluating organic chemical contaminants in sediment was proposed by McFarland (1984) and McFarland and Clarke (1986).

Tier I evaluation uses results of sediment chemical analysis to estimate theoretical maximum tissue residues that would occur in an exposed organism if all of a chemical of interest in the sediment were bioavailable. Tier II evaluation follows if the maxima calculated in Tier I are judged unacceptable by applicable criteria, levels of concern, or action levels. Tier II involves exposures of aquatic biota to sediment with time-sequenced sampling over a sufficient exposure period (e.g., on days 2, 4, 10, 17, and 30) to allow for projection of steady-state tissue residues using a kinetic model. Tier I calculations represent the maximum bioaccumulation that could occur from a given sediment. Tier II steady-state tissue residues represent the maximum bioaccumulation that is likely to occur in the field (i.e., projected achievable) under exposure conditions similar to those used in the laboratory. Comparison of potential bioaccumulation from Tier I with projected achievable bioaccumulation in Tier II results in a quantitative estimation of bioavailability of chemicals in the sediment under investigation.

This note briefly describes the two-tiered approach and presents an example using laboratory exposures of an aquatic organism to a harbor sediment contaminated with polychlorinated biphenyls (PCBs). More information on the theoretical background and derivation of the approach is found in McFarland (1984) and McFarland and Clarke (1986).

ADDITIONAL INFORMATION OR QUESTIONS: Contact the authors, Mr. Victor McFarland (601) 634-3721 (FTS 542-3721), or Ms. Joan Clarke (601) 634-2954 (FTS 542-2954), or the manager of the Environmental Effects of Dredging Programs, Dr. Robert M. Engler (601) 634-3624 (FTS 542-3624).

Tier I Evaluation

In the first tier evaluation, knowledge of the distribution of a chemical between sediment and organism is required. Neutral organic chemicals such as PCBs are distributed primarily in the lipids of organisms (Könemann and van Leeuwen 1980, Geyer et al. 1982, Mackay 1982) and in the organic carbon fraction of sediment (Karickhoff 1981). Based on the work of Könemann and van Leeuwen (1980) and Karickhoff (1981), neutral organic chemicals were calculated to have a preference factor of 1.72 for organism lipid over sediment organic carbon. This means that the maximum possible chemical concentration that could result in an organism's lipids would be 1.72 times the concentration of that chemical in the sediment organic carbon. This calculated maximum is called the lipid bioaccumulation potential (LBP):

$$LBP = 1.72(C_s/fOC) \tag{1}$$

where

LBP = equivalent concentration in organism lipid in the same units of concentration as C_s

C_s = concentration of chemical in the sediment (any units of concentration may be used)

fOC = decimal fraction organic carbon content of the sediment

LBP represents a maximum possible contaminant concentration in lipid if the sediment is the only source of that contaminant to the organism.

In practice, sediment would be analyzed for the concentration of a neutral organic chemical of concern and for organic carbon content. LBP would be calculated using Equation 1 and would indicate maximum bioaccumulation potential in the lipid of any organism. It is generally desirable to convert LBP to a whole-body bioaccumulation potential (WBP) for a particular organism of interest. This is done by multiplying LBP by that organism's lipid content (expressed as a decimal fraction of wet weight), as determined by lipid analysis or from reported data:

$$WBP = LBP(fL) \qquad (2)$$

where

WBP = maximum whole-body bioaccumulation potential in the same units of concentration as LBP

fL = decimal fraction of an organism's lipid content

If the calculated WBP is acceptable by whatever criteria are applied (e.g., the Food and Drug Administration (FDA) limit of 2 parts per million (ppm) PCB in the edible portions of fish and shellfish), then the sediment evaluation need go no further. If the calculated level is not acceptable (e.g., greater than 2 ppm PCB), then further evaluation could involve biological testing in Tier II.

Caveats: Two important assumptions are implicit in these calculations: (1) no metabolic degradation or biotransformation of the chemical and (2) total bioavailability of sediment-associated chemical to the organism. Estimations involving WBP, then, are inherently conservative in that they will present a worst-case prediction of bioaccumulation if sediment is the only source of the contaminant to the organism.

Tier II Evaluation

In the second tier evaluation, aquatic organisms are exposed to contaminated sediment under constant laboratory conditions for a sufficient period of time for bioaccumulation to occur. If exposure were continued under constant conditions, then a steady state would eventually be achieved in which maximum bioaccumulation would have occurred and the net exchange of the contaminant between sediment and organism would be zero. In practice it is not likely that steady state will be reached in any period of time short enough for economical laboratory testing. By taking samples sequentially over a short period of constant exposure, a simple kinetic model can be used to project tissue concentrations at steady state (Blau et al. 1975). A computational form of this model integrated for constant exposure is:

$$C_T = \frac{k_1 C_W}{k_2}\left(1 - e^{-k_2 t}\right) \tag{3}$$

where

C_T = concentration of chemical in organism
k_1 = uptake rate constant
C_W = concentration of chemical in exposure medium
k_2 = elimination rate constant
t = time

This model can be fitted to time-sequenced exposure data using an iterative nonlinear regression method, such as those in the SAS NLIN procedure (SAS 1985).

As duration of exposure increases, the term $e^{-k_2 t}$ approaches zero, and

$$C_T = \frac{k_1 C_W}{k_2} = C_{ss} \tag{4}$$

in which C_{ss} is the whole-body concentration of chemical at steady state.

If steady state is not achieved for a contaminant of interest during the laboratory exposure, then C_{ss} can be projected using the time-sequenced exposure data in a nonlinear regression procedure, as described above.

The projected achievable C_{ss} in an organism can then be compared with the potential maximum bioaccumulation WBP estimated from sediment chemistry in Tier I and is expressed as the proportion p of WBP projected at steady state:

$$p = \frac{C_{ss}}{WBP} \tag{5}$$

If all of the chemical of concern in the sediment to which an organism is exposed were bioavailable, then p would equal 1. Any value of $p < 1$ indicates less-than-complete bioavailability of the chemical of concern in a sediment under investigation. The magnitude of p is a numerical expression of bioavailability that could be of assistance in decisionmaking: for example, in evaluating several disposal alternatives.

Example Using PCB-Contaminated Sediment

Figure 1 presents a flow chart illustrating the steps in the Tier I and Tier II evaluations of contaminated dredged material. Example data from tests of a PCB-contaminated harbor sediment (shown in the following tabulation) demonstrate the calculation of maximum potential bioaccumulation (WBP) and the bioavailability expressed as the proportion p of WBP actually achieved.

In this example, freshwater mussels were exposed to sediment having four levels of PCB contamination (high, medium, low, and reference) for 30 days at 20° C in a flow-through aquarium system under constant exposure conditions. Tissue samples were taken for chemical residue analysis on days 2, 4, 10, 17,

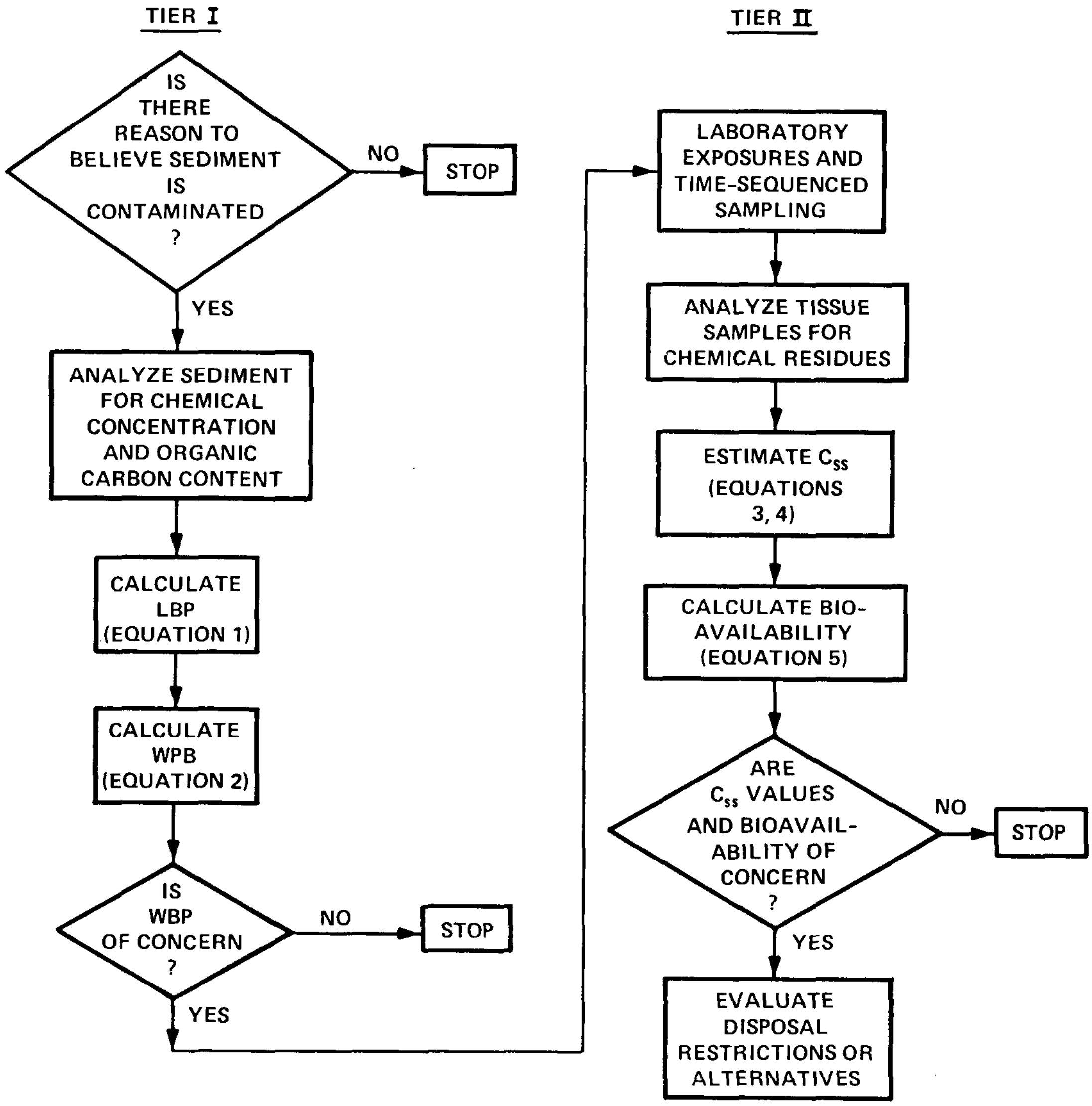

Figure 1. Flow chart for Tier I and Tier II evaluations

a. Tier I evaluation.

Level of Contamination	Sediment: Total PCB C_s, ppm	Sediment: Organic Carbon fOC	LBP, ppm (Eq. 1)	Mussels Lipid Fraction, fL	WBP, ppm (Eq. 2)
High	44	0.016	4730	0.0029	13.7
Medium	33	0.016	3548	0.0031	11.0
Low	4.0	0.016	430	0.0025	1.07
Reference	0.45	0.015	51.6	0.0026	0.13

b. Tier II evaluation.

Level of Sediment Contamination	Total PCB C_{ss}, ppm (Eq. 3, 4)	Bioavailability p (Eq. 5)
High	1.1	0.0802
Medium	0.87	0.0791
Low	0.83	0.7721
Reference	0.054	0.4025

and 30; and the residue data were used to calculate C_{ss} for total PCB. Details of the experimental design and analysis are described in McFarland and Clarke (1986).

Total PCB concentrations in the sediment ranged from <1 ppm in the reference sediment to 44 ppm in the highly contaminated sediment. LBP values calculated from these concentrations ranged from about 50 to over 4000 ppm. These values represent maximum total PCB concentrations that could occur in the lipids of any aquatic organism exposed to the sediment as the only source of contamination and where that source was totally biologically available. Converting LBP values to a whole-body basis for mussels having a lipid fraction of approximately 0.003 (i.e., 0.3 percent), yielded WBP values that ranged from 0.13 ppm maximum possible bioaccumulation for mussels exposed to the reference sediment to over 13 ppm for mussels exposed to the highly contaminated sediment.

The final step of a Tier I evaluation is a regulatory decision concerning the potential for adverse environmental impact of the sediment analyzed. The regulator might decide, for example, that any sediment having a potential (WBP) for total PCB bioaccumulation greater than the 2-ppm FDA action level would require further evaluation for actual bioaccumulation. Based on the example data for this freshwater sediment, further evaluation (Tier II) would be indicated for the sediment with high and medium levels of contamination, but not for the low contamination or the reference sediment. Tier II calculations using all four sediments are presented in this note for the sake of illustration.

Tier II projected steady-state tissue concentrations C_{ss} of total PCB ranged from 0.054 ppm for mussels exposed to the reference sediment to 1.1 ppm for mussels exposed to the highly contaminated sediment. These values are clearly much lower than the calculated potential maximum tissue residues (WBP). The regulator might now decide that the PCB content of the sediment under evaluation did not pose a threat to a mussels fishery located near the proposed disposal site under conditions similar to the experiment, since the projected actual PCB bioaccumulation is less than the FDA action level of 2 ppm for edible portions of fish and shellfish.

However, the potential for PCB bioaccumulation in other organisms of greater lipid content exposed to the same sediment might exceed the FDA action level. Using the nomograph shown in Figure 2, it is possible to quickly estimate WBP for organisms of various lipid contents, providing the approximate contaminant concentration C_s and organic carbon content fOC of the sediment are known. The procedure for using the nomograph is as follows.

<u>STEP 1.</u> Determine the lipid content of an organism of interest, either from previously reported values or from laboratory analysis, and express the lipid content as percent of whole-body wet weight, rather than as decimal fraction.

<u>STEP 2.</u> Locate the value on the right-hand vertical axis that corresponds most closely to that lipid content.

<u>STEP 3.</u> Follow the sloped line until it intersects the sediment concentration C_s . C_s may be expressed in any units of concentration and may be selected from any of the four ranges: 0.1-1.0; 1-10; 10-100; or 100-1000.

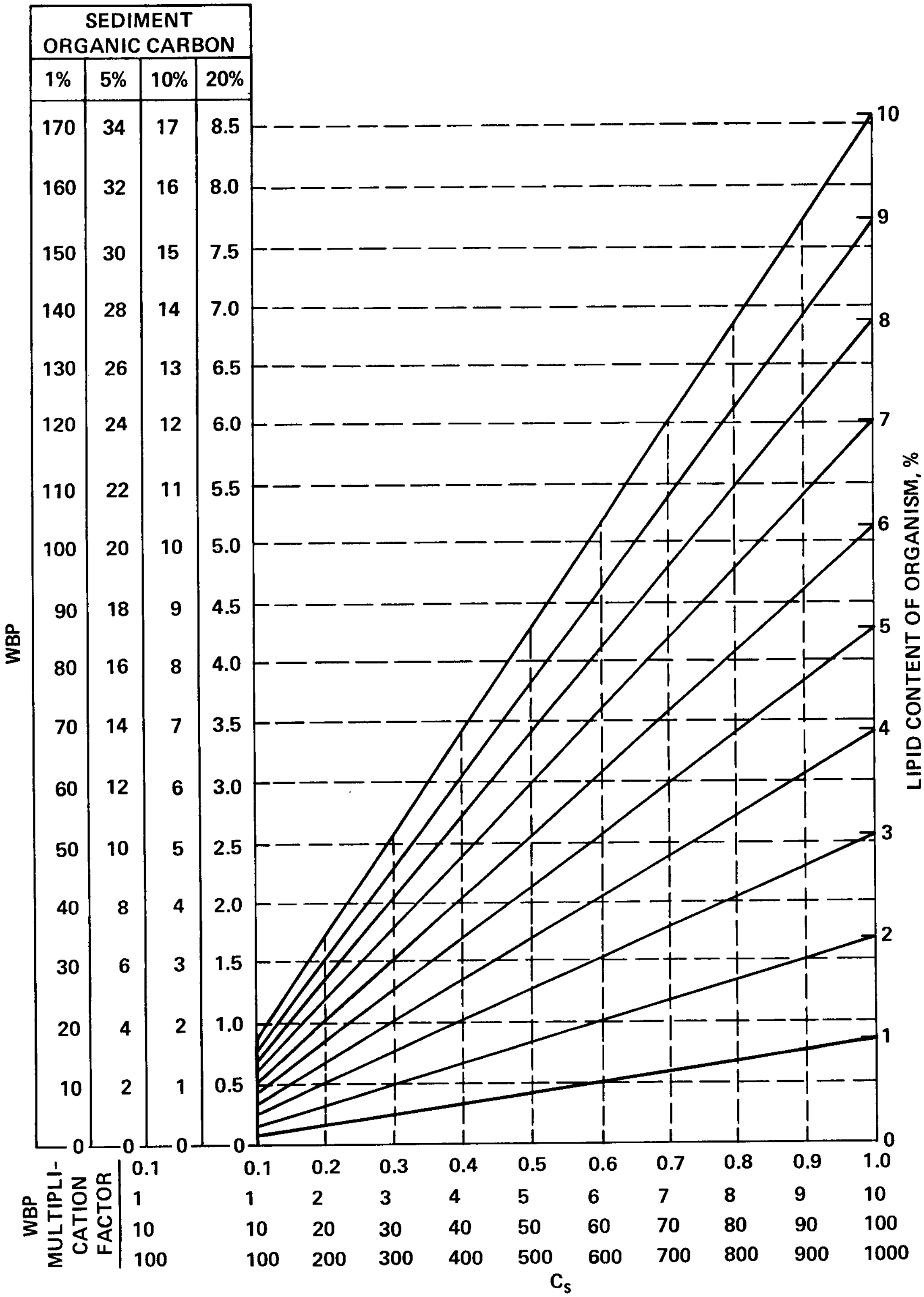

Figure 2. Nomograph for determining bioaccumulation potential (WBP)

STEP 4. From that point, read across to the left-hand vertical axis and select the WBP value from the appropriate sediment organic carbon column expressed as percent of sediment dry weight.

STEP 5. Multiply WBP by the factor (0.1, 1, 10, or 100) corresponding to the selected C_s range. WBP will then be in the same units of concentration as C_s .

The lipid scale as well as the C_s scale of the nomograph can be changed by orders of magnitude by adjusting the WBP scale in the same manner. For example, if the organism of interest is a mussel having 0.3 percent lipid content, one would simply follow the 3-percent lipid line and divide the appropriate resulting WBP value by 10. If the sediment concentration C_s of a contaminant falls above or below the C_s ranges shown on the nomograph, then the units of concentration can be changed (e.g., change 0.02 ppm to 20 parts per billion). Interpolation between lipid lines or between organic carbon columns is straightforward because all relationships are proportional. For example, for sediment organic carbon content of 3 percent (fOC = 0.03), WBP would be 1/3 the WBP value at 1 percent organic carbon, 5/3 the WBP value at 5 percent organic carbon, 10/3 the WBP value at 10 percent organic carbon, or 20/3 the WBP value at 20 percent organic carbon.

To illustrate the use of the nomograph, the regulator may be interested in assessing the potential for bioaccumulation of total PCB by a fish of 5 percent lipid content exposed to the highly contaminated sediment (44 ppm PCB). The regulator would trace the 5-percent lipid line to a C_s value of 44 and then read across to the 1-percent organic carbon column to obtain a WBP value of about 38×10 or 380 ppm. Since the organic carbon content of the sediment is 1.6 percent, a more precise estimate can be made by dividing 380 by 1.6 to obtain a maximum whole-body bioaccumulation potential of 238 ppm. Such a high WBP value might prompt the regulator to impose disposal prohibitions or restrictions without further sediment evaluation. Alternatively, the regulator might decide to conduct Tier II evaluations to project actual PCB bioaccumulation from the highly contaminated sediment by that fish species, as well as to evaluate bioavailability under various disposal options.

The final aspect of the Tier II evaluation in this example, then, is the consideration of bioavailability. Proportion p of projected bioaccumulation C_{ss} to bioaccumulation potential WBP for mussels ranged from <0.1 for high and medium contamination to 0.4 for the reference sediment and 0.8 for low contamination. Since p is so low for the more highly contaminated sediments, environmental factors that could enhance bioavailability should be considered. Suspension of contaminated sediment in the water column during dredging and disposal operations, for example, would increase the surface area for desorption and could at least transiently increase concentrations of desorbing chemicals available to fish and filter-feeding animals. This is particularly true in freshwater systems. On the other hand, freshwater bivalves often close up when turbidity increases, thus limiting their exposure to contaminants desorbing from suspended particulates.

Future Research

Research is being conducted at the WES to define the roles of suspended contaminated sediment, soluble and microparticulate organic carbon, organism life-history strategies, and other environmental variables in determining the bioavailability to aquatic biota of chemicals associated with sediment that must be dredged. From these findings, methods for evaluating the ecological impact of dredging and disposal operations are being developed that will have improved utility and interpretability compared to present methods.

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EEDP-01-9
February 1988

Environmental Effects of Dredging Technical Notes

A PROCEDURE FOR DETERMINING CAP THICKNESS FOR CAPPING SUBAQUEOUS DREDGED MATERIAL DEPOSITS

PURPOSE: This note presents preliminary information on using a procedure to ascertain the thickness of a cap of natural material necessary to isolate a contaminated sediment under aquatic disposal conditions.

BACKGROUND: When testing required under Public Law 92-532 (the Ocean Dumping Act) demonstrates that aquatic disposal of dredged material may cause unreasonable degradation of the marine environment, ocean disposal of that material may be prohibited. Capping of the contaminated material by material suitable for ocean disposal has been accepted by the Convention on the Prevention of Marine Pollution by Dumping of Waste and Other Matter (London Dumping Convention) as an alternative to other disposal methods (such as confined land disposal). For this option to be operational (rather than being restricted to experimental situations), it must be demonstrated that capping isolates the contaminated material under a wide range of conditions.

A prime concern about capping as an acceptable disposal method is its efficiency in isolating contaminated dredged material from the water column and from both pelagic and benthic biota. Much work has addressed this concern (Brannon et al. 1985, 1986; Gunnison et al. 1986, 1987; Palermo et al., in preparation). In these studies, the effectiveness of capping in chemically and biologically isolating a contaminated sediment from the overlying water column was studied using a two-step process that involved small- and large-scale experimental units.

The small-scale laboratory tests were used to experimentally assess the cap thickness needed to chemically isolate a contaminated dredged material by following changes of dissolved oxygen, ammonium-nitrogen, and orthophosphate-phosphorus in the overlying water column. The large-scale laboratory tests were used to:

- Determine the effect of cap thickness in preventing movement of contaminants into the biota.
- Determine the effect of bioturbation on the effectiveness of capping.
- Validate results that were obtained in the small-scale test.

Based on the results of these studies, a research procedure has been modified into a laboratory test suitable for field use.

The effective cap thickness for a biological and chemical seal provides the isolation necessary to control the movement of contaminants out of the contaminated dredged material into the overlying water column and to prevent direct contact (through bioturbation) between aquatic biota and contaminants. This estimated thickness does not allow for hydrodynamic forces that may result in scouring and resuspension of cap material and, possibly, the material beneath the cap. Procedures to predict and offset the effects of hydrodynamic processes require engineering considerations. In addition, since capping is still considered an experimental procedure under some water depth and hydrodynamic conditions, the site should be monitored once the cap has been emplaced. For a discussion of such capping-related concerns, see Environmental Laboratory (1987), Truitt (1987a,b), and Palermo et al. (in preparation).

ADDITIONAL INFORMATION AND QUESTIONS: For additional information on the procedure described in this article, contact the authors, Mr. Thomas Sturgis, commercial and FTS (601)634-2805, and Dr. Douglas Gunnison, (601)634-3873, or Dr. Robert M. Engler, Manager of the Environmental Effects of Dredging Programs, (601)634-3624.

Small-Scale Laboratory Test for Field Application

To allow Corps Districts to estimate the cap thickness that will chemically isolate a contaminated sediment from the overlying water column, a laboratory test is needed that is accurate and easily used. Such a test has been developed based on the work of Brannon et al. (1985, 1986), Gunnison et al. (1986), and Palermo et al. (in preparation).

Dissolved oxygen (DO) depletion, ammonium-nitrogen, and orthophosphate-phosphorus are used as tracers because they are easy and inexpensive to measure. A cap thickness that is effective in preventing the movement of these inorganic constituents will also be effective in preventing the movement of organic contaminants that are more strongly bound to sediment (e.g., polynuclear aromatic hydrocarbons (PAHs), petroleum hydrocarbons, and polychlorinated biphenyls (PCBs)). The behavior of soluble reduced inorganic species (e.g., arsenic) will also be similar to the tracers.

Dissolved oxygen depletion in the water column is normally not a problem in an open-water disposal environment, due to mixing and reaeration of the water column. However, DO depletion can be used as a tracer for determining the effectiveness of a cap in isolating an underlying contaminated dredged material having an oxygen demand exceeding that of the capping material. A cap thickness that is effective in preventing or reducing the diffusion of DO into the contaminated sediment will also prevent or reduce the diffusion of DO-demanding species from the contaminated sediment into the overlying water column. Once an effective cap thickness has been achieved, there will be no significant difference in oxygen depletion rates between the contaminated sediment with cap material and the cap material alone.

A similar rationale is applicable for using ammonium-nitrogen and orthophosphate-phosphorus as tracers. These constituents are released only under anaerobic conditions. However, if the layer of cap material is thick enough to prevent the diffusing materials in the underlying contaminated dredged material from reaching the water column, the release rates from the capped contaminated sediment will be the same as from the cap material alone.

