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Section 01 - Aquatic Disposal
Technical Notes
EEDP-01-19 through EEDP-01-33

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Section 01—Aquatic Disposal
Technotes EEDP-01-19 through EEDP-01-36

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Environmental Effects of Dredging Technical Notes

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

PURPOSE: This is the third 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 in the series describe factors relating 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.

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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.

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Biotransformation (138-148)

Biotransformation is the process by which foreign chemical substances are enzymatically oxidized, reduced, cleaved, rearranged, or conjugated within the metabolically active organs of biota. In Phase I of metabolic detoxication, foreign chemical compounds are biotransformed to reactive metabolites (bioactivation) enabling them to undergo further chemical reactions. Bioactivation is usually followed by conjugation with endogenous substrates in Phase II of metabolic detoxication, and excretion from the organism. Biotransformation can also result in activation of a foreign compound to a toxicant of higher potency than the parent compound, as is the case with the polynuclear aromatic hydrocarbon (PAH) benzo[a]pyrene. There are substantial differences among organisms in their ability to biotransform chemicals. For example, the PAHs tend to bioaccumulate in certain amphipods and bivalves that do not possess the enzyme systems necessary to metabolically detoxify and eliminate them. These same PAHs are found in much lower concentrations in most fishes because the fishes do possess the necessary enzyme systems for biotransformation and degradation.

The effect of biotransformation is to reduce the amount of unchanged chemical that is bioaccumulated by an organism. However, in some cases, metabolites may be bioaccumulated rather than excreted. For example, dichloro-diphenyl-trichoroethane (DDT) is metabolized to the -dichloroethane (DDD) and then to the -dichloroethylene (DDE) by most aquatic organisms. Over time DDT residues in organisms diminish, while DDE residues increase.

Depuration (149-155)

Depuration refers to the elimination of toxic substances from an aquatic organism by all processes and occurs concomitantly with uptake of chemicals. Steady-state bioaccumulation is considered to exist when the net loss of a chemical by depuration is equal to the net gain by uptake. Removal of an organism to conditions of lower exposure favors depuration over uptake. In most cases, depuration is a biphasic process; first, chemicals in the bloodstream or in tissues with high blood-exchange are depurated, and then the same chemicals in storage tissues such as depot fat are mobilized and eliminated over a longer period of time.

Diet (56, 93, 156-166)

Recent studies assign a greater role to contaminated food as a major pathway for bioaccumulation of contaminants in aquatic organisms. High levels of chemicals in the tissues of water column-dwelling fishes that are exposed to only very low concentrations of chemicals in the water are explained on the basis of ingestion of contaminated food. Benthic infaunal species that ingest sediment as a part of their diet probably also receive a large part of their body burden through feeding. Dietary accumulation is dependent on feeding and clearance rates and on the ability of organisms to assimilate chemicals. If the food that an organism ingests is highly contaminated relative to the water that the organism respires, and if the quantity of contaminated food ingested is also large, diet is likely to be the dominant pathway for bioaccumulation.

Feeding Type (167-169)

Bioaccumulation through dietary exposure is also influenced by the manner in which an organism obtains its food, i.e., the organism feeding type. The assimilation efficiency of organic chemicals and organometalloids from food to predatory fish ranges about 65-95 percent. Deposit feeders that ingest contaminated sediment assimilate these contaminants with about 20-40 percent efficiency. Filter feeders are intermediate or similar to deposit feeders in their assimilation efficiencies.

Kinetics of Uptake and Elimination (81, 141, 153, 170-196)

The rates at which chemicals are taken up and eliminated by organisms are major determinants of both the time required for and the magnitude of bioaccumulation. Bioaccumulation in aquatic organisms is most commonly viewed as a one-compartment model in which uptake and elimination take place simultaneously, but at different rates. Rate constants for uptake (k_1) and for elimination (k_2) can be calculated from empirical data or can be estimated from physiocochemical parameters such as K_{ow}. The ratio, K_1/K_2, is the bioconcentration factor (BCF) of a chemical when the exposure medium is water. The same model has been applied to contaminants in sediments, in which case k_1/k_2 is the sediment bioaccumulation factor (BAF) and relates chemical concentration in sediment to steady-state concentration in exposed organisms. A fundamental requirement of the simple kinetic model is that exposure concentration be constant. At constant exposure, the concentration of chemical in the tissues of an organism at steady state exceeds the concentration in the exposure medium by the magnitude of the BCF (or BAF). Although steady-state conditions rarely occur in the real world, the concept is a useful simplification that makes kinetic calculations possible.

For nonmetabolizing neutral organic chemicals, the rate of elimination is inversely correlated with hydrophobicity, resulting in very slow elimination for the higher molecular weight hydrophobic compounds. However, rates of uptake for such chemicals are generally rapid. Such compounds are taken up and eliminated passively, and bioaccumulation usually follows first-order kinetics as described above. The slower the rate of elimination, the greater will be the magnitude of bioaccumulation for a given chemical. For chemicals that are metabolized, rates of elimination can be accelerated by mixed-function oxidase (MFO) induction (see "Mixed-Function Oxidases"). Animals receiving prior exposure to the same or to similar chemical substances may develop the ability to depurate those substances rapidly by having synthesized greater quantities of MFOs. Bioaccumulation under these circumstances is reduced compared to bioaccumulation in similar but chemically naive animals.

In a few studies first-order kinetics have also been able to describe the uptake and elimination kinetics of metals. However, the kinetics of metal bioaccumulation are influenced by many more variables than are the kinetics of hydrophobic organics, and simple models are generally less successful in describing them.

Lipid Content (87-88, 171, 197-205)

Hydrophobic chemicals tend to be stored in body lipids, predominantly in fat. Lipids are organic substances of biological origin that are insoluble in water. Lipids include structural substances such as phosphatides, substances that are involved in various biochemical reactions such as steroids and carotenoids, and the fats and waxes. Fats are relatively inert substances that constitute reserve energy stores for biota. Storage lipids composed primarily of fats have the highest affinity for neutral chemicals. In general, the higher the total lipid content of an organism, the greater its capacity for bioaccumulation of hydrophobic chemicals. The total lipid content of an aquatic organism or the lipid content of specific tissues is now frequently used as a basis for normalizing the concentration of neutral organic chemicals found in organisms. Normalization of concentration data makes it possible to make interspecies comparisons of bioaccumulation. Analytical procedures for lipids in biota have not yet been standardized for environmental samples. Total lipids are often measured gravimetrically as the residue after evaporation of a hexane extract prepared for analysis of organic contaminants in aquatic biota.

Metabolic Rate (14, 145, 150, 156, 176, 194, 206-211)

Metabolic rate affects bioaccumulation in several ways. A high rate of metabolism is usually accompanied by increased rates of oxygen uptake. Rates of oxygen uptake closely parallel rates of contaminant uptake from water in aquatic species. However, rates of biotransformation and excretion may also be accelerated by increases in metabolic rate. The net effect on bioaccumulation depends on whether the uptake or depuration process is favored. For example, elevated metabolic rate during a period of reduced external contamination in most cases would tend to favor increased elimination of chemicals and a reduction in body burden.

Metallothioneins (212-222)

Low molecular weight sulfur-containing proteins that bind certain metals (metallothioneins) are produced in the kidneys, livers, gills, and digestive organs of most aquatic organisms. Metallothioneins primarily are involved with regulating the metabolism of essential trace metals. However, for many organisms metallothioneins also provide a measure of protection against the toxic effects of certain metals, mainly Cu, Cd, Zn, and Hg. The synthesis of metallothioneins is inducible in a manner analogous to MFO induction. Low-level acute or chronic exposure of biota to certain metal ions can produce a tolerance to the toxic effects of those metals through the induction of metallothioneins. The consequence of metallothionein induction to bioaccumulation is an enhanced ability of organisms to accumulate certain metals before the appearance of toxic effects.

Mixed-Function Oxidases (138, 140, 147, 223-235)

Mixed-function oxidases (MFOs), also referred to as monooxygenases, are cytochrome P-450-dependent intracellular enzymes that function mainly in the oxidative metabolism of lipoidal endogenous compounds such as steroids, and in the first phase of detoxication of foreign organic compounds. Several types of MFOs may be found in metabolically active organs of all vertebrates, including fishes, and of most invertebrates. In fishes and aquatic invertebrates, the most highly developed MFO systems are those that catalyze the biotransformation of planar aromatic lipid-soluble chemicals like the PAHs into more water-soluble compounds. Increasing the water solubility of such compounds makes them more easily eliminated. Exposure of an inducible organism to low PAH concentrations or to sufficiently similar compounds, such as the coplanar polychlorinated biphenyls (PCBs), stimulates (induces) the synthesis of appropriate MFOs for detoxication of those compounds. Subsequent exposure of the induced organism to the same or to a chemically similar organic chemical can be met with an increased capacity to eliminate the chemical and a reduced level of bioaccumulation.

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Environmental Effects of Dredging Technical Notes

FACTORS INFLUENCING BIOACCUMULATION OF SEDIMENT-ASSOCIATED CONTAMINANTS BY AQUATIC ORGANISMS; GLOSSARY AND BIBLIOGRAPHY

PURPOSE: This is the fourth 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 in the series describe factors relating to contaminants, sediment and water, and biota. This note contains a glossary of terms and a bibliography of key and recent publications in the scientific literature containing supporting data and discussion on each topic. 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.

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Glossary

Absorption: assimilation of a chemical into biological tissue by capillary, osmotic, chemical, or solvent action.

Adsorption: condensation of gases, liquids, or dissolved substances on the surfaces of solids.

Assimilation efficiency: speed and effectiveness with which a chemical in food is incorporated into the tissues of an organism.

Bilipid: membrane formed of two separate sheets of lipid molecules which orient themselves so that the polar headgroups are exposed to the outer aqueous environment and the nonpolar tails are exposed to each other.

Bioaccumulation potential: equilibrium concentration of a foreign compound that could result in an organism's tissues given unlimited time and an absence of degradative and gradient effects.

Bioavailability: extent to which the fraction of the total chemical in the environment is available for uptake by an organism.

Biphasic: having two separate and distinct stages or periods.

Body burden: total concentration of a chemical in an organism taken up from the environment.

Cation: positively charged ion.

Cationic exchange capacity: extent to which negatively charged groups of a sediment matrix are able to exchange one cation for another.

Coprecipitate: inclusion of ions in a precipitation reaction by physical association rather than chemical bonding.

Complexation: bonding of metal ions with organic molecules.

Conjugation: addition reactions in which large chemical groups or entire compounds such as sugars and amino acids are covalently added to endogenous or foreign organic chemical compounds in metabolic detoxication.

[redacted]

Free ion: unbound charged particle in solution.

Functional group: an assemblage of atoms that imparts chemical activity to a molecule.

Humic material: complex heterogeneous substance produced in soils and aquatic sediments by the decay and decomposition of organic matter, chiefly of plant origin.

Hydrolysis: double decomposition reaction involving the splitting of water into its ions and the formation of a weak acid or base or both.

Hydrous oxides: amorphous, noncrystalline and permeable structures composed primarily of the oxides of iron and manganese and formed on mineral particles.

In situ: in the natural or original position.

Induction: stimulation of synthesis of enzymes through an increase in available substrate for enzymatic action.

Ion-exchange resin: permeable solid containing chemically bound charged groups to which ions are electrostatically bound and exchangeable with other ions of like charge.

Ionic strength of solution: relative concentration of charged particles in a solution.

Labile fraction: portion of a compound that readily undergoes physical, chemical, or biological change.

Manganic: substance composed of, relating to, or containing manganese, especially those in which the manganese is trivalent.

Micelle: water-soluble molecular aggregate composed of molecules containing both polar and nonpolar components that form with the polar units oriented to the outside of the aggregate and the nonpolar groups to the inside.

Normalization: expression of concentration data for a chemical in a complex mixture on the basis of one component of the mixture that is thought to account for most of the association of the chemical with the mixture.

One-compartment model: kinetic model in which the organism is considered as an integrated unit in terms of uptake and elimination; individual internal distribution and disposition rates are not considered.

Organometalloid: complex formed by binding of a metallic ion with an organic ligand.

Partitioning: distribution of a chemical between two immiscible solvents or phases.

Passive equlibration: equalization of concentration of a chemical substance on both sides of a membrane without the use of energy consuming processes.

Perfusion: pumping of a fluid through an organ or a tissue.

Polyvalent: substance that can have more than one valence state.

Protonation: uptake of hydrogen ions by a molecule to give an overall positive charge.

Steady state: state at which the competing rates of uptake and elimination of a chemical by an organism are equal and the net exchange of chemical is zero.

Substrate: chemical, usually a biogenic macromolecule, that serves as a reactant in biochemical transformation processes.

Van der Waals/London forces: relatively weak electrostatic attraction between atoms and molecules arising from fluctuations in their electron distributions as the electrons circulate in their orbitals.

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ments information in Technical Note EEDP-01-12 (Clausner 1988) on using sea bed drifters (SBDs) to site and monitor feeder and stable berms.

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Introduction

A nearshore berm consists of dredged sand placed in a long mound in shallow water (usually less than 25 ft), often parallel to shore or bottom contours. Typically they are constructed from maintenance dredged sand using split hull hopper dredges and are 4 to 10 ft high above surrounding topography, 400 to 700 ft wide at the base, and over 5,000 ft long.

Nearshore berms have several advantages over conventional offshore disposal. Often, placing sand close to the inlet from which it was removed may be cheaper than disposal in designated offshore sites or directly on the beach. For example, costs per cubic yard for the various disposal options from Fire Island Inlet, NY, were:

US Army Engineer Waterways Experiment Station, Environmental Laboratory
PO Box 631, Vicksburg, Mississippi 39181-0631

Beach Nourishment	Historical Disposal Site	Feeder Berm
$5.50 (bid)	$4.00 (estimated)	$2.23 (actual)

The historical disposal site was approximately 2 miles from the dredging site and 0.5 mile offshore.

Nearshore berms also have potential benefits for beaches. Since the berm forms a barlike feature, it can dissipate incident wave energy by inducing wave breaking. As the berm disperses over time, it contributes quality sand to the nearshore system. The berm may also act as a partial block to the loss of beach materials to deeper water during storms. A nearshore berm may also move onshore, contributing visible amounts of sand to the dry beach. In this case, the nearshore berm can be termed a feeder berm. However, the research and field experience to define the combinations of sediment characteristics and environmental conditions necessary for onshore movement of sand are not complete. Therefore the term nearshore berm, which only describes where the feature is placed without inferring its ultimate contribution to the littoral system, is generally preferred.

Physical monitoring of nearshore berms involves measuring changes in elevation and volume through successive bathymetric surveys. Most monitoring plans will also include taking sand samples along the berm and possibly on the beach to measure changes in grain size. Beach profiles are often taken to determine changes in response to the berm. Because of the limited experience with nearshore berms, design guidance is not yet available. Consequently, measurement of the driving forces--waves and currents--has been included on some projects.

Physical monitoring is needed to more directly quantify the physical benefits of nearshore berms, verify performance, and check construction. This technical note summarizes the monitoring plans used or proposed for several nearshore berms and concludes with monitoring recommendations for nearshore berms in general.

Biological benefits of nearshore berms are also possible. The major potential benefit would be increasing fisheries value resulting from a change in bathymetry or grain size which may attract other types of fish not normally found at the site. However, monitoring programs to date have not investigated this aspect, so this technical note will focus solely on physical monitoring.

Monitoring Programs For Existing Nearshore Berms

During 1987, three nearshore berms were constructed--one off Sand Island, AL, and two along the southern shore of Long Island, at Gilgo and Lido Beaches, NY. (The Lido Beach project is not discussed here for lack of available information.) Hands (in preparation) discusses interim monitoring results for the Sand Island nearshore berm. McLellan, Truitt, and Flax (1988) present detailed information on the Gilgo Beach nearshore berm. A nearshore berm was completed off south Padre Island, TX, in January 1989. Monitoring procedures for each project are summarized in Table 1.

Generalized Nearshore Berm Monitoring Guidelines

The following generalized nearshore berm monitoring guidelines have been synthesized from the experiences and recommendations described above. Since the number of berm projects is limited and data analysis continues, modifications to these recommendations are likely. The most important recommendation is to begin the initial monitoring phase as soon as possible after construction is completed. Shallow placement of the berms makes them particularly susceptible to rapid sediment dispersion.

Bathymetry

Bathymetry is the backbone of nearshore berm monitoring, providing volume and elevation change information, and should be included on all projects. Survey lines should be run perpendicular to the berm alignment at a 200-ft spacing, continuing from the breakers, across the berm, out to closure depth. This depth will typically be from 20 to 30 ft on the East and Gulf Coasts, and 30 to 45 ft on the West Coast. Nearshore berms have rarely migrated onshore intact. Instead they have generally dispersed or spread, with the major movement in the alongshore direction. Therefore surveys should extend from a minimum of 1,000 ft updrift of the berm to 2,000 ft downdrift. Preconstruction, immediate

Table 1

Summary of Nearshore Berm Monitoring Activities

Monitoring	Projects		
	Fire Island/ Gilgo Beach, NY	Brazos-Santiago/ Padre Island, TX	Mobile Bay/ Sand Island, AL
Surveys			
Hydrographic Surveys of Nearshore Berm	100 ft spacing between lines. Pre-, mid-, post-, 1 mo, every 2 months.	500 ft spacing between 14 lines, each 3500 ft. Pre-, post-, 1 mo, quarterly.	200 ft spacing between 42 lines, each 2000 ft long. Pre-, every 2 weeks for 2 mo, every 2 months.
Beach Profiles	500 ft spacing. Pre-, mid-, post-, quarterly.	1000 ft spacing, 11 lines. Post-, 6 mo, 12 mo.	None
Sediment Samples			
Nearshore Berm	Pre-, post-, 2 mo.	12 grab, 6-10 cores. Per survey.	31 grab, min. 200 ft apart. Per Survey.
Beach	Pre-, post-, 2 mo.	Undetermined.	None
Waves/Currents	LEO	LEO	Nearshore wave/current gages Offshore wave/meteorological
Side-scan Sonar	None	None	Pre-, two post-surveys.
Seabed Drifters	None	Bundles released from 4 sites, each survey.	Bundles of 50 released from 6 sites, each survey.

(Continued)

Table 1 (Concluded)

Monitoring	Projects: Fire Island/ Gilgo Beach, NY	Brazos-Santiago/ Padre Island, TX	Mobile Bay/ Sand Island, AL
Aerial Photography	Pre-, post-, 3 mo.	Post- 1 additional.	2 mo, 7 mo.
Characteristics			
Length (ft)	7,500	5,300	8,000 ft
Width (ft)	500	*	500-700 ft
Elevation (ft)	1-9 ft, 4 ft avg	*	6-8 ft
Amount of Material (cu yd)	420,000 cu yd	225,000 cu yd	464,000 cu yd
Water Depth of Base	-16 ft mlw	-22 to -19 ft mlw	-19 ft mlw

* This information still unavailable pending analysis of monitoring data.

postconstruction, and quarterly surveys are recommended, with a minimum of surveys twice per year, e.g., late winter/early spring (March/April) and late summer/early fall (September/October).

Fathometer surveys should be of high quality since the volume percentage of the berm represented by a ±0.5 ft error band is large. Microwave positioning is a must. Tide corrections, based on a nearby open-water tide gage if possible, are also required, as are vessel squat and speed of sound corrections. Clausner and Hands (1988) and Fredette et al. (in preparation) discuss these surveying and positioning factors in greater detail.

Often the final construction acceptance survey may be used as the initial monitoring survey. If surveys are to be performed by a combination of Field Operating Activities personnel and private contractors, data compatibility and consistency must be assigned. This is particularly true if volume change and elevation data will be analyzed by computer. Figure 1 shows the contours associated with the berm created off Gilgo Beach.

Beach profiles

The need for beach profiles will be a function of the purpose of the nearshore berm. If the nearshore berm is intended to provide beach protection or nourishment, then beach profiles will be needed to quantify the effects. As the depth of the berm increases, probable short-term effects on the beach will

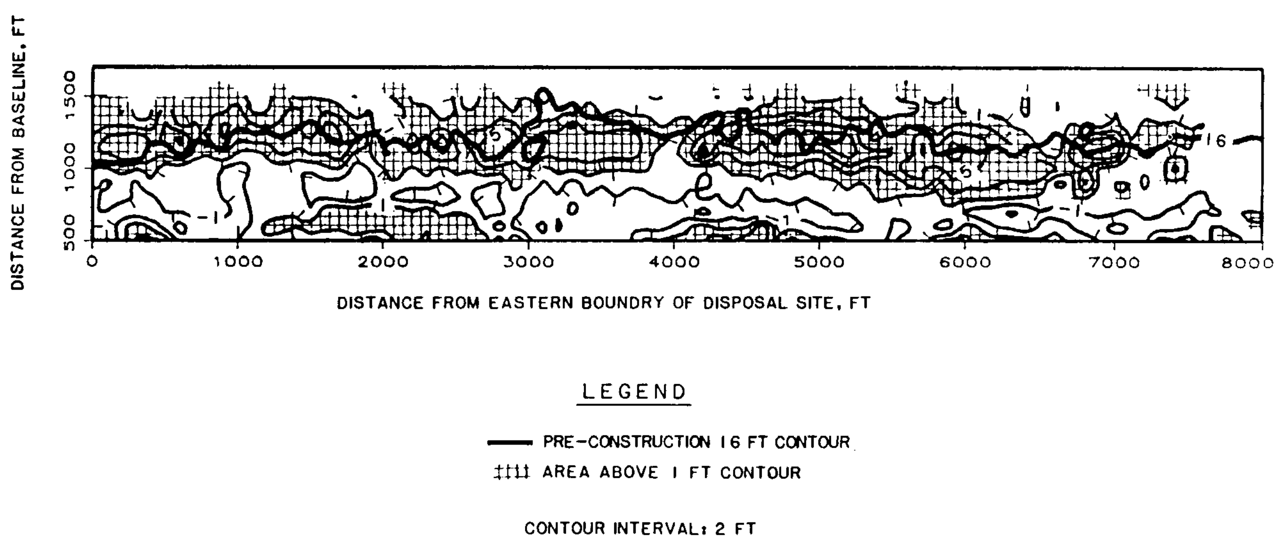

Figure 1. Contour difference plot of Gilgo Beach feeder berm

decrease, reducing the need for frequent beach profiles. If the nearshore bermis intended strictly to save money by reducing haul distances and is being placed where beach erosion is not a problem, beach profiles may not be needed. Nonetheless, potential claims of adverse effects due to the berm probably make it prudent to take a limited number of pre- and postconstruction profiles.

Beach profiles with 500-ft spacing should be adequate for most projects. These profiles should be taken at the same frequency as bathymetry if possible, and should extend updrift and downdrift of the berm. To better define the benefits of a berm, a control section of the beach, some distance away from the berm with similar erosion history, should also be surveyed.

Sea-sled surveys (Clausner, Birkemeier, and Clark 1986) are a highly accurate option to Fathometer surveys. One advantage of sled surveys is that they can measure the profile through the surf zone. In many cases, sled survey lines can easily extend from the subaerial beach seaward across the nearshore berm out to closure depth, allowing monitoring of the entire profile simultaneously.

Sand sampling

Sand samples should be taken and analyzed to help determine migration of the berm. Usefulness of this technique will be reduced if grain-size distributions of the berm and native material are similar. Ten samples per mile of berm, with the samples distributed between the crest and flanks should be sufficient. Samples should be obtained during the bathymetric surveys if possible. Grain-size analysis using 1/4 phi sieves should be obtained for each sample. Control samples from adjacent areas would provide a measure of natural variability.

Short cores can be taken to show depths to which sediments are being worked by waves and currents. Cores can be X-rayed to show sediment reworking and subsampled for grain-size analysis at different elevations. This level of monitoring is not recommended for most nearshore berm projects.

Waves/currents

Measurements of the forces driving movement of nearshore berms are very desirable, but quality long-term measurements of waves and currents are expensive at present. Ideally, directional wave and alongshore current measurements would be taken both on seaward and landward side of the berm. This should produce data on wave height reduction due to the berm, modification of wave direction due to refraction over the berm, and changes in alongshore/cross-shore currents.

The cost of installing and maintaining instruments, combined with data analysis costs, will generally make these coastal process measurements practical only for a limited number of research efforts such as Sand Island. In addition, as mentioned earlier, fishing/shrimping activities make it difficult to protect gages.

However, a District may plan to use nearshore placement and berm construction repeatedly for maintenance dredged material disposal. Then wave/current measurements from the initial placement could provide input into a numerical model to extrapolate the effects of the waves and currents on the berm for future placements.

A low-cost alternative to instruments are Littoral Environment Observation (LEO) measurements (Schneider 1981). However, LEO data only allow qualitative estimates of wave height, direction, and alongshore currents. Training, supplies, and processing LEO data cost approximately $3,000 for the first year, and $2,000 per year for subsequent years.

Sea bed drifters

SBDs are umbrella-shaped, near-bottom current drogues. They are perhaps more useful as devices to help site berms, but can be used on existing nearshore berms to provide insight as to direction of prevailing bottom currents. In addition, public involvement in return of the drifters can generate good, low-cost public relations. McLellan and Burke (in preparation) describe in detail an SBD study used to site the Brazos-Santiago Pass/Padre Island berm. See EEDP Technical Note 01-12 for details on actual use, and Hands (1987) for a review of earlier deep-water SBD studies.

Aerial photography

Aerial photography is standard practice for many monitoring projects. It is not very expensive and gives a continuous picture of the beach. While beach profiles provide much more accurate information on changes, aerial photography can, at low cost, provide information on beach changes for miles beyond the project boundaries (e.g., accretion at an adjacent jetty fillet). Use of aerial photography to directly monitor the berm is limited to cases of exceptionally clear water or very shallow berms (less than 4 ft).

It is recommended that aerial photography be included in nearshore berm projects. Aerials should be flown at least twice a year (at low tide) at times coinciding with profiles and surveys if possible. Color photography is recommended at a maximum scale of 1:4,800.

Side-scan sonar/subbottom profilers

Side-scan sonar produces an acoustic picture of the bottom, while subbottom profilers produce an acoustic image of sediment layers below the bottom surface. Based on experiences at Sand Island, neither instrument is recommended for monitoring nearshore berms in general. Both of these instruments are discussed in greater detail in EEDP Tech Note 01-10 (Clausner and Hands 1988).

Diver observations

Diver observations are probably not required for general nearshore berm monitoring. Divers can give information on small-scale processes and biological activity, take short cores, and maintain bottom instrument packages. However, their expense is probably not justified for most projects.

Wind

Wind data may prove useful in supplementing nearshore current observations by providing a measurement of this primary driving force. Wind data are often useful in interpreting SBD movements. If it appears that wind data should be used in interpreting nearshore berm performance, availability of wind data from local airports, the National Climatic Data Center, and local Coast Guard Stations should be checked.

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EEDP-01-22
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Environmental Effects of Dredging Technical Notes

Chronic Sublethal Sediment Bioassays for the Regulatory Evaluation of Marine and Estuarine Dredged Material: Proceedings of a Workshop

Purpose

This note summarizes the proceedings of a workshop held April 3-5, 1990, at the US Army Engineer Waterways Experiment Station, Vicksburg, MS. The purpose of the workshop was to solicit input from technical experts regarding techniques for evaluating the chronic sublethal effects of sediments on aquatic biota. This input will be used to help direct subsequent research and development by the US Army Corps of Engineers.

Background

The US Army Corps of Engineers uses an effects-based approach for the regulatory evaluation of dredged material. Bioassays are conducted to determine the toxicity of sediments and the bioaccumulation potential of sediment-associated contaminants. Survival of appropriate sensitive test species is used to measure acute sediment toxicity. This endpoint is quantal; that is, the test species either lived or died. Interpretation, therefore, is relatively straightforward.

Animals exposed to sediment normally accumulate contaminants at a slow rate compared to animals exposed to contaminants in aqueous solutions. Thus, sediment exposures connote chronic chemical exposures. Such chronic, low-level exposures are often not fatal but may elicit one or more subtle sublethal responses in the organism. These biological responses are designed to be adaptive. However, the chemical exposure may be of sufficient magnitude or duration that the

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organism's survival potential is impaired. Methods to accurately determine the chronic sublethal effects of sediment are not well developed. Moreover, the ability to discern adaptive from maladaptive sublethal responses (that is, interpret test results) is even more rudimentary.

The US Army Corps of Engineers has statutory authority for evaluating chronic impacts of dredged material. Regulations implementing Section 103 of the Marine Protection, Research and Sanctuaries Act (PL 92-532) state that, "[M]aterial shall be deemed environmentally acceptable for ocean dumping only when . . . no significant undesirable effects will occur due either to chronic toxicity or to bioaccumulation . . . ." Likewise, regulations implementing Section 404(b)(1) of the Clean Water Act (PL 92-500) state that, "[T]he permitting authority shall determine in writing the potential short-term or long-term effects of a proposed discharge of dredged or fill material on the physical, chemical and biological components of the aquatic environment . . . ."

In response to this statutory authority and technical need, a new research work unit, Chronic Sublethal Effects of Contaminated Dredged Material on Aquatic Organisms, was initiated within the Long-Term Effects of Dredging Operations (LEDO) Program.

Additional Information or Questions

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Introduction

Workshop participants were welcomed by COL Fulton, Commander and Director of the US Army Engineer Waterways Experiment Station (WES), and Dr. Tom Dillon, Workshop Chairman. Each participant was asked to briefly introduce himself and describe his technical background. All participants are recognized technical experts representing private industry, academia, and the Federal government (Table 1). A number of scientists from the US Environmental Protection Agency R&D laboratories (Narragansett, RI; Gulf Breeze, FL; and Newport, OR) were invited but were unable to attend.

Table 1. Workshop Participants

Mr. Steven Bay	Southern California Coastal Water Research Project, Long Beach, CA
Dr. Scott Carr	US Fish & Wildlife Service, Corpus Christi, TX
Dr. Ed Casillas	NOAA/National Marine Fisheries Service, Seattle, WA
Dr. Ted DeWitt	Oregon State University, Newport, OR
Dr. Jay Means	Louisiana State University, Baton Rouge, LA
Dr. David Moore	University of South Carolina, Columbia, SC
Dr. Frank Reilly	ASCI Corp., McLean, VA
Dr. John Scott	SAIC Inc., Narragansett, RI
Dr. Jack Word	Battelle Marine Research Laboratory, Sequim, WA
Dr. Tom Dillon, Chairman	US Army Engineer Waterways Experiment Station (WES)
Ms. Joan Clarke	WES
Dr. Robert Engler	WES
Ms. Freda Gibson	WES
Dr. Tom Fredette	New England Division, Waltham, MA
Mr. John Wakeman	Seattle District, Seattle, WA

Dr. Robert Engler presented the national perspective on the Corps' regulatory program for dredged material testing (Engler and others 1988). He explained the "effects-based" approach and the tiered testing protocol. He noted that while efforts are underway within the R&D community to develop chronic sublethal sediment tests, there are no generally accepted standard bioassays appropriate for the routine regulatory evaluation of dredged material. For that reason, use of these tests in a regulatory environment is restricted to special situations, for example, when biologically important bioaccumulation is observed with no concomitant acute toxicity.

Mr. John Wakeman, US Army Engineer District, Seattle, indicated that in the Pacific Northwest there is a broad-based constituency calling for the use of chronic sublethal testing in regulatory programs. In response, the Puget Sound Water Quality Authority has directed the Washington Department of Ecology to develop and eventually incorporate chronic sublethal tests into a variety of regulatory and monitoring activities. They have identified a 20-day growth bioassay with the marine benthic polychaete, *Neanthes arenaceodentata,* as a desirable chronic sublethal test (Johns and Ginn 1990). This procedure is also being considered by the Puget Sound Dredged Disposal Analysis (PSDDA) program for evaluating dredged material. The Corps' technical opinion is that more development is needed before this test is ready for use in a regulatory context.

Dr. Tom Fredette, US Army Engineer Division, New England (NED), described the New England Division's extensive monitoring program at aquatic relocation sites for dredged material. Although historical monitoring goes back to the 1930s, formal testing under the Disposal Area Monitoring System (DAMOS) did not begin until the 1970s. NED recently modified their testing protocol to include a 10-day bioassay with the amphipod *Ampelisca abdita* and 28-day bioassays with *Macoma balthica* and *Nereis virens*. Contaminant bioaccumulation potential will be assessed with the latter two species. Dr. Fredette indicated they would use a chronic sublethal sediment bioassay if it was technically sound, fully developed, ecologially relevant, and could be used in lieu of current testing procedures.

Dr. Frank Reilly summarized the results of a related workshop recently held at WES. The subject of that workshop was genotoxicity. This is a specific category of sublethal test that is being evaluated at WES under a separate but parallel effort. To avoid duplication, genotoxic endpoints were not addressed in any great detail during the current workshop. Details of the genotoxicity workshop will be reported in a future Environmental Effects of Dredging Technical Note (Reilly and others in preparation).

Dr. Dillon outlined objectives of the workshop and charged the participants with providing their best specific technical guidance. To initiate discussions, a hypothetical regulatory situation was described to the attendees (a permit action for marina dredging). They were asked to recommend a chronic sublethal sediment bioassay. They were specifically requested to address each of the workshop objectives by indicating how long the test would be run, what sublethal endpoint(s) would be monitored, what test species would be used, and how the test would be interpreted in terms of issuing or denying the permit. Response to this mock regulatory exercise is summarized in Table 2 and formed the basis for subsequent discussions at the workshop.

Workshop Objective 1: How Long is "Chronic"?

Current regulatory bioassays for evaluating the toxicity of dredged material may last up to 10 days. Since these are typically referred to as "acute" toxicity tests, one could infer that "chronic" tests are longer than 10 days. But how much longer? From the participants' response (Table 2), 3-6 weeks appears to be an appropriate timeframe. However, it was not possible to reach consensus on a time-specific criterion for the term "chronic" because the lifespan of aquatic animals can range from a few days to many months. The participants felt a specific time for chronic would be too restrictive.

Instead, two important characteristics of chronic sediment exposure were identified. The exposure should include a substantial portion of the life cycle or number of life stages and allow sufficient time so that contaminant steady-state is approached in the tissues. Although the terms "substantial" and "sufficient" are qualitative, they provide necessary flexibility since duration of life cycles and time to steady-state vary tremendously. This concept of chronic requires that one demonstrate, or at least convincingly argue, that the two constraints have been met.

Table 2. Characteristics of Chronic Sublethal Sediment Bioassays Suggested by the Workshop Participants

Test Duration, days (unless otherwise noted)	Sublethal Endpoint(s)*	Test Organism
28	reproduction	amphipod
14	challenge	any species
28	growth, reproduction	amphipod
30	growth, reproduction	amphipod
10-40	growth, reproduction	amphipod
20-60	growth, reproduction	amphipod
28-60	growth, reproduction	amphipod
20	growth	polychaete
28	growth	polychaete
30	growth	polychaete
20-60	growth, reproduction	polychaete
21-120	growth, reproduction	polychaete
45-60	growth, genotoxic biomarkers	polychaete
45-60	growth, genotoxic biomarkers	adult bivalve
not specified	genotoxic biomarkers	adult bivalve
2-4	development, genotoxic aberrations	larval bivalve
2-4	development, genotoxic aberrations	larval sea urchin
15 min.	bioluminescence inhibition	Microtox®

* All participants suggested survival also be reported.

Workshop Objective 2: Identify Appropriate Sublethal Endpoints

Only one endpoint in acute lethality tests is possible, percent survival. In contrast, the number of sublethal endpoints is almost infinite. They can be arranged according to three levels of biological organization: biochemical/cellular, organismic or whole animal, and populations/communities (Figure 1). The ultimate goal of most environmental protection programs is the maintenance of healthy viable populations and communities. Consequently, effects on populations and communities have the highest ecological relevance and societal importance. However, response sensitivity at this level of biological organization is generally low and predictive methods are not well developed. At the other extreme, biochemical/cellular endpoints may be quite sensitive but their ecological relevance is often unclear. Evaluating the response of individual whole organisms represents a judicious compromise between response sensitivity and ecological relevance. This approach, referred to as the surrogate toxicological approach, is used by many regulatory agencies (including the Corps) in evaluating contaminant-related perturbations.

During workshop discussions, the attendees upheld the proposition that the ability of an organism (or population of organisms) to reproduce and remain viable is of paramount ecological importance. Desirable sublethal endpoints should assess this capability. The participants identified two sublethal responses almost exclusively which fulfill this requirement: growth and reproduction (Table 2).

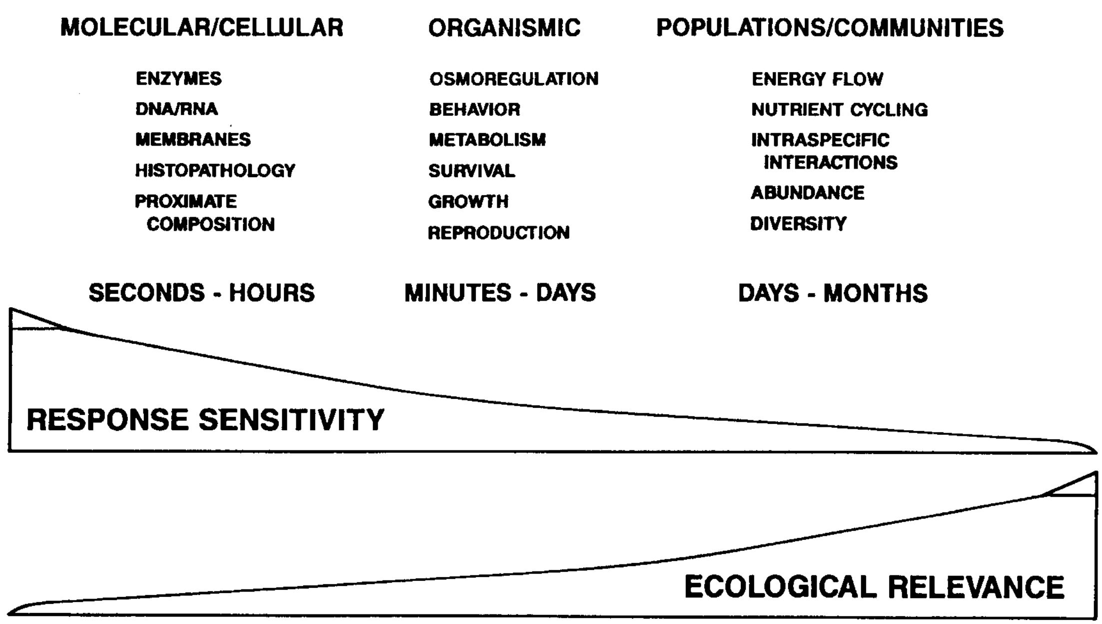

Figure 1. Levels of Biological Organization

Growth is a measure of change in mass or dimension. It can be expressed as a rate function or in absolute terms. The participants indicated that measuring growth in individuals was superior to estimating individual growth from survival and biomass. Measuring biomass alone was deemed unacceptable. Because growth and reproduction both represent competing demands on a usually limited energy source, the participants felt it was extremely important to distinguish between somatic growth and gametic growth (that is gametogenesis) in both measurement and interpretation.

The maintenance of viable populations is dependent on two factors, reproduction and survival. Sublethal measures of reproductive success would therefore seem to have greater intrinsic value than observations of growth. However, the costs associated with evaluating reproductive success are usually much greater (that is, longer and more complex experiments) than those that just measure growth. Also, most participants felt that if growth was adversely affected, reproduction would likely be affected to some degree. The question then arose, "Is growth an acceptable surrogate measure for reproduction?" After much discussion, the following conclusions were agreed upon:

1. The most desirable sublethal measure is reproduction, especially if expressed in terms of population viability.

2. If one measures only growth, the relationship between growth and reproduction *must* be thoroughly researched and quantitatively expressed.

3. The biological importance of any growth diminution must be interpreted in light of the relationship between growth and reproduction.

4. Enhanced growth due to experimental treatment is possible. If that result is observed, then treatment effects on reproductive success *must* be evaluated.

5. Measures of growth must distinguish between somatic and gametic growth.

Workshop Objective 3: Identify Appropriate Test Species

Before discussing individual species, the participants were asked to formulate a prioritized list of criteria for use in selecting appropriate test species. The ranked criteria are shown below.

1st - Intimate contact with sediment
2nd - Amenable to testing*

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* Includes the following:
  - Unaffected by nontreatment influences (for example, sediment grain size).
  - Readily available from lab cultures or field collections.
  - Reasonable cost.
  - Appropriate endpoints (that is, growth or reproduction).
  - Defined precision in control and reference response.
  - Logistically feasible.

3rd - Ecological relevance and sensitivity (tied)
4th - Economic importance

Prioritization of the first two selection criteria (intimate contact with sediment and amenable to testing) were clear choices among the participants. Ranking the next two criteria (ecological relevance and sensitivity) was more equivocal. As a result, they have been rated equally. The participants felt that the last criterion (economic importance) should be considered only when all other factors are equal. However, this criterion may become very important if, for example, a commercially important species is at clear and demonstrated risk.

Amphipods and polychaetes were the participants' main choices for test species (Table 2). Among the amphipods, chronic sublethal effects methodologies are most developed for *Ampelisca abdita* (Scott and Redmond 1989). *Ampelisca* are estuarine infaunal tube dwellers that occur from the intertidal environment down to about 60 m. They are surficial detrital feeders but must be fed an algal diet in the laboratory. The life cycle can be completed in 28-30 days at 20-25° C. Most tests have been conducted with field-collected animals. Current research is focused on the development of culture techniques, appropriate feeding rations, and improvement of the survival of lab-reared young. Also under development are a partial life-cycle test protocol (<20-day test) and a demographic population model.

Another recommended amphipod was *Grandidierella japonica*. Its distribution, life cycle, and feeding habits are similar to *Ampelisca abdita*. However, it is a much larger amphipod and constructs a membranous tube that is not as substantial as that of *Ampelisca*. It can be maintained in the laboratory on ground fish flake food. While *Grandidierella japonica* appears to be a good candidate species, test method development for this species lags behind that of *Ampelisca abdita*.

Word and others (1989) have reported that the sensitivities of *Ampelisca abdita* and *Grandidierella japonica* were similar to another amphipod, *Rhepoxynius abronius* in static 10-day bioassays. The latter species has been used extensively in acute toxicity sediment bioassays. However, it appears to be inappropriate for chronic sublethal testing. *Rhepoxynius* is an annual species. Gravid females are available only once a year. Attempts to culture this species have been unsuccessful. Attempts to culture other amphipod species (for example, *Lepidactylus* sp., *Leptocheirus* sp., and *Eohaustorius* sp.) as a prelude to chronic sublethal sediment bioassays are underway.

Development of a chronic sublethal sediment test method with polychaetes has focused on one species, *Neanthes arenaceodentata*. This polychaete is unique among the family Nereidae in that it has a nonplanktonic larval stage. Development is direct and a full life cycle can be completed in about 120 days. Cultures are easily maintained in the laboratory and organisms are widely available. *Neanthes arenaceodentata* has been used to evaluate the chronic sublethal effects (that is, growth and reproduction) of a variety of contaminants (Reish 1985). In addition, a population dynamics model has been constructed for this species (Pesch and others 1987).

Mr. John Wakeman indicated early in the workshop that growth in this species is being used as a sublethal sediment test in the Puget Sound area. However, important technical questions remain before this test can be used in the regulatory environment. Two research issues that need to be resolved are the relationship between growth and reproduction and the effects of important nontreatment factors (for example, ammonia, grain size, and feeding).

Dr. Ed Casillas described another polychaete species, *Armandia brevis*, which has recently been examined as a chronic sublethal test species. This polychaete is an obligate deposit feeder found in the shallow intertidal zone of Puget Sound and is available nine months of the year. The majority of the worm's adult growth occurs during the 20-day growth test. Current research is focused on culture techniques and the influence of nontreatment factors.

A life-cycle test using another marine worm, *Dinophilus gyrociliatus*, was described by Dr. Scott Carr. This species attains a maximum length of 1 mm and has a life cycle of 10 days at 20°C. *Dinophilus* is easy to culture and its short life cycle allows reproductive endpoints to be evaluated quickly. Because of its small size, it is not possible to test sediments directly with this organism. Instead pore water is extracted from sediments and used in the bioassay (Carr, Williams, and Fragata 1989). To obtain pore water, sediments are pressurized in a Teflon container with compressed air and the resultant effluent is filtered and frozen. Prior to testing, samples are thawed and adjusted to standard water quality conditions. The advantage of this procedure is that samples can presumably be frozen for extended periods of time and nontreatment effects such as grain size are avoided.

However, some participants were concerned that other more serious artifacts may be introduced using this procedure. For example, adsorption of contaminants during the extraction process or to the walls of the test vessel may occur. Also, the mass of contaminants may be depleted during static chronic exposures. Dr. Word presented evidence which suggests that pore water characteristics and subsequent toxicity are dramatically affected by the physical disturbance of sediment. Dr. Jay Means went on to explain that contaminant bioavailability is highly dependent on the type and amount of colloidal material in the pore water (Sigleo and Means 1990). This material is most certainly altered in pore-water extractions. Furthermore, contaminants that are tightly bound to sediment particles under hypoxic reducing conditions become mobile and available for biouptake when aerobic oxidizing conditions are imposed (Folsom and others 1988). For these reasons, it is highly unlikely that extraction and subsequent testing of pore water even remotely simulates sediment exposure. Pore-water exposure is also contrary to the highest priority criterion for organism selection identified by the workshop participants, namely, intimate contact with the sediment.

Additional candidate test species other than amphipods and polychaetes were identified by the attendees. Mr. Steven Bay described sediment bioassays he has conducted using the white sea urchin, *Lytechinus pictus*. This epibenthic urchin is a surface deposit feeder found at depths of 1-100 m off the southern California coast. It can be spawned and raised in the laboratory. Sublethal endpoints are results are dependent on the level of significance selected by the investigator (for example, $P < 0.05$ versus $P < 0.01$) as well as the experimental design (for example, number of replicates).

The participants grappled with what constitutes a biologically important difference between test and reference sediments. The discussion revolved around two issues. Most participants felt it was very important to characterize the variability of the sublethal response both in the presence and absence of reference sediment. For example, if the response normally varied by 20 percent, then a difference of at least that amount would have to occur before the results were considered biologically important (assuming statistical significance). The converse would also be true if the normal variability was small.

The second issue concerned the biological importance of the sublethal response itself. Reproduction and growth were identified as the most desirable sublethal endpoints. However, even these endpoints do not reflect potential impacts on the population--that level of biological organization which is most important ecologically and to society. The workshop participants could only identify one vector to link effects observed on individual organisms to population level impacts: demographic models.

Demographic models were originally developed to estimate probabilities, human mortalities, and future population patterns (Euler 1970 (originally published in 1760), Lotka 1925). Ecologists began using these models this century to examine life history characteristics of nonhuman species (Pearl and Miner 1935, Leslie 1945, and Ricker 1954). Marshall (1962) was the first to use demographic models in ecotoxicology. Very simply, these models integrate life history information (survival and reproduction) into population statistics such as the intrinsic rate of population increase or finite growth rate. Thus, the response of the individual becomes a population level response. The theory supporting these demographic models predicts population decline/extinction at levels defined *a priori*. For the most part, these models have not been field verified. In addition, the models make certain assumptions such as the absence of intraspecific competition and population steady-state, which are rarely met in nature. The participants felt that while demographic models represented an excellent way to express sublethal response to contaminated sediment, much more work was needed. Two areas specifically cited were verification of the models' predictive capability and improvement of experimental conditions under which the model parameters are generated.

Workshop Summary

Important points made at workshop are listed below:

1. Presently no chronic sublethal sediment bioassays have been developed to a point where they can be used by the Corps of Engineers for the regulatory evaluation of dredged material.

2. A precise chronological definition for the term "chronic" is not possible. However, participants did agree that a chronic sediment exposure should include a substantial portion of the life cycle or number of life stages and allow sufficient time so that contaminant steady-state is approached in the tissues of the exposed animals.

3. The ability of an organism (or population of organisms) to reproduce and remain viable is of paramount ecological and societal importance. Desirable sublethal endpoints should strive to assess this capability.

4. Reproduction and growth were identified as the most desirable sublethal endpoints. Where possible, results should be expressed in terms of population level impacts.

5. The relationship between growth and reproduction must be established if growth is used as the sublethal endpoint.

6. Measures of growth should differentiate between somatic and gametic growth.

7. Prioritized criteria for species selection were developed and are:

1st - Intimate contact with sediment
2nd - Amenable to testing
3rd - Ecological relevance and sensitivity (tied)
4th - Economic importance

8. Amphipods and polychaetes were identified as the most desirable test species.

9. Sublethal sediment bioassay methods are most developed for the amphipod *Ampelisca abdita* and for the polychaete *Neanthes arenaceodentata*. Additional test development is still required for both species.

10. Other candidate species were identified but considerable test development is required.

11. Statistically significant results do not necessarily imply that biologically important differences exist between test sediments and reference sediments.

12. Additional work is required to fully develop interpretive guidance for chronic sublethal sediment bioassays.

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Environmental Effects of Dredging Technical Notes

Evaluation of Sediment Genotoxicity: Workshop Summary and Conclusions

Purpose

This Technical Note summarizes the proceedings of a workshop that was held March 6-8, 1990, at the Environmental Laboratory, US Army Engineer Waterways Experiment Station. The purpose of the workshop was to gain guidance from recognized authorities for the development of sediment bioassays of genotoxicity, that is, mutagenicity, carcinogenicity, immunotoxicity, teratogenicity, and histopathologic potential. The conclusions of the workshop are being used to identify existing genotoxicity bioassays that show promise for application in evaluating sediments, to recommend modifications for testing sediments, and to help direct subsequent research and development of bioassays of genotoxicity by the US Army Corps of Engineers.

Background

The US Army Corps of Engineers (USACE) is responsible for maintaining, extending, and improving the Nation's waterways. In carrying out its mission, the USACE now dredges or regulates the dredging of more than 230 million cubic yards (cu yd) in maintenance and about 70 million cu yd in new dredging operations annually, at a cost of about $500-600 million. Additionally, about 150 million cu yd dredged by others are regulated by permits issued by the USACE. Regulatory responsibilities of the USACE in this context involve the annual review of 10,000-30,000 dredge and fill permit applications nationwide. The authority of the USACE stems from Section 10 of the Rivers and Harbors Act of 1899, Section 404 of the Clean Water Act (Public Law 92-500, as amended), and Section 103 of the Marine Protection, Research, and Sanctuaries Act ("Ocean Dumping Act," Public Law 92-532, as amended). Compliance with both laws involves, among other things, the avoidance of "unacceptable adverse impacts," and Section 103 specifically prohibits "known *carcinogens, mutagens, or teratogens or materials suspected to be carcinogens, mutagens, or teratogens by responsible scientific opinion.*" These substances are prohibited under Section 227.6, *Constituents prohibited as other than trace contaminants.* In Section 103, constituents are identified as trace contaminants if, as a result of bioassays, there is *"reasonable assurance . . . that when the materials are dumped, no significant undesirable effects will occur due either to chronic toxicity or to bioaccumulation. . . ."* The nature of tests that are mandated under Section 103, are clearly effects based. Chemical inventories of sediments can be included, but regulatory decisions regarding trace contaminants must be based on the results of bioassays.

About 5-10 percent of the maintenance and a much smaller part of the new-work dredged material is considered contaminated and is unacceptable for unrestricted open-water disposal. Both law and the public interest require that these contaminated sediments be identified and disposed of in the most environmentally responsible manner possible. In the regulatory evaluation of contaminated sediments for dredging and disposal, bioassays are used to assess acute toxicity and bioaccumulation. As yet, no bioassay methods exist that are recognized as appropriate for detection of carcinogenic, mutagenic, or teratogenic effects on aquatic organisms of dredged material proposed for open-water disposal.

Headquarters, US Army Corps of Engineers has authorized the start of a new work unit under the Long-Term Effects of Dredging Operations (LEDO) Program to develop dredged sediment genotoxicity bioassays for application in regulating disposal operations. In this context the term "genotoxicity" is used broadly to encompass all carcinogenic, mutagenic, or teratogenic effects of chemically contaminated sediments in aquatic biota, whether mediated through genetic or epigenetic mechanisms.

The purpose of the Genotoxicity Workshop was to provide guidance to the USACE for developing bioassays addressing this problem. For implementation, bioassays must be predictive, interpretable, reliable, and economical. In regulatory testing of contaminated sediments, bioassays have until now involved only whole organisms. Bioassays for sediment genotoxicity (as defined), it is anticipated, may properly include the use of biomarkers. Bioassays can be operationally defined as the exposure of a biological system (whole organisms or tissues) to test conditions to find changes caused by the test conditions. "Biomarkers" are indicators of chemical effects on biological functions that are observed at suborganismal levels. Bioassays and biomarkers may be recommended in a suite of related tests that together imply the probability of genotoxic effects. In order to be most persuasive, relationships must be describable in terms of mechanisms of action whenever possible, as opposed to being merely correlating. Ideally, the probability of effects such as cancer or lethal defects at birth will be established by the early observation of biomarkers.

Workshop participants were selected for their technical expertise in various phases of genotoxicity, as well as their familiarity with aquatic sediment processes

- What is/are the relationship(s) between sediments and tumors?

- Are there effects at the population level of organization?

- How many tumors are too many?

The research perspective of the US Environmental Protection Agency (EPA) was presented by Dr. Susan Cormier of the EPA Office of Research and Development, Environmental Monitoring Systems Laboratory (EMSL). The focus of EMSL is ecological, not human health effects. Ecological events can occur within a [redacted] biotransformation of the contaminants, biochemical and cellular changes due to the contaminants or the transformed contaminants, and certain biologically meaningful endpoints such as reproductive impairment or death. These biologically meaningful endpoints can result in ecological damage by affecting population densities of species and effecting changes at the community level of biological organization. Bioindicators can be defined as indicative of change or degradation at any level of the hierarchy. Organism level effects (for example, reproductive effects and tumors) can be monitored by the use of sentinel species or biomonitors and can signal habitat degradation. However, these changes are gross in nature. What is needed is a suite of indicators of change at the cellular or biochemical level of organization, namely, biomarkers.

Different types of biomarkers include those that indicate general health, those that indicate changes in specific organismic functions, and those that indicate nonspecific exposure to chemicals or damage due to specific toxicants. The EPA expects to use a suite of biomarkers and is currently investigating the usefulness of several of these. The tests that are selected will have broad responsiveness and sensitivity, with applicability to a wide range of species. They must also be fast, inexpensive, and reproducible.

Mr. Victor McFarland, Team Leader of the Aquatic Contaminants Team, CMRCG, presented the Corps research perspective regarding genotoxicity testing. The required attributes of a genotoxicity assay for USACE regulatory use include the ability to clearly relate the genotoxicity to sediment-associated contamination. The tests must indicate ecological relevant effects. They must also share the attributes of reliability, reproducibility, cost effectiveness, and ease of use.

A suite of biomarkers were discussed by the workshop participants. Table 2 gives the names of the genotoxicity biomarkers or assays that were discussed. The biomarkers or assays were ranked by consensus in three categories. Ease of use was discussed for each biomarker or assay and a ranking of high, medium, or low ease of use was ascribed to each. The biomarkers' ecological relevance and relevance to the USACE mission requirements regarding clear relationships to sediment toxicity were ranked as either of high, medium, or low relevance. The biomarkers and assays were also ranked on a cost-per-sample basis as either inexpensive or expensive. Comments were allowed for each bioassay or biomarker, and it is here that the developmental stage of each was discussed. Very few of the assays are ready to be tested for use in the regulatory arena, and none are ready for use without some development and interpretive guidance. The usefulness of each biomarker or assay for specific organisms was also discussed.

Conclusions

After the second day of the workshop, four participants met to draft a consensus paper outlining the results, conclusions, and recommendations of the workshop. The authors were: Dr. Richard Lee, Mr. Victor McFarland, Mr. Francis J. Reilly, Jr., and Dr. Robert Spies. The consensus draft was presented to the entire workshop on the third day of the workshop and corrections and additions were made and have been incorporated in the following paragraphs.

Biomarkers and Bioassays

To achieve the workshop objectives, the group decided that it will be necessary to develop biomarkers of exposure, integrators of effects at higher levels of biological organization, and general indicators of genotoxic potential. After evaluation of a number of biomarkers and bioassays, the participants agreed that the following showed the most potential for development in the framework of a four- to five-year program:

Biomarkers of Exposure. None of these taken singly is sufficient to prove causality of tumor development or other genotoxic effects, but all are indicative of exposure to potentially genotoxic chemicals. There are three main groups of biomarkers of exposure: proteins/enzymes, bile metabolites, and DNA adducts. The induction (increased production) of detoxifying proteins such as cytochrome P-450 or P-450-dependent enzymes (AHH, EROD, and ECOD) can be measured either via specific protein assays, antibody reactions, or by measurement of messenger RNA (mRNA). Certain genotoxic agents can be easily detected as metabolized compounds excreted in the bile of exposed organisms. Reaction products formed from genotoxic chemicals and nucleic acids can also be detected.

General Indicators of Genotoxicity. Changes in an organism's genetic integrity due to exposure to genotoxicants can be measured by mutations, chromosomal abnormalities, or DNA strand breaks. A host of bioassays exist that can be performed in either the test tube (*in vitro*), or in an intact organism (*in vivo*). Those that may be applicable to sediment genotoxicity testing are discussed in the following paragraphs.

Methods that detect chemically caused mutations may be either *in vitro* or *in vivo*: *In vitro* methods that have been applied to sediment contaminants include the Chinese Hamster Ovary (CHO) test, the Syrian Hamster Embryo (SHE) test, and the Ames Salmonella Test. *In vivo* bioassays for mutagenicity that may be applicable to contaminants in sediments are measurements of oncogene formation in an organism following exposure (for example, K-ras).

Cytogenetic techniques are methods that detect damage done to the chromosomes of cells. Chromosome anaphase or telophase abnormalities, micronuclei formation, or sister chromatid exchange can be applied either *in vitro* using CHO or SHE cell lines, or *in vivo* using medaka or other fishes, or invertebrates such as sea urchins or polychaete worms.

Methods that detect DNA damage are mainly *in vivo*, but are indicative of direct genotoxicity. These include adduct formation, minor nucleotide formation, DNA strand breaks, and unscheduled DNA synthesis (UDS) or repair.

Integrators of Genotoxic Effects. None of the biomarkers or subchronic bioassays are sufficient in themselves to predict tumors or terats. Also, many of the chemicals that are implicated in the etiology of cancer or birth defects and that may be sediment contaminants are not initiators (direct acting on genetic material). Some of the workshop scientists estimated that up to 40 percent of carcinogens in sediments are promoters that create the conditions necessary for a genetically damaged cell to become neoplastic (for example, dioxin and PCBs). For this reason the participants agreed that integrated assays of genotoxic effects should be part of the testing strategy.

Bioassays of whole animal carcinogenesis and gross abnormalities in the embryonic development of exposed organisms should be included in the testing protocol. To date, most existing work on tumorogenesis and teratogenesis has been accomplished using vertebrates. Therefore, if scientifically valid assays of integrated genotoxic effect are to be obtained in a timely manner, the ultimate objective of sediment genotoxicity testing must involve the development of a suitable fish model for direct assessment of tumor production and developmental abnormalities. The much more rapid and less costly biomarkers and subchronic bioassays listed above should at the same time be tested for responsiveness to sediment chemical contaminants. As a data base consisting of results of the short-term and the chronic test methods develops, establishing predictive relationships will be possible. Eventually, a testing suite consisting of certain biomarkers and subchronic bioassays will evolve as acceptable predictors of the real potential for genotoxic effects.

Test Species

Cytogenetic and metabolic studies, studies of neoplasm development, and studies of development abnormalities will all require early decisions on suitable species. The group thought that it would be desirable to first choose a single species that could be used throughout the country, followed by later development of regional species. Consideration was made of animals living in the water column versus benthic infaunal organisms. It was agreed that fish models are the best subjects for genotoxicity testing due to the body of literature that already exists. The Japanese medaka shows great promise because of the large amount of work that has been done to date. Medaka is very susceptible to neoplasm production as a result of exposure to carcinogens and mutagens. Additionally, the medaka can easily be cultured under a variety of environmental regimes and varying salinities.

A major disadvantage is the small size of the fish for carrying out some of the proposed biomarker assays. However, it was agreed that a first task is to determine whether a fish in the laboratory can be made to develop cancer from exposure to contaminants in sediment. This must be done reliably and consistently, or the subchronic bioassays and biomarkers will be of little use. It is realized that a larger species will be needed for other types of endpoints.

Invertebrates do not appear to be suitable for cancer studies, but may be useful for other purposes. The polychaete, *Neanthes arenaceadentata*, has been used in cytogenetic studies, and oyster larvae have been used to detect chemically induced developmental abnormalities. Invertebrate genotoxicity testing is much less developed than are tests using fish.

Research and Implementation Strategy

Many biomarker assays are not conducive to being carried out with whole sediment. Thus, extracts of sediments are often used for these tests. Extracting the genotoxic agents from sediments eliminates any possibility of making an assessment of the bioavailability of such substances. Substances that are genotoxic when extracted may not be genotoxic in the real world. Therefore, positive indications in these assays should not be the sole basis for regulatory decisions, but they should be used in combination with other more ecologically relevant whole sediment exposures to indicate potential for effects.

Various types of extraction procedures will have to be examined as surrogates for bioavailability of genotoxic compounds. Other exposure routes should be investigated. An example of the above might be exposure of benthic infauna (for which no biomarkers are available) to contaminated sediments, followed by their use as a food source for fish (for which there are ample biomarkers of genotoxicity).

Once the appropriateness of the fish models has been confirmed for some of the genotoxic endpoints with polluted sediments, for example, tumor formation, then the strategy should be to determine which of the remaining biomarkers in the above list are responsive to such exposures.

Further research should then be limited to those biomarkers that have responded well to the trial testing as discussed above. However, it should be recognized that new biomarkers will arise and should be tested. New techniques will probably become available because of the speed with which molecular biology is developing. A data base will have to be developed for the selected biomarkers and assays of genotoxicity prior to interpretive guidance. The data base should show a linkage of the effects measured with sediment contamination, as well as dose-responsiveness.

Table 1

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Table 2

Working List of Biomarkers Discussed at the Genotoxicity Workshop

Biomarker	Relevance*	Ease of Use**	Cost †	Comments
Bile metabolites	M	H-M	I	Large vertebrate
DNA adducts	H	L	E	Not in all organisms
Peroxisome proliferation	L	H	I	Needs development
Minor nucleotides		H	I	Needs development
DNA flow cytometry	H	H	E	High initial cost
Chromosome aberrations	H	M	E	Depending on species, time consuming, problems with interpretation
Oncogenes	H	M	E	Available now in some fish, for example, flounder
Single strand breaks	H	H	I	Available now in medaka
EROD (ethoxy resorufin oxygen deethylase)	M	H	I	Fish sensitive, but negative results are hard to interpret
Glutathione-S-transferase	L	H	I	Low response except in medaka, overlap with EROD
P450	L	H	I	Overlap with EROD

(Continued)

* Biomarkers were ranked on their applicability to sediment testing as well as their ecological relevance; H = highly relevant, M = moderately relevant, L = little relevance.

** Biomarkers were ranked on their projected ease of use by contract laboratory personnel; H = easy to use, M = moderately difficult, L = very difficult.

† Biomarkers and assays were ranked on the expense of performing the assay. Taken into account were cost of startup, and quality control considerations (for example, required replicates for reproducibility).

Table 2 (Concluded)

Biomarker	Relevance	Ease of Use	Cost	Comments
Oxyradical Scavenging Enzyme	L	M	E	Needs development
Stress Protein	L	M	E	Needs development
Histopathology	M	M	E	Important in liver
Embryological Development				
Gross	H	H	I	Inexpensive
Fine	H	H	E	Very expensive
CHO	H	H	I	Requires sediment extract
SHE	H	H	I	Requires sediment extract
AMES/HPTLC (High Performance Thin-Layer Chromatography)	H	H	I	Requires sediment extract, but identifies chemical class of mutagen
Fluctuation	M	H	I	Field or lab applicable
Fish Carcinogenesis	H	H	E	At least 3 month exposure required, does not work for aromatic amines
UDS (Unscheduled DNA Synthesis)	Unknown	H	E	Needs development
Macrophage*				
Other Immunocompetence*				

* Should be considered outside of genotoxicity.

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Environmental Effects of Dredging Technical Notes

Literature Review for Residue-Effects Relationships with Hydrocarbon Contaminants in Marine Organisms

Purpose

The purpose of this literature review was to identify potential residue-effects relationships involving hydrocarbon contaminants which are described in the scientific literature. That information will be used to develop guidance for interpreting the results of bioaccumulation experiments conducted in the regulatory evaluation of dredged material.

Background

Work Unit 31771, "Environmental Interpretation of Consequences from Bioaccumulation," of the Long-Term Effects of Dredging Operations (LEDO) Program is designed to generate interpretive guidance for evaluating data produced by Corps field offices or their permit applicants. This guidance results from identifying residue-effects relationships through laboratory experiments and literature reviews. Previous investigations have focused on two classes of environmental contaminants--heavy metals and chlorinated hydrocarbons. The current effort examines residue-effects relationships with hydrocarbon contaminants by a literature survey.

Hydrocarbons are an extremely complex class of environmental contaminants consisting of aliphatic, cyclic, aromatic, and heterocyclic compounds (Blumer 1976). Most of the toxicity of petroleum hydrocarbons to aquatic organisms is due to the aromatic fraction (Anderson and others 1974, Rice, Short, and Karinen 1977, Neff and others 1976). Because aromatic hydrocarbons are composed of one or more aromatic rings they are called polycyclic aromatic hydrocarbons (PAHs).

PAHs are ubiquitous environmental contaminants (Neff 1979). They are most often associated with the accidental release of petroleum, but may also originate from pyrolytic and biogenic sources. Origin notwithstanding, PAHs tend to partition into sediments due to their hydrophobic nature. Consequently, when sediments are scheduled for dredging, the bioavailability of PAHs to aquatic organisms may need to be evaluated.

In 1987, the US Army Engineer Waterways Experiment Station (WES) conducted a workshop in which experts recommended that 15 of the 16 priority pollutant PAHs should be analyzed during the regulatory evaluation of dredged material (Clarke and Gibson 1987). Naphthalene, a diaromatic hydrocarbon, was omitted from the list because the workshop participants felt it was too volatile for routine chemical analysis and did not persist in sediments. It was also felt that if high levels of naphthalene were present in sediment, its effects would be manifested as mortality in acute toxicity bioassays.

Subsequent to that workshop, a tiered testing protocol for dredged material containing hydrocarbon contaminants was developed (Jarvis and Clarke 1990). One of the tiers (Tier III) includes bioaccumulation testing using deposit-feeding organisms that have little or no metabolic capability for PAHs. For example, most fish and aquatic invertebrates rapidly metabolize PAHs while marine bivalves have little or no such capability (Lee, Sauerheber, and Benson 1972, Varanasi 1989). However, the interpretive guidance to assess the results of these sediment bioaccumulation tests is currently lacking.

Additional Information

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Approach

Published literature reporting the effects of PAHs on marine organisms was reviewed. Only investigations which examined organismic endpoints in bivalve molluscs such as growth, reproduction, behavior, and metabolism were included. Bivalve molluscs were emphasized because they have little or no biotransformation capability and they are the species of choice for assessing the bioaccumulation potential of PAHs in dredged material (Jarvis and Clarke 1990). Organismic sublethal effects are desirable endpoints for the regulatory evaluation of dredged material for reasons previously discussed (Dillon 1984). Anderson (1977) also concluded that growth, reproduction, and behavior may be the most sensitive and meaningful biological measures when the effects of petroleum hydrocarbons on aquatic organisms are being evaluated.

More than 30 technical journals and 10 data base literature search services (for example, Biosis, Pollution Abstracts, and National Technical Information Service) were used in this review. Over 100 publications were individually reviewed. For each paper included in this review, the following information was recorded: test species, exposure conditions, hydrocarbon tissue concentration, and corresponding biological effects.

Analysis

Publications which contained both hydrocarbon residue and biological effects information for marine bivalve molluscs are shown in Table 1. All investigations evaluated the effects of crude or refined oil exposed via water or sediment. Laboratory investigations slightly outnumbered field studies and all the latter focused exclusively on exposure via oiled sediment. There were no acute exposures. Laboratory exposures ranged from 28 days to 16 months. The duration of field studies ranged from 38 days to 6 years. The longer term exposures were part of monitoring studies conducted after the accidental release of petroleum. All investigations were limited to only four species of bivalve molluscs--the filter-feeding blue mussel, *Mytilus edulis*; the soft-shell clam, *Mya arenaria;* and the deposit-feeding bivalves, *Macoma* sp. and *Protothaca staminea.*

Direct and indirect measures of growth were the most popular biological endpoints. One such measure, Scope For Growth (SFG), has been studied extensively in the mussel *Mytilus edulis* (Bayne 1985). This endpoint is an instantaneous measure of growth based on the amount of calories consumed less the amount required for maintenance and lost via excretion. If there are excess calories after calculating SFG, the mussel is said to have a positive SFG. Negative SFG values are generally indicative of stressful conditions and have been strongly associated with diminished reproduction in this mussel.

Another measure of growth, Condition Index (CI), evaluates the amount of bivalve tissue relative to its shell size or volume (Lawrence and Scott 1982). The advantage of measuring growth via this endpoint is that differences among molluscs in their shell size are normalized. If the CI is reduced, then the amount of tissue relative to its shell size or volume has decreased. One underlying assumption is that a change in tissue mass occurs more rapidly than shell size. This is a reasonable assumption to make.

Tissue concentrations in most investigations are expressed as aromatic hydrocarbons--total, diaromatic, or triaromatics. The range of concentrations spans four orders of magnitude. Three investigations reported residues as total aliphatics, while two reported total hydrocarbons. To more clearly evaluate potential residue-effects relationships, those aromatic hydrocarbon tissue concentrations in Table 1 which are associated with adverse biological effects were ranked in descending order (Table 2). The highest tissue concentrations (about 200-300 $\mu g/g$) are reported as total aromatics, while the lowest concentrations (about 0.01-1.0 $\mu g/g$) are found when residues are expressed as di- or triaromatic hydrocarbons. Remaining tissue residues are in the double-digit $\mu g/g$ range.

A wide variety of analytical methods were used to analyze for hydrocarbons in bivalve tissue (Table 3). Most investigators used gas chromatography (GC) or high performance liquid chromatography (HPLC). With appropriate extraction techniques, either can be used to quantify both aromatic and aliphatic hydrocarbons. Three studies used ultraviolet absorption or fluorometry which are specific to aromatic hydrocarbons. Total hydrocarbons were analyzed in two papers using infrared spectrometry and gravimetric analysis.

Conclusions

Only a small proportion (about 10 percent of publications reviewed contained information on both the biological effects of hydrocarbons and the corresponding tissue residues in marine bivalves. Similar results were reported earlier for other environmental contaminants and aquatic biota (Dillon 1984). This small data base greatly restricts the ability to generate quantitative guidance on hydrocarbon residue-effects relationships. In addition to a small data base, variations in analytical methods reduce the effectiveness of any potential guidance.

Despite these difficulties, some general qualitative trends are apparent from the data reviewed. For example, biological effects are associated with relatively high tissue concentrations (about 200-300 μg/g) when those data are expressed as total aromatics. Lower body burdens are observed if aromatic hydrocarbons groups (for example, di- and triaromatics) are reported individually (about 1-100 μg/g) or together (about 0.01-1.0 μg/g). Moore and others (1987), in reviewing numerous papers on the effects of petroleum on field-exposed mussels, reported a similar range of effects-related tissue concentrations (1-100 μg/g) for di- and triaromatic hydrocarbons. Anderson (1977, 1979) reviewed the effects of petroleum hydrocarbons on fish, crustaceans, and polychaetes and found adverse effects at tissue concentrations of 0.2-0.6 μg/g total naphthalenes or 0.2-10.0 μg/g total aromatics.

Are these data sufficient to provide interpretive guidance for the regulatory evaluation of dredged material? Unfortunately the answer is no. The data base is too small and does not provide any specifics regarding the 15 individual PAHs on the priority pollutant list.

Two approaches for developing the needed guidance on PAHs are possible. One approach is the generation of site-specific guidance based on tissue concentrations in organisms collected in and around the disposal site environs. This so-called matrix approach assumes a local policy of "no further degradation" and that the environmental status quo is acceptable. The advantage to this approach is that numerical guidance can be generated with relative ease. There are three primary disadvantages. The field-collected organisms must be the same or closely related to the sediment bioassay test species. The toxicological significance of the bioassay results is unknown. For example, how does one interpret results where only one of the 15 priority pollutant PAHs is accumulated or 3 out of 15 or 8 of 15 are accumulated? Finally, there is no allowance for ecological interpretation. All comparisons are statistical. Tissue concentrations slightly but significantly above matrix values are treated the same as grossly elevated residues but different from concentrations slightly below but significantly different from matrix values.