Chemical tracers

More than one tracer (ammonium-nitrogen, orthophosphate-phosphorus, and DO depletion) should be considered for each application (Brannon et al. 1985, 1986; Gunnison et al. 1986; Palermo et al., in preparation). In a laboratory study conducted with dredged material from Everett Harbor, Washington, the DO depletion rate of the cap material was not significantly different from that of the contaminated sediment (Palermo et al., in preparation). This precluded the use of DO depletion as a tracer in evaluating cap effectiveness. In studies using sediments from Dutch Kills, New York, and Black Rock Harbor, Connecticut, orthophosphate-phosphorus was unsuitable as a tracer, while DO depletion and ammonium-nitrogen were suitable (Brannon et al. 1985, 1986; Gunnison et al. 1986).

Another reason for using more than one tracer is the variation of chemical and biochemical properties in sediments. Frequently, the contaminated sediment and the proposed capping material will be so different that a chemical property of the contaminated sediment will be easily distinguishable from that same property of the cap material. However, when the cap material has chemical properties similar to the contaminated sediment, chemical differences are harder to distinguish. In such a case, if only one tracer is measured and negative results are obtained, a second series of tests is necessary.

Water analysis

The release rates of ammonium-nitrogen and orthophosphate-phosphorus should be determined in accordance with procedures recommended by Ballinger (1979).

The depletion rate of DO should be determined using either the azide modification of the Winkler method, as described in Standard Methods (APHA 1986), or a DO meter.

Sediment collection

Samples of contaminated sediment should be collected that are representative of sediment to be dredged. Samples of the proposed capping material should also be taken. To ensure that sediment samples are not diluted with large volumes of water, a clamshell dredge or similar device should be used to sample both contaminated sediment and capping material. Representative subsamples of both materials should be taken for initial bulk analysis and characterization. All sediments should be placed into polyethylene-lined steel barrels, sealed, and stored at 4° C until tested.

Sediment preparation

Sediment samples should be composited and mixed, using a motorized mixer (to ensure a homogenous sediment sample). Any unused sediment may be returned to the containers, stored at 4° C, and later discarded if there is no further need for the sediment.

Handling of highly contaminated sediments

The following procedure, which outlines safety equipment, sediment handling, cleanup operations, and disposal, is used by the Environmental Laboratory for handling highly contaminated sediment. This procedure is not intended to replace any existing procedures; however, it can serve as a guide and supplement the existing safety procedures.

All individuals involved in handling contaminated sediment are required to use protective equipment and to submit to blood and urine tests. The protective equipment consists of:

- A full-face chemical cartridge respirator (with an organic chemical cartridge and dust filter).
- A pressure-demand airline respirator, when handling sediment with PCB concentrations $\geq$2,500 ppm.
- A polyethylene- or saran-coated tyvek disposable coverall.
- Inner PVC laboratory gloves with outer neoprene gloves.
- Neoprene rubber boots.
- Surgical scrubs.

Blood and urine sampling is intended as a monitoring procedure to ensure the safety of the individual handling the sediment. It is recommended that background blood and urine screening be performed for those contaminants of concern before project testing begins and upon completion of the project. In cases of exposure to highly contaminated sediment over a long period (6 months or longer), blood and urine sampling should be done every 3 months.

Contaminated sediment must be handled in a well-ventilated building in order to control the concentration of particles in the air. For example, PCBs will adsorb strongly to any surface, and a small amount of contaminated sediment solids in the air can have very high concentrations of PCBs adsorbed on them, making inhalation of this dust very dangerous. Also, polyethylene sheeting should be placed under all test and mixing apparatus as a contamination preventive measure. This polyethylene sheeting will prevent needless contact with the laboratory surface and make cleanup easier.

Cleanup is an essential part of a safe laboratory environment. The procedure is as follows:

- Contaminated sediment should be removed from all equipment using machine wipes. Used wipes are considered hazardous and should be disposed of in the same manner as coveralls (see below).
- All equipment is rinsed in the laboratory sink after cleaning. The sink is then thoroughly cleaned.
- The polyethylene sheeting is disposed of in a disposal drum.
- Lids are fastened securely on the drums.
- Coveralls (used as protective clothing) and surgical scrubs (worn underneath the coverall rather than personal clothing) are removed and placed in a disposal drum.
- The disposal drum is labeled and disposed of according to US Department of Transportation guidelines (1984).

Materials

The following items are required to conduct the laboratory test:

- Twelve 22.6-ℓ cylindrical plexiglass units, 120 cm in height and 15.5 cm in diameter attached to a 30-cm, 2-plexiglass base (see Figure 1). The units should be fitted with a sampling port.
- Twelve plexiglass plungers, 80 cm in length with a wire hook attached at the top.
- Twelve pint-size bottles of mineral oil.
- Six aquarium pumps (two small-scale units per pump) or some other source of air supply.
- Twelve 1-cm-long airstones.
- Two plexiglass tubes, 130 cm in length, 7.28-cm inside diameter.
- Two large funnels, 40.8-cm top diameter, 6.60-cm outside diameter at the base.
- Tygon tubing, 3.02-mm inside diameter.

Test procedure

Step 1 - Adding contaminated sediment to the units. The contaminated sediment should be mixed, then placed in the bottom of nine small-scale units to a depth of 10 cm (Figure 1). It is important to add the sediment carefully to avoid splashing on the sides of the units.

Step 2 - Adding capping material. The capping material is mixed and then added in thicknesses of 22 and 35 cm in triplicate to six of the units

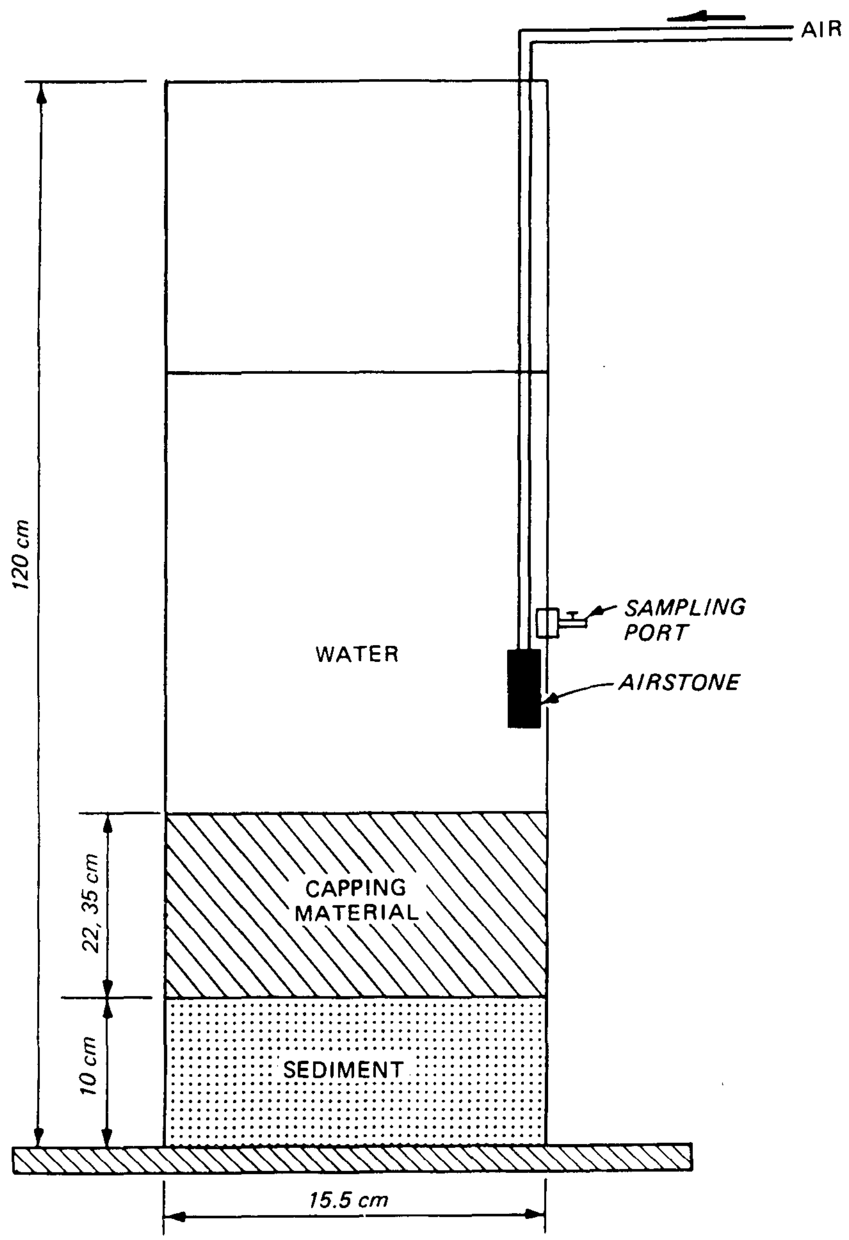

Figure 1. Small-scale experimental unit

containing the contaminated sediment (Figure 1). The remaining three units with contaminated sediment receive no cap. An additional three units receive 10 cm each of capping material only. Units containing contaminated sediment alone and units with capping material alone serve as controls. The 22- and 35-cm cap thicknesses were selected based on results of studies conducted by Brannon et al. (1985, 1986) and Gunnison et al. (1986). The experimental setup for the small-scale laboratory test is shown in the following table:

Small-Scale Units	Sediment
1-3	Contaminated sediment only
4-6	Cap material only
7-9	Contaminated sediment + 22-cm cap
10-12	Contaminated sediment + 35-cm cap

Step 3 - Water addition and unit aeration. For an estuarine or marine simulation, 10 ℓ of artificial seawater is prepared using artificial sea salts to achieve the salinity of the proposed disposal area. For a freshwater simulation, 10 ℓ of either distilled or reverse osmosis water is used. The water is added as gently as possible to each small-scale unit and allowed to equilibrate for 3 days while being aerated. Aeration will ensure that the DO concentration in all units is at or near saturation (within ±0.5 mg/ℓ) at the start of the test.

After 3 days of aeration, the airstone is removed, and a plunger and mineral oil are added. The plunger is used for daily mixing to prevent the establishment of concentration gradients in the water column and to ensure a well-mixed column. Mineral oil is used to seal the surface of the water column from the atmosphere to allow the development of anaerobic conditions in the water column. The plunger is suspended between the sediment and the mineral oil. Mixing should be done in a manner that will not disturb the sediment in the bottom of the unit or breach the mineral oil on the surface of the water. After mixing, the plunger is left suspended in the water column.

Step 4 - DO measurements. Water samples should be taken immediately after aeration for initial DO determination. Dissolved oxygen should then be measured daily until the DO is depleted in the water column of the uncapped contaminated sediment. The consequences of reducing the volume of the water column by taking DO samples is accounted for by multiplying the DO concentration (milligrams per liter) by the volume of water remaining in the unit after a given sampling. (See the Calculations section that follows.)

Step 5 - Water sampling and preservation. Water samples to be analyzed for ammonium-nitrogen and orthophosphate-phosphorus should be taken immediately after the DO is depleted (day 0) and subsequently on days 15 and 30. These water samples should be cleared of particulate matter by passing through a 0.45- m membrane filter, preserved by acidification with concentrated hydrochloric acid (HCl) to pH 2, then stored at 4° C. After the water column is sampled on day 30, all water samples (days 0, 15, and 30) should be analyzed. Results from previous small-scale studies (Brannon et al. 1985, 1986; Gunnison et al. 1986; Palermo et al., in preparation), have shown that complete anaerobic conditions are achieved in the water column within 30 days.

Data interpretation and analyses

The results from these laboratory tests will indicate which of the thicknesses (22 or 35 cm) will reduce overlying-water oxygen demand and transfer of ammonium-nitrogen and orthophosphate-phosphorus from the contaminated sediment to the level of the cap material alone.

Oxygen-depletion rates and ammonium-nitrogen and orthophosphate-phosphorus release rates should be determined by performing linear regression analyses of mass uptake or release per unit area (milligrams per square meter) versus time. Means and standard deviations should be determined for the triplicates, and t-tests should be conducted to determine the statistical significance of differences between the means. Rates plotted are the means and standard deviation of three replicates and represent values greater than the controls.

Calculations

The rates in this test are defined as milligrams per square meter per day. This may be determined by:

$$Tt = Pd \times Vr$$

then

$$Ra = Tt/Au/day$$

where

Tt = tracer total concentration (mg) in the unit

Pd = tracer dissolved concentration (mg/ml) as determined by chemical analysis

Vr = volume of water (ml) remaining in the water column after a given sampling

Ra = rate of release or mass uptake ($mg/m^2/day$)

Au = area (m^2) of the unit

day = number of days of study

The recommended thickness (22 or 35 cm) can then be evaluated by comparing the release rates (Ra) of tracers through the thicknesses tested to the release rates of tracers from the capping material alone. For a given thickness to be considered effective, its release rates must equal those from the capping material alone, or there should be no statistically significant difference.

Figure 2 is an example graph showing oxygen depletion rates of the Black Rock Harbor sediment capped with sand plotted against cap thickness (centimeters). It is important to note that a series of cap thicknesses ranging from 2 to 26 cm were evaluated. The data points on the graph are means and standard deviations of three replicates. Results show that a 22-cm cap of

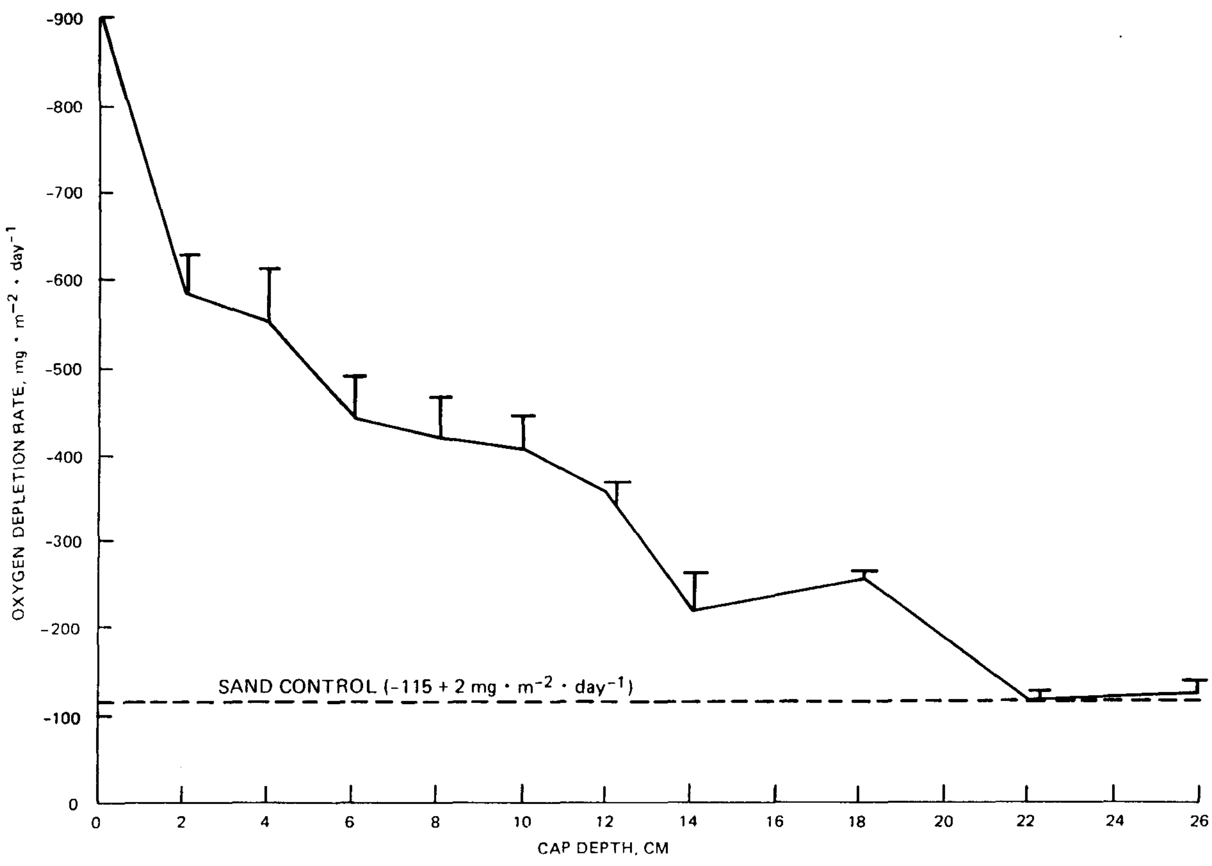

Figure 2. Effect of sand cap depth on overlying water oxygen demand

sand resulted in inhibition of oxygen demand equal to that of the sand cap itself, thus indicating a seal effective in isolating the overlying water column from oxygen demand due to Black Rock Harbor sediment. In this case, the recommended thickness for reducing oxygen demand on the overlying water by the contaminated sediment would be 22 cm.

The test described here will evaluate only the 22- and 35-cm thicknesses of caps. An alternative to using two capping thicknesses is to conduct a series of tests using capping thicknesses ranging from 2 to x cm. Through this approach, the effective cap thickness to chemically isolate the contaminated sediment can be determined.

The thickness predicted by this test is for a chemical seal only and does not include allowances for bioturbation.

<u>Bioturbation</u>

The importance of bioturbation by burrowing aquatic organisms to the mobility of contaminants cannot be overestimated. In addition to the disruption (breaching) of a thin cap that can result when organisms actively work the surface sediments, there is the problem of the direct exposure of the burrowing organisms to the underlying contaminated sediment.

The thickness needed to prevent breaching of cap integrity through bioturbation can be determined indirectly from other information sources. For example, the benthic biota of US coastal and freshwater areas have been fairly well examined, and estimates of the depth to which benthic animals burrow should be available from regional authorities.

<u>Estimating required cap thickness</u>

The thickness required to obtain a complete chemical and biological seal (TR) is provided by the equation:

$$TR = TP + DB$$

where

TP = predicted thickness (cm) to obtain a chemical seal

DB = depth (cm) to which the deepest burrowing organism in the region can reach (obtained by consultation with authorities or bioturbation in the region)

A cap thickness is needed that will maintain its efficacy under the long-term effects of hydrodynamic forces. The hydrodynamic forces may result in erosion and transport of the cap material, thus reducing the efficacy of the cap. If hydrodynamic forces are severe enough, other precautions, such as armoring cap surface, may need to be taken. For additional information on engineering considerations to offset hydrodynamic forces, see Truitt (1987a,b).

References describing the application of both the small- and large-scale tests to several Corps projects are available in Brannon et al. (1985, 1986), Gunnison et al. (1986), Environmental Laboratory (1987), and Palermo et al. (in preparation). A detailed description of the development of the small-scale test is given in Gunnison et al. (1987).

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Environmental Effects of Dredging Technical Notes

The Environmental Effects of Dredging Technical Notes have been published since June 1985, and the reaction from the field offices has been very encouraging. The responses have noted quality and timeliness of subject matter, and ease of keeping up with new and innovative ideas in dredging-related areas.

Although the primary distribution is to the field offices of the Corps of Engineers for use by personnel involved with all aspects of dredging and disposal projects, these Technical Notes are not just "WES to Field." They are intended to include "Field to WES" and "Field to Field." Field input is highly desirable to disseminate to other offices new techniques or a unique application developed by Corps field offices. WES will collect and publish appropriate material and fully credit the source. Every Corps professional involved in dredging projects in the Corps of Engineers is a partner in the Technical Notes and is encouraged to contribute.

The information presented in the Technical Notes is based on state-of-the-science procedures and state-of-the-practice field demonstrations. However, these are considered interim in nature. Consequently, they may not be final procedures or approaches in all cases. Engineer Manuals and other implementation manuals will provide the more formal guidance.

Suggestions on subject material and input from the Corps field for Technical Notes are invited and should be addressed to Commander and Director, US Army Engineer Waterways Experiment Station, ATTN: CEWES-EP-D, PO Box 631, Vicksburg, MS 39180-0631.

Subject material can be in any of the following areas:

1. Aquatic Disposal
2. Upland Disposal
3. Wetland/Estuarine Disposal
4. Regulatory (Testing and Interpretation)
5. Design, Construction and Operations
6. Management
7. Beneficial Uses
8. Miscellaneous
9. Equipment

EEDP-01-10
February 1988

Environmental Effects of Dredging Technical Notes

ACOUSTIC TOOLS AND TECHNIQUES FOR PHYSICAL MONITORING OF AQUATIC DREDGED MATERIAL DISPOSAL SITES

PURPOSE: This article provides interim guidance on the use of acoustic tools and techniques for physical monitoring of aquatic (open-water) dredged material disposal sites. The information presented is taken from the "Guidelines for Biological and Physical Monitoring of Aquatic Dredged Material Disposal Sites" (Fredette et al., in preparation).

BACKGROUND: Increased coastal and marine dredging, limited upland disposal sites, and a drive to reduce dredging costs combine to increase the need for open-water disposal of dredged material relatively close to shore. Effective monitoring of disposal activities is necessary to prevent adverse physical and biological impacts resulting from such disposal operations. Lack of guidance on monitoring was identified as a problem at the Long-Term Management Strategy Workshop in August 1985 (US Army Engineer Waterways Experiment Station, in preparation), leading to a Dredging Operations Technical Support (DOTS) management task to provide needed guidelines to the Corps field offices.

The focus of the guidelines and of this article is on dredged material that has complied with the guidelines of the Clean Water Act (Section 404) and the Ocean Dumping Act (Section 103), i.e. material that is acceptable for open-water disposal. Consequently, chemical concerns associated with contaminated sediments are not addressed.

These guidelines were developed under the DOTS Program, and the tools and techniques recommended are being further evaluated under DOTS and through cooperative studies with Corps of Engineer District Offices.

A series of articles on monitoring aquatic dredged material disposal sites is planned for the Environmental Effects of Dredging Technical Note Series. Future topics include biological monitoring, a sediment profiling camera, sampling tools, measurement of engineering properties of disposed sediments, dredged material consolidation, and other topics as information becomes available.

ADDITIONAL INFORMATION: Contact one of the authors, Mr. James E. Clausner, (601)634-2009; or Mr. Edward B. Hands, (601)634-2088; or the Environmental Effects of Dredging Programs (EEDP) Manager, Dr. Robert M. Engler, (601)634-3624.

Introduction

Monitoring programs should provide information the site manager needs to make decisions concerning continued disposal operations. Potentially adverse physical and biological impacts resulting from disposal should be defined before initiating a monitoring program. This will allow the design of a monitoring program to address those factors that will provide the site manager with information needed to modify the disposal operation prior to creating any substantial adverse effect. The size and cost of the monitoring program should be based on the size and cost of the project. These objectives can be met with a tiered monitoring program based on predetermined trigger levels. Exceeding a predetermined monitoring trigger level provides the manager with an early warning and calls the next higher (more detailed) tier of monitoring.

In some cases, concern is limited to physical impacts such as increased shoaling that may create a navigation hazard. In cases where biological resources are of concern, the impact may stem from physical processes such as burial or change in grain size of the substrate. Because physical impacts drive the biological changes, and because physical impacts are more easily measured, the first tier of any monitoring program should include basic physical measurements.

A physical monitoring strategy combines remote techniques covering broad areas (bathymetry, side-scan sonar, subbottom profiles) with direct measurements (cores, grab samples, sediment profiling camera) at chosen locations to verify the information provided by the broad-area techniques. Direct and indirect measurements of sediment transport are also used, including current meters, sediment traps, reference rods, and near-bottom current drogues (seabed drifters). Other physical monitoring methods include remote sensing applications and measurement of engineering properties of disposal sediments. Navigation and positioning systems must be chosen with care to ensure that the monitoring data collected have sufficient accuracy.

This technical note discusses the broad-area acoustic tools and techniques of bathymetry, side-scan sonar, and subbottom profilers. These tools complement each other in monitoring activities. Bathymetry provides topographic measurements of the disposal area, side-scan sonar gives qualitative surface topography and distinguishes between sediment types, and subbottom profilers show subsurface layers.

Acoustic Monitoring Tools and Techniques

Bathymetry

Probably the most fundamental measurement of a disposal site is bathymetry. For most applications, bathymetric surveys are the primary tool for determining where the material has been placed and how much remains on-site. Bathymetric surveys usually require microwave positioning accuracy (National Oceanic and Atmospheric Administration 1976, Hart and Downing 1977). Standard quality control measures and equipment include precision depth sounders (200 kHz or higher, narrow beam), tide and squat corrections, and a bar check (speed of sound correction). Even with all these accuracy-improving techniques, Morton, Stewart, and Germano (1984) reported an estimated repeatability of ±0.7 ft. The accuracy of an individual depth sounder measurement is estimated at 0.1 ft under ideal conditions, with more typical accuracies of 0.3 to 0.7 ft (Clausner, Birkemeier, and Clark 1986).

Some sources of error vary rapidly during the survey. Waves, signal ambiguities, and some components of positioning contribute randomly changing errors that are both positive and negative. These random variations tend to "average out" in volume change calculations. Hands (1976) showed that 80 percent of the sounding errors canceled out over 1,000-m profiles. Morton, Stewart, and Germano (1984) provide an additional discussion of percent errors in volume change.