The second approach is the ecotoxicological approach, which requires more effort than the matrix approach, but provides additional interpretive latitude. Here the toxicological significance of the priority pollutant PAHs is determined individually and as a group. Ideally, the model organism is the same as or closely related to the sediment bioassay test species. Next, guidance on the ecological significance of bioaccumulation is developed by generating residue-effects relationships for the individual PAHs. With these data, the ecological and toxicological importance of PAH bioaccumulation can be interpreted in a technically sound manner.

Naphthalene was not included on the experts' list of PAHs. This omission may warrant further consideration because many of the residue-effects papers reported diaromatic (naphthalenic) hydrocarbons concentrations, di- and triaromatic hydrocarbons are the major constituents in mussels from oil-contaminated environments (Boehm and others 1982, Farrington and others 1982), and di- and triaromatic hydrocarbons contribute most to the toxicity of petroleum (Neff and others 1976, Rice, Short, and Karinen 1977, Anderson and others 1974).

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Table 1

Literature Reporting Tissue Hydrocarbon Concentrations in Marine Bivalves and Corresponding Biological Effects

Reference	Contaminant	Organism	Exposure Time	Exposure Concentration	Biological Effect*	Tissue Concentration
1**	Prudhoe Bay crude oil	*Protothaca staminea*	54 days (field)	850-1,237 µg/g (sediment)	CI reduced	0.184 diaromatics 0.104 triaromatics 0.428 aliphatics
2	Prudhoe Bay crude oil	*Macoma inquinata*	55 days (laboratory)	616-1,233 µg/g (sediment)	CI reduced	1.15-5.21 total naphthalenes 0.14-0.42 aliphatics
			38 days (field) (exp. 2)	364-1,144 µg/g (sediment)	CI reduced	0.01-0.07 total naphthalenes 0.42-0.46 aliphatics
3	Prudhoe Bay crude oil	*Macoma balthica*	180 days (laboratory)	30-3,000 µg/L (water)	CI reduced; growth reduced	81-350 total aromatics 68-240 aliphatics

(Continued)

* Text of Footnote

** Numbered references are given at the end of the table; full bibliographic information is given in the References section.

NOTE: Tissue concentrations are given in micrograms per gram wet weight, unless otherwise noted.

Table 1 (Continued)

Reference	Contaminant	Organism	Exposure Time	Exposure Concentration	Biological Effect	Tissue Concentration
4	Bunker C	*Mya arenaria*	6 years after spill (field)	3,800 µg/g (sediment)	Growth reduced	267 total aromatics
5	Bunker C	*Mya arenaria*	6 years after spill (field)	5,115 µg/g (sediment)	Growth reduced	157 total aromatics
6	Diesel oil	*Mytilus edulis*	8 months (laboratory) (exp. 1)	28-125 µg/L (water)	SFG reduced	2.9-68.5 di- + triaromatics
			8 months (laboratory) (exp. 2)	28-125 µg/L (water)	SFG reduced; growth reduced	0.71-128.1 di- + triaromatics
7	Diesel oil	*Mytilus edulis*	8 months (laboratory)	30-130 µg/L (water)	SFG reduced; feeding reduced	21-24 di- + triaromatics
8	North Sea crude oil	*Mytilus edulis*	28 days (laboratory)	36 µg/L (water & food)	SFG reduced; food absorption reduced	21.8-78.3 aromatics (digestive gland)
					Oxygen consumption elevated	8.8-16.2 aromatics (remaining tissue)
3	Prudhoe Bay crude oil	*Macoma balthica*	180 days (laboratory)	30 µg/L (water)	Oxygen consumption unaffected	81 total aromatics 68 aliphatics

(Continued)

(Sheet 2 of 4)

Table 1 (Continued)

Reference	Contaminant	Organism	Exposure Time	Exposure Concentration	Biological Effect	Tissue Concentration
				300-3,000 μg/L (water)	Oxygen consumption reduced	130-150 total aromatics 160-240 aliphatics
9	No. 2 fuel oil	*Mya arenaria*	28 days (laboratory)	4,500 μg/L (water)	Oxygen consumption elevated	20-30 total hydrocarbons
10	No. 2 fuel oil	*Mya arenaria*	28 days (laboratory)	43.7-60.7 mg/L (water)	Oxygen consumption unaffected	60-145 total hydrocarbons
11	No. 6 fuel oil	*Mya arenaria*	1 year after spill (field)	11.8 mg/g (sediment)	Oxygen consumption elevated	661 total hydrocarbons
12	Diesel oil	*Mytilus edulis*	4-16 months (laboratory)	29-123 μg/L (water)	Gamete number reduced	14.7-25.4 diaromatics 3.4-7.4 triaromatics
8	North Sea crude oil	*Mytilus edulis*	140 days (laboratory)	30 μg/L (water)	Gamete production unaffected	152 aromatics (digestive gland) 22.9 aromatics (remaining tissue)
2	Prudhoe Bay crude oil	*Macoma inquinata*	55 days (field) (exp. 1)	88-1,233 μg/g (sediment)	Abnormal surfacing in sediments	0.01-0.05 total naphthalenes

(Continued)

(Sheet 3 of 4)

Table 1 (Concluded)

Reference	Contaminant	Organism	Exposure Time	Exposure Concentration	Biological Effect	Tissue Concentration
						0.06-0.14 aliphatics
3	Prudhoe Bay crude oil	*Macoma balthica*	180 days (laboratory)	30-3,000 µg/L (water)	Burrowing rate (unaffected)	81-350 total aromatics
						68-240 aliphatics

1. Augenfeld and others 1980
2. Roesijadi and Anderson 1979
3. Stekoll, Clement, and Shaw 1980, Clement, Stekoll, and Shaw 1980
4. Gilfillan and Vandermeulen 1978
5. Thomas 1978
6. Widdows, Donkin, and Evans 1987
7. Widdows, Donkin, and Evans 1985
8. Widdows and others 1982
9. Stainken 1976
10. Stainken 1978
11. Gilfillan and others 1976
12. Livingstone and others 1985

(Sheet 4 of 4)

Table 2

Effects-Level Tissue Concentrations (µg/g wet weight) of Hydrocarbons in Marine Bivalves*

Reference	Aromatics		Aliphatics	Total Hydrocarbons
3	81-350	total	68-240	
4	270	total		
5	160	total		
8	22-78	total (dig. gland)		
	8.8-16	total (rem. tissue)		
6	0.71-130	di- + triaromatics		
6	2.9-68	di- + triaromatics		
7	21-24	di- + triaromatics		
12	15-25	diaromatics		
	3.4-7.4	triaromatics		
2	1.2-5.2	diaromatics	0.14-0.42	
1	0.18	diaromatics	0.43	
	0.10	triaromatics		
2	0.01-0.07	diaromatics	0.42-0.46	
2	0.01-0.05	diaromatics	0.06-0.14	
11				660
9				20-30

* Residues are taken from Table 1 and rounded to two significant figures; see Table 1 for references.

Table 3

Methods Used to Analyze Bivalve Tissues for Hydrocarbon Content*

Analytical Method	Reference
Gas chromatography/glass capillary column	1, 2, 3
Gas chromatography/packed column	10
Ultraviolet absorbance	4
Ultraviolet absorbance with GC/MS confirmation	8
High performance liquid chromatography	6, 7, 12
Fluorometry	5
Infrared spectrometry	9
Gravimetric	11

* See Table 1 for references.

EEDP-01-25
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Environmental Effects of Dredging Technical Notes

Interim Results: The Relationship Between Sediment Organic Carbon and Biological Uptake of Contaminants

Purpose

This technical note describes testing conducted to determine the partitioning of contaminants between sediment organic carbon and sediment interstitial water, assess the effects of sediment organic carbon upon bioaccumulation of a selected polychlorinated biphenyl (PCB) and polycyclic aromatic hydrocarbon (PAH) by two organisms, and investigate the accuracy of the apparent preference factor as a predictive tool by comparing predicted uptake with actual uptake.

Background

The US Environmental Protection Agency is authorized to develop and implement sediment quality criteria (SQC) under Section 304(a) of the Clean Water Act. SQC, when promulgated, will profoundly affect US Army Corps of Engineers (USACE) dredging and disposal operations. Aquatic disposal of dredged material and selection of aquatic disposal sites will be based on SQC. Most SQC approaches currently under development involve a determination of the relationship between contaminant concentrations in sediment and biological effects on organisms exposed to the contaminated sediment. The USACE is presently investigating the link between contaminant levels in sediment and sediment geochemistry, as well as contaminant levels and effects in aquatic organisms. Knowledge of these interactions will provide the USACE with a means of evaluating the adequacy of proposed SQC approaches for estimating the potential impacts of dredged material disposal.

Additional Information

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Dr. Judith C. Pennington, (601) 634-2802, Mr. Victor McFarland, (601) 634-3721, or the manager of the Environmental Effects of Dredging Programs, Dr. Robert M. Engler, (601) 634-3624.

Introduction

Sediment organic carbon has been identified as the most important factor controlling partitioning of nonpolar organic contaminants between sediment and organisms (McFarland and Clarke 1986, McElroy and Means 1988) and between sediment and water (Karickhoff 1981). Many studies have also shown that partitioning of nonpolar organic compounds is strongly related to the octanol-water partitioning coefficient of the compound. Sediment concentrations expressed on a total organic carbon (TOC) basis have been used to predict concentrations of nonpolar organic compounds in organisms (Ferraro, Lee, and Ozretich 1990, Ferraro and others 1991, Lake, Rubinstein, and Pavignano 1987, McElroy and Means 1988, McFarland and Clarke 1986, and Rubinstein and others 1987). This method is currently being pursued by the US Environmental Protection Agency (EPA) to predict interstitial water concentrations for regulatory purposes (Brannon and others 1990).

The EPA approach to predicting interstitial water concentrations is called the Equilibrium Partitioning (EP) approach. The approach allows estimation of the concentration of a contaminant in interstitial water from sediment contaminant concentrations normalized to organic carbon. The calculated interstitial water concentrations are then compared to water quality criteria. If the predicted sediment interstitial water concentration for a given contaminant exceeds its respective chronic water quality criterion, the sediment would be categorized as contaminated by the EP procedure (Brannon and others 1990).

A procedure for investigating the relationship between sediment-bound contaminants and biota has been developed and tested (Brannon and others 1989). The procedure makes use of contaminants labelled with carbon-14 (radiotracer). Contaminants were introduced to sediments in a manner that closely simulated introduction of contaminants in the aquatic environment. Initial results showed that radiotracers provided a means for examining sediment geochemistry/bioavailability relationships that are consistent with results obtained in traditional laboratory and field studies. Results of radiotracer studies can be used to develop models for real-world conditions provided that the radiolabelled compound does not biodegrade during the course of the test, and the radiolabelled compound behaves as would the nonlabelled compound.

Results indicated that equilibration of contaminants with both sediment and the lipid pool of organisms occurred rapidly. Therefore, long exposures for bioaccumulation testing are unnecessary.

The laboratory experiments described in this note were designed to examine the relationships between sediment organic carbon and sediment interstitial water, the effects of sediment organic carbon upon bioaccumulation of a selected polychlorinated biphenyl (PCB) and polycyclic aromatic hydrocarbon (PAH) by two organisms, and the accuracy of the apparent preference factor as a predictive tool.

Materials and Methods

Three sediments were used in this study, Oakland Inner Harbor sediment from Oakland, CA; Red Hook sediment from the New York Bight, NJ; and a mixture of sediment from Brown's Lake, a freshwater lake in Vicksburg, MS, with sediment from a salt-marsh channel in Louisiana. The mixed sediment provided a test of organic matter different from that in the two saline sediments (Oakland and Red Hook).

Two organisms having different feeding modes were used in this study: clams (*Macoma nasuta*), which burrow into and deposit-feed on surficial sediments via an incurrent siphon, and worms (*Nereis virens*), which burrow into and ingest the sediment. Clams and worms were exposed to each of the three sediments amended with 4 µg of either [^{14}C] PCB 153 ([^{14}C]2,2′,4,4′,5,5′-hexachlorobiphenyl) or [^{14}C]fluoranthene per gram dry sediment weight using methods described previously (Brannon and others 1989). At all sampling periods, concentrations of PCB 153 and fluoranthene were determined in the overlying water, interstitial water, foam plugs for trapping volatiles, and clams and worms at all sampling periods. Details of the experimental procedure used in this study are given elsewhere (Brannon and others 1991).

Results and Discussion

Interstitial Water

Concentrations of free and bound (complexed with dissolved organic carbon and microparticulates) fluoranthene in interstitial water during 15 days of organism exposure are presented in Figure 1. Significant differences in both free and bound interstitial water fluoranthene concentrations for sediment containing either worms or clams were observed in all sediments tested. These differences may be a function of the manner in which the organisms disturb the sediment and process carbon or the increased bioaccumulation of organic contaminants from sediments low in organic carbon.

The ability of EP to predict interstitial water PCB 153 and fluoranthene concentrations in sediment was tested by comparing estimated K_{OC} with measured K_{OC} values. K_{OC} is the partition coefficient for sediment organic carbon and is one of the key components used in EP for predicting interstitial water concentrations. Estimated K_{OC} values were computed by substituting values of log K_{OW} (octanol/water partition coefficient) for fluoranthene (5.5) (Tetra Tech 1985) and PCB 153 (6.92) (Hawker and Connell 1988) in the equation of Karickhoff (1981) that relates K_{OW} to K_{OC}. Measured values of K_{OC} were determined by dividing the TOC

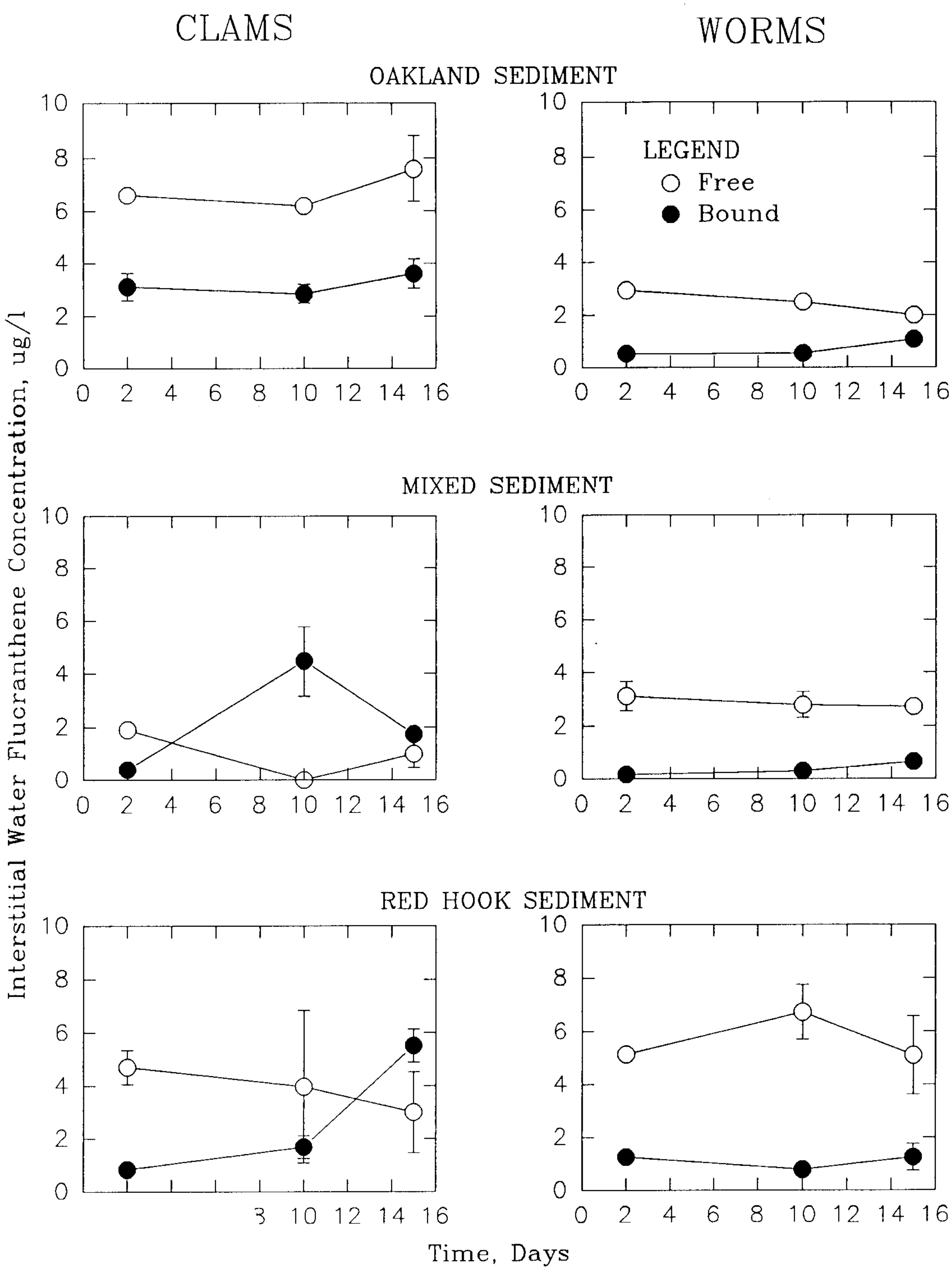

Figure 1. Free and bound (associated with dissolved TOC and microparticulates) interstitial water concentrations of ^{14}C-labelled PCB 153 and fluoranthene during bioaccumulation testing

normalized sediment concentration of either PCB 153 or fluoranthene by the free interstitial water concentration of the respective compounds.

Comparison of measured and estimated K_{OC} values for the 15-day sampling (Figure 2) showed that agreement was poor for both fluoranthene and PCB 153. Because of the log scale of the figure and the log nature of K_{OC} values, a difference of one unit is an order of magnitude difference in partitioning between water and sediment TOC. Measured K_{OC} was consistently lower than estimated K_{OC} for PCB 153, but showed no consistent pattern for fluoranthene. Therefore, EP did not provide accurate estimates of free interstitial water concentrations of PCB 153 and fluoranthene in the sediments tested. Such inaccuracy could result in sediment categorizations that are inconsistent with the actual environmental impacts of the dredged material.

An additional problem was identified that may frequently occur in sediment from industrial areas. TOC concentrations measured using whole sediments were:

Sediment	**Percent TOC**
Oakland	1.08
Mixed	2.84
Red Hook	4.63

Investigation of the Red Hook sediment revealed numerous small lumps of shiny black coal. Sorption of PCB and fluoranthene on such surfaces should be minimal in comparison to sorption on sediment organic matter because of the tremendous difference in surface area. Passage of the sediment through a No. 40 mesh sieve to remove coal prior to TOC determination resulted in a 37 percent reduction in sediment TOC to 2.92 percent. This TOC concentration was then used to compute measured K_{OC} and apparent preference factors because of the insignificant role of the coal fraction as a sorptive phase for fluoranthene or PCB 153 compared to other forms of sediment TOC. The TOC without coal was used to generate the Red Hook data (Figure 2).

Measured K_{OC} values for PCB 153 (average log $K_{OC} = 6.34$) were in better agreement with the estimated K_{OC} value of 6.5 using the sediment humic + fulvic acid fraction rather than sediment TOC. Measured K_{OC} values for fluoranthene (average log $K_{OC} = 5.7$) consistently exceeded the estimated K_{OC} value (5.09). These results demonstrated the possible utility of the humic + fulvic acid fraction and suggest that fractions of sediment organic matter other than TOC are potentially useful for predictive purposes. However, more research is needed before such approaches can be made useful.

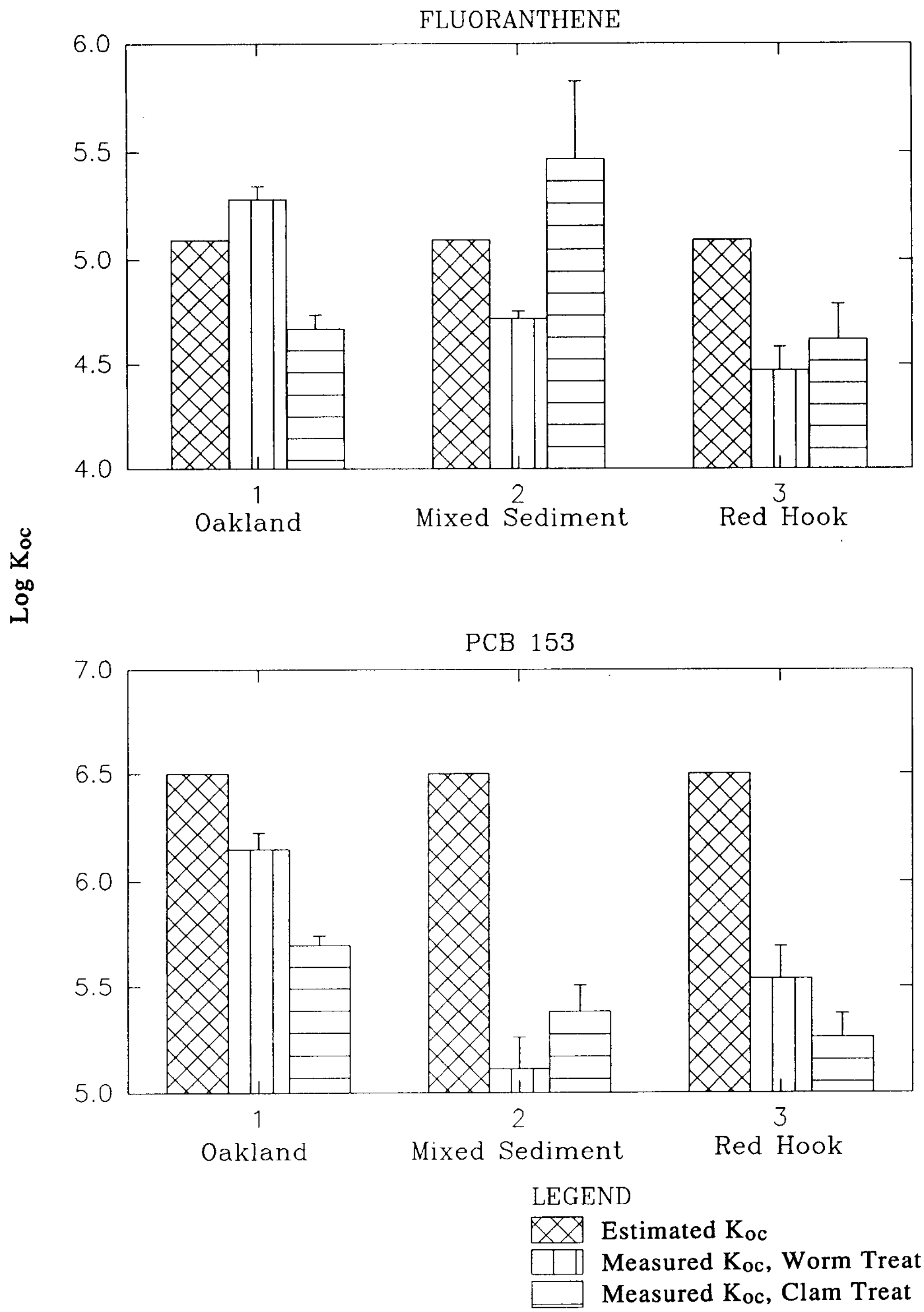

Figure 2. Estimated and measured K_{oc} values following 15 days of incubation

Relationship between Interstitial Water and Overlying Water

Results indicated that volatilization from the overlying water can be a loss pathway for both PCB 153 and fluoranthene during bioassay experiments. Volatile losses from bioassay tests were significantly higher in the worm treatments than in clam treatments for both PCB 153 and fluoranthene in all but Red Hook sediment, where losses for the two treatments were comparable. Volatile losses for the three sediments averaged 0.21 percent of the total fluoranthene and 0.17 percent of the total PCB 153. Possibly as a result of such losses, interstitial water and overlying water concentrations were not significantly correlated, except for fluoranthene in the clam treatment ($r = 0.72$, $p < 0.05$) (Brannon and others 1991). Consequently, the linking of interstitial water concentrations to biological effects in the overlying water column may be difficult.

Apparent Preference Factor

Apparent preference factors (APFs) calculated at 10- and 15-day sample intervals for both clams and worms were in close agreement in all sediments except Oakland Harbor (Brannon and others 1991). Slow stabilization of Oakland Harbor PCB 153 and fluoranthene APFs was unexpected, because only 10 days had been required for PCB 52 to attain steady state APFs in Oakland Harbor sediment during a previous study (Brannon and others 1989). Tissue concentrations and interstitial water concentrations in worm and clam treatments were unrelated (Brannon and others 1991).

The values of 15-day APFs (Table 1) were similar to those for other empirical determinations reported in the literature for both field and laboratory studies and studies using both spiked and "naturally" contaminated sediment (Figure 3). The observations in this study are not only consistent with, and supportive of, results of a previous study (Brannon and others 1989), but also indicate good correspondence between laboratory results using spiked sediments and results

Table 1
APFs (and Associated Standard Errors) for Clams and Worms Following 15 Days of Exposure to Sediment Containing Fluoranthene or PCB 153

Sediment	Clams		Worms	
	Fluoranthene	PCB	Fluoranthene	PCB
Oakland	3.77 (3.4)	4.79 (1.77)	0.8 (0.17)	4.78 (0.65)
Mixed	2.47 (0.79)	Samples lost	1.05 (0.18)	1.41 (0.29)
Red Hook	0.55 (0.84)	0.49 (0.19)	3.31 (1.27)	4.79 (2.50)

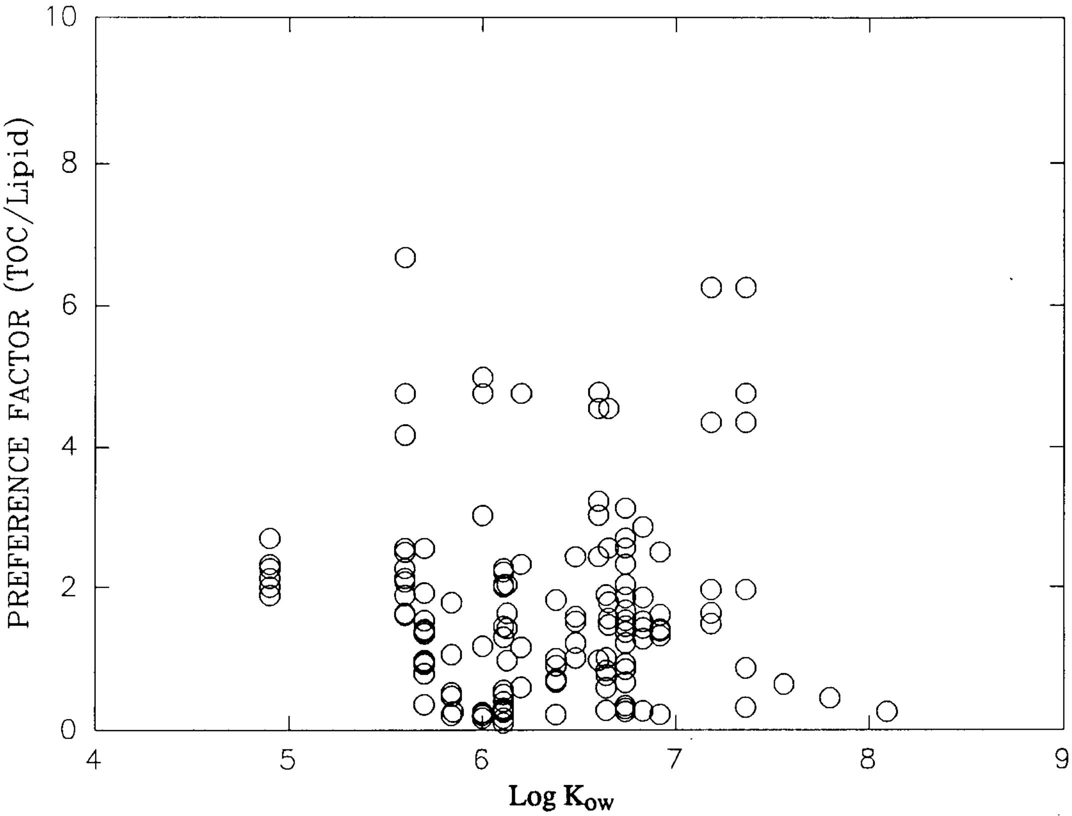

Figure 3. Literature preference factor values (Bierman 1988, Clarke, McFarland, and Dorkin 1988, Ferraro and others 1990, Ferraro and others 1991, Lake, Rubinstein, and Pavignano 1987, McElroy and Means 1988, Pruell and others 1990, and Rubinstein and others 1987)

with field-contaminated sediments and biota. In addition, results of Brannon and others (1991), McElroy and Means (1988), and Brannon and others (1989) showed rapid attainment of constant preference factors, implying that long exposures for the purposes of bioaccumulation testing are not necessary for PCBs and fluoranthene.

Summary of Findings

Values of K_{OC} measured using free interstitial water concentrations of fluoranthene and PCB 153 were either substantially higher or lower than estimated K_{OC} values. The data indicated that concentrations of PCB 153 and fluoranthene in interstitial water will be either overestimated or underestimated when using equilibrium partitioning, estimated K_{OC} values, and TOC. In a regulatory framework, predictive methods with a high degree of uncertainty are not a good foundation upon which to base pass/fail decisions. The geochemistry affecting interstitial water concentrations must be better understood before rigid regulatory criteria based upon predicted interstitial water concentrations are promulgated.

The sediment humic + fulvic acid fraction was investigated as a method to normalize sediment concentrations and predict interstitial water concentrations.

Measured K_{OC} based on the sediment humic + fulvic acid fraction were in close agreement with estimated K_{OC} values for PCB 153, but not for fluoranthene. These results demonstrated the potential usefulness of examining discrete fractions of sediment TOC as a means of normalizing sediment concentrations, but also indicated that much work remains to be done in this area. Interstitial water and overlying water concentrations were not significantly correlated, except for fluoranthene in the clam treatment. These results demonstrate the difficulty in linking interstitial water concentrations to biological effects in the overlying water column.

Bioaccumulation of PCB 153 and fluoranthene by worms and clams was observed in all sediments. Even though tissue concentrations increased as time of exposure increased, APF values showed that steady state was reached between sediment-bound contaminants and organism lipid pools. No relationship was found between tissue concentrations of worms or clams and interstitial water concentrations of contaminants. This result suggests that interstitial water may not be the primary source of contaminant exposure for sediment-associated organisms.

The APFs for PCB 153 and fluoranthene in worms and clams were in close agreement with field and laboratory values reported in the literature. These results imply that long exposures for bioaccumulation testing are not necessary for PCBs and fluoranthene. The presence of coal in the Red Hook sediment demonstrated that care must be exercised when using TOC values for sediment from industrial areas. However, the use of sediment TOC in conjunction with partition coefficients, such as APFs, is a viable approach for predicting bioaccumulation of nonpolar organic contaminants by infaunal organisms.

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Environmental Effects of Dredging Technical Notes

Data Base for Polychlorinated Dioxins and Polychlorinated Furans

Purpose

This note provides initial information on the development of a computerized data base concerning dioxin and polychlorinated furans in aquatic media including sediments, animals, and bioassay results. The ultimate goal of compiling this data is to provide Corps elements with numerical and descriptive guidance so sediment, tissue, and bioassay results concerning dioxins and polychlorinated furans in the environment can be related to the potential for biological and environmental effects.

Background

The aquatic disposal of dredged material is regulated under two federal statutes: Section 404(b)(1) of the Clean Water Act, as amended (PL 92-500) and Section 103 of the Marine Protection, Research, and Sanctuaries Act, as amended (PL 92-532). The US Army Corps of Engineers (USACE) is responsible for ensuring that sediments are dredged and disposed in a manner that will not have an unacceptable adverse impact on the environment.

Over the past few years sediments in several areas slated for dredging have been demonstrated to have small, but measurable amounts of dioxins and polychlorinated furans present as contaminants. At least one dredging project was delayed because dioxin was detected by the US Environmental Protection Agency in a nearby paper plant settling pond. The mention of dioxin has become enough to cause public outcry and endanger the future of some dredging projects.

The extremely high cost of dioxin and polychlorinated furan determination in environmental samples coupled with the vanishingly small quantities of these analytes present in most environmental samples simply do not allow cases of suspected dioxin and polychlorinated furan contamination to be routinely evaluated like other contaminants.

Dioxins are a class of chlorinated, two-ring compounds. The word dioxin is commonly used interchangeably to mean either the class of compounds known as dioxins or, more commonly, to mean the most toxic member of the class of compounds, 2,3,7,8-tetrachloro dibenzo-*p*-dioxin. Polychlorinated furans are a closely related class of compounds that are also chlorinated, two-ring compounds. Figure 1 shows a polychlorinated dibenzodioxin (PCDD), and Figure 2 shows a polychlorinated dibenzofuran (PCDF). Chlorine atoms are substituted at each of the numbered sites on the two molecules, and the resultant name of the compound is given by the substitution numbers and a prefix designating the total number of substitutions. For example, 2,3,7,8-tetrachloro dibenzo-*p*-dioxin is substituted with four chlorines (tetra-) at the sites numbered 2, 3, 7, and 8. The names of the compounds are often abbreviated to the substituted sites, a hyphen, and the first letter of each word in the compound. For example, **2,3,7,8-t**etrachloro **d**ibenzo-*p*-**d**ioxin is abbreviated to 2,3,7,8-TCDD and **2,3,6,7,8-p**entachloro **d**ibenzo**f**uran is abbreviated to 2,3,6,7,8-PCDF. The chemicals that make up the class of compounds known as dioxins are all known as dioxin congeners. There are 75 poly**c**hlorinated **d**ibenzo**d**ioxin (PCDD) congeners. Likewise, the members of the class of compounds known as **p**olychlorinated **d**ibenzofurans (PCDF) are known as polychlorinated furan congeners. There are 135 polychlorinated furan congeners (Rappe 1984).

The relative toxicities of PCDD and PCDF congeners are the subject of current controversy. There is some agreement that congeners of both classes that are substituted in the 2, 3, 7, and 8 positions are toxic or more toxic than congeners not substituted in those positions. Congeners not substituted in the 2, 3, 7, and 8 positions are deemed less toxic or nontoxic. The relative toxicities of these compounds are discussed by McFarland and others (in preparation).

PCDDs and PCDFs are the product of incomplete combustion in the presence of chlorine or the product of certain industrial chlorination processes. They are released into the environment via industrial fugitive emissions or by the application of contaminated herbicide (Miller, Norris, and Hawkes 1973). It is clear that some natural mechanism for the synthesis of PCDDs and PCDFs also exists, particularly octachloro dibenzo-*p*-dioxin (Hashimoto, Wakimoto, and Tatsukawa 1990).

Additional Information

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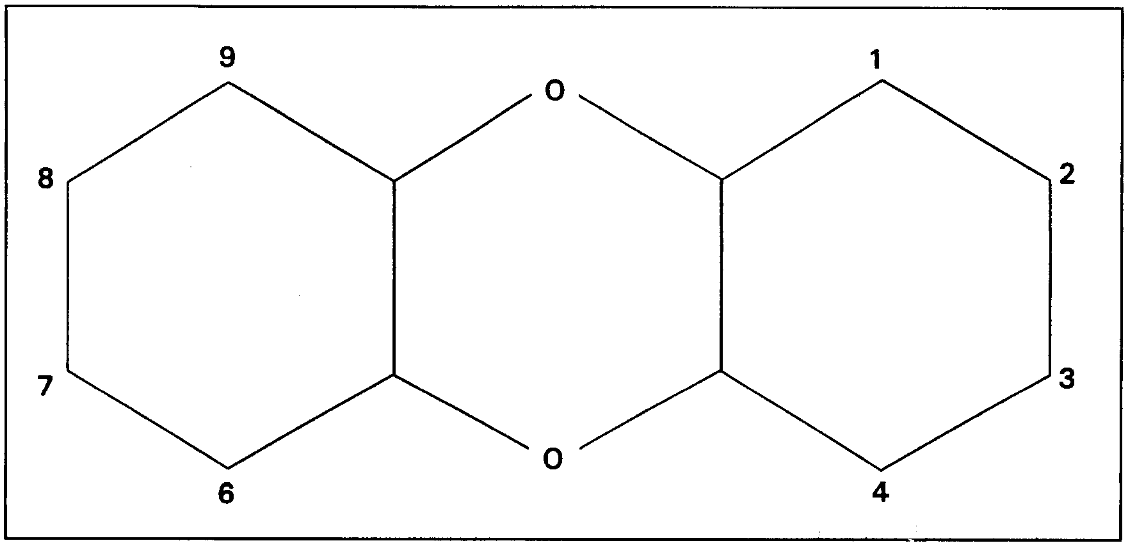

Figure 1. Polychlorinated dibenzo-p-dioxins (PCDDs); chlorine atoms are substituted at one or more of the numbered locations on the molecule

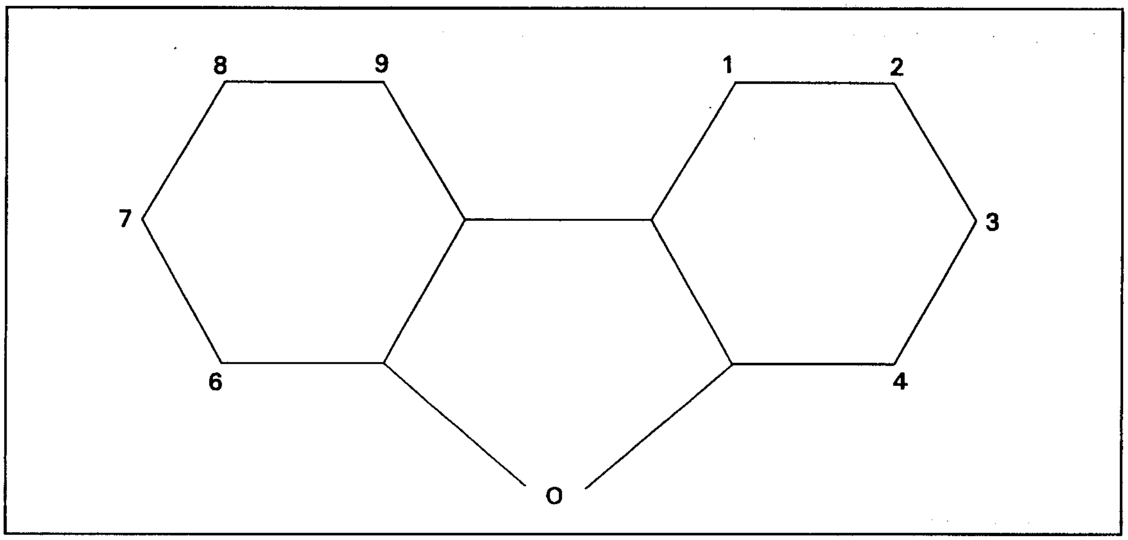

Figure 2. Polychlorinated dibenzofurans (PCDFs); chlorine atoms are substituted at one or more of the numbered locations on the molecule

Approach

In order to construct this data base the published literature was reviewed. A good deal of literature is available concerning PCDDs and PCDFs in general and the mammalian health effects of these classes of compounds. This data base, however, was specifically limited to references of an aquatic nature. More than 20 technical journals were reviewed in their entirety for the information contained in the data base. The papers selected for inclusion in the data base were used to construct nine tables, the contents of which are:

- Table 1a - Dioxin Levels in Selected Sediments
- Table 1b - Furan Levels in Selected Sediments
- Table 2a - Dioxin Residues in Bioassay Exposed Organisms
- Table 2b - Furan Residues in Bioassay Exposed Organisms
- Table 3a - Dioxin Residues in Field-Collected Organisms
- Table 3b - Furan Residues in Field-Collected Organisms
- Table 4a - Dioxin Levels in Fish-Eating Birds and their Eggs
- Table 4b - Furan Levels in Fish-Eating Birds and their Eggs
- Table 5 - Dioxin Residues Associated with Known Biological Effects

Analysis

The tables will be published in their entirety in a Miscellaneous Paper currently being prepared, but are too extensive to be included in this note. Significant information, however, can be gleaned from the tables that could be of immediate benefit to USACE field elements. Summaries of the information contained in these tables have been prepared. The information contained in Table 5 has been summarized and commented on in a technical note (Gibson and Reilly 1992). The information gleaned from all the tables and some other sources has been included in a Miscellaneous Paper dealing with Toxic Equivalency Factors (McFarland and others, in preparation).

Tables 1a and 1b summarize data published concerning PCDD and PCDF levels in selected sediments and other aquatic substrates including some data concerning fly ash settling ponds, sewage sludge, and other types of sludges. There are over 750 data arranged by congener with individual data concerning each congener's concentration, the location of the collection, and the reference from which the data were obtained. In some cases, the detection limit is given, particularly when a nondetect value was reported. Some data are grouped by congener class (for example, the total of all hexa congeners). Detection levels reported for nondetect data range from 0.2 part per trillion (pptr) for 1,2,3,7,8-PCDD from the Baltic Sea to 88 pptr for sediments from the Saigon River in South Vietnam. The lowest reported value of 0.001 pptr was for total PCDDs in treated sludge collected near Ontario. The highest reported sediment value was over 99,000 pptr OCDD in sediments collected near Sheffield, United Kingdom.

Tables 2a and 2b summarize data published concerning PCDD and PCDF levels in organisms that were exposed to materials contaminated with PCDDs and PCDFs under bioassay conditions. Over 140 individual measurements were reviewed. The data base is arranged by congener and by organism. The level of exposure and the tissue residue of the exposed organism is given, as are some salient features concerning the exposure. Data exist concerning the uptake of PCDD in freshwater fish and invertebrates and saltwater invertebrates. The only PCDF data currently available are for carp exposed to fly ash.

Several interesting facts can be extracted from this data set. Under a variety of exposure times and conditions, the bioaccumulation of various PCDD and PCDF congeners was assessed. In no reported case did the organism ever bioaccumulate a higher concentration of contaminant than the concentration of the substrate to which they were exposed. Other information from this data set seems to indicate that because of differences in bioavailability of the various congeners, it would be inadvisable to apply Toxic Equivalency Factors (TEQ Methodology) to sediment concentrations to perform an estimation of environmental risk.

Tables 3a and 3b summarize the published data concerning PCDD and PCDF levels in field-collected organisms from a variety of contaminated and relatively uncontaminated areas. Over 1,500 individual measurements are reviewed. The data base is arranged by individual congener, although some data are given as summations of congener groups (for example, total hexa). The specific organism and tissue (if available) as well as the concentration and collection information are given for each measurement. Only a few of the studies reported give environmental levels that resulted in the reported tissue residues. From the scant information concerning the relationship between exposure and resultant tissue concentration, it is possible to conclude that the organisms nearly always show higher tissue residue concentrations than reported for water at the collection site, but never show higher tissue concentrations than sediment concentrations reported for the collection site.

Tables 4a and 4b summarize the published data concerning PCDD and PCDF tissue residues reported for fish-eating birds and their eggs. Over 30 individual measurements are given. Most of the measurements are for 2,3,7,8-TCDD and 2,3,7,8-TCDF. Some measurements summarize all the PCDD or all the PCDF congeners, but no studies summarized report on any other congeners for either PCDDs or PCDFs. Information concerning exposure levels was not given in any of these studies. The highest level reported for total PCDDs was 214 pptr in a night heron collected from Lake Michigan, Wisconsin, but the highest total for 2,3,7,8-TCDD was only 59 pptr, in a night heron collected near Green Bay, Wisconsin. The highest value reported for 2,3,7,8-TCDF was from a kingfisher collected near Sheboygan, Wisconsin. The highest total PCDFs level reported in the literature was 53 pptr reported in a night heron collected near Lake Michigan, Wisconsin.

Table 5 summarizes the data concerning PCDD residues in organisms associated with specific biological effects. Published data linking effects with known residues rather than nominal doses are limited to two studies. Both studies are limited to data concerning the 2,3,7,8-TCDD congener. It is not currently possible to develop an effects threshold or a no-effects concentration. Further information concerning residue levels and effects can be found in Gibson and Reilly (1992).

Summary

The PCDD and PCDF Data Base was developed to provide information concerning the environmental concentrations and effects of PCDD and PCDF contaminated sediments. Nearly 2,500 individual PCDD or PCDF measurements have been reviewed, summarized, and included in the data base. This information is available to aid field elements by providing specific numerical guidance concerning contamination by PCDDs and PCDFs and the potential for effect due to dredging and disposal of sediments that may be contaminated with PCDDs or PCDFs.

It appears clear from several lines of evidence that organisms exposed to sediments contaminated by PCDDs or PCDFs will not bioaccumulate levels of PCDD or PCDF higher than the concentrations of the sediments to which they are exposed. Equally clear is the fact that there is a real lack of the information necessary to make informed decisions regarding the biological consequences associated with body burdens of PCDDs and PCDFs. The PCDD and PCDF Data Base will be published in a later paper with a complete analysis of the data.

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Environmental Effects of Dredging Technical Notes

A Computer-Assisted Expert System for Interpreting the Consequences Of Bioaccumulation In Aquatic Animals (*COBIAA*)

Purpose

This technical note describes a prototype expert system being developed to assist managers and scientists in the interpretation of bioaccumulation test results and their potential effect on the disposal of dredged material. This is a microcomputer (MS-DOS™) based system, operating in the Microsoft Windows™ environment.

Background

Two types of sediment bioassays may be conducted in the regulatory evaluation of dredged material: toxicity tests and bioaccumulation tests. *COBIAA* (for *Co*nsequences *O*f *B*ioaccumulation *I*n *A*quatic *A*nimals) is an expert system being developed to help interpret results of bioaccumulation tests, which incorporates toxicity data in the final decision. Because regulatory decisions are based on both types of bioassays, this note includes a brief overview of the two types.

Additional Information

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Interpreting Toxicity Tests

Sediment bioassays designed to evaluate the toxicity of dredged material are called toxicity tests. Typically, survival or some biologically important sublethal endpoint, such as growth or reproduction, are measured in appropriate test species following laboratory exposure. All sediment toxicity tests are conducted to meet two objectives: evaluate material with high precision and accuracy and predict potential biological impacts in the field.

Most research and development has focused on the first objective. Little effort has been devoted to field verifying sediment bioassays. Currently, sediment bioassays cannot be considered precise predictors of biological impacts in the field. Testing the requisite number of animals from a variety of trophic levels under simulated realistic field conditions is too time- and resource-intensive, especially for a regulatory program. Rather, the approach adopted jointly by the U.S. Army Corps of Engineers and the U.S. Environmental Protection Agency (EPA) is to evaluate worst-case scenarios using appropriate test species. Animals are spatially confined under rigorously defined exposure conditions to the project sediment for a period of days to weeks. Survival, growth, and/or reproduction are measured and compared to a reference sediment treatment. Results are analyzed statistically and an evaluation is made regarding the *potential* for unacceptable adverse impact due to the project dredged material. Statistical significance cannot be used to *predict the occurrence* of field impacts. Rather, the results of sediment toxicity tests are used to *project the possibility* and the relative magnitude of potential impacts.

Reference Sediment Approach

The second way to facilitate projections of potential field impacts from sediment bioassays is to make *relative* rather than *absolute* comparisons. This is accomplished by using a *reference* sediment. The reference sediment is selected to simulate, as closely as possible, the disposal site environs in the absence of dredged material disposal. In laboratory experiments, the biological response in the project dredged material is compared to that in reference sediment. If results are indistinguishable, one infers that the potential for unacceptable adverse impacts is low to nonexistent.

Interpreting Bioaccumulation Tests

Bioaccumulation tests are conducted to demonstrate whether environmental contaminants have the potential for moving from the sediment matrix into aquatic animals. As with toxicity tests, *relative* comparisons are carried out using a reference sediment. If there are no significant differences in bioaccumulation between project and reference sediments, one concludes that the potential for bioaccumulation does not exist. If, however, significant bioaccumulation is observed, one must interpret the importance of the resultant tissue residues. To date, this interpretation has varied widely and lacked a sound technical framework. The *COBIAA* computer-assisted expert system will help provide this framework.

Interpreting bioaccumulation data is generally more difficult than interpreting results of sediment toxicity bioassays, involving a weighing and balancing of many factors, some intuitive and some quantitative. All these evaluations are carried out in the context of two categories of potential target receptors: aquatic organisms and humans.

Need for an Expert System

Very few people have the knowledge and expertise to interpret bioaccumulation and toxicity data to reach a conclusion concerning the appropriate disposal of dredged material. This interpretation of the data is approached by different people in varying ways, creating a lack of consistency. *COBIAA* is based upon the procedure followed by an expert in this field to assess this type of data. Using *COBIAA* will allow managers and scientists to follow the same determination process in spite of not having the necessary experimental background needed to make these kinds of decisions. To provide flexibility, *COBIAA* will permit defaults to be modified, but will require the user to input a justification for these changes. *COBIAA* does not attempt to provide the definitive answer to the question of whether a particular sediment should be dredged or whether the sediment is acceptable for ocean disposal, but adds to the knowledge base needed to make this decision.

Figure 1 shows the underlying structure of the decision process used by *COBIAA*. Each step of the hierarchy represents the attributes or categories that *COBIAA* uses to reach a decision. None of the basic attributes (T1, T2, B1, B2, B3, B4, and B5) is more or less important than the others in the decision process. The

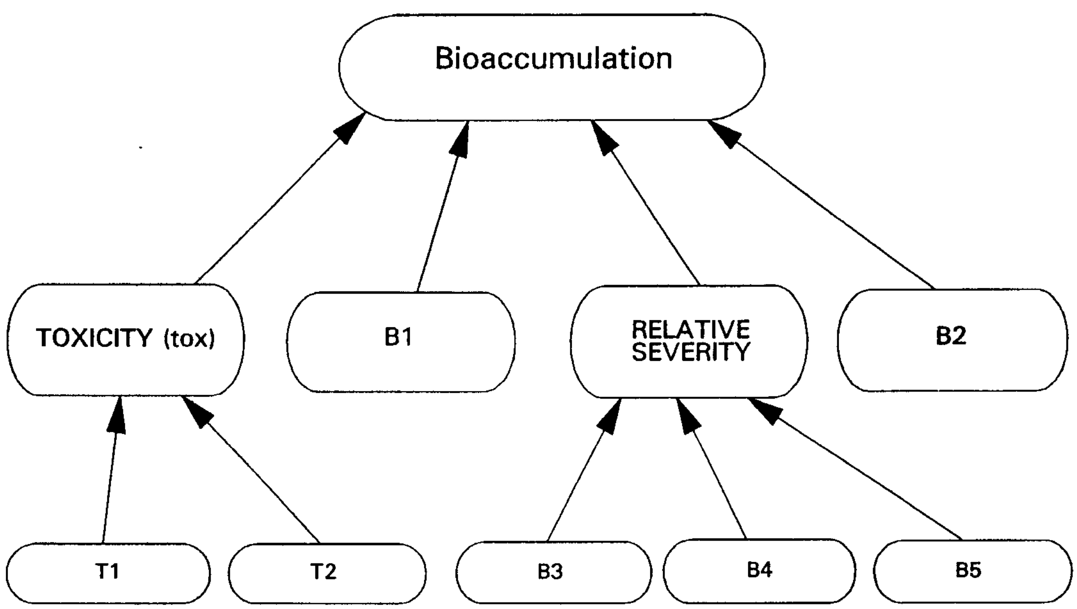

Figure 1. ***COBIAA* attribute hierarchy**

other two attributes, toxicity and relative severity, are used as intermediate steps in the decision process to pool the data from several basic attributes. This pooling of data from several attributes allows them to change the impact each individual attribute has on the decision.

COBIAA is most properly viewed as a decision support system (DSS), with an expert system component, for solving an ill-defined management problem as described previously. As a DSS, *COBIAA* is made up of three components: a compilation of rules and attributes, a set of data, and a user interface.

Overview of the Attribute Hierarchy

The basic attributes are divided into two types, those based on toxicity data and those based on bioaccumulation data. *COBIAA* is designed to interpret bioaccumulation data, but in so doing uses the toxicity data to assist in this interpretation.

- Attribute T1 — number of different species showing toxicity when exposed to the same test sediment. This attribute attempts to take into account the sensitivity levels of various species to contaminants. Possible values of the T1 attribute are:

Range of Values	Level of Concern
0	Low
1	Medium
> 1	High

- Attribute T2 — magnitude of toxicity above reference caused by the same test sediment used in attribute T1. If the toxicity levels of the test sediment are only slightly above reference, there is less cause for concern than if they are twice the reference. Possible values of the T2 attribute are:

Range of Values	Level of Concern
0 percent	Low
1-20 percent	Medium
> 20 percent	High

- Attribute B1 — number of phylogenetic groups showing statistically significant bioaccumulation relative to reference levels. This attribute attempts to account for varying levels of sensitivity to bioaccumulation in different taxa of animals. Possible values of the B1 attribute are:

Range of Values	Level of Concern
< 2	Low
2	Medium
> 2	High

- Attribute B2 — proportion of contaminants of concern bioaccumulated to concentrations statistically exceeding reference levels. This attribute attempts to determine the severity of the bioaccumulation problem. Possible values of the B2 attribute are:

Range of Values	Level of Concern
< 10 percent	Low
10-50 percent	Medium
> 50 percent	High

- Attribute B3 — magnitude of test sediment bioaccumulation above reference levels. This is a different measure of the severity of contaminant bioaccumulation. Possible values of the B3 attribute are:

Range of Values	Level of Concern
0-20 percent	Low
21-100 percent	Medium
> 100 percent	High

- Attribute B4 — toxicological importance of contaminants bioaccumulated from the test sediment to concentrations exceeding reference levels. The contaminant rankings are based on EPA Water Quality Criteria (Lee and others 1991), which indicates that certain contaminants are of more concern if bioaccumulated than others. Contaminants not listed in the EPA table are not assigned an attribute value and therefore not used by *COBIAA* in the decision process for those contaminants. Possible values of the B4 attribute are:

Range of Values	Level of Concern
6	Low
3-5	Medium
1-2	High

- Attribute B5 — magnitude of contaminant concentrations (micrograms per gram wet weight) in tissues of test organisms. This attribute uses the actual level of tissue residues as an indication of the severity of concern. Possible values of the B5 attribute are:

Range of Values	Level of Concern
< 0.1	Low
0.1-1.0	Medium
> 1.0	High

Data Requirements

The data used by *COBIAA* are readily available for most dredging projects and use both test and reference site data. As alluded to above, these data include: animals tested, contaminants analyzed for, toxicity data (number of animals showing toxicity), and bioaccumulation tissue residue concentrations. Default data files supplied with *COBIAA* contain any available toxicological ranking of contaminants and U.S. Food and Drug Administration (FDA) criteria (Lee and others 1991).

COBIAA's data are organized into data sets. A toxicity data set is defined as an animal name, the reference toxicity, and the test toxicity. A bioaccumulation data set is defined as an animal name, a single contaminant, the reference tissue residue level, and the test tissue residue level of that contaminant. Each project or site will consist of multiple data sets encompassing as many animals and contaminants for which data exist. These data sets are stored in a file on the computer's hard disk and may be edited at any time.

COBIAA's User Interface

The *COBIAA* user interface is an interactive environment within which the user accesses the program files, enters and edits data, and analyzes those data using rules contained within the expert system component. The *COBIAA* prototype incorporates a graphical user interface and a mouse to navigate through the program menus easily. Because the user interface will likely change significantly from this prototype version to the production version of *COBIAA*, the specifics of system design and development will not be discussed in detail. A sample data entry screen, shown in Figure 2, provides an indication of the type of user interface currently employed. A Microsoft Windows™ based program, *COBIAA* requires the minimum computer hardware needed to run Microsoft Windows™, an 80386 microcomputer with hard disk, VGA monitor, 4 megabytes of memory, and a mouse. A printer is used if available to generate printed output of the decision and logic used to arrive at a decision. The expert system portion of *COBIAA* was developed using CLIPS, an expert system shell developed by the National Aeronautics and Space Administration.

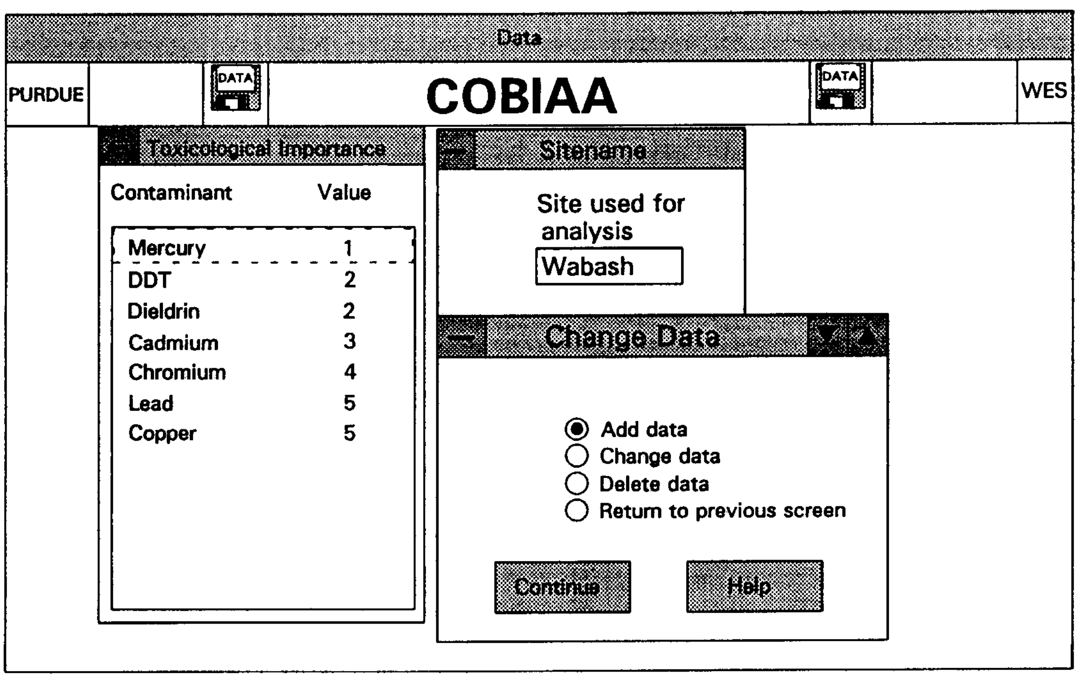

Figure 2. Example data entry screen from *COBIAA*

How *COBIAA* Works

COBIAA uses the data provided from bioaccumulation and toxicity tests to evaluate the potential for unrestricted disposal, disposal with restrictions, or whether not enough information is available requiring a regional authority decision (RAD) (Lee and others 1991). This evaluation process is accomplished by looking at the data in discrete sections, comparing them to available default values (for example, FDA criteria) and comparing the test data to the reference data. Each data set is evaluated in turn and each attribute or category is determined according to preset rules. Each attribute is assigned a value depending upon these rules. Based upon the previously defined conditions, the attributes (B1, B2, B3, B4, B5, T1, and T2) are assigned a value of low, medium, or high. The two toxicity attributes (T1 and T2) are then combined into a single toxicity attribute (tox). Similarly, attributes B3, B4, and B5 are combined to create the relative severity attribute. The toxicity, relative severity, and B1 and B2 attributes are then combined using another set of rules to reach the final decision.

The final decision reached by *COBIAA* will be one of three conclusions. If the contamination in the dredged material appears to be of little concern, then *COBIAA* will recommend disposal with no restrictions. If the material appears to be of high concern, than *COBIAA* will recommend disposal with restrictions. This option includes no disposal as well as other possible restrictions (Francingues and others 1985). The third possible conclusion is that there is not enough information available to select one of the first two conclusions and a RAD is required. This RAD may be to select conclusions 1 or 2 or may require that more information (for example, new testing) be provided and resubmitted to *COBIAA*. Dillon and Lutz (1991) provides more information concerning the types of decision categories that fall within the three conclusions presented here.

Status of *COBIAA*

The software is currently in the prototype development stage. A working version exists, but is being constantly modified based upon feedback from the expert and several test users. The prototype is anticipated to be ready for field testing during the second quarter of FY 93.

Conclusions

COBIAA, a decision support system, is being designed to provide a consistent and easy-to-use method for interpreting bioaccumulation tissue residue data as applied to dredged material disposal. Following the procedures set forth by an expert in this field will enable the user to analyze toxicity and bioaccumulation data to determine possible disposal options. *COBIAA* will provide a consistent nationwide methodology for interpreting bioaccumulation data.

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Environmental Effects of Dredging Technical Notes

Risk-based Testing of Dredged Material for Aquatic Disposal Evaluations

Purpose

This technical note describes a risk-based framework for testing and evaluating dredged material scheduled for open-water disposal.

Background

In 1989, the Environmental Advisory Board (EAB) recommended to the Chief of Engineers that risk assessment methods be incorporated into the Corps' dredging program. The Chief accepted these recommendations the following year (Anonymous 1990). To examine the feasibility of incorporating risk-based assessment technologies, a review of the risk assessment process was recently conducted (Dillon 1992). This technical note describes an approach for using risk-based test methods in the regulatory evaluation of dredged material being considered for open-water disposal.

Additional Information

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Current Dredged Material Testing and Evaluation

The Corps' statutory authority for the transport and disposal of dredged material into the ocean or waters of the United States comes, respectively, from

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section 103 of the Marine Protection, Research, and Sanctuaries Act of 1972 (Public Law 92-532) and section 404(b)(1) of the Federal Water Pollution Control Act of 1972 (Public Law 92-500), as amended. Both laws require that there shall be "no unacceptable adverse impacts" on the environment. This statutory language implies that some "adverse impacts" resulting from dredging operations are permitted as long as they are not "unacceptable." These evaluative criteria strongly suggest a risk-based approach for identifying acceptable "adverse impacts" and when "unacceptable adverse impacts" may be anticipated.

However, contaminant testing of dredged material for aquatic disposal allows only a quantal response (U.S. Environmental Protection Agency (EPA) and U.S. Army Corps of Engineers (USACE) 1991); that is, after testing, the material is classified as either suitable for open-water disposal or not suitable. Intermediate judgments are not possible with the current test procedures. The dredged material manager does not have the technical basis for deciding to what degree the project material is "acceptable" or "unacceptable." Instead, the manager must rely on "best professional judgment" to fill the technical void and provide the necessary managerial flexibility. Rightly or wrongly, the Corps has been severely criticized for what is perceived by some as an overreliance on "best professional judgment" and a decision-making process that is too flexible.

Advantages of Risk-based Assessment Methods

The need for a risk-based approach to testing dredged material can be found in the milieu of Corps' decision-making:

- A regulatory decision will always be made.
- This decision will always be based on incomplete data.
- Data which are available will always have some uncertainty.
- Everyone will accept a certain level of risk and uncertainty.
- Achieving zero environmental risk is not possible.
- Managing for near-zero risk is often cost-prohibitive.

Ultimately, a decision regarding specific project dredged material will be made and documented in the Record of Decision (ROD). This decision must be justified but should not be qualified. That is, the ROD should read "Yes, because . . ." or "No, because . . ." not "Yes, but . . ." or "No, but . . .". The justification supporting the regulator's decision presently relies heavily on "best professional judgment." Risk assessment offers a technically sound, quantitative alternative to best professional judgment. It would provide the decision-maker with estimates of environmental risks allowing the decision-maker to balance risks with potential benefits and would also permit the relative risks associated with different management options to be evaluated (USEPA and USACE in preparation).

Another advantage of risk-based assessments is that they address uncertainties explicitly. Instead of ignoring the uncertainties associated with *all* data sets, risk assessments are designed and conducted in a way which quantitates this uncertainty. Technical findings of a risk assessment are expressed in terms of probability statements. In contrast, current dredged material test methods appear as quantal statements. Expressing results as probability distributions recognizes the uncertainties involved and provides a quantitative framework for managerial flexibility (Morgan 1984 and Finkel 1990).

A risk-based approach to testing, therefore, can provide the dredged material manager with a more rational basis for decision-making where subjective evaluations are required. Test results are expressed as a continuum of alternative solutions, each with its own probability of adverse environmental impact. It was these characteristics of risk assessment and the Corps' decision-making environment which prompted the EAB recommendations.

Synopsis of the Risk Assessment Process

A decade ago, the National Academy of Sciences (NAS) recommended a unified, generic process be used by Federal government agencies to assess the health risks posed by anthropogenic chemicals (National Research Council 1983). The NAS risk assessment paradigm (as it came to be known) has been the blueprint for virtually every risk assessment conducted since that time. While details of individual risk assessments vary, they all contain three major elements — exposure assessment, effects assessment, and risk characterization. In exposure assessment, the spatial and temporal distributions of chemicals and chemical mixtures are determined relative to the target receptor of concern. Effects assessment determines the magnitude of chemical toxicity by conducting dose-response experiments in the laboratory with appropriate test species. The third element, risk characterization, integrates exposure and effects assessment data to produce a numerical estimate of chemical risk. Despite the complex jargon and voluminous publications on the subject, all risk assessments consist of just these three simple elements.

Risk-based Framework for Testing Dredged Material

The framework described below is based on what is known and what knowledge must be acquired. It draws heavily upon existing dredged material test methods and is based on current understanding of the fate and effects of contaminated sediment. The framework also suggests some assessment activities which require additional research and development or have not yet been developed. Topics requiring future evaluation include:

- Quantitative probability-based models accurately simulating in-situ exposures.
- Appropriate experimental designs for generating probability-based exposure-response curves.

- Technically sound interpretive guidance for biologically and ecologically important endpoints which have societal value.
- Models to more closely couple the probability-based exposure and effects information.
- Development of formal uncertainty analysis procedures.
- Procedures for accurately communicating environmental risks to nontechnical audiences.

Exposure Assessment

Exposure assessment determines the spatial and temporal distributions of contaminants or contaminant mixtures. In the environment, these distributions often appear as logarithmic functions. Figure 1 presents a hypothetical example of this type of distribution. Note that the mean, a statistic routinely used to portray data sets, does *not* represent the most probable exposures.

Various types of spatial and temporal exposure distributions are associated with the aquatic disposal of dredged material. High concentrations of suspended material may exist for a very short time (minutes to hours) in the water column immediately following disposal. This type of exposure distribution is characterized as both time- and space-limited. Consequently, the *probability* of exposure is very low. In contrast, exposure to low concentrations of suspended sediment has a higher probability of occurrence. Sediment

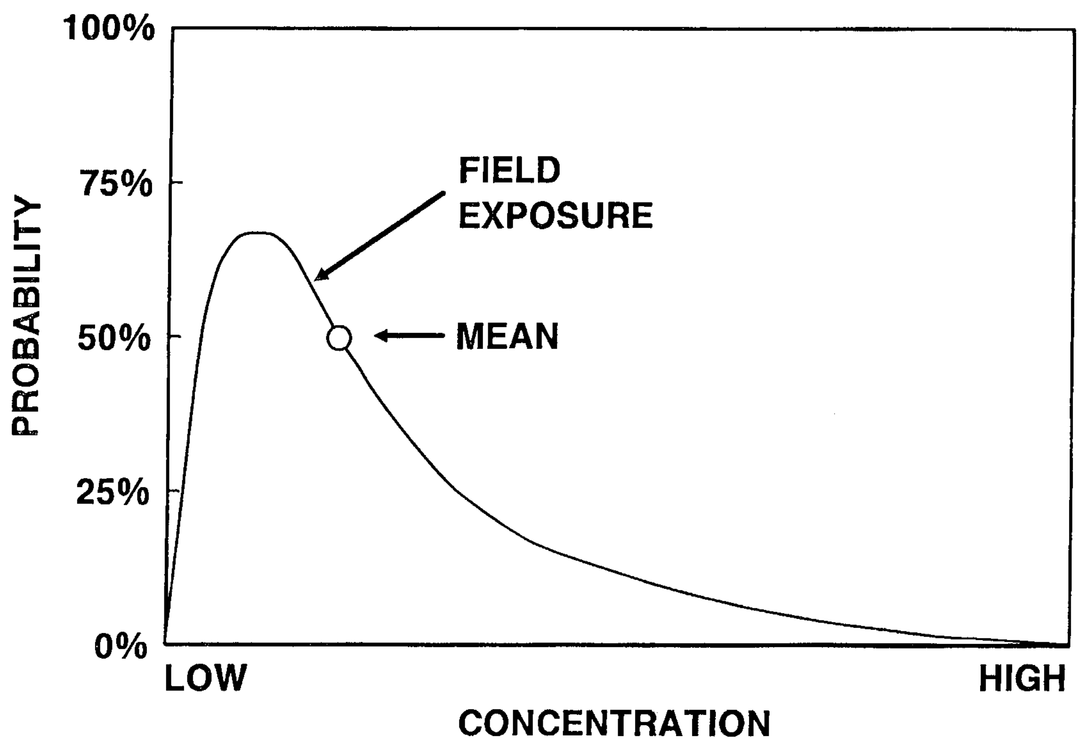


Figure 1. Hypothetical logarithmic probability distribution of environmental contaminants or contaminant mixtures

resuspension may occur frequently and can involve spatially expansive areas. Hence, exposure to low concentrations of suspended sediment is neither time-nor space-limited. A biological component is associated with this latter type of exposure distribution. Many target species of concern (benthic organisms) live at or near the sediment-water interface where sediment resuspension is most intense.

For deposited dredged material, exposure distributions can also vary spatially and temporally. Immediately following point-dump disposal in non-dispersive waters, a discrete mound of material is created on the bottom (Germano and Rhoads 1984). However, material can spread outward radially from the central mound, creating a spatially broad yet relatively thin layer of material surrounding the central mound. Over time, the finer grained material may be winnowed out via currents and resuspension events. Thus, the qualitative nature of the deposited sediment exposure will change temporally.

Effects Assessment

In traditional effects assessment studies where human health is the primary concern, laboratory animals are exposed to a range of chemical concentrations and their biological response to each concentration determined. These data are used to construct dose-response curves. The dose-response curve establishes chemical-specific causality and documents the magnitude of chemical toxicity. Laboratory results are then extrapolated in two ways — from the surrogate test species to the target species of concern and from high laboratory concentrations to low environmentally realistic exposures. The first extrapolation is necessary because toxicity tests with the most common target species of concern, *Homo sapiens*, are not possible. High chemical doses are used in the laboratory because statistically significant responses are not detectable at low concentrations. Not surprisingly, both types of extrapolations introduce considerable uncertainty. Appropriate extrapolation models are still debated in the scientific community (Cothern, Coniglio, and Marcus 1986 and Lu and Sielken 1991).

Effects assessment for dredged material differs from the usual chemical-specific approach in several important aspects. One of the most important differences is based on the fact that dredged material is a complex mixture of chemicals. The chemical composition of sediment samples is rarely ever completely characterized. For that reason, establishing chemical-specific causality with dose-response curves is not possible. Instead, sediment exposure is substituted for the chemical dose to produce an exposure-response curve (Figure 2). Sediment exposure-response curves have two distinct advantages over the standard chemical-specific dose-response curve approach. First, because aquatic organisms (not humans) are the primary target species of concern, effects-based testing can be conducted with that species or a phylogenetic sibling. This eliminates the need for extrapolations between disparate species. Second, environmentally realistic sediment exposures can be included in the experimental design (see horizontal axis in Figure 2). This negates the need for extrapolation models estimating low environmentally realistic exposures from high laboratory doses. Eliminating these dubious extrapolations greatly reduces the

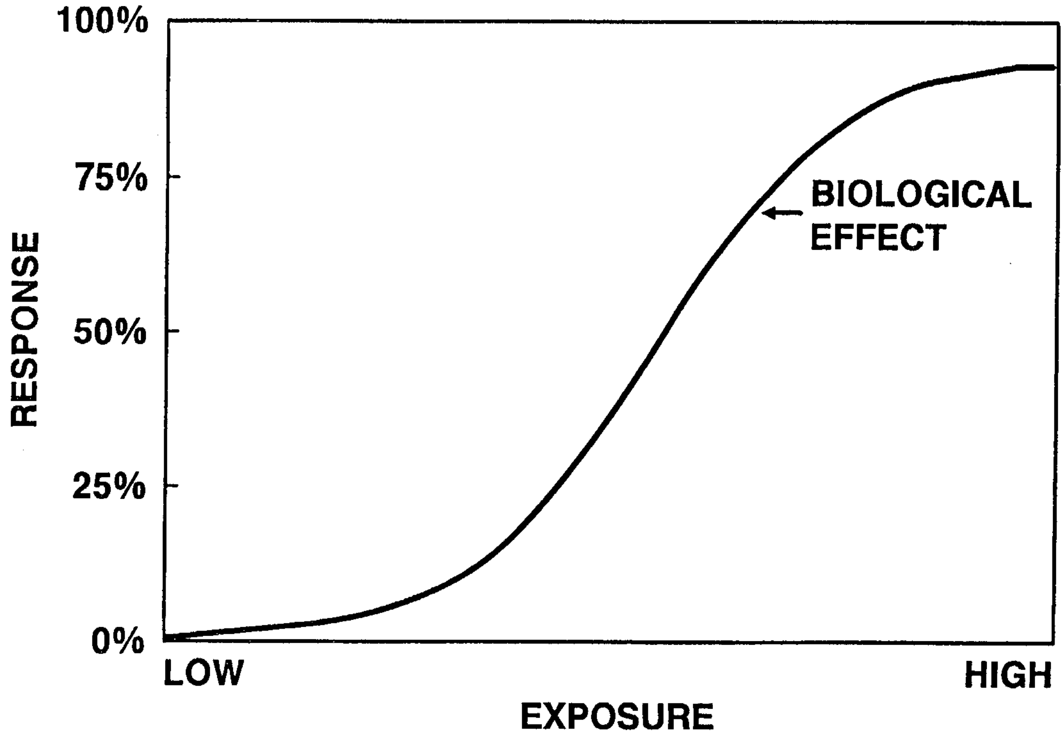


Figure 2. Hypothetical sediment exposure-response curve

uncertainty associated with the sediment exposure-response data and with the subsequent estimate of environmental risk.