Other critical items to consider in bathymetric survey planning are the density, pattern, and extent of the survey grid. The complexity of the survey effort should depend on the intent of the monitoring program. If the bathymetric survey is being conducted to verify the formation of significant mounds, or other changes in bathymetry, a minimal density survey plan may be adequate. Conversely, if the survey's purpose is to make an accurate measurement of the volume of material contained in a mound, closely spaced survey lines (i.e. 100- to 200-ft spacing) may be necessary. One or two crossed lines can be used to verify survey accuracy. Appropriately spaced parallel survey lines are preferred over a grid pattern due to the reduced ship time required. The survey pattern should be at right angles to the anticipated bathymetric slope or contour lines. Spacing will be a function of the size of the area, and a trade-off between accuracy and cost. When attempting to estimate volume of contaminated material, or the thickness of a sand cap over contaminated material, distances between survey lines of 50 to 80 ft may be required and cross surveys are a must.

Bathymetric surveys should extend beyond the area of interest to include areas "not affected" by the disposal operation. Initially, the survey boundaries should be 100 to 200 percent longer than the disposal site. For large sites (greater than 2 miles on a side) this figure may be reduced to 50 to 100 percent. As time passes and no changes occur, the area surveyed may be reduced, or expanded in the direction of material movement. Controlled disposal at precise coordinates or at marker buoys may reduce the required survey area to only a fraction of the total permitted disposal site.

Several new computer-integrated sounding systems have potential applications for monitoring disposal sites. (See Fredette et al., in preparation, for detailed information.)

Bathymetric surveys are often an expensive portion of a monitoring study. Proper scheduling to coincide with other monitoring activities may be cost-effective.

<u>Side-scan sonar</u>

Surface characteristics of the seafloor can be mapped using side-scan sonar. This tool uses acoustic energy projected laterally from a pair of transducers housed in a towed "fish." The received signal is transmitted through the tow cable to the shipboard receiver, which processes the signal and prints the record. The resulting image of the bottom is roughly similar to a continuous, oblique aerial photograph. However, the interpretation of side-scan sonar records requires some training and experience. Side-scan sonars for disposal site monitoring should usually be operated at a frequency of 100 or 500 kHz. The lower frequency has a greater range, but provides less detail than the higher frequency.

A survey run at 500 kHz distinguishes differences in bottom texture that can be used to map suspected variations in grain size. For example, moderately graded 0.25- and 0.13-mm sands may be identified (Figure 1). Spacing and orientation of sand ripples recorded on the sonograph can be used to interpret grain-size variations and direction and magnitude of sediment movement. Because ripples form more readily in sands than silts, and are usually larger for a coarser sand size, discrimination between placed and in-situ sediments may be further enhanced.

If bed-form or grain-size differences are substantial, a 100-kHz survey

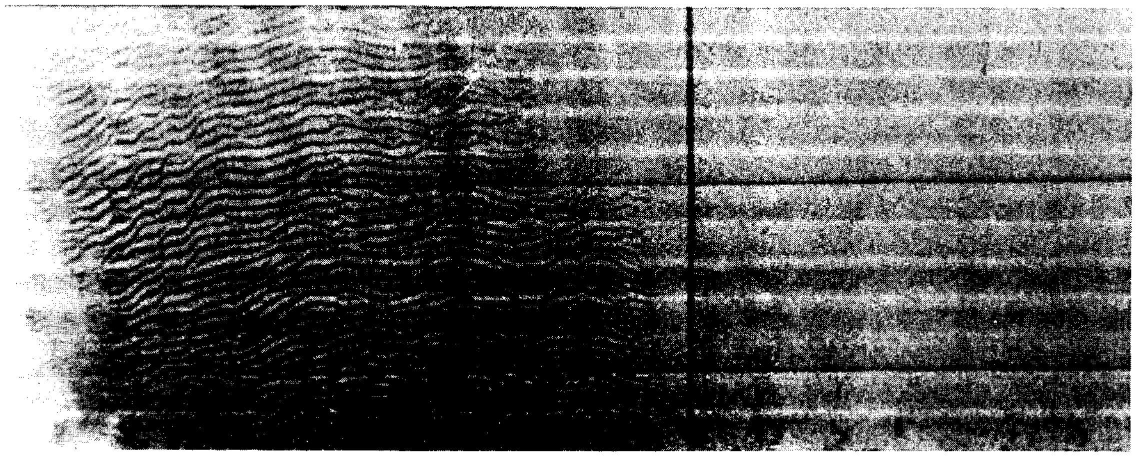
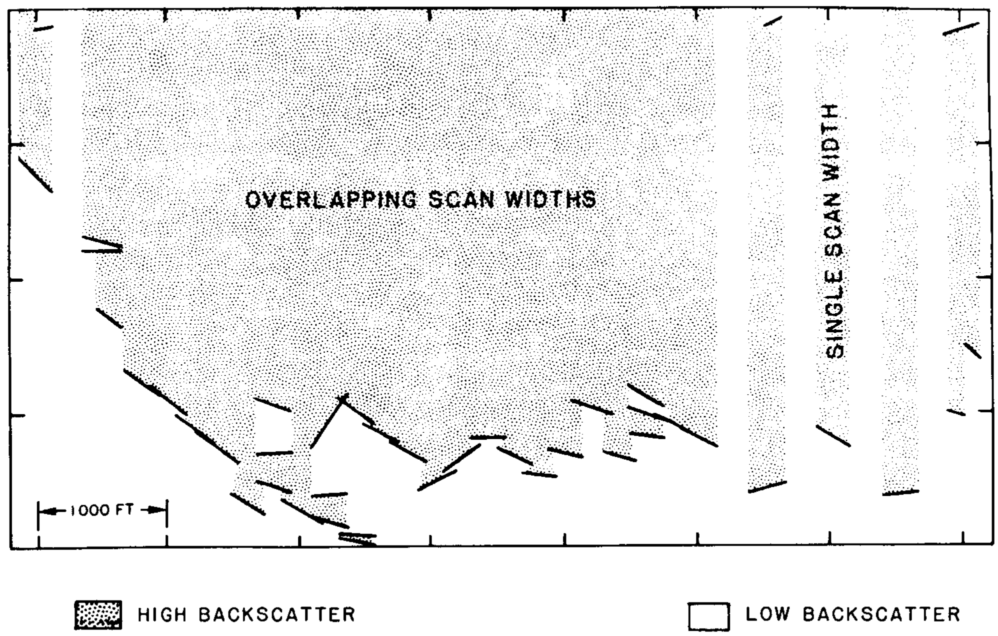


Figure 1. Side-scan sonar record of Dam Neck Disposal Site showing the difference between the native sand bottom (left) and disposal sediments (right)

may be preferred for its wider coverage in spite of poorer resolution. The lower frequency system may cover 200 to 400 m of bottom (depending on water depth) in a single scan as compared with 100 m for the 500-kHz system. Trial surveys with both frequencies are recommended when surveying unfamiliar areas. The grid spacing and overlap between the tracks, if any, will be a function of the purpose of the survey and the positioning system used. Complete coverage with 30- to 50-percent overlap should be required only for contaminated material, or to check coverage of capping operations. Relatively few tracks with no overlap may be appropriate for determining whether or not a stable deposit has begun to spread. A discrete track spacing of three times the swath width is recommended.

Overlapping coverage obtained with closely spaced survey lines, as in Figure 2a, allows precise and continuous mapping of the edges of disposal deposits. Side-scan surveys delineate the edge of disposal deposits more accurately than bathymetric surveys, provided the released and native sediments have distinctive backscatter characteristics. Definitive backscatter is likely, as the two materials frequently have different grain-size characteristics. Even if the grain sizes and reflection characteristics of the native

A. PREDISPOSAL



B. POSTDISPOSAL

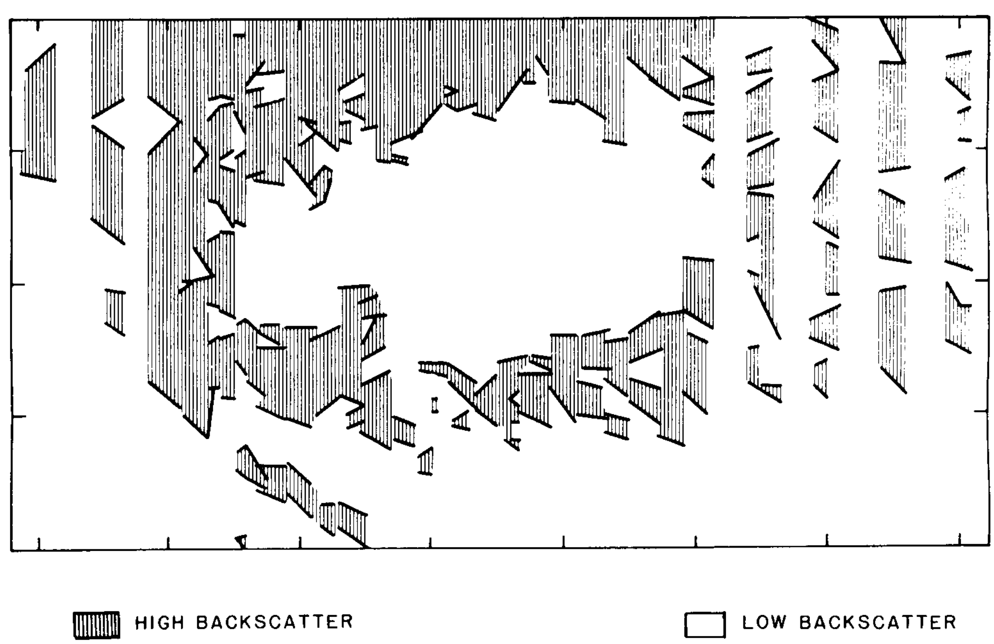


Figure 2. Predisposal and postdisposal maps of the Dam Neck Disposal Site produced from side-scan sonar records. In map B, the large low-backscatter area in the center represents the footprint of the disposal mound. Smaller areas scattered farther afield represent thin deposits of the finer grained material

and disposed material are similar, differences in bed forms can still be observed on a side-scan sonar record. To increase the probability of observing bed-form differences between native and disposal sediments, side-scan sonar surveys should be conducted as soon as possible after disposal. Deep water and less active driving forces may increase the allowable time between disposal and survey.

Individual side-scan sonar strips may be combined to observe a large area at one time. Heavy lines on each scan in Figure 2a indicate distinct contacts between high- and low-backscatter regions in the predisposal, native sediment population. Note that these contacts, which were identified on each scan separately, often match longitudinally when composed in the map view.

The low-backscatter region along the base of the predisposal survey indicates a silty bottom. The same low-backscatter region can be seen in the postdisposal survey 5 months later (Figure 2b). Reappearance of the same boundary on both surveys and the close match from one scan to the next within each survey establish position control accuracy.

The new low-backscatter area at the center of the postdisposal survey delineates the major deposit. Outlying low-backscatter patches represent many shallow depressions which now contain the finest disposal material that eroded from the central deposit.

At the edge of the major deposit and in outlying patches, the disposed material thins to a surface film. Bathymetry should be run in conjunction with side-scan surveys to determine where deposits are thick enough to warrant attention. These areal techniques extend and strengthen one another.

Subbottom profilers

The principles of subbottom acoustical profiling are fundamentally the same as those in acoustic depth sounding; however, subbottom acoustical systems employ a lower frequency, higher power signal to penetrate the shallow sediments of the seafloor. The signal is reflected from interfaces between sediment strata of different acoustic impedance. Subbottom technology was originally developed to search for deep petroleum traps. In contrast, the interest in disposal site monitoring is on high-precision, shallow penetration, to detect stratification within and just below deposits. Medium-power, high-resolution subbottom equipment on the order of 25 to 50 joules and 3.5 to 14 kHz best suits this type of application. The configuration of sediment layers within the disposed deposit can indicate characteristics such as degrees and uniformity of compaction, while the shape of the predisposal bottom may indicate subsidence of the underlying seafloor. Such settling, if unidentified, could be mistakenly interpreted as a loss of dredged material from the disposal site.

Geophysical surveys are now frequently conducted during archaeological (cultural resource) evaluation of potential disposal sites in the United States. Follow-up subbottom surveys, sediment cores, and geotechnical measurements may be needed to confirm the extent to which compaction and subsidence contribute to apparent losses of material from disposal mounds. Since subbottom surveys are usually performed in conjunction with bathymetric and/or side-scan sonar surveys, spacing and grid dimensions are usually related to those used for the other surveys. A significant thickness (at least 2 ft) of disposed material that is acoustically distinct from the predisposal seafloor will be necessary to obtain beneficial information from subbottom records. Under these restrictions, subbottom information may be used to check the thickness of a protective cap, but this information should be verified with core results.

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EEDP-01-11
March 1988

Environmental Effects of Dredging Technical Notes

CONTAMINANT MODELING

PURPOSE: This note provides initial information on contaminant models that are potentially applicable to situations where the presence of toxic materials in sediments complicates Corps of Engineers (CE) dredging activities.

BACKGROUND: Public concern about environmental contamination and increased regulatory requirements by local, State, and other Federal agencies mandate that Corps managers comprehensively address questions related to the presence of contaminants in dredged material. Modeling, in conjunction with field and laboratory evaluations, provides a valuable tool for answering questions raised when the presence of contaminated sediments complicates dredging operations. The emphasis by regulatory agencies on the use of models is steadily increasing. As a result, CE managers must be adequately informed about availability, capability, and applicability of various contaminant fate models.

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Background

As the need for applying contaminant fate models in the evaluation of dredging-related problems increases, CE managers must be familiar with the applicability and limitations of these models. Industry, academia, and government agencies have developed numerous contaminant fate models, many of which are neither easily accessible nor adequately documented and supported. This discussion focuses on five readily available, well-documented models--MINTEQ, EXAMS, MEXAMS, HSPF, and TOXIWASP. Characteristics of these models are summarized in Table 1. These five models are supported by the US Environmental Protection Agency (USEPA), which also continually refines and upgrades them. Acceptance and support of these models by this national regulatory agency increases the likelihood that one or more may be recommended to CE

Table 1

Summary of USEPA Contaminant Model Characteristics

Model	Chemicals	Aquatic Systems	Spatial Resolution	Temporal
MINTEQ	Metals	All	Zero-dimensional (0D) (mixed reactor)	Steady-state
EXAMS	Organics	Lake, river, tidally averaged estuary	Variable (0,1,2,3D)	Time-varying concentration with quasi-steady flow and loading
MEXAMS	Metals	Lake, river, tidally averaged estuary	Variable (0,1,2,3D)	Time-varying concentration with quasi-steady flow and loading
HSPF	Organics	River, run-of-river reservoir	1D longitudinal	Fully time-varying
TOXIWASP	Organics	All	Variable (0,1D,2D,3D)	Fully time-varying

managers for use in permit applications. A subsequent WES technical report will be prepared to discuss in detail the use of these USEPA models, and a variety of other contaminant fate models, in the evaluation of dredging-related problems.

USEPA Contaminant Fate Models

MINTEQ (Felmy, Girvin, and Jenne 1984) calculates aqueous geochemical phase equilbria for seven priority metal pollutants (arsenic, cadmium, copper, lead, nickel, silver, and zinc) using environmental variables, including pH, ionic strength, and temperature. MINTEQ is a mixed reactor or zero-dimensional model, i.e., there is no spatial resolution of the system modeled. MINTEQ is strictly an equilibrium model and cannot provide information about time-varying processes. However, MINTEQ does give an estimate of aqueous speciation and predicts the removal from solution of different metallic species by adsorption and precipitation. Toxicity of a metal varies with the form of the metal, so information about the proportion of each species present, as well as total metal concentration, is important in evaluating environmental impacts. An extensive thermodynamic data base for the seven metals MINTEQ simulates is included in the model package. An interactive preprocessor is available with MINTEQ to aid the user in setting up input data sets.

In dredging-related applications, MINTEQ can be used in assessing potential increases in toxicity, mobility, and bioavailability of metals in dredged materials at a disposal site, particularly when the chemical environment at the disposal site is significantly different from the dredging site. Additionally, MINTEQ can effectively aid in assessing the behavior of metals entering receiving waters from upland disposal facility runoff or water released from confined aquatic facilities.

EXAMS, the Exposure Analysis Modeling System (Burns and Cline 1985), a steady-flow compartment model, calculates the concentration and distribution of organic compounds in a system under a given pollutant load; EXAMS further determines the persistence of compounds in the system after the loading is removed. This model is applicable to systems where an assumption of constant pollutant loading and steady flow is reasonable over a period of a few weeks. Spatial resolution in one, two, or three dimensions can be obtained with EXAMS, depending on the number and arrangement of the compartments. EXAMS requires input of flow distribution within the system and calculates flow through the compartments based on volume conservation. EXAMS calculates dissolved and sediment-bound contaminant concentrations in both the water column and benthic layers. A major technical strength of this model is its handling of chemical kinetic processes such as hydrolysis, photolysis, microbial degradation, and volatilization. EXAMS has been designed so it can be easily applied for screening the behavior of numerous organic compounds and can be run interactively for rapid evaluation of scenarios. EXAMS is user friendly and provides on-line "help" to explain command options and input requirements.

EXAMS could be useful in screening for the presence of various contaminants in the sediments of lakes or streams with known or suspected loadings or for calculating contaminant concentrations downstream from a disposal facility. Additionally, EXAMS may have application in evaluating action/no-action alternatives where steady flow and loading assumptions are valid. EXAMS could be applied in estuarine applications where tidally averaged values for the flow are considered.

Limitations in handling suspended solids and sediment-water interactions diminish EXAMS' usefulness in analyzing many dredging-related problems. EXAMS does not simulate solids concentration in the water column; i.e., solids concentration must be supplied as input to the model. EXAMS does not include solids settling/resuspension processes. Net exchange of contaminants between the bed solids and suspended solids and between the water column and pore water is lumped into a single exchange coefficient. Loss of contaminants through burial is not accounted for in this model.

MEXAMS, Metal Exposure Analysis Modeling System (Felmy et al. 1982), combines the metal equilibrium model of MINTEQ with the transport structure of EXAMS, allowing calculation for a constant loading of the steady-state distribution of heavy metal species throughout a water body and persistence in the system after removal of the loading. Applicability of MEXAMS is similar to EXAMS except it simulates the metals in the MINTEQ data base (arsenic, cadmium, copper, lead, nickel, silver, and zinc) rather than organic compounds.

HSPF, Hydrologic Simulation Program - FORTRAN (Donigan et al. 1984), a one-dimensional model for nontidal rivers and unstratified lakes, is coupled with a watershed hydrologic model and nonpoint-source runoff algorithms. HSPF simulates organic pollutants in a time-varying mode. Sediment transport is calculated for three particle sizes (sand, silt, clay). HSPF has been used to evaluate best management practices for controlling nonpoint-source pollution from surface runoff, i.e., to determine impacts on receiving water quality from changes in watershed land use. This model may prove useful in examining effects of different watershed land use options on sediment quality or on sediment contributions to water quality problems that could impact dredging operations.

TOXIWASP, Toxics Water Analysis Simulation Program (Ambrose, Hill, and Mulkey 1983) is a time-varying, multidimensional, box-type model for simulating transport and fate of toxic organic chemicals in rivers, lakes, estuaries, or coastal waters. TOXIWASP segments can be arranged in a zero-, one-, two-, or three-dimensional configuration to achieve any required spatial resolution. Time-varying or steady-state flows can be used in WASP simulations and must be supplied to the model as input. For complex multidimensional water body applications, a separate hydrodynamic model simulation would probably be required in conjunction with the TOXIWASP applications. Three different size classes of sediment and contaminant concentration for each class are simulated in the most recent version of the model. TOXIWASP simulates multiple bed layers and allows net deposition or erosion of the bed surface and removal of contaminants from the system through burial. TOXIWASP incorporates chemical kinetics similar to EXAMS. As in EXAMS and HSPF, numerous parameters related to environmental and pollutant characteristics are required.

Although setting up TOXIWASP may prove data intensive and time consuming, the model's flexibility, including the ability to perform time-varying calculations, to simulate multiple sizes of suspended sediment and contaminant concentrations associated with each fraction, and to simulate a dynamic bed and loss of contaminant through burial, makes its potential great in dredging-related activities. While TOXIWASP is designed for organics, transport of metals, without speciation, could be performed within the model's framework. TOXIWASP should also be applicable in evaluating the action/no-action alternatives; i.e., predicting the effects of dredging or not dredging based on ambient conditions. TOXIWASP could be applied to evaluate the impacts of different dredging options on water quality. Furthermore, TOXIWASP is applicable in evaluating the impact of disposal operations on ambient conditions. For example, if linked to system hydrodynamics, TOXIWASP could be used to evaluate the impacts of a disposal site on ambient water and sediment quality. TOXIWASP could be used to simulate operations of a confined disposal facility to estimate return-flow concentrations from the site under different operational regimes.

In addition to the USEPA models, a variety of other models exist to model the fate of contaminants in aquatic systems. Several are capable of time-varying one- or two-dimensional system hydrodynamics, sediment transport, and first-order contaminant loss. Others make simplifying assumptions such as steady flow or mixed reactors to facilitate ease of application. Others consider partitioning/uptake of contaminants in the biota as well as the water column and sediment. Some trace biomagnification of contaminants through the food chain. Several models calculate concentration distributions through a series of sediment layers in the bed. PC-based spreadsheet type analytical models show utility for quick estimates of contaminant concentrations. A comprehensive discussion of available contaminant fate models and their potential application to dredging-related problems will be presented in a WES technical report scheduled for completion in 1988.

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EEDP-01-12
March 1988

Environmental Effects of Dredging
Technical Notes

USE OF SEABED DRIFTERS FOR LOCATING AND MONITORING DREDGED MATERIAL PLACEMENT SITES

PURPOSE: This note provides information on using seabed drifters (SBDs) to help locate optimum sites for placing dredged material, for both nearshore berms (feeder berms) and offshore mounds (stable berms). In addition, guidance is given for using SBDs to monitor potential sediment transport pathways at existing dredged material placement sites. Some of the information provided is based on results from DUCK 85 and SUPERDUCK, two large coastal processes experiments conducted during the fall of 1985 and 1986, respectively, on the outer banks of North Carolina. Seabed drifter investigations during SUPERDUCK were directed specifically toward offshore dredged material placement applications. Other guidance is based on monitoring associated with past and ongoing projects of Corps Districts.

BACKGROUND: This technical note is one of a series on monitoring of dredged material disposal sites. As mentioned in the first note of this series, "Acoustic Tools and Techniques for Physical Monitoring of Aquatic Dredged Material Disposal Sites" (Technical Note EEDP-01-10), increased use of nearshore disposal sites often requires additional monitoring. This is necessary to meet local, state, and Federal requirements to minimize adverse physical and biological impacts by determining the fate and stability of dredged material placed underwater.

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Introduction

Predicting the path of transported sediments is difficult. The physics of sediment erosion and transport are not completely understood. The complex interactions between waves and currents which erode sediment and the lack of sufficient information on space and time variations of near-bottom currents which transport the material make the predictions of numerical models subject to debate. SBDs can provide low-cost documentation of bottom-current

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circulation and may indicate possible paths for sediment movement. Their predictive capabilities are limited, however, because direct correlations between SBD movement and bottom sediment movement are not available. The second major limitation is that the path taken by the SBD and the rates of movement along that path are not usually known, although tracking is possible, as discussed later. Still, SBDs have a wide, although qualitative application to problems in coastal engineering. The focus of this note is on using SBDs to locate and monitor nearshore dredged material placement sites.

Following a brief description of seabed drifters, methods of deployment and recovery are presented. Recommendations on how to best use SBDs for siting and monitoring both feeder berms and offshore disposal mounds follow.

Physical Characteristics of SBDs

SBDs are commercially available, umbrella-shaped, plastic drogues with plastic tails (Figure 1). Typical disk size is 18 cm with four 2-cm vent holes. A 55-cm-long tail is most often used. Because the plastic is slightly buoyant, brass weights are attached to the tail to keep the SBDs on or near the bottom. Experiments have shown that 14-g weights are sufficient for this purpose (Hands 1987). Flume tests have shown that SBDs move at various percentages of the mean current speed, ranging from 73 percent at 15 cm/sec to 85 percent at 50 cm/sec, approaching 91 percent as the current increases.

Assembled SBDs are bulky, so it is usually best to put them together at the study site. Assembly takes 1 to 2 min per drifter. A hammer or crimping tool is needed to attach the weights.

SBDs are usually deployed with a postage-paid, waterproof card (usually polyvinyl chloride paper) to identify the time and location of recovery. Persons recovering the SBD and attached card are asked to fill in time, date, and exact location of the recovery (Figure 2). Small maps printed on the cards may assist in identifying recovery points when few landmarks are available. The card surface must be suitable for writing on with pen or pencil.