Various experimental designs can be used to generate sediment exposure-response curves. For suspended sediments, an exposure gradient can be created in two ways. One approach creates a range of suspended sediment concentrations (mg/L) from a single project sample. The second holds the suspended sediment concentration constant and varies the proportion of project material (for example, 0, 10, 50, or 100 percent). For deposited sediments, a similar approach can be taken by proportionally diluting project sediment with the reference sediment. Alternatively, a known or suspected field gradient can be evaluated by using field-collected sediment samples representing that gradient.

In designing a sediment exposure-response experiment, one must select an appropriate biological response endpoint (see the vertical axis in Figure 2). In the past, sediment bioassays have measured percent survival following acute exposure ($\leq$ 10 days). Most dredged materials, however, are not acutely lethal. Therefore, a new generation of sediment bioassays is emerging which examine more subtle, sublethal endpoints following longer (chronic) sediment exposures (Dillon in preparation). Growth and reproduction are two desirable sublethal endpoints for chronic sediment bioassays (Dillon, Gibson, and Moore 1990). They are sensitive and relatively easy to measure and have high ecological and biological relevance. They have the added advantage of being easily understood by the public. The disadvantage of sublethal endpoints is the lack of technically sound interpretive guidance. While death is easy to discern and interpret, sublethal endpoints encompass a range of responses and each requires a slightly different interpretation. For example, what is the significance of a 5 percent decrease in growth? Is a 10 percent decrease twice as bad or just marginally worse? Interpretive guidance to answer these questions must be generated before chronic sublethal sediment bioassays can be fully used.

Risk Characterization

The exposure and effects assessment information is combined in the last stage of the risk assessment process — risk characterization. This technical integration produces an estimate of environmental risk (Figure 3). Figure 3 was created by superimposing Figure 1 onto Figure 2. One can use this information to project the probability of potential impacts. For example, in the hypothetical data set, the most probable field exposure will occur with a frequency of about 65 percent (Figure 4a). Because this exposure is associated with a very low probability of adverse impacts (≅ 2 percent), one concludes that the environmental risk is very low. The average or mean field exposure (Figure 4b) is associated with a slightly higher incidence of adverse effects (≅ 5 percent). At the other end of the spectrum (Figure 4c), a very high frequency of adverse biological effects (≅ 100 percent) is associated with sediment exposures that are very rare (≅ 2 percent). Whether these sediment-induced adverse impacts are judged "acceptable" or "unacceptable" depends on the interpretive guidance used to explain the biological and ecological importance of test results.

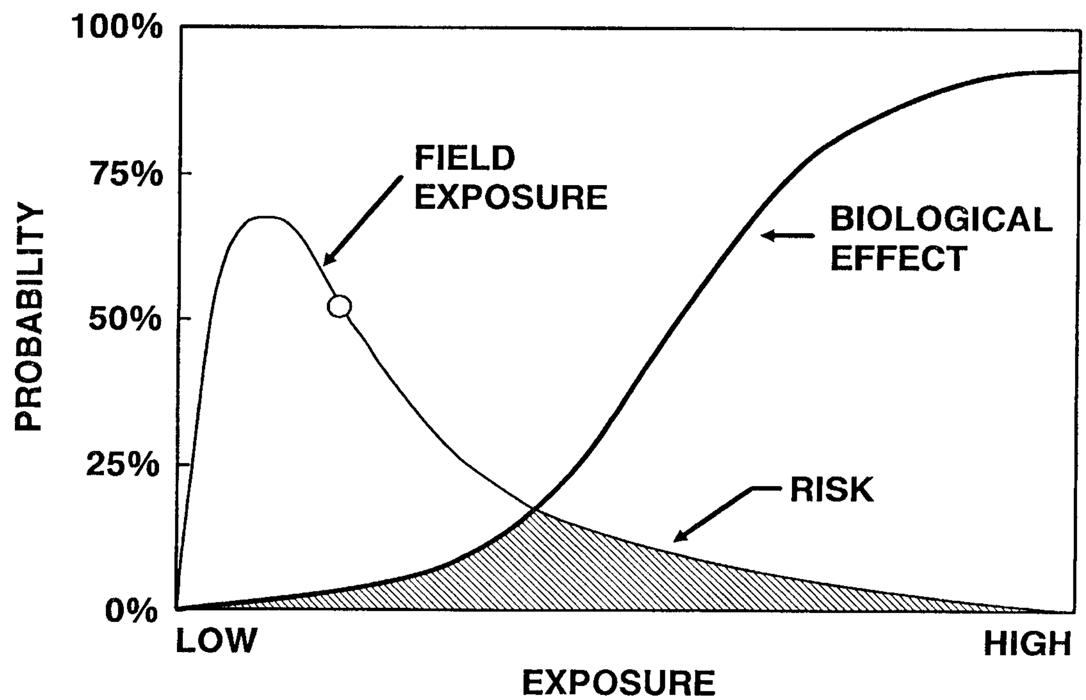
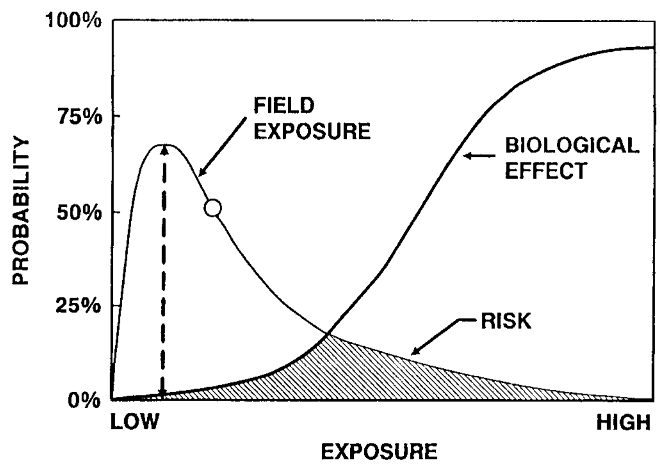
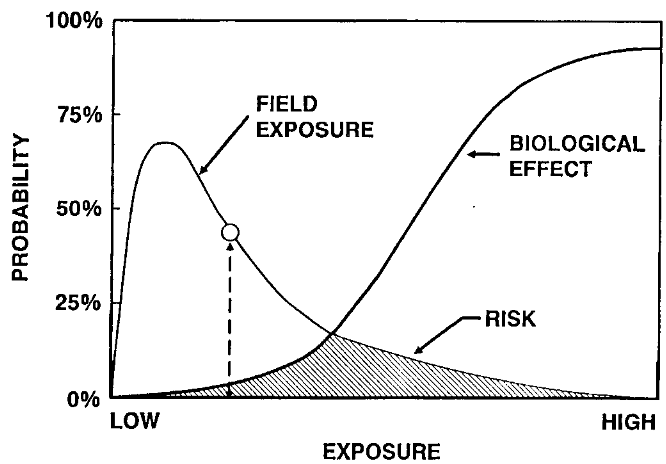


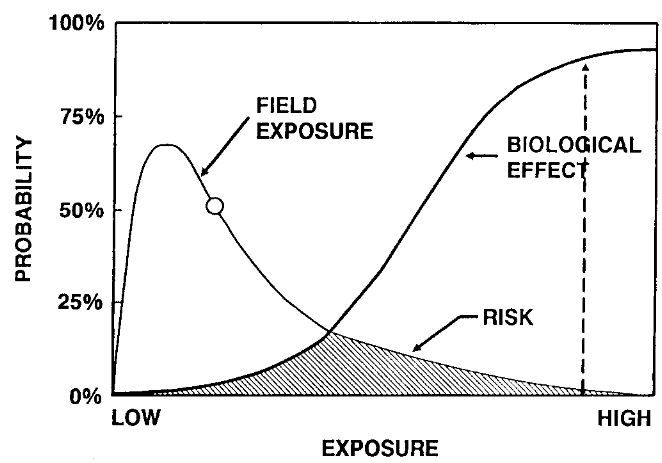
Figure 3. Technical integration of exposure assessment and effects assessment information to yield estimates of probable environmental risk (hatched area)



a. Highest probability event



b. Average probability event



c. Lowest probability event

Figure 4. Use of exposure assessment and effects assessment information to project the relationship between exposure event probabilities and their associated biological effects

Risk-based Management of Dredged Material

Once risk-based dredged material testing has been completed, possible management alternatives are evaluated. These can range from no action to extensive (and perhaps expensive) management. All chemical risks are managed by controlling exposure. This includes contaminated sediments. The intrinsic toxicity of dredged material (that is, the exposure-response curve in Figure 2) can rarely, if ever, be altered.

One popular and effective management technique for deposited dredged material is capping (Shields and Montgomery 1984, Brannon, Hoeppel, and Gunnison 1987, and Palermo in preparation). Project material found to be initially unacceptable for open-water disposal is covered with a cap of acceptable material. This cap physically isolates the unacceptable material and, by reducing the contaminant exposure potential, renders it acceptable. This reduction is shown graphically (Figure 5) using the previous example. Similar risk-based comparisons can be carried out to evaluate other management alternatives such as confined disposal areas or even the no action alternative. Exposure to contaminants in the water column may be reduced by managing the frequency, location, or volume of material disposed. Risk-based technical evaluations also facilitate the weighing and balancing of potential environmental impacts with other management considerations, such as engineering feasibility,

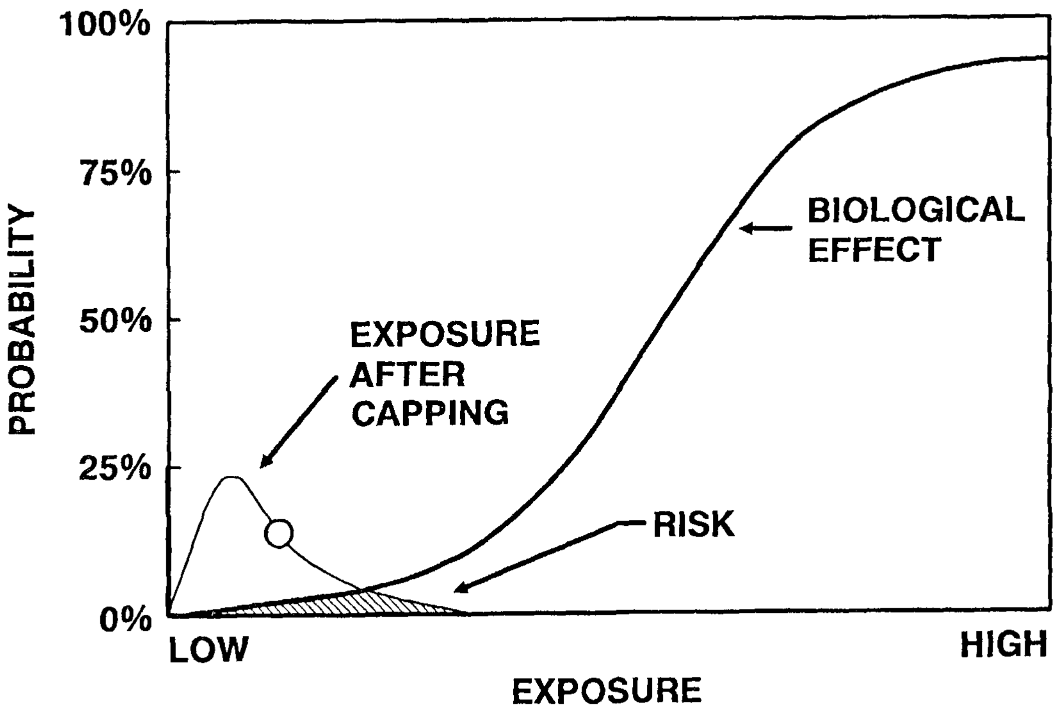


Figure 5. Use of exposure assessment and effects assessment information to quantify the reduction in environmental risk achieved through capping

benefits, and costs. Even qualitative considerations, such as the socio-political decision-making environment, would be facilitated with risk-based testing.

In the future, the Corps will probably become intensively involved in environmental or cleanup dredging. The Corps has three separate authorities for conducting this type of nonnavigational dredging. The oldest, but least used, is section 115 of the Federal Water Pollution Control Act of 1972 (Public Law 92-500). The second, more familiar authority is the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) or Superfund. The 1986 reauthorization of this law, (Superfund Amendments and Reauthorization Act (SARA) (Public Law 99-499)), included the Department of Defense's Defense Environmental Restoration Program (DERP) as section 211. The third authority, also the most recent, is section 312 of the Water Resources Development Act of 1990. Under all three authorities, the Department of Defense and the Corps are required to follow the procedural and substantive assessment techniques recommended by the EPA. The guiding framework for those assessment technologies is environmental risk assessment.

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EEDP-01-29
December 1992

Environmental Effects of Dredging Technical Notes

Determining the Developmental Status of Sediment Toxicity Bioassays

Purpose

This technical note describes events in the generic development of sediment toxicity bioassays for the evaluation of dredged material under section 103 of the Marine Protection, Research, and Sanctuaries Act of 1972 (Public Law 92-532) and section 404(b)(1) of the Federal Water Pollution Control Act of 1972 (Public Law 92-500), as amended. This technical note was written for four reasons:

- To facilitate determining the technical progress of any proposed bioassay by describing its requisite developmental steps.
- To provide the scientific community and regulatory agencies a logical, sequential framework for developing sediment toxicity tests.
- To identify gaps in knowledge and indicate where additional research is needed.
- To suggest a process to the regulatory agencies for evaluating and incorporating a sediment bioassay once it has been accepted by the scientific community.

Background

Sediment toxicity tests are often conducted in the regulatory evaluation of dredged material. Developing these tests requires research on a variety of topics. Some tests are intuitively more developed and more appropriate for regulatory application than others. However, judging the developmental status of individual tests has been difficult because specific criteria are lacking. This technical note provides initial guidance on this subject by describing the steps taken to develop a sediment toxicity bioassay. However, even technically sound sediment toxicity tests may not be appropriate for the regulatory evaluation of dredged material. Again, specific guidance for judging the appropriateness of proposed tests is needed.

Additional Information

Contact the author, Dr. Thomas M. Dillon, (601) 634-3922; the manager of the Environmental Effects of Dredging Programs (EEDP), Dr. Robert M. Engler, (601) 634-3624; or the manager of the Dredging Operations Technical Support (DOTS) Program, Mr. Thomas R. Patin, (601) 634-3444.

Approach

No written guidance exists for judging the developmental status of sediment toxicity tests with regard to the regulatory evaluation of dredged material. For that reason, input was obtained from nearly 40 individuals in the scientific community and regulatory agencies via telephone. Persons contacted represent a geographic balance of the Federal government, private industry and academia (Table 1). Each person was briefed on the purposes of the project, as described above. They were then asked to describe in their own words the characteristics they would expect to see in a fully developed sediment toxicity test intended for the regulatory evaluation of dredged material. Not all persons sought could be reached for comment. For that reason, interested individuals are encouraged to provide written comments to either the author or the EEDP manager.

Analysis

Results of the telephone survey suggest that most people believe sediment bioassays are developed in an orderly, sequenced fashion. Practitioners know this is not always the case. However, it does suggest that new or proposed tests are judged in a similar fashion by asking the question "How far along in the developmental process is the test?" For that reason, much of the input received during the telephone survey was consolidated into a developmental paradigm for sediment toxicity tests (Phase I). Persons contacted, from both the technical and regulatory communities, strongly indicated that any proposed bioassay must be acceptable to the scientific community. Criteria for judging this acceptance are included in Phase II. Phase III is a description of a process for incorporating a sediment toxicity bioassay into the regulatory evaluation of dredged material after it has been accepted by the scientific community. The above steps are summarized in Table 2.

Table 1. Persons Contacted in Telephone Survey

W. T. Adams	ABC Laboratories, Columbia, MO
R. W. Alden	Old Dominion University, Norfolk, VA
D. D. Anderson	U.S. Army Corps of Engineers, St. Paul District, St. Paul, MN
G. T. Ankley	U.S. Environmental Protection Agency, Environmental Research Laboratory, Duluth, MN
S. M. Bay	Southern California Coastal Water Research Project, Long Beach, CA
G. A. Burton	Wright State University, Dayton, OH
D. J. Call	University of Wisconsin-Superior, Superior, WI
E. Casillas	NOAA, National Marine Fisheries Service, Seattle, WA
P. M. Chapman	EVS Consultants, Ltd., North Vancouver, BC
D. C. Cowgill	U.S. Environmental Protection Agency, Great Lakes National Program Office, Chicago, IL
P. A. Dinnell	University of Washington, Seattle, WA
J. L. Dorkin	U.S. Environmental Protection Agency, Region V, Chicago, IL
T. Fredette	U.S. Army Corps of Engineers, New England Division, Waltham, MA
L. Glenbowski	U.S. Army Corps of Engineers, New Orleans District, New Orleans, LA
J. F. Hall	Texaco, Inc., Port Arthur, TX
D. J. Hansen	U.S. Environmental Protection Agency, Environmental Research Laboratory, Narragansett, RI
K. B. Hollar	U.S. Environmental Protection Agency, Region VI, Dallas, TX
C. G. Ingersoll	U.S. Fish and Wildlife Service, National Fisheries Contaminant Research Laboratory, Columbia, MO
D. R. Kendall	U.S. Army Corps of Engineers, Seattle District, Seattle, WA
J. O. Lamberson	U.S. Environmental Protection Agency, Environmental Research Laboratory, Newport, OR
J. M. Lazorchak	U.S. Environmental Protection Agency, Cincinnati, OH
S. K. Lemlich	U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS
J. A. Miller	U.S. Army Corps of Engineers, North Central Division, Chicago, IL
D. Nacci	Science Applications International Corporation, Narragansett, RI
M. K. Nelson	U.S. Fish and Wildlife Service, National Fisheries Contaminant Research Laboratory, Columbia, MO
P. S. Oshida	U.S. Environmental Protection Agency, Region IX, San Francisco, CA
W. H. Peltier	U.S. Environmental Protection Agency, Region IV, Athens, GA
R. J. Pennington	U.S. Army Corps of Engineers, Jacksonville District, Jacksonville, FL
S. I. Rees	U.S. Army Corps of Engineers, Mobile District, Mobile, AL
J. R. Reese	U.S. Army Corps of Engineers, North Pacific Division, Portland, OR
B. Ross	U.S. Environmental Protection Agency, Region IX, San Francisco, CA
N. I. Rubinstein	U.S. Environmental Protection Agency, Environmental Research Laboratory, Narragansett, RI
K. J. Scott	Science Applications International Corporation, Narragansett, RI
J. D. Smith	U.S. Environmental Protection Agency, Region X, Seattle, WA
J. F. Tavolaro	U.S. Army Corps of Engineers, New York District, New York, NY
M. L. Tuchman	U.S. Environmental Protection Agency, Region V, Chicago, IL
F. J. Urabeck	U.S. Army Corps of Engineers, Seattle District, Seattle, WA
C. I. Weber	U.S. Environmental Protection Agency, Cincinnati, OH
J. Q. Word	Battelle Northwest Pacific Laboratory, Sequim, WA

Table 2. Milestones in the Technical Development and Regulatory Adoption of Dredged Material Toxicity Bioassays

Phase I — Developmental Paradigm for Sediment Toxicity Bioassays

- Present rationale for developing the bioassay.
- Select appropriate test species.
- Select biological test endpoint(s).
- Characterize contaminant dose-response.
- Develop test procedure.
- Construct statistical design.
- Specify quality assurance/quality control.
- Evaluate test "ruggedness."
- Generate interpretive guidance.
- Conduct bioassay with range of dredged material.

Phase II — Evaluation by the Scientific Community

- Peer-reviewed publications.
- Interlaboratory evaluations.
- Intertest comparisons.
- Acceptance by the scientific community.

Phase III — Evaluation by Federal Regulatory Agencies

- Joint EPA/Corps committee evaluation.
- Training with detailed written protocol.
- Round-robin testing by contract laboratories.
- Joint EPA/Corps committee approval.

Phase I — Development of the Test Method

Present Rationale for Developing the Bioassay

The test proponent must clearly depict how the sediment bioassay will be used in the regulatory evaluation of dredged material. Obviously, this requires some knowledge on the part of the test proponent of the regulatory *milieu*. This knowledge should be acquired before test development. Otherwise, considerable resources may be expended in developing a test for which there is no practical use. For example, is the bioassay intended to evaluate bedded or suspended sediments? Is it designed for early tier screening or later evaluations? Is it designed to help implement section 103 of the Marine Protection, Research, and Sanctuaries Act of 1972 (Public Law 92-532) or section 404(b)(1) of the Federal Water Pollution Control Act of 1972 (Public Law 92-500), as amended? Can it be performed by the contracting community or is it restricted to research and development laboratories? Is the cost of the proposed test in line with current bioassays or would it be expensive to run and require a considerable capital outlay?

Select Appropriate Test Species

Selection of an appropriate test species is the second and arguably the most critical step in developing a sediment bioassay. Its importance is derived from the fact that biological response is used to "assay" the toxicity of sediment-associated contaminants in dredged material. This biological response, in effect, becomes a "toxicity meter." The following selection criteria must be met:

- *Compatible with test media.* Infaunal species (tube-building or free-burrowing) are used to evaluate bedded sediments while epibenthic, planktonic, or nektonic species are used with suspended sediments.
- *Ecologically, commercially, recreationally important or indigenous.* The biology and natural history of the test species must be documented. For example, what is its ecological function with regard to carbon flow and nutrient cycling.
- *Available throughout the year.* Sufficient numbers of healthy test organisms must be readily available throughout the year either through laboratory cultures or field collections. If cultured, there must be performance criteria for assessing the culture's viability and a published standard operating procedure (SOP) for culturing. If field collected, there must be an acclimation SOP and the effect of seasonality on bioassay results must be documented. For example, what is the seasonal influence of gametic cycle, ambient temperature, recent food availability, and water quality?
- *"Handleable."* Good survival in the negative control treatment and consistent response in the positive control must be achievable on a routine basis by contract laboratories.
- *Documented contaminant sensitivity.* The sensitivity to major classes of contaminants must be documented; details are provided below.

Select Biological Test Endpoint(s)

Sediment toxicity tests have traditionally measured survival as the primary test endpoint. While this will always continue to be true, a new generation of sediment bioassays that examine sublethal endpoints is now being developed (Dillon in press). These tests typically involve longer (chronic) sediment exposures. The potential number of sublethal endpoints is virtually infinite and includes responses at all levels of biological organization (biochemical, cellular, organismic, population, and community). However, the practical number of sublethal endpoints is much smaller because they must be ecologically relevant, not too difficult to measure, and easily understood outside the scientific community. Reproduction and growth are often cited as two highly desirable sublethal test endpoints (Dillon, Gibson, and Moore 1990). The type of bioassay test endpoint has a major impact on the type of interpretive guidance required (see below).

Characterize Contaminant Dose-Response

A fundamental principle in toxicology is that no chemical is either inherently toxic or inherently safe. Rather, it is the amount or internal *dose* experienced by the biological receptor that renders a substance toxic or therapeutic (Klaassen, Amdur, and Doull 1986). The quantitative relationship between internal dose and the response that dose elicits is called the dose-response curve. This curve was borrowed early in the formative years of aquatic toxicology to assess the relative toxicities of environmental contaminants (for example, see Sprague 1969). However, it has been used in a significantly different manner. Chemical dose was replaced by external exposure concentration. In other words, the exposure concentration became a *surrogate* for internal dose (Connolly 1985).

One of the many uses of the exposure-response curve in aquatic toxicology was to seek the "most sensitive species." The results of this search have been equivocal. Reviews of aquatic toxicity data (Klapow and Lewis 1979, Thurston and others 1985, Mayer and Ellersieck 1986, and Slooff, van Oers, and de Zwart 1986) as well as convincing theoretical arguments (Cairns and Niederlehner 1987) suggest that seeking a "most sensitive species" may be much like the quest for the "holy grail."

For dredged material bioassays, seeking the "most sensitive species" is even more problematic because sediments are *mixtures* of chemicals. Some of these chemicals are identified by laboratory analysis, but many more are present but never analyzed. The mixture problem is confounded by the fact that these chemicals are embedded in a very complex, heterogeneous geological matrix. Contaminant bioavailability and in situ exposures are affected by these characteristics in a manner not easily understood. For these reasons, dredged material evaluations use "effects-based" testing, that is, allowing the biological response of the test species to integrate the availability and toxicity of all sediment-associated contaminants. Clearly, identifying the "most sensitive species" under these conditions would be quite difficult. Rather, the goal should be to *characterize* the causal relationship between test species' response and major classes of contaminants (for example, metals, chlorinated hydrocarbons, low- and high-molecular weight petroleum hydrocarbons, and pesticides). In a return to fundamental toxicological principles, this characterization should be based on internal *dose* rather than external concentration (Connolly 1985).

Develop Test Procedure

The experimental protocol is a detailed description of how the proposed test will be conducted. It includes but is not limited to:

- Treatment of sediment before, during, and after the test.
- Treatment of test organism before, during, and after the bioassay.
- Physical conditions (for example, temperature, photoperiod, and aeration).
- Replicate description (for example, size and animals/replicate).

- Feeding.
- Daily activities (for example, visual observations and water quality).
- Duration of test.
- Test termination procedures.
- Measurement of test endpoint.

Construct Statistical Design

Statistical design is the a priori description of what types and amounts of data are required to adequately test a given hypothesis and how these data will be analyzed. It includes but is not limited to:

- Hypothesis formulation.
- Level of statistical significance.
- Randomization procedures.
- Number of treatments.
- Number of replicates per treatment.
- Population sampling.
- Hypothesis testing (data reduction/data analysis).
- Power analysis.
- Sensitivity analysis.

Specify Quality Assurance (QA)/Quality Control (QC)

QA/QC is the administrative and technical steps taken to ensure reliable data are produced with specified precision and accuracy. It includes but is not limited to:

- Analysis of intratest variability.
- Analysis of variability at different levels of biological response.
- Acceptable response in negative controls.
- Consistent response in positive controls.
- Development of performance criteria.
- Use of control charts.

Evaluate Test "Ruggedness"

The American Society for Testing and Materials (ASTM) (1992a) defines "ruggedness" as the "insensitivity of a test method to departures from specified test or environmental conditions." Some of these conditions are identified when the initial test procedure is developed. However, others deal with the intrinsic properties of the sediments and require additional study. Examples include the effects of grain size, interstitial ammonia and sulfides, presence of indigenous fauna, and organic carbon. It has been shown that these factors can and do bias results of acute lethality sediment bioassays (DeWitt, Ditsworth, and Swartz 1988, and Ankley, Katko, and Arthur 1990). Their potential influence will no doubt increase when test duration increases and more sensitive endpoints are examined (that is, chronic sublethal sediment bioassays). It is therefore incumbent upon the test proponent to evaluate these factors. Guidance for evaluating test "ruggedness" has been provided by ASTM (1989). Results should be summarized as a matrix of conditions under which the test should or should not be conducted.

Generate Interpretive Guidance

The bioassay proponent must provide the technical basis for interpreting the biological and ecological importance of test results. Interpretive guidance should not be confused with statistical significance. The latter is an arbitrary (but hopefully not capricious) means of judging numerical data within a specific level of confidence. Interpretive guidance, on the other hand, explains the biological importance of the observed results. For example, if a project sediment causes a statistically significant 5 percent decrease in survival or growth, is that truly detrimental to the organism? Would a 10 percent decrease be twice as "bad" or only incrementally injurious? A more concrete example can be found in contemporary sediment bioassays conducted with two of the most commonly used species — *Rhepoxynius abronius* and *Ampelisca abdita*. An observation of 30 percent mortality in *R. abronius* is probably much worse than 30 percent mortality in *A. abdita* simply because the former is an annual species and the latter has multiple broods per season. Generating interpretive guidance for sublethal endpoints represents an even greater challenge than that required for survival data.

Conduct Bioassay with Range of Dredged Material

Once a draft protocol has been developed, the test should be conducted on a range of well characterized sediments representing suspected low and high toxicity. Gauging the success (or failure) of this initial sediment testing will be directly dependent on the preceding research and development. If sufficient time and effort has been devoted to the issues described in Phase I above, this initial foray with natural sediments should result in only minor adjustments to the protocol. Too many sediment bioassays probably enter this phase prematurely.

Phase II — Evaluation by the Scientific Community

Peer-Reviewed Publications

The test proponent must communicate the research results in peer-reviewed publications. This activity serves several functions. First, it permits simultaneous access to the test protocol to everyone in the scientific community. This examination promotes and focuses scientific debate. Before publication, knowledge is anecdotal and typically limited to informal communications between colleagues. Acceptance for peer-review publication, however, does not necessarily imply endorsement nor acceptance on the part of the scientific community. In fact, some editors will publish marginal manuscripts in an effort to induce scientific debate.

Second, increased scrutiny brought on as a result of peer-review publication will greatly increase the probability that weakness in a proposed test method will be discovered — a healthy process. Exposing weaknesses does not necessarily disqualify any bioassay. On the contrary, it usually leads to significant improvements. At the very least, it helps define the test's limits of applicability.

Third, in a good, well written journal article, the author will identify knowledge gaps and recommend important areas for further research and development. At this point in its development, the proposed sediment bioassay is beginning to move out of its laboratory of origin and into the larger family of research laboratories.

Interlaboratory Evaluations

If there are sufficient resources and technical interest, the proposed method will be conducted by other research and development laboratories. This is an important and critical step in the evolution of any test method. Interlaboratory evaluations can be designed to accomplish one or more goals.

- Improve specific aspects of the test method via targeted research.
- Expand the domain of bioassay response with other dredged material.
- Evaluate interlaboratory variation.
- Compare response with other sediment bioassays (see below).

Intertest Comparisons

Once an initial draft protocol has been modified and refined through debate and research in the scientific community, it is ready for comparison to other sediment bioassays. For this comparison to be meaningful, it must be conducted in an equitable fashion; that is, same sediment, same time, same place, same temperature, and so forth. Intertest comparisons under dissimilar circumstances are not valid. One purpose of the intertest study is to examine how frequently and with what precision a particular bioassay indicates toxicity relative to other sediment bioassays. It is *not* designed to identify the "most sensitive bioassay." As with species sensitivity, finding the single most sensitive sediment bioassay is probably not achievable. Most intertest studies recognize this fact and recommend using a battery of sediment bioassays (Burton and others 1989, Giesy and Hoke 1989, Long and Buckman 1989, and Pastorok and Becker 1990).

Acceptance by the Scientific Community

The scientific community has developed little written guidance for accepting or rejecting individual bioassays. Instead, a "survival of the fittest" process usually takes place. Over time, some bioassays are examined and used with greater frequency, while others receive less and less attention. Eventually, some tests disappear from laboratory evaluation altogether. This is usually a slow but healthy process. Close scrutiny by many investigators ensures "survival of the fittest"; that is, tests that work and are biologically meaningful. If this process has one weakness, it is determining precisely when a particular test has been accepted (or rejected) by the scientific community. Many of those contacted during the telephone survey indicated that being able to discern when the scientific community had made this judgment was very important to them.

Probably the most discrete temporal event connoting scientific acceptance of a sediment bioassay is publication by ASTM's Subcommittee E47.03 on Sediment Toxicity. However, the reader should realize that even these ASTM documents are not step-by-step "cookbooks." In ASTM parlance, these reports are *guides* — "a series of options or instructions that do not recommend a specific course of action" (ASTM 1992b). The lack of an instruction manual does not mean that the Subcommittee members cannot make a decision. Rather, it reflects the true state-of-the-practice in sediment toxicity testing.

Phase III — Evaluation by Federal Regulatory Agencies

Joint EPA/Corps Committee Evaluation

Open-water disposal of dredged material is evaluated under regulations implementing portions of two laws: section 103 of the Marine Protection, Research, and Sanctuaries Act of 1972 (Public Law 92-532) and section 404(b)(1) of the Federal Water Pollution Control Act of 1972 (Public Law 92-500), as amended. Joint Federal regulatory responsibility is vested with the U.S. Army Corps of Engineers and the U.S. Environmental Protection Agency. These agencies have created two permanent joint committees to oversee the technical implementation of these laws and regulations. Because the regulations include sediment bioassays in the evaluation of dredged material, it is logical that these two committees review and judge the appropriateness and acceptability of proposed sediment bioassays.

The basis for evaluating a sediment bioassay is much broader than just technical soundness. As public servants and custodians of the public welfare, regulatory agencies are required to balance resource expenditures with benefits received in all Federal actions. They must be able to explain to the public or, in the case of permitted activities, to the private sector, precisely why the test is being conducted, what information it will yield, and how that information will be used in decision-making. Important criteria used by regulatory agencies in evaluating a sediment bioassay include but are not limited to:

- Relevant and appropriate for the intended use.
- Founded in the applicable laws and regulations.
- Accepted by the scientific community.
- Accompanied by interpretive guidance.
- Demonstrated track record with a variety of dredged material.
- Cost-effective.
- Able to sustain judicial review.
- Simplified "cookbook" version of the bioassay available.
- "Doable" in a routine fashion by contract laboratories.

Training with Detailed Written Protocol

Once a technically sound method has been developed and accepted by the scientific community, some level of training is highly desirable. Accompanying this training should be a simplified step-by-step instruction manual. This instruction manual should be based on the appropriate detailed technical documentation, but should not include extraneous material not required for conducting the bioassay in a technically sound manner.

Round-robin Testing with Contract Laboratories

Contract laboratory performance is analyzed by round-robin testing. The purpose is to evaluate the laboratories' technical ability to conduct the test, establish market-based costs for conducting the bioassay, determine interlaboratory variability, and expand the track record for this bioassay with a greater variety of dredged material. Use of these round-robin data to determine the acceptability of specific project materials will be made on a case-by-case basis.

Joint EPA/Corps Committee Approval

Once the above steps have been completed, the EPA/Corps joint technical committees should formally approve (or disapprove) a particular sediment toxicity test.

Sediment Bioaccumulation Bioassays

The focus of this technical note was on the technical development and potential regulatory use of sediment toxicity bioassays. The same approach can be applied to sediment bioaccumulation bioassays. In that case, many of the Phase I elements (test development) would be different. However, much of Phase II and Phase III activities (evaluation by the scientific and regulatory communities, respectively) would be very similar.

Future Activities

This technical note provides initial guidance for determining the developmental status of sediment toxicity tests for the regulatory evaluation of dredged material. It will form the basis for a workshop to be conducted in FY 93. The purpose of the workshop will be to comment on the content and completeness of this technical note. Participants will be charged with prioritizing developmental milestones and assigning attributes such as "must," "should," and "could" to each milestone. Invited participants will be those who are actively involved in developing and regulating with dredged material toxicity bioassays. Following the workshop, final guidance will be published as a technical note.

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Environmental Effects of Dredging Technical Notes

Environmental Effects Evaluation for Thalweg Disposal of Dredged Material

Purpose

This technical note describes the general concept of thalweg disposal and presents information on the potential environmental effects of thalweg disposal, including water quality, habitat alteration, and fate of sediments. This note also presents the results of studies done on the environmental consequences of thalweg disposal at four test sites on the upper Mississippi River.

Background

The thalweg of a river is defined by a line whose course is given by connecting the lowest points along the streambed for each transect. The thalweg's course passes through pools at river bends and through crossings between the bends. During high-discharge events along a river system, pool areas scour and crossings accrete material. The opposite takes place during low-discharge periods, but with a lower magnitude of change. Blockages to navigation generally occur at the crossings.

The concept of thalweg disposal is to dredge the shallow reaches and dispose the dredged material in a downstream pool. Thalweg disposal is a form of open-water disposal and is regulated under Section 404 of the Clean Water Act (CWA). The "Guidelines for Specification of Disposal Sites for Dredged or Fill Material," outlined in 40 CFR 230, apply (U.S. Environmental Protection Agency (EPA) 1980).

Additional Information

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Concept of Thalweg Disposal

Thalweg disposal refers to the practice of disposing of dredged material by discharge into the naturally occurring scour holes within a river—a form of open-water disposal specific to these locations. A more rigorous description has been given by the U.S. Army Engineer District (USAED), Rock Island, as follows: "Thalweg disposal is placement of dredged material in a deep-water portion of the channel thalweg where it will become a natural element of the sediment transport system, and will be assimilated into the system with minimal impacts to either the sediment transport system or the environment" (Nanda and Baker 1984). In practice, thalweg disposal mimics a cut-and-fill operation, whereby a shallow crossing is dredged and the material is moved into a downstream pool. Thalweg disposal is therefore similar to the natural process of low-water scour and accretion of crossings and pools, although greater in rate and magnitude. Theoretically, if the volume to be dredged is small compared with the total annual transport, the energy increment used to move the sediment from crossing to pool should have little overall effect on the regime of the river (Lagasse 1975).

By definition, the thalweg of a river follows the line connecting the lowest points along a streambed. The thalweg will meander back and forth across the riverbed in response to the changing course of the river, as shown in Figures 1 and 2. At many locations within the thalweg, the depth is sufficient to permit dredged material disposal without interference to navigation. Figure 3 illustrates this concept, before and after disposal.

Thalweg disposal has been proposed as a disposal alternative for uncontaminated sediments and as an alternative to the use of sidecasting dredges, which have the disadvantage of high disturbance and a tendency for redeposition of material in the cut. Thalweg disposal offers potential economic advantages, eliminating the need to transport dredged material to confined disposal sites, and the costs associated with acquisition, development, and maintenance of those sites. In some cases, thalweg disposal constitutes the environmentally preferred alternative (personal communication, January 1993, Richard M. Baker, USAED, Rock Island).

Information on the implementation of thalweg disposal is provided in *Environmental Effects of Dredging Technical Notes* EEDP-01-31, "Implementation Approach for Thalweg Disposal of Dredged Material" (Olin 1993).

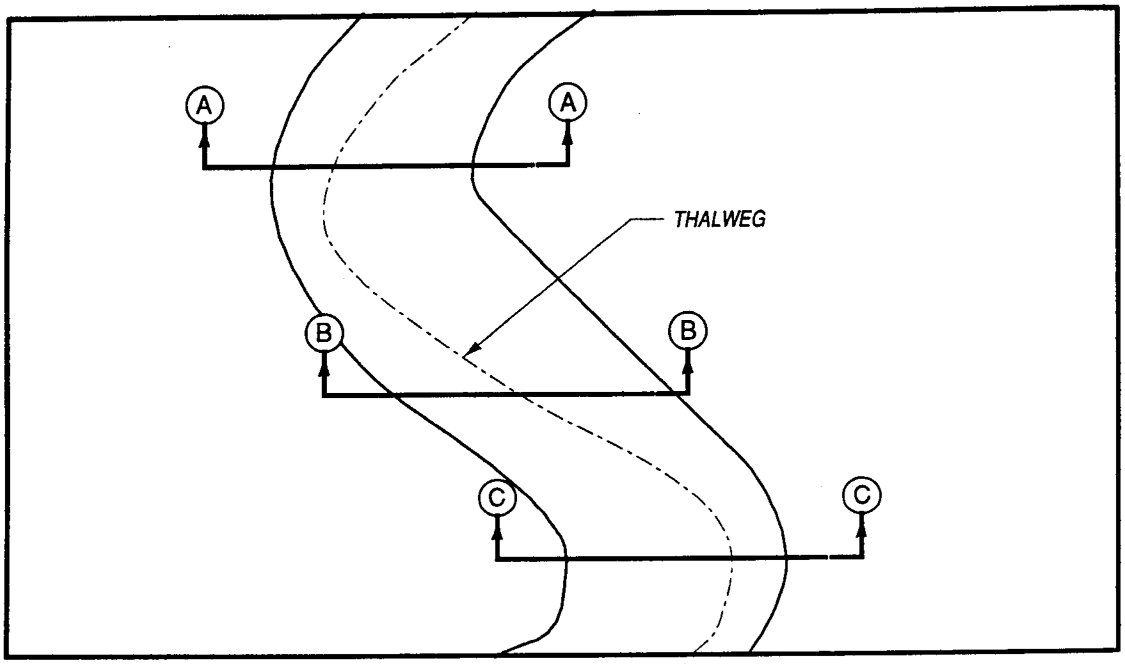

Figure 1. Line of the thalweg

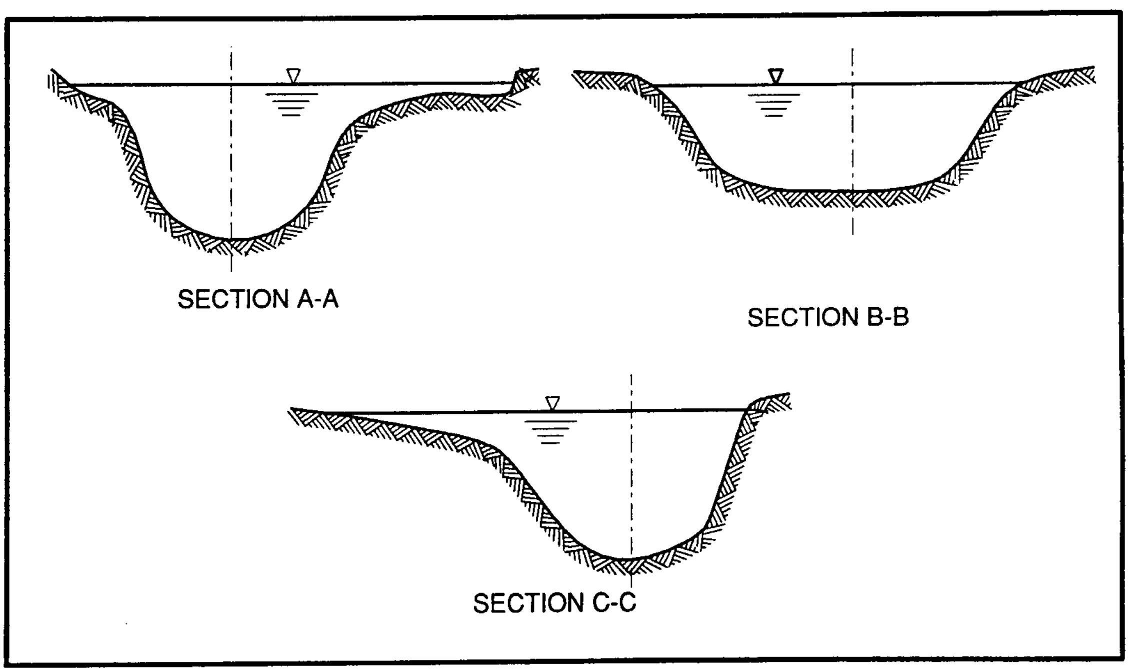

Figure 2. Section depicting location of the thalweg

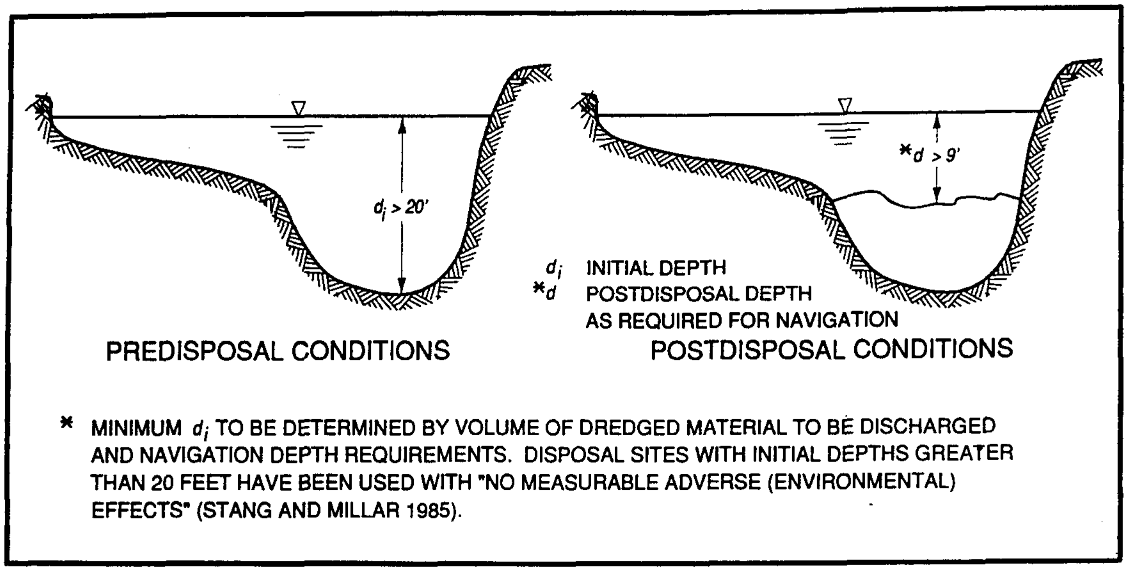

Figure 3. Section view of disposal site

Environmental Effects

Potential environmental concerns of thalweg disposal include the effect on water quality (due to increased turbidity and resuspension of any contaminants), habitat alteration (due to changes in the existing substrate), and the fate of sediments.

Environmental consequences of thalweg disposal were studied at four test sites on the upper Mississippi River (Paddock and McCown 1984, McCown and Paddock 1985). The USAED, Rock Island, has supported studies to demonstrate the viability of thalweg disposal for noncontaminated dredged material (McCown, Paddock, and Ditmars 1984, Paddock and McCown 1984, and McCown and Paddock 1985). In addition, studies were conducted to determine the relative value of various habitats within the riverine system in order to establish environmentally sound criteria for implementation of the procedure (Lubinski 1984, Stang and Nickum 1985b).

Water Quality Effects

Because thalweg disposal is a form of open-water disposal, the suitability of the material for open-water disposal from the standpoint of contaminants must be determined. A tiered evaluation approach is used (Environmental Protection Agency/U.S. Army Corps of Engineers (EPA/USACE) 1991; EPA/USACE, in preparation).

An initial screening for contamination is designed to determine, based on available information, if the sediments to be dredged contain any contaminants in forms and concentrations likely to cause unacceptable impacts to the environment. Materials considered for thalweg disposal may be excluded from testing as specified in 40 CFR 230.60. However, if the material does not meet the exclusions, the contaminants must be addressed with respect to their potential for biological effects or release through applicable pathways.

Water column contaminant impacts must be considered from the standpoint of water quality (chemical) and toxicity (biological). Benthic impacts must be considered from the standpoint of toxicity and bioaccumulation. Detailed descriptions of the initial screening for contamination and testing and of the assessments for the tiered approach are available in EPA/USACE (1991) and EPA/USACE (in preparation).

Turbidity and suspended solids in the water column will be increased to some degree during thalweg disposal, with the degree of the effect depending upon the disposal method. Typically, dissipation occurs rapidly after disposal, and the effects are transient. However, the acceptability of a discharge is regulated under State water quality certification requirements and Section 404 of the CWA (EPA 1980).

Habitat Alteration

Possibly the most significant effects of aquatic disposal are seen as a result of burial of the benthos. Some species are capable of migrating upward through the imposed sediment load, but most surface-dwelling life forms cannot, and therefore die. Mussels, periphyton, invertebrates, and dormant fish populations can suffer mortality. Effects on larval fish are thought to be minor, as they are not bottom-dependent for food or shelter (Stang and Millar 1985).

The effects of thalweg disposal are not altogether permanent. Reestablishment of species on the disposal site begins within several months, and near-complete recovery is achieved within 1 to 2 years (USACE 1983). Usually, opportunistic species are the first to repopulate a disposal site. Species diversity at the site is low, often for several months; however, diversity can recover over a period of years.

From the perspective of the ecosystem as a whole, it is desirable to protect species diversity, as well as species with identified recreation and ecological value or endangered species. Because of the potential for adverse effects of thalweg disposal on the benthos, it is important to adhere to responsible site selection procedures, with an important objective being to avoid valuable habitats both within and near the intended disposal site.

Aquatic biota will differ from location to location, as will their habitat. Because of the dynamic nature of the riverine environment, generalizations are somewhat difficult to make, but some features emerge as consistently important to a wide variety of aquatic life forms. Substrate type is one such feature, with coarse and stable substrates being important to a wide variety of fish species for egg laying and protection from high-velocity water. Typically, these substrates consist of hard sand and clay with mixtures of gravel, cobbles, bedrock, shells, and boulders or logpiles (Lubinski 1984).

Sand substrate and sand dunes in a dynamic environment are of less value to species on the upper Mississippi, although this may not be true in other locales. Deep holes have been demonstrated to be important to catfish for overwintering, and catfish have been found in the deep scour holes located in outside bends on the upper Mississippi (Stang and Nickum 1985a). However, this habitat is generally considered less valuable because of the high water velocities found there. It has been demonstrated that where revetments, dike fields, weirs, and other hydraulic or natural "structures" exist, catch per unit effort and species numbers tend to be higher (Stang and Nickum 1985b). Studies in other locales would reveal other species and associated habitats of importance.

Preservation of species diversity and endangered species is regulated by the Endangered Species Act of 1978, as amended. While general characteristics of valuable aquatic habitat are known, the annual and seasonal variability in the use of any site by the aquatic community would indicate a need for examination of the proposed disposal site and its immediate area (which may be potentially affected by disposal) prior to use. This may be accomplished by sampling, diving surveys, and other methods that will provide a rapid assessment of substrate and the organisms present (personal communication, 16 September 1992, Dr. Andrew C. Miller, U.S. Army Engineer Waterways Experiment Station).

Effective sampling can be accomplished to varying degrees of reliability, and may not be feasible in all cases. Therefore, to further minimize the potential for adverse effects, thalweg disposal should be

- Restricted to those sites constituting the least valuable habitat to species of importance.
- Restricted to disposal of materials of similar grain size to those of the disposal site (personal communication, January 1993, Richard M. Baker, USAED, Rock Island).
- Seasonally restricted as appropriate for local conditions and habitat uses.

Fate of Sediments

Lubinski (1984) suggested that, after placement in the thalweg, dredged material either remained at the site or was assimilated into the bed load, where it could then migrate in response to water currents.

Migration of sediment is an important environmental consideration, which can potentially impact important habitats downstream from the disposal site.

Thalweg disposal has been used to some extent on the lower Mississippi River, and the Rock Island District has used and studied the procedure on the upper Mississippi River. Lower Mississippi dustpan dredging operations use the procedure when the river stage is such that access to holes downstream from the extraction site is possible. While there may be some movement of sediment out of the disposal site, in this area it constitutes a small fraction of the bed load, and effects are considered to be negligible (personal communication, August 1992, Larry Rabalais, USAE Division, Lower Mississippi Valley).

Studies of the movement of sand, tagged with fluorescent dye, from four test sites on the upper Mississippi River (Savannah Bay, Whitney Island, Gordons' Ferry, and Duck Creek) were conducted by the Argonne National Laboratory. Results of a 9-month observation of the Savannah Bay site (Paddock and McCown 1984) correlated closely with results obtained at the Whitney Island and Gordons' Ferry sites. This investigation revealed that contours of the disposal mound had been altered and dunes had developed, similar to the original bottom configuration of the river. Movement of tagged sand from the original site was observed, apparently confined to within the thalweg, and occurred in response to high river discharge.

At the Gordon's Ferry site, sampling that was conducted after a 5-year flood event (at a time approximately 20 months after disposal) revealed downstream movement of tagged sand for a distance of approximately 1,000 m.

Tagged material redistributed outside the thalweg was thought to be primarily fines and not representative of the characteristics of typical dredged material. It was concluded that "virtually no movement of dredged material into side channels occurs where the thalweg is at least 10 to 20 feet deeper than the channel inlet. Where the side channel inlet and the thalweg are of similar depth, however, migration of material into the side channel can be assumed. Side channel accretion may be due to sand input from the channel border area" (personal communication, January 1993, Richard M. Baker, USAED, Rock Island).

In the sites tested and sampled, the disposal mounds were eradicated by the first flood. Tagged sand appeared to have been incorporated into the bed forms of the natural channel (Ditmars, McCown, and Paddock 1986). A similar experiment conducted in a more complex reach with submerged wing dams on either bank resulted in a return to original depth within 5 months (September to January) after disposal (Ditmars, McCown, and Paddock 1986). Further monitoring of other, more diverse sites will be necessary to determine whether the behavior of these sites is representative.

As part of investigations conducted at Waterways Experiment Station for the St. Louis District, two tests in a physical movable-bed model were conducted for the Dogtooth Bend reach of the middle Mississippi River (miles 39.6 to 20.2). Considerable channel stabilization work has been done at this location, including weirs, dikes, and revetments, all designed to increase channel depth and improve navigation. The model used granulated coal as both dredged material and bed medium. Plastic particles were mixed with the dredged material to act as tracers. One test examined disposal along the opposite bank from the dredge cut and in scour holes off the ends of dikes. True thalweg disposal was not examined. Sediment transport, rate of movement, and areas of deposition were examined and recorded. Preliminary results were encouraging in that, for the limited testing performed, material deposited in the scour holes at the stream end of dikes did not negatively impact the navigation channel in the two bends and crossing downstream of the disposal site. However, a more intensive study would be needed to determine if results were representative of the behavior of sediments in natural channels.

Summary

"Thalweg disposal is placement of dredged material in a deep-water portion of the channel thalweg where it will become a natural element of the sediment transport system" (Nanda and Baker 1984). Thalweg disposal mimics the natural low-water scour and accretion of crossings and pools.

Of primary concern are the potential adverse effects on water quality due to increased suspended solids, possible resuspension of contaminants, short- and long-term effects on the aquatic environment from alteration of the existing habitat, and effects on the immediate area resulting from sediment migration from the disposal site. The environmental effects of thalweg disposal are minimized, however, when the procedure is appropriately implemented and the disposal site appropriately located. In some cases, thalweg disposal may be the environmentally preferred alternative.

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May 1993

Environmental Effects of Dredging Technical Notes

Implementation Approach for Thalweg Disposal of Dredged Material

Purpose

This technical note introduces the concept of thalweg disposal and associated considerations for implementation, including disposal site selection, environmental and regulatory considerations, and suitable dredging methods and equipment. Monitoring procedures are also outlined.

Background

The thalweg of a river is defined by a line whose course is given by connecting the lowest points along the streambed for each transect. The thalweg's course passes through pools at river bends and through crossings between the bends. During high-discharge events along a river system, pool areas scour and crossings accrete material. The opposite takes place during low-discharge periods, but with a lower magnitude of change. Blockages to navigation generally occur at the crossings.

The concept of thalweg disposal is to dredge the shallow reaches and dispose the dredged material in a downstream pool. Thalweg disposal is a form of open-water disposal and is regulated under Section 404 of the Clean Water Act (CWA). The "Guidelines for Specification of Disposal Sites for Dredged or Fill Material," outlined in 40 CFR 230, apply (U.S. Environmental Protection Agency (EPA) 1980).

Additional Information

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Concept of Thalweg Disposal

Thalweg disposal refers to the practice of disposing of dredged material by discharge into the naturally occurring scour holes within a river—a form of open-water disposal specific to these locations. A more rigorous description has been given by the U.S. Army Engineer District (USAED), Rock Island, as follows: "Thalweg disposal is placement of dredged material in a deep-water portion of the channel thalweg where it will become a natural element of the sediment transport system, and will be assimilated into the system with minimal impacts to either the sediment transport system or the environment" (Nanda and Baker 1984). In practice, thalweg disposal mimics a cut-and-fill operation, whereby a shallow crossing is dredged and the material is moved into a downstream pool. Thalweg disposal is therefore similar to the natural process of low-water scour and accretion of crossings and pools, although greater in rate and magnitude. Theoretically, if the volume to be dredged is small compared with the total annual transport, the energy increment used to move the sediment from crossing to pool should have little overall effect on the regime of the river (Lagasse 1975).

By definition, the thalweg of a river follows the line connecting the lowest points along a streambed. The thalweg will meander back and forth across the riverbed in response to the changing course of the river, as shown in Figures 1 and 2. At many locations within the thalweg, the depth is sufficient to permit dredged material disposal without interference to navigation. Figure 3 illustrates this concept, before and after disposal.

Thalweg disposal has been proposed as a disposal alternative for uncontaminated sediments and as an alternative to the use of sidecasting dredges, which have the disadvantage of high disturbance and a tendency for redeposition of material in the cut. Thalweg disposal offers potential economic advantages, eliminating the need to transport dredged material to confined disposal sites, and the costs associated with acquisition, development, and maintenance of those sites.

The USAED, Rock Island, has reported costs of approximately $1.80 to 2.00 per cubic yard for thalweg disposal. Unit costs are influenced by the amount of material to be dredged and the distance to the disposal site. Typically, 4,000 to 5,000 ft of pipeline is required for a hydraulic dredging and disposal operation. Monitoring requirements of the disposal process and long distances can in some cases increase the cost of thalweg disposal over that of other riverine disposal methods (personal communication, January 1993, Richard M. Baker, USAED, Rock Island).

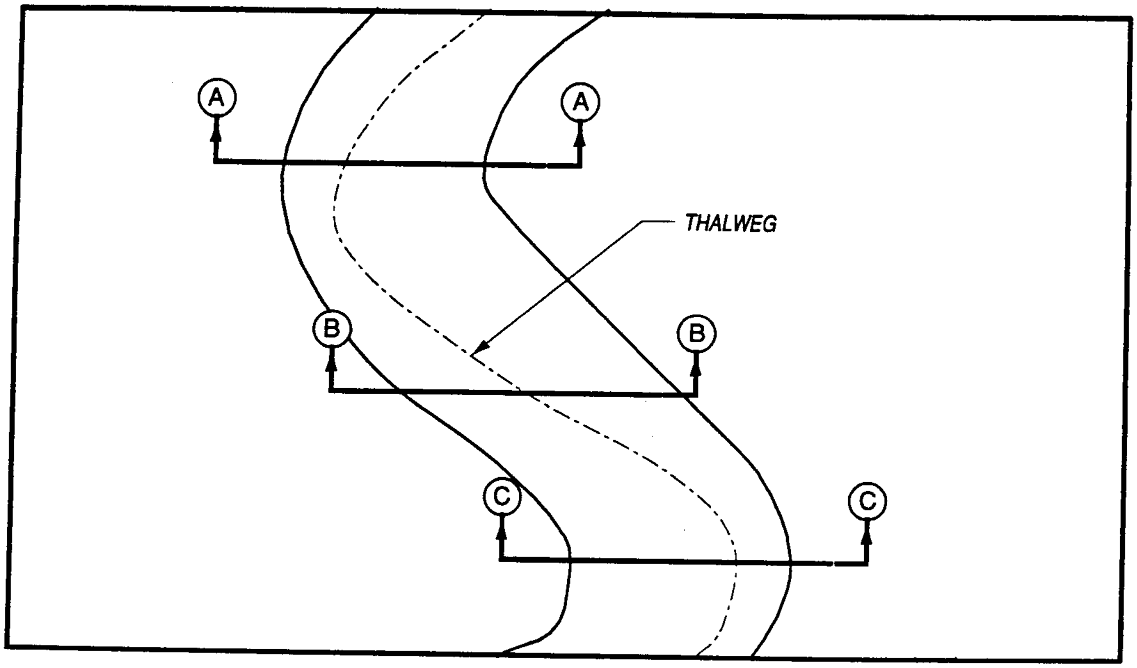

Figure 1. Line of the thalweg

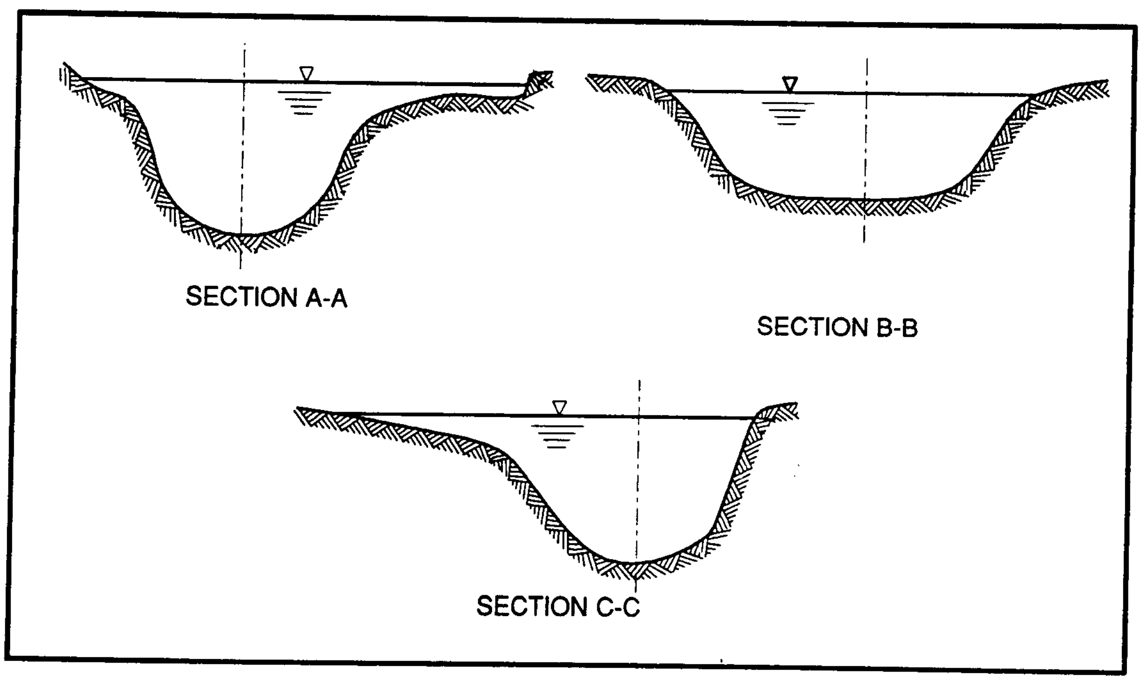

Figure 2. Section depicting location of the thalweg

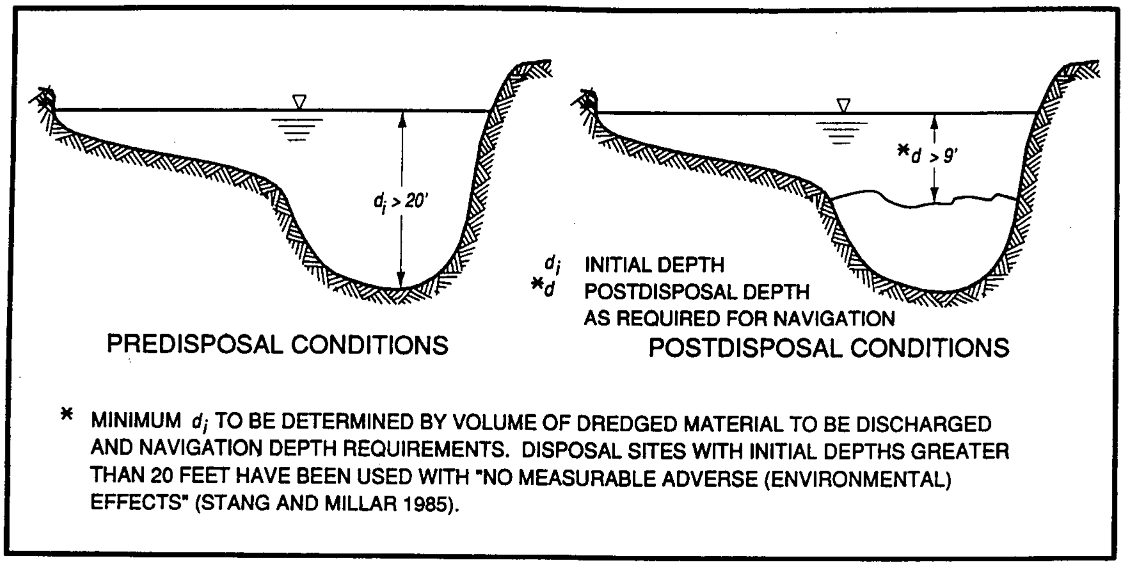

Figure 3. Section view of disposal site

Implementation

Application

Thalweg disposal has the potential to be an economic form of dredged material disposal which, when appropriately implemented, will have a minimal impact on the environment. River reaches requiring relatively low-volume dredging are the best candidates for thalweg disposal, while river reaches with divided flow are marginal candidates (although such reaches are most commonly dredged). Reaches that require heavy dredging should not be considered for thalweg disposal (Simons and Chen 1980). Thalweg disposal is most applicable to clean, sandy sediments, although in some cases it may be used for contaminated sediments as well, depending upon the nature and degree of contamination and the relative locations of extraction and disposal sites (EPA 1980).

Decision Structure

Implementation of thalweg disposal involves the reconciliation of various factors, including regulatory requirements, habitat preservation, and technical feasibility. Figure 4 illustrates the interdependence of these factors. The chronology of the decision structure will depend on what information is most readily available initially, coupled with those criteria that are most likely to be the limiting factors in the decision-making process. For example, the availability of potentially less damaging, or existing, disposal alternatives may negate further investment in evaluation of thalweg disposal. The availability of suitable disposal sites in reasonable proximity to dredging sites or the presence of contaminated sediments may possibly be determined from existing information, thus determining the next appropriate areas of inquiry and minimizing the evaluation process.

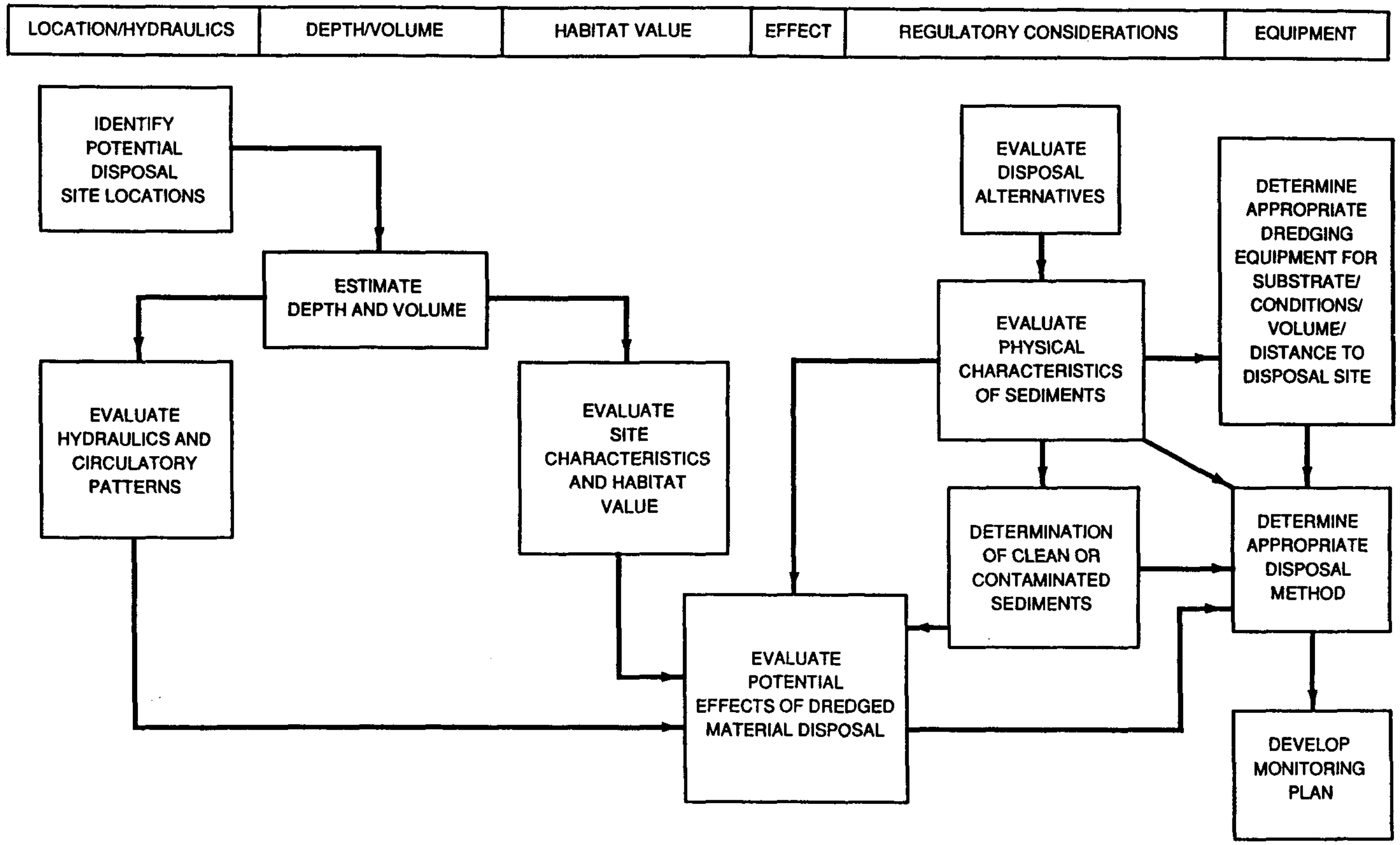

Figure 4. Interaction among operational/environmental and regulatory parameters in thalweg disposal

A feasibility study could begin with identifying locations where dredging is anticipated, or where it has historically been necessary, as well as potential thalweg disposal sites downstream of these areas which, from existing bathymetric information, would appear to be of suitable depth. Disposal alternatives for these locations could then be evaluated. If thalweg disposal appears to be a viable and justifiable alternative, further study would be initiated, according to the areas of importance outlined in Figure 4. Determinations can thus be made regarding the availability of suitable disposal sites, the technical and economic feasibility of the process, and environmental and regulatory acceptability. In general terms, the approach would be:

- **Collect available information**

  —Bathymetric surveys

  —Dredging logs

  —Sediment characterization (grain size, sediment chemistry, etc.)

- **Make preliminary determinations based on this information**

  —Proximity of potential thalweg disposal sites to dredging sites

  —Other disposal alternatives

  —Potential limitations to the process

    - Contaminated sediments
    - Obvious adverse site characteristics
    - Other considerations as indicated in Figure 4 and for which information is in hand

- **Proceed with further evaluation of most viable alternatives**

  —Addressing all areas of Figure 4:

    - Collect further information as required to proceed with evaluation
    - Identify concerns
    - Select most feasible and environmentally appropriate alternatives
    - Develop site-specific implementation plans
      - Boundaries of disposal site
      - Disposal volume
      - Seasonal disposal "window"
      - Monitoring and testing as required
      - Other pertinent considerations

Site Selection

Potential disposal sites should be identified well in advance of need so that site characteristics can be investigated. Environmental assessments and 404 evaluations may require as much as 2 years lead time prior to implementation of a site (personal communication, January 1993, Richard M. Baker, USAED, Rock Island). Preliminary identification can be made using dredging records and documented river geomorphology (Simons and Chen 1980). Sites that show evidence of high habitat value or potential for adverse effects due to undesirable alteration of current patterns and sediment movement are to be avoided.

As previously suggested, site selection will appropriately begin with identification of areas where disposal sites may be available within practical range of dredging sites (Simons and Chen 1980) and a review of available information to determine where the thalweg is of sufficient depth and volume to merit further consideration. Necessary depth will be determined from navigation requirements and the anticipated volume of dredged material to be discharged.

Detailed bathymetry of the disposal site and the reach 1 to 2 miles downstream must be obtained, provided the site is not first eliminated on the basis of other criteria (Figure 4). Potential disposal sites should then be evaluated on the basis of hydraulic characteristics. Millar (1986) recommends that the general morphology and hydraulics of the area for high- and low-flow periods be documented, and that bathymetric measurements be made of the disposal area and the thalweg/main channel for approximately 0.75 mile downstream.

The USAED, Rock Island, compiles detailed bathymetry of both the dredging and disposal sites, as well as nearby side channels and back waters in a 1- to 2-mile reach (personal communication, January 1993, Richard M. Baker, USAED, Rock Island). This distance will be site-specific, and typically will be determined in response to the concerns of local permitting agencies with respect to a particular river reach.

Depth

Stang and Millar (1985) recommend the use of sites that are at least 20 ft deep. Deeper holes will have correspondingly higher potential as disposal sites. Relative depth of the thalweg and side channel inlets is also of importance. The depth of the thalweg should exceed the depth of side channel inlets by at least 10 to 20 ft within potentially affected reaches (1 to 2 miles downstream of a disposal site) (personal communication, January 1993, Richard M. Baker, USAED, Rock Island).