Experience along US coasts has shown that the public will consistently report SBD recoveries without the inducement of a monetary reward. The satisfaction of participating in a scientific study with practical consequences appears to be sufficient motivation for most volunteers. Posters placed in local shops, marinas, and docks and local media exposure (television

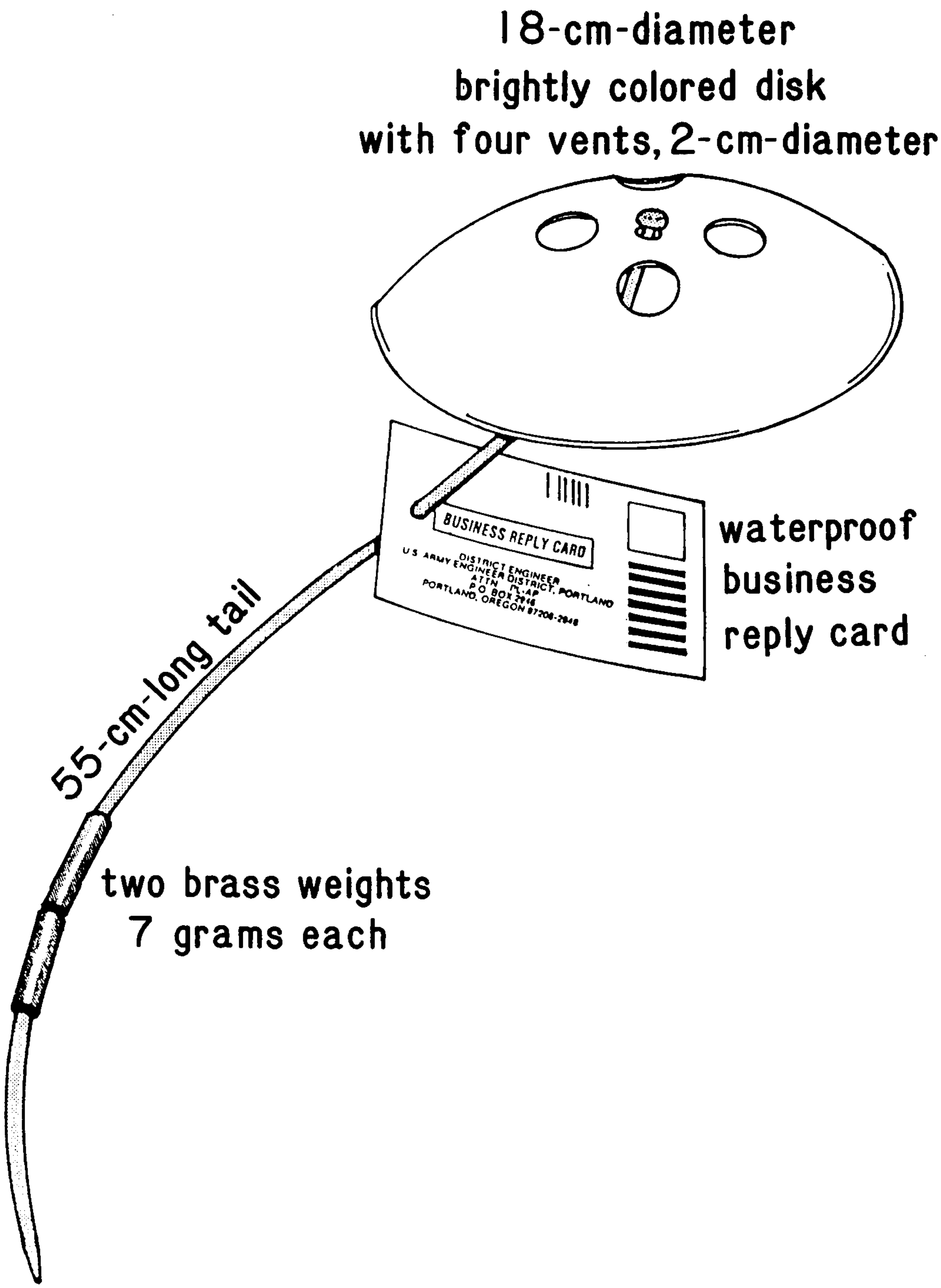

Figure 1. Seabed drifter

WES FORM 2365-4, 1 July 87

This device is part of a scientific study of ocean currents conducted by the US Army Corps of Engineers. Please help by completing this questionnaire (in pencil or ball-point) and placing it in any convenient mailbox. Keep or dispose of device.

EXACT RECOVERY LOCATION: ______________________

(May mark an X on the map) ______________________

DATE AND HOUR OF RECOVERY: ______________________

WAS THE CARD ATTACHED TO A PLASTIC STEM? YES NO

WAS THE STEM ATTACHED TO THE 7-INCH DISC? YES NO

DISC CARD STEM

Thank you for this vital assistance. Do you want return information? ______

YOUR NAME AND ADDRESS ______________________

______________________ TELEPHONE ______

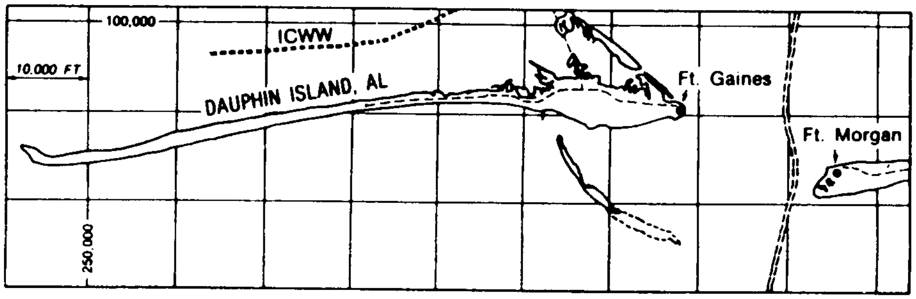

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Figure 2. Example of the waterproof information card used for a study near Dauphin Island, Alabama. The card can be customized for individual projects

and newspapers) alert the public to the nature of the study and provide instructions on what to do with the drifter and cards after recovery. Public participation is further motivated when the Corps follows up the voluntary return of an SBD card with a thank-you letter, describing the purpose of the project. In addition, public participation provides some low-cost public awareness of the Corps' role in attempting to solve coastal problems.

Deployment

SBDs are usually deployed in bundles of 25 to 30. A smaller number may make recovery of a statistically significant number of drifters difficult. More than 30 SBDs in a single bundle would be difficult to handle. Multiple releases of bundles of 25 to 30 are possible and may be used under certain situations described later. Typically, 25 to 30 individual SBDs are bundled together using 1/8-in. nylon (parachute) cord that is knotted to a salt ring (Figure 3). Recently, plastic cable ties have replaced the nylon cord, reducing the time needed to make the bundle. A piece of scrap metal or chain (approximately 2 to 5 lb) or a sandbag is used as weight to expedite descent. A second line tied to the weight is used to lower the weight and SBDs to the bottom. After 10 to 15 min, the salt ring dissolves, releasing the drifters at the bottom, after which time the weight can be recovered if desired. Should it not be practical to recover the weight, a bundle of SBDs with attached weight can safely free-fall through the water column.

Recovery

SBDs are usually recovered in two ways. For nearshore applications, the SBDs will wash up on shore and can simply be picked up. The other tested method of recovery is by trawling with a fishing boat. Sonic tags can be attached to the one or two of the SBDs to assist in recovery from fishing vessels or to track the path of individual SBDs. Sonic tags are small acoustical transmitters normally used to track fish. When the sonic tags are used in place of a weight on an SBD, they do not appear to adversely influence the motion of the SBD. The sonic-tagged SBDs can be tracked using a portable, directional hydrophone and receiver. Sonic tags cost $150 each, the receiver costs $1,400, and a hydrophone costs $800 (1987 dollars).

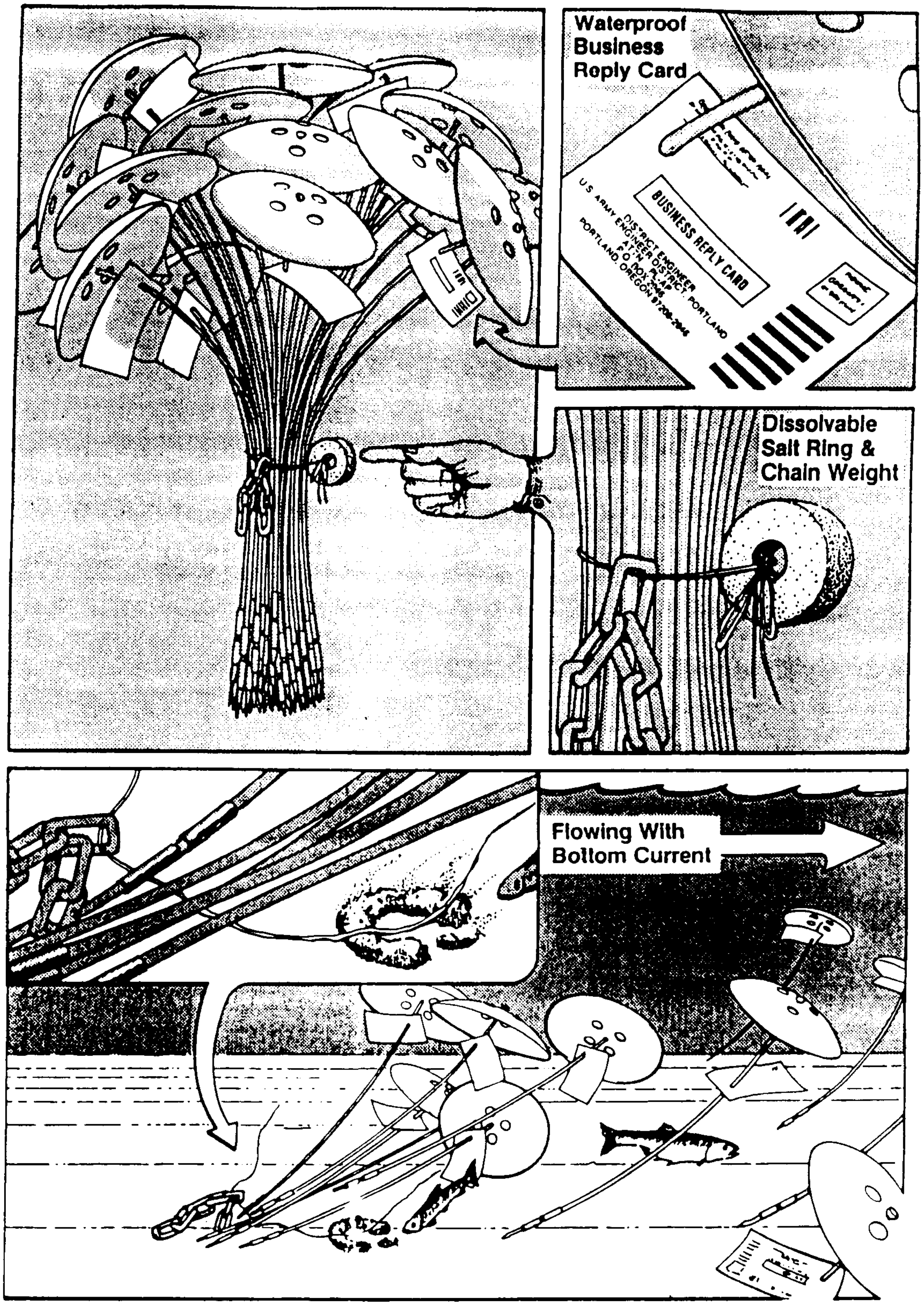

Figure 3. Seabed drifter deployment

Siting and Monitoring Feeder Berms

In the feeder berm concept, dredged sand is placed close to shore with the anticipation that the sand will remain in the nearshore sand prism and contribute to overall shore stability. Detailed guidance on design of feeder berms is not yet available. Monitored placements now under way will help develop a basis for design. Two monitored feeder berm projects have recently been constructed off the south shore of Long Island as part of the New York District's channel maintenance project at Fire Island and Jones Inlet, and a feeder berm was built off Sand Island, Alabama, as part of a National Demonstration project. SBDs are one part of the much larger monitoring effort at the Sand Island site. The Galveston District is now considering a feeder berm project near Brazos-Santiago Pass, Texas. An SBD study is planned to help locate the optimum site for sand placement. Many of the recommendations for that project are incorporated in this note.

SBDs have potential both to help determine the best feeder berm locations and to estimate in what direction the placed sediment will move. Studies before and after placement have value because the placed berm may alter the nearshore waves and currents to some degree. Whenever possible, SBDs should be used in conjunction with other measurements. Use of SBD drifter data without some knowledge of the forces that influence SBD movement (i.e., currents, waves, and wind) makes interpretation of these data very difficult.

Long-term current measurements at several elevations in the water column and at several locations at a site are most desirable, but may be too expensive for many projects. A less expensive alternative is to take current measurements over a tidal cycle at several locations during the periods SBDs are released. This would provide information on the tidal-induced current field but would not document wave-induced currents.

Wave data, both long-term and during the SBD releases, are also desirable when studying feeder berms. Long-term wave data indicate where placed sediment may move. Short-term measurements during SBD releases may aid in understanding SBD movements. While Wave Information Study-hindcasted wave data (Brooks and Corson 1984, McAneny 1986) are relatively inexpensive, most long-term measurements are expensive. Short-term, quality measurements are not practical in most cases due to the effort and expense associated with installation of a wave gage. Often, the only practical method of getting wave data is from visual observations, which is of fairly low reliability. Aerial photography can document wave refraction and diffraction patterns at a particular instant in time.

Wind data during deployment are often available from local airports and military or Coast Guard installations. A record of wind speed, duration, and direction for the extent of the SBD release and recovery period is usually easy to obtain. Recent studies have demonstrated the importance of local winds in driving nearshore currents (Hubertz 1984) and controlling the pattern of SBD returns (Hands 1987).

One of the most important aspects of using SBDs is deciding when to deploy them. Deploying SBDs over a variety of wind, wave, and tide conditions capable of transporting the placed sediments is desirable. However, the logistics of operating on short notice under potentially stormy conditions makes this difficult. A more reasonable scenario may be to identify a time period(s), say 1 to 2 months, when placed sediment is likely to move. During that time, intensive deployments could be made at several sites over 3- to 5-day periods. One deployment should be planned for the spring tide conditions when tidal currents are maximum. For example, in the siting plan for the Brazos-Santiago Pass project, 10 separate sets of releases at each of five sites over a variety of wind, wave, and tide conditions are planned. This plan requires 50 releases with 30 SBDs in each bundle for a total of 1,500 SBDs.

SBDs are usually released from a boat. Electronic positioning (preferably microwave, although well-calibrated LORAN C may be acceptable) is needed to determine location of the release sites. The same vessel can also be used to deploy current meters, track sonic tags attached to the drifters, and, possibly, trawl for drifters.

To ensure prompt, accurate recording of the location and time of SBD arrival onshore, persons involved in the operation should monitor the shore for at least a 24-hr period after deployment. A vehicle capable of traversing the beach, usually a four-wheel-drive or all-terrain vehicle with an accurate odometer, should be available. Special authorization from local authorities may be needed to use a vehicle on the beach. As mentioned earlier, a public awareness program will increase the likelihood of volunteer data reporting.

Ongoing studies indicate that SBDs tend to arrive onshore during rising tides, particularly during the last few hours before high tide. Experience from the DUCK Experiments (Hands 1987) has shown that high percentages of the drifters can be recovered (>80 percent) and that many will arrive onshore simultaneously (within 1 hr). Under favorable conditions (swell waves, offshore winds, and onshore bottom currents), SBDs released from 1,500 ft offshore came ashore in 6 hr. Thus, it may be essential that, following SBD releases, paid participants keep searching every few hours for several miles up and down coast from the release point.

While a majority of the SBDs may be recovered within a few miles of the release point, some travel considerable distances. Figure 4 shows the distribution of SBD returns based on deployment of 1,500 SBDs from six release sites on five separate occasions over 2 months at the Sand Island site near Mobile, Ala. Although the environmental conditions have not as yet been strong enough to cause significant movement of the berm, when the berm does move, these data should allow better interpretation of where the sand is moving.

The cost of the SBDs is usually only a small portion of the total cost of a comprehensive study. For example, less than 10 percent of the cost of the Brazos-Santiago Pass study designated for locating the feeder berm site was for the 1,500 SBDs (approximately $3.00 each).

When monitoring existing feeder berms, SBDs can play a supportive role to other methods of monitoring sediment dispersion. Changes in bathymetry and beach profiles, along with sediment sampling and side-scan sonar, are important techniques for documenting berm movement. Monitoring of the forcing functions, waves, and currents is recommended. The monitoring guidelines (Fredette et al., in preparation) and Technical Note EEDP-01-10 on acoustic tools provide guidance in this area.

Siting and Monitoring Offshore Berms

In addition to being potentially useful tools for siting and monitoring feeder berms, SBDs also have potential for siting and monitoring offshore sites, both stable berms (a permanent mound formed by placing dredged material at a specific location) and dispersive sites. Many of the methods and techniques used for monitoring feeder berms apply to other offshore areas as well. Usually, recoveries from the beach become less important, and recoveries from trawling become more important. Tracking SBDs with sonic transmitters to determine their path and locate optimum areas for trawling will also become more important.

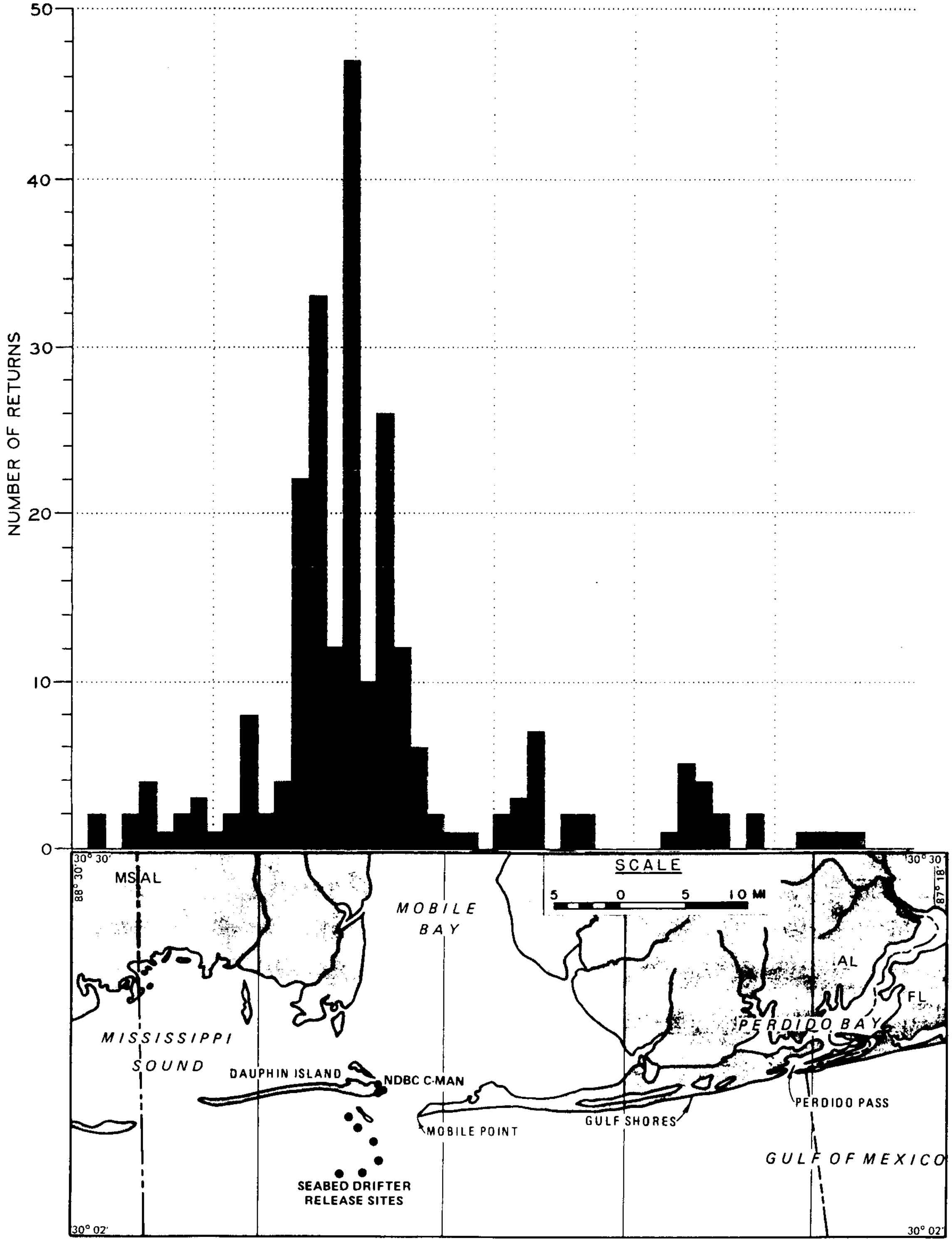

Figure 4. Longshore distribution of 191 Sand Island SBDs recovered as of 7 August 1987. A total of 1,500 drifters released between 3 March and 5 May 1987

Because the percentages of recovery are usually lower for deeper water, more drifters should be deployed. The number released will depend on the number of releases and the intensity of the search effort. When a series of closely spaced releases are planned along with sonic tracking and trawling, release quantities of 100 to 200 may be appropriate. For releases where local fishing efforts will be the primary source of recovery, larger releases of up to 300 to 1,000 or more SBDs may be necessary.

As with nearshore releases, bottom-current measurements will help to determine probable SBD paths and indicate potential areas for trawl searches. Electronic positioning for release and search vessels is also a requirement. In certain locations, tracking may be crucial. For example, SBDs released close to an inlet may travel in and out of the inlet on tidal currents before washing ashore some distance away from the inlet. Closely tracking the SBDs would provide data on whether the SBDs are entering the inlet. In these cases, placing the hydrophone on a very maneuverable boat or using two vessels, each with a hydrophone and receiver, have been suggested to more closely monitor SBD movements (Hicks 1986).

A potentially much more effective method of searching for SBDs may be to use side-scan sonar. Most commercially available sonic tags transmit at 74 kHz, which cannot be readily detected by 100-kHz commercial side-scan sonars. The Coastal Engineering Research Center of the US Army Engineer Waterways Experiment Station has had initial success with side-scan sonar detection of 100-kHz sonic tags. This new development should make tracking easier, faster, and more reliable.

The Portland District used SBDs to help determine whether sediments disposed in a site not far from the Siuslaw River entrance were making their way back into the inlet (Hicks 1986). Based on the pattern of SBD recoveries, current meter measurements, and surface dye movements during disposal operations, the Portland District recommended that the disposal site be moved farther offshore to deeper water.

Summary and Recommendations

SBDs are inexpensive bottom current-following drogues that show integrated bottom-current paths. With careful application they can give an indication of the potential paths of sediment transport. Thus, SBDs are promising tools for siting and monitoring both feeder berms and offshore sites, where movement of placed sediments is important. Their relatively low cost and high visibility in the public eye are the primary advantages. Lack of definite correlation between sediment movement and SBD movement and uncertainty about the path SBDs have followed are the major limitations. Consequently, SBD data should always be supplemented with as many coastal process measurements (such as currents, waves, and winds) as possible.

Existing and future projects will provide additional data on SBD effectiveness. When sufficient information becomes available, it will be included in a report or an updated version of this note.

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EEDP-01-13
April 1988

Environmental Effects of Dredging Technical Notes

RELATIONSHIP BETWEEN PCB TISSUE RESIDUES AND REPRODUCTIVE SUCCESS OF FATHEAD MINNOWS

PURPOSE: This technical note provides initial guidance for interpreting the biological consequences of bioaccumulation in aquatic organisms. Specifically, the relationship between polychlorinated biphenyl (PCB) tissue residues and reproductive success in the fathead minnow, *Pimephales promelas*, is examined.

BACKGROUND: The US Army Corps of Engineers often conducts, or requires to be conducted, an assessment of potential bioaccumulation of environmental contaminants from sediments scheduled for dredging and open-water disposal. At present, however, there is no generally accepted guidance to assist in the interpretation of the biological consequences of specific levels of bioaccumulation. To provide an initial basis for such guidance, the Environmental Laboratory of the US Army Engineer Waterways Experiment Station is conducting both literature data base analyses and experimental laboratory studies as part of its Long-Term Effects of Dredging Operations (LEDO) Program. This technical note discusses a portion of the laboratory effort.

ADDITIONAL INFORMATION OR QUESTIONS: Contact Dr. Tom M. Dillon, commercial or FTS: (601) 634-3922, or Dr. Robert M. Engler, Program Manager, Environmental Effects of Dredging Programs, (601) 634-3624.

Materials and Methods

Experimental design

Adult fathead minnows, *Pimephales promelas*, were obtained from Northeastern Biologists, Rhinebeck, N.Y. The response of *P. promelas* in a variety of toxicity tests has been shown to be representative of most freshwater fish (Suter et al. 1987). Following a 30-day acclimation period to all conditions except sediment, fish were placed in 40-ℓ glass aquaria containing a 2- to 4-cm layer of sediment. Sediments containing 0.82, 14.0, and 27.0 μg/g dry weight PCB, expressed as Aroclor 1254 equivalents, were identified as low, medium, and high treatments, respectively. Sediment was collected from

Sheboygan Harbor, Wis., an inland waterway known to contain substantial amounts of PCBs. Aquaria containing only clear water served as the control treatment. Moderate aeration and a continuous flow (80 mℓ/min) of aged, charcoal-filtered tap water were provided to all aquaria.

Fathead minnows were exposed for 16 weeks total to the four treatments. For the first 5 weeks fish were maintained at 20° C with a photoperiod of 12 hr light:12 hr dark. The water temperature was then increased 1 degree per day to 26° C and the photoperiod lengthened 1 hr every other day to 16 hr light:8 hr dark. Spawning substrates were also introduced into each aquarium. These steps were taken to induce gametogenesis, sexual dimorphism, and reproductive activities in fathead minnows (Denny 1987). Fecundity and frequency of egg production were monitored daily for the remainder of the 16-week exposure.