Fate of Sediments

Thalweg disposal can potentially have an effect on circulation patterns and water-level fluctuations, as well as contributing sediment to reaches and structures immediately downstream of the disposal site. Lubinski (1984) suggested that, after placement in the thalweg, dredged material either remained at the site or was assimilated into the bed load, where it could then migrate in response to water currents.

Thalweg disposal has been used to some extent on the lower Mississippi River, and the Rock Island District has used and studied the procedure on the upper Mississippi River. Lower Mississippi dustpan dredging operations use the procedure when the river stage is such that access to holes downstream from the extraction site is possible. While there may be some movement of sediment out of the disposal site, in this area it constitutes a small fraction of the bed load, and effects are considered to be negligible (personal communication, August 1992, Mr. Larry Rabalais, USAE Division, Lower Mississippi Valley).

Studies of the movement of sand, tagged with fluorescent dye, from four test sites on the upper Mississippi River (Savannah Bay, Whitney Island, Gordons' Ferry, and Duck Creek) were conducted by the Argonne National Laboratory. Results of a 9-month observation of the Savannah Bay site (Paddock and McCown 1984) correlated closely with results obtained at the Whitney Island and Gordons' Ferry sites. This investigation revealed that contours of the disposal mound had been altered and dunes had developed, similar to the original bottom configuration of the river. Movement of tagged sand from the original site was observed, apparently confined to within the thalweg, and occurred in response to high river discharge.

At the Gordon's Ferry site, sampling that was conducted after a 5-year flood event (at a time approximately 20 months after disposal) revealed downstream movement of tagged sand for a distance of approximately 1,000 m.

Tagged material redistributed outside the thalweg was thought to be primarily fines and not representative of the characteristics of typical dredged material. It was concluded that "virtually no movement of dredged material into side channels occurs where the thalweg is at least 10 to 20 feet deeper than the channel inlet. Where the side channel inlet and the thalweg are of similar depth, however, migration of material into the side channel can be assumed. Side channel accretion may be due to sand input from the channel border area" (personal communication, January 1993, Richard M. Baker, USAED, Rock Island).

In the sites tested and sampled, the disposal mounds were eradicated by the first flood. Tagged sand appeared to have been incorporated into the bed forms of the natural channel (Ditmars, McCown, and Paddock 1986). A similar experiment conducted in a more complex reach with submerged wing dams on either bank resulted in a return to original depth within 5 months (September to January) after disposal (Ditmars, McCown, and Paddock 1986). Further monitoring of other, more diverse sites will be necessary to determine whether the behavior of these sites is representative.

As part of investigations conducted at Waterways Experiment Station for the St. Louis District, two tests in a physical movable-bed model were conducted for the Dogtooth Bend reach of the middle Mississippi River (miles 39.6 to 20.2). Considerable channel stabilization work was been done at this location, including weirs, dikes, and revetments, all designed to increase channel depth and improve navigation. The model used granulated coal as both dredged material and bed medium. Plastic particles were mixed with the dredged material to act as tracers. One test examined disposal along the opposite bank from the dredge cut and in scour holes off the ends of dikes. True thalweg disposal was not examined. Sediment transport, rate of movement, and areas of deposition were examined and recorded. Preliminary results were encouraging in that, for the limited testing performed, material deposited in the scour holes at the stream end of dikes did not negatively impact the navigation channel in the two bends and crossing downstream of the disposal site. However, a more intensive study would be needed to determine if results were representative of the behavior of sediments in natural channels.

Hydraulics

The following are some general guidelines to site selection on the basis of hydraulics:

- Where a disposal site is located upstream of an island, adverse effects may result if the thalweg current is of equal force on both sides of the island, or if there is more force down the side channel than in the main channel (Millar 1986).
- Thalweg disposal should not be used where the depositional pool or downstream crossing is not of adequate depth to handle the material without further dredging (Millar 1986).
- No thalweg disposal site should be located within 2 miles upstream of a high-volume dredging site (Simons and Chen 1980). [The Rock Island District has used thalweg disposal within 1 mile of a high-volume dredging site with no adverse impacts on the site. Increased scour was noted on the next crossing. The cause-and-effect relationship in this instance was not determined, but the site has been used three times with similar results (personal communication, January 1993, Richard M. Baker, USAED, Rock Island).]
- When the disposal site is located adjacent to or immediately upstream from the entities listed below, the potential for adverse effects due to sediment movement from the disposal site, during and after disposal, exists. These entities include

  —Tributaries.

  —Hydraulic/navigational structures.

  —Water supply intakes.

  —Important habitat.

—Submerged artifacts.

—Recreational or commercial fisheries.

—Other sensitive areas, by site-specific determination. [The Rock Island District has not experienced any problems due to thalweg disposal near navigational structures (training works) (personal communication, January 1993, Richard M. Baker, USAED, Rock Island).]

- No side channels and backwater areas should be located within 1 mile downstream of the disposal site (Simons and Chen 1980). [Side channel entrance depth relative to thalweg depth appears to be the most critical factor. Where the thalweg is at least 10 to 20 ft deeper than the side channel inlet, movement of dredged materials into the inlet is not expected to occur. Where they are of similar depth, material migrating from the disposal sites may accrete in the side channel inlet (personal communication, January 1993, Richard M. Baker, USAED, Rock Island). This would not be reflected in a one-dimensional analysis that assumes equal transport through the cross section.]

The hydraulics of such locations must be carefully evaluated if they are to receive further consideration. Postdisposal monitoring, with an action plan for intervention, may be advisable. An intervention trigger might be a specified change in bathymetry, or increased turbidity above background levels (personal communication, January 1993, Richard M. Baker, USAED, Rock Island).

Environmental Effects

The habitat value of potential disposal sites must also be evaluated. In general, sites with the following characteristics should not be considered for thalweg disposal:

- Significant numbers of aquatic or benthic organisms.
- Presence of endangered species.
- High species diversity.
- Dormant species at time of disposal.
- Bottom-dwelling species.

Avoidance of sites with these characteristics is a primary objective in evaluation of potential disposal sites. A careful evaluation of depth, substrate, and water temperature will be the primary indicators of habitat potential of a site. Because the thalweg is a highly dynamic environment, its physical, chemical, and biological attributes may change on a seasonal basis or in response to changes in water level. In specifying a disposal site, these fluctuations should be taken into consideration. Thalweg disposal should be seasonally restricted as appropriate for local conditions and habitat use. Disposal should be restricted to materials with characteristics (grain size, level of compaction, etc.)

similar to the disposal site (personal communication, January 1993, Richard M. Baker, USAED, Rock Island).

In general, coarse stable substrates and structures that provide a current break have demonstrated habitat value. Areas with low velocities or where natural back eddies exist are also good habitat. Sites with high velocities and unstable substrates are generally least valuable as habitat. These conditions are often found on outside bends, which may appropriately be given first consideration in initial site evaluation.

Environmental Effects of Dredging Technical Notes EEDP-01-30, "Environmental Effects Evaluation for Thalweg Disposal of Dredged Material" (Olin 1993), gives detailed information on evaluating potential thalweg disposal sites for various environmental concerns.

Regulatory Considerations

Thalweg disposal is a form of open-water disposal and, as such, is regulated under Section 404 of the CWA. The "Guidelines for Specification of Disposal Sites for Dredged or Fill Material," as outlined in 40 CFR 230, apply to disposal site determination (EPA 1980). Under Section 404 of the CWA, specification of disposal sites and evaluation of dredged material for open-water disposal are addressed. Once a disposal site is specified, a contaminant evaluation of the material must be done. In general, material proposed for thalweg disposal will meet the exclusionary criteria outlined in 40 CFR 230.60, and the testing described in 40 CFR 230.61 need not be performed. The 404(b)(1) evaluation must include State water quality certification as described in Section 401 of the CWA.

Dredging Methods and Equipment

As with any dredging operation, selection of suitable equipment for the sediments, depth, traffic, and adjacent structures is the major consideration. In some cases, it may be necessary or desirable to minimize sediment resuspension during dredging and disposal, which places further requirements on equipment selection. Section 33 CFR 323.2(d) addresses the status of "de minimis incidental soil movement" resulting from "normal dredging operations."

In general, hydraulic dredging is suited to the extraction of loosely compacted materials and results in a slurry with a high water content. Thus, hydraulic dredging can minimize disturbance at the extraction site, but generally contributes to wider dispersion at the disposal site (Palermo and others 1992). Mechanical dredging is appropriate to a wider range of substrates, and materials removed by mechanical dredging remain at or near their in situ density. This minimizes turbidity at disposal.

Constraints on disposal options will be dictated by type of dredging equipment selected, the suspended solids requirements, and distance to disposal site. If dustpan dredges are used, the maximum pipeline length is approximately

800 ft, which limits thalweg disposal to this distance (personal communication, August 1992, Larry Rabalais, USAE Division, Lower Mississippi Valley). Pipeline cutterhead dredges can pump through 5,000 ft of pipeline without additional booster pumps (personal communication, September 1992, Dr. Michael Palermo, U.S. Army Engineer Waterways Experiment Station).

Materials suspended during disposal are regulated under Section 401 of the CWA. Open-ended pipeline disposal, above and parallel to the water surface, maximizes dispersion and produces a thin, widely spread sediment layer. Turbidity can be minimized by using submerged discharge or submerged discharge with diffusers for hydraulically dredged sediments. Where depths exceed 6 ft, dispersion can be decreased by vertically discharging the slurry through a 90-deg elbow at 1.5 to 3 ft below the water surface (Simons and Chen 1980). A vertically oriented, 15-deg axial diffuser with a cross-sectional area ratio of 4 to 1, followed by a combined turning and radial diffuser section that increases the overall area ratio to 16 to 1, can reportedly eliminate most turbidity (Simons and Chen 1980, citing Barnard 1978). Mechanically dredged sediment discharged from barges also results in lower suspended solids levels at disposal.

Hydraulic disposal of materials in discrete mounds to simulate the structure of large dunes has been implemented by the Rock Island District. The environmental advantages of this disposal method relative to disposal in one large mound are not yet known.

In general, the disposal method that is selected must allow for accurate placement of the material in the disposal site, must be technically and physically feasible, and must enable the discharge to conform to the requirements of Section 401 of the CWA.

Monitoring

The thalweg is a dynamic environment, and seasonal changes in physical, chemical, and biological attributes may occur. These changes should be taken into consideration when specifying a site for disposal; however, if a site cannot be located to avoid potential unacceptable adverse environmental effects, postdisposal monitoring may be necessary. If so, the guidance by Fredette and others (1990) should be followed. In particular, "a prospective monitoring program requires that changes in resources at risk be quantified and that the threshold at which changes become unacceptable be explicitly specified."

Although not "monitoring" in the regulatory sense, periodic checks of the area below the dredging site are recommended during dredging and disposal to identify problems that may develop during operations. This may consist of sounding the area with a bathometer every 2 to 3 days during operations to identify areas of excessive accretion or drift of dredged material back into the cut (personal communication, September 1992, Mr. Larry Rabalais, USAE Division, Lower Mississippi Valley). The Rock Island District recommends more frequent monitoring, as much as once every hour, until the rate and pattern of deposition for a particular site have been established. Postdisposal monitoring is also practiced to obtain data needed for documentation and justification of thalweg disposal (personal communication, January 1993, Richard M. Baker, USAED, Rock Island).

Summary

"Thalweg disposal is placement of dredged material in a deep-water portion of the channel thalweg where it will become a natural element of the sediment transport system" (Nanda and Baker 1984). It mimics the natural low-water scour and accretion of crossings and pools.

Thalweg disposal is an economically viable disposal alternative for appropriately located reaches that require low-volume dredging. Most suitable for clean sediments, the process may be used for disposal of contaminated sediments under certain circumstances. The process of implementing thalweg disposal requires an evaluation procedure by which the important considerations can be reconciled. Figure 4 illustrates the interdependency of the variables involved.

In most cases, potential disposal sites can be identified on the basis of dredging logs and existing bathymetric information. More extensive evaluation can then be restricted to the most promising sites. Location, depth, hydraulic characteristics, and habitat value must all be evaluated, in conjunction with regulatory requirements and feasible dredging techniques.

Thalweg disposal can be reconciled with regulatory requirements for dredged material discharges. As a form of open-water disposal, thalweg disposal is regulated under Section 404 of the CWA. The 404(b)(1) evaluation must include State water quality certification, based on Section 401 of the CWA.

Dredging equipment will be selected in much the same manner as for any dredging operation, with consideration given to sediment characteristics, depth, traffic, adjacent structures, and the presence of contaminants. Where existing turbidity is low, contaminants are present, or where required by regulation, dredging and disposal methods that minimize dispersion and levels of suspended solids may be necessary.

When a disposal site cannot be located to avoid potential unacceptable adverse environmental effects, postdisposal monitoring may be needed. If so, the guidance in Fredette and others (1990) should be followed. Bathymetric monitoring is advisable during and following disposal for accurate material placement and documentation of subsequent effects.

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Environmental Effects of Dredging Technical Notes

Managing Dredged Material Via Thin-Layer Disposal in Coastal Marshes

Purpose

This technical note describes how dredged material can be successfully managed in an environmentally sound manner in marshes by placing it in layers of 5 to 15 cm. (Unless otherwise indicated, all layer thicknesses indicated in this report refer to material that has undergone postdisposal consolidation.) Environmental studies of this process and of the regulatory history of thin-layer disposal in marshes are summarized. General planning and monitoring considerations are described, including descriptions of the types of equipment used to place dredged material in thin layers in marshes.

This note complements *Environmental Effects of Dredging* Information Exchange Bulletins, Volumes D-92-1, D-92-3, and D-92-5, which describe case histories of thin-layer disposal, and an upcoming *Environmental Effects of Dredging* technical note, which will provide additional detail on engineering aspects of managing dredged material by thin-layer disposal. Together, these documents provide guidance for the planning, execution, and monitoring of thin-layer disposal in marshes.

Background

Channels that pass through marshes can be difficult to dredge, because dredged material cannot be readily placed in marshes without impairing wetland functions. Hence, effort is spent finding scarce upland sites, or additional costs are incurred transporting material to other areas. To help alleviate this situation, several groups have proposed that thin-layer disposal (hydraulically placing dredged material in single layers of 5 to 15 cm) will reduce environmental impacts sufficiently that disposal in some marshes may become acceptable. If true, maintaining channels that pass through wetlands, especially those in remote areas, may be facilitated.

Although thin-layer disposal potentially can reduce environmental impacts in several types of habitat, few reviews have been conducted of the environmental effects of this disposal technique. This note and the earlier information bulletins provide additional reviews needed to determine when thin-layer disposal in marshes is an effective disposal option for dredged material.

Additional Information

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Regulatory History

Anecdotal accounts indicate that thin-layer disposal in marshes has been used intermittently as a management technique since the 1930s for channels that pass through marshes, although this practice reflected engineering constraints more than efforts to minimize environmental impacts. Early bucket dredges often could not place material far enough away from a canal to prevent it from slumping back into the canal (Williams 1944, McGhee and Hoot 1963). To remedy this situation, relatively low-pressure hydraulic dredges were used to spray material into marsh away from the canal bank. By the 1950s, bucket dredging technology had improved, and by the 1960s it was generally more cost effective than hydraulically dredging these canals.

During the 1970s and early 1980s, opposition to placing dredged material in marshes mounted, as the value of these habitats to fisheries and water quality became more clear (Davis 1973, Nixon 1980) and it was realized that wetlands outside disposal areas were also affected by these practices (Scaife, Turner, and Costanza 1983, Swenson and Turner 1987). In response to these concerns, relatively high-pressure spray technology was developed for placing dredged material in the late 1970s, but this practice has not been used widely.

Thin-layer disposal has been required (via Section 404 permits or analogous state approvals) for managing dredged material in marshes for only a few projects in Georgia, Louisiana, North Carolina and, possibly, Florida. (Thin-layer disposal has been done several times in Florida, but it is unclear whether this method was mandated by regulatory agencies or was simply the choice of construction managers.)

Thin-layer disposal in marshes as a technique for managing dredged material has received wide discussion only in Louisiana, where thousands of hectares of remote marsh are crisscrossed by oil-rig access canals. For 10 to 20 of these projects during the mid-1980s, several regulatory agencies, most notably the U.S. Fish and Wildlife Service and the National Marine Fisheries Service, suggested that Section 404 permits and analogous state approvals require that thin-layer disposal be used to minimize environmental impacts in and around disposal areas (Cahoon and Cowan 1988). However, cost considerations resulted in issuance of relatively few (5 to 10) permits with such stipulations (LaSalle 1992). At that time in Louisiana, hydraulic thin-layer disposal was 2 to 14 times more expensive than conventional bucket dredging (Cahoon and Cowan 1988). Further, dredging many of the access canals also involved constructing a dock or platform for drilling machinery, work that could be done using a bucket dredge's derrick. Thus, a bucket dredge resulted in less mobilization/demobilization cost for the overall project. In several cases, applicants for permits modified projects to make them acceptable to review agencies without having to resort to thin-layer disposal to minimize impacts (LaSalle 1992).

Environmental Effects

Case Studies

Instances of thin-layer disposal of dredged material in marshes or uplands were identified in Florida, Texas, Georgia, North Carolina, and Louisiana; however, only four formal studies of environmental effects of this management practice were found. Reimold, Hardisky, and Adams (1978) used 0.6-m^2 plots along St. Simons Sound, Georgia, to examine the effects of thin-layer disposal on *Spartina alterniflora* (cordgrass). Corrugated metal pipe was driven 122 cm into the ground to create each enclosure. Six layer thicknesses (8, 15, 23, 30, 61, and 91 cm), three dredged material types (sand, silty sand, and silt), and three discharge times (late winter, summer, and fall) were examined for up to 21 months (two growing seasons) after disposal. The layer thicknesses indicated above were prior to postdisposal consolidation and were achieved by shoveling material into the rings.

Layer thickness was the most important factor. Placement of material smothered most stems. Recovery of the vegetation occurred by either new shoots arising from rhizomes or by seeds germinating at the surface of the dredged material, the latter process being much slower than the former. Recovery from the 8- to 23-cm layers was generally from new shoots penetrating the dredged material, with seedlings accounting for the limited recovery of the 61- and 91-cm layers. More shoots emerged from the sandy and silty-sand material than from the silty material. However, shoots emerging from the silty material tended to have a higher biomass, perhaps reflecting the higher nutrient content of the material or reduced competition for nutrients from other shoots.

At the end of the experiment, there was little variation in vegetation abundance due to discharge time, and differences that were present partly reflected differences in length of the postdisposal monitoring (21, 16, and 11 months for the late-winter, summer, and fall discharges, respectively). It was unclear if, at the end of the experiment, complete recovery had occurred from the 8- to

23-cm layers. Biomass in these plots was considerably lower than in nearby reference marshes, but approximated levels seen in plot controls (enclosures that received no dredged material). Hence, the pipe used to create the enclosures may have introduced artifacts (for example, shading and reduced groundwater movement) that prevented full recovery of the vegetation to background levels.

Cahoon and Cowan (1988) semiquantitatively examined two brackish marshes in Louisiana up to 11 and 17 months after disposal of material excavated for small new-work channels and barge slips. At Dog Lake, about 14,400 m^3 of silty-clay material was placed in a layer 10 to 15 cm thick up to 70 m from the canal edge. At Lake Coquille, about 8,000 m^3 of silty-clay material was placed in a layer 18 to 38 cm thick up to 80 m from the edge.

At both sites, placement of dredged material smothered most of the aboveground vegetation. Eight to 14 months later (about one growing season), limited recolonization by *S. alterniflora*, *Salicornia* spp. (glassworts), and *Distichlis spicata* (saltgrass) was evident, presumably via new shoots emerging from old rhizomes. Three months later (midway through the second postdisposal growing season), vegetation cover had increased but had not yet reached the presumed predisposal levels. Only the Lake Coquille site had wetland area converted to upland habitat by dredged material, but the extent of this alteration was limited to less than 100 m^2 (0.025 acre). No obvious obstructions to water flow were created by the dredged material at either site.

In addition to Dog Lake and Lake Coquille, Cahoon and Cowan visited two floating roseau cane (*Phragmites australis*) marshes soon after approximately 15,000 m^3 of material was placed upon each. At both sites, no accumulation of dredged material was apparent because the material sank into the extremely soft substrate. However, at both sites, much of the standing vegetation had been crushed.

LaSalle (1992) returned to Cahoon and Cowan's Dog Lake and Lake Coquille sites in 1992, about 6 years after disposal. Both marshes had healthy stands of vegetation (Figure 1, upper panel). Species distributions and abundances in the Lake Coquille disposal area were similar to nearby reference areas. However, the Dog Lake disposal and reference areas differed in several ways. The disposal area consisted predominantly of *S. alterniflora* and *Salicornia* spp., whereas *D. spicata*, *Juncus roemerianus* (needle rush), and *S. alterniflora* dominated reference areas. Further, shoot density was about 20 percent less in the disposal area. In both areas, sediment cores exhibited a layered structure (Figure 2). The top few centimeters consisted of roots and rhizomes from the existing marsh. Below this was 10 to 20 cm of compact silt/clay material that appeared to be dredged material. Below this was another few centimeters of roots and rhizomes, which presumably represented the predisposal marsh. In contrast to Cahoon and Cowan's earlier observations, the apparent dredged material layer was thinner at Lake Coquille (10 to 15 cm) than at Dog Lake (15 to 20 cm).

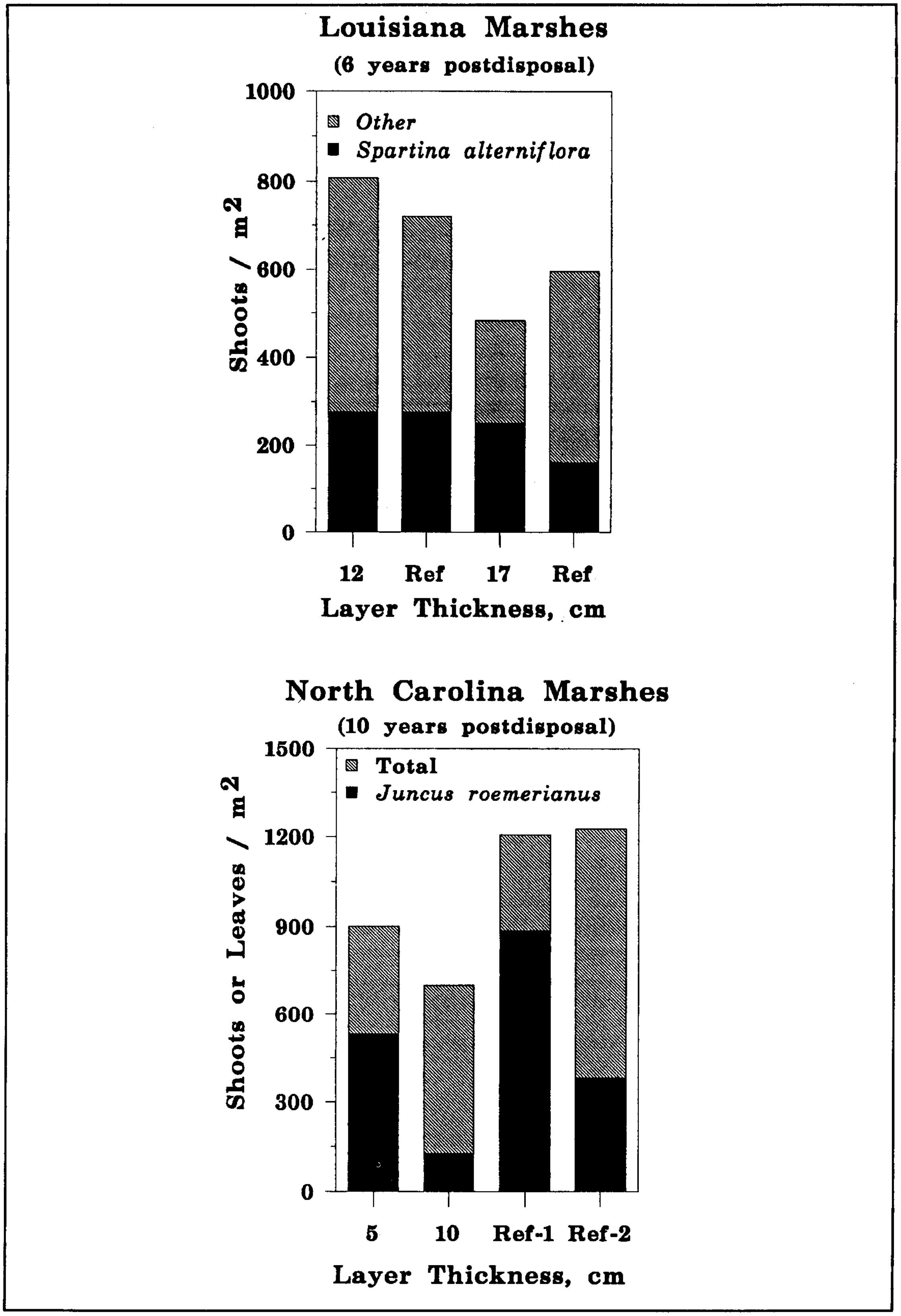

Figure 1. Vegetation densities in disposal areas of marshes used for thin-layer disposal. (For comparison, densities at reference sites are also given)

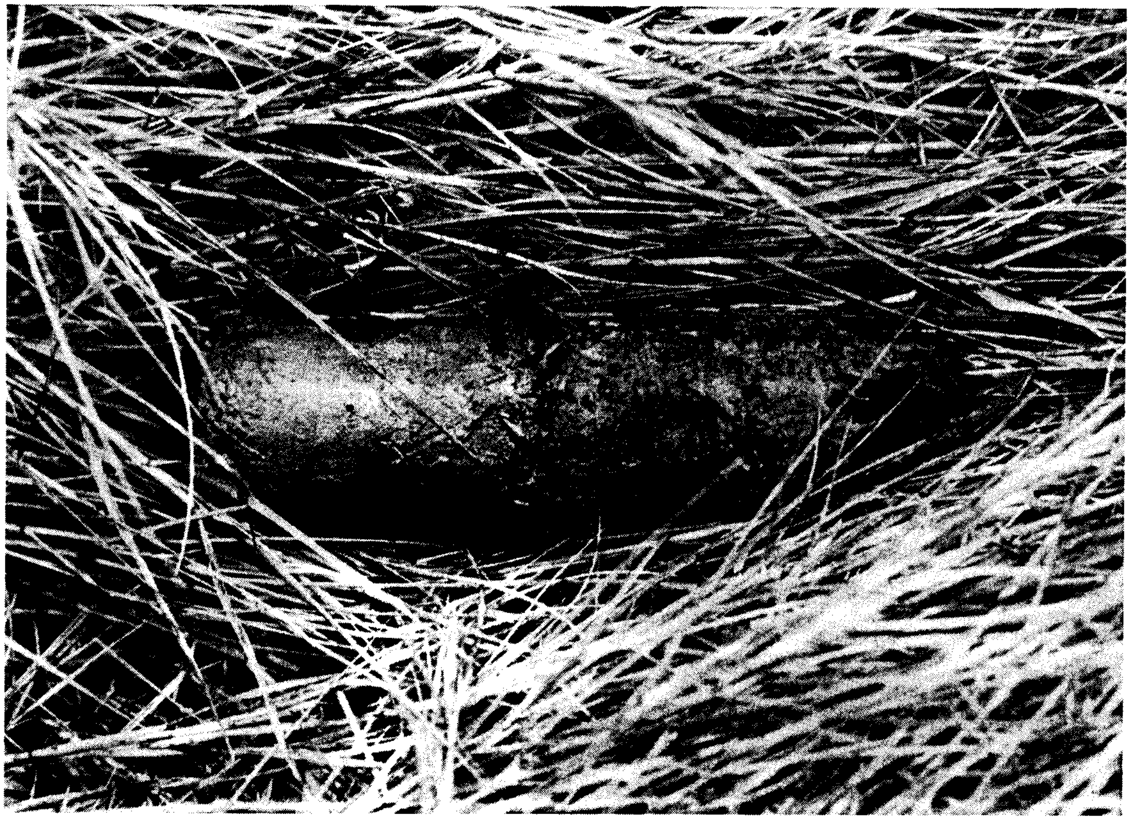

Figure 2. Sediment core from Lake Coquille, Louisiana, 6 years after thin-layer disposal

Wilber, Luczkovich, and Knowles (1992) examined an oligohaline marsh in Gull Rock, North Carolina, approximately 10 years after it had been used for thin-layer disposal of 8,000 to 12,000 m^3 of mostly silty maintenance material. The two disposal areas examined had healthy stands of vegetation, but nonetheless, some differences were apparent when compared to reference areas (Figure 1, lower panel). An area where the disposal layer was about 5 cm thick had slightly less *J. roemerianus* than an adjacent reference area, and shoot density was 25 percent lower. An area where the disposal layer was about 10 cm thick was dominated by *D. spicata* and *S. alterniflora,* whereas the reference areas were dominated by *J. roemerianus* and *D. spicata.* Shoot density at this site was 40 percent lower than at reference areas. Although there were small differences in the plant community, estimates of infauna abundance and use by fiddler crabs and larval fish were similar to reference areas.

Several other groups are currently examining environmental effects of thin-layer disposal. During January-March 1993, the city of Savannah, GA, placed about 30,000 m^3 of sandy material in a 10- to 20-cm layer of a tidal-freshwater forested wetland. The city will monitor disposal areas for 3 years. Plaquemines Parish, LA, is examining effects of thin-layer disposal projects at West Pointe-a-la-Hache and La Reussite. However, the Louisiana projects involve diverting fresh water to a brackish marsh, an intentional habitat change that limits the scope of inferences that can be drawn from the Plaquemines Parish studies.

Related Studies

Two common types of natural disturbance, dune overwash and wrack deposition, are qualitatively similar to thin-layer disposal of dredged material and provide some insight about the long-term effects of this management technique. Severe storms and hurricanes transport large mats (500 to 1,500 m^2) of dead vegetation into the upper region of marshes; the thickness of the wrack layer can be 20 to 30 cm. The wrack has the immediate effect of smothering existing vegetation. The area then recovers as wrack decomposes or is relocated by subsequent storms. Reidenbaugh and Banta (1980), Bertness and Ellison (1987), and Hartman (1988) examined wrack accumulation in *Spartina* marshes and concluded that almost complete recovery occurs in two growing seasons if roots and rhizomes are not killed. Knowles (1989) examined this process in a *Juncus* marsh and found that recolonization can occur at a similar rate, but species composition may change.

Marsh vegetation commonly occurs on the lee side of sand dunes along the eastern coast of the United States. When hurricanes and other storms overwash these dunes, sandy material often smothers this vegetation. Zaremba and Leatherman (1984) found that recovery from these disturbances via new shoots arising from roots and rhizomes varies with species, initial cover, and elevation. *Spartina patens* was able to penetrate up to 33 cm of material, and *S. alterniflora* was able to penetrate up to 24 cm of material. Less quantitative data are available demonstrating recovery by other marsh grasses and shrubs.

Other information indicates potential problems from thin-layer disposal in marshes. Mendelssohn, McKee, and Patrick (1981), King and others (1982), and DeLaune, Pezeshki, and Patrick (1987) discuss the effects of water-logged soils and high sulfide concentrations on marsh vegetation. In poorly drained soils, decomposition of organic material can lead to hypoxic conditions inconducive to plant growth. Since dredged material is placed hydraulically in a thin-layer operation and water volume can exceed material volume 10-fold, significant alteration of soils could occur. Finally, numerous studies of wetland creation (Broome 1989, Lewis 1989) show that elevation changes as small as 5 cm can significantly alter vegetation patterns.

A General Model for Marsh Recovery

The above studies can be synthesized into a conceptual model of how marshes respond to a thin-layer disposal event (Figure 3). Dredged material is hydraulically placed onto the marsh with some type of spray device. The distance of the spray and the texture of material within it depend upon equipment and operation. Placement of material will smother standing vegetation, although the cause may be the large amounts of water used in placement rather than dredged material itself. Rate of recovery depends upon layer thickness and the extent to which soil characteristics are altered. If a substantial number of roots and rhizomes survive the hypoxia and high sulfide conditions that often result from water-logged soil and decomposing vegetation, new shoots will arise. If enough new shoots penetrate the dredged material, new

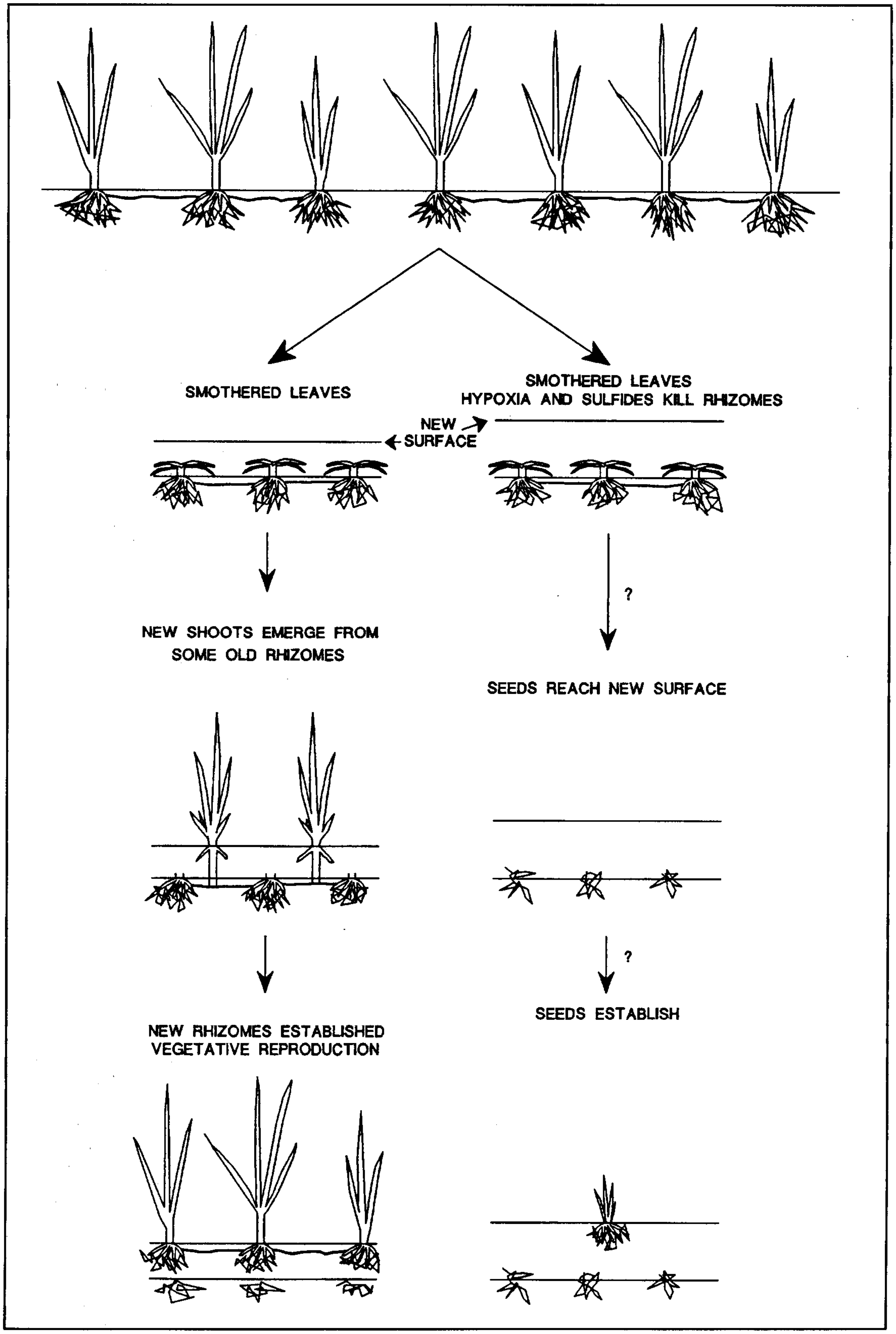

Figure 3. Illustration of conceptual model for marsh recovery after thin-layer disposal

adventitious roots and rhizomes will occur at the newly appropriate soil depth; old roots and rhizomes will be abandoned. Areas where new shoots did not arise will be subsequently colonized via vegetative growth.

This process generally requires two growing seasons to reach vegetation densities commonly found in marshes, but may require longer to reach predisposal levels. Marsh plants differ in their ability to withstand this type of stress and reproduce vegetatively. Hence, species composition of the new marsh may differ from the old marsh. However, if the new elevations remain within the marsh range (which varies with tidal range but can be crudely approximated by mean low water to mean higher high water), the new community will still be a marsh rather than an upland.

If too many roots and rhizomes are killed by altered soil conditions or if too few shoots penetrate the dredged material, the bulk of recolonization will be by seedlings, assuming marsh elevations are preserved. This method of recolonization will require considerably longer than two growing seasons to establish typical marsh vegetation patterns and may allow erosive forces to prevent recovery from occurring at all. Thus, the key to successfully managing dredged material in marshes with thin-layer disposal is placing material in a manner such that severe hypoxia and sulfide levels do not result and new shoots can penetrate the dredged material. Studies of thin-layer disposal in Louisiana and North Carolina show this goal can be reliably achieved with layers of 5 to 15 cm.

Project Planning and Monitoring

General Planning Considerations

Although detailed engineering analyses of thin-layer disposal in marshes have not been done, determining appropriate layer thickness and estimating a marsh's disposal capacity are the most important steps. Before exploring these steps, it is necessary to understand various aspects of dredged material solids concentration or volume, and the changes that occur during dredging, disposal, and postdisposal. Initially, the volume of sediment and its concentration are known in situ, yielding the total mass of solids to be dredged and disposed. During hydraulic dredging, water is mixed into the sediment to create a slurry that can be pumped; therefore, the volume of dredged material is 4 to 7 times as large as the in situ volume, particularly for new-work dredging of fine-grained material (Headquarters, U.S. Army Corps of Engineers 1987).

During disposal, the dredged material slurry undergoes sedimentation, and supernatant water runs offsite. The volume of dredged material continues to decrease as the material undergoes compression settling. Immediately following disposal, the material volume may still be 2 to 4 times larger than before dredging.

During postdisposal, the dredged material continues to densify by self-weight consolidation and desiccation. The rate of densification for thin-layer disposal will be fast since the drainage length for the water to escape from the material is small. Complete densification should occur in less than a year, with the actual rate a function of soil permeability, location of the water table, evaporation, and other climatic factors. Final volume may be somewhat less than in situ volume (0.7 to 1.1 times) for maintenance dredging and somewhat more than in situ (1.3 to 2 times) for new-work dredging.

Studies of thin-layer disposal in Louisiana and North Carolina show healthy stands of marsh vegetation atop 5- to 15-cm layers of dredged material. Since the thicknesses of these layers were measured months to years after disposal, these can be considered postconsolidation thicknesses. The question then arises, What were the immediate postplacement thicknesses? Reimold, Hardisky, and Adams (1978) found that 8- to 91-cm layers of dredged material shrink 10 to 40 percent in thickness by 10 days after placement, and the shrinkage rate was inversely related to initial layer thickness and did not differ between dredged material types. Using these results and assuming no other processes were involved, the immediate postplacement thickness of dredged material in the above studies would have been approximately 8 to 22 cm. Standard engineering analyses indicate that the immediate postplacement thickness depends on grain size, and could have been as much as 15 to 45 cm.

Elevation data from nearby wetlands should be used to determine which part of the postconsolidation range should be targeted. If similar marshes occur at elevations 10 to 15 cm higher than the ambient predisposal marsh, the upper portion of the range may be appropriate. Otherwise, a postconsolidation change of 5 to 10 cm should be targeted, unless this range would bring the marsh to upland elevations, in which case thin-layer disposal should not be attempted.

In practice, wetland thin-layer disposal sites have been sized by estimating the volume of material to be excavated per meter of channel length and then calculating how wide the disposal area needs to be to reduce that volume to a given thickness (personal communication, January 1993, R. Hallman and J. Sawyer, City of Savannah, Savannah, GA).

Figure 4 illustrates such calculations for a range of layer thicknesses. It should be noted that the disposal area width shown in Figure 4 is for the *total disposal area*. Thus, if material is placed on both sides of a channel, the two widths (one from each side of the channel) are summed to yield total disposal area width. In making these calculations for a specific project, certain special circumstances should be considered. If the channel makes a severe bend, disposal swaths may overlap, locally reducing disposal capacity. One advantage of thin-layer disposal is that placement of dredged material in creeks, sloughs, and other sensitive areas can be readily avoided by redirecting the discharge. However, the cost for such avoidance is reduced disposal capacity, which should be considered when planning a project.

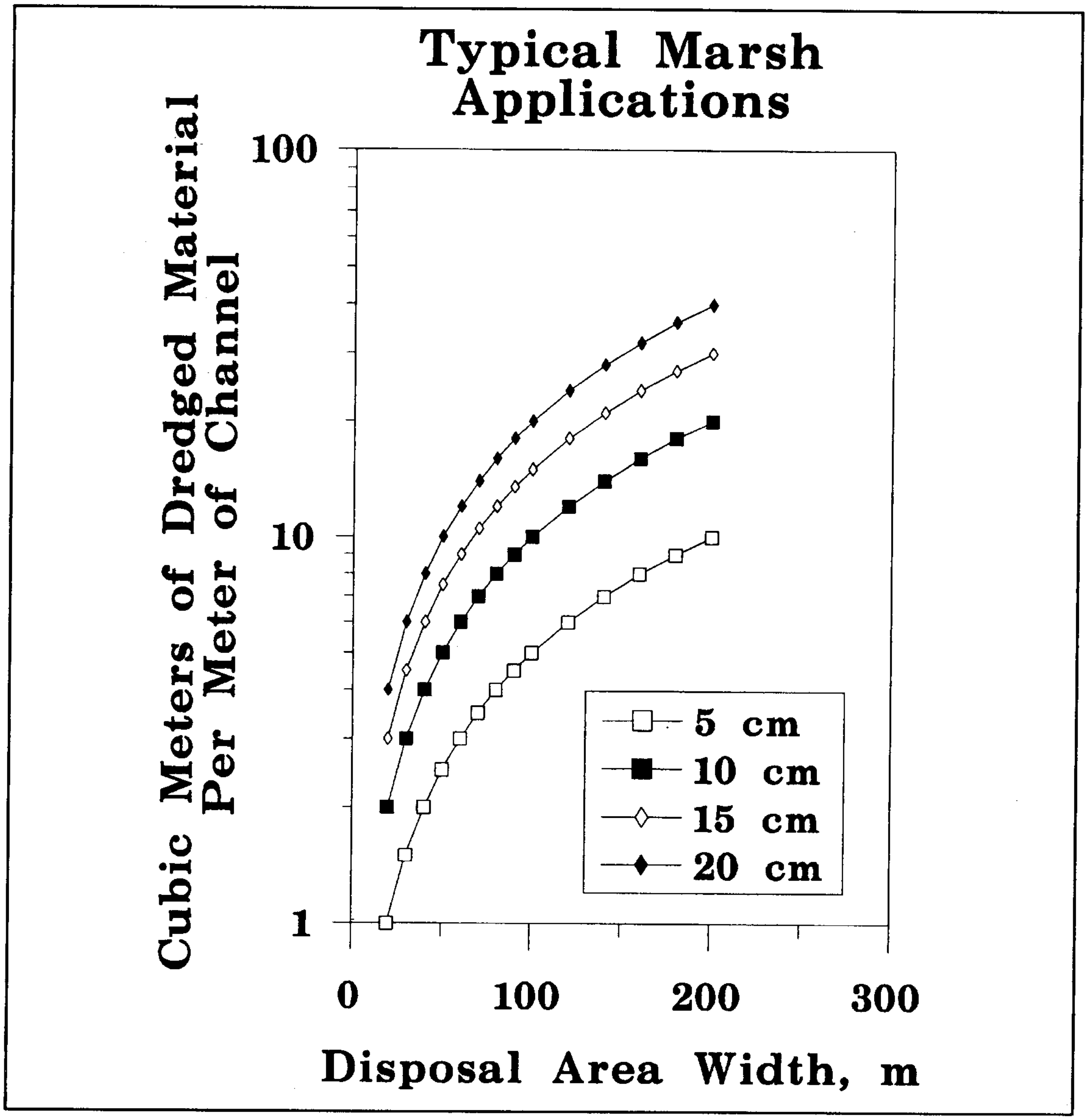

Figure 4. Nomogram that indicates relationship between total disposal area width and volume of dredged material per meter of channel for selected layer thicknesses

Equipment and Monitoring

At present, thin-layer disposal in marshes is accomplished by slurrying dredged material and spraying it onto nearby marshes. In almost all cases, the cutterhead, pump, and spray device occur on the same vessel; in a few cases, the pump and spray device were connected by a few hundred meters of pipe.

The type of cutterhead chosen is determined by the nature of the material to be dredged. Horizontal auger cutterheads have been used for fine material, and radial cutterheads for sandy material. In either case, the goal is to turn material into a fine slurry. Both high- and low-pressure hydraulic dredges can be used, although high-pressure dredges can spray material farther, which potentially increases disposal capacity. A high-pressure system that includes cutting blades in the pump impeller has been patented under the name JET-SPRAY, but other equipment can be used in these operations.

Since it is relatively easy to control the direction of the spray device, a thin-layer disposal operation can avoid marsh creeks, sloughs, and other sensitive areas within a disposal site. Although control at this level is relatively easy, precisely controlling the thickness of the dredged material layer has proven difficult. No thin-layer disposal site has been thoroughly examined to determine how close actual layer thicknesses were to target thicknesses. The limited available data indicate layer thickness will vary by at least 10 cm.

The variability in layer thickness probably results from several factors. First, there is a lack of real-time feedback from the disposal area to the dredge operator. Because of the large amounts of water involved in slurrying the material and because the marshes suitable for thin-layer disposal have little slope, water can accumulate in the disposal area and hide the dredged material layer from view, making it difficult to monitor. To deal with this situation, arrays of large buckets with bottom drain holes are often placed in the disposal area to catch dredged material. However, turbulence from the raining material may keep material in the bucket partially suspended if drains are not working properly. Second, trees and wind deflect the spray from its intended target. Third, although spray ranges can be 80 m, fallout along that range is not even leading to uneven accumulations. However, placing a deflector plate a few centimeters from the spray nozzle reduces this problem (personal communication, January 1993, J. Sawyer, Savannah, GA). Since the only way to deal with layers thicker than planned is to stop dredging, it is extremely important to accurately determine material volumes and disposal site capacity.

Other Uses of Thin-Layer Disposal Technology

Thin-layer disposal, as discussed here and in the previous Information Exchange Bulletins, is defined narrowly to focus on how the practice minimizes environmental impacts from dredged material disposal. However, thin-layer disposal technology has other applications suited to beneficial uses of dredged material. Eustacy and subsidence are increasing the submergence of many marshes in Louisiana, causing the marshes to deteriorate and disappear. Wilsey, McKee, and Mendelssohn (1992) have shown that *S. alterniflora* transplanted to these dieback areas is more likely to become established if elevations are raised 30 cm. Access to deteriorating interior marshes is a problem that requires technological innovation, but the basic principles of thin-layer disposal should still apply. Thin-layer disposal technology may also be useful in habitat creation projects where small changes in elevation are needed (for example, transforming shallow subtidal areas into intertidal marshes).

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Environmental Effects of Dredging Technical Notes

Trophic Transfer and Biomagnification Potential of Contaminants in Aquatic Ecosystems

Purpose

This technical note examines the potential (or lack thereof) of contaminants to biomagnify in aquatic ecosystems. This information will be useful in interpreting the environmental significance of regulatory-mandated dredged material bioaccumulation test results. Several chemical classes were examined, with emphasis placed on contaminants that are of immediate concern for management of dredged material. Major classes of contaminants of concern in dredged material management currently include metals such as mercury and cadmium; polycyclic aromatic hydrocarbons (PAHs), especially petroleum-derived PAHs; known or potentially carcinogenic compounds such as dioxin; and organochlorine compounds such as polychlorinated biphenyls (PCBs).

The scope of this study does not include air-breathing organisms (for example, marine mammals, sea turtles, reptiles, piscivorous birds, terrestrial biota). A more comprehensive review of the data presented herein is available in Suedel and others (1994).

Background

Potential ecological effects of sediment-associated contaminants are of concern, particularly in the context of dredged material management. Sediments can serve as contaminant sources for transport and exposure to aquatic biota, particularly when sediments are disturbed by physical perturbations such as storms, bioturbation, or dredging and aquatic placement of dredged material. Sediment-sorbed contaminants may accumulate sufficiently in the tissues of prey organisms to elicit direct adverse effects, and may be transferred to consumers through dietary intake or by increased concentrations in the water column. Aquatic organisms that bioaccumulate contaminants from water or sediment may transfer these contaminants to predators that forage on them.

Of special interest is the extent to which these sediment-associated contaminants can move through aquatic food webs and thus potentially affect organisms at higher trophic levels. This trophic transfer potential must be known in order to determine the environmental significance of the bioaccumulation of sediment-associated materials in aquatic organisms.

Additional Information

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Introduction

The terms bioconcentration, bioaccumulation, biomagnification, trophic transfer, and trophic transfer coefficient are defined below to avoid confusion, as they have been used inconsistently throughout the literature (Dallinger and others 1987).

Bioconcentration is the uptake of a contaminant by aquatic organisms where water is the sole contaminant source. Bioaccumulation is the uptake of a contaminant from both water and dietary sources. Biomagnification refers to the processes of both bioconcentration and bioaccumulation that result in increased tissue concentrations of a contaminant as it passes through two or more trophic levels (Macek, Petrocelli, and Sleight 1979).

Trophic transfer is defined as the transport of contaminants between two trophic levels (that is, prey to predator) (Swartz and Lee 1980). Trophic transfer coefficient (TTC) is the concentration of contaminant in consumer tissue divided by the concentration of contaminant in food sources (that is, preceding trophic level). A TTC is an approximate measure of the potential for a contaminant to biomagnify. Biomagnification occurs when concentrations of a material increase between two or more trophic levels (that is, TTC >1) and is a subset of trophic transfer, which refers to any movement of a material between trophic levels (that is, TTC can be greater than or less than 1). If trophic transfer is determined to be substantially >1, biomagnification is said to occur. If a TTC value is ≤1, biomagnification is judged not to take place.

Approach

This review was conducted in two phases. In Phase I, information from the published literature demonstrating contaminant trophic transfer (or lack thereof) in laboratory and field experiments was reviewed and summarized. Studies examining annelids and molluscs as potential first-level bioaccumulators of contaminants from sediments were emphasized since these organisms are used extensively to assess regulatory-mandated sediment bioaccumulation potential (U.S. Environmental Protection Agency/U.S. Army Corps of Engineers 1991). Whenever possible, results were expressed quantitatively as chemical-specific TTCs.

In Phase II, the TTCs and estimates of overall potential for contaminant trophic transfer through aquatic food webs from Phase I were compared with appropriate data from published aquatic food web models. Phase II was designed to determine the applicability of laboratory and modeling results in predicting contaminant-specific trophic transfer potential. General conclusions were then drawn concerning whether biomagnification (with regard to categories of contaminants and groups of organisms) occurs within aquatic systems and, if so, its relative frequency of occurrence, magnitude, and estimates of uncertainty.

Peer-reviewed literature was obtained from a variety of sources including electronic database and chain-of-citation searches. Approximately 300 articles published since 1969 were obtained and screened for relevant information. Over 100 manuscripts from the published literature were selected for detailed review based on the reporting of contaminant tissue data, allowing for the determination of contaminant TCC values.

Emphasis was placed on articles containing measured contaminant tissue concentrations of organisms comprising potential predator-prey relationships of aquatic food webs. As part of this review, results from laboratory experiments were compared with field results whenever possible.

Results and Discussion

Phase I, Literature Review—Metals

Most metals that were examined showed potential for trophic transfer uptake from food, but not in sufficient quantities to result in biomagnification (Figure 1). Those metals that showed a propensity to biomagnify include arsenic and methyl mercury, and perhaps mercury.

Arsenic was the only compound examined that showed a clear trend of increased TTC values with increased trophic level. This relationship was found only in marine food webs, as no supporting data for freshwater aquatic food webs was found. Cadmium also appears to biomagnify in aquatic food webs; however, all TTC values for cadmium calculated in this review as >2.4 were for marine gastropods. Some marine gastropod species apparently have the

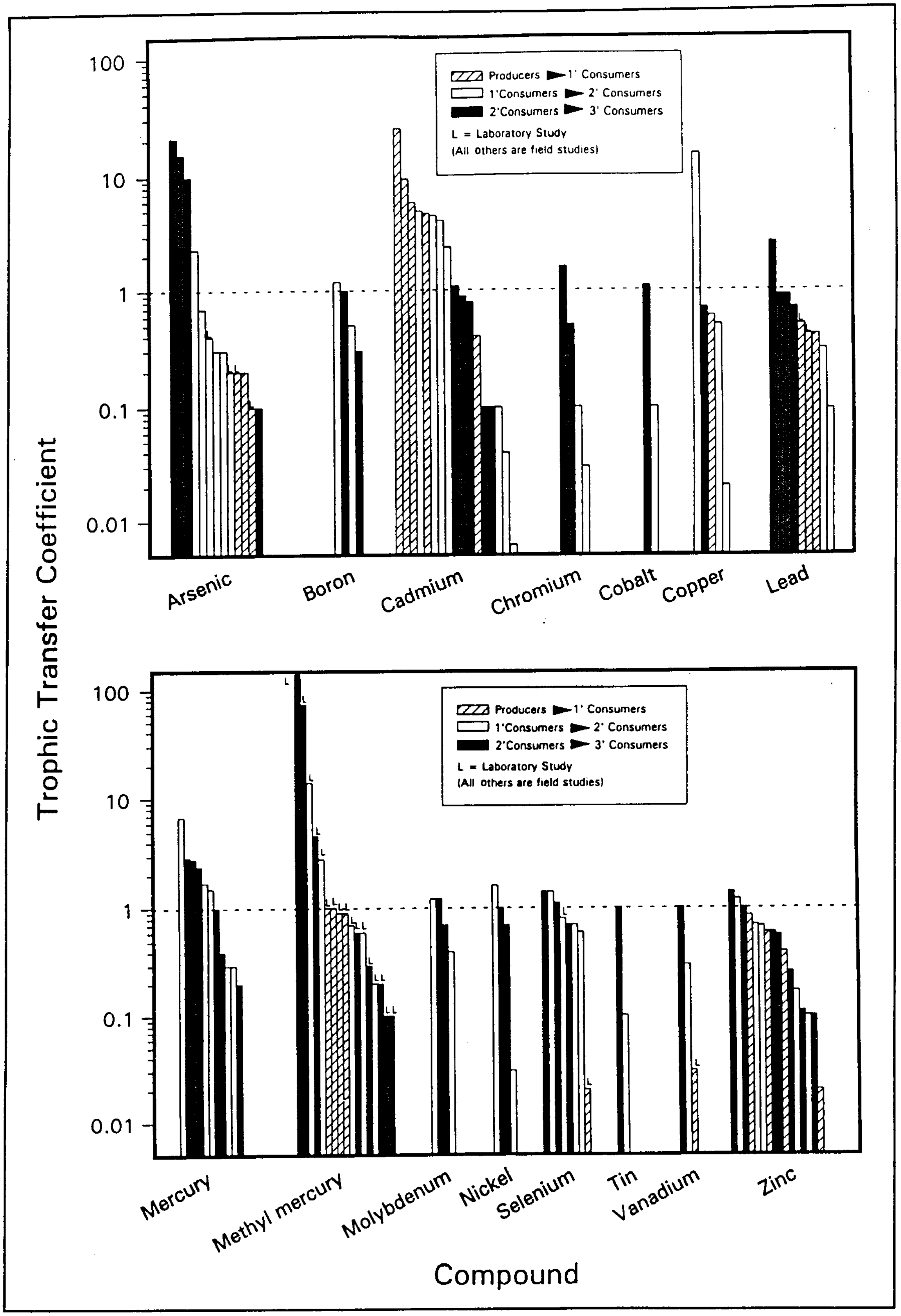

Figure 1. Trophic transfer coefficients for metals examined in this study. (TCC values >1 indicate a potential for biomagnification in aquatic ecosystems)

capacity to sequester cadmium in their tissues (the physiological significance for this is unknown).

Concentrations of most metals were often higher in tissues of producers and primary consumers than top-level carnivores (Klump and Peterson 1979; Ward, Connell, and Anderson 1986). Often, organisms feeding directly on sediments (such as crab and shrimp) and filter-feeders (such as bivalve molluscs) had the highest metal body burdens (LeBlanc and Jackson 1973; Hardisty and others 1974; Ward, Connell, and Anderson 1986; Kiorboe, Mohlenberg, and Riisgard 1983).

These results were consistent with the findings of Bryan (1979) and Dallinger and others (1987). Bryan (1979) noted that food web transfer was a significant source of metals to predator species such as fish. He noted that fish tissue levels were dependent primarily on the ability of the fish to excrete or store the contaminant. In addition to fish, decapods, polychaete worms, and bivalve molluscs were found to have the ability to regulate some essential metals such as zinc and copper, but not nonessential metals such as cadmium and lead (Bryan 1979, Bryan and Langston 1992, Lewis and Cave 1982). The fact that oysters, other bivalve molluscs, and aquatic organisms such as fish accumulate some metals for physiological requirements must be considered before concluding that biomagnification of these metals is occurring in aquatic food webs.

Phase I, Literature Review—Organics

From the data reviewed, PCBs, DDT, DDE, and toxaphene have the potential to biomagnify in aquatic ecosystems (Figure 2). Most of the accumulation of these contaminants was in secondary and tertiary consumer organisms. However, few if any data were found for lower tropic level organisms for toxaphene and DDT. Studies examining DDT and PCB accumulation observed higher tissue burdens in top carnivorous species such as salmonids and bass (Oliver and Niimi 1988, Niethammer and others 1984) and were attributed to the lipophilic nature of these compounds and exposure duration. Top carnivores often had the highest lipid content and longest life spans relative to organisms at lower trophic levels. Generally speaking, biomagnification data were lacking for producers and primary consumers for most organic compounds. Other organics reviewed do no appear to biomagnify in aquatic ecosystems.

Phase II, Trophic Transfer Models

Bioenergetic-based models (food web models) are used to predict contaminant concentrations in organism tissues at several levels through aquatic food webs (Thomann and Connolly 1984, Thomann 1989). One of the most rigorous studies predicting food web biomagnification (or the lack thereof) of organic compounds by a food chain model was conducted by Thomann (1989). Thomann's model was used in this review to compare model predictions to "real world" biomagnification potential. This was accomplished by comparing model predictions with calculated TTC values for organic compounds examined in this review. Only tissue residue data for small fish-predacious fish food

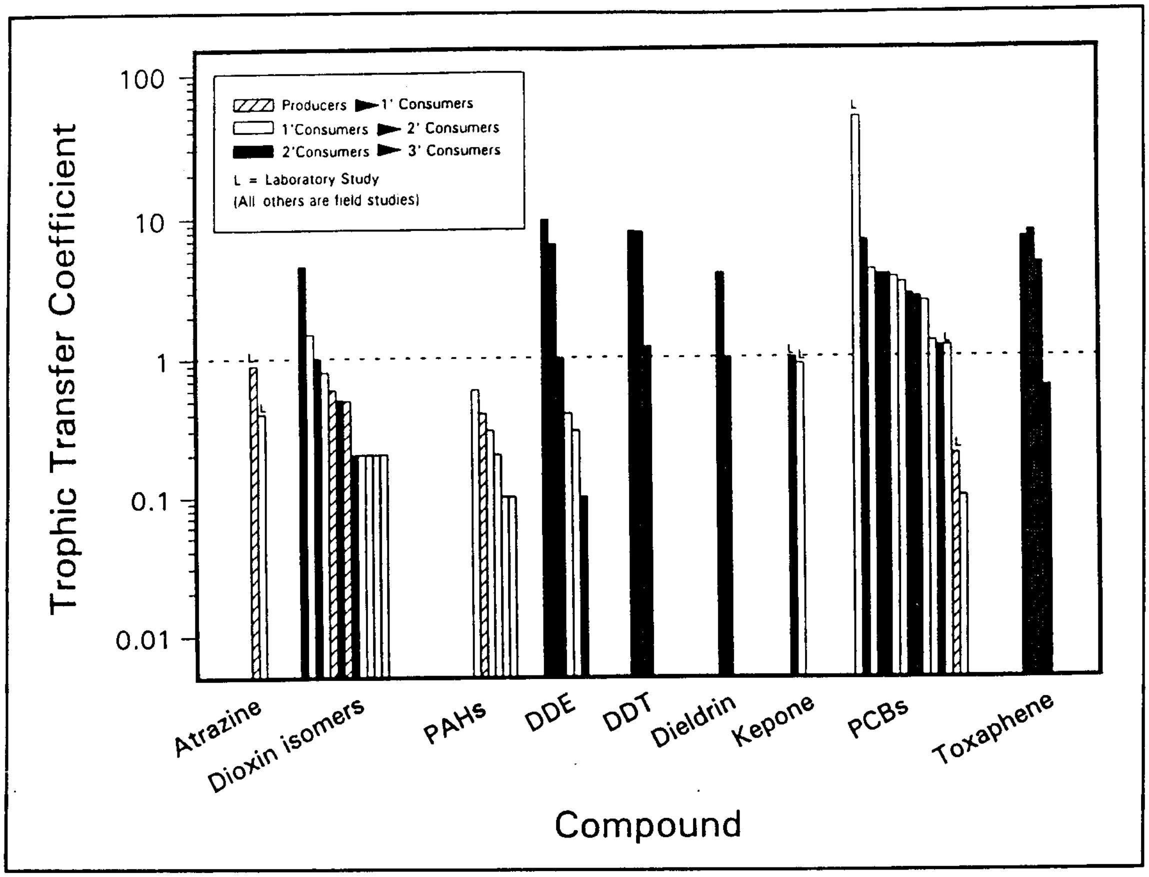

Figure 2. Trophic transfer coefficients for organic compounds examined in this study. (TCC values >1 indicate a potential for biomagnification in aquatic ecosystems)

chains obtained during this review and from the Thomann model were used for comparison. Trophic transfer coefficients for dioxin isomers (TCDD), PCBs (as Aroclor 1254), DDT, DDE, dieldrin, toxaphene, and kepone obtained from this review were plotted against values calculated from the model for these compounds (Figure 3).

As shown in Figure 3, the Thomann model generally provided numerically lower estimates of the potential for the organic compounds examined (log K_{ow} values between 5 and 6.5) to biomagnify in aquatic ecosystems. TTC values calculated by the model were 2 to 10 times lower than most median TTC values obtained for small fish-predacious fish food chains in this review.

For kepone, predictions of trophic transfer from the model and this literature review were virtually identical (TTC = 0.9). All TTC values for dieldrin and toxaphene from this review were higher than the TTC values predicted by the model, resulting in median values for these compounds considerably above model predictions. The model produced higher trophic transfer potential for Aroclor 1254 and 2,3,7,8-TCDD (log K_{ow} values between 6.5 and 7) than reported in this literature review.

Except for toxaphene and dieldrin, model predictions were within the range of TTC values found in this review.

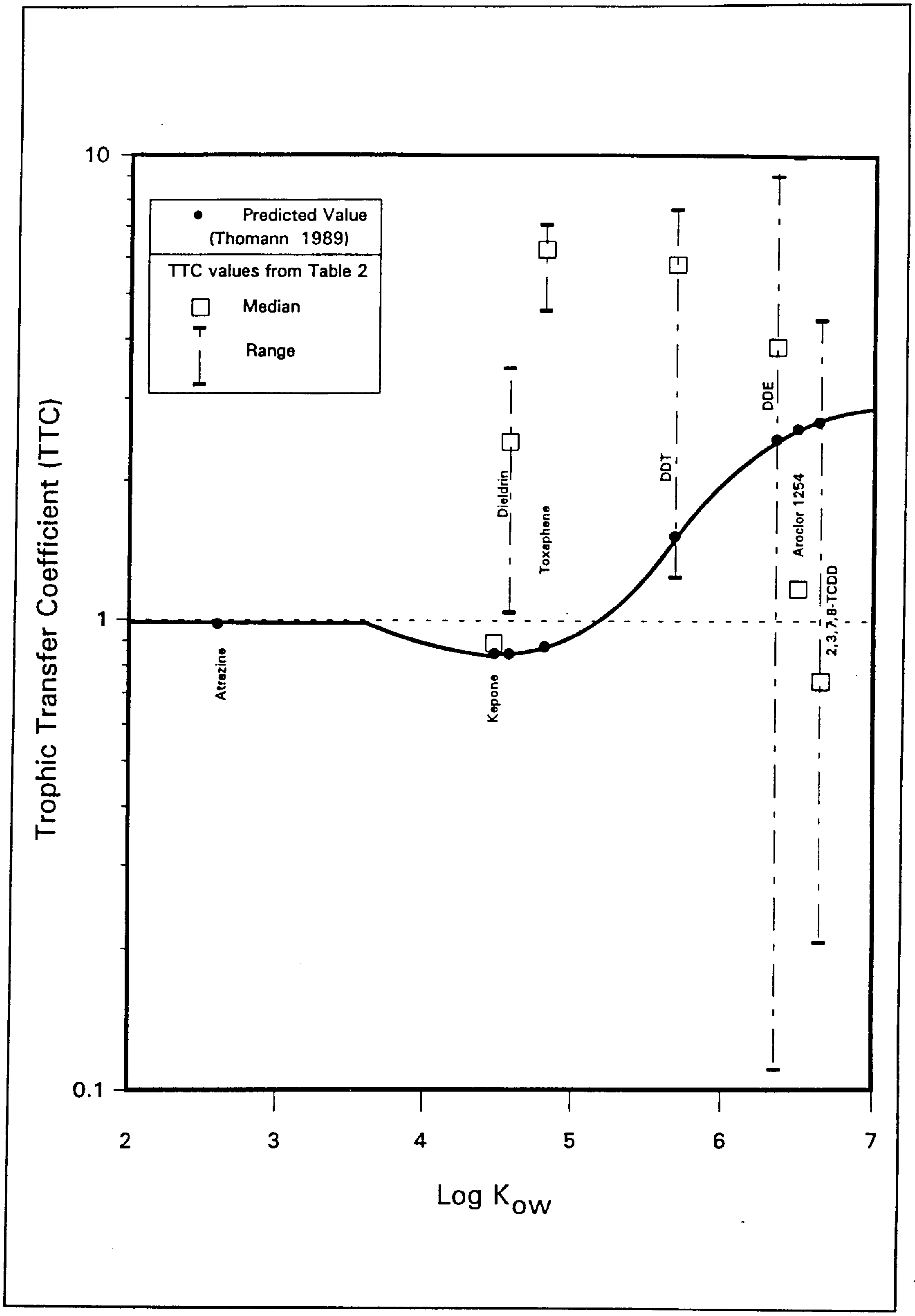

Figure 3. Plot of trophic transfer coefficients versus log K_{ow} values for selected organic compounds compared to values predicted by the food chain model by Thomann (1989)

Aquatic Food Web Biomagnification—Evidence

The data reviewed in this study are in general agreement with the results of other investigators examining the potential for aquatic food web biomagnification (Biddinger and Gloss 1984, Kay 1984). All three studies concluded that PCBs and methyl mercury have the potential to biomagnify in aquatic food webs (Table 1). As in this review, Biddinger and Gloss (1984) also concluded that total mercury and DDT have the potential to biomagnify. The results from this review and Biddinger and Gloss (1984) generally agree that only highly water-insoluble organic compounds have the potential to biomagnify in aquatic food webs (that is, DDT, PCBs). Results from Thomann's model also indicated that highly water-insoluble compounds (log k_{ow} values 5 to 7) showed the greatest potential to biomagnify. However, the model also included other organic compounds that were not observed to biomagnify in this study, such as TCDD (log K_{ow} = 6.6).

As was also observed in an earlier review by Kay (1984), studies examining contaminant biomagnification were often plagued by methodological problems. The variability observed in results for individual compounds may be attributed to many factors, including uncertainty regarding an organism's position in a food web, contrived laboratory food chains that do not effectively represent actual feeding relationships in the field, unknown feeding habits of organisms examined, inadequate sampling (that is, one sample at a given time and location), sampling at different times and locations, and lack of standardization of units of measurement (fresh weight, dry weight, lipid normalized).

Results reported for tissue levels based on wet weights or lipid normalized data can influence the TTC considerably, since percent water and percent lipid have been demonstrated to vary considerably with age, body weight, season, and physiological condition of the organism (Kay 1984). Often, the organisms examined within a particular study did not fit in a logical food chain, with several organisms potentially occupying a given trophic level. Many studies made no effort to identify predator-prey relationships between organisms and trophic levels, making it difficult to determine whether contaminant trophic transfer could actually occur.

In other studies, tropic levels were well defined but other factors that may affect conclusions regarding biomagnification potential were not considered. For example, if gut contents of predators were not analyzed, definitive statements regarding their food source(s) or proportions cannot be made. Individual age and size can influence body burdens, particularly for younger, smaller organisms with high potential for growth dilution (Bryan 1979). Methodological problems such as those listed above severely limit the conclusions that can be made regarding trophic transfer and biomagnification of most contaminants in aquatic ecosystems.

Few, if any, data exist on the potential for numerous organic compounds and metals to biomagnify in aquatic systems, especially those compounds that are not hypothesized to readily biomagnify. Thus, conclusions regarding their

Table 1. Compounds for Which Available Information Exists for Potential Food Chain Biomagnification to Occur in Aquatic Ecosystems

	Source		
Compound	Biddinger and Gloss (1984)	Kay (1984)	This Study
Most Evidence for Potential Biomagnification			
Methyl mercury	Yes	Yes	Yes
PCBs	Yes	Yes	Yes
Some Evidence for Potential Biomagnification			
Arsenic	No	No	Yes
Mecury (total)	Yes	No	Yes
Selenium	Yes	No	No
Zinc	Yes	No	No
Benzo[a]pryrene	No	Yes	No
DDT	Yes	No	Yes
DDE	—[1]	No	Yes
Dieldrin	—	Yes	No
Endrin	No	Yes	—
Kepone	—	Yes	No
Mirex	—	Yes	No
Toxaphene	—	—	Yes
No Evidence for Potential Biomagnification			
Beryllium	No	—	—
Boron	—	—	No
Cadmium	No	No	No
Chromium	No	No	No
Cobalt	—	—	No
Copper	No	No	No
Lead	No	No	No
Molybdenum	—	—	No
Nickel	No	No	No
Silver	No	No	—
Tin	—	No	No
Vanadium	—	—	No
Aldrin/dieldrin	No	No	—
Atrazine	—	No	No
Chlordane	No	—	—
Chlorinated benzenes	No	No	—
Chlorinated phenols	No	No	—
Endosulfan	No	No	—
Heptachlor	No	—	—
HCH	No	—	No
Lindane	—	No	—
PAH	No	No	No
Phthalate esters	No	—	—
TCDD	No	—	No

[1] Not examined.

potential to biomagnify cannot be made until data are available. From the data reviewed in this study and others, when the potential for food web biomagnification was evident in aquatic food webs, TTC values were generally between 1 and 10, rather than hundreds or thousands, as reported for non-aquatic food webs (Kay 1984).

Conclusions

Food web biomagnification of contaminants in freshwater and marine ecosystems is not well substantiated in the literature. Results of this review suggest that most metal and organic contaminants appear to have a low potential for trophic transfer and are therefore not likely to biomagnify in aquatic food webs. Data reviewed in this study indicate that DDT, DDE, PCBs, toxaphene, total and methyl mercury, and arsenic have the potential to biomagnify (Table 1; Figures 1-2). For most compounds examined, data were variable, with TTC values varying 2 to 3 orders of magnitude for arsenic, zinc, methyl mercury, and cadmium.

Evidence from this review suggests that most biologically available contaminants associated with sediment or dredged material may undergo trophic transfer but would not biomagnify in aquatic food webs. From the combined evidence of this and other reviews (Biddinger and Gloss 1984, Kay 1984), if sediment-associated PCBs and methyl mercury were to bioaccumulate in organisms such as bivalve molluscs and polychaetes, these two contaminants will likely have a greater potential to biomagnify in aquatic ecosystems than other contaminants (Table 1). If biomagnification of these contaminants takes place in aquatic food webs, the TTC values from bottom to top of the food webs are likely to be on the order of 1 to 10, rather than hundreds to thousands as observed for some nonaquatic food webs (Kay 1984).

Additional, carefully designed, scientifically defensible and repeatable research is needed to clarify whether these and numerous other compounds have the potential to biomagnify in aquatic food webs. Until then, predictions of whether aquatic organisms will experience biomagnification when exposed to these compounds will remain uncertain.

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