Chemical analysis

Fish were collected after 7 and 16 weeks of exposure, frozen, and saved for determinations of tissue PCB residues. All chemical analyses were conducted by the Tennessee Valley Authority, Chattanooga, Tenn., using a Hewlett-Packard gas chromatograph equipped with a 30-meter DB-5 fused capillary column and electron capture detector. The detection limit for Aroclor 1254 was 0.10 μg/g and for individual PCB congeners, 0.01 μg/g. Nomenclature for PCB congeners follows Ballschmiter and Zell (1980) as adopted by the International Union of Pure and Applied Chemists (IUPAC).

Statistical analysis

Treatment effects on all parameters were analyzed via one-way analysis of variance. Differences were considered statistically significant at $p < 0.05$. Square root or log 10 transformations were used if data sets were not homogeneous. Arc sine transformations were used for nonhomogeneous data expressed as percentages. Mean separation for homogeneous data was achieved via the Waller-Duncan k-ratio t test. When transformations were unsuccessful in achieving homogeneity a Proc Rank nonparametric procedure was used for mean separation (SAS Institute, Inc. 1985). PCB concentrations reported to be less than the detection limit were considered equal to the detection limit in all calculations. To facilitate residue-effects observations, the reproduction data were normalized to control values and expressed as a percentage.

Results and Discussion

A clear and inverse relationship existed between the reproductive success of sediment-exposed minnows and their internal PCB tissue concentrations (Figure 1). Reproduction was significantly impaired in fish exposed to the

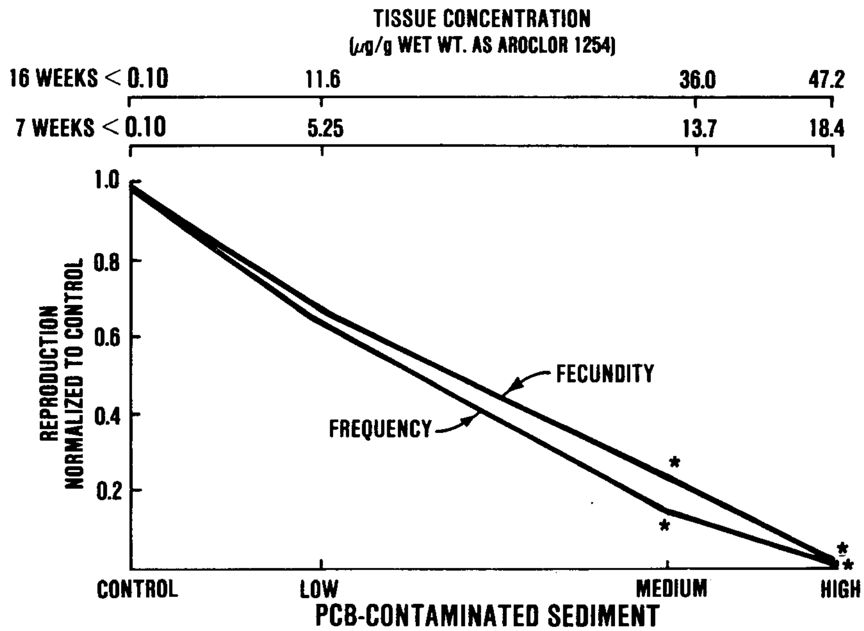

Figure 1. The relationship between reproduction and PCB tissue concentrations in fish exposed to PCB-contaminated sediment; asterisk indicates significantly different from controls

medium and high PCB-contaminated sediment treatments compared to fish in the controls and low PCB-contaminated sediment. The magnitude of this inhibition was large; approximately 80 to 100 percent of control values. Mean PCB concentrations in these affected fish ranged from 13.7 to 47.2 µg/g (ppm) wet weight, expressed as Aroclor 1254 equivalents. Bengtsson (1980) reported similar results for the saltwater minnow, *Phoxinus phoxinus*, chronically exposed to PCBs. In that study, mean tissue concentrations of 170 mg/kg (ppm) fresh weight were associated with a significant decrease in reproduction while fish with lower tissue concentrations, 0.2 to 15 mg/kg (ppm) fresh weight, were apparently unaffected.

Numerous reviews have demonstrated that background PCB residues in feral fish throughout the United States generally range from a few tenths of a ppm up to low single-digit ppm's (Peakall 1975, Butler and Schutzmann 1978, Wassermann et al. 1979, and Veith et al. 1981). Freshwater fish collected from some of the more highly industrialized waterways in Lake Michigan contained PCB concentrations (as Aroclor 1254) ranging from a few ppm wet weight to a high of 15 ppm (Veith 1975). Saltwater fish collected from one of the most highly PCB-contaminated estuaries in the United States, New Bedford Bay, Mass., generally had single-digit ppm concentrations of PCB with two fish species, cunner and American eel, being exceptionally high--30 to 130 ppm (Weaver 1984). In this experiment fathead minnows with significantly impaired reproduction had double-digit ppm PCB concentrations in their tissue. This level of PCB in the tissue is associated with fish inhabiting some of the more highly contaminated waterways in the United States.

Although the mechanism responsible for the observed decrease in fathead minnow reproduction is unknown, several possibilities, based on supporting data, can be considered. For example, one might suspect that the large decrease in reproduction in fish was due to high mortalities. However, this was not the case. At the end of the 16-week exposure, percent survival in all treatments was quite high (80 to 100 percent) and there was no significant effect among treatments for either male or female fish (Figure 2). Neither

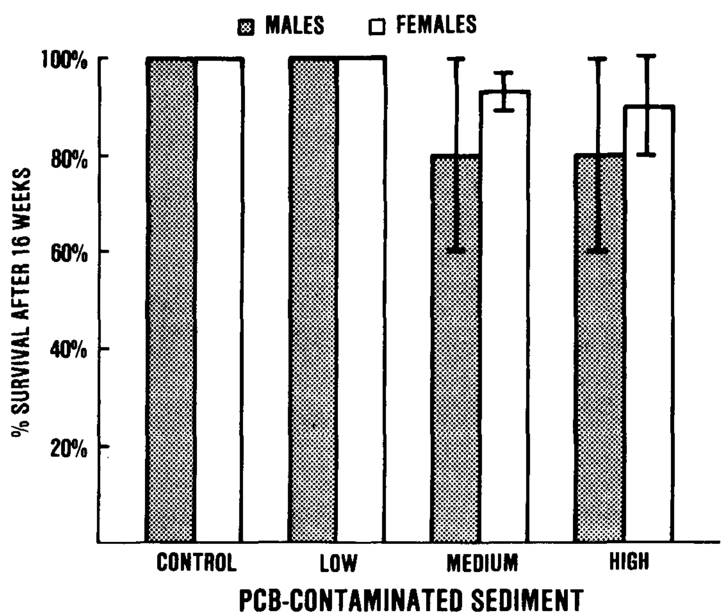

Figure 2. Survival of fish exposed to PCB-contaminated sediment for 16 weeks

does the impaired reproduction appear to be due to a lack of feeding or a general wasting of the exposed fish. There was no significant treatment effect on the mass of male or female fish at the end of the experiment (Figure 3).

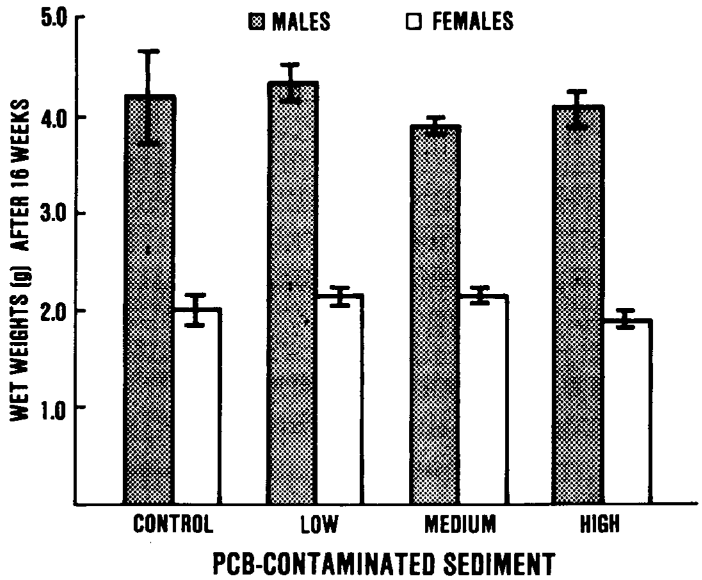

Figure 3. Wet weights of fish exposed to PCB-contaminated sediment for 16 weeks

There was, however, a significant increase in percent lipid at the end of the experiment in affected fish exposed to the medium and high sediment treatments (Figure 4). This indicates that lipid metabolism was somehow disrupted.

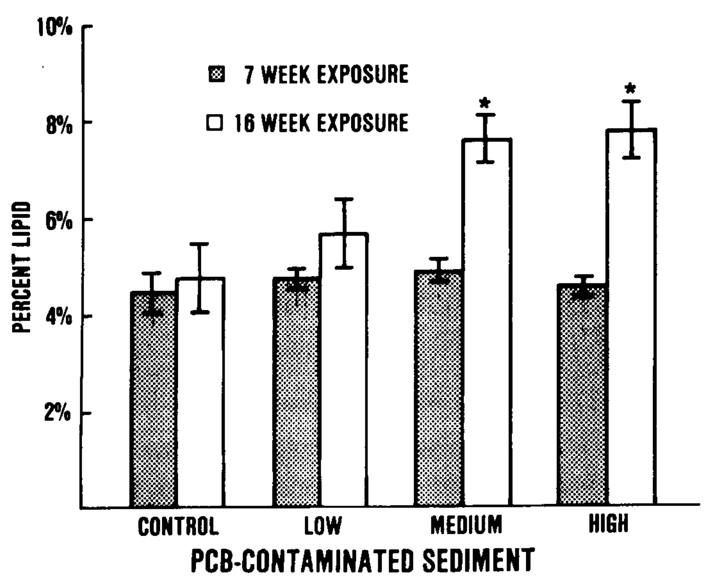

Figure 4. Percent lipid in fish exposed to PCB-contaminated sediment; asterisk indicates significantly different from controls

Elevated lipids, as well as changes in lipid-synthesizing enzymes, have been observed in PCB-exposed mammals (Hansell and Ecobichon 1974, Holub et al. 1975, and Ishidate et al. 1978). Some evidence suggests that these elevations are due to an inhibition of catabolism rather than an active synthesis of lipids (Ishidate and Nakazawa 1976). An increase in the proportion of saturated fatty acids has also been observed in saltwater fish exposed to PCBs (Caldwell et al. 1979). This response was similar to that observed in fish acclimating to elevated temperatures. Since the lipid data from this experiment is not qualitative, effects on specific lipid pools cannot be determined at this time.

The bioaccumulation of specific PCB congeners may have also adversely affected fish reproduction (Figure 5). Some of these congeners, especially IUPAC numbers 52, 128, 138, 153, and 180, are known or suspected inducers of the mixed-function oxidase (MFO) enzyme system in fish (Chambers and Yarbrough 1976, Lech et al. 1982, Clarke 1986, and Kleinow et al. 1987). As the name implies, the MFO system is responsible for the insertion of molecular oxygen into numerous organic substrates which are required for a variety of biochemical reactions. The MFO system is embedded in the lipid matrix of biological membranes and any disruption in that matrix will affect normal MFO activity (Stier 1976). Certain PCB congeners, because of their MFO-inducing properties, could be exerting a direct toxic effect by disrupting normal MFO activity. They may also be acting indirectly by generating excessive numbers of electrophilic metabolites. These highly reactive compounds, typified by the diol epoxide of benzo(a)pyrene, are known to interact detrimentally with biological membranes and genetic material (Ahokas 1979).

One of the major functions of the MFO system is the metabolism of lipids. Two specific types of lipid metabolism which are essential to fish reproduction are the synthesis of sex hormones (steroidogenesis) and the production of egg yolk. The latter process is dependent on the production of the phospholipoprotein, vitellogenin, in the liver. Both processes are initiated and controlled by the hypothalmic-pituitary-gonadal axis in fish (Peter and Crim 1979). Although many investigators have shown that exposure of fish to PCBs can have a significant effect on both steroidogenesis and vitellogenesis (Sivarajah et al. 1978, Hansson et al. 1980, Chen and Sonstegard 1984, Forlin et al. 1984, and Wester et al. 1985), no clear pattern of effect has emerged. This is probably due, at least in part, to the complexity of the

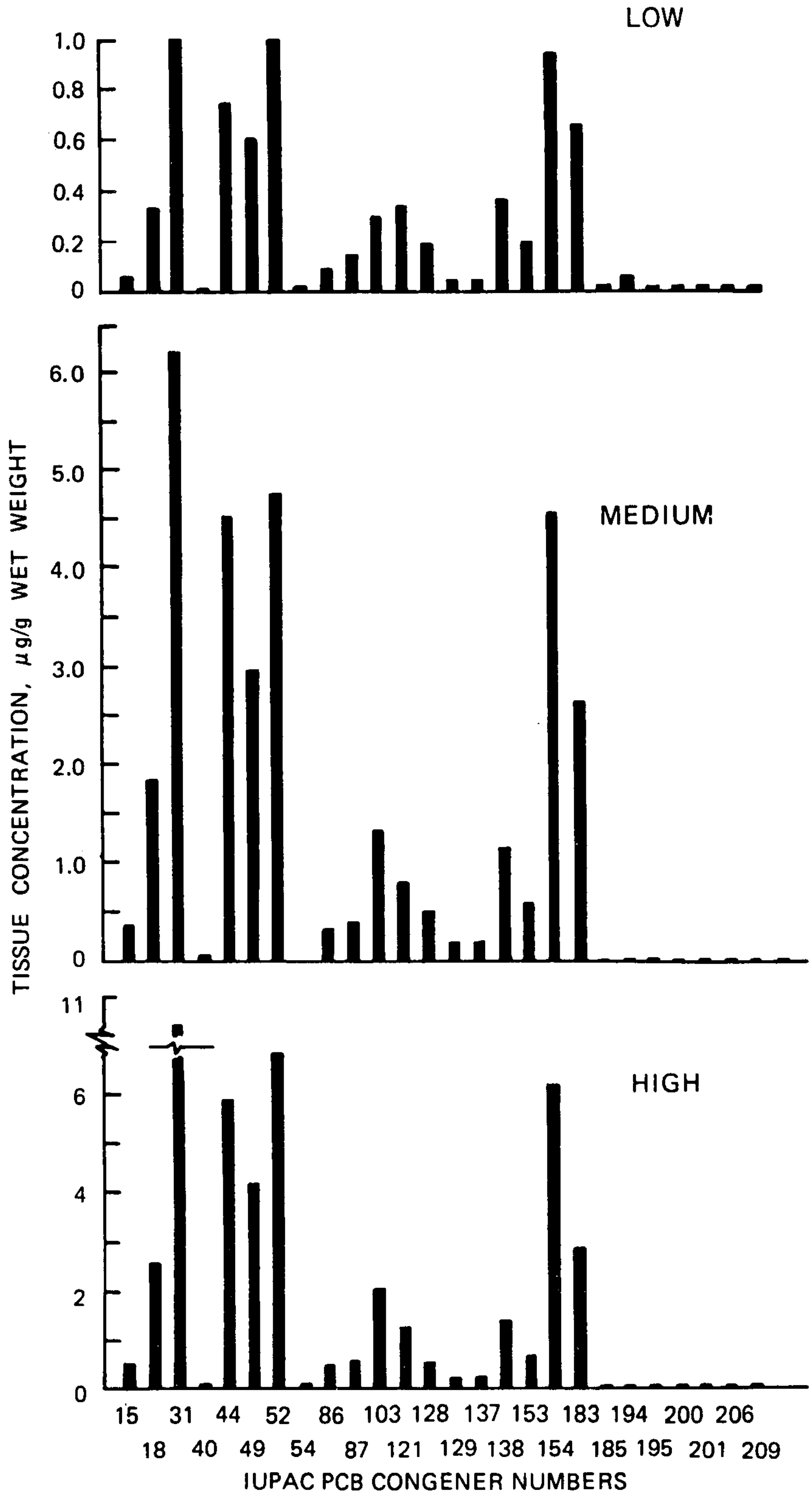

Figure 5. Concentrations of individual PCB congeners in fish exposed to PCB-contaminated sediment for 16 weeks

biochemistry of fish reproduction (Stegeman and Woodin 1984). Nevertheless, the accumulation of significant amounts of MFO-inducing PCB congeners, coupled with elevated lipid levels in the affected fish, strongly suggests that a disruption in lipid metabolism may be responsible for the impaired reproduction observed in fish exposed to PCBs and PCB-contaminated sediment.

Conclusions

Results from the testing of adult fathead minnows, *Pimephales promelas*, indicated that PCB-contaminated sediments had a significant deleterious effect of the species' fecundity and frequency of reproduction. Affected fish had double-digit ppm tissue concentrations of PCBs, expressed as Aroclor 1254. These tissue concentrations correspond to PCB residues of fish inhabiting highly contaminated waterways in the United States. The manner in which PCBs exert their detrimental effect on fish reproduction is unknown, but may involve some aspect of lipid metabolism. Pathways responsible for steroidogenesis and vitellogenesis may be especially vulnerable. Affected fish accumulated substantial amounts of specific PCB congeners which are known or suspected MFO inducers. Further investigation as to their potential biological effect will assist interpretation of the biological consequences of PCB bioaccumulation in aquatic animals.

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EEDP-01-14
December 1988

Environmental Effects of Dredging Technical Notes

INFLUENCE OF ENVIRONMENTAL VARIABLES ON BIOACCUMULATION OF MERCURY

PURPOSE: This note examines the effects of environmental factors on the bioavailability of mercury from sediment and describes results of a laboratory experiment to assess the influence of temperature, salinity, and suspended sediment on bioaccumulation of mercury in estuarine clams and killifish.

BACKGROUND: Public laws regulating dredged material disposal (Section 404 of the Clean Water Act and Section 103 of the Ocean Dumping Act) require ecological evaluation prior to the permitting of operations. Assessment of the potential for bioaccumulation of chemical contaminants in sediment, including heavy metals, is required as part of the evaluation process. Metals can represent significant sediment contamination in the vicinity of industrial and commercial point sources. Mercury, in particular, enters the aquatic environment in various forms from chloralkali and instrumentation plants, paints, pulp and paper manufacture, agricultural sources, and other nonpoint sources such as atmospheric deposition (Khalid et al. 1977). Because sediment serves as a sink for mercury, dredging and disposal operations can affect the bioavailability of mercury to aquatic organisms. In general, metals in sediment have low bioavailability in reduced environments such as aquatic disposal sites, but may be highly bioavailable in upland disposal sites where the dredged material is subject to drying and oxidation.

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Introduction

Mercury is among the most toxic of the heavy metals and thus can greatly concern regulators faced with the dredging and disposal of mercury-contaminated sediment. Acute toxicity tests have demonstrated the lethality of mercury to various aquatic organisms, including polychaetes (Warren 1976), adult and larval crabs (Vernberg and Vernberg 1972; McKenney and Costlow 1981, 1982), and daphnids (Khangarot, Ray, and Chandra 1987), especially under conditions of environmental stress. Mercury can exist in various forms in the environment, including inorganic mercury (Hg^0, Hg^{1+}, Hg^{2+}) and various organic complexes such as the highly toxic methyl mercury. Besides lethality, various sublethal effects in aquatic organisms have been attributed to methyl mercury exposure, notably interference with development or regeneration in tadpoles, fish, and crabs (Chang, Reuhl, and Dudley 1974; Weis and Weis 1978; Callahan and Weis 1983).

Methylation of mercury can occur either through biotic or abiotic processes (Nagase et al. 1982, 1984), although the environmental significance of abiotic methylation is probably minor (Berman and Bartha 1986b). In the aquatic environment, methylation of mercury is likely to occur wherever there is microbial activity, e.g., in the sediment, water column, and digestive tract of fish (Rudd et al. 1983). Methylation of mercury can occur in either anaerobic sediment (Hammer, Merkowsky, and Huang 1988) or aerobic sediment (Fagerstrom and Jernelov 1970). High sulfide concentrations inhibit methyl mercury production (Berman and Bartha 1986a).

Various investigators have reported high mercury concentrations in sediment and organisms in the vicinity of mercury pollution sources (Kiørboe, Møhlenberg, and Riisgård 1983; Mikac et al. 1985). Mercury concentrations in fish and crustaceans taken from the New York Bight, nearby New Jersey, and Long Island Sound ranged from 0.08 to 2.3 parts per million (ppm) (Roberts, Hill, and Tifft 1982). However, animals exposed to New York Harbor sediment in laboratory studies did not accumulate mercury even though sediment mercury concentrations ranged from 2 to 35 ppm (Rubinstein, Lores, and Gregory 1983). These investigators proposed that high organic or sulfide content in the sediment bound the mercury and rendered it unavailable. Weis, Weis, and Bogden (1986) reported no correlation between mercury bioaccumulation in the killifish *Fundulus* and mercury concentrations in the sediment to which the fish were exposed.

Bioaccumulation of mercury from sediment by aquatic organisms can be influenced by a number of environmental factors, including temperature, salinity, dissolved oxygen, pH and alkalinity, suspended sediment, organic carbon content of the sediment, and presence of other elements. Of these factors, the last may be one of the most important. Sulfide (S^{2-}), in particular, may be the primary regulator of Hg^{2+} activity in natural waters (Bjornberg, Håkanson, and Lundbergh 1988). If the pH is high or the redox potential is low, then sulfide activity will be high and virtually all mercury will be precipitated as very poorly soluble HgS. Conversely, at lower pH or higher redox potential, sulfide activity will be lower, and mercury activity and bioavailability will be higher. Two other elements, selenium and tellurium, interact with mercury in the same way as sulfide.

The influence of temperature and salinity on mercury bioavailability and toxicity is not well understood, and diverse observations have been reported. Parks, Sutton, and Hollinger (1984) noted that increases in water temperature result in increases in total mercury and methyl mercury concentrations in water. Weis, Weis, and Bogden (1986) reported a fivefold increase in mercury uptake by *Fundulus* during the summer months in a mercury-contaminated tidal creek, whereas Cossa and Rondeau (1985) found mercury bioaccumulation in mussels to be lower in summer than during other seasons. Smith, Green, and Lutz (1975) found temperature to have no effect on the rate of mercury uptake or elimination by freshwater clams. Olson and Harrel (1973) reported higher toxicity of mercury to the estuarine clam *Rangia cuneata* in fresh water than in salinities of 5.5 and 22 parts per thousand (ppt). In a factorial experiment, Khayrallah (1985) observed that the toxicity of mercury to amphipods was directly related to concentration and temperature, but inversely related to salinity and age of the test animals.

Several investigators noted increased mercury accumulation in biota at low dissolved oxygen levels (Weis, Weis, and Bogden 1986; Hammer, Merkowsky, and Huang 1988). Björnberg, Håkanson, and Lundbergh (1988) postulated that this phenomenon may be due to increased methylation of mercury under anoxic conditions.

Mercury partitions readily from water to suspended sediment (Sayler and Colwell 1976), and also to the organic fraction in oxidized surface layers of sediment (Langston 1982). In either case the mercury may be largely unavailable to organisms (Langston 1986, Rudd and Turner 1983). Rudd et al. (1983) noted that mercury methylation and bioaccumulation are inversely related to the concentration of mercury-binding particulates present. Breteler, Valiela, and Teal (1981) found the highest concentrations of mercury in animals living in marsh sediments lowest in organic matter. Contradictory data suggest that humic substances transfer mercury from sediment to the water phase and then to biota (Surma-Aho et al. 1986); thus, high humic content in sediment may be linked to high mercury content in biota (Björnberg, Håkanson, and Lundbergh

1988). Mercury in humic particles can be converted to bioavailable forms by microbial methylation.

The laboratory experiment described in this note was designed to assess the influence of temperature, salinity, and mercury-contaminated suspended sediment on bioaccumulation of mercury by clams (*Rangia cuneata*) and killifish (*Fundulus heteroclitus*). Other environmental factors such as dissolved oxygen and pH were held constant or nearly constant.

Materials and Methods

The experimental system used was the Flow-through Aquatic Toxicology Exposure System (FATES) developed at the US Army Engineer Waterways Experiment Station (WES). This system consists of 24 flow-through cylindrical aquaria having round bottoms and a 75-ℓ capacity. The entire system is controlled by a microcomputer that interfaces with valves and other mechanical equipment via microprocessor-based data acquisition and control hardware. Temperature, salinity, suspended sediment loads, and water flow-through rates are all controlled and may be set to whatever parameters are needed in the experiment. The system also incorporates a light level timer for day/night simulation. Commercially available artificial sea salt is mixed with aged, dechlorinated tap water when saltwater experiments are conducted in FATES.

Test sediment was collected from a mercury-contaminated tidal creek in the northeastern United States and held at 4° C until used. The sediment was diluted with deoxygenated water and mixed with a high-speed, shear-type mixer to provide a uniform, small particle-size slurry. The slurry was pumped into a stainless steel, cone-bottom hopper and kept in constant circulation to prevent settling. Once the hopper was pressurized with approximately 2 psi of argon gas to retard oxidation, the slurry was then ready for use in FATES.

The level of suspended sediment in each aquarium was maintained by a computer-controlled feedback system. A transmissometer head, located in each aquarium, measured suspended sediment level by light transmission every 10 to 15 min and if the level was low, an injector valve was activated to pulse a small amount of slurry into the aquarium. A recirculating pump dispersed the slurry uniformly throughout the aquarium. Average suspended sediment concentrations were maintained near the targeted levels.

The water flow-through volume in each aquarium was computer controlled at 300 mℓ/min, allowing 95 percent water replacement every 12 hr to maintain high water quality. Temperature of the aquaria was maintained with a heat-exchanger system. The computer checked the temperature of the heat exchangers several times every minute and added hot or cold water as needed to the heat exchangers to keep the temperature constant. All 24 aquaria were sequentially sampled every 6 hr for temperature, dissolved oxygen, pH, and conductivity. These data were written to a computer disk file and printout for later analysis.

The experiment consisted of six 7-day (168-hr) exposures in various temperature and salinity combinations (Table 1). During each exposure, suspended sediment concentrations were maintained in individual aquaria at nominal levels of 0, 5, 15, 25, 50 and 100 mg/ℓ. Each aquarium contained 1 to 2 ℓ of bedded sediment below a screen to prevent test animals from having direct contact with the sediment. In addition three control aquaria contained clean pea gravel with no bedded or suspended sediment. The assignment of controls and suspended sediment levels to aquaria was done in a random manner.

Table 1

Environmental Conditions Used for Each of 6 Runs

Run No.	Salinity ppt	Temperature °C
1	6.0	12
2	6.0	25
3	2.0	25
4	0.5	25
5	2.0	12
6	0.5	12

Killifish and clams were acclimated to experimental conditions for at least 10 days before each exposure. Tissue samples were taken when the animals were received at WES to determine any background residues of contaminants. Before the beginning of each run, environmental parameters were checked to verify that they were within the ranges needed. Once these were found to be acceptable, approximately 25 fish and 30 clams were placed in each of the 24 aquaria and Day 0 tissue, culture water and slurry samples were taken. During the next 7 days, the animals were not fed, but were checked daily and any dead ones removed. Total suspended solids were determined gravimetrically to verify the levels in each aquarium and unfiltered water samples were taken from each aquarium during the exposure. On Day 7 the animals were removed from each aquarium and allowed to depurate for 24 hours in clean flowing water at the same salinity as the run. The clams were then shucked and tissues of both clams and fish were frozen in separate glass jars by aquarium.

At the end of each exposure, bedded sediment was removed and stirred, and the aquaria were cleaned and refilled with water. The bedded sediment was placed back into each aquarium in preparation for the next run.

Water and tissue samples were analyzed for mercury using the cold vapor atomic absorption technique (American Public Health Association 1985) except for sample preparation. Water samples were prepared by continuous stirring while two 100-mℓ aliquots were removed. The first subsample was filtered through a 0.45-μm membrane filter while the second subsample was unfiltered. The subsamples were transferred to biological oxygen demand (BOD) bottles and analyzed by cold vapor atomic absorption for total mercury content. Tissue samples were prepared after thawing and homogenizing. Weighed subsamples were placed into digestion tubes, treated with nitric acid, and heated to 125° C until all tissue was dissolved (Evans, Johnson, and Leah 1986). The resulting solutions were evaporated to approximately 1.5 mℓ and diluted with distilled water to a known volume. Each subsample was transferred to a BOD bottle and analyzed by cold vapor atomic absorption. Appropriate US Environmental Protection Agency (USEPA) water and tissue quality control samples were run to verify proper functioning of equipment and procedures.

Data were analyzed using the SAS General Linear Models (GLM), Regression (REG), and MEANS procedures (SAS 1985). Values below detection limits were set equal to the detection limits for inclusion in analyses. Prior to analysis of variance (anova) or analysis of covariance (ancova), the assumption of homogeneity of variances was tested using Levene's test (Brown and Forsythe 1974), and a data transformation or nonparametric procedure employed if needed. Analyses of covariance also included a test of the ancova assumption of parallelism. Following significant anovas, means were compared using Duncan's multiple range test (two means), the Waller-Duncan k-ratio t-test (three or more means), or orthogonal contrasts (preplanned comparisons).

Functional regression equations were determined using geometric mean regression analysis when the independent variable could not be specified without error (Halfon 1985, Ricker 1984).

Results

Temperature and salinity measured in the aquaria during the six runs (Table 2) were close to the predefined experimental conditions listed in Table 1. Dissolved oxygen (DO) and pH were not controlled during the experiment but remained stable throughout all runs. DO was 9 to 10 mg/ℓ and pH was approximately 8 in all runs (Table 2).

Table 2

Mean Measured Physical Parameters for Each of the Six Runs

Physical Parameter	Run 1	Run 2	Run 3	Run 4	Run 5	Run 6
Temperature, °C	11.22 (0.80)	25.60 (0.80)	24.82 (0.46)	24.44 (0.37)	10.95 (0.60)	12.13 (1.06)
Salinity, ppt	6.0 (0.26)	6.6 (0.11)	2.4 (0.27)	1.0 (0.03)	2.1 (0.30)	1.0 (0.00)
Dissolved oxygen, mg/ℓ	9.37 (0.13)	9.50 (0.13)	9.60 (0.48)	9.62 (0.27)	9.56 (0.45)	10.03 (0.55)
pH	8.27 (0.06)	8.33 (0.13)	7.97 (0.07)	7.84 (0.06)	7.93 (0.05)	7.87 (0.10)
Total Suspended Sediment (TSS), 5 mg/ℓ	13.3 (6.1)	13.4 (9.9)	4.7 (2.5)	8.2 (3.2)	13.2 (7.5)	14.7 (6.0)
TSS, 10 mg/ℓ	18.1 (7.5)	14.8 (7.3)	8.3 (1.9)	17.8 (8.8)	18.1 (5.9)	18.8 (8.6)
TSS, 15 mg/ℓ	32.5 (9.0)	27.3 (8.6)	20.4 (4.8)	19.8 (5.3)	29.0 (9.8)	29.3 (6.4)
TSS, 25 mg/ℓ	30.7 (12.1)	35.8 (15.1)	25.7 (3.1)	25.3 (7.9)	30.8 (10.9)	27.8 (4.3)
TSS, 50 mg/ℓ	64.3 (22.9)	64.8 (34.9)	90.8 (41.3)	50.5 (7.4)	57.7 (8.0)	58.3 (2.5)
TSS, 100 mg/ℓ	127.9 (39.1)	110.2 (44.0)	91.8 (38.8)	120.0 (45.6)	121.8 (46.9)	119.8 (44.6)

Note: Standard deviations are given in parentheses. TSS values are listed by treatments in order of increasing nominal values.

Gravimetrically determined total suspended sediment (TSS) values did not reflect exactly the nominal suspended sediment levels assigned to each treatment, but generally did increase in a corresponding manner with the nominal levels (Table 2). Likewise, mercury concentrations in whole (unfiltered) water increased with increasing TSS. However, soluble mercury (in filtered water) was below or near detection limits regardless of TSS. Regression equations relating these parameters are given in Table 3.

Table 3

Regression Equations Relating TSS with Nominal Suspended Sediment Levels (NomSS) and Mercury Concentrations in Whole Water (HgWhole) and Filtered Water (HgSol)

Equation	No. of Samples	Anova Statistic	Probability, P	Adjusted Coefficient of Determination, R^2 percent
TSS = 3.616 + 1.138 NomSS*	140	1,499.389	0.0001	91.51
HgWhole = -0.744 + 0.313 TSS**	140	597.249	0.0001	81.09
HgSol = 0.273 + 0.005 TSS**	140	0.038	0.8453	0

* Linear least-squares regression.
** Geometric mean regression.

Clams exposed to mercury-contaminated suspended sediment accumulated significant amounts of mercury during all of the 7-day runs compared to clams that were not exposed to suspended sediment. However, bioaccumulation of mercury from suspended sediment by killifish was significant only in Run 2. In several other runs (3, 4, and 6), fish exposed to mercury-contaminated suspended sediment appeared to have lower concentrations of mercury than fish not exposed to suspended sediment. Orthogonal contrast statistics for these comparisons, mean bioaccumulation, and standard error of the mean for fish and clams are given in Table 4. In all runs, mercury levels were significantly higher in clams than in fish.

In all runs combined, clams exhibited a significant linear increase in mercury concentration with increasing amounts of TSS, and likewise with whole water mercury (HgWhole); whereas, fish did not. Regression equations relating mercury in clams (HgClam) and in fish (HgFish) with TSS and HgWhole for all

Table 4

Comparison of Mercury Bioaccumulation in Animals Exposed to Mercury-Contaminated Suspended Sediment with That of Animals Not Exposed to Mercury-Contaminated Suspended Sediment

Organism	Run	Anova Statistic	Probability, P	Mercury Tissue Concentration Mean (Standard Error, No. of Samples) ppm: No Suspended Sediment Exposure	Suspended Sediment Exposure
Clams	1	25.09	0.0001**	0.145 (0.0134, 12)	0.199 (0.0077, 18)
	2	39.73	0.0001**	0.153 (0.0071, 12)	0.209 (0.0059, 18)
	3	32.40	0.0001**	0.133 (0.0072, 12)	0.195 (0.0089, 18)
	4	27.97	0.0001**	0.132 (0.0071, 11)	0.208 (0.0097, 17)
	5	53.56	0.0001**	0.107 (0.0126, 12)	0.208 (0.0089, 18)
	6	212.19	0.0001**	0.083 (0.0066, 12)	0.189 (0.0058, 18)
Fish	1	1.64	0.2145NS	<0.026 (0.0028, 12)+	<0.035 (0.0057, 18)+
	2	15.43	0.0008**	<0.024 (0.0014, 12)+	<0.038 (0.0026, 18)+
	3	2.36	0.1405NS	0.053 (0.0033, 12)	0.046 (0.0027, 18)
	4	0.95	0.3419NS	<0.011 (0.0013, 9)+	<0.010 (0.0003, 18)+
	5	0.76	0.3942NS	<0.012 (0.0009, 12)+	<0.014 (0.0024, 18)+
	6	2.85	0.1067NS	<0.016 (0.0043, 12)+	<0.010 (0.0000, 18)+

Note: In the Probability column, NS indicates not significant at $P > 0.05$ and a double asterisk indicates highly significant at $P < 0.01$.

+ Means include values below detection limits that were set equal to the detection limits for inclusion in the data analysis.

runs combined are given in Table 5. Based on the regressions, clam tissue residues of mercury increase by about 1 part per billion (ppb) for each increase in TSS of 1 mg/ℓ (ppm), or by about 4 ppb for each increase in HgWhole of 1 ppb. However, changes in TSS or whole water mercury levels accounted for only 12 to 13 percent of the variation in clam tissue residues of mercury after the 7-day exposures, as evidenced by the adjusted coefficient of determination (R^2) values.

To assess the effects of temperature and salinity on mercury uptake in clams and fish, ancovas were run comparing bioaccumulation at the two temperatures (12° and 25° C) and at the three salinities (0.5, 2, and 6 ppt). TSS and HgWhole were each used as covariates in order to statistically remove any variation in bioaccumulation due to variation in these parameters. Any significant variation in bioaccumulation that remains can then be attributed to

Table 5

Geometric Mean Regression Equations Relating Mercury in Clams (HgClam) and Fish (HgFish) with TSS and with Mercury Concentrations in Whole Water (HgWhole)

Equation	No. of Samples	Anova Statistic	Probability, P	Adjusted Coefficient of Determination R^2 percent
HgClam = 0.146 + 0.00117 TSS	139	21.518	0.0001	12.94
HgClam = 0.145 + 0.00367 HgWhole	139	19.125	0.0001	11.61
HgFish = 0.009 + 0.00047 TSS	139	1.014	0.3157	0.01
HgFish = 0.009 + 0.0015 HgWhole	139	0.050	0.8238	0

the environmental factors of interest, temperature or salinity.

After statistical adjustment for the covariates, differences in bioaccumulation between the two temperatures were not significant for either fish or clams. Both organisms experienced slightly increasing mercury uptake with increasing salinity, but a significant difference was noted only for clams after adjusting for HgWhole as a covariate. In this case, clams exhibited significantly higher mercury concentrations at 6 ppt than at 0.5 ppt salinity. The mean tissue concentrations of mercury (not adjusted for covariates) in clams and fish following exposure to the various nominal TSS levels are shown in Figure 1 for the two temperatures, and in Figure 2 for the three salinities. Differences between the two organisms are far more apparent than any differences due to temperature, salinity, or TSS.

Discussion and Conclusions

Mercury uptake by killifish was clearly not influenced by temperature, salinity, or concentration of mercury-contaminated suspended sediment in this experiment. The fish simply did not bioaccumulate mercury under the conditions of exposure. It would appear that the sediment-associated mercury was not bioavailable to these estuarine fish under the experimental conditions. These results are consistent with those of Weis, Weis, and Bogden (1986), who

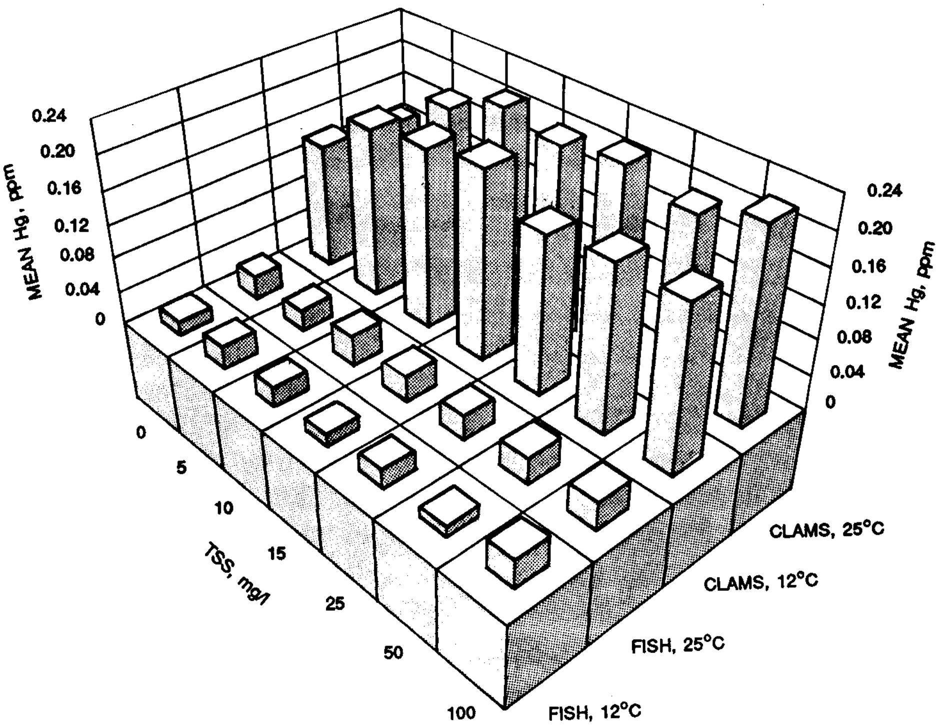

Figure 1. Mean mercury concentrations (ppm) in tissues of fish and clams exposed to the seven TSS treatments (0, 5, 10, 15, 25, 50, and 100 mg/ℓ) at two temperatures (12° and 25° C)

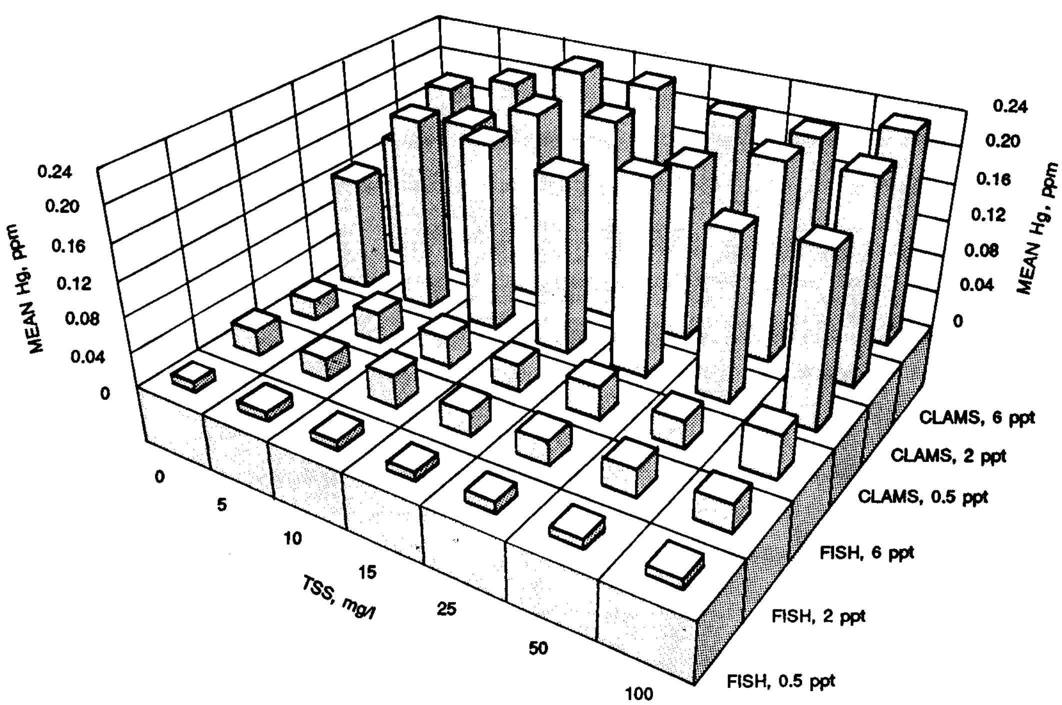

Figure 2. Mean mercury concentrations (ppm) in tissues of fish and clams exposed to the seven TSS treatments (0, 5, 10, 15, 25, 50, and 100 mg/ℓ) at three salinities (0.5, 2, and 6)

found no relationship between mercury uptake in killifish and salinity or mercury concentration in sediment to which the fish were exposed.

Clams, on the other hand, were able to accumulate mercury in this experiment, and had consistently higher mercury tissue residues than the fish. Mercury bioaccumulation in the clams appeared to be slightly enhanced by increasing salinity and increasing concentrations of mercury-contaminated suspended sediment. However, mercury tissue residues were not significantly enhanced at the higher temperature compared to the lower temperature.

Mercury content of the sediment from which the suspended particulate slurries were prepared ranged from about 80 to 100 ppm. This was two to three orders of magnitude greater than the mercury concentrations in tissues of animals exposed to those slurries. Clearly even the clams did not bioaccumulate mercury to any great extent in this experiment. The absence of detectable mercury in filtered water samples suggests that the mercury remained tightly sorbed to the suspended sediment. Binding of mercury in the organic fraction of the sediment could thus contribute to its lack of bioavailability, especially since total sediment organic carbon was quite high, in the range of 10-11 percent. The short duration of exposure (7 days) and continuous water exchange in this experiment may have also contributed to the lack of mercury uptake by organisms.

Preliminary data indicate that sulfide levels in the sediment were very high, around 20,000 ppm. However, the high sulfide levels probably had little influence on mercury bioavailability in this experiment because sulfide is rapidly oxidized in aqueous systems in the presence of dissolved oxygen and suspended sediment.* Sulfides would likely interact with mercury to form insoluble HgS only under anaerobic conditions.

In summary, temperature and salinity had little or no impact on uptake of mercury by estuarine fish and clams in the experiment described herein. Bioaccumulation of mercury by the clams appeared to be enhanced by increasing suspended sediment concentrations, but was still extremely low considering the high mercury content of the sediment. Mercury bioavailability may have been severely limited by the high sediment organic carbon content, if the mercury remained tightly bound in the organic fraction of the suspended sediment.

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* Personal communication, Dr. James Brannon, Environmental Laboratory, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

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EEDP-01-15
May 1989

Environmental Effects of Dredging Technical Notes

BIOACCUMULATION OF CHLORINATED CONTAMINANTS AND CONCOMITANT SUBLETHAL EFFECTS IN MARINE ANIMALS: AN ASSESSMENT OF THE CURRENT LITERATURE

PURPOSE: This note focuses on studies evaluating the sublethal effects of chlorinated organic contaminants on marine and estuarine organisms. Its objective is twofold: (1) to survey the literature for papers reporting both the sublethal effects of organohalogens and the corresponding body burdens in marine fish and invertebrates and (2) to provide a source of information for Corps field elements who have site-specific concerns (e.g., reproductive effects in a particular organism exposed to a specific organohalogen).

BACKGROUND: The US Army Corps of Engineers has the responsibility to ensure that contaminated sediments are dredged and disposed of in a manner that will not have an unacceptable adverse impact on the environment. The aquatic disposal of dredged material is regulated under two Federal statutes: Section 404(b)(1) of the Federal Water Pollution Control Act, as amended (PL 92-500) and Section 103 of the Marine Protection, Research, and Sanctuaries Act, as amended (PL 92-532). The regulations implementing these laws often require an evaluation of sediment toxicity and bioaccumulation potential prior to dredging and aquatic disposal.

Approximately 370 million cu m of sediments are dredged every year in the United States (Engler 1980). Approximately half of that volume is placed in open water. In most instances, dredged material is not acutely toxic to aquatic organisms. Therefore, decision-makers have had to rely less on toxicity data and more heavily on the results of bioaccumulation tests to evaluate potential impacts on the environment. There is very little interpretive guidance to assist in this evaluation (Peddicord and Hansen 1983). This report, produced under Work Unit 31773, Environmental Interpretation of Consequences from Bioaccumulation, of the Long-term Effects of Dredging Operations (LEDO) Program was designed, in part, to help provide that interpretive guidance.

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Approach

A literature search was performed for information on the sublethal effects of organohalogenated contaminants on marine and estuarine animals. Only those investigations which examined organismic endpoints (growth, reproduction, behavior, morphology, osmoregulation, and metabolism) were considered. The reasons for evaluating organismic sublethal endpoints are discussed in Dillon (1984). The scope of this literature review was large. More than 50 technical journals were individually reviewed (Table 1). Ten data base literature search services were also used to identify any additional papers (Table 1).

For every paper included in this review, the following information was recorded: contaminant, test animal, exposure time, contaminant exposure concentration, reported tissue concentration, and any observed biological effects. The test animal was identified by common name and/or phylogenetic group. Tissue concentrations were expressed on a wet-weight basis. Exposure concentrations were all reported as micrograms per litre (parts per billion) unless noted otherwise.

Analysis

Approximately 1,200 published papers reporting the sublethal effects of chlorinated contaminants on marine and estuarine animals were identified in the literature. Of these, only 37 papers (3 percent) contained both sublethal effects data and contaminant tissue concentrations (Table 2).

Growth and behavior were the most frequently examined sublethal endpoints, while metabolism and osmoregulation were the least examined. Effects on reproduction and morphology appeared to be intermediate choices. The test organisms used by most investigators were fish and arthropods. They appeared in 51 percent and 36 percent of the papers, respectively. The environmental contaminants most frequently tested were kepone (30 percent) and polychlorinated biphenyls (PCBs) (24 percent). Exposure to contaminants was mainly via aqueous solutions (73 percent) or food (27 percent). None of the residue-effects papers involved contaminated sediment.

Because only 3 percent of the sublethal effects investigations considered published concomitant tissue residue information, the data base for establishing quantitative residue-effects relationships is very limited. Variations due to interspecific differences, exposure regimes, and analytical capabilities diminish the ability to generate quantitative contaminant-specific guidance. However, a very broad generalization can be made based on data contained in Table 2. For marine and estuarine organisms with whole body tissue residues of chlorinated organic contaminants at or near steady-state, the level of concern associated with potential adverse sublethal effects is:

LOW for tissue concentrations <0.1 μg/g wet weight
MEDIUM for tissue concentrations 0.1-1.0 μg/g wet weight
HIGH for tissue concentrations >1.0 μg/g wet weight

These are not discrete thresholds, pass-fail values, or numerical criteria. Rather they are heuristic and are the only general guidance the data will allow.

Discussion

This review and assessment of the literature has shown that few laboratory investigations (3 percent) report both sublethal effects of chlorinated organic contaminants and tissue residue data. In an earlier review which included biochemical and cellular endpoints as well as organismic responses (Dillon 1984), a similar frequency of residue-effects in the published literature was noted (6 percent). This paucity of residue-effects information hampers the ability to generate contaminant-specific guidance for interpreting results of bioaccumulation tests. This does not mean, however, that evaluative techniques are nonexistent. There are several.

It is often desirable to make relative comparisons among bioaccumulation data rather than to infer specific effects from absolute tissue concentrations. For example, bioassay data from a specific project sediment can be compared to a reference value. This reference value may be generated in the laboratory by exposing bioassay organisms to sediment collected at or near the aquatic disposal site. The resultant tissue concentration is then the comparative standard. Exposure to reference sediment is carried out concurrently with project sediment bioassays. A reference value may also emerge from a consensus process in which tissue concentrations, representing indigenous organisms in a spatially discrete area, are identified (e.g., New York District matrix values). In both instances, bioassay results are interpreted from the standpoint of "no further degradation." It is important to note that although statistically significant differences may be observed in the laboratory, they do not necessarily imply that unacceptable adverse impacts in the environment are imminent or even inevitable.

In addition to ecological effects, tissue residue data may be interpreted in terms of human health issues. This can be done directly when the bioassay organism (or appropriate surrogate) is one commonly ingested by man. Numerical guidance for assessing contaminated seafood has been developed by agencies such as the US Food and Drug Administration and the Australian National Health and Medical Research Council. A summary of these data can be found in Peddicord et al. (1986). Local guidance in the form of action levels for seafood may also be available from state officials and US Environmental Protection Agency (USEPA) Regional Offices. If the bioassay does not involve an organism normally consumed by man, human health impacts can still be evaluated, albeit in a more circuitous manner. This is done by examining the potential for trophic transfer in the marine food web. Transfer may include the phenomenon of biomagnification, but this process is not common for many contaminants when trophic levels are strictly aquatic (Kay 1984). Biomagnification can become very important quantitatively when the trophic transfer process exits the aquatic environment. An in-depth technical discussion of this subject can be found in Biddinger and Gloss (1984).

When interpreting bioassay results, one must assess not only individual contaminants but also the impacts of multiple contaminants within the same tissue matrix. The first step in this analysis is, "How many and how much?" This approach can be quite useful in initial evaluations. For example, one would be very concerned if 10 out of 12 compounds were taken up in substantial amounts. The concern would lessen if only a few contaminants were accumulated and/or the magnitude of uptake was small. If only 1 out of 12 was accumulated to levels just above control or reference concentrations, the level of concern would be lower still.

Once a significant potential for bioaccumulation is established, the toxicological importance of the different contaminants must be considered. The potential for unacceptable adverse effects is elevated when toxic contaminants such as mercury, cadmium, and PCBs are accumulated. In contrast, concern lessens if less toxic compounds such as phthalates are found in the tissues of biota. How does one gauge relative toxicity? One of the better sources of information is the numerous toxicity tests conducted by the USEPA as part of their program to develop Water Quality Criteria (USEPA 1980). Additional guidance for determining the toxicological importance of various environmental contaminants can be found in Peddicord et al. (1986).

One question often asked when reviewing tissue residue data is, "How do interactions among the contaminants (e.g., synergism or antagonism) affect the organism?" All interactions that may (or may not) be occurring are expressed in the acute toxicity data. Therefore, this question is somewhat irrelevant for sediment bioassays. To determine the interactive effects among specific contaminants of concern for a particular marine organism, additional laboratory experiments would have to be conducted.

Summary

A review of the literature has shown that about 3 percent of studies investigating the sublethal effects of chlorinated organic contaminants on marine organisms contain both effects and concomitant tissue residue data.

The residue-effects information that is available (Table 2) can be very useful for interpreting the results of project-specific bioassays. It is believed that they also represent heuristic guidance, not to be confused with pass-fail or threshold criteria. For marine and estuarine organisms with whole body tissue residues of chlorinated organic contaminants at or near steady-state, the level of concern associated with adverse sublethal effects is generally:

LOW for tissue concentrations <0.1 µg/g wet weight
MEDIUM for tissue concentrations 0.1-1.0 µg/g wet weight
HIGH for tissue concentrations >1.0 µg/g wet weight

Although the paucity of residue-effects information hampers contaminant-specific guidance, other evaluative techniques are available for interpreting the biological importance of bioaccumulation. For evaluating potential ecological effects, comparisons to reference values derived either in the laboratory or by consensus agreements can be carried out. To assess the potential for human health impacts, bioaccumulation results can be compared directly to previously developed numerical guidance for contaminated seafood. Human health effects can also be evaluated indirectly by examining trophic transfer potential of contaminants in the marine food web. Finally, tissue residue information can be evaluated by determining the number of contaminants showing mobility, the magnitude of uptake relative to control and/or reference values, and the toxicological importance of contaminants that are bioaccumulated.

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Table 1

List of Scientific Journals and Data Base Search Services Used To Identify Published Papers

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Journals

Aquatic Toxicology
Archiv fur Hydrobiologie
Archives of Environmental Contamination and Toxicology
Australian Journal of Biological Science
Australian Journal of Marine and Freshwater Research
Australian Journal of Zoology
Biological Bulletin
Bulletin of Environmental Contamination and Toxicology
California Fish and Game
Canadian Journal of Fisheries and Aquatic Sciences
Canadian Journal of Zoology
Chemosphere
Comparative Biochemistry and Physiology
Critical Reviews in Environmental Control
Crustaceana
Developmental Biology
Ecotoxicology and Environmental Safety
Environmental Biology of Fish
Environmental Pollution Series A, B, and C
Environmental Research
Environmental Science and Technology
Environmental Toxicology and Chemistry
Estuaries
Fisheries
Fisheries Bulletin, U.S.
Hydrobiologia
International Review of Invertebrate Reproduction and Development
International Revue der Gesamten Hydrobiologia
Journal Applied Ecology
Journal of Crustacean Biology
Journal of Experimental Biology
Journal of Experimental Zoology
Journal of Fish Biology
Journal of Invertebrate Pathology
Journal of Pesticide Science
Journal of Plankton Research
Journal of Toxicological and Environmental Research
Journal of Water Pollution Control Federation
Journal of Zoology
Limnology and Oceanography
Marine Environmental Research
New York Fish and Game
New Zealand Journal of Marine and Freshwater Research
Oecologia
Oikos

(Continued)

Table 1 (Concluded)

Journals (Concluded)

Pesticide Biochemistry and Physiology
Pesticides Science
Physiological Zoology
The Progressive Fish Culturist
Quarterly Review of Biology
Science
Science of the Total Environment
Transactions, American Fisheries Society
US Environmental Protection Agency's Ecological Research Series
Water, Air and Soil Pollution
Water Pollution, Research and Control
Water Quality International
Water Research
Water Resources Research

Computerized Data Base Searches Services

Biosis
Water Resources Abstract
Aquatic Sciences and Fisheries Abstract
Chemical Abstracts
Life Sciences Collection
Zoological Record
NTIS (National Technical Information Service)
Dissertation Abstracts
Conference Papers Index
Pollution Abstracts

Table 2

Summary of Published Papers on the Biological Effects of Chlorinated Environmental Contaminants on Marine Organisms and Associated Tissue Residues

Parameter	Contaminant	Organism	Exposure Time	Exposure Concentration*	Tissue Concentration**	Biological Effect	Reference
Growth	Kepone	Sheepshead minnow	36 days	0.08-6.60	1-22	Growth inversely related to concentration	Hansen, Goodman, and Wilson 1977
Growth	Kepone	Sheepshead minnow	160 days	0-0.12	0-0.86	No effect on growth	Goodman et al. 1982
				0.39-0.78	1.1-5.0	Reduction in growth	
Growth	Kepone	Blue crab	28 days	0.15 μg/g food	0.069	Reduction in growth	Schimmel et al. 1979
Growth	Kepone	Blue crab	65 days	0.36-2.5 μg/g food	0.38-4.16	No effect on growth, ratio of carapace thickness to width inversely proportional to concentration	Fisher, Bender, and Roberts 1983
Growth	Methoxychlor	Crab	10 days	0.7	0.51	Reduction in growth	Bookout, Costlow, and Monroe 1976
Growth	Endrin	Sheepshead minnows	22 weeks	0-1.31	0.94	No effect on growth	Hansen, Schimmel, and Forester 1977
Behavior	Toxaphene	Killifish (embryo)	28 days	ND†-0.6	No data	Reduction in survival at all concentrations	Schimmel, Patrick, and Forester 1977
				1.3-6.5	No data	Erratic swimming, loss of equilibrium	
		Killifish (fry)	28 days	ND-0.6	ND-8.0	Reduction in survival at all concentrations	
				1.3-6.5	34-no data	Reduction in survival at all behavior, loss of equilibrium	
		Killifish (juvenile)	28 days	ND-0.8	ND-24.7	Reduction in survival at all concentrations	
				1.7-3.4	102-no data	Erratic swimming, loss of equilibrium	

(Continued)

* Exposure concentrations are expressed in units of micrograms per litre (μg/ℓ) unless noted otherwise.
** Tissue concentrations are expressed in units of micrograms per gram (μg/g) wet weight whole animals unless noted otherwise.
† ND - Nondetectable, <0.2 μg/ℓ in water, <0.2 μg/g in tissue.

(Sheet 1 of 5)

Table 2 (Continued)

Parameter	Contaminant	Organism	Exposure Time	Exposure Concentration	Tissue Concentration	Biological Effect	Reference
		Killifish (adults)	28 days	ND-0.9	ND-6.1	Reduction in survival	
				1.7-3.8	No data	Erratic swimming, loss of equilibrium	
Behavior	Kepone	Sheepshead minnow	28 days	0-1.9	0.26-11	Erratic swimming behavior and a reduction in feeding rate both increased as concentrations increased	Hansen, Goodman, and Wilson 1977
Behavior	Kepone	Blue crab	65 days	0.36-1.64 μg/g food	0.38-1.73	No effect on behavior	Fisher, Bender, and Roberts 1983
				2.26-2.50 μg/g food	2.54-4.61	Excitable behavior during feeding, reduced ability to locate and consume food	
Behavior	Methoxychlor	Crab	15 days	1.8-32	0.11-1.59	Hyperactivity, inability to maintain an upright position, difficulty in locating and coinsuming food	Armstrong et al. 1976
Behavior	Mirex	Grass shrimp	14-16 days	0.011-0.130	0.02-0.20	Diminished ability to avoid predation	Tagatz 1976
Behavior	Mirex	Oyster	10 weeks	0.038	1.3-28	Diminished ability to withstand predation	Tagatz et al. 1976
		Mussel	10 weeks	0.038	1.6-2.0	Diminished ability to withstand predation	
Behavior	PCB DDT	Fish	Field collected	No data	110 7	Decreased ability to maintain position while swimming in a current	Olofsson and Lindahl 1979
Reproduction	Kepone	Sheepshead minnow	90-133 days	0.041-0.074	0.15-0.56	Increased number of eggs/female/day; fertility unaffected	Goodman et al. 1982
				0.12-0.39	0.86-3.0	Number of eggs/female/day unaffected; fertility unaffected	
				0.78	5.0-6.8	Decreased number of eggs/female/day, reduced fertility	
Reproduction	Kepone	Sheepshead minnow	28 days	0.05-0.80	0.26-4.7	Production of normal embryos	Hansen, Goodman, and Wilson 1977
				1.9	11	Production of abnormal embryos	

(Continued)

(Sheet 2 of 5)

Table 2 (Continued)

Parameter	Contaminant	Organism	Exposure Time	Exposure Concentration	Tissue Concentration	Biological Effect	Reference
Reproduction	PCB (Aroclor 1254)	Cod	5-1/2 months	1-50 μg/g wet food	0.06-5.3 (testes)	Disruption in production of sex steroids from testes	Freeman, Sangalang, and Flemming 1982
Reproduction	PCB (Aroclor 1254)	Sheepshead minnow	4 weeks	0-10.0	0.52-170	No effect on number of eggs fertilized	Hansen, Schimmel, and Forester 1973
Reproduction	PCB (Aroclor 1016)	Sheepshead minnow	29 days	1-10	5.4-110 (adults) 4.2-66 (eggs)	No effect on egg fertility, hatching, or subsequent survival of progeny	Hansen, Schimmel, and Forester 1975
				32	200-1,100 (adults)	100 percent mortality in adults	
Reproduction	Endrin	Sheepshead minnow	23 weeks, 1 generation	0.027-0.12	0.20-1.0 (adults) 0.09-0.87 (eggs)	No effect on reproduction	Hansen, Schimmel, and Forester 1977
				0.31	0.94 (adults) 1.80 (eggs)	Reducted fertilization and early hatching, high mortalities	
				0.72	No data		
Reproduction	PCB	Flounder	Field collected	No data	5.0-317 ng/g (ovaries)	Reduced viable hatch at PCB tissue concentrations above 120 ng/g	Von Westernhagen et al. 1981
	DDD				3.0-30.3 ng/g (ovaries)	Hatch viability not correlated with tissue concentration of any other contaminant	
	DDE				0.1-62.0 (ovaries)		
	Hexachlorobenzene				0.06-2.0 (ovaries)		
	Dieldrin				0.1-49.0 (ovaries)		
	Heptachlorepoxide				0.08-3.0 (ovaries)		
Reproduction	PCB	Fish	Field collected	No data	19-241 ng/g (ovaries)	Reduced viable hatch at PCB tissue concentrations above 18 ng/g	Hansen, Von Westernhagen, and Rosenthal 1985
	DDD				<1-16.0 (ovaries)	Hatch viability not correlated with tissue concentration of any other contaminant	
	DDE				<1-34.0 (ovaries)		
	Dieldrin				<1-8.1 (ovaries)		
	Hexachlorobenzene				<1-8.6 (ovaries)		
	Heptachlorepoxide				<1-8.9 (ovaries)		

(Continued)

(Sheet 3 of 5)

Table 2 (Continued)

Parameter	Contaminant	Organism	Exposure Time	Exposure Concentration	Tissue Concentration	Biological Effect	Reference
	α Hexachlorocyclohexane				<1-9.2 (ovaries)		
	γ Hexachlorocyclohexane				<1-12.1 (ovaries)		
Morphology	PCB (Aroclor 1254)	Cod	5-1/2 months	1-50 μg/g wet food	0.04-2.1 (kidney)	Disruption in production of adrenal hormones from kidney	Freeman, Sangalang, and Flemming 1982
					0.02-0.98 (muscle)	No effect on histopathology of kidney	
					10.1-374 (liver)	Degeneration of liver's fatty tissue	
Morphology	PCB (Aroclor 1254)	Cod	5-1/2 months	1-50 μg/g food	0.06-5.3 (testes)	Response intensified as tissue concentration increased and progressed from testicular fibrosis to inhibition of spermatogenesis and finally to complete disintegration of the testes	Sangalang, Freeman, and Crowell 1981
Morphology	PCB (Aroclor 1254)	Cod	5-1/2 months	1-50 μg/g wet food	0.02-0.98 (muscle)	Hyperplasia of gills with disrupted blood spaces	Freeman, Sangalang, and Flemming 1982
Morphology	PCB (Aroclor 1254)	Shrimp	35 days	0.6-0.7	2 (muscle) 21 (hepatopancreas)	Increased occurrence of viral pathogen	Couch and Courtney 1977
Morphology	Kepone	Crab	65 days	0.36-2.50 μg/g food	0.38-4.61	Carapace thickness-to-width ratio inversely related to concentration	Fisher, Bender, and Roberts 1983
Morphology	Kepone	Killifish	28 days	0-1.9	0.26-11	Response intensified as tissue concentration increased. Response progressed from deformed vertebral column, hemorrhaging near brain, and darkened posterior to increased hemorrhaging and fin rot	Hansen, Goodman, and Wilson 1977
Morphology	Dieldrin	Oyster	43 days	1-100	25.6-2,685*	No effect on fibrous or cellular components of gills, gut, or mantle, no inflammation or infiltration of leucocytes	Emanuelsen, Lincer, and Rifkin 1978

(Continued)

* Data originally reported on a dry-weight basis were converted to wet weight assuming 80 percent body water.

(Sheet 4 of 5)

Table 2 (Concluded)

Parameter	Contaminant	Organism	Exposure		Tissue	Biological Effect	Reference
			Time	Concentration	Concentration		
Osmoregulation	Pentachlorophenol	Fish	5 days	100	37.1	Reduction in total osmotic pressure	Thomas, Carr, and Neff 1981
Osmoregulation	Methoxychlor	Crab	7 days	10	0.31 2.0 (gill)	No effect on total osmotic pressure	Caldwell 1974
		Crab	14 days	10	1.0 2.5 (gill)	No effect on total osmotic pressure, sodium or potassium regulation but magnesium regulation was disrupted	
Osmoregulation	DDT	Crab	50 hr	Single injection of 100 μg/kg	0.06 (gill) 1.5 (hepatopancreas)	Sodium and potassium regulation in the gill disrupted	Neufeld and Pritchard 1979
Metabolism	PCB (Aroclor 1016)	Horseshoe crab	96 days	0.35-71.5	0.08-92.8	No ecologically significant change in oxygen consumption	Neff and Giam 1977
Metabolism	Halowax 1099 (chlorinated naphthalene)	Horseshoe crab	96 days	22-70	0.51-5.7	Highly variable oxygen consumption	Neff and Giam 1977
Metabolism	Kepone	Blue crab	65 days	0.36-1.64 μg/g food	0.38-1.73	No effect on oxygen consumption	Fisher, Bender, and Roberts 1983
				2.26-2.50 μg/g food	2.54	Elevated rates of oxygen consumption	

(Sheet 5 of 5)

EEDP-01-16
May 1989

Environmental Effects of Dredging Technical Notes

SEASONAL RESTRICTIONS ON DREDGING OPERATIONS IN FRESHWATER SYSTEMS

PURPOSE: This note summarizes the status of seasonal restrictions on dredging operations in freshwater navigable waterways. The information presented is based on replies received from a questionnaire sent to all US Army Corps of Engineers (CE) District and Division offices that conduct O&M dredging operations in freshwater systems.

BACKGROUND: Restrictions on dredging operations are used to protect various types of aquatic resources. CE Districts are often required to restrict or suspend dredging operations during a defined period of time to prevent real or perceived detrimental impacts on important species of invertebrates, fish, and birds. The magnitudes of these potential impacts are often speculative or not technically supportable, but are imposed nonetheless until new information becomes available.

Restrictions have historically been placed on dredging operations occurring in coastal systems. However, as dredging of freshwater navigable waterways increases, resource management agencies are imposing new restrictions resulting in contractual delays, increased project costs, and other complications in maintaining a navigable channel throughout the year. To ensure that valuable aquatic resources are adequately protected and prevent unwarranted delays in dredging, a more complete understanding of the scope and nature of seasonal restrictions is needed to assist Corps offices in planning and implementing dredging operations.

ADDITIONAL INFORMATION: Contact one of the authors--Mr. Larry Sanders, (601) 634-2976, or Mr. Jack Killgore, (601) 634-3397--or the Environmental Effects of Dredging Programs (EEDP) Manager, Dr. Robert M. Engler, (601) 634-3624.

Introduction

Resource management agencies often restrict the time and location of CE dredging/disposal operations to minimize potential impacts to important aquatic resources. These restrictions are a major concern to the Corps because they create an added impact on the cost and scheduling of dredging operations. In many cases, the imposition of restrictions is based on limited information relating to the behavior and survival of species such as fish, birds, and mussels associated with dredging operations. However, since protection of important aquatic resources is an important issue in any navigation-related activity, seasonal restrictions will continue unless perceived impacts are determined to be unwarranted.

A recent survey of coastal and Great Lakes CE Districts indicated that dredging can be delayed, or even cancelled because of the potential effects of elevated suspended sediment concentrations on fish survival, turbidity plumes on the behavior of migrating fishes, dissolved oxygen reduction on aquatic species survival, and physical disturbance of spawning and feeding grounds (LaSalle et al., in preparation). In order to obtain a more complete understanding of the scope and nature of seasonal restrictions throughout the United States, a survey of inland waterway CE District and Division offices was also conducted in the Fall of 1988 and the results are presented herein.

Twenty-six individuals, representing 29 District and Division offices, were asked to provide the following information on existing restrictions: (1) the subject of the restriction, (2) the specific reason(s) for the restriction, (3) the project type and specific activities of concern, (4) the dates of restriction, and (5) the agency suggesting the restriction. In addition there was a section for general comments.

Survey Results

Approximately 70 percent of the people asked to participate in the survey responded. Table 1 identifies those District and Division offices that responded to the survey. In most cases, similar restrictions applied to both coastal and inland waterway CE offices, although the species of concern varied by geographical region. Six Districts reported no restrictions because of limited dredging requirements, while other Districts indicated extensive restrictions usually due to the presence of commercially valuable or threatened species of fish.

Subject of restrictions. The most common subject of seasonal restrictions was related to either individual or groups of sport and anadromous fishes (Table 2). Other topics included endangered species (sturgeon, mussels), water quality, migratory waterfowl, and nesting birds. The highest number of individual species being protected through seasonal restrictions was reported by the Pacific Ocean Division (11) followed by the Walla Walla District (5).

Reason for restriction. The primary reason given for seasonal restrictions was the potential impacts on fish and their habitat (Table 3). Those issues of major concern were loss of habitat; disruption of spawning for certain species; entrainment of fish eggs and larvae; high mortality of eggs and larvae due to smothering and clogging of gills caused by suspended sediment; change in functional utilization of habitat for feeding, cover, and overwintering; and potential blockage of migratory pathways of various anadromous species due to their reluctance to pass through turbidity plumes. Other reasons listed included noise impairment on migratory birds, potential degradation of water quality (primarily turbidity and dissolved oxygen (DO) reduction), and concern regarding contaminant release.

Project type or activity of concern. Maintenance dredging was the most common operation affected by seasonal restrictions (Table 4). Other operations included channel improvement, bank reshaping, commercial sand and gravel dredging, and hopper dredge overflow. Disposal operations listed were in-water, overboard, and upland. An increase in barge/scow travel to approved disposal sites was also mentioned which would directly increase the cost of disposal operations. Restrictions often applied to dredging projects regulated by the CE under Section 10 of the Rivers and Harbor Act and Section 404 of the Clean Water Act.

Dates of restrictions. Restrictions usually occurred in the spring and early summer. As a result, most dredging took place during the winter when most species of concern have migrated out of the area or were not involved in spawning or rearing in the vicinity of the dredging or disposal operations. In some cases, Districts are required to monitor water quality and other potential physical impacts during dredging operations to ensure there are no detrimental impacts to existing resources. If any significant changes occurred, then dredging may be suspended by the resource agency.

Agencies suggesting the restrictions. Restrictions were placed upon CE dredging operations by one or more state resource agencies. In order that Federal activities are consistent with approved state management programs, the CE complies with restrictions usually through memorandums of agreement pending further evaluation of potential impacts. Federal agencies such as the US Fish and Wildlife Service, the US Environmental Protection Agency, and the National

Marine Fisheries Service also took part in the negotiations, particularly if endangered species or commercial fish stocks were involved.

Discussion of Survey Results

The nature of seasonal restrictions imposed on CE dredging operations is similar in both coastal and inland waterway districts. Most restrictions are related to activities which may have a potential negative impact on fish and their habitat, such as physical disruption of spawning sites or degradation of certain water quality parameters. A resource of particular concern is anadromous fish species, including salmon, striped bass, and shad.

There is often inadequate data to substantiate the validity of certain seasonal restrictions, but concern over potential impacts necessitates compliance with the resource agency's decisions. Direct impacts of dredging operations on aquatic resources, such as benthic burial or alteration of spawning sites, require that restrictions be placed on certain areas or during specific time periods. However, other issues of concern that are not well-defined or for which there is no direct evidence to support impact statements can be investigated through well designed field studies. For example, the question of turbidity plumes altering migratory pathways of anadromous fishes is a concern shared by many CE offices, but no direct evidence supports such a contention. As new information is obtained, the validity of certain restrictions should be reevaluated.

Reference

LaSalle, M. W., Homziak, J., Lunz, J. D., Clarke, D. G., and Fredette, T. J. "Seasonal Restrictions on Dredging and Disposal Operations," Technical Report (in preparation), US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Table 1

Corps of Engineers Divisions and District Offices Responding to Survey

Division/District	Abbreviation
Memphis	LMM
St. Louis	LMS
Kansas City	MRK
Omaha	MRO
Baltimore	NAB
New York	NAN
Norfolk	NAO
Philadelphia	NAP
Buffalo	NCB
Detroit	NCE
Walla Walla	NPW
Ohio River	ORD
Huntington	ORH
Nashville	ORN
Pittsburgh	ORP
Pacific Ocean	POD
Jacksonville	SAJ
Mobile	SAM
Savannah	SAS
Wilmington	SAW
Sacramento	SPK
Fort Worth	SWF
Galveston	SWG
Little Rock	SWL

Table 2

Summary on the Subject of Seasonal Restrictions

Subject of Restriction	CE Divisions/Districts
Sport fish	SPK, NAN, ORN, NCB, ORP, NCE, NPW, NAO, ORD, SAJ, SAM, NAP
Anadromous fish*	SPK, NAB, SAW, NPW, SAM, NAP, NAN, SAJ, NAO
No restrictions	SWG, LMS, SWF, MRO, LMM, MRK
Fisheries (general)	NCB, ORH, ORD, SAJ
Fish spawning (general)	SPK, ORN, ORP, NCE
Bird nesting	POD, NCE, SWL, SPK
Water quality	NAN, NCE, SAJ, SAS
Sturgeon	NAN, NPW, SAM, NAP
Recreational activities and/or aesthetics	SAJ, NCE
Mussels/snails	POD, SWL
Shrimp	SAW, POD
White bass	ORN
Goby's	POD

* Anadromous fish which were mentioned included striped bass (*Morone saxatilis*), Chinook salmon (*Oncorhynchus tshawytscha*), steelhead (*Salmo gairdneri*), blueback herring (*Alosa aestivalis*), and American shad (*Alosa sapidissima*).

Table 3

Reasons for Seasonal Restrictions

Topic	CE Divisions/Districts
Avoid degradation of recreational waters, fisheries, and aquatic vegetation	SPK, NAN, ORN, NCB, ORP, NAB, SAW, NCE, ORH, NPW, NAO, SAJ, SAM, POD, NAP, SAS
Degraded water quality (DO, temperature, turbidity)	NAN, NCB, NPW, SAJ, NAP, SAS, SPK, SPK, NCE, ORH, NAO
Turbidity influence on fish spawning and migration	SPK, NAN, ORN, NCB, NCE, ORH, NPW, NAO, SAJ, NAP
Loss or change in functional use of fish habitat	SPK, NAN, NCB, ORP, NCE, ORH, NPW, NAO, SWL
Physical disturbance of fish spawning habitat	SPK, NCE, ORH, NAO, NCB
Protect fish nursery habitat	NAN, NCB, SAW, NPW
Noise impairment of migratory bird nesting	SPK, SWL, POD
Entrainment	NPW, SAM, NAP
Release of contaminants	NCE, NPW
Preservation of shallow-water habitat	ORP
Benthic burial	SAS

Table 4

Project Type or Activity of Concern Affected by Seasonal Restrictions

Project Type	CE Divisions/Districts
Maintenance dredging	NAN, SAW, ORH, ORD, SAM, SAS
Dredging (general)	NCB, NAB, POD, NAP
Commercial sand and gravel dredging	ORP, SWL
Hopper dredge overflow	NCE, NPW
In-water construction	ORH, POD
Upland disposal	NAB, SAJ
Bank reshaping and toe trench excavation	SPK
Channel improvement and bendway removal	ORN
In-water disposal	NPW
Barge/scow travel to disposal sites	NAO
Hydraulic cutterhead dredging	NAO
Bucket dredging or overboard disposal	NAP
Blasting and fill	NAP

EEDP-01-17
July 1989

Environmental Effects of Dredging Technical Notes

FACTORS INFLUENCING BIOACCUMULATION OF SEDIMENT-ASSOCIATED CONTAMINANTS BY AQUATIC ORGANISMS; FACTORS RELATED TO CONTAMINANTS

PURPOSE: This is the first technical note in a series of four which outlines and describes the principal factors that determine uptake and retention of chemicals by aquatic organisms. The first three notes describe factors related to contaminants, sediment and water, and biota. The fourth note is a glossary and bibliography. The information contained herein is intended to assist Corps of Engineers environmental personnel in activities requiring a working knowledge of concepts and terminology in the subject of chemical uptake, retention, and elimination by aquatic organisms exposed to contaminated sediments.

BACKGROUND: Bioaccumulation is the general term used to refer to the uptake and storage of chemicals by organisms from their environment through all routes of entry. Bioaccumulation includes bioconcentration, which is the direct uptake of chemicals from water alone, and is distinguished from biomagnification, which is the increase in chemical residues taken up through two or more levels of a food chain. Assessments of the potential for bioaccumulation of toxic substances associated with dredged sediments are often required in evaluations of permit requests. Thus, familiarity with the fundamental physical, biological, and chemical factors affecting bioaccumulation is necessary for performing evaluations of the ecological impacts of dredging operations. Additionally, a basic understanding of the concepts and terminology of bioaccumulation is increasingly required of environmental personnel who are involved in dredging and disposal operations which may involve contaminated sediments and legal personnel involved with regulation and litigation.

These notes are intended to serve as a source of basic information and to provide a guide to the scientific literature for each topic discussed. The emphasis is on factors affecting bioaccumulation of sediment-associated chemicals. A brief discussion of each factor is given and a list of references is provided. The references are extensive and frequently bear on more than one topic. An effort has been made to select both historically important works and the most recent research reports in each area. Numbers in parentheses following the subject headings locate the references for each subject. Papers referenced are alphabetized for each subject for easy identification of those most pertinent to the reader's interest. The glossary of technical terminology is presented in the fourth note in the series.

The subjects discussed in these notes reflect current research for which new findings constantly appear in the literature. Consequently, the discussions and interpretations are based on inference and best judgement regarding the interactions of factors influencing bioaccumulation and represent the best understandings of the authors. Readers are encouraged to consult the literature cited.

ADDITIONAL INFORMATION: Contact the authors--Mr. Victor A. McFarland, (601) 634-3721; Mr. Charles H. Lutz, (601) 634-2489; or Mr. Francis J. Reilly, (601) 634-4148--or the manager of the Environmental Effects of Dredging Programs, Dr. Robert M. Engler, (601) 634-3624.

Fugacity (1-12)

If a chemical is introduced into a closed system consisting of two immiscible phases, e.g., an oil and water, it will distribute itself between the two phases until equilibrium is reached. At equilibrium there will continue to be an escape of individual molecules from the oil to the water, and vice versa, but the net exchange of chemical mass between the two phases will be equal to zero. This tendency of a chemical to escape from a phase is referred to as its "fugacity" (from the Latin fuga for "flight"). Fugacity is a corollary function of chemical potential and just as, by definition, equilibrium exists when the chemical potential in all phases of a closed system is equal, equilibrium also exists when the fugacities of a chemical in the phases are equal.

At the low concentrations typical of environmental contamination, fugacity and concentration can be directly related by a constant that quantifies the ability of a phase to contain the chemical. An example of this "containing ability" is the mass of a chemical that could be solubilized in a given volume of water under standard conditions. A different mass of the same chemical could be solubilized in an equal volume of an organic solvent, such as oil. That is, the containing ability of the oil would be more or less than the containing ability of the same volume of water for the chemical.

Partition coefficients express the concentration differential between two phases at equilibrium. This is the essence of the concept of "equilibrium partitioning" and is fundamental to understanding the processes of chemical bioavailability and bioaccumulation. Fugacity and equilibrium partitioning are thermodynamic concepts, i.e., they are independent of rates of change (kinetics) or of rate-influencing processes. Rate-influencing processes determine how long it takes to reach an equilibrium between chemical concentration in a source phase and in a sink phase, but not how much chemical will be in each when the equilibrium is reached. "Bioaccumulation potential," as the term is used throughout these notes, is intended in the thermodynamic sense. The bioaccumulation potential of a chemical is dependent on the fugacity of chemicals and the containing abilities of sediment and organism as the phases of concern. The actual quantity of chemical that may be bioaccumulated is influenced by a multitude of variables. The discussions that follow consider many of these rate-influencing variables as well as variables affecting bioaccumulation potential.

Hydrophobicity (13-18)

Literally, "fear of water," hydrophobicity is the property of neutral (uncharged or nonpolar) organic molecules that causes them to associate with surfaces or with organic solvents rather than to remain in aqueous solution. The presence of a neutral molecule causes the highly charged molecules of water in its vicinity to link up in what has been described as a "shaky cage" structure around the neutral molecule. This structuring of water is energetically unfavorable and the neutral molecule tends to seek a less energetic phase if one is available. Animal lipids, mineral surfaces, or associations of other neutral molecules are examples of phases that are less energetic than water. In an operational sense, hydrophobicity is the reverse of aqueous solubility.

The octanol/water partition coefficient of a chemical (K_{ow}, log K_{ow}, or log P) is a measure of its hydrophobicity. K_{ow} is a constant that describes the magnitude of the difference between the solubility of a chemical in water and its solubility in the organic solvent, octanol. Octanol serves as a good surrogate for animal lipids in the laboratory as organic chemicals are soluble in both to about the same extent. Because organic chemicals accumulate in the lipids of organisms, hydrophobicity measurements provide good indications of the tendency for organic chemicals to bioconcentrate and bioaccumulate. Bioconcentration factors (BCF) increase with increasing hydrophobicity up to a log $K_{ow} \cong 6.00$. At hydrophobicities greater than log $K_{ow} \cong 6.00$, BCFs tend to decrease.

Solubility (13, 19-24)

In general, as the water solubility of chemicals increases, bioaccumulation decreases. Water solubility favors rapid uptake of chemicals by organisms but at the same time favors rapid elimination. Any physical or chemical process that increases the water solubility of a chemical decreases the tendency for that chemical to bioaccumulate. Organic chemicals that form weak acids or bases and those that can be protonated (e.g., sulfonate and tertammonium surfactants) bioaccumulate to lower concentrations than do neutral organics not only because they are more reactive, but also because these processes make them soluble. Compounds such as chlorinated phenols are sometimes referred to as "hydrophobic acids" because the chlorinated benzene nucleus favors partitioning to lipid and other organic phases, while the phenolic oxygen confers aqueous solubility. Such compounds are also reactive and do not usually bioaccumulate to high levels. Organic compounds that do not dissociate (neutral or nonpolar organics) are increasingly insoluble as molecular mass increases and are the most highly bioaccumulating. Although the ionized forms of heavy metals such as mercury, cadmium, and lead are soluble in water, these substances bind with tissues and thus are actively bioaccumulated by organisms.

Stability (25-30)

For chemicals to bioaccumulate, they must be stable, conservative, and resistant to degradation. Metals are inherently conservative since they are elemental in nature. Metals are taken up by organisms either as ions in solution or as organometallic complexes. Complexation of metals may facilitate bioaccumulation by increasing bioavailability. Organometalloids that are taken up by organisms may hydrolyze, allowing the free metal ion to bond ionically or covalently with sulfhydryl, amino, purine, and other reactive groups present in endogenous substrates.

Organic compounds with structures that protect them from the action of enzymes or from nonenzymatic hydrolysis tend to bioaccumulate. However, chemicals such as the phosphate ester pesticides (e.g., parathion and malathion) do not bioaccumulate because they are easily hydrolyzed and the products eliminated. Polynuclear aromatic hydrocarbons (PAHs) are also easily broken down and eliminated by most fishes and many other organisms, although some invertebrates, such as bivalve mollusks and amphipods, have low metabolizing capability for these compounds. The presence of electron-withdrawing substituents on PAHs tends to stabilize these compounds. Chlorines, for example, are bulky, highly electronegative atoms that tend to protect the nucleus of an organic molecule, such as a PAH, against chemical attack. Highly chlorinated organic compounds such as the polychlorinated biphenyls (PCBs) bioaccumulate to high levels because they are easily taken up by organisms and cannot be readily broken down and eliminated.

Stereochemistry (14, 31-42)

The spatial configuration or shape, i.e., stereochemistry, of a neutral molecule affects its tendency to bioaccumulate. Molecules that are planar, such as PAHs, dioxins, or certain of the PCBs, tend to be more lipid soluble than globular molecules of similar molecular weight. For neutral organic molecules, planarity generally correlates with higher bioaccumulation unless the molecule is easily metabolized by an organism, as is the case, for example, with PAHs in most fishes.

Hydrophobicity and transport across biological membranes are affected by the size as well as shape of molecules. Hydrophobicity of neutral molecules generally increases with molecular mass, volume, or surface area. Neutral molecules that have cross-sectional dimensions greater than about 9.5 Å have been described as "sterically hindered" in their ability to penetrate the polar surfaces of the cell membranes in fish gut or gill tissue. The limited bioaccumulation of compounds such as octachlorodibenzo-p-dioxin (9.8 Å) or decabromobiphenyl (9.6 Å) has been attributed to steric hindrance. Many of the properties of molecules that play a role in bioaccumulation (e.g., hydrophobicity, solubility, vapor pressure, and dissociation constant) may be predictable from their moleculaar structures.

EEDP-01-18
July 1989

Environmental Effects of Dredging Technical Notes

FACTORS INFLUENCING BIOACCUMULATION OF SEDIMENT-ASSOCIATED CONTAMINANTS BY AQUATIC ORGANISMS; FACTORS RELATED TO SEDIMENT AND WATER

PURPOSE: This is the second technical note in a series of four which outlines and describes the principal factors that determine uptake and retention of chemicals by aquatic organisms. The first three notes describe factors related to contaminants, sediment and water, and biota. The fourth note is a glossary and bibliography. The information contained herein is intended to assist Corps of Engineers environmental personnel in activities requiring a working knowledge of concepts and terminology in the subject of chemical uptake, retention, and elimination by aquatic organisms exposed to contaminated sediments.

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Factors Relating to Sediment

Eh and pH (43-46)

Water or sediment pH (acidity/basicity) and Eh (oxidation-reduction potential) affect the concentration of metals and, to some extent, organic chemicals that are present in bioavailable form in natural systems. Natural waters are weakly to strongly oxidized and mildly acidic to mildly alkaline. Sediments, in contrast, are generally reduced and nearly neutral in pH. In sediments, both iron and manganese are in the divalent, relatively soluble state. Oxidation to the ferric and manganic forms favors formation of insoluble hydrous oxides that can coprecipitate or adsorb other soluble metallic and organic species, thus reducing their bioavailability and potential for bioaccumulation. Insoluble sulfides of many heavy metals that are formed under reduced conditions are rapidly oxidized, when conditions change to aerobic, first to elemental sulfur and then to sulfate releasing the soluble metal ion. The processes of iron and manganese oxidation produce hydrogen ions; sulfide oxidation produces sulfate, and the result of these oxidations is an increase in acidity. Acidic conditions favor the solution of free metal ions but also favor the formation of insoluble hydrous oxides that tend to reduce the concentration of metal ions in solution by adsorbing them. The interactions of these two processes are thus in opposition.

As a general rule, free ions tend to be present in greater abundance and are thus more bioavailable at low pH and under oxidizing conditions. Under reducing conditions, metals are present largely as insoluble sulfides and are not bioavailable. Trace metals associated with sediments that are not bound in the sediment crystal matrix are present either as ions, complexes, or precipitates. Aqueous concentration of the free ions is regulated by solubility of the precipitates under prevailing conditions of Eh and pH.

Hydrous ferric and manganese oxides (45, 47-55)

Hydrous ferric and manganese oxides form amorphous aggregates that contain large amounts of water. These aggregates have surface areas many times greater than those of clay minerals. Hydrated metal ions and soluble complexes are able to diffuse through the aggregate structures in addition to being surface adsorbed. The effect is analogous to that of an ion-exchange resin in that metals can be concentrated by the aggregates. Rates of adsorption and desorption processes are variable according to conditions of external concentration, pH, Eh, temperature, and to a limited extent ionic strength of solution. Additionally, hydrous oxides which form rapidly when reduced sediments are oxidized may scavenge soluble metals and organic chemicals from the water column by coprecipitation. In low organic carbon substrates, hydrous oxides may also play a significant role in reversible sorption of organic chemicals. Hydrous oxides thus affect bioaccumulation indirectly by influencing the sorption, and thus the bioavailability, of chemicals associated with sediments.

Kinetics of adsorption/desorption (56-65)

Adsorption and desorption of hydrophobic contaminants to and from natural sediments have been described as biphasic processes having a labile (rapid) component and a nonlabile (slow or resistant) component. Sediment particle size, organic carbon content, and relative hydrophobicity of individual chemicals are major factors influencing rates of sorption. About 10-60 percent of the sorption capacity of sediment particles typically appears to be accounted for in the labile fraction; i.e., adsorption or desorption occurs in a matter of minutes. Sorption to or from the remaining sites (nonlabile fraction) takes place over a period of days to weeks in laboratory experiments. Highly hydrophobic chemicals tend to sorb slowly. It has been estimated that chemicals having sediment/water equilibrium distribution coefficients (K_p or K_D) greater than 10^5 will likely require more than a year to completely desorb from a sediment. Kinetics of desorption, then, are of particular interest in estimating the bioavailability of hydrophobic chemicals from sediments. Estimation methods that rely on equilibrium distribution of chemicals among environmental phases may overestimate the bioavailable fraction of a chemical in sediments, depending on the time frame allowed for equilibration. Rates of sorption processes involving metals are strongly pH and Eh dependent.

Oil and grease (66-69)

Oil and grease (O&G) is a nonspecific determination often included in sediment chemical inventories. O&G is primarily composed of nonbioaccumulating alkanes; however, in sediments O&G may affect the bioavailability of other chemicals that do bioaccumulate. If present in a sufficiently high concentration to constitute a discrete phase, O&G may concentrate organic chemicals in a manner similar to sediment organic carbon. In effect, O&G could add incrementally to the total organic carbon (TOC) pool in a sediment (see "Sediment organic carbon"), thus reducing the bioavailability of organic chemical contaminants to biota. However, the mass contributed by total O&G in sediment is usually insignificant compared with the mass represented by humic TOC, and can usually be disregarded.

Particle interactions (70-75)

The desorption of contaminants from sediment particulates is apparently affected by physical interactions among the particles. Inverse correlations between particulate concentrations in suspensions of sediments and the partition or distribution coefficients between the particulates and water have been reported for both metals and organic chemical contaminants. These observations appear contrary to equilibrium partitioning theory as partition coefficients are descriptive of absolute conditions and are subject primarily to fundamental changes in physical properties (temperature, pressure, and state), and not to secondary changes in physical conditions, such as concentration. A possible explanation of the "sliding partition coefficient" is that increasing the concentration of particulates in a suspension increases the frequency of collision between particulates. Collisions between particulates bearing organic carbon to which contaminating chemicals are "loosely sorbed" could result in an increase in the solution-phase concentration of contaminants.

In bedded sediments where the particles are at rest, partition coefficients are constant, and for hydrophobic chemicals K_{OC} (see "Sediment organic carbon") describes equilibrium distribution with the interstitial water. However, in dilute suspensions where particulates are highly organic, it is K_{OW} rather than K_{OC} that best describes partitioning. It has been suggested that only about 40 percent of the surface organic carbon of particles makes up the lining of the pores in bedded sediments and thus only 40 percent of the organic carbon by mass is available for exchange with the interstitial water. This would account for the fact that for most neutral chemicals K_{oc} is about 40 percent of K_{ow}. The hypothesis postulates that reduced surface area within the pores and loose sorption at the surfaces of particulates account for the difference in partitioning.

The particle interaction effect is still not fully explained or accepted in the scientific community. However, the existence of such an effect could have substantial implications for contaminant bioavailability during dredging operations that produce high turbidity. The suspension of high levels of contaminated sediments during disposal operations could conceivably increase the concentration of desorbed chemicals in the water column. Such an effect would amount to an increase in bioavailability for exposed organisms because the amount of unbound chemical present would be greater than could be expected from simple desorption. The effect of particle concentration on solution-phase concentration of chemicals has been modeled, but reported research that sheds light on whether, and to what extent, such processes affect bioavailability under natural conditions is lacking.

Sediment organic carbon (62-63, 76-91)

Sediment organic carbon consists primarily of humic matter and may constitute as much as 10-20 percent of navigation channel sediments. Ranges for harbor sediments are generally on the order of 1-4 percent and may be much less than 1 percent in very sandy sediments. The organic carbon in sediments is primarily responsible for sorption of neutral organic chemicals such as PCBs or PAHs; mineral surface adsorption sites for such compounds become important only when the sediment TOC is very low, perhaps less than 0.5 percent. Organic carbon behaves as though it were an organic solvent in competition with the lipids of biota for distribution of any neutral organic chemicals that are present. For neutral organic chemicals the TOC content of the sediment is the primary determinant of bioaccumulation potential. The bioaccumulation potential of a sediment is the concentration of a chemical in an organism's tissues that would result from exposure to a contaminated sediment if an equilibrium chemical distribution could be established between the sediment and the organism. Bioaccumulation potential is a thermodynamic concept independent of rates of desorption, transport, uptake, or elimination. For a given concentration of a neutral chemical on a whole sediment basis, high TOC content reduces bioaccumulation potential, and lower TOC proportionally increases it.

TOC provides a basis for normalizing chemical concentration data among sediments of differing origin so that comparisons can be made. This is accomplished by dividing the concentration of chemical in the sediment by the concentration of TOC in the sediment, expressed as a decimal fraction. For example, two sediments, one having 2 ppm PCB and 6 percent TOC, and the other having 1 ppm PCB and 3 percent TOC would both have 33 ppm PCB on an organic carbon-normalized basis and would have the same bioaccumulation potential.*

The concentration of PCB in the interstitial water of the two sediments would also be the same. The partition coefficient that describes equilibrium distribution of neutral organic chemicals between sediment and water, K_{oc}, is calculated using organic carbon normalization of concentration data. In the example above, if the PCB were analyzed as Aroclor 1254 (log $K_{oc} \cong 6.05$), the organic-carbon normalized concentration of PCB in the sediment (33 ppm) would be divided by $10^{6.05}$ to get 29 pptr (parts per trillion), the expected equilibrium concentration of PCB in interstitial water. Since the solution phase concentration is the most bioavailable, these calculations lead to an estimate of bioaccumulation potential. Application of a bioconcentration factor (BCF) to the interstitial water concentration gives an estimate of chemical concentration that could be expected in an exposed organism. If the appropriate log BCF for a representative organism $\cong 5$, the bioaccumulation potential would be 2.9 ppm.

Metals also associate with the organic carbon fraction of sediments. However, the association is primarily by active bonding with functional groups rather than by passive equilibrium. In the case of metals, there is no simple relationship between TOC and bioavailability or bioaccumulation potential.

Sediment particle size (85, 92-94)

As sediment particle size decreases, the surface area of the particles per unit mass of sediment increases. Increasing the surface area increases the number of negatively charged sites for adsorption, and therefore the number of cations that can be carried on the sediment. The sediment surface also provides sorption sites for neutral organic chemicals that associate through van der

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* For a discussion of calculations involving organic carbon normalization and estimates of bioaccumulation potential and bioabailability, see *Environmental Effects of Dredging Technical Note* EEDP-01-8.

Waals/London forces. Sediment particulates may have coatings of humic matter, and most of the organic carbon is associated with the finer-grained material. For these reasons chemical contaminants in sediments are associated primarily with the fine-grained fraction of sediments. Infaunal organisms that dwell in and/or ingest fine-grained material are potentially exposed to higher environmental concentrations of chemicals than are those in coarse-grained sediments, and usually reflect this in their higher bioaccumulation. The same is true of filter-feeding organisms that select small-sized particulates for ingestion.

Sediment suspension (95-104)

Dredging or disposal operations that involve the suspension of sediments can, at least transiently, increase the concentration of associated chemical contaminants in the water column. The increase is not a simple linear function of the mass of sediment suspended because the contaminant-bearing TOC of the suspended sediment fraction is typically higher than the TOC of the consolidated deposited sediment. However, particulate organic matter can act as a scavenger of metals and organic chemicals from solution, thus reducing the bioavailable fraction in the water column.

In sea water the presence of divalent cations (Mg^{++}, Ca^{++}) can cause resuspended particulate, colloidal, and soluble organic matter to flocculate and settle from the water column. Under these conditions, lower molecular weight organic acids can be precipitated as metal fulvates and humates. Trace elements may coprecipitate with flocculated material. Suspension of uncontaminated sedimentary material has been demonstrated to reduce the bioavailability of contaminants by adsorbing them from solution. Conversely, the suspension of contaminated sediments in clean water has been reported to result in bioaccumulation by exposed organisms. In such cases fugacity favors desorption from particulates to the water, and chemicals including PCBs, kepone, lead, and mercury that are bound with the particulates may be made bioavailable.

Factors Relating to Water

Dissolved organic carbon (105-119)

Dissolved organic carbon (DOC) in natural systems is composed primarily of humic substances produced by the degradation of dead plant material. Humic and fulvic acids make up 40-80 percent of DOC and are defined according to the effects of pH on their precipitation from aqueous solution. These organic acids are structurally complex colloidal and subcolloidal compounds containing large numbers of functional groups (e.g., phenolic, hydroxylic, and carboxylic acid) and straight and branched alkyl side-chains. The functional groups make these large molecules, micelles, and aggregates water soluble and also provide cationic-exchange sites for metal ions in solution. The alkyl chains provide sites for adsorption of hydrophobic chemicals.

The concentration of DOC, or humic and fulvic acids, affects bioavailability and, thus, bioaccumulation of chemicals by aquatic biota. Reduced uptake in aquatic organisms has been demonstrated when metals or neutral organic chemicals are added to water containing uncontaminated humic acids. In the water column high DOC concentrations appear to reduce bioaccumulation by adsorbing neutral organic contaminants and making them less available to organisms. Metals such as copper and zinc may be more or less available depending on salinity and suspended particulate concentrations.

<u>Hardness (118, 120-124)</u>

Elevated concentrations of polyvalent cations, primarily calcium and magnesium, in water reduce the bioavailability of toxic metallic species. The interactions of hardness, alkalinity, and pH have been studied in the context of toxicity, rather than bioaccumulation. However, since bioavailability is a determinant of both toxicity and bioaccumulation of metals, it is reasonable to assume that increased water hardness may also reduce bioaccumulation of metals through a reduction in metal bioavailability. The influence of hardness on bioaccumulation of most organic compounds is negligible.

<u>Salinity (125-137)</u>

Salinity affects bioaccumulation both directly and indirectly. The mechanisms involve effects on physicochemical processes including desorption and solubility as well as effects on physiological processes such as osmoregulation, membrane permeability, and respiration rate and volume. In salt water there may also be competition among free ions for tissue binding sites.

For organic contaminants, especially neutral organics, increasing salinity usually decreases the water solubility of the compounds. Both particulate organic carbon, and dissolved organic carbon are inversely related to salinity. Since bioavailability of neutral organics is also inversely related to TOC, the decrease in organic carbon with increasing salinity may under some conditions actually enhance bioavailability of neutral organics to organisms.

The relationship of salinity to metal bioaccumulation is more complex and element specific. Metals in solution have been reported to bioaccumulate to higher concentrations as salinity decreases, but the opposite may also be true. Increasing salinity decreases the binding strength of Cd, Cu, Mn, and Zn to inorganic ligands, both by the competition of other major cations for binding sites and by favoring the formation of chloride complexes. The free ion is the form of greatest bioavailability, but the variable amounts of dissolved and particulate carbon (related to salinity) confound the picture by providing sites for complexation. In general, Se solubility and bioavailability are inversely related to salinity; Zn uptake is unrelated to salinity; Cu results are erratic and are especially affected by organic complexation; Pb uptake increases with increasing salinity; Hg binds very tightly to particles and does not respond to salinity changes; and Cd uptake is inversely related to salinity.